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June 15, 2012

HAND DELIVERED

Jeanine Townsend
Clerk to the Board
State Water Resources Control Board
1001 I Street, 24th Floor
Sacramento, California 95814

Re: In the Matter of Own Motion Review of Waste Discharge Requirements Order No. R5-2010-0114 [NPDES No. CA0077682] for Sacramento Regional Wastewater Treatment Plant, Issued by the California Regional Water Quality Control Board, Central Valley Region
COMMENTS TO A-2144(a)(b) – JULY 8 BOARD WORKSHOP

Hi Jeanine:

Please find enclosed for filing an original of the following documents:

1. June 15, 2012, Letter to Ms. Jeanine Townsend, Clerk to the Board, State Water Resources Control Board, from Stan Dean, District Engineer, Sacramento Regional County Sanitation District;
2. Sacramento Regional County Sanitation District's Response To Draft Order On Own Motion Review;
3. Sacramento Regional County Sanitation District's Request For Admission Of New Evidence And For Official Notice, including Exhibits A through J;
4. Declaration Of Theresa A. Dunham In Support Of Sacramento Regional County Sanitation District's Request For Admission Of New Evidence And For Official Notice; and
5. Proof of Service.

Also enclosed are copies of the face sheets for each document; please stamp and return the face sheets via our courier. Per my earlier email, a CD containing PDFs of these documents is also provided.

Thank you, and if you have any questions please contact me.

Very truly yours,

Crystal

Crystal Rivera, Secretary to
Paul S. Simmons

Encs.

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Attorneys for SACRAMENTO REGIONAL COUNTY SANITATION DISTRICT

BEFORE THE

CALIFORNIA STATE WATER RESOURCES CONTROL BOARD

In the Matter of Own Motion Review of Waste Discharge Requirements Order No. R5-2010-0114 [NPDES No. CA0077682] for Sacramento Regional Wastewater Treatment Plant, Issued by the California Regional Water Quality Control Board, Central Valley Region.	SWRCB/OCC File Nos. A-2144(a) and A-2144(b) (consolidated) **DECLARATION OF THERESA A. DUNHAM IN SUPPORT OF SACRAMENTO REGIONAL COUNTY SANITATION DISTRICT'S REQUEST FOR ADMISSION OF NEW EVIDENCE AND FOR OFFICIAL NOTICE**

I, Theresa A. Dunham, declare:

1. I am an attorney and shareholder in the law firm of Somach Simmons & Dunn, special counsel for Sacramento Regional County Sanitation District.

2. Exhibit A attached to the Sacramento Regional County Sanitation District's Request for Admission of New Evidence and For Official Notice (District's Request) is a true and correct copy of the February 1987 *Wastewater Disinfection for Health Protection*, Sanitary Engineering Branch, California Department of Health Services.

3. Exhibit B attached to the District's Request is a true and correct copy of the Memorandum to Office of Drinking Water Management Staff, from Office of Drinking Water, Peter A. Rogers, Chief (Aug. 18, 1992, retyped in November 2000) re: Uniform Guidelines for Disinfection of Wastewater, including attached Uniform Guidelines for the Disinfection of Wastewater.

4. Exhibit C attached to the District's Request is a true and correct copy of the December 28, 2011 Central Contra Costa Sanitary District's Response to Comments on the Tentative Order No. R2-2011-XXXX, NPDES Permit CA0037648.

5. Exhibit D attached to the District's Request is a true and correct copy of the October 31, 2011 San Luis & Delta-Mendota Water Authority and State Water Contractors' Comments on Tentative Order No. R2-2011-XXXX (NPDES No. CA0037648) for the Central Contra Costa Sanitary District Wastewater Treatment Plant.

6. Exhibit E attached to the District's Request is a true and correct copy of the December 8, 2011 Bay Area Clean Water Agencies' comments on the Tentative Order for the Central Costa Sanitary District, No. R2-2011-XXXX, NPDES Permit No. CA0037648.

7. Exhibit F attached to the District's Request is a true and correct copy of the California Regional Water Quality Control Board, San Francisco Bay Region, Response to Written Comments on October 2011 Tentative Order for Central Contra Costa Sanitary District Wastewater Treatment Plant.

8. Exhibit G attached to the District's Request is a true and correct copy of the California Regional Water Quality Control Board, San Francisco Bay Region, Order No. R2-2012-0016, NPDES No. CA0037648 (adopted February 8, 2012).

9. Exhibit H attached to the District's Request is a true and correct copy of the *Numeric Nutrient Endpoint Development for San Francisco Bay Estuary: Literature Review and Data Gaps Analysis*, L. McKee, M. Sutula, A. Gilbreath, J. Beagle, D. Gluchowski, and J. Hunt, Southern California Coastal Water Research Project (Technical Report 644 – June 2011).

10. Exhibit I attached to the District's Request is a true and correct copy of the April 27, 2012 Letter to Mike Chotkowski, Department of Interior, United States Fish and

Wildlife Service, from Sacramento Regional County Sanitation District, Comments on Endangered and Threatened Wildlife and Plants; 12-month Finding on a Petition to List the San Francisco Bay-Delta Population of the Longfin Smelt as Endangered or Threatened, Docket No. FWS-R8-ES-2008-0045.

11. Exhibit J attached to the District's Request is a true and correct copy of the March 2, 2012 Letter to Municipal Wastewater Dischargers from Bruce H. Wolfe, Executive Officer, California Regional Water Quality Control Board, San Francisco Bay Region, subject Water Code Section 13267 Technical Report Order Requiring Submittal of Information on Nutrients in Wastewater Discharges.

I declare under penalty of perjury under the laws of the State of California that the foregoing is true and correct. Executed this 15th day of June 2012 at Sacramento, California.

Theresa A. Dunham

Theresa A. Dunham

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STATE OF CALIFORNIA

STATE WATER RESOURCES CONTROL BOARD

In the Matter of Own Motion Review of Waste Discharge Requirements Order No. R5-2010-0114 [NPDES No. CA0077682] for Sacramento Regional Wastewater Treatment Plant, Issued by the California Regional Water Quality Control Board, Central Valley Region.	SWRCB/OCC File Nos. A-2144(a) and A-2144(b) (consolidated) **PROOF OF SERVICE**

I am employed in the County of Sacramento; my business address is 500 Capitol Mall, Suite 1000, Sacramento, California; I am over the age of 18 years and not a party to the foregoing action.

On June 15, 2012, I served **a CD** containing a true and correct copy of:

(1) **June 15, 2012, Letter to Ms. Jeanine Townsend, Clerk to the Board, State Water Resources Control Board, from Stan Dean, District Engineer, Sacramento Regional County Sanitation District;**

(2) **Sacramento Regional County Sanitation District's Response To Draft Order On Own Motion Review;**

(3) Sacramento Regional County Sanitation District's Request For Admission Of New Evidence And For Official Notice, including Exhibits A through J;

(4) Declaration Of Theresa A. Dunham In Support Of Sacramento Regional County Sanitation District's Request For Admission Of New Evidence And For Official Notice; and

(5) Proof of Service.

XXX (by mail) on all parties in said action, in accordance with Code of Civil Procedure § 1013a(3), by placing a true copy thereof enclosed in a sealed envelope, with postage fully paid thereon, in the designated area for outgoing mail, addressed as set forth below.

SEE ATTACHED SERVICE LIST

I declare under penalty of perjury that the foregoing is true and correct. Executed on June 15, 2012, at Sacramento, California.

Crystal Rivera

SOMACH SIMMONS & DUNN
A Professional Corporation

SERVICE LIST
SWRCB/OCC File Nos. A-2144(a) and A-2144(b) (consolidated)

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STATE OF CALIFORNIA

STATE WATER RESOURCES CONTROL BOARD

In the Matter of Own Motion Review of Waste Discharge Requirements Order No. R5-2010-0114 [NPDES No. CA0077682] for Sacramento Regional Wastewater Treatment Plant, Issued by the California Regional Water Quality Control Board, Central Valley Region.	SWRCB/OCC File Nos. A-2144(a) and A-2144(b) **SACRAMENTO REGIONAL COUNTY SANITATION DISTRICT'S RESPONSE TO DRAFT ORDER ON OWN MOTION REVIEW**

By letter dated May 14, 2012, the Chief Counsel of the State Water Resources Control Board (State Board) transmitted a proposed order on the State Board's own motion review (Draft Order) of Central Valley Regional Water Quality Control Board (Regional Board) Order No. R5-2010-0114 (Permit). Sacramento Regional County Sanitation District (District or SRCSD), the owner and operator of the Sacramento Regional Wastewater Treatment Plant (SRWTP), hereby provides comments, certain objections, and other responses to the Draft Order.

I. INTRODUCTION

The Draft Order fails to acknowledge fundamental errors in the Permit. In some cases, the Draft Order proposes an altered logic for the outcomes in the Permit, but these attempted cures do not succeed. The State Board should not adopt the Draft Order.

The three subjects taken up in the Draft Order are the Permit's requirements regarding pathogens and filtration (tertiary filtration and disinfection), ammonia effluent limitations, and nitrate effluent limitations. In the case of each of these subjects, the Permit is built upon flawed regulatory/legal logic and unsupported technical conclusions. These matters were the subjects of a Petition for Review filed by the District on January 10, 2011,[1] and the District here addresses the regulatory and technical flaws that would be allowed or introduced by the Draft Order.

The first issue is tertiary filtration, for which the relevant Permit requirements would require an estimated $1.2 - 1.3 billion in capital expenditures and $44 - 46 million in additional annual operation and maintenance (O&M) costs.[2] With respect to regulatory issues, the Draft Order's analysis relies upon the existence of a regulation (an unspecified narrative water quality objective) that does not exist. As a result, the Draft Order does not do what it should have done, which is to conduct an exacting review of: whether the Regional Board complied with the Porter-Cologne Water Quality Control Act (Porter-Cologne),[3] specifically Water Code sections 13263(a) and 13241; and whether findings required to implement such provisions are responsive to law and supported by the record, and reasonable. As discussed in the District's Petition and below, the Permit fails these tests. With respect to technical issues related to the Permit's tertiary filtration requirements, the Draft Order – like the Permit – unfortunately relies on selective or unfounded discussion that cannot be supported. Also, and very troubling, the Draft Order – like the Permit itself – altogether ignores highly relevant, uncontroverted evidence, without explanation. Further,

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[1] *In the Matter of the Sacramento Regional County Sanitation District's Petition for Review of Action and Failure to Act by Regional Water Quality Control Board, Central Valley Region, in Adopting Waste Discharge Requirements Order No. R5-2010-0114 (NPDES No. CA0077682) and Time Schedule Order No. R5-2010-0115 for Sacramento Regional County Sanitation District, Sacramento Regional Wastewater Treatment Plant* (Petition). The Petition is incorporated by reference in its entirety.

[2] Pages 18-25 of the Petition discuss compliance costs and underlying evidence in detail.

[3] Wat. Code, § 13000.

it states factual propositions that are wrong or have no record support or are directly contrary to all record evidence on the subject.

The Permit's ammonia and nitrate effluent limitations would amount to an additional $.8 billion expense (and $31 million in annual O&M costs), with the ammonia compliance constituting the greatest amount of the costs. These two requirements have similar defects in regulatory logic, but are discussed separately herein.

In the case of ammonia, ultimately the regulatory logic of the Permit amounts to the following: "An effluent limitation of 33 milligrams per liter (mg/L) is too high. Therefore, the effluent limitation shall be 1.8 mg/L." The Draft Order does nothing to alter this non sequitur. In fact, while sweeping the Regional Board's specific Permit justifications under the rug, the Draft Order creates new problems. For example, it states that the Regional Board did things that the Regional Board did not do. It repeatedly cites and relies upon a U.S. Environmental Protection Agency (U.S. EPA) guidance document for a proposition that document does not contain. The misstatement of the document is presumably a mistake. But it is not appropriate. The Draft Order pairs the mistake with novel and erroneous interpretations of regulations that do apply, to perpetuate the non sequitur identified above. Beyond the regulatory problems, the Permit and Draft Order's technical conclusions concerning ammonia are unfounded. The Permit and Draft Order rely on hypotheses to conclude that current effluent ammonia concentrations are "too high." In sum, the scientific underpinnings are lacking, but even if there were universal consensus about "the science," the specific effluent limitations imposed are incorrect and improper.

With respect to nitrate, the Draft Order identifies the glaring error in the Permit's regulatory logic. The Permit's nitrate limitations must be set aside. But the Draft Order should not, as it proposes to do, bias or prejudice the results of future decisions concerning nitrate or nutrients.

The District understands, of course, that the Draft Order is on the State Board's own motion. However, fully 17 months have passed since the District filed its Petition, and the District believes the State Board has had time to conduct a robust analysis of the Permit. The

District submits, respectfully, that in many important respects the Draft Order is superficial and not careful; it would not do a great deal more than embrace extremely aggressive regulation of the District. The Draft Order also avoids an issue that the State Board undoubtedly recognizes to be highly significant.[4] Under these circumstances, there is very little value in the State Board adopting any order at all.[5]

If, however, the State Board does adopt the Draft Order or anything like it, the State Board should respect the real-world problems faced by the District. The State Board should recognize that uncertainty concerning what the District must or must not ultimately design and build creates real-world problems.

II. DISCUSSION

A. Tertiary Filtration Requirements (Pathogens)

The Permit requirements for further pathogen reduction at the SRWTP – generally referred to as tertiary filtration – are unjustified. The Draft Order does not acknowledge the Regional Board's obligations under state law or evaluate whether the Regional Board complied with applicable law. It condones the Permit's departure from reason and judgment, and it does not even acknowledge the major economic burden that the requirements would impose on the Sacramento region. The Draft Order does not evenhandedly evaluate all evidence and ignores important evidence.

The Draft Order does not address, at all, several issues that were identified in the District's Petition related to the Permit's tertiary filtration requirements. While the State Board may not have been obliged, in own motion review, to address all matters raised by the Petition, it is not clear why the Draft Order discusses only some of the issues that relate to tertiary filtration. The

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[4] Specifically, the Regional Board applied State Board Resolution 68-16 ("Antidegradation" policy) in a manner that the District considers unlawful, but that is at least extremely atypical. (See Petition, pp. 133-165.) With respect to antidegradation, both the regulatory issues and adequacy of the Regional Board's technical approach would, it seems, merit the attention of the State Board.

[5] The Draft Order notes the existence of pending litigation concerning the Permit. (Draft Order, p. 3, fn. 7.) The District's present filings are without waiver of any position that has been or may be asserted in that case with regard to any issue.

District reiterates all the matters in its Petition. The discussion below is directed primarily to the Draft Order, while identifying some of the other issues that must be considered.

Initially, while the Draft Order describes the Permit's new requirements requiring tertiary filtration / disinfection, it does not provide a complete regulatory context. The only regulation related to pathogens that applies to the SRWTP discharge is the water quality objective for coliform bacteria identified in the Water Quality Control Plan for the Sacramento and San Joaquin River Basins (Basin Plan). That water quality objective provides:

> In waters designated for contact recreation (REC-1), the fecal coliform concentration based on a minimum of not less than five samples for any 30-day period shall not exceed a geometric mean of 200/100 ml, nor shall more than ten percent of the total number of samples taken during any 30-day period exceed 400/100 ml.[6]

The Permit does not implement this water quality objective. It is much more stringent. The Permit's total coliform requirements (2.2 Most Probable Number [MPN] per 100 mL as a 7-day median, and as otherwise specified in the Permit) are based on Department of Public Health (DPH) "Title 22" regulations that prescribe effluent quality for certain uses of recycled water "that has been transported from the point of treatment or production to the point of use without an intervening discharge to waters of the State."[7] Specifically, under DPH regulations, the "2.2 MPN" requirement applies where effluent is used directly for irrigation of "food crops," impoundments of recycled water for unrestricted recreation, and certain other uses.[8] The new Permit limitations for BOD, TSS, and turbidity are coupled with the new total coliform requirements, and represent limits that can be achieved with filtration technology.[9]

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[6] *Water Quality Control Plan for the Sacramento and San Joaquin River Basins* (4th ed. Rev. Sept. 2009) (Basin Plan), p. III-3.00.

[7] Cal. Code Regs., tit. 22, § 60301.200.

[8] Cal. Code Regs., tit. 22, §§ 60301.220, 60304(a), (b), 60305.

[9] As characterized in the Permit, the new BOD and TSS requirements are "based on tertiary treatment." (Permit, p. F-17.) The turbidity specification is also based on the capabilities of tertiary filtration. (Permit, pp. F-78 to F-79.) All of the described filtration requirements are subject to the Permit Compliance Schedule. (Permit, pp. 30, 33; see also, Staff Report, Sacramento Regional County Sanitation District, Sacramento Regional Wastewater Treatment Plant, Proposed NPDES Permit Renewal and Time Schedule Order, Sacramento County (Staff Report), p. 29, Table 8 [tertiary requirements include BOD, TSS, total coliform, and turbidity].) The Permit generally refers to all of these provisions collectively as "tertiary treatment" or "tertiary filtration."

Order No. 5-00-188, the District's predecessor permit, contained effluent limitations for disinfection/pathogens as follows: 23 MPN/100 mL as a median weekly average and 500 MPN/100 mL as a daily maximum not to be exceeded in any consecutive two days.[10] While these requirements are themselves more stringent than necessary to implement any applicable water quality objectives, they are acceptable to the District. Limitations for BOD and TSS in Order No. 5-00-188 were based on applicable requirements of the Clean Water Act (CWA).[11] The previous limitations for total coliform, BOD, and TSS are adequate and appropriate. The State Board should determine that the Permit's filtration requirements are improper. The Regional Board staff prepared a "Disinfection Alternative No. 1" based on 23 MPN/100 mL, with BOD and TSS limitations based on CWA requirements.[12] The State Board should order that final effluent limitations for coliform, BOD, and TSS shall be those provided in Disinfection Alternative 1. Those limitations are also identified in paragraph 6.B.ii of the District's Petition.[13]

1. Draft Order's Plain Legal Error and Resultant Failure of Review of State Law Compliance

Porter-Cologne is the source of the Regional Board's power. Porter-Cologne and other state laws also impose obligations on the Regional Board. The District maintains that the Regional Board failed to comply with applicable law, including provisions that mandate, and are designed to ensure, informed decisions, reasonableness, and balance.[14]

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[10] Order No. 5-00-188, pp. 13-14, and fn. 4. To the extent a request is necessary, the District requests the State Board take official notice of the orders of the State Board and Regional Board and other official acts of public agencies cited herein and relied upon by the District, in accordance with section 648.2 of title 23 of the California Code of Regulations.

[11] See Order No. 5-00-188, p. 13. The regulations implementing the CWA require effluent quality for BOD and TSS of 30 mg/L as a 30-day average. (40 C.F.R. § 133.102.) The actual performance of the SRWTP is significantly superior to the CWA "30-30" requirements for BOD and TSS. (See Permit, p. F-6 [Table F-2].)

[12] See Disinfection Alternative No. 1, Sacramento County Sanitation District [sic], Sacramento Regional Wastewater Treatment Plant, Proposed Waste Discharge Requirements and Time Schedule Order (NPDES No. CA0077682); Regional Water Quality Control Board, Central Valley Region Board Meeting – 9 December 2010, Item #6 (document distributed November 24, 2010), Administrative Record (AR) at SRCSD_BM_01, BM_1I, p. 3. The interim effluent limitations under the Permit are similar, but not identical to, Disinfection Alternative 1. (See Permit, p. 16.)

[13] Petition, p. 4.

[14] See Wat. Code, §§ 13000, 13001, 13241, 13263(a); Petition, pp. 43-53; pp. 7-10, *post*.

The Draft Order excuses the Regional Board's noncompliance with applicable state law in the adoption of tertiary filtration requirements, on the premise that because such requirements implement existing narrative water quality objectives from the Basin Plan, the CWA prohibits compliance with state law.[15] This is wrong. There is no such narrative water quality objective – nor does the Draft Order actually identify any such narrative water quality objective. Thus, full compliance with Water Code sections 13263(a) and 13241, and with basic tenets of administrative law, was required.[16] Through the vehicle of its foundational assumption regarding non-existent narrative water quality objectives, the Draft Order would avoid any consideration of whether the Regional Board complied with the law and whether required findings were appropriate and supported by evidence. The Draft Order would additionally minimize the significance of compliance with state law. Below and in section II.A.3, the District explains these points in more detail.

Water Code section 13263(a) requires that, in the adoption of waste discharge requirements, the Regional Board must consider the water quality objectives reasonably required to protect beneficial uses and the provisions of Water Code section 13241. Water Code section 13241 in turn requires the consideration of a variety of factors, one of which is economics.[17] (The Draft Order uses the phrase "consideration of costs" as apparent "shorthand"

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[15] Draft Order, p. 9.

[16] It is unnecessary here to address whether noncompliance with state law would have been excusable if the Regional Board had been implementing a mandate resulting from a narrative water quality objective and the CWA, because the Regional Board was not so acting.

[17] Water Code section 13241 provides:

> Each regional board shall establish such water quality objectives in water quality control plans as in its judgment will ensure the reasonable protection of beneficial uses and the prevention of nuisance; however, it is recognized that it may be possible for the quality of water to be changed to some degree without unreasonably affecting beneficial uses. Factors to be considered by a regional board in establishing water quality objectives shall include, but not necessarily be limited to, all of the following:
>
> (a) Past, present, and probable future beneficial uses of water.
>
> (b) Environmental characteristics of the hydrographic unit under consideration, including the quality of water available thereto.
>
> (c) Water quality conditions that could reasonably be achieved through the coordinated control of all factors which affect water quality in the area.
>
> (d) Economic considerations.
>
> (e) The need for developing housing within the region.
>
> (f) The need to develop and use recycled water.

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for Water Code section 13241.[18] That is an incomplete statement, and the specific, actual requirements of Porter-Cologne are discussed elsewhere. The immediate consideration is *whether* the Regional Board was required to comply with state law obligations.)

There is an adopted, numeric water quality objective for coliform in the Basin Plan.[19] The coliform effluent limitation in the Permit, and other provisions related to tertiary filtration, are much more stringent than necessary to implement this water quality objective.

The State Board has recognized that a complete analysis of the Water Code section 13241 provisions is required when the Regional Board proposes to adopt effluent limitations more stringent than those required by existing water quality objectives. If a Regional Board takes this approach, ". . . the rationale for the more stringent limitations must be explained in the permit findings . . . . In addition, the RWQCB must consider the factors specified in Water Code Section 13241[.]"[20] That is, if the Regional Board chooses to implement a more stringent objective on a permit-specific basis, it "must consider the factors specified in Water Code Section 13241."[21] The State Board has further explained that, "when a Regional Board includes permit limits more stringent than limits based on an applicable numeric objective in the relevant basin plan, the Regional Board must address the section 13241 factors in the permit findings. These factors include, among others, economic considerations, environmental characteristics of the hydrographic unit under consideration, and the need for recycled water."[22] Thus, the Regional Board must make findings related to each of the provisions of Water Code section 13241.[23] The

[18] Draft Order, pp. 8-10.

[19] See p. 5, *ante*, and Basin Plan, p. III-3.00 (water quality objective for fecal coliform).

[20] *In the Matter of the Petition of City and County of San Francisco, et al.*, State Board Order No. WQ 95-4 (Sept. 21, 1995), p. 13; see also *In the Matter of the Petitions of Napa Sanitation District, et al.*, State Board Order No. WQ 2001-16 (Dec. 5, 2001), p. 24.

[21] *In the Matter of the Petition of the Cities of Palo Alto, Sunnyvale and San Jose*, State Board Order No. WQ 94-8 (Sept. 22, 1994), p. 9.

[22] *In the Matter of the Review on Own Motion of Waste Discharge Requirements Order No. 5-01-044 for Vacaville's Easterly Wastewater Treatment Plant*, State Board Order WQO 2002-0015 (Oct. 3, 2002), p. 35, footnote omitted.

[23] See, e.g., State Board Order WQO 2002-0015, *supra*, pp. 35, 72 (issue remanded and Regional Board directed to revise its findings to expressly address Wat. Code, § 13241 factors which had not been addressed); see also State Board Order WQ 95-4, *supra*, pp. 13-14, 32 (permit remanded to Regional Board for failure to consider the factors specified in Wat. Code, § 13241).

State Board's Chief Counsel has explained that, in these types of circumstances, a regional board has an affirmative duty to develop and consider information on the section 13241 factors and engage in a "balancing" of factors to develop objectives consistent with the statute.[24] A regional board must "set forth findings to bridge the analytic gap between the raw evidence and ultimate decision or order."[25] Further, the findings must also be supported by evidence in the record.[26]

The Draft Order asserts that compliance with these requirements was unnecessary. The Draft Order argues that the tertiary filtration requirements were "calculated" to "implement existing narrative water quality objectives set forth in the Basin Plan" and thus that the Regional Board *could not* consider section 13241 factors.[27] The Draft Order then states that the "Basin Plan's narrative water quality objectives for the Delta" are water quality standards under the CWA.[28] It states that the total coliform limitations "are implementing existing water quality objectives" and thus economic considerations (and by extension all provisions of Water Code sections 13263(a) and 13241) are legally irrelevant.[29]

There are two enormous defects in these contentions.[30] First, there are no existing narrative water quality objectives in the Basin Plan that are implemented through the Permit's total coliform limitations or any of the related tertiary filtration requirements of the Permit.[31] Second, even if such narrative water quality objectives existed, there is no logic in the Permit or the Draft Order that reflects how, as a matter of law, those narrative water quality objectives

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[24] Memorandum dated January 4, 1994, to Regional Water Board Executive Officers, from William R. Attwater, Chief Counsel of the State Board, re: Guidance on Consideration of Economics in the Adoption of Water Quality Objectives, AR at SRCSD_CORR_1002, Larry Walker (1) Section 1 (Attwater Memorandum), p. 3.

[25] *Topanga Assn. for a Scenic Community v. County of Los Angeles* (1974) 11 Cal.3d 506, 515 (*Topanga*); see State Board Order WQ 95-4, *supra*, pp. 10, 13.

[26] *Topanga*, *supra*, 11 Cal.3d, pp. 514-515.

[27] Draft Order, p. 9.

[28] Draft Order, p. 9.

[29] Draft Order, p. 9.

[30] The statements also are inconsistent with the Draft Order's reference to the Regional Board's exercise of discretion in adopting the "conservative" tertiary filtration requirements. (Draft Order, pp. 5-6.)

[31] Indeed, the State Board itself has recognized that, even in an effluent-dominated water (EDW), tertiary filtration requirements are more stringent than necessary to implement any Basin Plan water quality objective. (See, e.g., State Board Order WQO 2002-0015, *supra*, pp. 33, 35.) (The Sacramento River is, of course, not an EDW.)

require the specific limitations or requirements in issue.[32] The suggestion that federal law mandates the requirements is unaccompanied by any rule or logic that would apply to other regulated discharges in the Central Valley, California, or the United States.

Thus, the Regional Board was required to comply fully with Water Code sections 13263(a) and 13241, and other principles of state law applicable to the adoption of the requirements in issue. By concluding otherwise, the Draft Order errs severely.

2. The Draft Order Is Based on Other Errors, Oversights, and Omissions

For the reasons identified above, the Draft Order is inadequate and the Permit's tertiary filtration requirements must be vacated.[33] There are, however, further issues in the Permit and Draft Order that are inaccurate, misleading, or otherwise inappropriate. In addition, both the Permit and Draft Order ignore significant, uncontroverted evidence that undermines their conclusions. In this regard, the Draft Order ultimately is a statement of position in favor of tertiary filtration. Thus, while the Permit and Draft Order fail to acknowledge or follow the requirements for developing permit requirements, the extra-legal case they advance is neither objective nor accurate.

a. Dilution of Effluent

Daily dilution of the SRWTP effluent in the Sacramento River is always greater than 20:1, and ordinarily it is considerably greater. The Permit findings, based on evidence in the record, identify the average dilution of effluent as 50:1, and the actual value is somewhat greater than that.[34]

Under these circumstances, it is the Regional Board's standard practice to impose effluent limitations of "23 MPN," not 2.2 MPN as it imposed here. During the five-year period ending

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[32] If it were true that, for example, the Permit total coliform limitations implement an existing narrative water quality objective, there would presumably be an analysis in the Permit that said as much. But there is not. (See also Petition, pp. 27-29.) Nor is there any analysis that in any way would support a conclusion that the various Permit requirements related to tertiary filtration are related to any adopted water quality objective.

[33] As described above, the State Board should direct that relevant Permit requirements shall instead be consistent with those identified in Regional Board staff's "Disinfection Alternative No. 1." (See p. 6, *ante*, and fn. 12.)

[34] Permit, p. F-74; see Staff Report, p. 30; see also AR at SRCSD_CORR_1002 (District's October 2010 Comments and Evidence Letter), pp. 8, 12.

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with adoption of the Permit, the Regional Board issued 18 permits to municipal wastewater dischargers who discharge to surface water where dilution is greater than 20:1. In *16 of those 18 situations*, the effluent limitations were "23 MPN," not 2.2 MPN.[35] And in the case of the two exceptions, the agencies receiving these permits had, for their own reasons, decided to construct tertiary facilities.[36] Thus, the Permit treats the District differently than other municipal agencies in the Central Valley region, without justification.[37]

The Draft Order is simply wrong when it states that DPH guidelines require or even recommend or propose a median 2.2 MPN to protect MUN when dilution is less than 100:1.[38] Rather, relevant here, for discharges to surface water that are used for food crop irrigation or body contact recreation, DPH has stated that it recommends 2.2 MPN when dilution of effluent is less than 20:1, and considers 23 MPN to be adequate and appropriate where dilution is 20:1 or greater. Thus, in a letter to the Regional Board dated April 8, 1999, DPH indicated it would consider

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[35] District's October 2010 Comments and Evidence Letter, p. 12.

[36] "List of Reviewed Region 5 Permits: Tertiary Coliform Limits and Available Dilution"; District's October 2010 Comments and Evidence Letter, pp. 12-13; Petition, pp. 30-31; see also *In the Matter of the Own Motion Review of Waste Discharge Requirements Order No. R5-2003-0031 for the City of Woodland*, State Board Order No. WQO 2004-0010 (June 17, 2004), pp. 9, 19 (State Board modified permit so as to require meeting tertiary filtration requirements only when effluent receives less than 20:1 dilution in receiving water).

[37] The Draft Order is very misleading when it states that the SRWTP is one of "three remaining" wastewater treatment plants that discharge within the Delta and "only provide secondary treatment to its effluent." (Draft Order, p. 2.) This statement implies that boundary drawn by the legislature in Water Code section 12220 is somehow relevant to treatment requirements. In fact, within the geographic area defined as the "Delta," there are numerous discharges where the Regional Board has found that the receiving water does not provide 20:1 dilution. (See Order No. R5-2008-0154 (City of Stockton), p. F-38; Order No. R5-2007-0113 (City of Lodi), p. F-32; Order No. R5-2009-0095 (City of Manteca), p. F-46; Order No. R5-2007-0036-01 (City of Tracy), p. F-46.) Thus, presence within "the Delta," is not a meaningful criterion, and the State Board should remove the misleading text from the Draft Order.

[38] Draft Order, p. 5. Here, the Draft Order inexplicably refers to a 1980 guidelines document that has been superseded and irrelevant for 25 years. (Draft Order, p. 5 and fn. 15.) Indeed, the State Board has even issued an order describing the document as having been "rescinded." (*Order Amending North Coast Regional Board Cease & Desist Order No. 85-35 for the City of Santa Rosa Subregional Wastewater Treatment, Reuse & Disposal Facilities*, State Board Order No. 2000-04 (March 15, 2000), ¶ 10.) In fact, in *1987*, the Department of Health Services (i.e., DPH) issued disinfection guidelines. (*Wastewater Disinfection for Health Protection*, Sanitary Engineering Branch, California Department of Health Services (Feb. 1987), Appendix D.) Under these guidelines, the recommendation for discharge to water used for domestic use was *23 MPN* where the average dilution is greater than 20:1. (*Id.*) In 1992, the DPH Office of Drinking Water notified its staff and the regional water quality control boards that certain provisions of the 1987 guidelines, including the 20:1 dilution ratio related to wastewater discharge where there is domestic use, were "no longer applicable" and the need "no longer exists." (Memorandum to Office of Drinking Water Management Staff, from Office of Drinking Water, Peter A. Rogers, Chief (Aug. 18, 1992) Re: Uniform Guidelines for Disinfection of Wastewater.) It further stated that the current guidelines "should only be referred to for recreational situations." (*Ibid.*) In summary, the source referred to in the Draft Order was reviewed, evaluated, and updated twice since 1980. It is plainly incorrect to cite an inapplicable document.

wastewater discharged to water bodies with identified beneficial uses of irrigation or contact recreation and where the wastewater receives dilution of more than 20:1 to be adequately disinfected if the effluent coliform concentration does not exceed 23 MPN/100 mL as a 7-day median and effluent coliform concentration does not exceed 240 MPN/100 mL more than once in any 30-day period. DPH reiterated this advice in a letter dated July 1, 2003: "A filtered and disinfected effluent should be required in situations where critical beneficial uses (i.e., food crop irrigation or body contact recreation) are made of the receiving waters unless a 20:1 dilution ratio (DR) is available. In these circumstances, a secondary, 23 MPN discharge is acceptable . . . . For wastewater discharges into streams that experience tidal influences, an instantaneous DR of less than 20:1 is acceptable as long as the average for each day exceeds 20:1."[39]

The Regional Board has restated and relied upon this approach in many other permits. For example:

> In a letter to the Regional Water Board dated 8 April 1999, DPH indicated it would consider wastewater discharged to water bodies with identified beneficial uses of irrigation or contact recreation and where the wastewater receives dilution of more than 20:1 to be adequately disinfected if the effluent coliform concentration does not exceed 23 MPN/100 mL as a 7-day median and if the effluent coliform concentration does not exceed 240 MPN/100 mL more than once in any 30 day period. In a subsequent letter dated 1 July 2003, DPH states that a *filtered and disinfected effluent should be required in situations where critical beneficial uses (i.e. food crop irrigation or body contact recreation) are made of the receiving waters unless a 20:1 dilution ratio is available. In these circumstances, a secondary, 23 MPN discharge is acceptable*. DPH considers such discharges to be essentially pathogen-free.[40]

As stated above, the daily dilution of SRWTP effluent in the Sacramento River is greater than 20:1 and the Permit characterizes average dilution as 50:1. There is no justification for the extremely "conservative"[41] tertiary filtration requirements. Further, had the District been

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[39] Letter dated July 1, 2003, to Thomas R. Pinkos, Executive Officer, RWQCB, from David P. Spath, Chief, Division of Drinking Water and Environmental Management, Department of Health Services, AR at SRCSD_CORR_2187, p. 1.

[40] Order No. R5-2010-0019 (City of Chico), pp. F-27 to F-28, emphasis in original.

[41] Draft Order, pp. 4-5.

discharging at its *full* permitted flow during the period January 1, 1998, through January 1, 2010, there would have been zero days with average dilution less than 20:1.[42]

The Draft Order attempts to rescue the Permit's departure from normal practice. For example, presumably evoking DPH's recommendation for 2.2 MPN when dilution of effluent is below 20:1 in water used for irrigation of food crops, the Draft Order refers to the existence of agricultural diversions within a few miles of the location where the SRWTP diffuser discharges.[43] Whatever specific crops may be irrigated with such diversions, the Draft Order completely ignores the specific evidence in the record (which is passively acknowledged in the Permit)[44] that agricultural diversions do not divert water having less than 20:1 dilution.[45] It is simply improper to paper over this uncontroverted evidence with vague generalities. The Draft Order also states, with no reference to any evidence whatsoever, that dilution "at or near" the outfall is "less than 20 to 1" and there is potential for "double dosing" due to tidal influence.[46] This is again a case of verbal arm-waiving in lieu of analysis. Of course, as in *any* discharge situation, there is *some* area where dilution is limited. For example, in the case of the District, there is a small area at the

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[42] District's October 2010 Comments and Evidence Letter, p. 12. Certain other material in the record that refers to the probability of occurrence of less than 20:1 dilution is based on calculations assuming the once-requested, increased permitted flow of 218 mgd ADWF. The value cited above is based on 181 mgd ADWF.

[43] Draft Order, p. 7.

[44] Permit, p. F-78 (undiluted effluent not drawn into agricultural intakes).

[45] During the course of Permit development, information on irrigation use of the Sacramento River was provided to the Regional Board. There is, first, uncontroverted evidence in the record from a knowledgeable engineer who works with 25 Reclamation Districts in the Delta. (See Letter dated December 15, 2004, to K. Landau, Regional Board, from R. Seyfried, SRCSD, re: NPDES Permit Responses to Comments Raised at Meeting of November 19, 2004, AR at SRCSD_CORR_210.) None of the types of pumps used for irrigation go much below the surface, with a typical depth between 5 feet and 10 feet below mean sea level. In fact, they are shallow enough that they run the risk of the pump cavitating at low tide. In addition, the pipes from these pumps do not stick out horizontally into the water. They draw water near the riverbank and, in general, outside the direct influence of the SRWTP effluent plume, which emanates from a diffuser located on the river bottom in the middle of the river.

Further, modeling (calibrated and validated with multiple dye studies) demonstrates that up to 700 feet downstream of the discharge, no effluent (diluted or undiluted) is present in the river within approximately 100 feet of either riverbank. Typically, dilution is far greater than 20:1. At Harmonic Mean Flows, the river:effluent flow ratio is 56:1 for 181 mgd of effluent flow. At critical low river flows as represented by the lowest 7-day average flow expected to occur once in ten-years (7Q10) (i.e., 5820 cfs), dilution is 21:1 at a discharge rate of 181 mgd. River flows as low as the 7Q10 occur infrequently. Between 1970 and 2009, river flow was at or below 5820 cfs approximately 0.58 percent of the time. (District's October 2010 Comments and Evidence Letter, p. 8.) In short, there is no evidence of any appreciable risk related to irrigation of food (or other crops) that would necessitate filtration.

[46] Draft Order, p. 4.

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diffuser outlets, on the bottom of the Sacramento River where dilution of effluent would be less than 20:1 but the effluent mixes rapidly in the river. Tidal influence can affect the dilution of effluent at a given location at a given moment. However, DPH's recommendation is that where there are tidal influences, "an instantaneous [dilution ratio] of less than 20:1 is acceptable as long as the average for each day exceeds 20:1."[47] The circumstances of the SRWTP meet this test. Further, to the extent there may, due to tidal influence, be any location where dilution is less than 20:1 at the surface, this is an entirely transient condition at a location where there is no effluent at all at other times.[48]

Finally, for unknown reasons the Draft Order refers to the potential for diversions for a peripheral canal or tunnel to be located downstream of the SRWTP discharge location.[49] This may relate to the erroneous statement in the Draft Order suggesting that there is a general DPH recommendation concerning dilution and treatment levels with respect to discharges to water that may be used for MUN; as stated previously, the Draft Order is simply wrong about that subject.[50]

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[47] Letter dated July 1, 2003, to Thomas R. Pinkos, Executive Officer, Regional Board, from David P. Spath, Chief, Division of Drinking Water and Environmental Management, AR at SRCSD_CORR_2187, p. 1.

[48] The Draft Order's reference to the uncommon occurrence of "double dosing" (Draft Order, p. 4 and fn. 12), is misleading and ultimately meaningless. Contrary to the suggestion of the Draft Order (fn. 12) "double dosing" does not imply doubling the concentration of pollutants in the river and there is no evidence cited that would support that it does. In fact, the District submits that record evidence demonstrates that the statement in footnote 12 is a very extreme overstatement. The record is replete with water quality modeling and dye studies that describe all hydrodynamic conditions and the ratio of background river water to effluent under a full range of hydrologic conditions. Although the Draft Order refers to no evidence whatsoever in its discussion of tidal influence, the District presumes the Draft Order *might* relate to argument submitted by other parties. For example, it was pointed out that the District measured an occurrence of dilution of 10:1 at the surface, 175 downstream of the diffuser. (Results of November 2007 Dye Study of Effluent Discharge to the Sacramento River at Freeport, California (Flow Science Incorporated), Attachment A, Brown & Caldwell Data Report (AR at SRCSD_CORR_0332), pp. A-19, A-20.) This is a transient condition at low flow influenced by a tidal reversal, a worst-case condition enduring for less than ½ hour. At all other times, there would be no influence at that location *at all*. (Thermal Plan Exception Justification for the Sacramento Regional Wastewater Treatment Plant (Robertson-Bryan, Inc., 2010), AR at SRCSD_CORR_0994, Att. 100, pp. 8, 9; Technical Memorandum re Revised Analysis of the Effect of SRWTP Effluent Discharge on Sacramento River Water Temperature (Flow Science Inc., 2010), AR at SRCSD_OTHER_166, pp. 4, 5.) In other words, the SRWTP contribution to risk at this location is less than the risk that would be suggested by the overall risk assessment in the February 2010 Risk Assessment Report discussed below.

[49] Draft Order, p. 7. Also, here, the Draft Order refers to a report dated November 18, 2010. (Draft Order, p. 7, fn. 27.) To the best of the District's knowledge, this document (Progress Report on the Bay Delta Conservation Plan (Nov. 18, 2010)) is not in the record and has not been proposed for inclusion in the record. Accordingly, citation to or reliance upon this document is improper. Nor are potential diversions for a peripheral canal "drinking water intakes."

[50] See p. 11, *ante*, and fn. 38.

It has been pointed out by others that the concentration of cysts or occysts in raw water can affect the amount of treatment that drinking water plants must provide under U.S. EPA's Long Term 2 Enhanced Surface Water Treatment Rule (LT2ESWTR). As described in the District's Petition, ambient water quality at diversions in the Sacramento River and Delta is *not* such as to require additional treatment under LT2ESWTR.[51] Further, the possibility of a peripheral canal is, by itself, irrelevant to this subject. Treatment requirements under the LT2ESWTR are not based on water in the Sacramento River at Freeport; they are based on water at the water treatment plant.[52] In short, the State Board should reject the Draft Order's implied contention that the Sacramento region must spend unreasonable sums in order to enable the peripheral canal.

In summary, with respect to the dilution of SRWTP effluent that occurs in the Sacramento River, the Permit is a departure from ordinary practice. While the District believes the general DPH 20:1 recommendation is conservative, its application to the District in the same manner as other dischargers would have yielded a different result. The Draft Order makes non-specific, undocumented, and unsuccessful attempts to identify some slight deviation from satisfaction of the conditions; in this respect, the Draft Order sets aside reasonable judgment and ignores the enormous burden the Permit would impose on the Sacramento region without a commensurate showing of reasonable protection of beneficial uses.

b. Quantitative Risk Assessment

As discussed in section A.2.a above, the Regional Board's standard permitting practice leads to a 23 MPN permit, not a 2.2 MPN permit. This conclusion is strongly supported by quantitative risk assessment and analysis by a national expert. A paragraph of the Draft Order includes discussion of a quantitative microbial risk assessment prepared during the development of the Permit. Overall, the Draft Order's treatment of this issue disregards Porter-Cologne and uncontroverted evidence, and for these and other reasons is improper. The District discusses these issues below.

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[51] Petition, p. 47, and fn. 147.

[52] See 71 Fed. Reg. 654, 657 (June 5, 2006); 40 C.F.R. § 141.703.

i. Context for Risk Assessment

Persons who ingest water directly from surface water bodies – such as in certain recreation activities – are at risk of acquiring gastrointestinal illness due to the presence of pathogens in the ingested water.[53] In this regard, U.S. EPA has identified acceptable levels of risk for all ambient surface waters, in its "Ambient Water Quality Criteria."[54] This U.S. EPA acceptable risk level is 0.8%, or 8 illnesses per 1000 bathers/swimmers.[55] The national criteria are applied extensively throughout the United States.[56] The Draft Order correctly states that the U.S. EPA recommendations are for public health protection from recreational contact with pathogens in waters subject to wastewater discharges.[57] The Draft Order is also correct that the U.S. EPA recommendations are not binding on states;[58] they were developed for use by states in establishing their own water quality standards.[59] The District does not insist that the U.S. EPA criteria must be applied to the Sacramento River. But the U.S. EPA recommendations are used extensively throughout the country[60] and at the very least provide valuable context and perspective. Further, risk levels from the U.S. EPA Recreation Criteria Document have been used in recent U.S. EPA regulations adopting *regulatory* criteria for various states. In 2000, Congress passed the Beaches Environmental Assessment and Coastal Health Act of 2000 (Pub.L. No. 106-284 (Oct. 10, 2000) 114 Stat. 870) (BEACH Act) which required states to adopt either the U.S. EPA 1986 Criteria or criteria "as protective" as the U.S. EPA recommendation. The

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[53] [Written] Testimony/Comments of Charles P. Gerba, Ph.D., Related to Draft NPDES Permit for the Sacramento Regional Wastewater Treatment Plant, submitted on October 11, 2010, AR at SRCSD_CORR_1002 (Gerba Written Testimony), pp. 1-2; Meeting, State of California, Central Valley Regional Water Quality Control Board, Partial Transcript (Dec. 9, 2010), Tiffany C. Kraft, CSR, AR at SRCSD_BM_13 (Hearing Transcript), p. 208:20-25.

[54] *Ambient Water Quality Criteria for Bacteria – 1986* (U.S. EPA, Jan. 1986, EPA440/5-84-002), AR at SRCSD_OTHER_370 (U.S. EPA Recreation Criteria Document).

[55] U.S. EPA Recreation Criteria Document, p. 9; Hearing Transcript, p. 210:21-25. As was pointed out by DPH, the February 2010 Risk Assessment Report inadvertently cited a 19 per 1000 swimmers threshold that applies to salt water rather than the 8 per 1000 acceptable risk that is applicable to freshwater recreation. The oversight is not material.

[56] See, e.g., Gerba Written Testimony, p. 5.

[57] Draft Order, p. 6.

[58] Draft Order, p. 6.

[59] See Petition, p. 37.

[60] Gerba Written Testimony, p. 5; see Hearing Transcript, p. 215:9-12.

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U.S. EPA's 2004 Water Quality Standards for Coastal and Great Lakes Recreation Waters promulgated water quality criteria for the remaining states that had not yet adopted protective criteria, putting in place regulatory criteria corresponding to an illness rate of 0.8% for swimmers (the U.S. EPA criteria value) in freshwater.[61]

As noted in the Draft Order, notwithstanding the Regional Board's ordinary practice, based on DPH recommendations, of requiring 23 MPN where there is substantial dilution, Regional Board staff also sought a further recommendation from DPH with regard to pathogens and disinfection.[62] Because *Cryptosporidium* and *Giardia* are less susceptible to inactivation by chlorine disinfection[63] than coliform, subsequent inquiry focused on the risk of illness from these organisms based on ingestion of river water. DPH staff initiated a preliminary evaluation of risk in the Sacramento River, but it was agreed that there were significant problems and uncertainties with that work.[64] DPH and Regional Board staff then endorsed the recommendation that an expert risk evaluation be conducted by Dr. Charles Gerba. Dr. Gerba is a Professor of Environmental Microbiology at the University of Arizona, and a renowned expert on microbial risk assessment, wastewater disinfection, and related issues. Among other things, he has produced over 500 articles, including textbooks, in environmental science and risk assessment. He has served as an advisor to multiple federal and state agencies, and conducts research on

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[61] 69 Fed. Reg. 67218, 67232 (Nov. 16, 2004), codified at 40 C.F.R. § 131.41 ("EPA is promulgating water quality criteria that correspond to an illness rate of 0.8% for swimmers in freshwater[.]").

[62] The District considers the request as an adjunct to the 20:1 policy that ultimately served to confirm the lack of need for filtration. (See also Letter dated June 9, 2009, to Ken Landau, Regional Board, from Robert Seyfried, SRCSD, re: Comments on Letter to Carl Lischeske (May 11, 2009) Requesting a Health Risk Assessment for Sacramento Regional Water Treatment Plant Discharge to the Sacramento River, AR at SRCSD_CORR_0441.)

[63] Footnote 14 of the Draft Order is not relevant and should be removed. (See Draft Order, p. 5.) The footnote refers to a study of particulate matter and ultraviolet disinfection of various wastewater effluents. (Draft Order, p. 5.) The SRWTP uses chlorine disinfection, followed by dechlorination prior to discharge. (Draft Order, p. 2.) Chlorine is a far more effective disinfectant than UV radiation in waters that contain particulate matter. (See also District's October 2010 Comments and Evidence Letter, p. 9.) In addition, and possibly related, the Draft Order refers to unspecified "issues with particle-associated pathogen indicators in the District's effluent." (Draft Order, p. 7.) In the absence of citation, it is impossible to know what the supposed "issues" referenced in the Draft Order are. Importantly, there is no evidence of any "issue" related to the effectiveness of the chlorine disinfection at the SRWTP under the circumstances, dosages, and contact times used in the disinfection.

[64] See, e.g., Letter dated August 23, 2010, to Ken Landau, Regional Board, from Stan Dean, SRCSD, re: Review of Department of Public Health Records Pertaining to SRCSD NPDES Permit Renewal Recommendation, AR at SRCSD_CORR_0707 (District's August 2010 Letter), p. 1.

microbial fate and transport in the environment and wastewater treatment.[65] DPH identified contact recreation as the most sensitive use for purposes of analysis: that is, if contact recreation is adequately protected, other uses that could be affected by pathogens will be protected.[66] With interaction and input by Regional Board staff and DPH, Dr. Gerba prepared a draft report and then a report dated February 23, 2010.[67] Dr. Gerba also subsequently submitted written testimony in October of 2010, and testified and presented evidence at the Regional Board hearing.[68] Dr. Gerba's work and testimony, none of which is disputed in the record, is discussed more specifically in section A.2.b.iii below.

ii. Draft Order's Abandonment of Porter-Cologne

The District believes that the Draft Order overlooks the requirements of Porter-Cologne and other state law in many respects. Among them is the statement regarding NPDES permitting that states that in California "CDPH determines the level of risk" and the State Board and Regional Board "establish waste discharge requirements that mitigate the risk to the level identified by CDPH."[69] This is not the law of California.[70] The Regional Board, in establishing waste discharge requirements, must comply with Water Code sections 13263(a) and 13241, and other applicable legal principles identified in section A.1. DPH recommendations should be considered appropriately in the Regional Board's discharge of its obligations. But DPH is not subject to the Water Code, and the Draft Order's suggestion that anything DPH might propose must be implemented, is illogical and inconsistent with law.

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[65] See Gerba Written Testimony, p. 1 and Attachments to Gerba Written Testimony; SRCSD Hearing Exhibits, PowerPoint, AR at SRCSD_BM_10, slide 30.

[66] See, e.g., Permit, p. F-75 ("DPH determined that if contact recreation is protected then agricultural irrigation and other Delta beneficial [sic] uses that could be impacted by pathogens would also be protected.").

[67] *Estimated Risk of Illness from Swimming in the Sacramento River*, Report for Sacramento Regional County Sanitation District (SRCSD), Charles P. Gerba, Ph.D. (Feb. 23, 2010), SRCSD_OTHER_147 (February 2010 Risk Assessment Report).

[68] Gerba Written Testimony, pp. 1-5; Hearing Transcript, pp. 208:14-221:20; SRCSD Hearing Exhibits, PowerPoint slides 31-40.

[69] Draft Order, p. 5.

[70] See, e.g., State Board Order WQO 2002-0015, *supra*, p. 36 (DPH recommendations not binding).

The Draft Order demonstrates that departure from state law can cause the eclipse of reason. For example, it identifies the DPH recommendation that SRWTP effluent not cause an incremental increase in risk to REC-1 uses of more than 1 in 10,000,[71] and states that the District's February 2010 Risk Assessment "indicated that the combined average risk of infection from *Giardia* and *Cryptosporidium* for one swimming exposure is reported as 2.4 in 10,000 upstream of the District's outfall and 3.6 in 10,000 downstream of the District's outfall."[72] The Draft Order here ignores the extreme conservatism of the recommendation, which is not employed in other regions or states and is far more stringent than the risk level that is the foundation for U.S. EPA national criteria. It also ignores uncontroverted evidence, discussed below, that indicates the extremely conservative recommendation to be *met* under the current 23 MPN discharge. But importantly here, the Draft Order does not appear to recognize what it is actually saying. Specifically, the Draft Order is asserting that the SRWTP represents a change in risk of 1.2 in 10,000. The logic of the Permit and Draft Order, then, is that a change in risk of 1.0 in 10,000 would be fine, but a change in risk of 1.2 in 10,000 means that the District must incur a $1.2 - 1.3 billion capital cost and additional increased annual operation costs of $45 million. Plainly, this is illogical.

Similarly, the Draft Order embraces a sweeping conclusion that "any" incremental change in risk whatever from exposure to wastewater from the SRWTP "does not protect beneficial uses" and thus, apparently is unacceptable.[73] This absolutist statement flouts Porter-Cologne and is not sensible. Likewise, the directive that the SRWTP's water be "essentially pathogen-free"[74] departs from Porter-Cologne, procedurally and substantively, as more fully set forth in section A.1.

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[71] Draft Order, pp. 6, 7.

[72] Draft Order, p. 6.

[73] Draft Order, p. 7.

[74] Draft Order, p. 8.

iii. The Draft Order Does Not Characterize the Evidence on Risk Assessment Completely or Fairly

Overall, the Draft Order does not recognize that the risks and changes in levels of risk under discussion all are quite low. In addition, like the Permit, the Draft Order avoids discussion of critical evidence including uncontroverted evidence. The District here provides context and discussion of evidence that has been avoided altogether.

(a) Preparation of Assessment

As identified above, Dr. Gerba performed a quantitative microbial risk assessment to determine the risk of acquiring gastrointestinal illness from *Giardia* and *Cryptosporidium* via ingestion of river water. The analysis relied upon standard microbial risk assessment methods.[75] The analysis calculated risks of illness based on compiled ambient water quality data from four locations: Veteran's Bridge, which is 8 miles upstream of the SRWTP discharge; Freeport (sometimes referred to as "Freeport Marina"), which is immediately upstream of the discharge; Cliff's Marina, which is approximately 0.5 miles downstream of the discharge; and River Mile 44, which is approximately 1.5 miles downstream of the discharge.[76] It also calculated risk of a 20:1 blend of upstream river water and effluent, a condition hypothetically assumed to exist at all times in the assessment.[77]

The report compared these risks to acceptable risk levels identified by U.S. EPA in U.S. EPA's "Ambient Water Quality Criteria."[78] As noted above, this U.S. EPA acceptable risk level is 0.8%, i.e., 8 illnesses per 1000 bathers/swimmers.[79] The report also notes that in the case

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[75] Gerba Written Testimony, p. 1.

[76] February 2010 Risk Assessment Report, pp. 4, 9; Hearing Transcript, pp. 213:21-214:1; SRCSD Hearing Exhibits, PowerPoint slides 36-39.

[77] February 2010 Risk Assessment Report, pp. 3-5; Hearing Transcript, pp. 211:12-18; SRCSD Hearing Exhibits, PowerPoint slides 37-39. As water moves further downstream, potential impacts attributable to the SRWTP discharge diminish. (See, e.g., Gerba Written Testimony, p. 3.) The February 2010 Risk Assessment Report, on page 5, relates certain data on the frequency of occurrence of dilution of 20:1. These frequencies are based on an assumed permitted 218 mgd ADWF rather than 181 mgd. The report was prepared before the District decided to withdraw its request for an increase to 218 mgd as permitted flow.

[78] U.S. EPA Recreation Criteria Document.

[79] U.S. EPA Recreation Criteria Document, p. 9; Hearing Transcript, p. 210:21-25. As was pointed out by DPH, the February 2010 Risk Assessment Report inadvertently cited a 19 per 1000 swimmers threshold that applies to salt

of recreational waters, risk of illness is used rather than risk of infection. Forty to fifty percent of persons infected actually experience a gastrointestinal illness.[80]

For purposes of the February 2010 Risk Assessment Report, very conservative, and conservatively compounding, assumptions were employed. For example, the February 2010 Risk Assessment Report used a conservative assumption with respect to the viability of *Giardia* cysts in SRWTP effluent. Not all the cysts or oocysts in measured water are viable (capable of causing an infection).[81] While no data exist on the percentage of *Giardia* cysts in secondary-treated wastewater that are viable, such data do exist for *Cryptosporidium* oocysts. This percentage value was used for *Cryptosporidium*, but it was also simply, and very conservatively, assumed in the February Report that an equal percentage of *Giardia* cysts from the SRWTP were viable.[82]

In addition, although the U.S. EPA acceptable or recommended risk levels (and the DPH recommendations) are based on one swimming or bathing exposure (also referred to as swimming activity day), the February 2010 Risk Assessment Report calculates risk from both one day of swimming activity and ten days of swimming activity.[83]

Also, the February 2010 Risk Assessment Report assumed that each individual swallows 100 mL of water during a day of swimming activity. This is *two to sixteen times greater* than amounts typically used in such risk assessments. U.S. EPA studies indicate that *37 mL* is a more appropriate value for a day of swimming. Nonetheless, the 100 mL assumption was applied throughout, unquestionably representing a very conservative assumption.[84]

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water rather than the 8 per 1000 acceptable risk that is applicable to freshwater recreation. The oversight is not material.

[80] February 2010 Risk Assessment Report, p. 9; Hearing Transcript, p. 209:5-7.

[81] February 2010 Risk Assessment Report, p. 7; Hearing Transcript, p. 212:6-12.

[82] February 2010 Risk Assessment Report, p. 7; Gerba Written Testimony, p. 3; Hearing Transcript, p. 212:15-18.

[83] Gerba Written Testimony, p. 2; Hearing Transcript, p. 212:18-19; SRCSD Hearing Exhibits, PowerPoint slide 34. The Draft Order does not reflect objectivity or fairness in its treatment of the February 2010 Risk Assessment Report's presentation of ten days of swimming activity. Specifically, the February 2010 Risk Assessment Report showed that even if the risk to swimmers were multiplied by a factor of ten, the risk would still be *far* below the U.S. EPA acceptable risk values. (February 2010 Risk Assessment Report, pp. 9-10.) That does *not* mean, as the Draft Order implies (Draft Order, pp. 6-7), that the risk of illness actually increases by a factor of ten.

[84] February 2010 Risk Assessment Report, p. 8; Gerba Written Testimony, p. 2; Hearing Transcript, pp. 212:20-213:2.

The resultant risk calculations are generally reflected in Tables 3-5 of the February 2010 Risk Assessment Report.[85] Thus, for example, referencing Table 4 and using the applicable conservative assumptions, the calculated average risk of illness from ingesting *Cryptosporidium* for a swimmer at Veteran's Bridge is 1.20×10^{-5} (or, 1.2 in 100,000), and at River Mile 44 it is 1.27×10^{-5} (or, 1.27 in 100,000).[86]

The February 2010 Risk Assessment Report found that for all scenarios evaluated, even combining risks from the two protozoa under the suite of conservative assumptions, the risk was below the U.S. EPA recreational criteria accepted risk value by two to three orders of magnitude.[87]

(b) Letter From DPH and Response

DPH wrote to Regional Board staff on June 15, 2010, after review of the February 2010 Risk Assessment Report.[88] DPH pointed out (not specifically referencing, but presumably using, Table 5 on p. 16 of the February 2010 Risk Assessment Report) that the calculated risk of illness reflected for swimmers was on average 1.3 per 10,000 at Veteran's Bridge (upstream), 1.2 per 10,000 at Freeport (upstream), 1.8 per 10,000 at Cliff's Marina (.5 mile downstream), and 3.4 per 10,000 at River Mile 44 (1.5 miles downstream).[89] The "bottom line" recommendation in the DPH letter was that SRCSD's effluent not cause an additional risk of infection greater than 1 in 10,000.[90]

In a letter of June 30, 2010, the District responded to the DPH letter, noting the extremely conservative nature of the DPH recommendation, the high cost of filtration, and the fact that the February 2010 Risk Assessment Report used extremely conservative assumptions. The District

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[85] February 2010 Risk Assessment Report, pp. 14-16.

[86] February 2010 Risk Assessment Report, p. 15.

[87] February 2010 Risk Assessment Report, pp. 9-10; Hearing Transcript, p. 211:18-20; SRCSD Hearing Exhibits, PowerPoint slide 33.

[88] Letter dated June 15, 2010, to Kenneth D. Landau, Regional Board, from Gary H. Yamamoto, P.E., DPH, re: Request for Health Risk Assessment for Sacramento Regional County Sanitation District (SRCSD) Discharge to Sacramento River, Sacramento County, AR at SRCSD_CORR_0573 (DPH June 2010 Letter).

[89] DPH June 2010 Letter, p. 2.

[90] DPH June 2010 Letter, p. 3.

also pointed out that even with all the conservative assumptions, the difference at .5 miles downstream was not statistically significant, and while the difference at 1.5 miles downstream was statistically significant, the value may be influenced by different factors such as the marina or other inflows. In addition, there were certain misstatements in the DPH letter that required clarification or correction. The District also noted that, even though the risk level recommendation proposed by DPH was extremely conservative, the level could be met if just one of the conservative assumptions were more realistic.[91] In written testimony subsequently submitted in October, Dr. Gerba states his agreement with the content of the District's June 30 letter in this regard as related to the microbial risk analysis, in addition to addressing additional topics discussed below.[92]

(c) Permit Discussion of February 2010 Risk Assessment Report and Uncontroverted Evidence

The Permit contains severe mischaracterizations or misunderstandings regarding the February 2010 Risk Assessment Report.[93] The Permit does not meaningfully consider the exceptionally small risks, or that they were the product of very conservative assumptions. Further, the Permit *does not address at all* Dr. Gerba's written testimony or testimony at the hearing which supplements the February 2010 Risk Assessment Report with further analysis. Nor is there any evidence disputing Dr. Gerba's analysis or testimony, a fact that undercuts much of the discussion in the Permit and a fact that is ignored in the Draft Order.

For example, as discussed below, the Permit does not consider in any way Dr. Gerba's uncontroverted testimony and analysis concerning inactivation of *Giardia* through the SRWTP treatment processes.[94]

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[91] See Letter dated June 30, 2010, to Ken Landau, Regional Board, from Stan Dean, SRCSD, Subject: California Department of Public Health letter dated June 15, 2010, AR at SRCSD_CORR_0594 (District's June 2010 Letter), pp. 2-4; see also District's August 2010 Letter. The District notes that in the District's June 2010 Letter (p. 3) there is discussion of the frequency of occurrence of 20:1 dilution, but this is based on assumed permitted flow of 218 mgd rather than 181 mgd.

[92] Gerba Written Testimony, p. 2.

[93] See Petition, pp. 37-40.

[94] See pp. 24-27, *post*.

As described above, the District explained in June of 2010 that if even one of the conservative assumptions employed for generating tables in the February 2010 Risk Assessment were made more realistic, the DPH recommendation, as stringent as it is, may well be met. The Tentative Permit released by Regional Board staff three months later (in September 2010) stated: "it is possible that further refinement of the Discharger's health risk assessment would demonstrate that the Discharger already achieves the health risk recommended by DPH."[95]

In October of 2010, the District transmitted written testimony of Dr. Gerba.[96] In his written testimony and testimony at the Regional Board hearing, Dr. Gerba described the preparation and outcomes of the February 2010 Risk Assessment Report. He expressed his conclusion and expert opinion that the "SRWTP discharge does not result in a meaningful increase in risk to recreationists of waterborne disease."[97]

In addition, Dr. Gerba explained that, subsequent to completion of the February 2010 Risk Assessment Report, he had also considered the effect of current SRWTP disinfection practices on the viability of *Giardia* cysts: "The impact of chlorination on the discharge from the [SRWTP] was not considered in this [February 2010 Risk Assessment Report's] assessment of *Giardia* viability. *Giardia is much more susceptible to inactivation by free chlorine and chloramines than Cryptosporidium*[.]"[98]

As described below, Dr. Gerba went on, in his October written testimony (which was incorporated as part of the District's Comments on the September Tentative Permit),[99] to discuss *Giardia* inactivation by the chloramines that are formed in the disinfection process.[100] Preliminarily, however, it requires emphasis that this information is uncontroverted in the record,

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[95] California Regional Water Quality Control Board, Central Valley Region, Tentative Order No. R5-2010-XXXX [NPDES No. CA0077682] Waste Discharge Requirements for the Sacramento Regional County Sanitation District, Sacramento Regional Wastewater Treatment Plant (Sept. 3, 2010) (hereafter, September Tentative Permit), p. F-75.

[96] Gerba Written Testimony; Hearing Transcript, p. 208:14-18.

[97] Gerba Written Testimony, p. 5; see Hearing Transcript, p. 215:14-19.

[98] Gerba Written Testimony, p. 3, emphasis added; see also Hearing Transcript, p. 215:14-19; SRCSD Hearing Exhibits, PowerPoint slide 40.

[99] District's October 2010 Comments and Evidence Letter, p. 14.

[100] Gerba Written Testimony, pp. 3-5.

and the Regional Board ignored it entirely. In this regard, the District's comment letter submitted in October simultaneously with Dr. Gerba's Written Testimony stated:

> However, *Giardia* is much more susceptible to inactivation by free chlorine and chloramines than *Cryptosporidium* and therefore would experience greater inactivation by chloramines in the SRWTP effluent before discharge . . . . Dr. Gerba provides further analysis and conclusions in accompanying material [i.e., the written testimony], which constitutes additional comment and evidence.[101]

The Regional Board "Staff Response to Comments" did not respond to this comment at all. This is significant because, alone, consideration of inactivation of *Giardia* results in risk values associated with the SRWTP being lower still than under the assumptions of the February 2010 Risk Assessment Report.[102]

Dr. Gerba's analysis, as described in his testimony, leads to the conclusion that in assessing in-river risks, the risk of illness from *Giardia* associated with the discharge is essentially eliminated, and the proper focus in assessing discharge-related risk is thus *Cryptosporidium*.[103] Dr. Gerba explained that chloramines are formed as a result of chlorine use in the disinfection process. He analyzed *Giardia* inactivation from chlorine/chloramines based on U.S. EPA guidance as a function of contact time and temperature of the SRWTP effluent. He confirmed that there are no in-river risks from *Giardia* attributable to the effluent. Accordingly, *Cryptosporidium*, not *Giardia*, is the appropriate microbe to consider in evaluating SRWTP's risks to recreaters from ingestion of river water.[104]

The data related to in-river risk from *Cryptosporidium* are in Table 4 of the February 2010 Risk Assessment Report,[105] and are depicted on PowerPoint slides 38 and 39 of SRCSD's Hearing Exhibits. The calculated risks for a swimming day are:

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[101] District's October 2010 Comments and Evidence Letter, p. 11, citation omitted.

[102] In addition, federal regulations require responses to significant comments raised during the public comment period. (40 C.F.R. § 124.17(a)(2).) Absence of a response to this evidence and comment is thus inconsistent with federal regulations.

[103] Hearing Transcript, pp. 213:16-19, 215:14-16, 221:8-20.

[104] Hearing Transcript, pp. 213:16-19, 215:14-16, 221:8-20; SRCSD Hearing Exhibits, PowerPoint slide 35 ("*Cryptosporidium* represents the only microbial risk from SRWTP discharge.").

[105] February 2010 Risk Assessment Report, p. 15.

Veteran's Bridge:	1.2:100,000
Freeport:	1.04:100,000
Cliff's Marina:	1.09:100,000
River Mile 44:	1.27:100,000[106]

Even assuming for the sake of argument that the differences are statistically significant (which they are not), they are trivial, and for each location the risk of illness is approximately *1:100,000*. The September Tentative Permit had recognized that refinement of the February 2010 Risk Assessment Report could support that the SRWTP already meets the extremely conservative DPH recommendation.[107] When such a refinement was presented, it was ignored.[108] It was completely inappropriate for the Regional Board to ignore the evidence, and it is inappropriate for the State Board to ignore it.

(d) Summary of Evidence Related to Risk Assessment

The District does not concur that the DPH "recommendation" for a change in risk of infection of no more than 1 in 10,000 is an appropriate basis for regulation. First, it advocates extremely costly treatment based on a risk value or change in risk that is unduly low. Indeed, the value is based on drinking water standards applicable to tap water, not recreation.[109] Second, the value is not based on consideration of ambient water quality conditions or the relative significance or insignificance of any change in water quality that may be caused by the SRWTP. In other words, it is disconnected from development of water quality-based effluent limitations (WQBELs) related to ambient water quality objectives. Third, DPH does not consider the factors provided in Water Code sections 13263(a) and 13241, which the Regional Board must do.[110]

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[106] February 2010 Risk Assessment Report, p. 15; SRCSD Hearing Exhibits, PowerPoint slides 38-39.

[107] September Tentative Permit, p. F-75.

[108] Instead, after submittal of the District's written comments and written testimony of Dr. Gerba in October 2010, the passage from the September Tentative Permit that had recognized that the conservative recommendation may be met, was *deleted* from the final revisions of the permit presented for Regional Board consideration in December. (See "Underline/Strikeout" version of the California Regional Water Quality Control Board, Central Valley Region, Order No. R5-2010-XXXX [NPDES No. CA0077682] Waste Discharge Requirements for the Sacramento Regional County Sanitation District, Sacramento Regional Wastewater Treatment Plant, Sacramento County (November Redline Tentative Permit), p. F-80.)

[109] See also Gerba Written Testimony, p. 2 ("In my experience spanning 33 years, I have not encountered a regulatory agency using a 1:10,000 risk threshold for contact recreation in surface waters.").

[110] See section II.A.1, *ante*, and II.A.3, *post*.

With that said, however, the uncontroverted evidence in the record is that the DPH recommendation *is met* with *current* treatment. In particular, the uncontroverted evidence is:

The SRWTP does not increase risk of illness from *Giardia* in the river, due to inactivation of *Giardia* in the specific disinfection circumstances of the SRWTP.

and

Increased risk of illness from *Cryptosporidium* contributed by the SRWTP is much less than 1 in 100,000.[111]

The Regional Board did not consider this evidence at all and the Draft Order does not consider it. Again, the District reiterates that the DPH position is inappropriate. However, that position was that the SRWTP not increase the risk of infection by more than *1* in *10,000*. There is uncontroverted evidence in the record that the SRWTP does not cause an increase of this magnitude.

3. Noncompliance With Obligations Under Porter-Cologne and Administrative Law

As described in section A.1 above, the Regional Board was required to comply with state law prior to consideration of tertiary filtration requirements. State law includes Water Code sections 13263(a) and 13241. Given that the Regional Board was proposing to adopt a permit/order with effluent limitations more stringent than necessary to implement Basin Plan water quality objectives, findings were required with respect to all factors identified in the Water Code. Those findings must be supported by evidence. The Regional Board was required to engage in a meaningful balancing of factors. And the findings must "bridge the analytic gap" between the raw evidence and ultimate decisions.[112] The tentative permit released by the Regional Board for public comment did not even address these requirements of the Water Code

[111] Translated to risk of infection, this would mean much less than 2 in 100,000. All the values discussed above ignore potential contribution of other sources between the point of discharge and River Mile 44. The Draft Order refers to the 2009 draft report that preceded the February 2010 Risk Assessment Report. (Draft Order, p. 6, fn. 22.) Based on the draft report, the incremental change in risk of illness associated with *Cryptosporidium* discharge would be between zero and 2.9 per 100,000. (*Estimated Risk of Illness from Swimming in the Sacramento River near Freeport*, Report for the Sacramento Regional County Sanitation District, Charles P. Gerba (Sept. 24, 2009), AR at SRCSD_OTHER_131 (2009 Draft Report), Table 3.)

[112] *Topanga, supra,* 11 Cal.3d at p. 515; see State Board Order WQ 95-4, *supra*, pp. 10, 13.

and state law.[113] After the District's comments noted the error, findings were included in the Permit as adopted.[114] Those findings are unexamined in the Draft Order as a result of the Draft Order's erroneous assumption that the tertiary filtration requirements were all somehow legally required.[115]

In fact, the relevant Permit findings are superficial, incorrect, unsupported by (or directly contrary to) the evidence, and not consistent with the Water Code. Immediately below, the District reiterates relevant discussion from the Permit, and also addresses other matters discussed above and in the Draft Order that may relate to these subjects. The District emphasizes that the Draft Order's failure to review the Regional Board's compliance with state law is a severe shortcoming.

a. Water Code Section 13263(a)

Under Water Code section 13263(a), the Regional Board must take into consideration, among other things, "the water quality objectives reasonably required" to protect beneficial uses. Nowhere does the Permit, or do findings in the Permit related to the filtration requirements, identify such water quality objectives or address this issue in any way. Nor does the Draft Order address this issue. For this reason alone, the Permit is unlawful.

b. Water Code Section 13241

In its hurriedly crafted and superficial Water Code section 13241 "findings," the Regional Board did no more than advocate advanced treatment. Each of the Water Code section 13241 factors, and the deficiencies of Regional Board's findings, is addressed below. Also, to the extent the Draft Order discusses issues that are related to findings, those issues are addressed below as well as in section A.2 of this memorandum.

Water Code section 13241(a) requires the Regional Board to consider the "[p]ast, present, and probable future beneficial uses of water." Here, the findings accurately list the

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[113] See Petition, p. 45; see also September Tentative Permit.

[114] District's October 2010 Comments and Evidence Letter, pp. 6-7; November Redline Tentative Permit, pp. F-77 to F-78; Permit, pp. F-74 to F-75.

[115] See section II.A.1, *ante*.

beneficial uses of the Sacramento River and Delta.[116] However, certain other discussion of beneficial uses merits attention.

In particular, Regional Board staff requested that the District conduct the recreational user risk assessment described in the District's Permit and below. As the Permit recites, contact recreation is considered the most sensitive use, such that, if it is protected, other beneficial uses will be protected.[117] However, the Permit as adopted and the Draft Order include generalized reference to irrigation use (AGR) and Municipal (MUN) use.[118] As discussed in section A.2.a, there is no evidence of any adverse effect to irrigation use; nor does the Draft Order cite any such evidence. With respect to municipal use, there is no evidence of any risk or any meaningful effect on risk to consumers of water of any kind; nor did DPH itself or anyone else identify any such risk as a concern. The nearest drinking water intake is the Barker Slough Pumping Plant, which is approximately 40 miles downstream of the discharge.[119] The California Urban Water Agencies (CUWA) stated that pathogens from the SRWTP "are not currently impacting drinking water quality/treatment[.]"[120] Similarly, a group of Delta export contractors recommended that disinfection requirements remain the same for existing flows.[121] The Permit refers to unspecified

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[116] Permit, p. 7.

[117] See, e.g., Permit, p. F-75 ("DPH determined that if contact recreation is protected then agricultural irrigation and other Delta beneificial [sic] uses that could be impacted by pathogens would also be protected.").

[118] November Redline Tentative Permit, p. F-76; Permit, p. F-78; Draft Order, p. 7.

[119] Permit, p. F-36. As stated in the District's October 2010 Comments and Evidence Letter (p. 11) and reflected in the record: *Giardia* and *Cryptosporidium* are not detected frequently in State Water Project waters according to the 2006 State Water Project Sanitary Survey. The source of waters for all of the drinking water treatment plants analyzed was classified as Bin 1 (no additional treatment required under LT2ESWTR). (District's October 2010 Comments and Evidence Letter, p. 11 [referencing California State Water Project Watershed Sanitary Survey, 2006 Update, prepared for the State Water Project Contractors Authority by Archibald Consulting, Richard Woodward Water Quality Consultants, Palencia Consulting Engineers (June 2007), AR at SRCSD_OTHER_208].)

[120] California Urban Water Agencies' February 1, 2010, Letter to K. Harder, *Comments on Issue Paper on NPDES Permitting Renewal Issues Drinking Water Supply and Public Health for the Sacramento Regional Wastewater Treatment Plant*, AR at SRCSD_CORR_0500, p. 2.

[121] Letter dated February 1, 2010, to Kathy Harder, Regional Board, from Walter Wadlow, Alameda County Water District, et al., re: Comments on Drinking Water Supply and Public Health Issues Concerning the Sacramento Regional Wastewater Treatment Plant NPDES Permit Renewal, AR at SRCSD_CORR_0499 (Wadlow Letter), p. 15. As of the date of the Wadlow Letter, the District had requested an increase in flow from the currently permitted flow of 181 mgd to 218 mgd, a request that was later withdrawn. (Letter dated February 1, 2005, from Wendell Kido, District Manager, SRCSD, to Ken Landau, Assistant Executive Officer, Regional Board subject: Application for NPDES Permit Renewal for the Sacramento Regional Wastewater Treatment Plant (SRWTP), NPDES Permit No. CA0077682, AR at SRCSD_OTHER_053; Letter dated June 11, 2010, from Mary Snyder, District Engineer,

"small drinking water systems throughout the Delta" and suggests such systems "may" divert surface water with no treatment at all.[122] Again, there is no evidence of such use or where it supposedly occurs, let alone any evidence of a risk of any kind caused by the SRWTP to any consumers of water. In short, the Permit (and Draft Order) suggestions regarding MUN use are a red herring. And as discussed in section A.2.a, references in the Draft Order to a possible peripheral canal intake are unexplained and irrelevant. As *DPH* identified, contact recreation is the appropriate focus.

In this regard, the District certainly concurs that the Regional Board should regulate for the reasonable protection of the REC-1 use. However, it was of little relevance for the Permit findings to say that the entirety of the Sacramento River and Sacramento-San Joaquin Delta support many recreational user days per year.[123] This number greatly overstates the use of the lower Sacramento River below the SRWTP discharge. The recreation use downstream of SRWTP mentioned in the Draft Order is sport fishing.[124] Risk calculations referred to in the February 2010 Risk Assessment Report and Permit are based on a day of swimming. Risks associated with fishing and boating are much lower still than those associated with swimming.[125] And, any effect on risk that could be attributable to the SRWTP diminishes as water moves downstream due to fate and transport processes and any additions of flow from other sources.[126] Again, the District does not dispute that downstream waters should have protection of the REC-1 beneficial use consistent with the Water Code, but the Permit is not forthright in regard to the

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SRCSD, to Pamela Creedon, Executive Officer, Regional Board re: Request for Change in Permitted Capacity for the Sacramento Regional Wastewater Treatment Plant (SRWTP), AR at SRCSD_CORR_0567; Permit, p. 4.) Both CUWA as cited in the preceding footnote and the individual contractors in the Wadlow Letter advocated filtration for increases in discharge above current actual flow levels up to the 218 mgd that was contemplated as of the time the letters were sent, but there was no technical justification offered for this position.

[122] November Redline Tentative Permit, p. F-77; Permit, p. F-78.

[123] Permit, p. F-95.

[124] Draft Order, p. 7.

[125] Gerba Written Testimony, pp. 2-3.

[126] Gerba Written Testimony, p. 3.

nature and extent of the affected recreational beneficial use. Discussion beyond saying REC-1 is a beneficial use must be objective.[127]

Water Code section 13241(b) requires the Regional Board to consider the "[e]nvironmental characteristics of the hydrographic unit under consideration, including the quality of water available thereto." The Regional Board failed to consider, or make findings on, this factor. The "findings" for section 13241(b) state that, "[t]he environmental characteristics of the hydrographic unit, including the quality of available water, will be improved by the requirement to provide tertiary treatment for this wastewater discharge."[128] This finding is meaningless. The hydrographic unit under consideration is, presumably, the lower Sacramento River. The quality of water available thereto would include background or upstream Sacramento River water quality. The Regional Board should have addressed levels of coliform or protozoa that exist in the absence of any discharge.

The Permit findings under section 13241(b) also state that tertiary treatment "will allow for the reuse of the diluted wastewater for food crop irrigation and contact recreation activities that would otherwise be unsafe according to recommendations from DPH."[129] The lower Sacramento River is not "unsafe," nor is there evidence that it is unsafe or has been pronounced unsafe by DPH or other health agencies. Again, the Permit findings do not address at all the existence of risks that exist without any discharge. The Regional Board's purported "finding" is merely another argument for advanced treatment, and is not in any way responsive to the Water Code. The Draft Order ignores this defect altogether.

Water Code section 13241(c) requires the Regional Board to consider the "[w]ater quality conditions that could reasonably be achieved through the coordinated control of all factors which affect water quality in the area." The new finding in the Permit on this issue is merely a

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[127] In addition, email correspondence from Mr. Lischeske of DPH dated July 27, 2009, states: "Since a relatively small number of people actually get in the Sacramento River below the SRCSD outfall, we don't have a large population to protect from exposure to the effluent." (AR at SRCSD_CORR_707, RE "Scope of Work for Health Risk Assessment for SRCSD.")

[128] Permit, p. F-79.

[129] Permit, p. F-79.

statement that "[f]ishable and swimmable water quality conditions can be reasonably achieved through the coordinated control of all factors that affect [sic] water quality in the area," with a description of categories of discharges.[130] The general recitation of the goals of the Clean Water Act, unaccompanied by any analysis, is insufficient. The Regional Board must address the quality of water that can be achieved in the lower Sacramento River. Further, there is simply no evidence that the Sacramento River and Delta are not "swimmable" today, or that the very minor effect on water quality from requiring filtration for the SRWTP discharge would convert the receiving water from "non-swimmable" to "swimmable." The Draft Order fails to evaluate this finding or underlying error.

Water Code section 13241(d) requires of the Regional Board to account for economic considerations. With regard to economics, the Permit "findings" include the following:

> The loss of beneficial uses within downstream waters, without the tertiary treatment requirement, which includes prohibiting the irrigation of food crops and prohibiting public access for contact recreational purposes, would have a detrimental economic impact.[131]

This finding borders on the absurd. There is no evidence whatsoever that any such prohibitions – which have never occurred – will occur, let alone any evidence of economic impacts. The Permit "finding" regarding section 13241(d) also merely recites a range of estimates of capital costs to SRCSD and its ratepayers of tertiary filtration, without any specific finding or consideration of consequences, reinforcing that the consideration of costs is perfunctory.[132] In its lone mention of any of the requirements of state law, the Draft Order at least implies[133] that the mere receipt and acknowledgment of evidence concerning costs is sufficient for compliance with Water Code section 13241(d). It is not sufficient simply to receive information and relate what the

[130] Permit, p. F-79.

[131] Permit, p. F-79.

[132] Permit, p. F-79; see also Attwater Memorandum, e.g., p. 3 (the obligation to take into account economic considerations includes "both the cost of providing treatment facilities and the economic value of development").

[133] In this regard, the Draft Order states on pages 8-10 that the Regional Board had no obligation to consider economics at all, and thus the fact that economic considerations "were presented" and commented upon was more than the law requires. This legal premise is incorrect and the Draft Order does not purport to articulate where the Regional Board's obligations under Water Code section 13241(d) are.

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information says. The costs of compliance is a *factor* that must be considered and if costs are significant, the Regional Board must articulate why the action is reasonable based on facts, logic, and other factors identified in the Water Code.[134]

The Permit also contains a finding identified under Water Code section 13241(d) and elsewhere stating that tertiary filtration for pathogens may also reduce concentrations of other pollutants.[135] There is no finding or *evidence* of any meaningful change in water quality that results with respect to other pollutants. In fact, the Permit actually ignores evidence that reductions in other pollutants from filtration would result in an immeasurable or *de minimus* change in ambient water quality. For example, the Permit vaguely states that filtration "will" reduce total organic carbon (TOC), without suggesting how much or whether there would be any meaningful benefit.[136] There is uncontroverted evidence in the record that the effects on water quality would be insignificant.[137]

The District stated in comments:

> Page F-75 of the Tentative Permit states that tertiary filtration will or may reduce discharge of other water quality constituents to an unspecified degree. The Regional Board has, of course, authority to require WQBELs where appropriate (and the Tentative Permit proposes WQBELs for some of the described water quality constituents). The Regional Board may not dictate how the District achieves compliance. The general reference to potential effects of filtration does

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[134] Attwater Memorandum, pp. 1, 5; see State Board Order WQO 2002-0015, *supra*, p. 31; State Board Order WQ 2001-16, *supra*, pp. 32-33; State Board Order WQ 95-4, *supra*, pp. 13-14.

[135] Permit, pp. F-77, F-79 to F-80. The Draft Order states without citation that the Permit made certain findings related to reductions of other pollutants. (Draft Order, p. 8.) But the Draft Order does not evaluate any such findings or suggest they are relevant. Nor does the Draft Order examine any actual evidence related to the issue.

[136] Permit, p. F-77.

[137] For example, incremental reduction in TOC concentrations resulting from advanced treatment technologies (including filtration) were specifically evaluated and modeled, and are reflected on pages 4-38 and 4-39 (Figures 4-16 and 4-17) of Technical Memorandum: Analysis of Costs and Benefits of Advanced Treatment Alternatives for the Sacramento Regional Wastewater Treatment Plant (Larry Walker Associates, May 2010), AR at SRCSD_CORR_547. In those figures, Train A and Train C include filtration. (*Id.*, p. III, Table ES-3.) Trains D and E include also reverse osmosis to varying degree. (*Ibid.*) The report finds that the very slight changes in receiving water concentrations, even with the reverse osmosis alternatives, would likely not be measurable. (*Id.*, pp. 4-37 to 4-38.) And, there is no basis whatever to suggest that this immeasurable change would have meaning for beneficial uses. Similar analyses were performed for other parameters mentioned in the Permit, with similar conclusions. (See *id.*, pp. 4-13 to 4-15 [copper], 4-40 to 4-41 [mercury].) It should be noted that the "improvement" shown in this report is overstated because there is an assumed discharge and treatment of 218 mgd. (*Id.*, p. III.) Similar to the vague suggestions regarding reduction of other pollutants, qualitative Permit references to "much cleaner" effluents are hollow and merely argumentative. (Permit, p. F-97.) Further, for all discharges, WQBELs should be developed in accordance with applicable law and policy.

> not support the requirement. With respect to BOD and dissolved oxygen specifically, the District has proposed that the SRWTP be regulated to limit discharge of oxygen-demanding substances. The Tentative Permit makes no demonstration that reductions in the listed constituents will provide an important incremental benefit in terms of compliance with objectives or protection of beneficial uses.[138]

Indeed, the Staff Response to Comments acknowledges that the "additional benefits" of filtration identified in the Permit, whatever they may be, are "not reasons for requiring the level of treatment."[139] Again, the Permit reflected a shotgun of arguments seeking to justify a result, without a hard look at real evidence, application of judgment, or adherence to state law. The Draft Order unfortunately accepts that approach.

Water Code section 13241(e) requires the Regional Board to consider "[t]he need for developing housing within the region." Major increases in utility costs or connection fees adversely affect housing. The Permit findings and discussion related to Water Code section 13241(e) ignore altogether any comment or evidence in the record of adverse effects on the need for developing housing in the region.[140] Instead, the finding is that the requirement "will not adversely impact the need for housing in the area any more than for other adjacent communities."[141] While the finding is vague, there is no evidence to support it. Further, the finding does not comply with the statute in any event, as the statute does not invite such comparisons to other communities, vague or otherwise. The finding goes on to say that "[t]he potential for developing housing in the area will be facilitated by improved water quality[.]"[142]

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[138] District's October 2010 Comments and Evidence Letter, p. 16.

[139] Response to Written Comments for Sacramento Regional County Sanitation District Sacramento Regional Wastewater Treatment Plan Tentative Waste Discharge Requirements (Dec. 9, 2010), AR at SRCSD_BM_01G (Staff Response to Comments), p. 17.

[140] See, e.g., Letter dated October 8, 2010, to Kenneth D. Landau, Regional Board, from Dennis M. Rogers, Building Industry Association, re: Comments on the Tentative Waste Discharge Requirements (NPDES Permit No. CA0077682) and Time Schedule Order for Sacramento Regional County Sanitation District (SRCSD), Sacramento Regional Wastewater Treatment Plant (SRWTP), AR at SRCSD_CORR_0992; see also District's October 2010 Comments and Evidence Letter, p. 15 (filtration requirements "would adversely affect the need to develop housing in the region, by driving up the cost of housing through increased connection fees and users charges which directly affect the cost of living in a house").

[141] Permit, p. F-80.

[142] Permit, p. F-80.

Again, there is no evidence in the record that would support that the extremely small change in Sacramento River quality that would result from filtration of the SRWTP discharge will facilitate the potential for housing at some (unspecified) location. The findings under this provision also again state that downstream water would not be "safe" for irrigation or recreation in the absence of filtration; as discussed above, this is unfounded.[143]

Water Code section 13241(f) requires the Regional Board to consider the "need to develop and use recycled water." The Regional Board failed to do so, and its finding is not supported by evidence in the record. The finding states that "[t]he need to develop and use recycled water is facilitated by providing a tertiary level of wastewater treatment that will allow for a greater variety of uses in accordance with CCR, Title 22."[144] The evidence does not support this finding. The District does not dispute that there is a broader range of potential direct re-use with tertiary effluent than secondary effluent. This does not, however, mean that recycling use (at some undefined location or locations) is promoted by requiring filtration of all flows at SRWTP (including even peak wet weather flows) prior to discharge to the Sacramento River. The Regional Board was informed by the District on this point as follows:

> The Regional Board must also consider the need to develop and use recycled water. (Wat. Code, § 13241(f).) Implementing full Title 22 tertiary treatment at SRWTP would significantly reduce the incentive and ability to recycle water, by diverting potential resources away from recycled water projects to a major filtration and disinfection treatment project. To the extent recycled water uses require tertiary effluent, the demand can be met by sizing facilities (or, potentially, constructing satellite or scalping facilities) to meet the demand. Demand for recycled water only equates to a fraction of SRWTP flow. Expensive, advanced treatment for the entire flow requires allocation of additional funds that do not serve projected recycled water needs. Thus, requiring full tertiary treatment at SRWTP would act as a substantial economic disincentive to the development and use of recycled water by the District and would hinder rather than facilitate the development of recycled water in the Sacramento region.
>
> Additionally, the District needs to partner with willing water purveyors to implement recycled water projects in their service areas since the District is not a water purveyor. Most of these water purveyors have other water supplies that are more readily available and less expensive compared to the use of recycled water at this time. Lack of funding is one of the key elements that affect the implementation of recycled water projects throughout the state and the Sacramento

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[143] Permit, p. F-80.

[144] Permit, p. F-80.

area. Thus, requiring full tertiary treatment at SRWTP will exacerbate this problem.[145]

The findings do not consider these facts, and the Staff Response to Comments document does not even address this comment and information.

The factors to be considered under Water Code section 13241 are not limited to those specifically enumerated in subdivisions (a)-(f).[146] In this instance, one other consideration is energy demand, which would include effects on greenhouse gas emissions. Uncontroverted evidence at the hearing established that the energy demands (ignoring construction itself) for operation of microfiltration facilities would be equivalent to the demand of 13,000 homes.[147] In its comments on the September Tentative Permit, the District stated, that, "energy demands associated with new treatment processes (and associated greenhouse gas emissions) must be considered to satisfy the Regional Board's obligations under sections 13241 and 13263 of the Water Code."[148]

The Staff Response to Comments does not respond to this comment at all, and the Regional Board ignored the issue.

In summary, the Permit is not in conformity with Water Code sections 13263(a) and 13241 and the Regional Board's other obligations under state law. In another circumstance, when the State Board concluded that the Regional Board could properly impose tertiary filtration on a discharge to an EDW, but the Regional Board had failed to comply with its obligations related to section 13241, the State Board remanded and stayed the tertiary filtration requirements.[149]

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[145] District's October 2010 Comments and Evidence Letter, p. 15.

[146] See Wat. Code, §13241 ("Factors to be considered . . . shall include, but not necessarily be limited to, [subdivisions (a)-(f)].").

[147] Hearing Transcript, p. 174:8-10.

[148] District's October 2010 Comments and Evidence Letter, p. 15.

[149] State Board Order WQO 2002-0015, *supra*, pp. 35, 75-76. It is notable that upon remand the Regional Board adopted a permit that differed from the permit that was challenged in regard to tertiary filtration. In particular, the Regional Board on remand did not require year-round tertiary filtration for discharges to the EDW in issue; rather, the requirement is seasonal. (Order No. R5-2008-0055-01 (City of Vacaville Easterly Wastewater Treatment Plant), pp. 9-11.)

4. Conclusion Related to Tertiary Filtration Requirements

The Permit's tertiary filtration requirements are unnecessary and unfounded. The State Board should order that they be vacated and replaced with the requirements of "Disinfection Alternative No. 1" as identified by Regional Board staff.[150] In the alternative and at the very least, the requirements should be remanded for a proper and lawful consideration of issues.

B. Effluent Limitations for Ammonia

As in the case of the tertiary filtration requirements, the Permit's effluent limitations for ammonia are based on regulatory flaws and technical flaws. The Draft Order does not resolve these problems. In this regard, the Draft Order focuses on denial of mixing zones but misses a fundamental issue before the State Board: "Has the Regional Board adopted WQBELs in accordance with all applicable requirements?" Below, the District addresses both technical and regulatory issues. The District does not here, however, restate all the points it has provided in the record and in the Petition with regard to "the science." The District emphasizes that the technical foundations for the Permit and Draft Order's statements about the science is lacking.[151] Without any modification of that position, the primary focus below is on the regulatory issues.

In brief, while the District strongly disputes the Regional Board and Draft Order's characterizations of "the science" pertaining to downstream impacts, the simple fact is that if there is potential for exceedance of the Basin Plan narrative water quality objective for toxicity, whether far downstream or elsewhere, at concentrations lower than the most commonly used numeric criteria (i.e., U.S. EPA's ammonia criteria), WQBELs must still be developed in accordance with federal and state law. The Regional Board must first properly interpret the narrative toxicity objective with an appropriate numeric criteria that is considered to be protective of the beneficial use, and then the Regional Board can derive effluent limitations with consideration of an approvable mixing zone.

In the Permit, the Regional Board presumed that ammonia loads and concentrations would remain the same as current levels and did not consider what effluent concentration is necessary to

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[150] See p. 6, *ante*, and fn. 12.

[151] See, e.g., Petition, pp. 55-124.

avoid adverse effects "far downstream" in Suisun Bay or anywhere else.[152] The Regional Board's failure to consider the effluent limitations actually needed to avoid effects in the "far downstream" areas resulted in the adoption of WQBELs equal to U.S. EPA criteria and arbitrary denial of mixing zones.[153] The Draft Order, although modifying the actual logic somewhat, would merely perpetuate the error.

1. General Regulatory Framework

There are certain matters that are beyond reasonable dispute that pertain to effluent limitations for ammonia. First, the effluent limitations are WQBELs. WQBELs are adopted in NPDES permits to ensure compliance with water quality standards.[154] Water quality standards consist of the designated uses of the waters and the water quality criteria necessary based on such uses.[155] The CWA delegates the development of water quality standards to the states.[156] Under Porter-Cologne, water quality standards consist of the beneficial uses to be protected, which are akin to "designated uses" under federal law, and water quality objectives, which are equal to water quality criteria as referred to under section 303(c) of the CWA.[157] Water quality objectives are defined to mean the "limits or levels of water quality constituents or characteristics which are established for the reasonable protection of beneficial uses of water or the prevention of nuisance within a specific area."[158] The CWA does not specifically define "water quality criteria" for purposes under section 303(c) of the Act; however, such water quality criteria are considered to be "elements of State water quality standards, expressed as constituent concentrations, levels, or narrative statements, representing a quality of water that supports a particular use."[159] When setting water quality criteria or the Porter-Cologne equivalent, "water quality objectives," the

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[152] See, e.g., Permit, pp. F-40 to F-41, F-54 to F-58, F-95, J-1 to J-12; Petition, p. 111.

[153] Permit, pp. F-40 to F-41; see Draft Order, p. 13; see, e.g., Petition, pp. 58-65.

[154] 33 U.S.C. § 1311(b)(1)(C); 40 C.F.R. § 122.44(d)(1).

[155] 33 U.S.C. § 1313(c)(2); 40 C.F.R. § 130.3.

[156] 33 U.S.C. § 1313(c)(2); 40 C.F.R. § 130.3.

[157] 33 U.S.C. § 1313(c)(2); Wat. Code, §§ 13050, 13241, 13242.

[158] Wat. Code, § 13050(h).

[159] 40 C.F.R. § 131.3(b).

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state must comply with both state and federal law.[160] Such water quality objectives are adopted into the Water Quality Control Plans, as developed for each region by the Regional Boards.[161] In this case, the applicable plan is the Basin Plan.

Second, with respect to ammonia, the applicable beneficial uses include those associated with aquatic life (e.g., warm freshwater habitat and cold freshwater habitat).[162] There are no numeric water quality objectives for ammonia in the Basin Plan.[163] Rather, the relevant water quality objective is the narrative toxicity objective,[164] which states:

> All waters shall be maintained free of toxic substances in concentrations that produce detrimental physiological responses in human, plant, animal, or aquatic life. This objective applies regardless of whether the toxicity is caused by a single substance or the interactive effect of multiple substances. Compliance with this objective will be determined by analyses of indicator organisms, species diversity, population density, growth anomalies, and biotoxicity tests of appropriate duration or other methods as specified by the Regional Water Board.
>
> The Regional Water Board will also consider all material and relevant information submitted by the discharger and other interested parties and numerical criteria and guidelines for toxic substances developed by the State Water Board, the California Office of Environmental Health Hazard Assessment, the California Department of Health Services, the U.S. Food and Drug Administration, the National Academy of Sciences, the U.S. Environmental Protection Agency, and other appropriate organizations to evaluate compliance with this objective.
>
> The survival of aquatic life in surface waters subjected to a waste discharge or other controllable water quality factors shall not be less than that for the same water body in areas unaffected by the waste discharge, or, when necessary, for other control water that is consistent with the requirements for "experimental water" as described in *Standard Methods for the Examination of Water and Wastewater*, latest edition. As a minimum, compliance with this objective as stated in the previous sentence shall be evaluated with a 96-hour bioassay.
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> In addition, effluent limits based upon acute biotoxicity tests of effluents will be prescribed where appropriate; additional numerical receiving water quality objectives for specific toxicants will be established as sufficient data become available; and source control of toxic substances will be encouraged.[165]

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[160] See 40 C.F.R. § 131.11; see also Wat. Code, §§ 13241, 13242, 13377. To distinguish between state-established water quality criteria adopted pursuant to section 303 of the CWA, and other federal water quality criteria which will be discussed later, we will hereafter refer to the Porter-Cologne term, "water quality objective."

[161] Wat. Code, §§ 13050(j), 13240, 13241.

[162] Permit, pp. 6-7.

[163] See Basin Plan, pp. III-2.00 to III-9.00.

[164] Permit, p. F-55.

[165] Basin Plan, pp. III-8.01-9.00, italics in original.

Separate and apart from water quality objectives (i.e., water quality criteria pursuant to section 303(c) of the CWA), are advisory-level water quality criteria developed and published by U.S. EPA under section 304(a) of the CWA.[166] Under section 304(a), U.S. EPA is required to develop and make available to the states criteria for water quality that reflect the latest scientific knowledge. These criteria are to be revised from time to time as necessary.[167] Such criteria and revisions thereto must be "published in the Federal Register and otherwise made available to the public."[168] Although the development of such criteria are required by the CWA, the water quality criteria developed under section 304(a) are themselves recommended criteria and they are not equivalent to state water quality objectives.[169] It is thus important to distinguish between the regulatory section 303 "criteria" and the non-regulatory section 304(a) "criteria." Here, the relevant section 303 criterion is the narrative water quality objective for toxicity. Adopted and potential numeric section 304(a) criteria are also relevant to the Permit and Draft Order.

For example, and directly applicable in this case, when establishing WQBELs through interpretation of a narrative water quality objective, the applicable federal regulation gives the states three options.[170] The second option, referred to as Option B, allows for the use of U.S. EPA's section 304(a) criteria and states: "Establish effluent limits on a case-by-case basis, using EPA's water quality criteria, published under 304(a) of the CWA, supplemented where necessary by other relevant information; . . . ."[171] Under Option B, the use of section 304(a) criteria is not required. "[], EPA is not requiring states to use EPA's water quality criteria. EPA is offering the water quality criteria as one of three options available to the state for interpreting and applying narrative water quality criteria. EPA's water quality criteria provide one reasonable

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[166] 33 U.S.C. § 1314(a).

[167] 33 U.S.C. § 1314(a)(1).

[168] 33 U.S.C. § 1314(a)(3).

[169] See *American Paper Institute, Inc. v. U.S. EPA* (D.C. Cir. 1993) 996 F.2d 346, 349 ("As required by the CWA, *see* U.S.C. § 1314(a)(1), the EPA has promulgated a set of *recommended* numeric criteria for certain listed pollutants that the states can, and quite often do, refer to in selecting appropriate criteria." [Emphasis in original.]).

[170] See 40 C.F.R., § 122.44(d)(1)(vi).

[171] 40 C.F.R. § 122.44(d)(1)(vi)(B).

approach for interpreting state narrative water quality criteria because EPA's criteria account for the effects of a pollutant on aquatic life and human health."[172]

When developing section 304(a) criteria for the protection of aquatic life beneficial uses, U.S. EPA follows its *Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses* (1985 Guidelines). In general, such development includes a thorough review of pertinent information to determine if there is sufficient, acceptable data to derive numerical national water quality criteria.[173] In addition to guiding U.S. EPA's development of 304(a) criteria, the 1985 Guidelines may also be used by the states to develop site-specific criteria when the numerical national water quality criteria may not be appropriate.[174] "Criteria produced by these Guidelines are intended to be useful for developing water quality standards, mixing zone standards, effluent limitations, etc."[175]

As is identified in the Permit and the Draft Order, U.S. EPA has developed section 304(a) criteria for ammonia. The currently applicable criteria are contained in the *1999 Update of Water Quality Criteria for Ammonia* (1999 Criteria). U.S. EPA is in the process of updating the ammonia criteria, however, the update is not yet final.[176]

2. State and Regional Board's Use of the 304(a) Criteria for Ammonia

As the regional water quality control boards began to fully implement the state's *Policy for Implementation of Toxics Standards for Inland Surface Waters, Enclosed Bays, and Estuaries of California* (2005) (SIP) and adopt WQBELs to protect narrative water quality standards such as the narrative toxicity objective, the regional water quality control boards have determined that the 1999 Criteria are appropriate for application throughout California.[177] Specifically, the

[172] 54 Fed. Reg. 23868 (June 2, 1989).

[173] 1985 Guidelines, p. iv; see also U.S. EPA's Technical Support Document for Water Quality-Based Toxics Control (TSD), p. 34 ("The development of national numerical water quality criteria for the protection of aquatic organisms is a complex process that uses information from many areas of aquatic toxicology.").

[174] See 1985 Guidelines, p. 3.

[175] 1985 Guidelines, p. 2.

[176] Permit, pp. J-3 to J-4; see also <http://water.epa.gov/scitech/swguidance/standards/criteria/current/index.cfm> (as of June 14, 2012).

[177] See, e.g., Order No. R5-2007-0036-01 (City of Tracy), p. F-35; Order No. R1-2011-0016 (Forestville Water District), pp. F-31 to F-36; Order No. R7-2010-0022 (City of Brawley), p. F-38. The water quality control plan for

Regional Board has found the 1999 Criteria to be protective in all cases, except for discharges from the SRWTP. This is true for others that discharge to the Delta.[178]

In the District's case, the Regional Board found that the 1999 Criteria were not sufficiently protective of far downstream aquatic life beneficial uses.[179] "[R]ecent studies on ammonia and the POD of the Delta indicate USEPA's criteria may not be adequately protective of some other sensitive resident Delta species."[180] To make these findings in the Permit, the Regional Board relied on studies conducted by Dr. Swee Teh.[181] Besides Teh's work, the Regional Board also identified other studies to argue the 1999 Criteria are not sufficiently protective. For example, the Regional Board refers to studies with respect to ammonia inhibition of nitrogen uptake by diatoms in Suisan Bay.[182] The primary study relied on by the Regional Board for this hypothesis was prepared by Dr. Richard Dugdale and others.[183]

The Draft Order proposes to uphold the Regional Board's findings, stating, among other things, that standard U.S. EPA toxicity testing procedures for the Teh studies were followed "as much as possible."[184] Although the Regional Board and the Draft Order both find that the 1999 Criteria are not protective, and both refer to Teh's work and Dugdale's work, neither entity

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the Los Angeles Region incorporates the 1999 Criteria as water quality objectives for ammonia. (Water Quality Control Plan, Los Angeles Region, Basin Plan for the Coastal Watersheds of Los Angeles and Ventura Counties (1995), Amendments to the Water Quality Control Plan - Los Angeles Region with respect to Inland Surface Water Ammonia Objectives (April 25, 2002) <http://www.waterboards.ca.gov/losangeles/board_decisions/basin_plan_amendments/technical_documents/2002-011/03_1020/03_1020_dn__amonia_NH3_BPA_%20(clean)_%20020403.pdf> (as of June 14, 2012), at pp. 1, 3 and 5.)

[178] See, e.g., Order No. R5-2007-0036-01 (City of Tracy), p. F-35; Order No. R5-2007-0113 (City of Lodi), pp. F-23 to F-24; Order No. R5-2008-0179 (Town of Discovery Bay Community Services District), p. F-18 to F-19; Order No. R5-2009-0095 (City of Manteca), pp. F-40 to F-41; Tentative Order R5-2012-XXXX (City of Modesto), adopted on June 7, 2012, pp. F-50 to F-52.

[179] Permit, pp. J-1 to J-2; see Draft Order, p. 13.

[180] Permit, p. J-2.

[181] Permit, pp. J-2 to J-3.

[182] Permit, pp. F-56, pp. J-5 to J-7.

[183] Permit, p. J-5.

[184] Draft Order, p. 16.

used this information to actually modify the 1999 Criteria, which can occur and would be consistent with U.S. EPA's 1985 Guidelines.[185]

Rather than revising the 1999 Criteria after determining that they were insufficient, the Regional Board continued to use the 1999 Criteria to determine reasonable potential and calculate WQBELs, and then denied an otherwise approvable mixing zone. As is discussed in detail below, this approach does not comply with state and federal regulations for establishing WQBELs.

3. Regulatory and Legal Deficiencies

To illustrate the regulatory and legal issues associated with the ammonia limitations in the Permit, and as is proposed to being upheld in the Draft Order, the District assumes (hypothetically and for purposes of illustration only) that there is perfect consensus on various scientific issues relied upon in the Permit and Draft Order. Thus, for example, the District assumes that "the science" demonstrates that there are negative effects on the copepod *P. forbesi* in Suisun Bay from ambient ammonia concentrations that are far lower than those identified for the protection of aquatic life in the 1999 Criteria.[186]

As explained in the Introduction, the ultimate logic of the Permit is: "an effluent limitation of 33 mg/L as an average monthly limit is too high; therefore the effluent limitation shall be 1.8 mg/L as an average monthly limit."[187] This proposition is, of course, a non sequitur. Assuming 33 mg/L is "too high," why is the answer 1.8 mg/L? Why not 13 mg/L or 7 mg/L? The answer is that the Permit's imposition of the WQBEL relies on slight of hand, which is endorsed and perpetuated – albeit in a different form – in the Draft Order.[188]

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[185] 1985 Guidelines, p. 3 ("In addition, with appropriate modifications these National Guidelines can be used to derive criteria for any specific geographical area, body of water (. . .), or group of similar bodies of water, if adequate information is available concerning the effects of the material of concern on appropriate species and their uses.").

[186] See Draft Order, p. 16 (". . . ammonia concentrations as low as 0.36 mg/L of nitrogen impaired *P. forbesi's* reproduction and juvenile life-stage survival.").

[187] The values cited here for purposes of illustration are the Permit's interim and final average monthly effluent limitations for ammonia. (Permit, pp. 16 (Table 7), 14 (Table 6).) The interim effluent limitation is treated here as current discharge quality for purposes of illustration. (See also, Permit, p. F-104.)

[188] Relevant to the discussion here are the implications of such WQBELs, which would result in the need to provide full nitrification of the SRWTP discharge. Footnote 71 of the Draft Order states that the Regional Board required full nitrification, presumably because of the WQBELs adopted. Footnote 71 of the Draft Order is incorrect and

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a. Process for Calculating WQBELs

In the normal course of developing WQBELs, the calculation of the actual WQBEL is the last step of the process. Prior to calculating the WQBEL, the Regional Board shall: (1) identify the applicable water quality objectives; (2) identify the lowest objective for the pollutant at issue; (3) identify effluent and ambient data for the pollutant; and, (4) determine if there is reasonable potential for the effluent to cause or contribute to a violation of the lowest objective for the pollutant at issue.[189] If it is determined that there is reasonable potential, then the Regional Board must calculate a WQBEL.[190] Consideration with respect to the use of mixing zones and dilution credits in this process varies slightly, depending on the type of pollutant and the policy used by the Regional Board. The SIP applies to priority toxicant pollutants.[191] Ammonia is not one of 126 priority pollutants that are automatically subject to the SIP.[192] Thus, the Regional Board had discretion to calculate WQBELs by using the procedures established in the SIP, or those provided in the TSD.[193] Under the SIP procedures, if there is reasonable potential, the Permit must include a WQBEL.[194] The dilution credit is then factored into the WQBEL calculation if a mixing zone is granted.[195] Under the TSD approach, the establishment of a mixing zone and dilution credits may be used as part of the reasonable potential step.[196] In other words, under the TSD, where there is

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inappropriate for technical, regulatory, and legal reasons. First, the footnote has a statement that partial nitrification would only convert the ammonia to nitrate is unsupported by any evidence and the District disputes the statement. It is not correct or, at a minimum, is not necessarily correct. Second, the footnote refers to oxygen demand as a justification for "full" nitrification. The Basin Plan contains a water quality objective for dissolved oxygen. To the extent the Regional Board seeks to regulate for dissolved oxygen, it can and must comply with applicable laws and regulations for the development of WQBELs. In addition, the Regional Board and State Board may prescribe effluent limitations but may not require a particular form of treatment. (Wat. Code, § 13360.)

[189] The steps identified here summarize the steps as set forth in the SIP and the TSD. (SIP, pp. 6-7; TSD, pp. 47-49, 51-59.) Under the TSD, dilution is considered as part of the reasonable potential analysis. (TSD, pp. 47, 49, 52-53.) However, when using the SIP procedures, dilution is considered when calculating the WQBEL. (SIP, pp. 7-13.)

[190] 40 C.F.R. § 122.44(d)(1); SIP, p. 6.

[191] SIP, p. 3.

[192] Draft Order, p. 11.

[193] Draft Order, p. 11.

[194] SIP, p. 6.

[195] SIP, pp. 7-13.

[196] See, e.g., TSD, pp. 52-53.

sufficient dilution and a mixing zone is established, there may not be reasonable potential for the pollutant in question after consideration of dilution and no WQBEL is necessary.[197]

Dilution is factored into a WQBEL calculated under the SIP after the Regional Board has determined whether granting a mixing zone is appropriate.[198] Further, when calculating WQBELs for the applicable water quality objective, one does not simply affirm that there is evidence of concern with the current pollutant load. Instead, WQBELs are calculated in a manner to determine what is the quality of effluent necessary to meet the water quality objective, thereby protecting the beneficial use.

Specifically, once the Regional Board has determined that there is reasonable potential, the Regional Board must then calculate the actual WQBELs.[199] The actual calculation of the WQBEL includes the applicable water quality objective, which must be expressed numerically.[200] According to the federal regulations, when reasonable potential occurs due to a violation or potential violation of a narrative criterion, as is here, the Regional Board has three options for deriving the numeric value that is necessary for establishing such limits.[201] The options are: "(A) Establish effluent limits using a calculated numeric water quality criterion . . . using a proposed State criterion, or an explicit State policy or regulation interpreting its narrative water quality criterion . . . ; (B) Establish effluent limits on a case-by-case basis, using EPA's water quality criteria, published under section 304(a) of the CWA, supplemented where necessary by other relevant information; or (C) Establish effluent limitations on an indicator parameter for the pollutant of concern; . . . ."[202] In theory, the numeric value used to interpret a narrative water quality objective to determine reasonable potential, and the numeric value used to interpret the narrative water quality objective to calculate WQBELs could be different. However, as a practical matter, the same numeric value is used in both steps. Accordingly, a fundamental step

[197] See, e.g., TSD, pp. 52-53.

[198] See SIP, p. 15.

[199] 40 C.F.R. § 122.44(d)(1).

[200] SIP, pp. 7-8.

[201] 40 C.F.R. § 122.44(d)(1)(vi).

[202] 40 C.F.R. § 122.44(d)(1)(vi).

in this process is to interpret the narrative objective with the numeric value for determining reasonable potential, and for calculating WQBELs if reasonable potential exists.

b. Standard Process for Considering Mixing Zones

The federal regulations provide the states discretion with respect to establishing mixing zone policies.[203] Such policies exist in California, and two are applicable here, the SIP and the Basin Plan.[204] The mixing zone policy in the Basin Plan incorporates the procedures and guidelines from U.S. EPA's Water Quality Standards Handbook, and the TSD. As indicated in the Draft Order, for non-priority pollutants such as ammonia, the Regional Board may use the SIP and the TSD as guidance for "determining whether and to what extent to allow dilution credits and a mixing zone."[205] When using either one, or both, the Regional Board is required to clarify the methodology used in the Permit's findings.[206] In this case, the Permit references both policies. However, the Permit's determinations with respect to mixing zones and dilution credits more closely follow the SIP, except for statements in the Permit alleging that the Regional Board considered the procedures and guidelines in the TSD for determining the size of the mixing zone.[207] The Draft Order suggests that the Regional Board used both as guidance.[208] Because the Permit more closely follows the SIP, and the policy considerations in the TSD with respect to granting mixing zones are similar, the standard process here is framed in accordance with the SIP.

The SIP, which has been approved by U.S. EPA, establishes a methodology for considering mixing zones and dilution credits. In general, the SIP provides that the allowance of a mixing zone "is discretionary and shall be determined on a discharge-by-discharge basis."[209] As a first step, it must be determined if there is receiving water available to dilute the discharge.[210]

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[203] 40 C.F.R. § 131.13.

[204] See, e.g., Permit, p. F-28.

[205] Draft Order, p. 11.

[206] *In the Matter of the Petition of Yuba City*, State Board Order WQO 2004-0013 (July 22, 2004), p. 6.

[207] Permit, pp. F-38, F-40.

[208] Draft Order, p. 11.

[209] SIP, p. 15.

[210] SIP, p. 15.

Such a determination is made on a pollutant-by-pollutant basis. The SIP then provides a table of critical receiving flows applicable to year-round mixing zones for the three types of objectives included in the California Toxics Rule: acute aquatic life, chronic aquatic life, and human health. Relevant to ammonia here is the chronic aquatic life criteria.[211] The SIP incorporates two different approaches for determining if mixing zones are appropriate based on if the discharge is completely-mixed or incompletely-mixed.[212] Discharges from the SRWTP are considered to be subject to the approach for incompletely-mixed discharges.[213]

Under this approach, the Regional Board may consider approving mixing zones and granting dilution credits after the discharger has completed an independent mixing zone study.[214] There is no dispute that such a mixing zone study was prepared and approved for use by the Regional Board in the development of this Permit.[215]

The SIP includes 11 conditions that must be met in granting a mixing zone.[216] In its consideration, the Regional Board evaluated all 11 conditions with respect to chronic aquatic life criteria.[217] Based on its evaluation of the 11 conditions, the Regional Board alleged that three conditions were not met for ammonia.[218] According to the Permit, the Regional Board used these three conditions as its basis for denying a chronic mixing zone for ammonia.[219] The Draft Order calls out two of these and effectively supports the District's position that the Regional Board's use of two of the conditions ("compromise the integrity of the entire water body" and "adversely impact biologically sensitive or critical habitats") was not applicable to the Regional Board's

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[211] Permit, pp. F-36 to F-38.

[212] SIP, p. 16.

[213] Permit, p. F-31.

[214] SIP, pp. 16-17.

[215] Permit, pp. F-33 to F-34; see also Letter dated April 2, 2009, to Mary K. Snyder from Kenneth D. Landau re: Acceptance of Sacramento Regional County Sanitation District's Dynamic Mathematical Model for Use in NPDES Permit Renewal for the Sacramento Regional Wastewater Treatment Plant (AR at SRCSD_CORR_0422.)

[216] SIP, p. 17.

[217] Permit, pp. F-36 to F-38.

[218] Permit, pp. F-36 to F-37.

[219] Permit, pp. F-40 to F-41.

denial of a mixing zone for ammonia.[220] Lastly, the SIP states that a regional board shall deny or significantly limit a mixing zone and dilution credit as necessary, "to protect beneficial uses, meet the conditions of this Policy, or comply with other regulatory requirements."[221]

In all cases, the Regional Board's use of discretion with respect to denying mixing zones and dilution credits is not unfettered. As directed by the State Board, ". . . while regional boards have discretion in allowing mixing zones and dilution credits, they must explain the denial of a mixing zone based on facts of the discharge."[222] While any environmental impact of granting a mixing zone must of course be considered, the cost to the discharger of denying a mixing zone is also among the factors to be considered.[223] Moreover, regional board decisions must be supported by evidence in the record, and such decisions must be supported by findings that link their ultimate conclusions to the evidence.[224]

Accordingly, the Regional Board must follow the applicable policies, specifically identify the methodologies used, and support its decisions with findings that "bridge the analytic gap"[225] between the facts used and its decision when denying a mixing zone and dilution credits.

c. SRCSD Permit Process

The required process for calculating WQBELs and consideration of mixing zones contrast with what occurred in the Permit and with what the Draft Order would conclude. First, the Regional Board used the SIP methodologies with respect to determining reasonable potential, considering mixing zones, and calculating WQBELs.[226] Under step 1 of the SIP, the narrative toxicity objective is the most stringent applicable objective. The Regional Board identified the 1999 Criteria as the basis for its reasonable potential analysis under the SIP.[227] By virtue of

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[220] See Draft Order, p. 12, fn. 51.

[221] SIP, p. 17.

[222] State Board Order WQO 2004-0013, *supra*, p. 10.

[223] State Board Order WQO 2004-0013, *supra*, p. 12.

[224] *Topanga*, *supra*, 11 Cal.3d at p. 513.

[225] *Topanga*, *supra*, 11 Cal.3d at p. 515.

[226] Draft Order, p. 11; Permit, p. F-45.

[227] Permit, pp. 54-55.

selecting the 1999 Criteria here for step 1, the Regional Board determined that the 1999 Criteria were equivalent to the narrative toxicity objective, the "lowest (most stringent) water quality criterion or objective for the pollutant applicable to the receiving water."[228] However, as discussed previously, the Regional Board has asserted that it had sufficient evidence showing that the 1999 Criteria were not sufficiently protective.[229] If the 1999 Criteria are not sufficient, then the Regional Board improperly selected the 1999 Criteria in step 1 of the SIP.

After identifying the 1999 Criteria for step 1, the Regional Board then determined that there was reasonable potential for effluent from the SRWTP to cause or contribute to a violation of the narrative toxicity objective.[230] After finding reasonable potential, and since the Regional Board declared that it was following the SIP, the Regional Board was then required to determine if it was appropriate to grant a mixing zone.[231] When determining if a mixing zone is appropriate, the Regional Board must consider if there is assimilative capacity for the pollutant at issue. This determination is made, primarily, by comparing the lowest (most stringent) objective identified in step 1 of the SIP to the receiving water. Because the Regional Board identified the 1999 Criteria as representing the lowest (most stringent) objective in step 1 of the SIP, it is also the value that the Regional Board should use to determine if assimilative capacity exists.[232] Although not specifically articulated in the Permit, there is no dispute that there was assimilative capacity in the receiving water using the 1999 Criteria and that the receiving water would meet the 1999 Criteria at the edge of the proposed mixing zone for ammonia.[233] Moreover, the Regional Board acknowledged that the proposed chronic mixing zone was approvable, and that dilution of 29:1

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[228] SIP, p. 6.

[229] Draft Order, p. 13 ("The record indicates that existing levels of ammonia in the receiving water are not protective of aquatic life beneficial uses downstream of the discharge even though the receiving water does not exceed the 1999 Criteria.")

[230] Permit, p. F-55.

[231] SIP, p. 6.

[232] See SIP, pp. 15-16; Permit, pp. F-40 to F-41, F-54 to F-55, J-1; Draft Order, p. 11.

[233] Permit, p. J-1.

was otherwise available for chronic criteria.[234] However, despite this acknowledgment, the Regional Board denied the mixing zone because it would "compromise the integrity of the entire water body," "adversely impact biologically sensitive or critical habitats," and "produce undesirable or nuisance aquatic life."[235] The Regional Board's denial for these three reasons, which are three of the eleven conditions in the SIP, are not explained in the Permit and no evidence is provided to show why these conditions are not met for ammonia. The Regional Board's failure to explain its decision here fundamentally fails to comply with existing law, which requires the Regional Board to include findings that explain its decision.[236]

Although not expressed in the "mixing zone" section of the Permit (i.e., Permit, pp. F-36 to F-40), the Regional Board allegedly also denied mixing zones for ammonia "due to concerns of toxicity impacts attributed to the heavy loading of ammonia into the Delta, downstream of the discharge, referred to as 'far-field impacts'."[237] Further, other considerations for denial of the mixing zone, as expressed in the Permit, are not specifically related to the SIP's 11 conditions, but relate to why the 1999 Criteria were not protective in downstream waters outside the proposed mixing zone. For example, the reasons for denial include:

> (1) Recent studies suggest that ammonia at ambient concentrations in the Sacramento River, Delta and Suisun Bay may be acutely toxic to native *Pseudodiaptomus forbesi (copepod)*.
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> * * * *
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> (3) Recent studies provide evidence that ammonia from the SRWTP discharge is contributing to the inhibition nitrogen uptake by diatoms in Suisun Bay.
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> * * * *

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[234] See Permit, p. F-36 ("[t]he chronic mixing zone meets the requirements of the SIP as follows: . . . ."); see also Permit, p. F-39 (dilution credit of 29:1); see also Permit, p. F-55 ("As discussed in Section IV.C.2.d of the Fact Sheet, an allowance for chronic aquatic life dilution may be granted.").

[235] Permit, p. F-40. The Draft Order is critical of the Regional Board's denial of the mixing zone based on the first two reasons. (See Draft Order, p. 12, fn. 51 ["These reasons from the SIP have their origin in the TSD and are [sic] more aptly address the sizing of an approved mixing zone rather than the initial approval or denial of a mixing zone."].) The Draft Order is silent with respect to the third issue regarding nuisance.

[236] *Topanga*, *supra*, 11 Cal.3d at p. 513; State Board Order WQO 2004-0013, *supra*, p. 10.

[237] May 4, 2011 Memorandum to James Herink, Staff Counsel, from Pamela C. Creedon, Executive Officer, Subject: Petitions for Review of Waste Discharge Requirements, Order No. R5-2010-0114 (NPDES No. CA0077682) and Time Schedule Order No. R5-2010-0115, Sacramento Regional County Sanitation District, Sacramento Regional Wastewater Treatment Plant, Sacramento County, SWRCB/OCC Files A-2144(a) and A-2144(b), Central Valley Water Board Response (Regional Board Response to Petitions), p. 34.

(5) Downstream of the discharge point, ammonia may be a cause in the shift of the aquatic community from diatoms to smaller phytoplankton species that are less desirable as food species.

(6) Regardless of whether ammonia is directly or indirectly contributing to the POD, ammonia is shown to affect adult *Pseudodiaptomus forbesi* reproduction at concentrations greater than or equal to 0.79 mg/L. And nauplii and juvenile *Pseudodiaptomus forbesi* are affected at ammonia concentrations greater to or equal 0.36 mg/L. These ammonia concentrations can be found downstream of the discharge. The beneficial use protection extends to all aquatic life and not limited to pelagic organisms.

(7) USEPA expects to public the 2009 Ammonia Criteria Update which includes more stringent ammonia criteria for freshwater mussels compared with criteria for salmonids in early 2011. Freshwater mussels reside in the Upper Sacramento River above and likely below the SRWTP discharge.[238]

Regardless of its concerns with the 1999 Criteria, the Regional Board then calculated WQBELs using the 1999 Criteria and applied them at the end-of-pipe with no dilution.[239] In doing so, the Regional Board used its concerns with adequacy of the 1999 Criteria to arbitrarily deny the mixing zone, rather than using the information for selecting the appropriate numeric value to interpret the narrative objective.

d. Regional Board's Process Violated the Law and Reason

Contrary to the Draft Order's proposed findings, the Regional Board's approach to arriving at WQBELs for ammonia in the Permit are improper.

According to the Draft Order, the Regional Board allegedly established the ammonia WQBELs using Option B of the federal regulations.[240] In doing so, the Draft Order claims the Regional Board was not otherwise relying on an explicit state policy.[241] The Regional Board's actual record on this issue is unclear. The Permit states that the effluent limitations were based on the 1999 Criteria with no dilution credit.[242] However, such a statement could mean that the Regional Board invoked use of Option B *or* Option A from the federal regulations. Under Option B, effluent limitations are established on a "case-by-case basis, using [U.S.] EPA's water

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[238] Permit, p. F-56, footnote omitted.

[239] Permit, p. F-57.

[240] Draft Order, p. 12.

[241] Draft Order, p. 12.

[242] Permit, p. F-57.

quality criteria, published under section 304(a) of the CWA, *supplemented where necessary by other relevant information*."[243] Under Option A, effluent limitations are established:

> [U]sing a calculated numeric water quality criterion for the pollutant which the permitting authority demonstrates will attain and maintain applicable narrative water quality criteria and will fully protect the designated use. Such a criterion may be derived using a proposed State criterion, or an explicit State policy or regulation interpreting its narrative water quality criterion, *supplemented with other relevant information* which may include: EPA's Water Quality Standards Handbook, October 1983, risk assessment data, exposure data, information about the pollutant from the Food and Drug Administration, and current EPA criteria documents; or . . . .[244]

The explicit state policy relevant under Option A is contained in the Basin Plan, which states that the Regional Board shall consider all "direct evidence of beneficial use impacts, all material and relevant information submitted by the discharger and other interested parties, and relevant numerical criteria and guidelines developed and/or published by other agencies and organizations (e.g., . . . USEPA . . .). In considering such criteria, the Board evaluates whether the specific numerical criteria, which are available through these sources and through other information supplied to the Board, are relevant and appropriate to the situation at hand, and therefore, should be used in determining compliance with the narrative objective."[245] Considering the Basin Plan's reference to numerical criteria published by U.S. EPA, and the Regional Board's limited description with respect to which option it actually invoked, it is not feasible to discern which option the Regional Board relied on to calculate WQBELs. More importantly, such a distinction is irrelevant. Both Option A and Option B provide that the Regional Board may supplement the process with "other relevant information."[246]

Under the arguments proposed in the Draft Order, "other relevant information" in Option B is related directly to deciding whether to allow dilution credit in a WQBEL calculated from 304(a) criteria rather than adjusting the 304(a) criteria itself based on other relevant information.[247] The Draft Order's interpretation and rationalization of Option B in this manner is

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[243] Draft Order, p. 12 citing 40 C.F.R. § 122.44(d)(1)(vi)(B), emphasis added.

[244] 40 C.F.R. § 122.44(d)(1)(vi)(A), emphasis added.

[245] Basin Plan, p. IV-17.00.

[246] 40 C.F.R. § 122.44(d)(1)(vi)(A)-(B).

[247] Draft Order, pp. 12-13.

unsupportable and is inconsistent with proper application of the applicable regulation. Option B, and for that matter Options A and C, is to be "used as one of three options *to interpret state narrative water quality* criteria until the state adopts a numeric water quality criterion for the pollutant."[248] In other words, 304(a) criteria and "other relevant information" are to be considered when interpreting the narrative objective – not at the end of the process after the WQBEL is calculated. Likewise, "other relevant information" in Option A is also to be considered when interpreting the narrative objective.

Thus, the primary question before the State Board with respect to the ammonia WQBELs is, did the Regional Board properly interpret the narrative toxicity objective for both determining reasonable potential and calculating WQBELs. This consideration would include both the near-field and far-field impacts.

Relevant to these procedures, the Permit indicates that the applicable numeric water quality criteria for interpreting the narrative toxicity objective are the 1999 Criteria. However, at the same time, the Regional Board and the Draft Order both find that the 1999 Criteria are not sufficiently protective. For example, the Regional Board Response to Petitions reflects the Regional Board's conclusion that ambient ammonia concentrations equal to the 1999 Criteria are not adequate to fully protect the designated uses, not immediately downstream of the discharge, but at locations far downstream.[249] According to the Regional Board's Response, "other information" provides evidence that the narrative toxicity objective would still be violated if the 1999 Criteria are met just outside the small mixing zone requested by the District for meeting the 1999 Criteria.[250] The Regional Board states that Dr. Teh's preliminary study results are evidence of a violation of the narrative toxicity objective.[251] Likewise, the Regional Board contends that suppression of nitrate uptake in diatoms in spring in Suisun Bay is a violation of the narrative toxicity objective at a downstream location.[252] In both of these examples, the Regional Board has

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[248] 54 Fed. Reg. 23868 (June 2, 1989), emphasis added.

[249] Regional Board Response to Petitions, pp. 34-35, 37.

[250] See Regional Board Response to Petitions, pp. 34-35.

[251] Regional Board Response to Petitions, p. 47.

[252] Regional Board Response to Petitions, pp. 39-40.

identified specific numeric values (i.e., criteria), which it believes are applicable to far downstream receiving waters for determining if there is a violation of the narrative toxicity objective. However, by the Regional Board's own repeated admission, this information was not used to interpret the narrative toxicity objective, determine reasonable potential, and subsequently calculate WQBELs considering an approvable mixing zone for ammonia. Thus, the Regional Board effectively acknowledges that it did not follow the prescribed procedures in state and federal regulations and guidelines. Accordingly, the actual WQBELs for ammonia adopted in the Permit are arbitrary. They were not calculated in a manner consistent with the SIP, in that the narrative toxicity objective was not properly interpreted considering all available information the Regional Board considered to be relevant. Importantly, if WQBELs were calculated based on the interpretation of the narrative water quality objective in consideration of the "far downstream" concerns (with proper consideration of the recognized chronic dilution credit of 29:1),[253] they would not necessarily be the WQBELs that were adopted. Indeed, the District believes they would not be.

Similarly, the Regional Board's determination of impacts to diatoms or copepods "far downstream in the Delta," even if correct, does not support the WQBELs.[254] Here, for example, with regard to Delta phytoplankton, the Regional Board contends that phytoplankton inhibition is "evidence of a violation of the [Basin Plan] Narrative Toxicity Objective[.]"[255] It also contends that recent studies from which the Regional Board inferred impacts to copepods are evidence of "a violation of the Narrative Toxicity Objective[.]"[256] Accepting that as true, the Regional Board was required to develop WQBELs in a manner consistent with state and federal regulations by interpreting the narrative toxicity objective as applied downstream with all relevant information to establish downstream numeric criteria for effluent limitation derivation.

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[253] Permit, pp. F-38 to F-39.

[254] Regional Board Response to Petitions, p. 39.

[255] Regional Board Response to Petitions, p. 53.

[256] Regional Board Response to Petitions, p. 53.

With respect to the denial of the mixing zones for ammonia, the Regional Board has asserted that there can be no aquatic life impacts outside the mixing zone, and that since there are impacts that occur at locations many miles downstream, none of these waters are outside the "mixing zone."[257] In other words, and as discussed previously above, the Regional Board contends that the narrative toxicity objective is violated at locations far downstream (Suisun Bay). To reach this conclusion, the Regional Board must necessarily be interpreting the narrative toxicity objective with information provided by Dr. Teh with respect to copepods, and information pertaining to diatom inhibition. However, as shown above, the Regional Board failed to use of any of these available numeric values as "other relevant information" for interpreting the narrative toxicity objective and instead found in the Permit that the 1999 Criteria were the appropriate numeric criteria for interpreting the narrative toxicity objective and for calculating WQBELs.[258] Therein lies the problem. With current SRWTP effluent quality, the 1999 Criteria are met throughout the waterbody except within the acute and chronic mixing zones of 60 feet and 350 feet, respectively. Thus, when using the 1999 Criteria to interpret the narrative toxicity objective, the elements of the SIP for the allowance of mixing zones are met and effluent limitations far higher than the Permit imposed would be appropriate. Had the Regional Board properly interpreted the narrative toxicity objective with "other relevant information" to address its far downstream concerns, and then calculated WQBELs using the recognized chronic mixing zone of 350 feet with 29:1 dilution,[259] compliance with newly calculated WQBELs (whatever they may be) would ensure that the narrative toxicity objective is met throughout the waterbody except within the chronic mixing zone.

The Regional Board has thus admitted that it selected numeric water quality criteria for interpreting the narrative toxicity objective and calculated WQBELs that are not, in the Regional Board's judgment, protective of the aquatic life beneficial use. Otherwise, the 1999 Criteria, and the effluent limitations calculated therefrom, including with the consideration of dilution, should

[257] Regional Board Response to Petitions, p. 32.

[258] Permit, pp. F-54 to F-57.

[259] Permit, pp. F-36 to F-38.

be presumed to protect beneficial uses because the 1999 Criteria will be met outside of an approvable mixing zone.[260] Here, instead of interpreting narrative toxicity objectives in a manner as required by law, and granting a mixing zone in accordance with accepted procedures, the Regional Board confused its desire to require ammonia load reductions from the current levels with the denial of mixing zones for ammonia, thereby making such denial arbitrary. If the 1999 Criteria are not protective of the beneficial use, the Regional Board was obliged to use "other" information to interpret the narrative toxicity objective for deriving effluent limitations with consideration of an approvable mixing zone.[261]

The Regional Board's reluctance to use information to interpret the narrative toxicity objective raises other questions. The Draft Order takes special care to distinguish between using the other information (e.g., Teh studies) for denial of a mixing zone versus interpreting the narrative toxicity objective.[262] In fact, the Draft Order suggests that this information would not be adequate for purposes of interpreting the narrative toxicity objective. For example, in its discussion with respect to the Draft 2009 Criteria (discussed further below in section B.5), the Draft Order states that it would not be proper to use the Draft 2009 Criteria to interpret the narrative toxicity objective.[263]

Likewise, when discussing Teh's work (i.e., ammonia toxicity to copepods), the Draft Order finds that (under the circumstances) the Regional Board could use Teh's work as relevant evidence, "to deny an ammonia mixing zone."[264] However, the Draft Order makes no findings to suggest that the same information is adequate and relevant evidence for interpreting the narrative toxicity objective. If the "other relevant information" is not of sufficient scientific quality for interpreting the narrative toxicity objective, it cannot be sufficient for denying an otherwise approvable mixing zone. The level of scientific credibility with respect to the use of evidence

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[260] Permit, pp. F-38, J-1.

[261] 40 C.F.R. § 122.44(d)(1)(vi)(B).

[262] Draft Order, pp. 12-13.

[263] Draft Order, pp. 13-14.

[264] Draft Order, p. 16.

must be equally applicable in both steps of the process. Otherwise, the Draft Order is suggesting that the Regional Board's denial of a mixing zone lacks technical support and thus is contrary to the law and previous State Board precedential orders.[265]

4. Improper Use of the TSD

Besides relying, improperly, on "other relevant information" to deny a mixing zone rather than interpret the narrative objective, the Draft Order repeatedly turns to the TSD for support of the Regional Board's actions.[266] The use of the TSD in the Draft Order is improper for two reasons. First, the Regional Board used its discretion to follow the SIP and did not indicate that it was also relying on the TSD, except with respect to the sizing of mixing zones.[267] Absent indication in the Permit or findings that the TSD was used for other purposes, the Regional Board must comply with the SIP procedures and must be assumed to have been operating under the SIP.

Second and more importantly, the TSD "guidance" that the Draft Order relies upon is completely inapplicable. The Draft Order characterizes the TSD as follows: "The TSD provides guidance that, as in this case, where adverse effects have been observed far downstream, rather than confined to a mixing zone, mixing zones may be denied where such denial is used as a device to compensate for uncertainties in the protectiveness of water quality criteria."[268] A significant problem with the frequent reference to this guidance from the TSD is that, here, there *is no* uncertainty in the protectiveness of the narrative toxicity objective, the applicable water quality criterion.[269] Moreover, and more fundamentally, the principle stated in the Draft Order does not appear in the TSD. The language that does appear is the following, which only applies

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[265] See, e.g., State Board Order WQO 2004-0013, *supra*, p. 10.

[266] Draft Order, p. 13.

[267] Permit, pp. F-34 to F-40; see State Board Order No. 2004-0013, *supra*, p. 6 ("[The Regional Board] apparently used a method combining aspects of the TSD and SIP . . . for determining reasonable potential. In prior orders, we have held that a Regional Board may use the SIP implementation provisions as guidance for water quality-based toxics control. It appears that the Regional Board's methodology was appropriate, but it should clarify the methodology it used in findings.").

[268] Draft Order, p. 13.

[269] Even if "water quality criteria" in this context could be interpreted as referencing section 304(a) criteria rather than section 303 criteria, the statement would be inapplicable. The Regional Board determined that the 1999 Criteria are *not* sufficiently protective.

to the specific case of bioaccumulative pollutants. The actual language from the TSD is as follows:

> While fish tissue contamination tends to be a far-field problem affecting entire waterbodies rather than a narrow-scale problem confined to mixing zones, restricting or eliminating mixing zones for *bioaccumulative* pollutants may be appropriate under conditions such as the following:
>
> - Mixing zones should be restricted such that they do not encroach on areas often used for fish harvesting particularly of stationary species such as shellfish.
> - Mixing zones might be denied where such denial is used as a device to compensate for uncertainties in the protectiveness of the water quality criteria or uncertainties in the assimilative capacity of the waterbody.[270]

Ammonia is *not* a bioaccumulative pollutant. Accordingly, the TSD provision is not applicable here and fails to provide any support for the Regional Board's illogical actions to deny the mixing zone. Moreover, had the Regional Board followed the normal procedures for selecting a numeric value that it determined was protective to interpret the narrative toxicity objective, there would be no need to deny an approvable mixing zone claiming that there was uncertainty with respect to the water quality criteria.

5. Draft 2009 Criteria

The Permit also used the Draft 2009 Criteria as a reason to deny the mixing zone.[271] This in itself is problematic. The Draft Order attempts to "fix" the Regional Board's improper use of the unadopted Draft 2009 Criteria by claiming that, "the Central Valley Water Board used the scientific literature that is the basis for the Draft 2009 Criteria as 'other relevant information' to deny a mixing zone."[272] There are several problems with this claim. First, the record provides no evidence that the Regional Board did any such thing and findings in the Permit describe no such undertaking.[273] The Permit includes a discussion of the Draft 2009 Criteria.[274] Nowhere in this discussion does the Permit state the Regional Board reviewed or considered the "scientific

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[270] TSD, p. 34, emphasis added.

[271] Permit, p. F-56.

[272] Draft Order, p. 14.

[273] See, *Topanga, supra,* 11 Cal.3d at pp. 514-515.

[274] See Permit, pp. J-3 to J-4.

literature" that was the basis for the Draft 2009 Criteria. Rather, the Permit discusses the Draft 2009 Criteria, and studies that compared monitoring data to the calculated Draft 2009 Criteria.[275] In fact, the Permit finding merely states that the Draft 2009 Criteria themselves were a reason for denial – not the literature behind the draft criteria.[276]

Second, the Draft Order's statement conflicts directly with argument that the Regional Board has made. The Regional Board's reference to using the "scientific literature" appears for the first time in the Regional Board Response to Petitions. Here, the Regional Board states, "[t]herefore, the Central Valley Water Board used the scientific literature that is the basis of the new USEPA ammonia criteria *to interpret the Basin Plan's Narrative Toxicity Objective*," which in itself contradicts the Draft Order's finding that the Draft 2009 Criteria were used to deny a mixing zone.[277]

Third, the Draft Order cites four references (out of 170) from the Draft 2009 Criteria.[278] The Draft Order claims that the Regional Board relied on these four studies to deny the mixing zone – not the Draft 2009 Criteria.[279] However, after careful review of the record, the District cannot locate these four studies in the administrative record.[280] To the District's knowledge, the documents in question are not in the Regional Board's possession, or at the very least, their exact location has not been identified in the administrative record of the Permit. Moreover, nowhere does the Permit itself or the Regional Board refer to the Regional Board's use of these studies. Accordingly, the District objects to the Draft Order's reference to these four studies.[281]

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[275] Permit, pp. J-3 to J-4.

[276] Permit, p. F-56.

[277] Regional Board Response to Petitions, p. 61.

[278] Draft Order, p. 14, fn. 57; Draft 2009 Criteria, pp. 48-59.

[279] Draft Order, p. 14.

[280] The record includes the U.S. EPA Draft 2009 Update, but the referenced studies are not specifically included. (See SRCSD_OTHER_138 and 290.)

[281] The District also notes that the Draft Order refers to no actual record evidence that Unionid mussels are present at any given relevant location. (Draft Order, pp. 14-15.) Nor does it acknowledge the finding in the Permit that it "is not known" whether the mussel is in the lower Sacramento River near the SRWTP. (Permit, p. J-3.)

Finally, the Draft Order once again muddles the application of "other relevant information."[282] As is discussed previously, "other relevant information" in this context is to be used when interpreting the narrative toxicity objective for calculating WQBELs not when trying to justify denial of a mixing zone. In its own words, the Draft Order indicates that it would be improper to use the Draft 2009 Criteria "to interpret its narrative toxicity objective."[283] Rather than acknowledging that the Regional Board improperly used the Draft 2009 Criteria, the Draft Order manufactures a new theory that is not present in the Permit, or the Permit record. In fact, as noted above, this theory and the Draft Order directly contradict argument made by the Regional Board. If the studies underlying the Draft 2009 Criteria were appropriate for any purpose, it was for use in interpreting the narrative objective, and a mixing zone could have been allowed for compliance with the relevant numeric threshold.

6. Ammonia Toxicity to Copepods

As with the other issues, the Draft Order attempts to justify the Regional Board's actions by claiming that the Regional Board properly used ammonia toxicity to copepods as "other relevant information" in its denial of the mixing zone.[284] As explained above, "other relevant information" such as this needs to be considered when the Regional Board is interpreting the narrative toxicity objective. In particular, when the Regional Board evaluated the 1999 Criteria to determine if they were appropriate for interpreting the narrative toxicity objective, the Regional Board needed to consider ammonia toxicity to copepods to the extent that the Regional Board found the evidence appropriate. The 1985 Guidelines in fact advise that national numerical criteria should be modified to reflect site-specific conditions. "As an intermediate step in the development of standards, it might be desirable to derive site-specific criteria by modification of

[282] 40 C.F.R. § 122.44(d)(1)(vi)(B).

[283] Draft Order, pp. 13-14.

[284] Draft Order, p. 16.

national criteria to reflect such local conditions as water quality, temperature, or ecologically important species."[285]

Rather than using the information at the appropriate step, the Permit and Draft Order find the information was "relevant evidence" in a decision to deny a mixing zone.[286] The Draft Order itself suggests that Teh's study results might not be appropriate evidence in all circumstances. After acknowledging that Teh's studies were not peer reviewed and that standard toxicity testing procedures were not completely followed,[287] the Draft Order concludes that the use of this evidence was still sufficient *under these circumstances* (which was to support denial of the mixing zone).[288] In light of the questions and concerns with respect to Teh's studies, it is appropriate that the Regional Board and the Draft Order decline to use the information to interpret the narrative toxicity objective. It is not clear, however, how this inadequate information somehow becomes adequate for use in the denial of an otherwise approvable mixing zone. This inconsistent application is neither reasonable nor supported by the record.

7. Ammonia Toxicity Impacts on Critical Habitat

In continuation of its arguments to validate various conclusions based on disputed science, the Draft Order proposes to find that the Regional Board's determination with respect to affects on designated critical habitat were supported by the record.[289] In doing so, the Draft Order proposes to take official notice of the United State Fish and Wildlife Service's (USFWS) recent federal register notice with respect to adding Longfin smelt as a candidate species under the

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[285] 1985 Guidelines, p. 3; see also TSD, p. 34 ("The water quality standards regulation allows States to develop numerical criteria or modify EPA's recommended criteria to account for site-specific or other scientifically defensible factors.").

[286] Draft Order, p. 16; see Permit, p. F-56.

[287] The Draft Order ignores key evidence with respect to Teh's studies presented by the District: (1) Teh's study results were not just not peer reviewed, but they were preliminary and not published at the time that they were used by the Regional Board (Petition, pp. 73-81); (2) Teh's study results did not use environmentally representative conditions (Petition, pp. 74-76); (3) There are unexplained irregularities in the test results (Petition, p. 78); (4) Tests were conducted with novel test organisms, for which there are no established protocols and no comparable test results from other laboratories (Petition, p. 78; District's October 2010 Comments and Evidence Letter, p. 38.)

[288] Draft Order, p. 16.

[289] Draft Order, pp. 16-17.

federal Endangered Species Act (USFWS 2012 Notice).[290] The District objects to the reference of the USFWS 2012 Notice on several grounds. First, the State Board in its Interlocutory Ruling on Outstanding Motions clearly indicated its intent to "base its review on the documents and information that was before the Central Valley Water Board at the time the Permit was adopted."[291] Having made such a declaration, it is improper for the State Board to now expand this administrative record by taking official notice of new information.

Second, it is impermissible to take official notice of facts that are in dispute.[292] In addition, the official notice provision of State Board's regulations provide that official notice may be taken of "any generally accepted technical or scientific matter within the Board's field of expertise, provided parties appearing at the hearing shall be informed of the matters to be noticed."[293] Assertions in the USFWS 2012 Notice are not a "generally accepted" technical or scientific matter, nor is it in the State Board's field of expertise. For example, scientific information described by the USFWS 2012 Notice is disputed in this very matter.[294] Further, there is significant evidence in this record, let alone outside of this administrative record, that questions the statements contained in the USFWS 2012 Notice.

Third, the State Board's official notice of new evidence is an unlawful violation of the District's procedural rights. The Draft Order's proposed finding with respect to this issue violates applicable provisions of the Government Code and the State Board's regulations, and additionally deprives the District of constitutionally guaranteed procedural due process.[295] For example, the State Board's official notice regulation provides that, "[p]arties shall be given a reasonable opportunity on request to refute officially noticed technical or scientific matters in a manner to be determined by the Board or presiding officer."[296] However, the notice with the Draft Order states

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[290] Draft Order, pp. 16-17, fn. 63.

[291] State Board's November 22, 2011 Interlocutory Ruling on Outstanding Motions (Interlocutory Ruling), p. 3.

[292] See Draft Order, p. 17, fn. 63 ("The U.S. Fish and Wildlife Service found that the available scientific information warranted listing . . . . The listing is only cumulative of other evidence . . . .").

[293] Cal. Code Regs., tit. 23, § 648.2.

[294] See, e.g., Petition, pp. 68-73.

[295] See, e.g., Gov. Code, §§ 11425.10, 11513; Cal. Code Regs., tit. 23, §§ 648.2, 648.5.

[296] Cal. Code Regs., tit. 23, § 648.2.

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that, "[a]ll comments shall be based solely upon evidence contained in the record or upon legal argument."[297] Supplemental evidence will only be considered in accordance with section 2050.6 of title 23 of the California Code of Regulations. This is not consistent with the District's automatic right to refute any evidence or fact proposed for official notice.[298]

For these reasons and others discussed in the Petition, the Draft Order's findings with respect to ammonia toxicity and its alleged impacts to biologically sensitive or critical habitats are improper.

8. Final Ammonia Effluent Limitation Calculation

The Draft Order proposes to remand the Permit to the Regional Board for the purpose of reviewing and revising the calculation of the 1999 Criteria.[299] Although this direction alone would not cure the fundamental technical and regulatory defects that the District has identified, the District agrees that the Permit calculations are incorrect. However, they are incorrect for different reasons. Specifically, as the District explained in its October 2010 Comments and Evidence Letter, to calculate the 30-day Criterion Continuous Concentration (CCC), the Regional Board should have used paired effluent pH and temperature data rather than receiving water data because no dilution has been granted.[300] Further, two 30-day CCCs should have been calculated, one for each season.[301] By calculating two 30-day CCCs with paired effluent temperature and pH data, and by deriving the effluent limitations from the 1/10th percentiles of the seasonal CCC datasets (assuming no mixing zone for compliance with the 1999 Criteria), the resulting effluents would be an average monthly effluent limitation (AMEL) of 3.0 mg/L and a maximum daily

[297] Letter dated May 14, 2012, to Paul Simmons, et al., from Michael A.M. Lauffer, Chief Counsel, State Board, re: Board Workshop Notification (State Board May 14, 2012 Letter), p. 2.

[298] State Board May 14, 2012 Letter, p. 2. Section 2050.6 of title 23 of the California Code of Regulations provides the State Board with discretion to approve a request – acceptance of such information is not guaranteed. (Cal. Code Regs., tit. 23, § 2050.6(a)(3).)

[299] Draft Order, pp. 18, 24.

[300] District's October 2010 Comments and Evidence Letter, p. 94.

[301] District's October 2010 Comments and Evidence Letter, p. 94.

effluent limitation (MDEL) of 3.9 mg/L for March 1-October 31, and an AMEL of 3.6 mg/L and MDEL of 4.7 mg/L for November 1-February 29.[302]

There are in fact other circumstances that the Permit and Draft Order ignore with respect to calculating appropriate WQBELs for ammonia. The District has repeatedly indicated that it expects to reduce ammonia concentrations from current levels in order to ensure future, long-term compliance with dissolved oxygen water quality objectives.[303] The District has suggested that it would be appropriate to reduce ammonia concentrations by about one-half.[304] But nowhere does the Regional Board or the Draft Order determine if ammonia concentrations discharged at half the current level (i.e., about 15 mg/L assuming the interim AMEL) would affect aquatic life in the "far-field" downstream or outside an approvable mixing zone in the near field. The key principle is that the final WQBELs for ammonia need not be those specified in the Permit – but may be set at a level between the Permit levels and current discharge, and still be fully protective of downstream beneficial uses.

C. Nitrate Effluent Limitations

The District agrees with the Draft Order that the Permit's effluent limitations for nitrate are improper.[305] The District is very concerned, however, that the Draft Order prejudices the analysis that must occur in the future, related to a complex subject. Any language that implies that the effects of the SRWTP or other nutrient discharge are adequately understood and that nutrient limitations will be needed in the future should be eliminated from the Draft Order. Such language is unnecessary and unfairly prejudices decisions in the future based on incomplete information. The Draft Order describes an "NNE framework."[306] While there are no citations to

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[302] District's October 2010 Comments and Evidence Letter, p. 94.

[303] See, e.g., District's October 2010 Comments and Evidence Letter, pp. 41, 43; see also Hearing Transcript, p. 226:8-15 (testimony of Stan Dean).

[304] *Ibid.*

[305] The Draft Order states that the reasons for denying a nitrate mixing zone "were related to aquatic and ecological impacts." (Draft Order, p. 20.) In fact, the Permit does not actually identify any adverse impacts of nitrate. It merely concludes that regulatory standards under the SIP for granting a mixing zone are not met. (See Permit, pp. F-44 to F-45; Petition, pp. 127-128.)

[306] Draft Order, p. 22.

any record documents, the District agrees that the NNE framework required to begin to set numeric objectives in San Francisco Bay and the Delta is "currently under development."[307] No proposed numeric objectives for total N and total P in the Bay or Delta are publicly available or proposed.

In this regard, if the Draft Order is going to describe an NNE framework and what may be expected, it needs to also recognize other information that does not support that some expected outcome is also "likely." In a letter to Municipal Wastewater Dischargers in the San Francisco Bay area that is posted on the CalEPA website, the San Francisco Bay Regional Water Board states, "the NNE approach will likely require models that link ecological response indicators to nutrient loads and other management controls. This effort must be supported by accurate loading estimates from a variety of sources, including wastewater . . . . There have been published studies that have developed some loading estimates; however, these studies are outdated, inadequate, or limited geographically. Thus, nutrient loads to the San Francisco Bay estuary from external sources are still poorly understood . . . ."[308] The letter requires the collection of nutrient data to improve the understanding of nutrient loads and to otherwise support the NNE effort in San Francisco Bay.[309] A final report is required from wastewater dischargers in July 2014.[310] That information will then be used in models to support the eventual development of nutrient water quality objectives.[311]

Further, the findings made in the Draft Order relating to the impact of the SRWTP discharge in Suisun Bay are not objective, and are inconsistent with those being made as part of

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[307] Draft Order, p. 22.

[308] Letter dated March 2, 2012, to Municipal Wastewater Dischargers from Bruce H. Wolfe, Executive Officer, California Regional Water Quality Control Board, San Francisco Bay Region, subject Water Code Section 13267 Technical Report Order Requiring Submittal of Information on Nutrients in Wastewater Discharges, <http://www.waterboards.ca.gov/sanfranciscobay/water_issues/programs/planningtmdls/amendments/estuarineNNE/Nutrients%2013267%20Order%20-%203-12.pdf> (as of June 12, 2012) (March 2, 2012 Letter), at p. 2. In a separate filing, the District requests that this March 2, 2012 Letter be added to the record as supplemental evidence.

[309] March 2, 2012 Letter, pp. 1-2, 6-7.

[310] March 2, 2012 Letter, p. 7

[311] March 2, 2012 Letter, p. 2.

the NNE process being performed in the same area by the State and Regional Water Boards.[312] This inconsistency should be resolved by eliminating these unnecessary statements from the Draft Order.

D. Additional Issues

1. Additional Issues Raised in Petition

The Draft Order does not specifically address certain other issues that were raised in the Petition. The District acknowledges that its Petition was deemed denied due to passage of time[313] and that the State Board is not required in an "own motion" review, to address issues identified in the Petition. However, if the State Board adopts an order, the District encourages that it address other issues, which the District submits are straightforward.

a. Illegal NMDA Monitoring Requirement

One such issue is the Permit's requirement to monitor for N-nitrosodimethylamine (NDMA) in the SRWTP effluent using U.S. EPA analytical test method 521 (U.S. EPA Method 521).[314] As explained in the District's Petition and below, this requirement runs afoul of federal regulations and the SIP. In brief, federal regulations applicable here require that NDMA effluent monitoring be under a test method approach specified in 40 Code of Federal Regulations Part 136.[315] For NDMA, 40 Code of Federal Regulations Part 136 specifies multiple approved methods, but U.S. EPA Method 521 is *not* one of them. Moreover, the Regional Board has not asserted that the use of U.S. EPA Method 521 to monitor the SRWTP effluent for compliance purposes is legally permissible.[316]

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[312] Draft Order, p. 22.

[313] Cal. Code Regs., tit. 23, § 2050.5(b).

[314] U.S. EPA Method 521, Determination of Nitrosamines in Drinking Water by Solid Phase Extraction and Capillary Column Gas Chromatography with Large Volume Injection and Chemical Ionization Tandem Mass Spectrometry (MS/MS) (September 2004), U.S. EPA Document # EPA/600/R-05/054, AR at SRCSD_OTHER_049 (U.S. EPA Method 521); see Permit, p. E-6; Petition, pp. 8, 175-176.

[315] See 40 C.F.R. §§ 122.41(j)(4), 122.44(i)(l)(iv), 136.1(a)(1); Petition, pp. 175-176.

[316] In its response to the Petition, the Regional Board made argument as to reasons it required U.S. EPA Method 521, but it provided no response to the simple point that it was unlawful to do so. (See Regional Board Response to Petitions, pp. 83-84.) It is true that the District in the past used Method 521 to determine if NDMA was detected in effluent or influent. But the Permit requirement accompanies a new effluent limitation and is thus relevant to determination of compliance with effluent limitations. Given the exceptionally low effluent limitations and limitations of any monitoring technologies, the method required by the Permit is objectionable.

In addition, requiring the District to use U.S. EPA Method 521 for purposes of determining compliance with the Permit's effluent limitations for NDMA violates the SIP. The SIP applies to priority pollutants, such as NDMA, and requires that the Regional Board determine the analytical methods to be used to evaluate compliance with permitted effluent limitations.[317] As explained in the Petition, the SIP does not allow the use of U.S. Method 521, although it does furnish a potential mechanism under which the District could agree to monitoring methods that would not otherwise be allowed.[318]

U.S. EPA did not approve the use of U.S. EPA Method 521 as an alternate test procedure for the SRWTP discharge, nor has the District agreed to use U.S. EPA Method 521. For these reasons, the Permit requirement to use U.S. EPA Method 521 violates the SIP.

Because the requirement to monitor for NDMA in the SRWTP effluent violates federal regulations and the SIP, the State Board should vacate the monitoring requirement and order that monitoring for NDMA be conducted using an appropriate test method.

b. Other Stringent Limitations Unrelated to Protection of Beneficial Uses

In its Petition, the District also identified other Permit limitations that may create problems for the District, yet are unnecessary. In particular, the record evidence identified that approval of mixing zones for certain other constituents would fully meet the requirements of the SIP and other applicable considerations, and that the Permit's toxicity trigger for whole effluent toxicity of 8 TUc is essentially arbitrary.[319] These stringent provisions create obligations that in turn create risks of liability or other consequences for the District, but are not necessary to protect beneficial uses. The approach taken in the Permit may have adverse implications for other regulated agencies as well, and the District submits that the issues that were presented were appropriate for consideration.

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[317] SIP, p. 22.

[318] Petition, p. 176; see SIP, pp. 22-24.

[319] Petition, pp. 165-175. As filed, the Petition requested a determination that effluent limitations for chlorpyrifos and diazinon were improper. The District subsequently withdrew that portion of its Petition.

2. Additional Issues Pertaining to Supplemental Evidence and Other Materials

On page 3, the Draft Order refers to requests that have been made to file supplemental pleadings and augment the administrative record. The District provides the following related to that discussion of the Draft Order.

First, the Draft Order incorrectly implies that, during the period for review of the Petition, the District submitted numerous requests to supplement the record.[320] With its Petition, the District requested that the State Board consider one document that the Regional Board did not enter into its administrative record.[321] This document, which was delivered to the Regional Board at the Regional Board hearing on December 9, 2010, but was not admitted into the record, was simply a listing of District comments on the tentative permit to which the Regional Board Staff Response to Comments had not responded.[322] The District's Petition also stated that if the State Board did not take official notice of, or consider the document, the District incorporated the document by reference into the District's argument.[323] By contrast, the parties referred to as the "Water Agencies" submitted over 30 extra-record documents with their response to the Petition. In summary, the District did not file numerous requests to supplement the administrative record. In addition, the District is concerned that the Draft Order may be suggesting that these matters affected the timing of action. The District submits that resolution of evidentiary or record issues does not, or need not have, consumed significant work time or resources for the State Board.

The District also requested that the State Board consider a reply memorandum in accordance with section 2050.5(a) of title 23 of the California Code of Regulations.[324] While the District believes the proposed reply was appropriate and should have been considered, the District also notes here that the memorandum submitted was not limited to the proposed reply, and also

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[320] "During our review of the petitions and the administrative record, the District and interested persons submitted numerous requests to file supplemental pleadings and augment the administrative record." (Draft Order, p. 3.)

[321] Petition, pp. 16-17.

[322] See Petition, pp. 16-17.

[323] Petition, p. 18.

[324] Sacramento Regional County Sanitation District's Application for Leave to File Reply Memorandum; Reply Memorandum and Further Objections (June 9, 2011) (Reply Application).

included certain evidentiary objections related to responses to the District's Petition that had been filed by other parties.[325] The Draft Order does not appear to rely specifically on the material to which the District objected in that document. However, the District continues to assert those objections.

3. Other Issues Related to Draft Order

As described in specific sections of the preceding material, the Draft Order includes reference to various materials that are not within the Regional Board's record. This is inconsistent with the State Board's Interlocutory Ruling that its review would be based on the documents and information before the Regional Board.[326] It also impairs the legal rights of the District.[327]

In the meantime, as to the Draft Order, and in some cases as preliminary response to material now cited in the Draft Order, the District is separately submitting other evidence that should be considered in accordance with applicable procedures and regulations.

E. The State Board Should Recognize the District's Real-World Problems Under Compliance Schedule Mandates While the District's Ultimate Obligations Are Being Determined

The Permit was issued nearly one and one-half years ago. It contains very costly requirements, and deadlines for compliance that are very aggressive. The District has been seeking review of the requirements of the Permit since January of 2011. Under the Draft Order, the Permit's ammonia effluent limitations would be remanded for revised calculation, and effluent limitations for nitrate would be remanded. These developments affect planning assumptions and all the while the District continues to seek review of fundamental issues in the Permit. The State Board should recognize and be supportive of the District's legitimate need for the compliance schedules to be abeyed until there is resolution of the pending issues.

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[325] See Reply Application, p. 3 ("[T]he Reply Memorandum also articulates certain objections to the responses. Under any circumstances, the District is entitled to have its objections considered.") The relevant objections here were primarily to assertions on facts or technical issues not based on evidence in the record. (See, e.g., Reply Application, p. 8, fn. 15; pp. 11-14.)

[326] Interlocutory Ruling, p. 3.

[327] See, e.g., Gov. Code, § 11425.10(a)(1); Cal. Code Regs., tit. 23, § 2050 et seq.

III. CONCLUSION

The State Board should not adopt the Draft Order.

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BEFORE THE

CALIFORNIA STATE WATER RESOURCES CONTROL BOARD

In the Matter of Own Motion Review of Waste Discharge Requirements Order No. R5-2010-0114 [NPDES No. CA0077682] for Sacramento Regional Wastewater Treatment Plant, Issued by the California Regional Water Quality Control Board, Central Valley Region.	SWRCB/OCC File Nos. A-2144(a) and A-2144(b) (consolidated) **SACRAMENTO REGIONAL COUNTY SANITATION DISTRICT'S REQUEST FOR ADMISSION OF NEW EVIDENCE AND FOR OFFICIAL NOTICE**

INTRODUCTION

Pursuant to section 2050.6 of title 23 of the California Code of Regulations, as well as section 648.2 of title 23 of the California Code of Regulations, the Sacramento Regional County Sanitation District (SRCSD or District) requests that the State Water Resources Control Board (State Board) admit into the record for the above-captioned matter and consider the following documents:

1. *Wastewater Disinfection for Health Protection*, Sanitary Engineering Branch, California Department of Health Services (Feb. 1987); and Memorandum to Office of Drinking Water Management Staff, from Office of Drinking Water, Peter A. Rogers, Chief (Aug. 18, 1992, retyped in November 2000) re: Uniform Guidelines for Disinfection of Wastewater, including attached Uniform Guidelines for the Disinfection of Wastewater;

2. Central Contra Costa Sanitary District's Response to Comments on the Tentative Order No. R2-2011-XXXX, NPDES Permit CA0037648 (Dec. 28, 2011); State Water Contractors and San Luis & Delta-Mendota Water Authority's Comments on Tentative Order No. R2-2011-XXXX (NPDES No. CA0037648) for the Central Contra Costa Sanitary District Wastewater Treatment Plant (Oct. 31, 2011); Bay Area Clean Water Agencies' Comments on the Tentative Order for the Central Costa Sanitary District, No. R2-2011-XXXX, NPDES Permit No. CA0037648 (Dec. 8, 2011); California Regional Water Quality Control Board, San Francisco Bay Region, Response to Written Comments on October 2011 Tentative Order for Central Contra Costa Sanitary District Wastewater Treatment Plant; and California Regional Water Quality Control Board, San Francisco Bay Region, Order No. R2-2012-0016, NPDES No. CA0037648;

3. *Numeric Nutrient Endpoint Development for San Francisco Bay Estuary: Literature Review and Data Gaps Analysis*, L. McKee, M. Sutula, A. Gilbreath, J. Beagle, D. Gluchowski, and J. Hunt, Southern California Coastal Water Research Project (Technical Report 644 – June 2011);

4. Letter to Mike Chotkowski, Department of Interior, United States Fish and Wildlife Service, from Sacramento Regional County Sanitation District, Comments on Endangered and Threatened Wildlife and Plants; 12-month Finding on a Petition to List the San Francisco Bay-Delta Population of the Longfin Smelt as Endangered or Threatened, Docket No. FWS-R8-ES-2008-0045; and

5. Letter to Municipal Wastewater Dischargers from Bruce H. Wolfe, Executive Officer, California Regional Water Quality Control Board, San Francisco Bay Region, subject Water Code Section 13267 Technical Report Order Requiring Submittal of Information on Nutrients in Wastewater Discharges (March 2, 2012).

The District makes this request because admission of the aforementioned documents is necessary and appropriate in light of and to respond to new evidence and/or Permit justifications or statements introduced in the State Board's proposed order on its review of Waste Discharge Requirements Order No. R5-2010-0114 [NPDES No. CA0077682] for Sacramento Regional County Sanitation District, Sacramento Regional Wastewater Treatment Plant, Sacramento County) (Draft Order). This request is consistent with the State Board's May 14, 2012 Board Workshop Notification that provides: "Supplemental Evidence will not be permitted except under limited circumstances described in California Code of Regulations, title 23, section 2050.6."[1] This request includes material subject to official notice under section 648.2 of title 23 of the California Code of Regulations.

BACKGROUND

For the convenience of the State Board, the District here furnishes certain context. The District filed a Petition for Review of Order No. R5-2010-0114 (Permit) in January of 2011 (Petition). During part of 2011, there were outstanding objections concerning extra-record evidence submitted and relied upon by other parties, particularly the entities that have been referred to as the "Water Agencies." On August 19, 2011, the State Board's legal counsel issued a letter to the Petitioners and interested parties regarding pending responses and time limits.[2] Among other things, the August 2011 letter states:

> Other than comments on the draft order scheduled for the September 20, 2011 State Water Board Meeting and any comment period associated with a draft disposition of this matter, no other submissions will be accepted by the State Water Board.[3]

On September 19, 2011, the State Board determined that it would review the Permit on its own motion.[4] On November 22, 2011, the State Board's legal counsel issued an Interlocutory

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[1] State Board May 14, 2012 Letter, p. 2.

[2] Letter to Paul Simmons, Bill Jennings, Theresa Dunham and Cassie Aw-Yang, Petitioners of Sacramento Regional County Sanitation District and California Sportfishing Protection Alliance, from James Herink, Staff Counsel, State Water Resources Control Board, re Pending Responses and Time Limits (Aug. 19, 2011) (August 2011 Letter).

[3] August 2011 Letter, p. 2.

[4] State Board Order WQ 2011-0013.

Ruling on Outstanding Motions (Interlocutory Ruling). In its Interlocutory Ruling, the State Board stated that, "the State Water Board intends to base its review on the documents and information that was before the Central Valley Water Board at the time the Permit was adopted."[5]

However, contrary to the Interlocutory Ruling, the Draft Order contains references to documents and information that have been considered in the development of the Draft Order that were not before the Central Valley Regional Water Quality Control Board (Central Valley Water Board). For example, the Draft Order cites to four studies that are referenced in *U.S. EPA's Draft 2009 Update Aquatic Life Ambient Water Quality Criteria for Ammonia – Freshwater* (Draft 2009 Criteria) as being appropriate scientific literature for denial of a mixing zone by the Central Valley Water Board. But the referenced documents have not been located in the Central Valley Water Board's administrative record.[6] The Draft Order also proposes to take official notice of a document issued "after briefing was complete" that contains discussion of ecological issues related to the subjects in dispute.[7] In another instance, the Draft Order relies on and references specific guidelines that were not referenced or relied upon in the Permit process and have been superseded by more recent information.[8] The specific circumstances that provide the need for the Supplemental Evidence identified in this request are explained below for each individual document.

The District notes that there are other extra-record documents that came into existence after adoption of the Permit that provide evidence that refutes certain scientific or technical conclusions advocated or once advocated by interested parties who argue for stringent regulation of the District. However, the Draft Order does not appear to endorse such conclusions. Accordingly, the District has not proposed here that any and all recent evidence supporting its views be admitted.

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[5] Interlocutory Ruling, p. 3.

[6] See Sacramento Regional County Sanitation District's Response to Draft Order on Own Motion Review (June 15, 2012) (District's Response to Draft Order), p. 59.

[7] Draft Order, p. 17, fn. 63.

[8] See, e.g., District's Response to Draft Order, p. 11, fn. 38.

ARGUMENT

Pertinent to this matter, section 2050.6(a) requires any person requesting the State Board to consider extra-record evidence to provide a detailed statement of the nature of the evidence and facts to be proved and detailed explanation of why the evidence could not previously have been submitted.[9] Justifications for admission of Supplemental Evidence are provided for each document here. Also, section 648.2 provides for official notice of any matter of which state courts may take judicial notice.

1. *Wastewater Disinfection for Health Protection*, Sanitary Engineering Branch, California Department of Health Services (Feb. 1987) (1987 Disinfection Guidelines); and Memorandum to Office of Drinking Water Management Staff, from Office of Drinking Water, Peter A. Rogers, Chief (Aug. 18, 1992, retyped in November 2000) re: Uniform Guidelines for Disinfection of Wastewater, including attached Uniform Guidelines for the Disinfection of Wastewater (1992 Memorandum)

The District requests that the 1987 Disinfection Guidelines and 1992 Memorandum be considered for the following reasons.

a. Context

The documents identified relate to the statement in the Draft Order that guidelines of the California Department of Public Health (DPH) from 1980 "require a median MPN of 2.2 when a

[9] Section 2050.6(a) provides in full:

> If any person requests that the state board consider evidence not previously provided to the regional board, that person shall provide a statement that additional evidence is available that was not presented to the regional board or that evidence was improperly excluded by the regional board. Any request by a regional board to present additional evidence shall comply with (a)(1) through (3).
>
> (1) The request to present additional evidence and all supporting arguments shall be provided at the time the petition is filed, or as soon as the evidence becomes available thereafter.
>
> (2) The request to present additional evidence shall include a detailed statement of the nature of the evidence and of the facts to be proved. If the evidence was not presented to the regional board, the person requesting consideration of the evidence shall provide a detailed explanation of the reasons why the evidence could not previously have been submitted. If the person presenting the evidence contends that the evidence was improperly excluded, the request shall include a specific statement of the manner in which the evidence was improperly excluded.
>
> (3) If the state board, in its discretion, approves a request to present additional evidence, the proponent must submit the evidence in writing and must also provide it to the petitioner, the discharger (if not the petitioner) and the regional board. The state board may prescribe a time limit for submission of the additional evidence.

stream's low flow provides dilution of less than 100 to 1 to protect MUN use . . . ."[10] As stated in relevant part in the District's Response to Draft Order:

> Here, the Draft Order inexplicably refers to a 1980 guidelines document that has been superseded and irrelevant for 25 years. (Draft Order, p. 5 and fn. 15.) Indeed, the State Board has even issued an order describing the document as having been "rescinded." (*Order Amending North Coast Regional Board Cease & Desist Order No. 85-35 for the City of Santa Rosa Subregional Wastewater Treatment, Reuse & Disposal Facilities,* State Board Order No. 2000-04 (March 15, 2000), ¶ 10.) In fact, in *1987*, the Department of Health Services (i.e., DPH) issued disinfection guidelines. (*Wastewater Disinfection for Health Protection*, Sanitary Engineering Branch, California Department of Health Services (Feb. 1987), Appendix D.) Under these guidelines, the recommendation for discharge to water used for domestic use was *23 MPN* where the average dilution is greater than 20:1. (*Id.*) In 1992, the DPH Office of Drinking Water notified its staff and the regional water quality control boards that certain provisions of the 1987 guidelines, including the 20:1 dilution ratio related to wastewater discharge where there is domestic use, were "no longer applicable" and the need "no longer exists." (Memorandum to Office of Drinking Water Management Staff, from Office of Drinking Water, Peter A. Rogers, Chief (Aug. 18, 1992) Re: Uniform Guidelines for Disinfection of Wastewater.) It further stated that the current guidelines "should only be referred to for recreational situations." (*Ibid.*) In summary, the source referred to in the Draft Order was reviewed, evaluated, and updated twice since 1980 . . . .[11]

The documents provided here support the above statements. Further, apart from section 2050.6, the documents and their content are properly subject to official notice under section 648.2 of title 23 of the California Code of Regulations.

b. Evidence Was Not Presented to the Central Valley Water Board But Is Being Presented as Soon as It Became Evident That Supplemental Evidence Was Necessary

The documents identified were not presented during the Central Valley Water Board's proceedings because the issue or statement newly raised in the Draft Order was not an issue before the Central Valley Water Board. During the Permit proceeding, neither the Regional Board nor DPH nor any other party suggested that the superseded guidelines were relevant.[12] Because the 1980 Guidelines as referenced in the Draft Order were superseded and not an issue before the Central Valley Water Board, it was not necessary to submit the 1987 Disinfection Guidelines and 1992 Memorandum to the Central Valley Water Board during its proceedings.

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[10] Draft Order, p. 5 and fn. 15.

[11] District's Response to Draft Order, p. 11, fn. 38.

[12] See also District's Response to Draft Order, pp. 10-15 (discussing and identifying DPH dilution-based recommendations).

c. Nature of Evidence and Facts To Be Proved

The nature of the evidence is as described in part "a" above. The documents establish the facts in the quotation in part a. The content of the documents is as provided in the documents.

2. Central Contra Costa Sanitary District's Response to Comments on the Tentative Order No. R2-2011-XXXX, NPDES Permit CA0037648 (Dec. 28, 2011) (Contra Costa Comments); State Water Contractors and San Luis & Delta-Mendota Water Authority's (Public Water Agencies) Comments on Tentative Order No. R2-2011-XXXX (NPDES No. CA0037648) for the Central Contra Costa Sanitary District Wastewater Treatment Plant (Oct. 31, 2011) (Public Water Agencies Comments); Bay Area Clean Water Agencies' (BACWA) Comments on the Tentative Order for the Central Contra Costa Sanitary District, No. R2-2011-XXXX, NPDES Permit No. CA0037648 (Dec. 8, 2011) (BACWA Comments); California Regional Water Quality Control Board, San Francisco Bay Region, Response to Written Comments on October 2011 Tentative Order for Central Contra Costa Sanitary District Wastewater Treatment Plant (San Francisco Water Board's Response to Written Comments); and California Regional Water Quality Control Board, San Francisco Bay Region, Order No. R2-2012-0016, NPDES No. CA0037648 (Order No. R2-2012-0016)

The District requests that the Contra Costa Comments, BACWA Comments, Public Water Agencies Comments, San Francisco Water Board's Response to Written Comments, and Order No. R2-2012-0016 all be received as Supplemental Evidence for the following reasons. All these materials relate to an NPDES permit issued to Central Contra Costa Sanitary District in February 2012.

a. Evidence Was Not Available When the Central Valley Water Board Considered the District's Permit

The Central Valley Water Board adopted the District's Permit on December 9, 2010. The San Francisco Regional Water Quality Control Board (San Francisco Water Board) noticed and considered renewal of an NPDES permit for discharges from the Central Contra Costa Sanitary District's Wastewater Treatment Plant in the fall of 2011 through early 2012. Thus, the evidence was not available at the time of the Central Valley Water Board's proceedings with respect to the District's Permit. The District did not request consideration of this Supplemental Evidence prior to this time because the State Board indicated in its Interlocutory Ruling that its decisions would be made based on the documents and information that was before the Central Valley Water Board. However, the Draft Order includes references outside the record before the Central Valley Water Board that address and relate to disputed issues. It includes citations to documents not in the record that pertain to the subject matter of the evidence now submitted. Accordingly, the District now requests consideration of the Supplemental Evidence identified here. The timing of this request is soon after learning that the Draft Order would be based on information not before the Central Valley Water Board, and it is properly being submitted together with the District's Response to Draft Order.

b. Statement of the Nature of Evidence and Facts To Be Proved

The Supplemental Evidence identified here is probative of the fact that there is no scientific basis to conclude that ammonia discharges are adversely affecting the Bay-Delta ecosystem or organisms outside the mixing zones for ammonia that were requested by the District for the Permit. It consists of public records that also provide evidence on the San Francisco Water Board's decision on the Contra Costa NPDES permit with respect to ammonium impacts on Suisun Bay, and whether or not it is necessary to adopt water quality-based effluent limitations (WQBELs) for ammonia that would require the Central Contra Costa Sanitary District to remove ammonia from its effluent. The evidence and issues presented to the San Francisco Water Board, and its subsequent decision, are directly related to the issues before the State Board. Notably, the San Francisco Water Board reached a different conclusion than the Central Valley Water Board, and different than the one that is being proposed within the Draft Order. The nature of the evidence here includes: comments and technical reports submitted by the Central Contra Costa Sanitary District, a group of entities referred to as the "Public Water Agencies,"[13] and the Bay Area Clean Water Agencies on the San Francisco Water Board's Tentative Order for discharges from the Central Contra Costa Sanitary District's Wastewater Treatment Plant;[14] the San Francisco Water Board's Response to Written Comments, which documents certain reasons as to why the San Francisco Water Board determined it inappropriate to adopt WQBELs for ammonia that would require ammonia removal; and, the final order (Order No. R2-2012-0016) as it was

[13] This group appears to be substantially identical to the group known as "Water Agencies" in this matter.

[14] The submittal of the comments from the "Public Water Agencies" is necessary to provide the appropriate context for the San Francisco Water Board's Response to Written Comments, and its ultimate decision to adopt Order No. R2-2012-0016, which included WQBELs for ammonia with consideration of dilution. Such documents are offered exclusively for the purpose of providing such context.

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adopted by the San Francisco Water Board after considering all of the comments, San Francisco Water Board's Response to Written Comments, and testimony provided at the hearing. Collectively, these documents provide evidence that calls into question findings contained in the Permit and Draft Order.

The Draft Order finds that the Central Valley Water Board could consider Dr. Teh's laboratory work as "relevant evidence to support its decision to deny an ammonia mixing zone," and that "available scientific evidence indicates that ammonia toxicity to copepods is one of the contributing factors compromising the integrity of the entire water body."[15] The Supplemental Evidence provided is contrary to that conclusion. Also, the Public Water Agencies made similar statements and proffered essentially the same evidence in the Central Contra Costa Sanitary District proceeding that is relied upon in the Permit and Draft Order.[16] However, evidence was provided in that proceeding that seriously called into question the results of Dr. Teh's laboratory work and findings.[17] These identified documents of BACWA and Contra Costa (and their attachments) are provided as evidence of all their statements that establish that the Teh work and other material relied upon in the Permit and Draft Order are not an appropriate basis for regulation.

For example, the Teh data referred to in the Draft Order has been reviewed by expert toxicologists.[18] As BACWA explains, in this independent analysis of the data, in particular Teh's 31-day full life-cycle bioassay results with *P. forbesi* that is specifically referred to in the Draft Order, the test treatment at 0.36 mg/l is not statistically different from the control, which significantly changes the conclusions from the Teh data results and essentially eliminates concern about adult copepod toxicity.[19]

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[15] Draft Order, p. 16.

[16] See Public Water Agencies Comments, p. 8.

[17] See, e.g., BACWA Comments, pp. 3-6.

[18] BACWA Comments, p. 4.

[19] BACWA Comments, p. 4.

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In its Response to Written Comments that are specific to the Public Water Agencies' comments with respect to this issue, the San Francisco Water Board found that, "[a]vailable information may not be as conclusive as the Water Agencies suggest. The copepod ammonium toxicity is not an issue for Suisun Bay because the ammonia concentrations observed in Suisun Bay are well below the low observed effect concentration derived in studies." Further the San Francisco Water Board's Response to Written Comments states that, "[m]ore information is needed to understand the relative contributions of the various Suisun Bay ammonia sources to Suisun Bay ammonia concentrations and their impacts . . . ."[20]

The District offers this Supplemental Evidence to prove that the Teh laboratory work is in fact not an appropriate basis for regulation and that it is being questioned in other proceedings. The documents before the San Francisco Water Board are also being offered to dispute the Draft Order's conclusion that there is scientific evidence of ammonia toxicity to copepods, which in turn is a contributing factor to "compromising the integrity of the entire waterbody."[21] The Supplemental Evidence also demonstrates that the San Francisco Water Board determined that it was premature to adopt WQBELs for ammonia that would result in ammonia removal (i.e., nitrification) because it is first necessary to obtain information to better evaluate ammonia impacts on Suisun Bay and throughout the Region.[22]

Related, the District also offers the identified documents as Supplemental Evidence in response to the Draft Order's finding that the Central Valley Water Board properly concluded that ammonia toxicity to copepods, "is a likely factor adversely affecting candidate, threatened, or endangered species populations . . . in the Delta and that the Permit's findings are supported by the administrative record."[23] The facts to be proven with the aforementioned documents go to the Draft Order's findings on ammonia toxicity to copepods and the effect that this would have on

[20] San Francisco Water Board's Response to Written Comments, p. 5.

[21] Draft Order, p. 16.

[22] San Francisco Water Board's Response to Written Comments, pp. 5, 9 ("While we agree that there are good reasons to be concerned about Suisun Bay ammonium concentrations, we do not believe available information is yet sufficient to require nitrification by the District.").

[23] Draft Order, p. 17.

listed species, and are evidence that the Permit and Draft Order's findings are not correct. The Draft Order finds that "[a]s would be expected, ammonia's toxic effects on copepods also affects those species that feed on copepods."[24] As is discussed immediately above, the Contra Costa and BACWA Comments in conjunction with the San Francisco Water Board's Response to Written Comments and the San Francisco Water Board's adoption of Order No. R2-2012-0016, collectively, undercut conclusions related to ammonia toxicity to copepods. By extension, such posited effects on listed species do not have support.

Moreover, the Draft Order cites to the Central Valley Water Board's hypothesis on the inhibition of diatom primary production in Suisun Bay.[25] The Central Valley Water Board is hypothesizing that by allegedly inhibiting diatom production, ammonia is adversely affecting critical fish habitat by reducing available food for the listed and candidate species of concern. The Draft Order appears to concur with this hypothesis.[26] However, the evidence provided here demonstrates that while this possibility may be a basis for concern, such impacts from ammonium are not well understood and that additional study is necessary.[27]

Accordingly, the District offers the above-mentioned documents both as evidence on the technical matters as described and to prove that the Draft Order's findings with respect to the Central Valley Water Board's determinations on these issues are inconsistent with other, more recent, regional water quality control board decisions on NPDES permits that have similar circumstances to those addressed in the District's Permit.

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[24] Draft Order, p. 16.

[25] Draft Order, p. 17.

[26] Draft Order, p. 17.

[27] See BACWA Comments, pp. 7-8; see also San Francisco Water Board's Response to Written Comments, p. 5 ("The potential for ammonium from the District's discharge to inhibit phytoplankton productivity in Suisun Bay exists, but needs to be evaluated in the context of other possible factors that could also affect productivity.").

3. *Numeric Nutrient Endpoint Development for San Francisco Bay Estuary: Literature Review and Data Gaps Analysis*, L. McKee, M. Sutula, A. Gilbreath, J. Beagle, D. Gluchowski, and J. Hunt, Southern California Coastal Water Research Project (Technical Report 644 – June 2011) (NNE Development for San Francisco Bay Estuary Report)

The District requests that the NNE Development for San Francisco Bay Estuary Report be received as Supplemental Evidence for the following reasons.

a. Evidence Was Not Available When the Central Valley Water Board Considered the District's Permit

The NNE Development for San Francisco Bay Estuary Report was produced under a contract between the State Board and the Southern California Coastal Research Project, and was released in June 2011. Thus, the evidence was not available at the time of the Central Valley Water Board's proceedings with respect to the District's Permit. The District did not request consideration of this Supplemental Evidence prior to this time because the State Board indicated in its Interlocutory Ruling that its decisions would be made based on the documents and information before the Central Valley Water Board. However, the Draft Order reaches beyond the record before the Central Valley Water Board, and the District now requests consideration of the Supplemental Evidence identified here. The timing of this request is soon after learning that the Draft Order would be based on information not before the Central Valley Water Board, and it is properly being submitted together with the District's Response to Draft Order.

b. Statement of the Nature of Evidence and Facts To Be Proved

The NNE Development of San Francisco Bay Estuary Report is a public report prepared with State Board funding. The report was produced by the Southern California Coastal Water Research Project with review and guidance from the members of the San Francisco Bay Nutrient Numeric Endpoint Technical Advisory Team, whose members include: Katharyn Boyer, Romberg Tiburon Center, San Francisco State University; James Cloern, US Geological Survey; Richard Dugdale, Romberg Tiburon Center, San Francisco State University; and, Raphael Kudela, University of California at Santa Cruz.[28] The State Board determined that such a report

[28] NNE Development of San Francisco Bay Estuary Report, p. ii.

was necessary as part of its process for developing nutrient water quality objectives for the state's surface waters because San Francisco Bay represents California's largest estuary. Specifically, the State Board has determined it appropriate to develop an estuary-specific NNE framework for the San Francisco Bay Estuary.[29] The purpose of the NNE Development of San Francisco Bay Estuary Report was to review literature and data relevant to the assessment of eutrophication in San Francisco Bay. This information will then be used to formulate a work plan for developing estuary-specific NNEs.

The District proposes to include the NNE Development of San Francisco Bay Estuary Report to address facts at issue in the Draft Order. As discussed previously, the Draft Order finds that ammonia toxicity to copepods and the inhibition of diatom primary production is a likely factor adversely affecting candidate, threatened, or endangered species populations.[30] However, the NNE Development of San Francisco Bay Estuary Report provides evidence that ammonium affects on diatom blooms is not well understood and that additional work is needed to resolve this issue.[31] Thus, based on the information contained in the NNE Development of San Francisco Bay Estuary Report, which is the State Board's own document, the NNE Development of San Francisco Bay Estuary Report is being proffered as Supplemental Evidence that calls into question the Draft Order's findings with respect to ammonia toxicity impacts to candidate, threatened, or endangered species.

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[29] NNE Development of San Francisco Bay Estuary Report, p. iii.

[30] Draft Order, p. 17.

[31] See, e.g., NNE Development of San Francisco Bay Estuary Report, p. 153 ("the ecological importance of ammonium inhibition of spring diatoms blooms is not well understood relative to factors known to control primary productivity . . . ."); see also, *id.*, p. 155 ("Elevated ammonium concentrations have been suggested as a major mechanism by which spring diatom blooms appear to be suppressed in the North Bay and lower Sacramento River . . . . Despite this evidence, the ecological importance of ammonium inhibition of spring diatoms blooms is not well understood relative to factors known to control primary productivity, particularly in other regions of the Bay where water column chlorphyll a appears to be increasing. Thus the linkage between ammonium concentrations and Bay beneficial uses is not at this time universally accepted. San Francisco Bay TAT members agree that additional data synthesis is required to better understand the role of ammonium in SF Bay.").

4. Letter to Mike Chotkowski, Department of Interior, United States Fish and Wildlife Service from Sacramento Regional County Sanitation District, Comments on Endangered and Threatened Wildlife and Plants; 12-month Finding on a Petition to List the San Francisco Bay-Delta Population of the Longfin Smelt as Endangered or Threatened, Docket No. FWS-R8-ES-2008-0045 (SRCSD Comments on Longfin Smelt)

The District requests that the SRCSD Comments on Longfin Smelt be received as Supplemental Evidence for the following reasons.

a. Evidence Was Not Available When the Central Valley Water Board Considered the District's Permit

The District submitted its SRCSD Comments on Longfin Smelt to the U.S. Fish and Wildlife Service (USFWS) on April 27, 2012, in response to the USFWS's April 2, 2012 Federal Register Notice (USFWS 2012 Notice). Thus, the evidence was not available at the time of the Central Valley Water Board's proceedings with respect to the District's Permit. The District did not request consideration of this Supplemental Evidence prior to this time because the State Board indicated in its Interlocutory Ruling that its decisions would be made based on the documents and information that was before the Central Valley Water Board. However, the Draft Order reaches beyond the record before the Central Valley Water Board. Specifically here, the Draft Order proposes to take official notice of the USFWS 2012 Notice.[32] The District has separately objected to the inclusion of the USFWS 2012 Notice.[33] However, to the extent that the Draft Order proposes to receive via the USFWS 2012 Notice evidence on this issue, it is entirely appropriate for the State Board to also accept as Supplemental Evidence the SRCSD Comments on Longfin Smelt. The timing of this request is soon after learning that the Draft Order would be based on information not before the Central Valley Water Board, and it is properly being submitted with the District's Response to Draft Order.

b. Statement of the Nature of Evidence and Facts To Be Proved

The nature of the evidence here, the SRCSD Comments on Longfin Smelt, are comments submitted directly in response to the USFWS 2012 Notice regarding Longfin smelt. In the

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[32] Draft Order, p. 17, fn. 63.

[33] See District's Response to Draft Order, p. 62.

SRCSD Comments on Longfin Smelt, the District provides to the USFWS the best available science on water quality in the Bay-Delta. The SCRCD Comments on Longfin Smelt call into question various statements contained in the USFWS 2012 Notice.

The facts that the District intends to prove with the SRCSD Comments on Longfin Smelt are all those stated in that document, which is incorporated by reference. The USFWS 2012 Notice selected for reference in the Draft Order is not based on best available science, and the Draft Order's reliance on matters in the notice is thus by extension also not based on best available science.

Among other matters, the SRCSD Comments on Longfin Smelt provide specific evidence that relate to Teh's findings with respect to ammonia toxicity to copepods. Included as part of the SRCSD Comments on Longfin Smelt is an independent critique of Teh's report which is referenced in the USFWS 2012 Notice by Pacific Ecorisk. The Pacific Ecorisk review raises serious questions as to the validity of key results identified in Teh's study, and with Teh's data.[34] In sum, the Pacific Ecorisk review raises serious concerns with the validity of the key threshold value of 0.36 mg/L that is relied on to support findings of copepod toxicity. Thus, the Supplemental Evidence contained within the SRCSD Comments on Longfin Smelt question the USFWS 2012 Notice, and the Draft Order's findings related thereto.

5. <u>Letter to Municipal Wastewater Dischargers from Bruce H. Wolfe, Executive Officer, California Regional Water Quality Control Board, San Francisco Bay Region, subject Water Code Section 13267 Technical Report Order Requiring Submittal of Information on Nutrients in Wastewater Discharges (March 2, 2012) (NNE Letter)</u>

The District requests that the NNE Letter be received as Supplemental Evidence for the following reasons.

a. <u>Evidence Was Not Available When the Central Valley Water Board Considered the District's Permit</u>

The NNE Letter was issued on March 2, 2012. It was not available at the time of adoption of the District's Permit. The Draft Order goes beyond the record before the Central Valley Water

[34] See SRCSD Comments on Longfin Smelt, pp. 3-4.

Board or at minimum beyond issues that are addressed in the Permit but are addressed in the NNE Letter, and the District now requests consideration of the Supplemental Evidence identified here. The timing of this request is soon after learning that the Draft Order would be based on information not before the Central Valley Water Board, and it is properly being submitted together with the District's Response to Draft Order.

b. Statement of the Nature of Evidence and Facts To Be Proved

The NNE Letter is a communication from the San Francisco Water Board. It supports that ecological response to nutrient loads in the Bay-Delta estuary is unknown and a matter to be determined.

In this regard, the Draft Order describes a Nutrient Numeric Endpoint (NNE) framework that is under development.[35] It discusses effects or hypothesized effects of nutrients as to issues not raised in the Permit, such as Suisun Marsh Wetlands. Had that issue been posed in the tentative permit, the District would have provided comment and evidence. With respect to the NNE, and while there are no citations to any record documents, the District agrees that the NNE framework required to begin to set numeric objectives for nutrients in San Francisco Bay and the Delta is "currently under development." No proposed numeric objectives for total N and total P in the Bay or Delta are publicly available or proposed.

In this regard, if a proposed order is to go beyond matters that were subject to comment in the record and to describe an NNE framework and what may be expected, it is also necessary to recognize other information that does not support that some specific outcome is "likely" as stated.

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[35] Draft Order, p. 22.

CONCLUSION

The District respectfully requests that the State Board grant the requests.

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DATED: June 15, 2012

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DATED: June 15, 2012

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SANITATION DISTRICT

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EXHIBIT A

WASTEWATER DISINFECTION FOR

HEALTH PROTECTION

Sanitary Engineering Branch
California Department of Health Services

February 1987

EXHIBIT A

CONTENTS

INTRODUCTION

The Report, "Wastewater Disinfection for Health Protection,has been prepared to serve as a basis for a review and evaluation of the "Uniform Guidelines for Sewage Disinfection". The Guidelines aid the Department of Health Services in recommending disinfection requirements to Regional Water Quality Control Boards.

A major consideration in the treatment of domestic wastewater and in the specific action of discharge requirements is the protection of public health from waterborne disease. Historically, the Department has provided recommendations on disinfection needs for health protection based on the particular discharge situation. The recommendation is generally in terms of a limiting concentration of indicator organisms (total coliform bacteria) in the waste discharge.

In the period of 1972-76, the Department developed uniform disinfection guidelines for various discharge situations based on available public health and technical information so that there would be a reasonable degree of uniformity in the recommendations made to the various regional boards for discharges involving similar circumstances and a uniform degree of health protection provided for different discharge circumstances.

The Sanitary Engineering Branch, Department of Health Services, was requested by the State Water Resources Control Board to conduct a thorough review and reappraisal of the guidelines as a result of questions raised regarding disinfection needs by regional boards and challenges of disinfection requirements by waste dischargers. Inasmuch as a substantial amount of information has been generated over the last decade on wastewater disinfection and public health effects associated with various receiving water uses, a review of the guidelines was appropriate.

A number of specific issues associated with wastewater disinfection were identified by the State Water Resources Control Board, regional boards, and others, and the review report was structured to provide information pertinent to these issues. The report covers chlorination chemistry, cost and safety, health effects of microbiological contaminants, and health effects of organic chemical contaminants.

A Technical Advisory Committee of experts in wastewater treatment and disinfection, waterborne disease, indicator organisms, and related matters assisted in the development of information and in assessing the accuracy and completeness of the contents of the review report.

Based on the review report information, position papers were developed for each of the issues that had been identified. A health policy committee made up of health authorities and experts was formed to advise the staff on the public health aspects of the review report and on responsible public health positions to be taken on the disinfection issues. The Uniform Guidelines for Sewage Disinfection were then reviewed to determine conformance with the recommended position on each issue.

The issues, committee membership, and proposed uniform guidelines are contained in the appendices of the report.

Special thanks must be given the members of the Technical Advisory Committee and the Health Policy Committee. They spent a significant amount of time and effort in identifying and providing source material for the report, critically reviewing the staff work, assisting in the report write-up, and providing guidance to the staff.

PART I

CHLORINATION

CHEMISTRY, COST, AND SAFETY

TABLE OF CONTENTS

CHLORINATION

CHEMISTRY OF CHLORINATION

When chlorine gas is added to water, it hydrolyzes rapidly according to the equation:

$$Cl_2 + H_2O \rightleftharpoons HOCl + H^+ + Cl^-$$

The hydrolysis of chlorine and, hence, the formation of hypochlorous acid (HOCl) is complete within a few seconds at the water temperatures typically found at wastewater treatment plants. The concentration of HOCl is dependent on the total chlorine concentration and pH. The amount of molecular chlorine present increases as the pH decreases.

If hypochlorite solutions, e.g., sodium hypochlorite (NaOCl) or calcium hypochlorite [$Ca(OCl)_2$] are used, hypochlorous acid is also formed according to the following equations:

$$NaOCl + H_2O \rightarrow HOCl + Na+ + OH^-$$

$$Ca(OCl)_2 + 2H_2O \rightarrow 2HOCl = Ca^{++} + 2(OH)^-$$

Hypochlorous acid, which is classified as a weak acid, undergoes partial dissociation as follows:

$$HOCl \rightleftharpoons H^+ + OCl^-$$

Low pH favors formation of HOCl, while high pH favors the formation of hypochlorite ion (OCl^-), which is a less-effective disinfecting species. At pH 6 to 9, the reaction is essentially incomplete and both species are present to some degree. When added to water gaseous chlorine tends to lower the pH and hypochlorite tends to raise the pH. Therefore, the residual formed by gaseous chlorine would be a more effective disinfectant in poorly buffered waters. The distributions of HOCl and OCl^- as a function of pH are shown in the following table [Heim and Burris (1979)].

DISTRIBUTION OF HOCl AND OCl^- WITH pH @ 20 °C
Percent of Total Free Chlorine

pH	HOCl	$OClO^-$
6.0	96.8	3.2
7.0	75.2	24.8
7.5	49.1	50.9
8.0	23.2	76.8
9.0	2.9	97.1

Ammonia compounds and numerous other constituents are present in wastewater that react immediately with hypochlorous acid. Consequently, free chlorine (HOCl and OCl^-) is consumed or converted to combined forms of chlorine (chloramines). Various reduced inorganic substances ($S^=$, $SO_3^=$, HS^-, NO_2^-, FE^{++}, Mn^{++}, etc.) react with both free and combined chlorine to reduce these compounds so that the active chlorine compound is eventually reduced to the stable chloride ion, which is nonbactericidal.

There is usually an appreciable amount of ammonia nitrogen in wastewater. The ammonium ion (NH_4^+) exists in equilibrium with ammonia nitrogen (NH_3) and hydrogen ion as follows:

$$NH_4^+ \rightleftharpoons NH_3 + H^+$$

Ammonia predominates above pH 9.3, while the ammonium ion predominates below pH 9.3. There are three types of chloramines that can be formed when ammonia reacts with chlorine, i.e., monochloramine, dichloramine, and trichloramine. When ammonia is present in wastewater, the following reactions occur:

$$NH_3 + HOCl \rightarrow H_2O + NH_2Cl$$

$$NH_2Cl + HOCl \rightarrow H_2O + NHCl_2$$

$$NHCl_2 + HOCl \rightarrow H_2O + NCl_3$$

The extent to which the various chloramines are formed depends on the pH, temperature, contact time, and relative concentration of each reacting substance. Monochloramine and dichloramine, while substantially less effective disinfecting agents than free chlorine, are nevertheless effective disinfecting agents, although they do require a greater contact time to kill microorganisms. Monochloramine will predominate at pH 8, and dichloramine, which is approximately twice as strong a disinfecting agent as monochloramine, will predominate at pH 5. Lower pH values and high chlorine dosages favor the formation of trichloramine. Sepp and White (1981) reported that trichloramine only forms when the ratio of chlorine to ammonia nitrogen exceeds 12 to 1 by weight.

Chlorine may also combine with organic nitrogen compounds that are normally present in sewage, e.g., amino acids and proteinaceous substances. The extent of these reactions has not been well studied, nor has the disinfecting power of the chlorine compounds. However, it is generally concluded [Sepp and White (1981)] that ammonia chloramines are far superior to organic chloramines as disinfecting coumpounds, and that the process of chlorination is hindered by the presence of organic nitrogen.

Leidholdt (1982) estimated the effectiveness of the various chlorine residuals, as indicated in the following table:

TYPE OF RESIDUAL	CHEMICAL ABBREVIATION	ESTIMATED EFFECTIVENESS COMPARED TO HOCl
Hypochlorous acid	$HOCl$	1
Hypochlorite ion	OCl^-	1/100
Monochloramine	NH_2Cl	1/150
Dichloramine	$NHCl_2$	1/80
Trichloramine	NCl_3	No estimate

It is important to note that estimated effectiveness of the various chlorine compounds is based on relatively short contact times - five minutes or less. Given adequate contact time, however, combined forms of chlorine may be nearly as effective disinfecting agents as free chlorine. Studies by the County Sanitation Districts of Los Angeles County (1977) and Engineering-Science (1985) have indicated that contact times of 1-2 hours with combined chlorine provides similar disinfection capability under the same conditions.

A detailed treatise on breakpoint chlorination is unwarranted in this document, but it is worthwhile to briefly describe this phenomenon even though it is not a primary goal in wastewater disinfection. When chlorine is added to wastewater it will react with various inorganic substances and, in effect, some of the chlorine is "used up" (no residual forms and no disinfection occurs) to satisfy this initial demand. The remaining chlorine reacts with the organics and the ammonia to form both chloramines and chloro-organic compounds, that is, the combined chlorine residuals. As the chlorine dose is increased, the chloramine residual (mostly monochloramine), also increases. Eventually, a point is reached where further chlorine addition begins to destroy the chloramines and some of the chloro-organic compounds. The reduction of the chloramines continues to a point called the breakpoint, where any additional chlorine results in feee chlorine residuals. Because the free chlorine residual forms only after the breakpoint, the process is called breakpoint chlorination.

DECHLORINATION

In the early 1970s, studies by Esvelt et al (1971) of domestic wastewater discharged into San Francisco Bay indicated that chlorination increased the toxicity of all of the wastewaters examined. It was also found that dechlorinated effluent was less toxic than the chlorinated or unchlorinated effluent. Another study of San Francisco Bay by Stone et al (1973) concluded that chlorine residuals of 0.06 mg/l were damaging to Bay aufwuchs and phytoplankton. The toxicity attributed to the chlorine residuals was completely removed by dechlorination.

An EPA Task Force Report (1976) reported that adverse effects on freshwater aquatic organisms have been observed at chlorine residuals as low as 0.02 mg/l and in marine waters at residuals above 0.01 mg/l. A design optimization study by Sepp and Bao (1980) demonstrated that optimized pilot plants could reduce toxicity to test fish by an average of 43 percent and contained 50 percent less chlorine residual than full-scale plant effluents. However, the study also found that it is not possible to eliminate all of the chlorine-induced toxicity by design optimization alone, and that dechlorination is still necessary in situations where toxicity must be removed. Therefore, in California, many wastewaters are required to be dechlorinated prior to discharge in order to eliminate or minimize effects on the aquatic environment.

Dechlorination is the practice of removing all or part of the total combined chlorine residual remaining after chlorination. Dechlorination can be effected using reducing agents such as sulfur dioxide, sodium bisulfite, sodium sulfite, activated carbon, or by aeration for certain volatile forms of chlorine.

Sulfur dioxide (SO_2) is the major dechlorinating agent used in California. It is usually the chemical of choice because it is relatively inexpensive, easy to control, reacts quickly to completely remove free or combined chlorine residual, and chlorination equipment can be used to handle the gas. It is available as a liquified gas and is about 20 times more soluble than chlorine in water [Helz and Kosak-Channing (1984)]. Upon dissolving in water, sulfur dioxide rapidly hydrolyzes to form a weak solution of sulfurous acid (H_2SO_3), as follows:

$$H_2O + SO_2 \rightarrow H_2SO_3$$

The sulfurous acid, in turn, rapidly ionizes to an equilibrium mixture of bisulfite (HSO_3) and sulfite (SO_3=) as follows:

$$H_2SO_3 \rightleftharpoons H^+ + HSO_3^-$$

$$HSO_3^- \rightleftharpoons H^+ + SO_3^=$$

Equimolar concentrations of bisulfite and sulfite occur at pH 7.2. The sulfite reacts with both free and combined chlorine as follows:

Free Chlorine
$$H_2SO_3 + HOCl \rightarrow H_2SO_4 + HCl$$

Combined Chlorine
$$H_2SO_3 + NH_2Cl + H_2O \rightarrow NH_4HSO_4 + HCl$$

The reaction ratio of sulfur dioxide to chlorine is 9:1, which converts chlorine to chloride ion. In moderately well-buffered wastewaters, the acidity produced has a small effect on pH. The doses of sulfur dioxide required for dechlorination are small compared to the alkalinity in most cases. In weakly buffered waters, however, acidity is a potential problem.

For a more complete coverage of the chemistry and kinetics of wastewater disinfection and dechlorination, the following three publications are suggested for further reading:

Sepp, E., and White, G.C., Manual for Wastewater Chlorination and Dechlorination Practices. Publication No. 53-A, California State Water Resources Control Board, Sacramento, CA, March 1981.

White, G.C., Disinfection of Wastewater and Water for Reuse. Van Nostrand and Reinhold Co., New York, NY, 1978.

U.S. Environmental Protection Agency, Design Manual - Municipal Wastewater Disinfection. Report No. EPA/625/1-86/021, U.S. Environmental Protection Agency, Office of Water Research and Development, Cincinnati, OH, October 1986.

COST OF DISINFECTION

Although the Department of Health Services does not directly base its health-related standards and guidelines on economics, the "reasonableness" of water quality and treatment processes from an economic standpoint should be evaluated. Obviously, chlorine disinfection is a well-proven technology to reduce or remove pathogenic microorganisms in wastewater and, subsequently, the receiving waters. While it is difficult to obtain specific data relating chlorination costs to effluent coliform concentrations, information is available that adequately documents the cost of disinfection as a unit process and as a percentage of the overall treatment cost.

Hubly et al (1985) estimated disinfection costs for 1, 10, and 100 MGD wastewater treatment plants adjusted to a 1981 cost index. The cost estimates were based on a 1980 EPA study of 92 wastewater treatment construction projects. Capital costs for a chlorine disinfection system must include non-construction project costs in addition to the actual construction costs. These non-construction project costs include administration, legal costs, architect and engineering fees, inspections, and contingencies, which add 28 percent to new projects or 36 percent on enlargement or upgrading projects. Hubly et al (1985) used a median figure of 32 percent for non-construcion costs in their analysis.

The peculiarities of individual situations leads to large variations in capital costs, and the EPA study showed that individual construction projects varied by more than a factor of three in about 20 percent of their cases. Using the EPA data, Hubly et al (1985) determined the chlorination capital costs to be as follows:

Plant Size (MGD)	Capital Costs (1981 Index) ($)
1	96,000
10	446,000
100	2,198,000

Although no information was provided regarding the level of treatment or chlorine contact times and, hence, the size of the chlorination contact tanks, it is reasonable to assume that most of the plants provided biological secondary treatment and had theoretical chlorine contact times of 15 to 60 minutes.

Annual costs for several interest rates with 20-year amortization periods are as follows:

Annual Chlorination Capital Costs

Interest Rate (%)	Amortization Period (Yrs)	Plant Size (MGD) 1	10	100
0	1	$96,000	$446,000	$2,198,000
0	20	4,800	22,300	109,000
3	20	6,432	29,882	147,226
6	20	8,352	38,802	191,226
9	20	10,464	48,614	238,582
12	20	12,864	59,764	294,532
15	20	15,360	71,360	351,680

For this analysis, a 20-year amortization period has been assumed, and 9 percent was selected as a reasonable interest rate. The unit capital costs for chlorine are then estimated as follows:

Unit Chlorination Capital Costs

Plant Size (MGD)	Annual Cost ($)	Annual Unit Cost (cents/1000 gal)
1	10,464	2.9
10	48,614	1.3
100	239,582	0.7

The operation and maintenance (O&M) costs for chlorine disinfection include chlorine, labor, supplies, and power. Chlorine alone typically accounts for 50 percent or more of all O&M costs. The chlorine dose depends on the specific wastewater being treated, the degree of treatment, and the coliform limits that must be achieved. Hubly et al (1985) determined chlorine cost for two different dosages - 6 and 10 mg/l. The cost per unit disinfected wastewater is shown below for these two dosages. The unit costs are based on the assumption that the smaller plants will use 1-ton cylinders and the large plants will purchase chlorine in railroad tank cars.

Chlorine Costs

Plant Size (MGD)	Price ($/bl)	Dose (mg/l)	Annual Usage (tons/yr)	Unit Cost (Cents/1000 gal)
1	0.11	6	9.25	0.6
		10	15.29	0.9
10	0.11	6	92.5	0.6
		10	152	0.9
100	0.08	6	925	0.4
		10	1520	0.7

Labor is the second category of operating cost. Hubly et al (1985) assumed maintenance, labor, and supervision to be proportional to plant investment totaling 1 percent of cost. This assumption may be adequate for estimating generalized costs but may not accurately reflect incremental costs associated with different chlorine dosages. Labor cost is assumed to be $10/hour. The labor costs per 1000 gallons of wastewater treated are shown in the following table.

Labor Costs

Plant Size (MGD)	Annual Labor (Hrs)	Annual Costs ($)	Unit Costs (Cents/1000 gal)
1	460	4,600	1.2
10	1840	18,400	0.5
100	7360	73,600	0.2

Power costs are primarily for heating, lighting, and ventilation. Although electricity can still be bought at 4-5 cents per kilowatt-hour, new capacity generally costs at least 6 cents, and this number was used to calculate the power costs. The power costs for 1, 10, and 100 MGD plants were determined to be 0.2, 0.03 and 0.004 cents/1000 gallons, respectively.

Assuming that supplies for maintenance are 1 percent of capital costs, the total O&M costs for chlorine disinfection are as follows:

Chlorination O&M Costs
(Cents/1000 gal)

	1 MGD		10 MGD		100 MGD	
	6 mg/l	10 mg/l	6 mg/l	10 mg/l	6 mg/l	10 mg/l
Chlorine	0.6	0.9	0.5	0.9	0.4	0.7
Labor	1.2	1.2	0.5	0.5	0.2	0.2
Power	0.2	0.2	0.03	0.03	0.004	0.004
Supplies	0.03	0.03	0.02	0.02	0.08	0.08
Totals	2.0	2.4	1.1	1.5	0.6	0.9

The total chlorination costs (capital + operation and maintenance) based on the 1981 cost index are shown in the following table:

Total Chlorination Costs
(Cents/1000 gal)

Plant Size (MGD)	Chlorine Dosage 6 mg/l	10 mg/l
1	4.9	5.2
10	2.4	2.8
100	1.3	1.6

As can be seen from the table, chlorine costs decrease as the size of the facility increases. Both capital and O&M costs per unit of wastewater treated decrease as the plant capacity increases. At smaller treatment plants, the O&M costs are most sensitive to labor costs, while sensitivity to the cost of chlorine is dominant in larger plants.

Horvath (1986) determined the capital and O&M costs for chlorination <u>and</u> dechlorination of wastewater effluent at various wastewater reclamation plants operated by the County Sanitation Districts of Los Angeles County. Capital costs include site clearing, excavation, piping, and safety measures. The approximate costs for the electrical and instrumentation systems are assumed to be 10 percent of the capital costs. The capital costs are based on data generated by the Pomona Virus Study [County Sanitation Districts of Los Angeles County (1977)] tied to the May 1986 construction and materials cost index (ENR-5452) and are amortized at 10 percent for 20 years. Other assumptions include a theoretical contact time of two hours, and a dechlorination contact time of five minutes.

Table 1 indicates the actual flow, chlorine dosage and quantity used, and the unit cost of the chlorine and sulfur dioxide (for dechlorination) for 10 wastewater treatment plants, all of which provide at least secondary treatment of the wastewater. Tables 2 and 3 present the direct and indirect cost of chlorination and dechlorination, respectively, which includes operation, maintenance and instrumentation workers salaries, equipment services, upkeep and repair, safety measures, trouble-shooting, periodic hydro testing of the tanks, and other costs for miscellaneous services.

The total disinfection capital and O&M costs, based on data from the Pomona Wastewater Reclamation Plant (chlorine dosage = 9.5 mg/l) are shown in Table 4 for chlorination and Table 5 for dechlorination. The total cost for both processes is $116 per million gallons of tertiary treated effluent, which is 11.6 cents per thousand gallons. If <u>both</u> the chlorine and sulfur dioxide dosages were increased by 5.0 mg/l, the total unit cost would be increased by $16 per million gallons, which is 1.6 cents per thousand gallons.

A recent cost analysis of a proposed 30 MGD wastewater reclamation facility by Engineering-Science (1986) provided rough estimates of the chlorine dosages necessary to reduce the total coliform concentration in tertiary-treated effluent to 2.2/100 ml or less. It was estimated that, for a treatment chain consisting of secondary biological treatment, chemical coagulation, flocculation, sedimentation, filtration, and chlorine disinfection with a 90-minute theoretical contact time, 344 tons/yr of chlorine (@$311/ton) would be required to provide a dosage of 11 mg/l. It was assumed that the full 30 MGD capacity of the reclamation facility would be used for irrigation an average of 250 days per year. This would result in a chlorine cost of $107,000/year, or 1.4 cents/1000 gallons of treated effluent. An increase of 5 mg/l in the chlorine dosage would increase the unit chlorine cost by approximately six-tenths of a cent per thousand gallons.

Similarly, it was estimated that a treatment chain consisting of secondary biological treatment, low-dose coagulant addition, filtration, and chlorine disinfection with a 90-minute theoretical chlorine contact time would require 467 tons/year of chlorine to provide a dosage of 15 mg/l. This

Table 1

Chemical Costs for Wastewater Disinfection
by Chlorination and Dechlorination
[Horvath (1986)]

Facility	Flow (MG/day)	Dosage (mg/l)		Usage (Tons/yr)		Price ($/ton)		Total Cost ($Cl+SO_2$/yr)	Cost ($/MG)
		Cl	SO_2	Cl	SO_2	Cl	SO_2		
JWPCP[1]	350			8000	0	134	-	1,080,000	8.6
Los Coyotes[2]	37.5	8.25	3.0	500	180	176.5	251	133,450	9.75
SJC[2]	62.5	8.5	2.25	860	230	176.5	251	209,650	9.19
WN[2]	15	10.5	5.75	260	140	176.5	376	98,530	18.00
Long Beach[2]	19	9.0	6.0	280	185	176.5	251	95,855	13.82
Pomona[2]	10	9.5	3.0	150	20	176.5	376	33,995	9.30
Lancaster[3]	6.2	19.5	0.0	195	0	176.5	376	34,417*	15.2*
Saugus[4]	5.5	12.5	7.0	110	63	176.5	376	43,103	21.4
Valencia[2]	3.9	12.0	9.0	75	57	176.5	376	34,669	24.3
La Canada[5]	0.1	22	0.0	4.0	0	520	-	2,080*	52.0*

*Chlorination only

[1]Partial secondary treatment - must meet ocean shellfish criteria
[2]Filtered secondary treatment - median coliform requirement = 2.2/100 ml
[3]Oxidation pond treatment - median coliform requirement = 2.2 or 23/100 ml (seasonal)
[4]Secondary treatment - median coliform requirement = 2.2/100 ml
[5]Secondary treatment - median coliform requirement = 23/100 ml

Table 2

Chlorination Operation and Maintenance Costs (1985)
[Horvath (1986)]

Treatment Plant (Flow)	Direct	Indirect	Cost ($) Total	Per MG
Joint Water Pollution Control Plant (132,000 MG)	2,111,000	484,000	2,595,000	19.64
Los Coyotes (12,640 MG)	136,100	70,900	207,000	16.38
San Jose Creek (18,640 MG)	156,400	102,100	258,500	13.87
Whittier Narrows (4,110 MG)	77,800	28,400	106,200	25.84
Long Beach (6360 MG)	106,600	47,200	153,800	24.18
Pomona (4000 MG)	60,300	32,700	93,000	23.25
Lancaster (2,014 MG)	59,000	24,100	83,100	41.26
Saugus (1827 MG)	41,200	17,300	58,500	32.02
Valencia (1,333 MG)	35,000	19,000	54,000	40.51
La Canada (38 MG)	4,940	2,800	7,740	203.68

Table 3

Dechlorination Operation and Maintenance Costs (1985)
[Horvath (1986)]

Treatment Plant (Flow)	Direct	Indirect	Cost ($) Total	per MG
Joint Water Pollution Control Plant *	–	–	–	–
Los Coyotes (12,640 MG)	45,800	64,100	109,000	8.62
San Jose Creek (18,640 MG)	46,600	99,000	145,600	7.81
Whittier Narrows (4,110 MG)	31,000	24,800	55,800	13.58
Long Beach (6,360)	42,700	43,100	85,800	13.49
Pomona (4,000)	17,400	32,800	50,200	12.55
Lancaster *	–	–	–	–
Saugus *	–	–	–	–
Valencia (1,333 MG)	18,100	19,000	37,100	27.83
La Canada *	–	–	–	–

* Effluent not dechlorinated

Table 4

Cost Estimates for Chlorination Process
Plant Size = 10 MGD
[Horvath (1986)]

Capital Costs (Thousands of Dollars)

Chlorine Contact System		654
Chlorine Mixing System		109
Chlorine Feeding and Control System		262
Chlorine Handling and Storage Building		131
	Subtotal	1,156
Electrical (10%)		116
	Subtotal	1,272
Contingencies (20%)		254
	Subtotal	1,526
Engineering (15%)		229
	Total	1,755

Amortized at 10% for 20 years ($/million gallons)	56
Operation and Maintenance Costs* (Pomona WRP)	23
Total Process Cost ($/million gallons)	79

*Based on 9.5 mg/l chlorine dosage

Table 5

Cost Estimate for Dechlorination with Sulfur Dioxide
Plant Size = 10 MGD
[Horvath (1986)]

<u>Capital Costs (Thousands of Dollars)</u>		
SO^2 Contact System		44
SO_2 Mixing System		65
SO_2 Feeding and Control System		262
SO_2 Handling and Storage System		<u>131</u>
	Subtotal	502
Electrical (10%)		<u>50</u>
	Subtotal	552
Contingencies (20%)		<u>110</u>
	Subtotal	662
Engineering (15%)		<u>99</u>
	Total	761

Amortized at 10% for 20 years ($/million gallons)	24
<u>Operation and Maintenance Costs* (Pomona WRP)</u>	13
Total Process Cost ($/million gallons)	37

*Based on 2 mg/l chlorine residual

would result in a chlorine cost of $145,800 per year, or 1.9 cents/1000 gallons of treated effluent. Again, a 5 mg/l increase in chlorine dosage would increase the unit cost of chlorine by approximately six-tenths of a cent per 1000 gallons.

Culp et al (1980) tabulated the cost of various types and levels of wastewater treatment. The capital costs were based on a 1977 cost index, and it is assumed that the other costs are also based on a 1977 cost index. However, since the base year and cost indices for these other costs were not given, none of the costs were updated. Therefore, the values given may be significantly less than current costs, but the relative costs between the various unit processes are still valid.

Figures 1 through 4 are based on 1977 cost data from Culp et al (1980) and are indicative of estimates for the capital cost of chlorine contact basins and chlorine feed facilities, the operation and maintenance costs for chlorine contact tanks, feed system, and residual monitoring, and the annual energy requirements for chlorination systems.

Table 6, based on Culp et al (1980) data, indicates the unit cost (cents/1000 gallons) of treating wastewater by secondary and tertiary treatment processes, including chlorination and dechlorination. The chlorination costs are based on a dosage of 6 mg/l (@$250/ton of chlorine) and a chlorine contact time of 30 minutes. The dechlorinaion costs are based on a sulfur dioxide dose of 3 mg/l and a cost of $181 per ton of sulfur dioxide.

As can be seen in Table 6, the unit cost to treat wastewater is dependent on the size of the wastewater treatment facility and decreases as the plant capacity increases., The cost differential based on treatment plant design flow is reflected in each of the treatment processes shown in the table. For example, the combined chlorination and dechlorination unit costs vary from 6.5-7.2 cents/1000 gallons for 1 MGD capacity treatment plants down to 1.5-1.6 cents/1000 gallons for 50 MGD capacity treatment plants. Therefore, it can be seen that chlorination and dechlorination account for approximately 7-9 percent of the total cost (based on cost per 1000 gallons of treated wastewater) of an activated sludge secondary treatment facility and 6-8 percent of the total cost of a trickling filter secondary treatment facility.

The percentage of the cost that chlorination/dechlorination contributes to the total cost of treatment is substantially less for tertiary treatment facilities than for secondary treatment facilities. For example, Table 6 indicates that chlorination and dechlorination only account for 3-4 percent of the total unit treatment cost for a tertiary plant including biological oxidation, chemical coagulation with lime, filtration, and disinfection. It should be pointed out, however, that in many cases the tertiary treatment is provided to prepare the wastewater for disinfection. Hence, the need for tertiary treatment may be directly related to disinfection goals or requirements.

Figure 1

Estimated Capital Cost
Chlorine Contact Basin
[Culp et al (1980)]

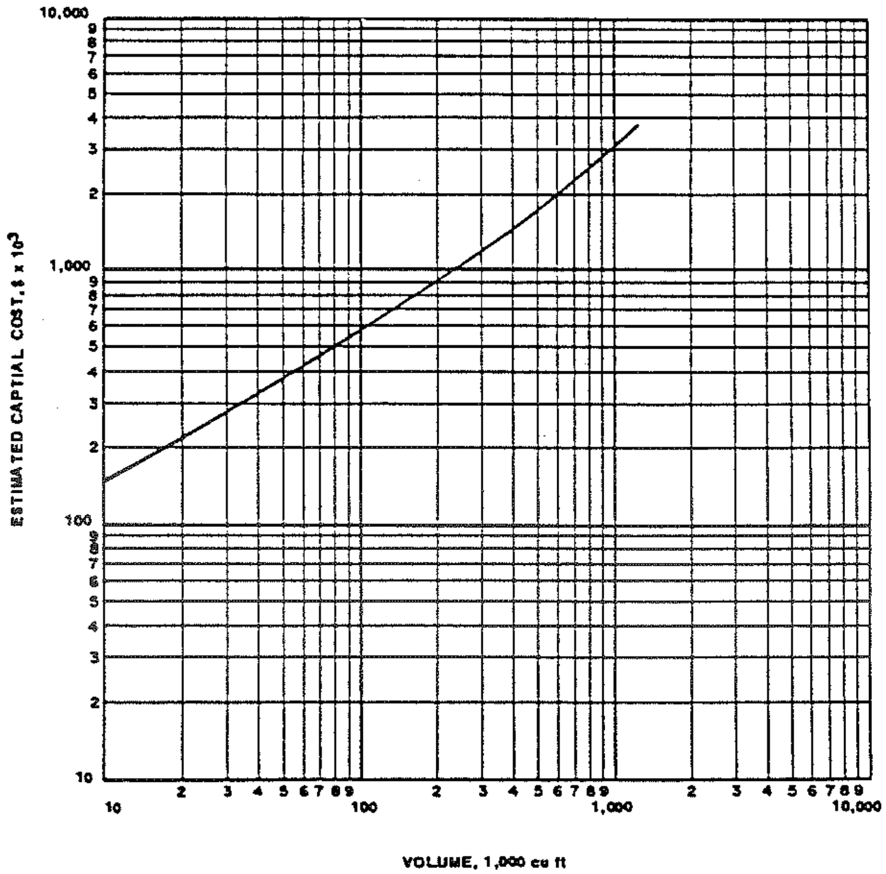


Figure 2

Estimated Capital Cost
Chlorine Feed Facilities
[Culp et al (1980)]

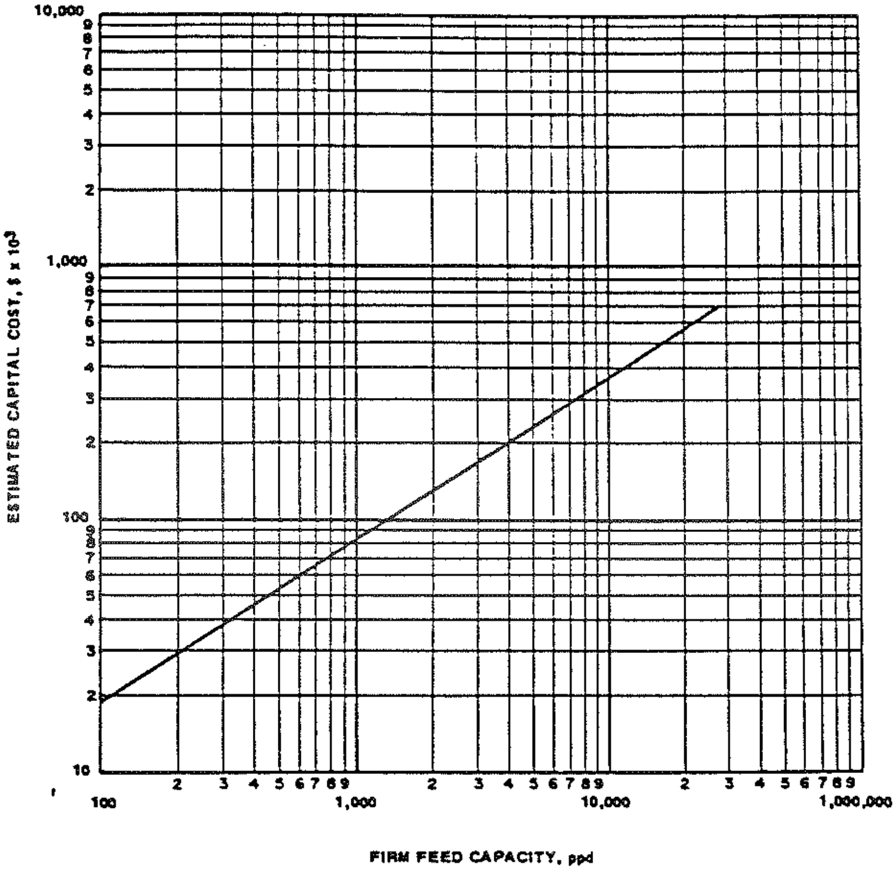


Figure 3

Operation and Maintenance Requirements
Chlorine Contact Tanks, Feed Systems, and Residual Monitoring
[Culp et al (1980)]

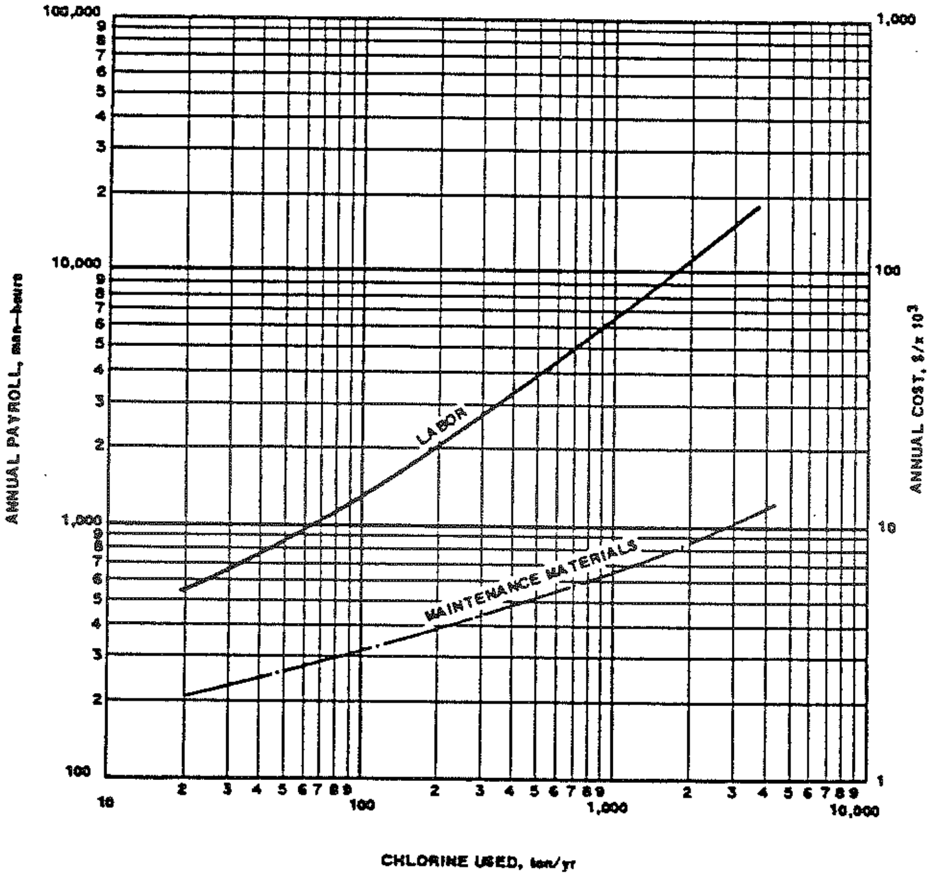


Figure 4

Annual Energy Requirements
Chlorination Systems
[Culp et al (1980)]

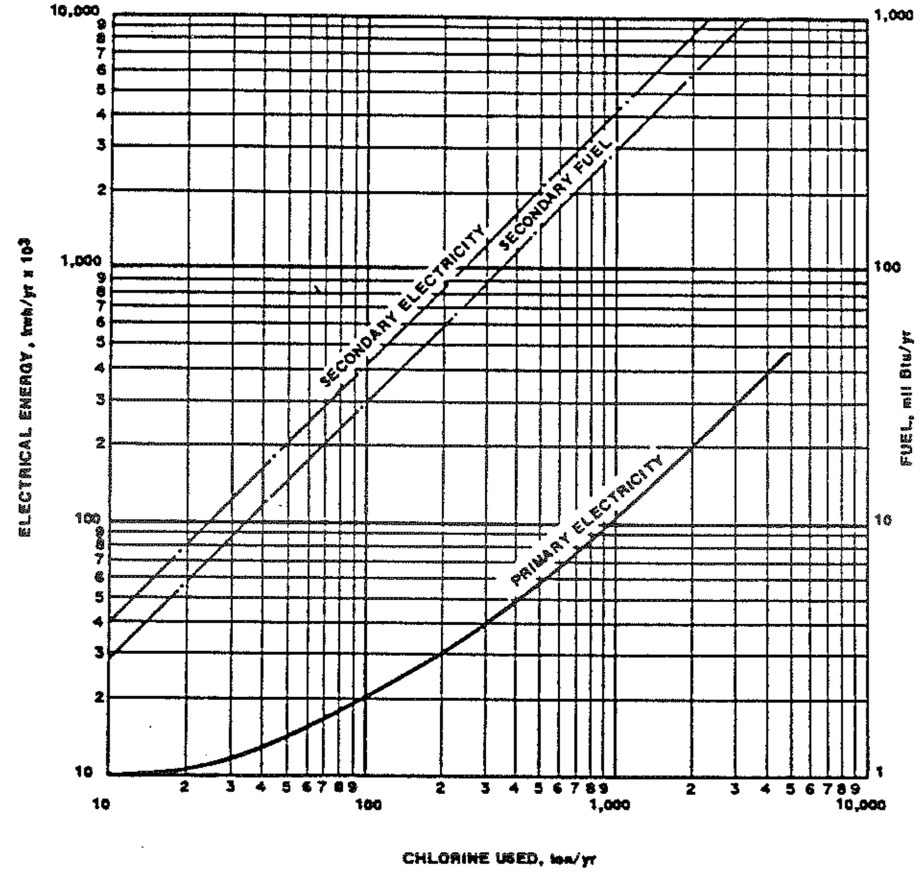


Table 6

Wastewater Treatment Process Costs
[Culp et al (1980)]

Treatment Provided	Unit Cost (cents/1000 gal) Plant Capacity (MGD) 1	10	50
Primary	39.5	15.2	11.1
+ Activated Sludge	70.2	27.9	20.2
+ Chlorination	74.9	29.7	21.4
+ Dechlorination	77.4	30.3	21.8
Primary	39.5	15.2	11.1
+ Trickling Filters	77.8	38.0	27.4
+ Chlorination	82.5	39.7	28.6
+ Dechlorination	85.0	40.3	29.0
Primary	39.5	15.2	11.1
+ Rotating Bio-Contactor	80.3	50.5	43.6
+ Chlorination	84.9	52.2	44.9
+ Dechlorination	87.5	52.8	44.9
Primary	39.5	15.2	11.1
+ Activated Sludge	70.2	27.9	20.2
+ Filtration (Polymer Added)	104.0	36.0	26.3
+ Chlorination	108.0	37.4	27.4
+ Dechlorination	110.6	38.0	27.8
Primary	39.5	15.2	11.1
+ Activated Sludge	70.2	27.9	20.2
+ Coagulation (Lime)	107.0	45.9	35.5
+ Filtration	148.8	55.4	43.2
+ Filtration	152.8	56.8	44.3
+ Dechlorination	155.3	57.4	44.7

Note: Unit costs include total O&M costs and capital costs amortized at 7 percent for 20 years.

As previously stated, the effect of increasing the chlorine and sulfur dioxide dosages has little effect on overall treatment costs for smaller facilities but has a more significant impact on large facilities. As an example, if both the chlorine and sulfur dioxide dosages were doubled for the data from Culp et al (1980), i.e., to 12 mg/l of chlorine and 6 mg/l of sulfur dioxide, the cost of treated effluent would increase by 0.8 cents/1000 gallons. This would increase the annual chemical costs by $3100 for a 1 MGD facility and $155,000 for a 50 MGD facility.

The resultant disinfection costs (chlorination and dechlorination) would then be 10 percent and 11 percent of the unit cost of treating the wastewater for 1 and 50 MGD activated sludge secondary treatment plants, respectively.

Disinfection costs can be reduced by increasing the chlorine contact time. Several disinfection models have been developed that show that microorganism destruction is strongly related to the product of contact time and chlorine residual. Therefore, increasing the chlorine contact time will result in a reduction in the chlorine dosage to meet any given coliform limit. The County Sanitation Districts of Los Angeles County (1977) presented a chlorination model based on data from three wastewater reclamation plants using secondary effluent. The model is shown in Figure 5, where: MPN is the chlorinated effluent MPN/100 ml, and MPN^{O} is the unchlorinated effluent MPN/100 ml.

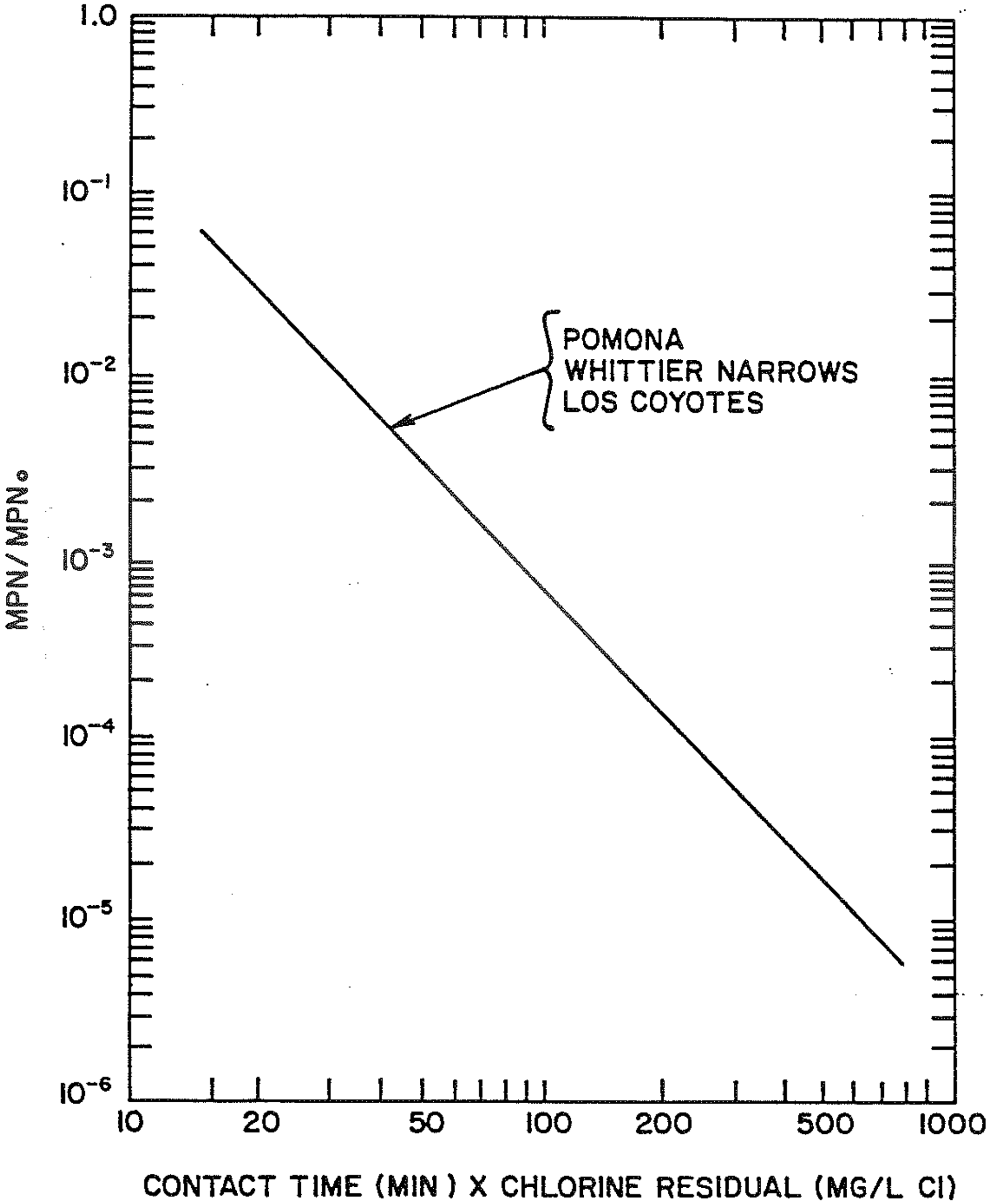
A similar model was developed during the Pomona Virus Study for disinfecting tertiary effluent. It was then mathematically determined that, for the Pomona WRP effluent, the most cost-effective chlorine contact time to meet a coliform limit of 2.2/100 ml was 2.8 hours. The optimum contact time is dependent on many factors, including the chemical and physical characteristics of the wastewater, construction costs, chemical costs, and the coliform level to be achieved in the final effluent. However, it is likely that, for minimizing the disinfection costs per unit of wastewater treated, the additional capital costs required to construct larger chlorine contact basins would be more than offset by the reduction in chlorine (and sulfur dioxide) costs.

SAFETY HAZARDS OF CHLORINE

While the use of chlorine to disinfect wastewater is an effective mechanism for the destruction of pathogens to accepted levels, various risks or hazards can be attributed to chlorine. The identification of risks associated with chlorine can be grouped according to production, transportation and handling, and use. Very little information is available regarding injuries to workers at plants producing chlorine. The amount of chlorine used as a wastewater disinfectant has very little effect on the total quantity of chlorine produced by industry. Therefore, if chlorine were totally eliminated as a wastewater disinfectant, the decrease in risk associated with the overall production of chlorine would appear to be minimal. Hence, the production hazards associated with chlorine used for wastewater disinfection are neglected in this document.

Figure 5

Chlorination Model
[County Sanitation Districts of Los Angeles County (1977)]



Health Hazards

The principal risks associated with the handling of chlorine are human exposure to liquid chlorine and occupational and public exposure to gaseous chlorine. Exposure to liquid chlorine is possible for the occupational workforce and can result in severe skin or eye burns. However, the most common exposure is to gaseous chlorine, which is the normal chlorine state at atmospheric pressure and normal temperatures.

Chlorine exists as an element only at very low pH (<2), and at the higher pH found in living tissue it is rapidly converted to hypochlorous acid. In this form, apparently, it can penetrate the cell and form N-chloroderivatives that damage cell integrity [Patton et al (1972)]. According to microbial test systems, chlorine can also disrupt cell wall permeability, which possibly explains its' ability to cause oedema and acute tissue injury.

The most important human exposure routes to gaseous chlorine are inhalation and eye and skin contact. Subjective complaints of odor and irritation of the eyes and upper respiratory tract are associated with short-term, low level exposures to gaseous chlorine. Hubly et al (1985) reported that the odor threshold of chlorine is approximately 0.2 ppm, while the World Health Organization (1982) reported that the odor threshold perception ranges from 0.02 to 2.0 ppm. The irritation threshold also ranges from 0.02 to 2.0 ppm. At and above concentrations of 1-2 ppm, irritation becomes a problem, and above 4 ppm it becomes intolerable [World Health Organization (1982)]. A National Academy of Sciences (1976) committee on medical and biological effects of environmental pollutants reported that there is little dose-response correlation for workers chronically exposed to chlorine below 1 ppm, while chronic exposure to 5 ppm can result in respiratory complaints, nausea, increased susceptibility to tuberculosis, and corrosion of teeth. High concentrations of chlorine irritate the skin, producing burning, stinging, inflammation, achrodermatosis, shrivelling, blistering, and development of nodules.

Acute exposure to chlorine presents both acute and latent effects. An exposure of 7 ppm for one hour can result in immediate throat and mucous membrane irritation. Hubly et al (1985) reported that higher concentrations lead to cough, conjunctivitis, pulmonary edema, and death. Exposure to 100 ppm is generally considered to be lethal in only a few seconds, which represents a significant risk to humans exposed to an accidental release of gaseous chlorine. Others have estimated that lethal doses may be higher than 100 ppm, however, and may require longer inhalation periods. For example, Gerchik (1939) noted in a review of the data that 1000 ppm of chlorine causes death in 5 minutes. Those exposed during physical exertion appear to be especially vulnerable. The latent effects of gaseous chlorine are less pronounced and often difficult to diagnose but may include broncospasm, especially in asthmatic people, and difficult or painful breathing [National Academy of Sciences (1976)].

At low concentrations, the acute effects of chlorine exposure are confined to the perception of a pungent odor and mild irritation of the eyes and upper respiratory tract. These symptoms are resolved shortly after exposure is stopped. Subjective reaction is variable and adaption has been reported with a resultant loss or diminution in the sensations of smell and irritation. As the chlorine concentration increases, symptoms become more severe and involve more distal portions of the respiratory tract. In addition to immediate irritation and associated pharoxysmal cough, victims manifest anxiety. At higher levels, there is dyspnoea, cyanosis, vomiting, headache, and a heightening of anxiety, especially in those prone to neurosis. Generally, with palliative treatment, the patient recovers within 2 days to 2 weeks. In more severe cases, complications, such as pneumonia, should be anticipated [World Health Organization (1982)].

There have been few fatalities following chlorine exposure. However, at significantly high concentrations, the chemical can cause shock, coma, respiratory arrest, and death. Unusual patterns in general mortality have not been reported, nor has chlorine been shown to induce mutagenic, carcinogenic, or teratogenic effects in human beings.

The occupational exposure limits for chlorine in air or work places vary in different countries from 0.344 to 1.032 ppm as time-weighted averages, and from 0.344 to 2.99 ppm as short-term exposure limits. Many years of engineering experience have reduced the potential for worker exposure to a minimum; however, occasional equipment failure does occur. Exposure is minimized through training and the use of respirators and other protective clothing.

Transportation Hazards

Determination of risks associated with the transportation and handling of chlorine is difficult to predict because of the variety of transportation modes available, the range in sizes of shipping containers, and most importantly, the lack of available data. Chlorine is transported as solid calcium hypochlorite, liquid sodium hypochlorite, or a compressed gas (liquid). However, almost all of the chlorine used for wastewater disinfection is transported as a compressed gas. The Chlorine Institute (1980) stated that, for economic reasons, the percentage of total chlorine shipped for wastewater disinfection in the form of hypochlorite is negligible.

Department of Transportation data for the years 1971 through 1980 indicate that there was only one reported commercial transportation accident involving chlorine that involved any fatalities. A 1979 major rail accident in Youngstown, Ohio resulted in 160 injuries and 8 deaths. The following tabulation illustrates the accident data involving transportation of chlorine for the years 1971 through 1980:

	Accidents	Deaths	Injuries
Railroad	72	8	247
Railroad (Excluding Youngstown)	71	0	87
Truck			
Cylinders to 114 kg	14	0	60
0.91 Ton Containers	4	0	15
Tanker Trucks	2	0	71
Barge	2	0	3

Hubly et al (1985) also reported that data obtained from the U.S. Bureau of Census showed that, in 1972, almost 85 percent of the chlorine was transported by rail in bulk containers, while 15 percent was transported by truck and less than 1 percent by water. Based on this information and other information on the breakdown of chlorine shipments by size and type of container, the following table was developed for the accident rates per metric ton-km:

	Deaths	Injuries	Chlorine Released
Railroad	4.3×10^{-10}	6.0×10^{-5}	3.3×10^{-5}
Railroad (Excluding Youngstown)	0	4.7×10^{-9}	2.3×10^{-5}
Truck:			
Cylinders to 114 kg	0	2.7×10^{-6}	1.2×10^{-4}
0.91 Metric Containers	0	1.4×10^{-8}	1.0×10^{-6}
Tank Truck	0	3.2×10^{-8}	4.6×10^{-8}
Barge	0	4.7×10^{-8}	N/A

The above table indicates that the accident rates for truck-transported cylinders are consistently higher than for the other categories listed. This is probably due to the fact that a greater number of cylinders are needed to carry a given amount of chlorine. The deletion of the Youngstown accident from the railroad totals does not greatly alter the relative ordering of the accident rates in the above table.

On-Site Hazards

Hubly et al (1985) stated that wastewater treatment plant on-site accident information is virtually impossible to obtain. However, a 1979 summary of AWWA data on accidents at water treatment plants indicated that 4 percent of the injuries were due to contact with radiations, caustics, toxic, and noxious substances. Exposure to chlorine would fall into this category.

The Colorado Division of Labor categorized accident types for disabling injuries from a statewide industrial base [Hubly et al (1985)]. Contact with radiation or caustics resulted in 3 percent of the accidents in 1977. In comparing the AWWA data to that for the State of Colorado, it appears that wastewater treatment plant accidents are very similar to those of all injuries. The accidents caused by exposure to chemicals, e.g., chlorine, is not significantly higher in water treatment plants and, by inference, wastewater treatment plants, than in the general industry. If the assumptions are made that: (a) the AWWA category "contact with radiations, caustics, toxic, and noxious substances" is almost entirely due to exposure to chlorine; and (b) the accident rate at sewage treatment plants is similar to that at water treatment plants, then a conservative estimate would indicate that 4 percent of accidents at sewage treatment plants are caused by chlorine.

Example

Using the preceding information, the relative risks of chlorine transportation and use at sewage treatment plants can be calculated. As an example, assume a sewage treatment plant treats 3790 cubic meters per day (1 MGD), utilizes a chlorine dosage of 6 mg/l, and has 6 full-time employees. Further, assume that the chlorine is obtained from a manufacturer located 644 kilometers (400 miles) from the treatment plant. If chlorine is transported by truck in 114 kg cylinders, the transportation risks will be as follows: deaths - 0; injuries - 0.014/yr; and the amount of chlorine released - 0.64 kg/yr. If the chlorine is transported by railroad using 0.91 metric ton containers, the calculated transportation risks are: deaths - 0; injuries - 0.000075/yr; and the amount of chlorine released - 0.17 kg/yr. As this example indicates, the truck transportation of chlorine in 114 kg cylinders has a significantly higher accident rate than rail transportation. However, relatively small shipments of chlorine using small cylinders do not pose a significant risk for human health and property damage. There is a low sensitivity for the transportation mode when dealing with small quantities of chlorine.

The same example presented above will result in 0.1 lost work days per year from chlorine exposure using the assumed accident rate of 4 percent. This calculation is based on the assumption that lost work time is approximately the same for each type of accident.

REFERENCES

Chlorine Institute (1980). "North American Chlor-Alkali Plants and Production Data Book."

County Sanitation Districts of Los Angeles County (1977). Pomona Virus Study - Final Report. California State Water Resources Control Board, Sacramento, CA.

Culp, G., et al. (1980). Wastewater Reuse and Recycling Technology. Noyes Data Corporation, Park Ridge, NJ.

Engineering-Science (1985). Fifth Annual Report of the Monterey Reclamation Study for Agriculture. Engineering-Science, Berkeley, CA.

Engineering-Science (1986). "Monterey Wastewater Reclamation Study for Agriculture - Projected Cost of Reclaimed water." Engineering-Science, Berkeley, CA.

Esvelt, L.A., Kaufman, W.J., and Selleck, R.E. (1971). Toxicity Removal from Municipal Wastewaters. SERL Report No. 71-7, Sanitary Engineering Research Laboratory, University of California, Berkeley, CA.

Gerchik, M. (1939). "Medical Experience of Americans with Chemical Poison Gas During the World War." Protor, 5(11), 173-179.

Heim, N.E., and Burris, B.E. (1979). Chemical Aids Manual for Wastewater Treatment Facilities. 430/9-79-018, Publication MO-25, U.S. Environmental Protection Agency, Washington, D.C.

Helz, G.C., and Kosak-Channing, L. (1984). "Dechlorinaion of Wastewater and Cooling Water." Environ. Sci. & Technol., 18(2), 48A-55A.

Horvath, R.W. (1986). Letter dated June 27, 1986 from Robert W. Horvath, County Sanitation Districts of Los Angeles County, to William F. Jopling, California Department of Health Services.

Hubly, D. et al (1985). "Risk Assessment of Wastewater Disinfection." EPA/600/2-85/037, Water Engineering Research Laboratory, Office of Research and Development, U.S. Environmental Protection Agency, Cincinnati, OH.

Leidholdt, R. (1982). "Breakpoint Chlorination - the Chemistry of Chlorine." Opflow, 8(10), 1-7.

National Academy of Sciences (1976). Chlorine and Hydrogen Chloride-Committee on Medical and Biologic Effects of Environmental Pollutants. National Academy of Sciences, Washington, D.C.

Patton, V. et al (1972). "Chlorination Studies. I. The Reaction of Aqueous Hypochlorous Acid With Cytosine." Biochem. Biophys. Res. Comm., 48(4), 880-884.

Sepp, E., and Bao, P. (1980). Design Optimization of the Chlorination Process. Volume I - Comparison of Optimized Pilot System with Existing Full-scale Systems. Report prepared for the Municipal Environmental Research Laboratory, Office of Water Research and Development, U.S. Environmental Protection Agency, Cincinnati, OH.

Sepp, E., and White, G.C. (1981). Manual for Wastewater Chlorination and Dechlorination Practices. Publication No. 53-A, California State Water Resources Control Board, Sacramento, CA.

Stone, R.W., Kaufman, W.J., and Horne, A.J. (1973). Long-term Effects of Toxicants and Biostimulants on the Waters of San Francisco Bay. SERL Report No. 73-1, Sanitary Engineering Research Laboratory, University of California, Berkeley, CA.

Task Force Report (1976). Disinfection of Wastewater. EPA-430/9-75-012, Publication MCD-21, U.S. Environmental Protection Agency, Washington, D.C.

White, G.C. (1978). Disinfection of Wastewater and Water for Reuse. Van Nostrand Reinhold Co., New York, NY.

World Health Organization (1982). "Chlorine and Hydrogen Chloride." Environmental Health Criteria 21, World Health Organization, Geneva.

PART II

HEALTH EFFECTS

MICROBIOLOGICAL CONTAINANTS

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FIGURES AND THEIR SOURCES

INDICATOR ORGANISMS

The literature contains criteria for the ideal indicator organism for determining the sanitary quality of water. There is general agreement that no single indicator organism meets all the criteria and different organisms or several organisms should be used in different circumstances. The criteria according to McFeters (1978) are:

- o The indicator should be applicable to all types of water.
- o It should be present in greater numbers than the pathogen.
- o It should not increase in the absence of a health hazard.
- o It should exhibit greater survival than pathogens to treatment or environmental conditions.
- o The test should be specific for the indicator organism.
- o The test should be simple, inexpensive, and rapid.
- o The indicator should be harmless.
- o The indicator should be proportional to the hazard.

A. <u>Specific Indicator Organism</u>

1. Coliform Bacteria

The coliform bacteria group is the indicator group which has seen the longest service as an indicator of fecal contamination. It is now being replaced or augmented by other indicator groups, except for treated drinking waters. The coliform bacteria group is defined as species of gram-negative rods which may ferment lactose with gas production (or produce a distinctive colony within 24 to 48 hours incubation on a suitable medium) at $35^{o}C$. There are strains which do not conform to the definition.

The total coliform group includes four genera in the Enterobacteriaceae family. These are Escherichia, Klebisella, Citrobacter, and Enterobacter. The Escherichia genus (E. coli species) appears to be most representative of fecal contamination of the group.

The test which defines the organisms in the group does not screen out organisms which may originate or multiply in soil and industrial wastes; consequently, the total coliforms generally have been in disfavor as a fecal indicator in the environment because of the recognized nonfecal sources of many strains.

2. Fecal Coliform Bacteria

For a number of years (commencing near the turn of the century) there has been a search for an indicator group which would be more selective than total coliforms for fecal contamination. A great deal of the developmental work was done in association with bacteriological standards for shellfish growing waters.

A fecal coliform bacteria group was established based on the ability to produce gas (or colonies) at an elevated incubation temperature ($44.5 \pm 0.2^{o}c$).

There is general agreement that the fecal coliform group reflects the sanitary quality of a surface water better than the total coliform group. According to Standard Methods (1985). "the membrane filter procedure gives 93 percent accuracy in differentiating the coliforms from warm-blooded animals and the nonfecal coliforms from other sources."

Recent work (Dufour et al. 1976: Caplenas. N. R. et al.. 1981) has indicated Klebisella originating from carbohydrate-rich industrial waste can show up positive as fecal coliform: consequently. there must be caution in depending on the sanitary significance of the fecal coliform group.

The fecal coliform bacteria appear to be more fragile in the environment and in response to disinfection than the total coliform group.

3. Klebisella

The total coliform population includes the genera Klebisella. The thermotolerant Klebiselia are also included in the fecal coliform group. Klebisella have been found in sources other than fecal pollution such as redwood tanks. vegetation. and textile plant wastes. It is a pathogen of the respiratory system. urinary tract. and other sites. There does not appear to be much to recommend it as an indicator of fecal contamination. and its characteristics tend to detract from the use of total and fecal coliform group of which it is a member.

4. E. Coli

The E. coli is one of the coliform bacteria population and is more representative of fecal sources than the other coliform genus.

There is little information on the relative occurrence and survival of pathogenic strains of E. coli (as compared to nonpathogenic) in the environment. It does not survive as well as enterococci in marine water.

The U. S. EPA has found a correlation of E. coli with gastrointestinal disease in fresh waters used for recreation and has proposed a standard using E. coli (or enterococci) as the indicator organisms.

5. Fecal Streptococci

This group had been used in conjunction with fecal coliforms to determine the source of recent fecal contamination (man or farm animals). Surface waters having a fecal coliform to fecal streptococci ratio of greater than four were suspected of receiving predominately human sewage, while waters with ratios of less than 0.7 were considered contaminated by wild or domestic animals. However, the animal strains of fecal streptococci tend to die away more rapidly, and the ability to differentiate sources is lost rather rapidly. The organisms are less prone to regrowth and are enumerated by a single-step membrane filter procedure, which is a benefit. Several strains appear to be ubiquitous and cannot be distinguished from the true fecal streptococci under usual analytical procedures, which detract from their use as an indicator organism.

6. Enterococci

Two strains of fecal streptococci -- S. faecalis and S. faecium -- are the most human specific components of the fecal streptococcus group. By eliminating the other strains through the analytical procedures, the two strains known as enterococci can be isolated and enumerated. Even these types have been found in the fecal wastes of lower animals, and some S. faecalis biotypes have been associated with vegetation. The enterococci are reasonably specific for fecal contamination.

The enterococci are generally found in lower numbers than other indicator organisms; however, they exhibit better survival in seawater.

7. Clostridium Perfringens

This is a spore-forming anaerobic persistent bacteria, and the characteristics make it a desirable indicator where disinfection is employed, where pollution may have occurred in the past, or where the interval before analysis is protracted.

It has found some use in tracing pollution and determining the sanitary quality of underwater sands and muds. Type A from human feces may also grow in the soil.

8. P. aeruginosa and A. hydrophila

These organisms may be present in sewage in large numbers. Both can be considered aquatic organisms and can be recovered in water in the absence of immediate sources of fecal pollution. One is the cause of "swimmer's ear" and folliculitis and the latter is implicated in wound infections and is a gastrointestinal pathogen. They would be most useful as a supplemental indicator of recreational water quality. They may multiply in the aquatic environment.

9. Standard Plate Count

The standard plate count is generally used to demonstrate the effectiveness of sanitary practices, i.e., bottled water and restaurant sanitation. It is used to identify the quality of shellfish meat (a $35^{\circ}C$ plate count shall not exceed 500,000 per gram). It does not differentiate fecal contamination from other forms and has seen little use in water quality investigations.

B. Uses of Indicator Organisms

The current uses of indicator organisms are summarized in Table 1 and are discussed below.

1. Disinfection Effectiveness

The objective of sewage effluent disinfection is the reduction or elimination of pathogenic agents prior to the release of wastewater into the environment. Two indicator groups have been used to demonstrate disinfection effectiveness: total coliform bacteria and fecal coliform bacteria.

In 1973, the U. S. EPA included a disinfection standard in the definition of secondary treatment which was to be achieved by all publicly-owned treatment works. The minimum levels of disinfection were expressed in terms of fecal coliform bacteria:

a. Geometric mean in a 30-day period shall not exceed 200/100 ml: and

b. Geometric mean in a 7-day period shall not exceed 400/100 ml.

Many states followed the EPA lead and adopted all or part of the federal disinfection standard (see Table 2).

Although Table 2 indicates that the federal standard has been applied by many states, a later report (EPA 1986) indicated that many states have several disinfection requirements, which differ from those shown, or vary, depending on the situation. Maryland requires disinfection to a total coliform bacteria level of 3 MPN/100 ml, unless a special exemption is granted. Over 45 states have multi-level standards for disinfection relative to the discharge stream water quality criteria, 15 states require effluent disinfection levels of 14 fecal coliforms per 100 ml for discharge into shellfish waters, and 9 states have more stringent disinfection standards than those imposed on shellfish water discharges.

In 1975, the EPA eliminated the disinfection standard from the definition of secondary treatment (Federal Register, August 15, 1975). The following are reasons given for elimination of the requirement:

a. Requiring a minimum level of disinfection for all sewage effluent discharges is of questionable benefit when human contact is remote.

b. Residual chlorine may be harmful to aquatic life.

c. Chlorination may form carcinogenic compounds in secondary effluents.

d. Production of chlorine used for disinfection may be 7 to 15 percent of the on-site energy demands of a secondary treatment plant.

(These issues are covered in other portions of the report.)

The points stated in support of the use of fecal coliforms as a measure of disinfection effectiveness are as follows:

a. Fecal coliforms are a better indicator of fecal pollution than the total coliform group.

b. Inasmuch as the quality of surface waters in Basin Plan criteria (for water quality protection) is in terms of fecal coliform concentrations, water discharges to these waters should also be in terms of fecal coliform bacteria.

c. Effluent disinfection requirements in other states employ fecal coliform bacteria.

The total coliform bacteria group has been recommended for use as an indicator of disinfection effectiveness by DHS in California for several reasons:

a. Total coliform bacteria are present in undisinfected municipal wastewater in greater numbers than fecal coliform bacteria or other indicators.

b. ~~The total coliform group is generally more resistant~~ to disinfection by chlorine and provides a more conservative measure of disinfection.

The second point may be particularly significant inasmuch as some pathogens, viruses in particular, have been demonstrated to be equal or more resistant to disinfection than indicator organisms.

A carefully conducted disinfection study by the University of California documented the greater resistance of coliforms than fecal coliforms, and concluded that total coliform bacteria provided a more conservative measure of disinfection efficiency (Collins 1972). Others have supported the position that the total coliform offers a margin of safety not associated with the fecal coliform group (Geldrich 1978).

There is little information on the relative inactivation of pathogenic bacteria and coliform bacteria in wastewater; however, there have been instances where waterborne salmonella have been present in the absence of total coliforms in water supplies suggesting that their resistance to inactivation may be at least equivalent to coliforms (McFeters 1978). A study in Long Island demonstrated that significant numbers of viruses (315.5 plaque-forming units (PFU)/gal.) were isolated in disinfected wastewater at a fecal coliform level of 230/100 ml (Vaughn 1978). Also, virus concentrations of 100 PFU/gal. were detected at a fecal coliform level of 11/100 ml and a total coliform level of 49/100 ml.

These results suggest that the more resistant indicator organisms is appropriate to measure disinfection effectiveness.

2. Freshwater Quality Bacteriological Indicator

DHS conducted bacteriological studies of river systems and other surface waters under an agreement with the State Water Resources Control Board during the 1960s. These studies included investigation of both coliform and fecal coliform bacteria and were carried out in order to provide information on which to base water quality objectives for these waters. The water studies included the Smith, Klamath, Russian, Eel, Truckee, and Colorado Rivers, and Lake Tahoe and its tributaries.

In the reports of these studies, the Department recommended water quality objectives in terms of fecal coliform levels for specific locations or zones of the rivers generally based on the existing high water quality levels, thereby conforming to the Board's "nondegradation policy". These objectives were subsequently incorporated into the appropriate "Basin Plans" of the State.

The Department recognized from these investigations that fecal coliform bacteria, used as a bacteriological indicator of surface water quality, was less affected by drainage, runoff, and other relatively nonfecal discharges. The fecal coliforms more accurately reflected the sanitary quality of the surface waters than did the total coliform group.

In May 1968, the Department sponsored and participated in a symposium on fecal coliform bacteria in water and wastewater (DHS 1968) in which experiences with fecal coliform bacteria were presented by the staff of the Department and other agencies. It was concluded by the Department that: "The finding that fecal coliform levels were relatively less influenced by many types of agricultural drainage, and the unknown causes of variation in the (total) coliform density would support the assertion that fecal coliforms are a better indicator organism than coliform bacteria in fresh surface waters; however, aftergrowth of fecal coliforms in freshwater may lessen their effectiveness." Also, "The fecal coliform group would be an effective indicator of the sanitary quality of saline waters. Survival characteristics were similar to those of the total coliform group, and the fecal coliform group more accurately indicated the sanitary quality of receiving waters during periods of storm runoff."

Aside from the objectives based on a "nondegradation policy", the Department has recommended a median fecal coliform level of 50/100 ml for coastal and mountain streams, and 100/100 ml for Central Valley rivers and the Delta based on existing water quality.

3. Marine Water Bacteriological Indicator -- Recreation

Coliform bacteria requirements for ocean water contact sports areas in California have been in use since the early 1940s. The latest regulations containing the coliform standard were adopted in 1958. Both the coliform standard and a fecal coliform standard are included for body-contact recreational waters in the Water Quality Control Plan for California Ocean Waters.

The recent EPA studies for marine recreational waters indicate that there is no clear relationship between gastroenteritis illness rates and fecal coliform concentrations (within typical standards). Enterococci have been proposed as the indicator organisms of choice; however, most of the studies of the recreation waters were conducted in the south and east. There is a need to determine the suitability of an enterococci standard for California marine waters.

4. Indicator Organisms for Shellfish Growing Waters

Originally, total coliform bacteria were used to assist in the evaluation of shellfish growing waters. A series of studies were carried out in the major commercial shellfish growing states and Canada to determine the relative responsiveness of fecal coliforms and total coliforms to conditions of pollution. It was concluded that the fecal coliform was a superior microbiological indicator of fecal pollution in estuarine waters, and now, alternative standards employing either of the two indicators may be used to identify approved growing waters.

In summary, several indicator organisms have been used, or are under study for use, for the several water uses. Coliform bacteria are used to indicate the sanitary quality of a drinking water after treatment and serves as a sensitive indicator of treatment effectiveness and possible reinfection of the water after treatment. Both coliform bacteria and fecal coliform bacteria are used to determine the suitability of shellfish growing waters. Fecal coliform bacteria are used for freshwater recreational waters in most states.

California has both a total coliform and a fecal coliform standard for ocean water sports areas. The EPA has recently proposed an *E. coli*, enterococci standard for freshwater recreation and an enterococci standard for marine waters. California currently uses total coliform bacteria as a measure of sewage disinfection effectiveness.

TABLE 1

Indicator Organisms Used in Criteria
For Water Uses

Drinking Water	Total Coliform
Freshwater Recreation	Fecal coliform E. coli Enterococci
Saltwater Recreation	Fecal coliform Total coliform Enterococci
Shellfish Growing Waters	Total coliform Fecal coliform
Agricultural Irrigation	Total coliform (for reclaimed water)
Wastewater Effluent Disinfection	Total coliform Fecal coliform

Table 2

Monthly Average Discharge Coliform Limits by State

State	Total Median	Fecal Median	Fecal Maximum	
ALABAMA		200		
ALASKA a		20 b		a. Includes discharge to drinking water supplies. b. Limit for outside of mixing zone/zone of dilution. No end of pipe limits. Definition of mixing zone varies in size.
ARIZONA a		200 b	800 c	a. As discharge to a domestic water source limit is 1,000/100 ml fecal for 30-day period with a single sample maximum of 4,000. b. Full body contact - recreation. c. Single sample - full body contact - recreation.
ARKANSAS		200 a		a. For discharge into Class A streams. 1,000 fecal/100 ml for discharge into Class B streams.
CALIFORNIA	23			
COLORADO x		200 bc		b. Recreational water. c. As discharge to a domestic water supply limit is 2,000/100 ml fecal for 30-day period.
CONNECTICUT		200 A	300 b	a. Uniform across state. b. Weekly average.
DELAWARE	1,000 a	200		a. For discharge into shellfish waters.
FLORIDA		200 a		a. Uniform across state.
GEORGIA		200	400	
HAWAII x		200 b		b. Recreational waters.
IDAHO		200 a		a. Weekly average.
ILLINOIS		200 a		a. General use streams.
INDIANA a		200	400	a. No requirement in winter.
IOWA x		200		
KANSAS x		200 a		a. Full body contact - recreation.
KENTUCKY		200 a	400 ab	a. Uniform across state. b. 10% of month maximum.
LOUISIANA x	200			
MAINE		200 a		a. Discharges are the same as the maximum of the receiving water.
MARYLAND		200		
MASSACHUSETTS x	1,000 a	200 a		a. Uniform across state.

Table 2 (Continued)

State	Total Median	Fecal Median	Fecal Maximum	
MICHIGAN		200 a	400 b	a. Recreational waters. b. 10% of month maximum.
MINNESOTA		200 ab	400 c	a. No requirement in winter. b. Uniform across state. c. 10% of month maximum.
MISSISSIPPI		200	400	
MISSOURI		200 ab		a. Full body contact - recreation. c. Maximum from 4/1 to 10/31 only.
MONTANA xa		200 b	400 c	a. As source of DWS: fecal <=50/ 100 ml. b. This is a water quality standard, not a waste discharge criteria. c. 10% of month maximum.
NEBRASKA		200 a	400	a. Class A streams.
NEVADA		200 a		a. Uniform across state.
NEW HAMPSHIRE x	240 a			a. Full body contact - recreation.
NEW JERSEY		200		
NEW MEXICO			500	
NEW YORK x		200		
NORTH CAROLINA		200		
NORTH DAKOTA		200	400 a	a. Weekly average.
OHIO x		1,000 a	2,000 b	a. No requirement in winter. b. 10% of month maximum.
OKLAHOMA x		200 a	400	a. This is the drinking water limit as well.
OREGON		200 a		a. Full body contact - recreation.
PENNSYLVANIA x		200 a		a. Uniform across state.
RHODE ISLAND		200 a	400	a. Uniform across state.
SOUTH CAROLINA		200	400	
SOUTH DAKOTA xa		200 b	400 c	a. Drinking water standard is 5,000/100 ml total coliforms (30-day average); 20,000/100 ml total coliforms for a single sample. b. This is a water quality standard, not a waste discharge criteria. c. Maximum for period of 5/1 to 9/30.
TENNESSEE x		200		
TEXAS x		200		
UTAH	2,000	200		
VERMONT	500 a	200 a		a. Uniform across state.
VIRGINIA		200		
WASHINGTON		200	400 a	a. Weekly average.

Table 2 (Continued)

State	Total Median	Fecal Median	Fecal Maximum	
WEST VIRGINIA x		200	400 a	a. Single sample - full body contact - recreation.
WISCONSIN		100 a		a. No coliform count required; requires 0.5 mg/l chlorine in discharge, if count is less than 100 to 200/100 ml; state alerted to check chlorine facilities.
WYOMING x		200	400 a	a. 10% of month maximum.

. Sometimes value is for a geometric mean. For a log normal distribution, the median and geometric mean are the same value.

x Given value is receiving-water standards for recreational waters. Waste discharge limit not to exceed standard if standard not being met. Otherwise, no limit on waste discharge.

HEALTH EFFECTS INFORMATION

Several literature reviews have searched out and presented available technical information and study results on indicator organisms and pathogenic agents in infected and disinfected wastewater and receiving waters and the health risks associated with various water uses. The coverage here is intended to summarize significant portions of the information in these reviews and other more specific information sources on infectious agents and indicator organisms. It is not intended to be all inclusive. Much more complete coverage of pathogenic agents and associated waterborne disease is presented in voluminous and detailed works such as Feachem et al., 1980 and 1983.

A. Untreated Domestic Wastewater

The presence and densities of pathogens and indicator organisms in untreated sewage is highly variable and is subject to a number of complex influences such as seasonal variations, industrial wastes, vaccination programs, and community disease patterns. Engelbrecht (1980), with these and other expressed reservations, presented. for illustrative purposes, the following range of densities of infectious agents in untreated municipal wastewater:

- Bacteria -- up to 10^4/l (for salmonella)
- Protozoa (cysts) -- up to 10^5/l
- Helminth (ova) -- up to 10^3/l
- Enteric Virus (PFU) -- up to 10^5/l

The diseases associated with the types of organisms within these four major groups is presented in Table 3. Newly recognized pathogenic bacteria are given in Table 3a. The coverage here is not total. Other viruses (Coronaviruses, Astroviruses and "small round" viruses) also may be waterborne. There is very little information on the presence or densities of the more recently identified or suspected waterborne pathogens in wastewater or surface waters; consequently, little coverage can be given these microorganisms in the report. Some, such as Campylobacter, Yersinia, and Norwalk-type viruses are considered to be significant agents of waterborne disease.

1. Enterovirus

It has been calculated that the <u>average</u> enteric virus density in domestic sewage in the United States is probably about 500 virus units per 100 ml (ASCE 1970). Based on the data presented by Cooper (1984), the estimate may be high although some studies have detected over 1,000 PFU/100 ml. It has also been stated by Engelbrecht (1980) that the observed number of enteric viruses may be one or two logs lower than the actual density due to the limitations of recovery and cultivation procedures. See Table 4 for reported viral densities and Table 5 for treatment removals.

It has been stated that, in developing countries, sewage must be assumed to contain 10^5 enterovirus per liter (Feachem 1983). In individual studies, the range of enterovirus in sewage is highly variable.

- In Haifa, Israel, sewage, from 1972 to 1974 contained a monthly average between 6×10^3 and 4.9×10^5 viruses per liter (Buras 1976).

- An early study in Michigan, 1955 to 1957, displayed the lack of sensitivity of early methods. In that study, enteroviruses were detected in only 1 percent and 14 percent of samples of Lansing and East Lansing sewage (Bloom 1959).

- In 1978, viruses were detected in all untreated sewage samples in Hawaii at concentrations ranging from 27 to 1.9×10^4 per liter (Fujioka 1978).

- Seattle sewage and storm water contained up to 1.3×10^3 viruses per liter (Heyward 1979). Maximum concentrations are found in late summer and fall, and viruses can survive for 100 days or more in sewage.

The variations may be due in part to the state-of-the art of laboratory techniques at the time the studies were done, and to the extent of health protection development in the country or area examined. Cooper (1984) found that the concentrations reported depended on the number of viruses tested. Where many types were tested, the concentration ranged from 200 to 1,600 per liter.

Table 3

Infectious Agents Potentially Present in
Raw Domestic Wastewater

Organism	Disease
1. Bacteria	
Shigella (4 spp.)	Shigellosis
Salmonella typhi	Typhoid Fever
Salmonella (~1700 spp.)	Salmonellosis
Vibrio cholerae	Cholera
Escherichia coli (Enteropathogenic)	Gastroenteritis
Yersinia enterolitica	Yersinosis
Leptospira (spp.)	Leptospirosis
2. Viruses	
Enteroviruses (67 types)	Gastroenteritis, heart anomalies, meningitis
Hepatitis A virus	Infectious hepatitis
Adenovirus (31 types)	Respiratory disease
Rotavirus	Gastroenteritis
Parvovirus (2 types)	Gastroenteritis
Norwalk Agent	Gastroenteritis (vomiting)
3. Protozoa	
Entamoeba histolytica	Amebiasis (Amoebic Dysentery)
Giardia lamblia	Giardiasis
Balantidium coli	Balantidiasis (Balantidial Dysentery)
4. Helminths	
Ascaris lumbricoides	Ascariasis
Ancylostoma duodenale	Ancylostomiasis
Necator americanus	Necatoriasis
~~Ancylostoma (spp.)~~	~~Hookworm~~
Strongyloides stercoralis	Strongyloidiasis
Trichuris trichiura	Trichuriasis
Taenia (spp.)	Taeniasis
Enterobius vermicularis	Enterobiasis
Echinococcus granulosis	Hydatidosis

Table 3a

Newly Recognized Pathogens

ORGANISM	SPECIES	COMMENTS	REFERENCE
Bacteria			
Campylobacter	jejuni, coli, fetus	Very common cause of gastroenteritis	Karmali 1985
Escherichia	coli	ETEC, EPEC, EIEC	Evens 1983
Salmonella	non-typhoidal		Karmali 1985
Shigella	sonnei, flexneri		Karmali 1985
Aeromonas	hydrophila, sobria, caviae	Assocciated with untreated water consumption	Holmberg 1986
Plesiomonas	shigelloides	Travel, shellfish consumption	Holmberg 1986a
Yersinia	enterocolitica		Karmali 1985
Vibrio	non-01	Often bacteremia in immunocompromised	Hughes 1978
	vulnificus	Now gastroenteritis in addition to bacteremia	Johnson 1985 Johnson 1986
	parahaemolyticus		Karmali 1985
	hollisae	Shellfish, catfish consumption	Morris 1982 Lowry 1986
	damsela	Primarily wound infections	Morris 1982
	mimicus	Resemble V. cholerae	Shadera 1983
	fluvialis		Hickman-Brenner 1984

Table 4

Reported Measurement of Viral Densities in Raw Wastewater Compiled by Englebrecht

PFU/L
7,000
5-11,000
30-110
Up to 100,000
100,000
50,000-100,000
6,000-492,000
Geometric mean = 5,650

Table 5

Viral Removal in Wastewater Treatment Processes

Process	Removal
Grit chambers	0-50%
Primary sedimentation	35%
	0-?%
Trickling filters	83-95%
	0-85%
Activated sludge	80%
	75-99%
Stabilization ponds	0-96%
	99%

2. Salmonella

Reported concentrations of salmonella in sewage are generally highly variable, and not much quantitative information is available. In Baltimore, sewage contained 500/100 ml, and in Houston, 8,000/100 ml (Feachem 1983). Salmonella at San Diego was very low (0.4/100 ml).

Using the Kerr and Butterfield equation, a salmonellosis morbidity rate of 18 per 100,000 and assuming that only 5 percent of the disease is reported, Cooper (1984) estimated the number of salmonella in wastewater would be 44.9 per million coliform or approximately 450/100 ml. He also found that the available data reported from 3-1,100/100 ml in wastewater and "effluent".

Based on the work of McCoy (1964) and Strobel (1968), Mechalas et al. (1972), estimated a salmonella to coliform ratio of 1:10,000.

3. Shigella

Based on an estimate of 0.2 to 4 percent of the populations excreting shigella, sewage is estimated to contain between 10 and 10^4 shigella per liter (Feachem 1983).

Using the relationship developed by Kerr and Butterfield, a shigella morbidity rate of 8 per 100,000, and assuming that 5 percent of the actual illnesses are reported, Cooper (1984) estimated the number of shigella per million coliform would be 30.9, approximately 300/100 ml.

It has been pointed out in an unreferenced statement (Engelbrecht 1980) that there are few reports of shigella bacteria being detected in wastewater. Feachem (1983) references several studies where shigella were not detected in the community wastewater even where the community was known to be infected. Inasmuch as controlled experiments indicate shigella survive longer than some other pathogenic bacteria (Table 20), negative findings may be due to low test sensitivity and the difficulty to detect and enumerate the organisms.

4. Helminth Ova

Denver sewage contained 5 to 110 ascaris eggs per liter (Wang and Dunlop 1954). Generally, in developed countries, the density is less than 100 per liter.

5. Protozoa

Denver sewage contained an average of 52 _Ent._ _coli_ cysts per liter (Wang and Dunlop 1954). It has been estimated that untreated sewage may contain 96,000 to 2,400,000 _Giardia_ cysts per liter when 1 to 25 percent of the population is infected (Cooper 1984). Feachem (1983) reported that up to 530 _Giardia_ cysts per liter were found in Chicago area untreated sewage. It has been estimated that, due to the inadequacy of laboratory techniques, the actual count may be 99 percent underestimated.

6. Pathogenic E. coli

Cooper (1984) reported that Geldreich had estimated that one percent of the total coliforms are pathogenic E. coli, and that this may be possibly as high as three to four percent.

7. Indicator Organisms

By definition, the total coliform group in sewage must be equal, to or in higher concentrations than, fecal coliforms.

Total coliform bacteria in untreated domestic wastewater is commonly m in the range of 10^6 to 10^8/100 ml (DHS 1968). A drawback to the use of the total coliform as an indicator of fecal contamination in surface waters is that the intestine of warm-blooded animals is not the sole source -- it may be found in large numbers in nature, i.e., soils, plants, and runoff from unpopulated areas may contain high densities. This is not a particular drawback when total coliforms are used in domestic wastewater to measure disinfection effectiveness.

Fecal coliform bacteria have been found to range around 30 to 35 percent of the total coliform content of undisinfected sewage (DHS 1968), although individual waste treatment results may vary greatly. Large numbers of certain thermotolerant organisms (Klebisella) which record positive in the fecal coliform test may be present in certain types of industrial wastes (Dufour 1984).

Untreated sewage contains between 10^5 to 10^8 fecal coliform and fecal streptococci/100 ml. Geldreich (1978) reported 21 towns in the United States had between 3.4×10^5 and 4.9×10^7 coliforms, and 7 towns had 6.4×10^4 - 4.5×10^6 fecal streptococci/100 ml.

From this and other individual studies, it appears that fecal coliforms are about one log order of magnitude greater than fecal streptococci.

Crook (1984) presented a summary of microorganisms in untreated domestic sewage which is given in Table 6.

Table 6

Microorganism Populations in
Untreated Domestic Wastewater

Organism	Concentration (No./ml)
Coliform	$0.5\text{-}1 \times 10^6$
Fecal streptococci	$5\text{-}20 \times 10^3$
Shigella	Present
Salmonella	4-12
Pseudomonas aeroginosa	102
Clostridium perfringens	507
Mycobacterium tuberculosis	Present
Protozoan cysts	100
Helminth ova	1
Enteric virus	1-492

From the foregoing it is apparent that ratios of indicator organisms to pathogens are estimates subject to significant limitations (and criticism). Pathogens are highly variable due to factors already mentioned and are not often enumerated in water and wastewater.

The refinement in virus detection and enumeration methods in recent years has made the date of a study almost as significant in an evaluation as the reported concentration. Some organisms (shigella, helminth eggs, protozoa cysts) are almost never looked for. Salmonella and shigella results are either positive or negative rather than quantified.

With these reservations, Table 7 presents typical concentrations and Table 8 present estimated ratios of pathogens and indicator organisms in municipal wastewater.

Table 7

Municipal Sewage (Untreated) Typical Concentrations/100 ml
Total coliform -- 10,000,000
Fecal coliform -- 3,000,000
Fecal streptococci -- 500,000
Virus -- 500
Salmonella -- 100 to 10,000
Shigella -- 1 to 500
Helminth -- 1 to 100
Protozoa -- 10 to 200

Table 8

Pathogens -- Indicator Ratios

Virus	Total Coliform	Fecal Coliform	Fecal Strept.
1 :	2×10^4	6×10^3	10^3
Salmonella			
1 :	$10^3 - 10^5$	$3x10^2 - 3x10^6$	$5x10^1 - 5x10^3$
Shigella			
1 :	$2x10^4 - 10^7$	$6x10^3 - 3x10^6$	$10^3 - 5x10^5$
Helminth			
1 :	$10^5 - 10^7$	$3x10^6 - 3x10^4$	$5x10^5 - 5x10^3$
Protozoan Cysts			
1 :	$5x10^4 - 10^6$	$1.5x10^4 - 3x10^5$	$2.5x10^3 - 5x10^4$

B. Treated Wastewater

Pathogenic agents are removed to different degrees by the different stages and types of conventional wastewater treatment. Table 9 presents typical removals reported in the literature. Although much greater removal efficiencies are ascribed to trickling filtration than activated sludge treatment with regard to cysts and ova, no particular reasons are given for this. Influent and secondary effluent concentrations and percent removals for various microorganisms is given in Tables 10, 11, and 12.

Reported removal percentages range from an undefined low of "limited" to a high of 99 percent for bacteria and viruses (Engelbrecht 1980). Where detention in the treatment process is extended (oxidation ponds and lagoon systems) the fecal coliform bacteria apparently disappear more rapidly than the total coliform group possibly due to the greater sensitivity of thermotolerant organisms to adverse environmental factors. With conventional secondary treatment, such as activated sludge and trickling filtration. it appears from several sources that the percent removal of indicator organism and bacterial pathogens is similar and the removal of enteric viruses is generally somewhat less (DHS 1968, Engelbrecht 1975).

Table 9

Removal of Representative Infectious Agents by Conventional Wastewater Treatment

Infectious Agents	Primary Treatment % Removed	Secondary Treatment % Removed: Activated Sludge	Secondary Treatment % Removed: Trickling Filter
Salmonella	15	90-99	90-99.9
Mycobacterium	40-60	5-90	70-99
Shigella	15	80-90	85-99
Amoebic cysts	Limited	Limited	10-99.9
Helminth ova	70-95	Limited	60-75
Viruses	Limited	75-99	0-85

Tertiary treatment processes such as lime coagulation. coagulation-rapid sand filtration, and mixed-media filtration have achieved three to five logs removal of bacteria and viruses.

Table 10
Influent Concentration Ranges for

Organism	Number/100 ml	
	Minimum	Maximum
Total coliforms	1,000,000	46,000,000
Fecal coliforms	340,000	49,000,000*
Fecal streptococci	64,000	4,500,000
Virus	0.5	10,000

* Apparently, the samples containing the maximum fecal coliform levels were not analyzed for total coliforms because fecal coliform levels can never exceed the total coliform levels.

Table 11
Microorganism Reductions by Conventional Treatment Processes

Microorganism	Primary treatment removal, %	Secondary treatment removal, %
Total coliforms	<10	90-99
Fecal coliforms	35	90-99
Shigella sp.	15	91-99
Salmonella sp.	15	96-99
Escherichia coli	15	90-99
Virus	<10	76-99
Entamoeba histolytica	10-50	10

Table 12
Secondary Effluent Ranges for
Pathogenic and Indicator Organisms

Organism	Number/100 ml	
	Minimum	Maximum
Total coliforms	45,000	2,020,000
Fecal coliforms	11,000	1,590,000
Fecal streptococci*	2,000	146,000
Viruses	0.05	1,100
Salmonella sp.	12	570

* Assuming removal efficiencies for fecal streptococci similar to the fecal coliform removal efficiencies.

The following are specific examples of pathogenic agent removals by various treatment processes. These examples are presented to indicate the type of information that is available. It would not be appropriate to draw conclusions from any individual study of limited scope. Study results may be highly variable due to differences in water quality, treatment efficiencies, and study procedure. The use of ranges of percent removals based on a group of studies appears more appropriate in evaluating relative effectiveness of treatment.

1. Primary Sedimentation

   - 0 to 83 percent virus removal (Feachem 1980).
   - 24 to 33 percent virus removal (Rao et al., 1977).
   - 35 to 47 percent virus removal (Naparstek et al., 1976)

2. Secondary Treatment

   - Trickling filter -- 9 to 11 percent virus removal (Sherman 1975). Only includes removal by the trickling filter.
   - Trickling filter -- 23 percent virus removal (Buras 1976). Includes primary and secondary treatment.
   - Activated sludge -- 68 percent virus removal (Glass 1980). Only includes removal by the activated sludge process.
   - Activated sludge -- 80 to 90 percent virus removal (Moore 1974). Only includes removal by the activated sludge process.
   - Activated sludge -- 92 to 99.9 percent virus removal (Malina 1974). Only includes removal by the activated sludge process.
   - Trickling filter -- 93 percent salmonella removal (Feachem 1983). Includes primary and secondary treatment.
   - Trickling filter -- 79 to 94 percent salmonella removal at three plants (Feachem 1983). Includes primary and secondary treatment.
   - Activated sludge -- 99.4 percent salmonella removal (Feachem 1983). Includes primary and secondary treatment.
   - Trickling filter -- 74 to 91 percent protozoa cyst removal (Feachem 1983). Includes primary and secondary treatment.
   - Activated sludge -- 83 percent protozoa cyst removal (Feachem 1983). Includes primary and secondary treatment.

3. Tertiary Treatment

- o Mixed media filtration results in two logs removal of viruses (USDI 1979).

- o Activated carbon filtration can remove two to five logs of viruses, but there may be a breakthrough with time (Engelbrecht 1976).

- o At Washington, DC, a complete water treatment process, following sewage treatment and disinfection to a free, available chlorine residual in excess of 2 mg/l after 20 minutes, resulted in a coliform- and virus-free water. Total plate count average 67/100 ml (Warner 1978).

A summary of water reclamation unit process fecal coliform and virus removal from Engelbrecht (1980) is presented in Table 13.

Table 13

Reduction of Fecal Coliform and Viruses by Various Reclamation Unit Processes

Unit Processes	Log Reduction* Fecal Coliform Per 100 ml	Log Reduction* Virus Per 1,000 ml
Coagulation - Al_{2SO4} or $FeCl_3$	<0.5	1-2
Lime treatment - pH 11.2	2	3-4
Lime treatment - pH 11.5	>5	4-5
Activated carbon adsorption	<0.5	5
Rapid sand filtration	1-2	1-2
Chlorination, <breakpoint	1	1
Chlorination, >breakpoint, low turbidity and nitrogen	>7	>7

* Results assumed to be achievable under favorable operating conditions.

4. Disinfection

- o The effectiveness of disinfection with chlorine depends upon a range of factors, including the presence of humic material or certain industrial wastes and pH. A chlorine residual of 2 mg/1 may achieve a 3-log reduction in coliform bacteria after a contact time of 30 to 60 minutes. The enteric bacterial pathogens could be expected to exhibit a similar reduction. Protozoan cysts and helminth eggs would not be inactivated significantly. Enteric viruses would be reduced, but to a lesser degree than bacteria under the same disinfection conditions. Figures 1, 2, and 3 indicate that viruses are generally less susceptible to disinfection by chlorination than are bacteria. (It should be noted that the figures indicate the response to free residual chlorine which is generally not achieved in wastewater disinfection.)

- o According to Engelbrecht (1980), in order to achieve a 3-to 4-log inactivation of viruses in secondary effluent with a contact time of 30 to 60 minutes, a chlorine residual of 4 to 10 mg/1 would be required.

- o Depending on pH and turbidity, a chlorine residual of 2 mg/1 may achieve a 3-log reduction in coliform after 30 to 60 minutes (Atkin, E. W. et al. 1977).

- o A reduction of 3 to 5 logs in bacteria and viruses would be achieved by lime addition to reach a pH of 11.0 or greater (Engelbrecht 1980).

The effectiveness of tertiary effluent chlorination on virus removal is shown in Table 14.

Cooper (1984) presented information from the literature on the relative effects of chlorination on pathogenic agents.

a. Shigella -- Disinfection is very effective. 1 to 3 mg/1 chlorine at 30 minutes results in 100 percent kill.

b. Campylobacter -- There is little available information.

c. *E. coli* -- 5 to 10 mg/1 will achieve a 5-log removal (in water).

d. Salmonella -- Most disinfectants appear effective. Most data is expressed in percent kill. 0.2 to 5 mg/1 dose of hypochlorite causes 90 to 99.99 percent removal.

e. Enteroviruses -- Data is generally given in percent kill. Polio, coxsackie, and ECHO viruses are more resistant to free available chlorine than are enteric bacteria. Different types of viruses exhibit different resistance. It may take 3 to 8 hours to achieve 99.7 reduction in polio and coxsackie virus by

1 mg/l combined chlorine (although a small increase in the residual may result in a significant increase in the disinfection rate).

f. Other Enteric Viruses -- There is no data for Norwalk agent. Chlorine appears effective against rotavirus and, although there is no direct data regarding infectious hepatitis virus, it appears resistant.

g. Ent. Histolytica -- Cysts of this organism are resistant to disinfection. One hour at 5 mg/l free available chlorine is required for destruction.

h. G. Lamblia -- The cysts apparently are more resistant to disinfection than bacteria and viruses. One to 8 mg/l free available chlorine has killed 90 to 99.9 percent at 10 to 60 minutes.

   Limited test results suggest that the cysts may be more sensitive to chloramines than are viruses (Hoff 1986).

Table 15 displays the density changes in three indicator organisms after chlorination of wastewater. Based on the results, enterococci is more resistant to chlorination and would be the preferred indicator of disinfection effectiveness of the three.

i. Comparison of Microorganism Inactivation

   It is difficult to compare the relative effectiveness of effluent disinfection on various microorganisms because of the different dosages, pHs, temperatures, and contact times that have been used in studies. In order to reduce the variables, the concept of C.t values has been developed.

The effectiveness of wastewater disinfection, as measured by the reaction of indicator organisms, is directly related to the product of the amperometric chlorine residual and the contact time. Collins et al. (1970), developed the following equation for coliform bacteria reduction based on laboratory studies of primary treated wastewater in stirred-batch reactors:

$$y/yo = (1 + 0.23\ ct)^{-3}$$

Where: y = coliform bacteria density at residence time t.

yo = initial coliform density

c = total amperometric chlorine residual in mg/l at time t.

t = time in minutes.

Table 14

WATER RECLAMATION PLANT VIRUS SAMPLING SUMMARY

Sample Type	Pomona WRP Number of Samples	Pomona WRP Median[a] Concentration pfu/1000 gal	San Jose Creek WRP Number of Samples	San Jose Creek WRP Median[a] Concentration pfu/1000 gal	Whittier Narrows WRP Number of Samples	Whittier Narrows WRP Median[a] Concentration pfu/1000 gal	Water Factory 21 Number of Samples	Water Factory 21 Median[a] Concentration pfu/1000 gal
Unchlorinated Effluent								
Primary	13	2.3×10^5	8	1.6×10^6	8	7.6×10^5	-	-
Secondary	12	6.5×10^3	14	5.5×10^2	9	5.6×10^2	11[d]	4.3×10^2
Tertiary	12[b]	5.2×10^2	-	-	-	-	-	-
Chlorinated Effluent								
Tertiary	33[b]	<2	43[c]	<2	38[c]	<2	10[e]	<1

a/ Virus concentrations expressed as probability medians, plaque forming units per 1,000 gallons.
b/ Pomona tertiary effluent receives activated carbon filtration.
c/ San Jose Creek and Whittier Narrows tertiary effluents receive dual media filtration.
d/ Influent to Water Factory 21.
e/ Activated carbon effluent stream from Water Factory 21 which has received additional advanced treatment.

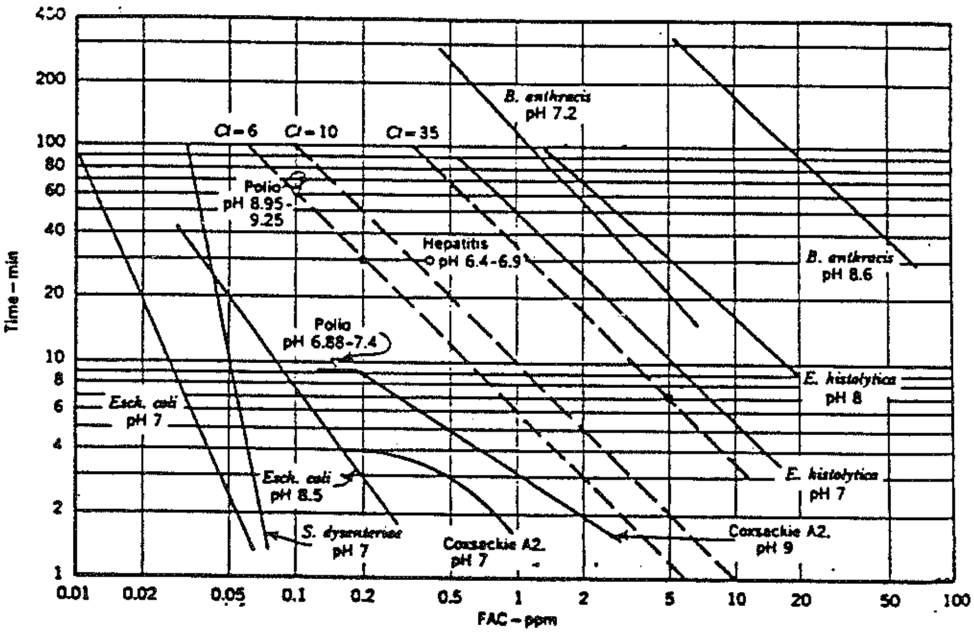


Fig. 1. Disinfection Versus FAC Residuals

Time scale is for 99.6 to 100 per cent kill. Temperature was in the range of 20–29°C. with pH as indicated.

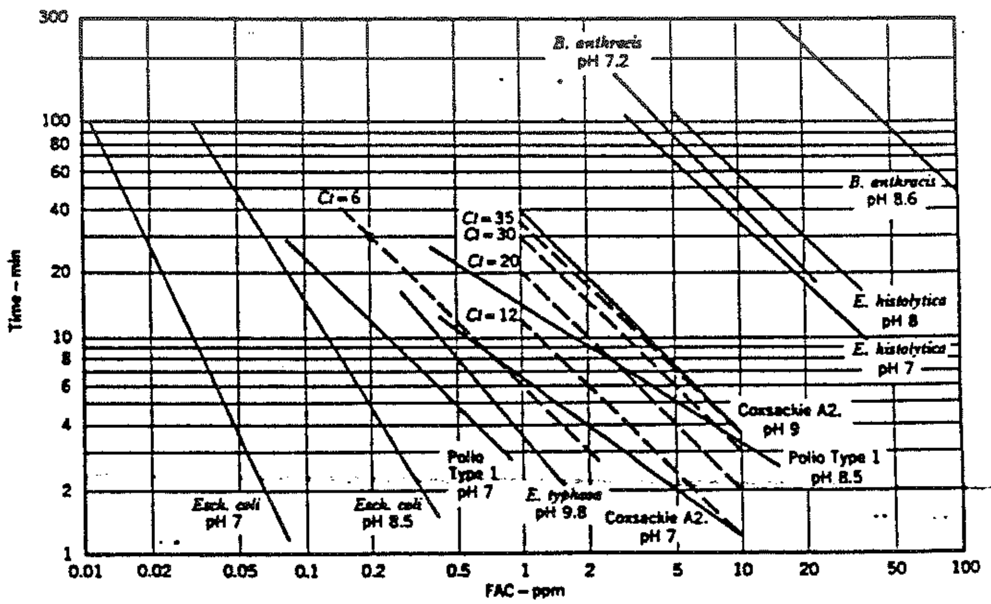


Fig. 2. Disinfection Versus FAC Residuals

Time scale is for 99.6 to 100 per cent kill. Temperature was in the range of 0–5°C, with pH as indicated.

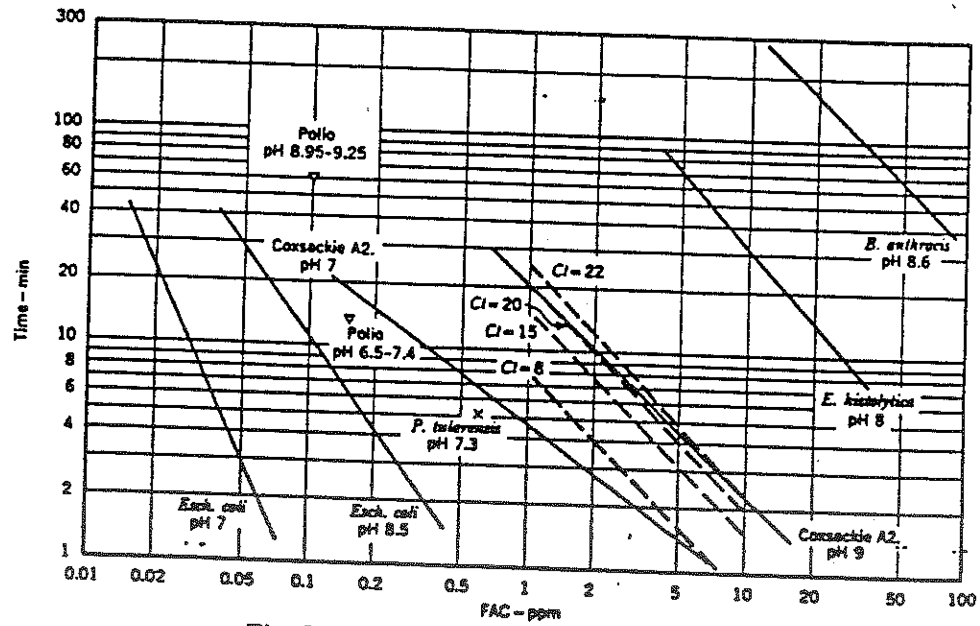


Fig. 3. Disinfection Versus FAC Residuals

Time scale is for 99.6 to 100 per cent kill. Temperature was 10°C, with pH as indicated.

Table 15

Log_{10} Density Changes in Fecal Coliforms,
E. Coli, and Enterococci After Chlorination

WTP[a]	Prechlor-ination Effluent[a]	Day	Residual Cl_2 (mg/l)	Indicator	Log_{10} Density/100 ml Prechlor-ination	Post ? Prechlor-ination	Log_{10} Reduction in Bacterial Density[b]
A	Pl	1	2.5	Fecal coliform	7.556	0.477	7.079
				E. coli	7.380	0.477	6.903
				Enterococci	5.477	0.000	5.477
		2	1.0	Fecal coliform	7.519	2.544	4.974
				E. coli	7.342	2.380	4.962
				Enterococci	5.785	2.000	3.785
D	STFE	1	0.25	Fecal coliform	6.079	0.301	5.778
				E. coli	5.982	0.000	5.982
				Enterococci	4.580	0.000	4.580
		2	1.0	Fecal coliform	6.613	0.301	6.312
				E. coli	6.462	0.000	6.462
				Enterococci	4.447	0.000	4.447
G	SFE	1	0.1	Fecal coliform	4.505	1.623	6.882
				E. coli	4.447	1.415	3.032
				Enterococci	3.362	1.114	2.248
H	SASE	1	3.0	Fecal coliform	4.301	0.000	4.301
				E. coli	4.000	0.000	4.000
				Enterococci	2.903	<0.000	>2.903
		2	2.5	Fecal coliform	4.580	1.176	3.404
				E. coli	4.322	0.778	3.544
				Enterococci	3.000	0.000	3.000
I	SASE	1	2.5	Fecal coliform	6.176	2.114	4.062
				E. coli	5.785	2.079	3.706
				Enterococci	4.477	1.623	2.854
		2	2.5	Fecal coliform	6.301	2.230	4.071
				~~E. coli~~	~~5.982~~	~~2.041~~	~~3.941~~
				Enterococci	4.462	1.602	2.860

a WTP, wastewater treatment plant; Pl, plant influent; SEFE, settled trickling filter effluent.

b Values not given for the following post-chlorination densities for all three indicators were undetectable (<1/100 ml): Plant B, days 1 (CL_2, 2.5 mg/l) and 2 (Cl_2, 2.5 mg/l), and 2 (Cl_2, 2.5 mg/l); Plant G, day 2 (Cl_2, 5.0 mg/l).

Stahl, J. (1971), developed a similar model for three secondary sewage treatment plants in the Los Angeles area:

$$y/yo = 27.22\ (1 + ct)^{-2.27}$$

Hoff (1986) used reported disinfection data to developed c.t values for various microorganisms. The work was developed for use in evaluating drinking water treatment needs; however, it does provide insight into the relative effectiveness that wastewater disinfection has on several microorganisms (see Table 16).

It can be seen that the c.t value to achieve 99 percent inactivation, where chloramine (combined chlorine) is the disinfectant, is much greater than that for free available chlorine (although a high pH was involved with chloramine). For combined chlorine, such as would be present in wastewater disinfection, a significantly higher c.t value is needed for virus inactivation than for E. coli inactivation. The available data on protozoan cysts indicate that they are resistant to several disinfectants. Data from tests at higher temperature (59°F) suggest that the cysts may be more sensitive to chloramines (combined chlorine as would result in wastewater disinfection) than viruses although this is not certain.

Parkhurst (1977) computed the relative removals of virus and coliform bacteria in chlorinated tertiary effluent. The slope of the regression line (for a certain range of removals) was 0.33 indicating that for a 1-log removal of coliform bacteria, a 0.33 log removal of viruses would occur indicating the lesser resistance of coliforms to chlorination.

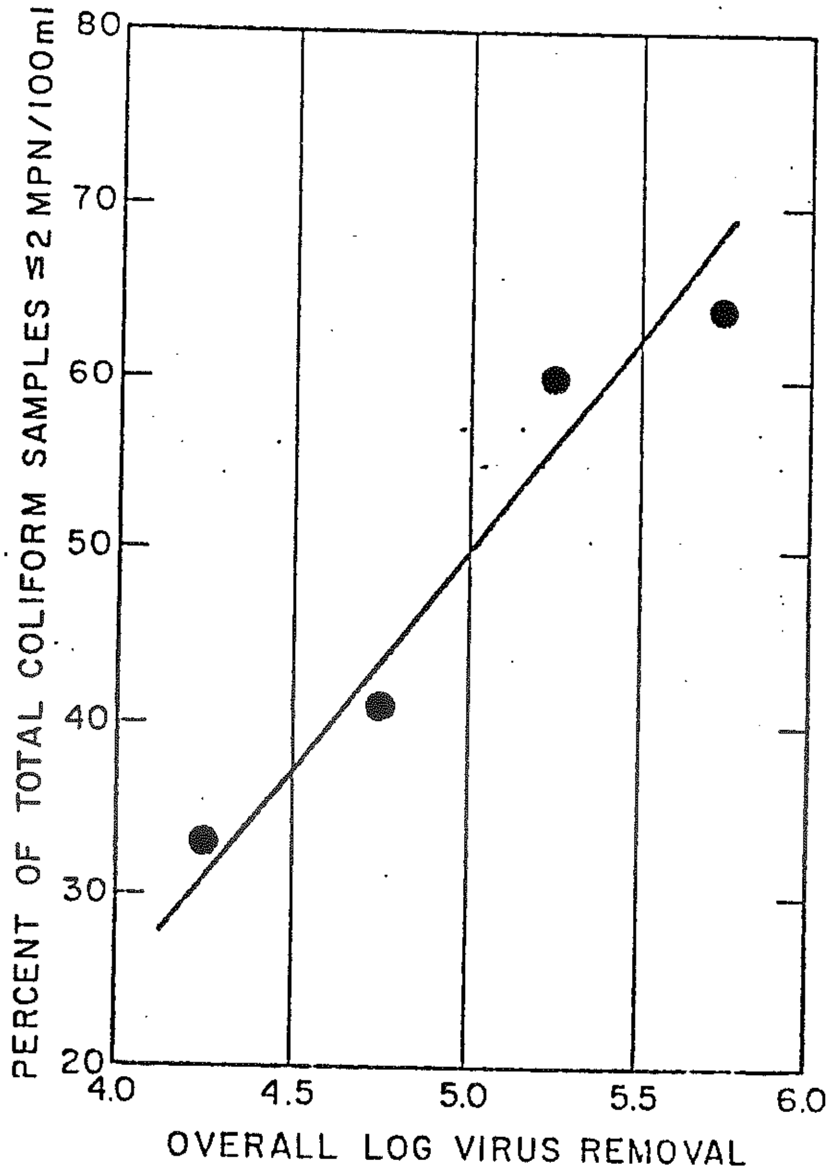
Figure 4 relates the virus removals associated with achieving a coliform bacteria level of 2.2/100 ml through the disinfection of tertiary effluent. The data indicate that when a median of 2.2/100 ml is achieved, approximately a 5-log virus removal has occurred. It should be noted that the results are of one effluent (Pomona) seeded with Polio I virus.

Table 16

Summary of C-t Value Ranges for 99% Inactivation of Various Microorganisms by Disinfectants at 5°C

	Disinfectant			
Microorganism	Free Chlorine pH 6 to 7	Preformed Chloramine pH 8 to 9	Chlorine Dioxide pH 6 to 7	Ozone pH 6 to 7
E. coli	0.034-0.05	95-180	0.4-0.75	0.02
Polio I	1.1-2.5	768-3740	0.2-6.7	0.1-0.2
Rotavirus	0.01-0.05	3806-6476	0.2-2.1	0.006-0.06
Phage f_2	0.08-0.18	-	-	-
G. lamblia cysts	47->150	-	-	-
G. muris cysts	30-630	-	7.2-18.5	1.8-2.0

Figure 4



FREQUENCY OF ACHIEVING COLIFORM STANDARD
AT VARIOUS VIRUS REMOVAL LEVELS
DISINFECTANT: CHLORINE

C. Regrowth

The phenomena of the regrowth or multiplication of indicator organisms in sewage and sewage effluents following dilution with fresh water and storage, or following disinfection and discharge to fresh receiving waters, has received a great deal of study since the 1930s. Not only is there the desire to understand the cause of such regrowth, there is concern with the public health implications: specifically, will bacterial pathogens exhibit the same type and degree of regrowth, and what effects will this have on the potential for waterborne disease?

Greenberg (1971) reported on the work of earlier investigators and the results of his own studies on the regrowth of indicator organisms. Hoskins and Butterfield had speculated that coliform bacteria aftergrowth following dilution and storage was due to both natural multiplication and the reduced presence of predators. Others, Hewkelekian, Rudolfs, and Gehm, supported the improved food supply and less predators, respectively. Further studies after World War II did little to resolve the issue, and the breakup of sewage particles was also suggested as an alternative to actual regrowth. Others have proposed that the apparent regrowth is the reactivation of bacteria which have been stunned but not killed by the disinfection process.

One experiment by Greenberg showed a substantial increase in coliform organisms (from <180 to >2.4 x 10^6) in secondary effluent which had received a chlorine dosage of 1 mg/l. Fecal coliform increased from <180 to 2,000/100 ml. There was no substantial regrowth of bacteria in oxidation pond effluent or soil-filtered tertiary effluent which had received from 1 to 5 mg/l doses of chlorine. This was ascribed to nutrient deficiencies, but also may be due, in part, to the presence of a chlorine residual over a longer period of time. Filtered river water containing 0.1 percent and 1 percent filtered sewage exhibited regrowth of coliforms.

Several studies investigated the possibility of "reviving" organisms which had been inactivated by various disinfectants. Incubation in combinations of metabolites and substrates allowed some multiplication; however, Garvie concluded that this was due to the multiplication of a few survivors and, when there were zero survivors, incubation in metabolites or enhancement media gave no recovery.

Greenberg's 1971 studies showed that aftergrowth is greatest for total coliform, less for fecal coliform, and relatively negligible for fecal streptococci (see Table 17).

Results of samples of sewage treatment plant effluent and river water from the Santa Ana River suggest the apparent ability of bacteria to regrow in a shallow warm river when sufficient nutrient material is present (see Case Examples). The results may be due, however, to uncontrolled sources or other factors. A significant portion of the river, as much as 60 percent, may be effluent from the treatment plants during the low flow summer conditions. The log mean of coliform bacteria in the discharges range from 2 to 43/100 ml and the concentrations at downstream river stations range from 8,000 to 17,000/100 ml. Enterococcus in the effluents range from 1 to 4/100 ml and from 40 to 450 in the river. E. coli is 8 to 60 in the effluents and 480 to 720 in the river. Finally, fecal coliform is 2 to 19/100 ml in the effluent and 180 to 6,000 in the river. If regrowth is involved, the results indicate that coliform bacteria displayed the greatest regrowth capability and enterococci and E. coli the least.

Regrowth of coliform and fecal coliform bacteria in sewage effluent and downstream receiving waters has been documented in several studies. In one study (Shuval, 1973) trickling filter effluent contained 5 x 10^6 coliforms per 100 ml which was reduced to 120/100 ml by chlorination. After storage for 3 days, the coliform content rose to 800/100 ml. A mixture of 1 percent undisinfected primary effluent and river water at Sacramento showed a 40-fold increase in coliform bacteria but only a 6-fold increase in fecal coliforms (DHS 1968).

At Santee, unchlorinated secondary effluent exhibited no regrowth, but disinfected effluent had increases of coliforms and fecal coliforms of similar magnitude after four to five days. The increase could be due to regrowth or reactivation. Ten percent mixture of disinfected effluent and Arcata Bay (marine water) water displayed no regrowth.

Studies with salmonella and shigella, according to Geldreich (DHS 1968) have indicated that these pathogenic bacteria can also go through aftergrowth or regrowth when conditions are suitable.

In general, regrowth or recovery of coliform and fecal coliform in chlorinated effluent and in effluent receiving-water mixtures has been found to be a real phenomena, not a breakup of clumps of organisms.

The total coliform group exhibits greater regrowth than fecal coliforms, major regrowth is more likely to be exhibited in fresh water than marine water, and regrowth of pathogenic bacteria is uncertain.

D. Receiving Waters -- Fresh

An evaluation of the results of a series of river studies including Red River of the North, Sacramento, Blackstone, Upper Mississippi, and Pigeon River (DHS 1968) provided some general findings regarding coliform and fecal coliform bacteria in fresh receiving waters. With a few exceptions, the fecal coliform bacteria die away faster than the total coliform after the first day or two. Composite die-away curves for fecal coliforms for summer and winter indicate much slower die-away rates in winter. Qualitative isolation of pathogenic salmonella downstream from sewage discharges, in several of the river studies, indicated that the persistence of these organisms is at least as long as fecal coliforms.

Table 17

Effect of chlorination on aftergrowth

SECONDARY EFFLUENT

Time, days	Chlorine dosage, mg/l: 0				1			
	Cl_2 residual, mg/l*	MPN/100 ml Coliform	Fecal Coliform	Fecal Strep	Cl_2 resid. mg/l*	MPN/100 ml Coliform	F. Coliform	F. Strep
0	0/0	7.9×10^6	2.3×10^6	79000	2.3/0	<1800	<1800	<180
1	0/0	240×10^6	0.7×10^6	7900	0.95/0	<180	<180	<18
2	0/0	1.3×10^6	0.078×10^6	7800	0.15/0	16000	180	<1.8
4	0/0	0.33×10^6	0.045×10^6	<180	0/05/0	$\geqq 2.4\times10^6$	780	23
6	0/0	0.33×10^6	<18000	<180	0/0	0.33×10^6	14000	<1.8
8	0/0	78000	20000	45	0/0	0.17×10^6	<1800	<1.8
11	0/0	79000	2000	4.5	0/0	0.49×10^6	2000	<1.8
14	0/0	130000	7800	4.5	0/0	0.079×10^6	2000	<1.8

Time, days	Chlorine dosage mg/l: 2				5			
	Cl_2 residual, mg/l*	MPN/100 ml Coliform	Fecal Coliform	Fecal Strep	Cl_2 resid. mg/l*	MPN/100 ml Coliform	F. Coliform	F. Strep
0	3.4/0	<1800	<1800	<180	7.8/	<180	<180	<18
1	1.4/0	<180	<180	<180	4.6/0	<18	<18	<18
2	0.65/0	3500	<1.8	9.1	3.25/0	130	<1.8	<1.8
4	0.15/0	3300	<180	--	2.55/0	490	45	79
6	0/0	170000	<180	<1.8	1.60/0	790	45	<1.8
8	0/0	28000	<180	<1.8	0.85/0	1400	20	<1.8
11	0/0	7900	<180	<1.8	0/0	1400	110	<1.8
14	0/0	13000	<180	<1.8	0/0	790	130	<1.8

Stream data compiled by Geldreich (1970) indicate a sharp rise in salmonella isolations (from 27.6 percent up to 85.2 percent) when greater than 200 fecal coliform per 100 ml were encountered in fresh water.

From a worldwide perspective, it has been stated that enteroviruses can be isolated in low concentrations from almost all surface waters receiving human wastes. The Missouri and Mississippi have contained up to 0.1 and 0.4 enteroviruses per liter, respectively (Feachem 1983). These concentrations for the viruses involved, if accurate, would not appear to be a significant health hazard.

Enteric virus survival times in water vary with the specific virus, temperature, and general quality of the water. Studies have shown survival times of 24 to 272 days at temperatures less than 10 C. Viruses have been found to survive longer in distilled water and grossly polluted water than in moderately polluted water. In Europe, where effluent disinfection is generally not practiced, the Seine at Paris contained 170 enterovirus per liter, and the Thames at London had 100/liter (World Health Organization (WHO) 1979, and Berg 1978).

Table 18 presents the relative concentrations of indicator organisms and viruses in the Ohio River at Cincinnati. It should be noted (Allen, M.J. 1978) that the viruses were isolated from 100-gallon samples. There have been several estimates of virus to coliform or fecal coliform bacteria in surface water. The Ohio River data indicated a virus-fecal coliform range from 1:23 to 1:11,000.

An estimate based on per capita excretions is 1:50,000 virus to coliforms in polluted waters.

A very generalized survival time for pathogens and fecal coliform bacteria in fresh water is given in Table 19 (Feachem 1983). A comparison of die-off rates for various microorganisms is given in Table 20. The enterococci appear to survive longer than the other indicator organisms and would be more reflective of the hardier pathogens than the other indicators.

Table 18

MICROBIOLOGICAL QUALITY OF OHIO RIVER WATER FROM CINCINNATI AREA

Location[1]	Total Coliforms (per 100 ml)	Fecal Coliforms (per 100 ml)	Standard Plate Count per ml	Virus[2] Isolates
Schmidt (467)[3]	11,000	6,500	37,000	0
Southside (475)[3]	49,000	17,000	62,000	3
Schmidt	85,000	23,000	79,000	9
Schmidt	38,000	11,000	48,000	1
Public Landing (470)[3]	3,400	370	20,000	16
Schmidt	105,000	43,000	162,000	17
Public Landing	35,000	3,500	112,000	1
Schmidt	99,000	42,000	193,000	31
Public Landing	48,000	7,200	46,000	12
Southside	135,000	33,000	79,000	59
Public Landing	7,300	1,200	8,300	3

[1] Sampling period 9/8/75 through 12/1/75
[2] From 100 gallon (380 liters) water samples
[3] Ohio River Mile

Table 19

Survival Times of Pathogens
In Freshwater and Sewage at 20-30 C

Pathogen	Days
Viruses	
Enteroviruses	<120, but usually <50
Bacteria	
Fecal coliforms	<60, but usually <30
Salmonella sp	<60, but usually <30
Shigella sp	<30, but usually <10
Vibrio cholerae	<30, but usually <10
Protozoa	
Ent. histolytica cysts	<30, but usually <15
Helminths	
Ascaris eggs	many months

Table 20

Comparative Die-Off Rates of Fecal Indicator Bacteria and Enteric Pathogens

Bacteria	Half-time[a] (Hours)	No. of Strains-Analyzed
Indicator Bacteria		
Coliform bacteria (avg.)	17.0	29
Entrococci (avg.)	22.0	20
Coliform from raw sewage	19.5	
Streptococci from raw sewage	19.5	
Streptococcus equinus	10.0	1
S. bovis	4.3	1
Pathogenic Bacteria		
Shigella dysenteriae	22.4	1
S. sonnei	24.5	1
S. flexneri	26.8	1
Salmonella enteritidis ser. paratyphi A	16.0	1
S. enteritidis ser. paratyphi D	19.2	1
S. enteritidis ser. typhimurium	16.0	1
S. typhi	6.0	2
Vibrio cholerae	7.2	3
S. enteritidis ser. paratyphi B	2.4	1

a The half time was determined graphically from the time required for a 50 percent reduction in the initial population.

Cooper (1984) has searched the literature for survival times of pathogens in fresh water. The following is a very brief summary of his findings.

1. Shigella -- 22 to 47 days. Generally a longer survival time than other bacterial pathogens. Survives longer at lower temperatures.

2. Campylobacter -- 4 to 28 days based on very little data. Survives longer at lower temperatures.

3. Pathogenic E. coli -- Survives longer in polluted water, higher temperatures, and lower pH.

4. Yersinia -- Very little data. Survives approximately 4 to 8 days and shorter than fecal coliform.

5. Enteroviruses -- Longer at low temperatures and high levels of pollution. Survives up to 160 days.

Several pathogenic agents, under certain circumstances, have been found to be viable but nonculturable by conventional test methods. High concentrations of viable Campylobacter jejuni (10^6/ml) were detected after incubation for 10 days in a stationary aquatic environment, although the cells could not be cultured by special plating.

E. Receiving Waters -- Marine

The survival time of indicator organisms, pathogenic bacteria (with the possible exception of shigella), and enteroviruses in marine waters are much shorter than in fresh waters. (Protozoa cysts do not appear to be affected by the water's salinity.) For example, 90-percent reductions of coliform bacteria in marine water may occur in 0.6 to 8 hours as compared to 20 to 100 hours in fresh water. Inasmuch as viruses survive for much longer periods than bacteria (99 percent reductions from several days to several months depending on the virus and temperature), there have been reports of virus isolations where few or no coliforms were present. At the Miami, Florida, ocean outfall, up to 3 enteroviruses per 400 liters were detected with fecal coliforms always less than 3 per 100 ml (Edmond et al. 1978).

In another study, enteroviruses were isolated from 43 percent (no quantification) of marine recreational water samples which met a total coliform standard of less than 1,000/100 ml and from 35 percent of shellfish growing waters having a total coliform content of less than 230/100 ml (Gerba 1979).

Fecal streptococci survive longer than fecal coliforms in the marine environment (15 to 70 hours), but fall far short of virus survival times (Baross et al. 1975).

Because of the discrepancies in survival times, there is a consensus that current indicator organisms do not serve well in the marine environment.

Cooper (1984) presented information from the literature on survival times of specific pathogens in marine waters.

1. Shigella -- Can be resistant to salt concentrations. Survives 15 to 70 days.

2. Pathogenic E. coli -- Rapid decline in seawater. Survives 1 to 7 days; longer with low temperatures and higher organic loading.

3. Salmonella -- The survival is shortened by salinity; however, it is increased by polluted waters. A decrease of 10^5 took 10 weeks with fecal pollution present.

4. Enteroviruses -- The viruses survive longer in summer in ocean water (60 to 80 days) than in winter (26 to 48 days). Survival may be longer (>100 days) in estuarine waters in the winter.

5. Other Enteric Viruses -- There is no specific survival data for rotaviruses, but they have been found in high concentrations in bay water. Hepatitis A is thought to be retained by oysters for up to two months.

F. Disease Associated with Exposure to Pathogens by Various Routes

There have been very few epidemiological studies of waterborne disease which have been able to provide information on all the major factors which would enable a thorough evaluation of disease risk associated with the quality of drinking, recreational, agricultural, or shellfish growing waters. The factors are:

- Disease rate;
- Pathogen concentrations;
- Indicator organisms concentrations; and
- Degree of exposure.

There are several general reasons for the lack of information on which to base firm conclusions regarding disease transmission for the several routes. First, waterborne disease incidence is greatly underreported. Acute gastroenteritis has been identified as clearly the most prevalent disease transmitted by water-related transmission routes. In the United States in 1980, there were 15,134 cases via drinking water, 137 cases via shellfish consumption, and 83 cases associated with recreational water use (Cabelli 1983). This is a nonreportable disease, and a substantial number of cases or even significant outbreaks may escape detection by the existing surveillance systems.

It has been estimated that the actual number of waterborne disease outbreaks may exceed those reported by two- to tenfold (Cabelli 1983, Lippy 1984). The two fold increase is based on the increased number of outbreaks involving drinking water when a voluntary reporting procedure was replaced by an improved procedure. The suspicion that as little as ten percent of the outbreaks are reported is based on a Washington State study which indicated that only one outbreak of foodborne disease in ten had been recognized and reported. The same might be true of waterborne disease. An even higher percentage of illnesses not associated with outbreaks may go unreported. Cooper (1975) has estimated this to be 95 percent.

ORGANIC CONSTITUENTS

Wastewater, by its' very nature, may contain a myriad of man-made and naturally-occurring organic compounds. Thousands of chemical products are in use which may find their way into wastewaters. Many organic constituents found in domestic wastewater derive from feces, urine, paper products, food wastes, detergents, cleaning agents, disinfectants, cosmetics, and skin excretion and contaminants from bathing. Pesticides, laundry products, paint products, polishes, and preservatives are discarded into wastewater less frequently, but in large concentrations. Domestic wastewater containing industrial wastes can contain significant quantities of chemical constituents of health concern, including solvents, plasticizers, pesticides, pigments, and polymers, to name a few.

The presence of organic materials in raw and treated domestic sewage is highly variable and dependent upon the source of the sewage and the degree of treatment which it receives. In terms of gross parameters, raw domestic sewage generally has the following organic composition: biochemical oxygen demand (BOD) - 175 to 225 mg/l, chemical oxygen demand (COD) - 325 to 450 mg/l, and total organic carbon (TOC) - 100 to 130 mg/l. The organic content of sewage after treatment is considerably less. The application of sedimentation and a biological system, e.g., activated sludge, will reduce the BOD to 15-25 mg/l, COD to 40-70 mg/l, and TOC to 15-25 mg/l. Additional treatment such as chemical coagulation, sedimentation, filtration, and carbon adsorption can produce an effluent with a BOD of 0.1-5.0 mg/l, a COD of 3-25 mg/l, and a TOC of 1-6 mg/l [California Department of Health and Cooper, (1975)].

According to Hunter (1971), organic compounds from the degradation of organic material in wastewater include a complex mixture of carbohydrates, amino acids, proteins, fatty acids, carboxylic acids, esters, nucleic acids, amino sugars, amides, aromatic and aliphatic hydrocarbons, phenols, polynuclear aromatic hydrocarbons, and more complex large molecular weight materials such as tannins, lignins, and humic materials. Humic materials are a generic type of organic substance classified according to solubility. These complex polymers range in molecular weight from several hundred to many thousand. Degradation products of humic material that could be found include benzoic acid, catechol, hydroxy- and di-hydroxy-benzoic acids, phenols, resorcinol, syringic acid, and vanillic acid. Humic materials are reported by Manka et al (1974) to comprise approximately 50% of the soluble organic matter in wastewaters. Jolley (1984) reported that hundreds of other constituents have been detected in wastewater at 1-200 ug/l concentrations, including alcohols, aromatic acids, indoles, phosphates, phthalates, polyols, purines, pyridines, and pyrimidines.

A review of the literature by Kowal (1981) indicated that, in medium strength sewage (700 ppm total solids content), approximately 75 percent of the suspended solids and 40 percent of the colloidal and dissolved filterable solids consist of organic material - primarily of proteins (40-60%), carbohydrates (25-50%), and fats and oils (10%). The more refractory and high-molecular weight water soluble organics predominate after secondary treatment, such as fulvic acid, humic acid, and kymathomelanic acid. Kowal (1981) reported that the organic priority pollutants most frequently used and discharged into domestic wastewater were predicted to be the following: benzene, phenol, 2,4,6-trichlorophenol, 2-chlorophenol, 1,2-dichlorobenzene, 1,4-dichlorobenzene, 1,1,1-trichloroethane, toluene, naphthalene, diethylphthalate, dimethylphthalate, trichloroethylene, aldrin, and dieldrin. Some individual compounds and classes of compounds typically found in raw domestic wastewater are shown in Table 5. Although many of the organic constituents are removed or reduced in concentration by conventional wastewater treatment through biodegradation, volatilization, and incorporation into sludge, some of the compounds are not significantly reduced.

One of the most recent and comprehensive studies of organic constituents in wastewater was done by Nellor et al (1984) on reclaimed wastewater from several treatment plants in Los Angeles County. The treatment chain included activated sludge treatment, followed by filtration, disinfection, and dechlorination. In addition to identification and quantification of several individual compounds, several miscellaneous groups of compounds were characterized in the treated effluent. Some of the miscellaneous groups of compounds found in sample residues were: petroleum and combustion byproducts - alkyl benzenes, naphthalene, alkyl naphthalenes, alkyl phenols, alkyl phenanthrenes, ketones, glycol ethers, branched chain aliphatics, and aldehydes; metabolic byproducts - fatty acids, aromatic acids, esters, and alcohols; and industrial materials - phthalates, styrenes, hydrocarbon solvents, and detergents.

The wide variation in reported organic levels in various wastewaters is not only due to the source of the sewage and degree of treatment. Sampling methods, laboratory analytical techniques, and the level of sophistication of laboratory instrumentation are also important factors. Although the organic fraction remains largely unidentified, several studies have documented the presence of specific organic constituents in treated wastewater. Table 6 presents organic compounds identified in both unchlorinated and chlorinated trickling filter and activated sludge effluents. Table 7 indicates the organic priority pollutants found in undisinfected activated sludge effluent from the Orange County Sanitation District's wastewater treatment plant. Table 8 indicates specific organic compounds identified in tertiary-treated effluent, i.e., activated sludge treatment followed by filtration, in Los Angeles County. These tables include selected organic constituents and in no way are meant to represent a total list of all organic constituents likely to be present in wastewater. Many more organic compounds are present in wastewater than are indicated in these tables.

Chlorinated Organic Constituents

Chlorine-organic material oxidation reactions constitute the major reacton of chlorine in water and wastewater. Jolley (1976) found that approximately 99 percent of the HOCl applied dose to wastewater was recovered as Cl^- from oxidation reactions. The oxidation reactions do not lead to any halogenated products by themselves. However the oxidized forms

Table 5

Individual Compounds and Compound Types Typically found in Raw Sewage
[Lauer et al (1984)]

<u>Chlorinated</u>

Chloroform
Dichloroethene
Trichloroethene
Tetrachloroethene
Trichloroethane
Dichlorobenzene (o,m,p)

<u>Aromatic</u>

Benzene
Toluene
Alkylated benzenes
Naphthalenes
Alkylated naphthalenes
Xylenes (o,m,p)
Aromatic aldehydes

<u>Aliphatic</u>

Straight chain, branched and cyclic-alkenes, alkynes and alkanes

<u>Carbonyl</u>

Ketones
Aldehydes
Carboxylic acids

<u>Other (individual)</u>

Decalin
Methyl decalin
Dimethyldisulfide
Phenol
Phosphoric acid-tributyl ester

<u>Other (class)</u>

1^o, 2o, 3^o Alcohols
Diols
Indenes
Phthalates
Triazoles
Acetamides
Indenones
Pyranones
Alkylated phenols
Quinolines
Amides
Pyrazoles
Nitriles
Furans
Oxazoles
Oxiranes
Indoles
Alkylated disulfides
Pyridines
Thiophenes

Table 6

Concentration of Organic Compounds in
Municipal Wastewater Treatment Plant Effluents
[Majeti and Clark (1982)]

Compound	Dayton, Ohio[1] Unchlorinated (ug/l)	Dayton, Ohio[1] Chlorinated (ug/l)	Cincinnati, Ohio[2] Unchlorinated (ug/l)	Cincinnati, Ohio[2] Chlorinated (ug/l)
Chloroform	0.3 to 1.4	0.4 to 12	0.1 to 0.7	0.5 to 12
Trichloroethylene	0.2 to 1.7	0.1 to 10	-	0.6 to 0.7
Benzidine	-	<0.1	-	<0.1
Vinyl Chloride	-	<1	-	<1
Benzene	-	0.2 to 40	-	0.3 to 3.8
PCBs	-	<1	-	-
Endrin	-	<1	-	-
Toxaphene	-	<1	-	-
Methanol	-	150 to 510	-	-
Ethanol	-	150 to 3,000	-	-
Acetone	-	50 to 300	-	20 to 400
2,3-Dithiabutane	-	-	-	1
Carbon Disulfide	-	-	-	2 to 8
1,1,1-Trichloroethane	-	1 to 15	-	-
Tetrachloroethylene	-	1 to 20	-	0.3 to 3
Toluene	-	1 to 10	-	1
Xylene	-	1 to 15	-	10
Acrolein	-	20 to 200	-	10 to 150
Acetaldehyde	-	90 to 1,350	-	100 to 560
Carbon Tetrachloride	-	3	-	-
Chlorodibromomethane	-	0.1	0.1	0.4 to 4.6
Dichlorobromomethane	0.1	0.1 to 0.4	0.1 to 0.3	0.1 to 8
Bromoform	-	0.1 to 0.3	-	0.2 to 0.3
1,3-Dichloroethane	1.4	0.1 to 4.6	-	-
Methylene Chloride	-	2 to 50	-	1 to 10

[1]Trickling filter secondary effluent
[2]Activated sludge secondary effluent

Table 7

Organic Priority Pollutants in Secondary Effluent
[McCarty et al (1982)]

Compound	No. Samples	Geom. Mean (ug/l)	95% Conf. Inter.	Spread Factor
Chloroform	99	3.5	3.1-3.9	1.84
Bromodichloromethane	98	0.46	0.42-0.50	1.57
Dibromochloromethane	100	0.71	0.65-0.78	1.55
Bromoform	96	0.46	0.39-0.53	1.99
1,1,1-Trichloroethane	99	4.8	4.0-5.8	2.65
Trichloroethylene	99	1.1	0.9-1.3	2.87
Tetrachloroethylene	101	3.6	2.9-4.5	2.97
Carbon Tetrachloride	99	0.05	0.04-0.06	2.18
Chlorobenzene*	110	0.13	0.10-0.16	3.09
1,3-Dichlorobenzene	108	0.25	0.21-0.29	2.29
1,4-Dichlorobenzene	109	1.9	1.7-2.1	1.64
1,2-Dichlorobenzene	110	0.74	0.61-0.89	2.65
1,2,4-Trichlorobenzene	110	0.31	0.25-0.38	2.85
Napthalene	109	0.11	0.08-0.16	4.39
Ethylbenzene	99	0.04	0.03-0.05	3.52
2,4-Dichlorophenol	23	0.16	0.03-0.99	9.28
2,4,6-Trichlorophenol	23	0.13	0.04-0.41	6.50
Pentachlorophenol	23	1.23	0.58-2.6	4.16
PCB (Aroclor)	86	0.40	0.32-0.49	2.27
Lindane*	86	0.11	0.10-0.12	1.55
DDT*	86	0.01	0.00-0.02	7.74
Di-n-butyl Phthalate	29	0.94	0.59-1.5	3.10
Diethyl Phthalate	29	1.14	0.35-3.7	10.2
Bis(2-ethylhexyl) Phthalate	29	11	5-27	9.5
Isophorone	13	0.30	0.04-2.3	14.7

*Concentration equal to less than value shown

Organic Priority Pollutants Detected in Secondary Effluent On Occasion but not Quantitatively Evaluated

Compound	No. Samples	No. Times Detected	Detection Limit	Max. Conc. Found
Phenanthrene	29	1	0.02 ug/l	0.06 ug/l
Anthracene	29	1	0.02 "	0.03 "
Fluoranthene	29	1	0.02 "	0.04 "
Pyrene	29	1	0.02 "	0.05 "
Butylbenzyl Phthalate	29	5	0.1 "	38 "
Di-n-octyl Phthalate	28	3	0.1 "	0.4 "
Phenol	13	1	0.05 "	12 "

Table 8

Results of Target Organic Analyses in Reclaimed Water Used for Groundwater Recharge in the Montebello Forebay[1]
[Nellor et al (1984)]

Target Compound	Reclaimed Water[2] Concentration (ug/l)
Methylene Chloride	4.9 - 56
Chloroform	5.8 - 84
Bromodichloromethane	<0.1 - 2.9
Dibromochloromethane	<0.1 - 2.1
Bromoform	<0.1 - 4.8
Carbon tetrachloride	<0.1 - 12
1,1-Dichloroethane	<0.1 - 1.8
1,2-Dichloroethane	<0.2 - 2.2
1,1,2-Trichloroethane	<0.2 - 21
Trichloroethylene	<0.1 - 19.4
Tetrachloroethylene	<0.1 - 17.4
Chlorobenzene	<0.1 - 1.0
1,4-Dichlorobenzene	0.2 - 10.2
1,2-Dichlorobenzene	0.2 - 6.0
Benzene	<0.2 - 2.9
Toluene	<0.1 - 1.3
Bis(2-ethylhexyl)phthalate	0.7 - 13
1,2,4-Trichlorobenzene	<0.1
2,4,6-Trichlorophenol	<0.2 - 0.7
2,4,5-Trichlorophenol	<0.2 - 0.9
2,3,4-Trichlorophenol	<0.4
2,3,6-Trichlorophenol	<0.2
3,4,5-Trichlorophenol	<0.9
Pentachlorophenol	<1.3 - 16
Lindane	<0.2 - 1.0
2,3',5-Trichlorobiphenyl	<0.1 - 0.7
2,2',4,4'-Tetrachlorobiphenyl	<0.2
Aldrin	<0.2 - 0.9
Phenanthrene	<0.2 - 1.7
Fluoranthene	<0.2
DDT	<0.2
Dieldrin	<0.4
Atrazine	<0.5
Simazine	<1.1
Phenylacetic acid	<0.7

[1]Concentration ranges for purgeable and non-purgeable target organic compounds.
[2]Chlorinated dual-media filtered effluent.

of many organic chemicals are much more reactive with chlorine in forming halogenated products than the parent compounds. Most of the chlorine demand of wastewaters is due to oxidation reactions with both organic and inorganic compounds and in reactions with nitrogenous materials.

Aqueous substitution reactions of chlorine with organic compounds which lead to stable carbon-chlorine bonds are quite limited in nature, and only one or two percent of the chlorine applied to a wastewater may end up incorporated into organic compounds [Jolley (1976)]. Three basic types of chlorine substitution reactions are observed in water and in order of their probable prevalence are: haloform reactions; chlorine substitution reactions with activated aromatic compounds; and chlorine addition to activated double bonds in unsaturated compounds [Richard (1980)].

In two field studies by Jolley (1973 & 1975), reaction yields (as chlorine) of nonvolatile, soluble chloro-organics were determined to be approximately one percent of the chlorine dosage applied to both primary and secondary sewage effluents after disinfection with 2-6 mg/l chlorine. Another study by Koczwara et al (1983), involving the breakpoint chlorination of wastewater effluents that had received activated sludge treatment, determined that chlorine incorporation into the organic matter was 0.019 to 0.067 mg Cl/mg C. It appears that most of the chlorine incorporated into organic constituents in wastewater is associated with constituents having a molecular weight less than 1000. Grady et al (1983) concluded that a major effect of chlorination of wastewaters is a reduction in the molecular size of constituents.

The production of THM compounds and possibly other halogenated volatile compounds appears to be the most widespread type of chlorine substitution reaction that occurs during water disinfection. Chloroform is one of the principal products and is generally formed in concentrations exceeding that of any other single identified halogenated product. Various haloform precursor compounds that may exist in water and wastewater may be divided into two groups. The first group includes methyl ketone compounds such as acetone and alcohols, acetaldehydes, and some unsaturated compounds which can be oxidized to a methyl ketone structure. The second group includes various aromatic compounds found in natural humic material which appear to be ubiquitous in water and wastewater.

Chlorination substitution reactions with activated aromatic compounds and aromatic nitrogen-containing compounds appear to be widespread in water but do not appear to take place to the same extent as the haloform reaction. Most aqueous chlorine substitution reactions with aromatic compounds occur at low pH and are limited by neutral pH in natural waters. However, many organic compounds found in water may contain activating substances which allow chlorination reactions to proceed at neutral pH. Wastewaters contain many aromatic compounds that potentially can react with chlorine to form chlorinated aromatic products. Included in these are phenols, aromatic aldehydes and carboxylic acids, phenol analogues such as cresols, hydroquinones, and anisoles, and many aromatic compounds found as components of aquatic humic material. Richard (1980) reported that the majority of aromatic compounds found in humic material contain activating substituents such as hydroxyl, ether, and methoxyl groups. Some of these compounds include benzoic acid, salicylic acid, hydroxybenzoic acid, vanillic acid, phenoxyacetic acid, and phthalic acid.

Numerous examples of chlorine substitution reactions with single ring aromatic compounds during water chlorination have been reported. Chlorine substitution into polynuclear aromatic hydrocarbons and biphenyl to form polychlorinated biphenyl compounds during water chlorination has also been reported by Harrison et al (1976) and Reinhard et al (1976). One of the most significant chlorination reactions that has been shown by Jolley (1974) to occur during chlorination of water is the formation of chlorinated pyrimidines, purines, pyrroles, and indoles.

Chlorine addition reactions occur to a limited extent with unsaturated fatty acids, with unsaturated sites on hydrocarbons, with unsaturated sites in the side chains of phenylpropanoid lignin units, and with unsaturated sites in humic material.

The chlorinated compounds are of particular concern because of the increasing awareness of the potential effects of halogenated hydrocarbons. Jolley et al (1982) state that a major portion of the mutagenicity associated with chlorinated wastewater is not associated with the active chlorine residuals but appears to be associated with constituents that are not reduced by thiosulfate. There is some information available concerning the toxicology of selected compounds that are by-products of wastewater chlorination. A wide variety of chlorinated organics are formed during the chlorination of wastewater, as indicated in Table 9.

Health Effects

Some of the organic constituents found in treated wastewater are known or suspected carcinogens, teratogens, or mutagens; however, most wastewaters have not been adequately characterized. Unfortunately, only a few studies have been performed which address the health effects of organic constituents identified in wastewater. Nellor et al (1984) identified several constituents in tertiary effluent that exceeded health-related levels. Therefore, it is likely that most secondary effluents will also have organic compounds that exceed health-related concentrations for drinking water. It is beyond the scope of this document to present health-related data on all of the potentially hazardous compounds. In addition, as the technology continues to improve and as more organic compounds are identified in wastewater, health-related research will undoubtedly expand the present list of known hazardous compounds.

Detailed data on many individual organic constituents which may be present in drinking water, and their associated health effects, are presented in four reports prepared by the National Research Council of the National Academy of Sciences [National Academy of Sciences (1977), (1980), (1982), & (1983)]. A general discussion of the presence of hazardous and potentially hazardous organic compounds in treated wastewater will suffice to illuminate the concerns of discharges of wastewater into watercourses used

Table 9

Chloro-Organic Constituents in Chlorinated Municipal Wastewater Effluents
[Jolley (1984)]

<u>Alcohols</u>

Chloromethylbenzyl alcohol
Dichloromethylbenzyl alcohol, 10 ppb
Trichloromethylbenzyl alcohol, 50 ppb

<u>Aliphatic hydrocarbons</u>

Chlorocyclohexane, 20 ppb
Chloroform
Chloromethylbutene, 285 ppb
Dibromochloromethane
Dichlorobutane, 27 ppb

<u>Amines</u>

N-Methyltrichloraniline, 10 ppb

<u>Amino acids</u>

Chlorotyrosine
Dichlorotyrosine

<u>Aromatic acids and esters</u>

Chlorobenzoic aicds, 0.3-1 ppb
Chlorohydroxybenzoic acid, 1 ppb
Chloromandelic acid, 1 ppb
Chlorophenylacetic acid, 0.4 ppb
Chlorosalicylic acid, 0.2 ppb
Tetrachlorophthalate
Trichlorophthalate

<u>Aromatic hydrocarbons</u>

Chlorocumene
Chloroethylbenzene, 21 ppb
Chlorotoluenes
Dichlorobenzenes, 10 ppb
Dichloroethylbenzene
Dichlorotoluene
Tetrachloromethylstyrene
Trichlorocumene
Trichloroethylbenzene, 20 ppb
Trichloromethylstyrene, 10 ppb

<u>Dibenzofurans</u>

Dichlorodibenzofuran

<u>Ethers</u>

Dichloromethoxytoluene, 32 ppb
Tetrachloromethoxytoluene, 40 ppb
Trichlorodimethoxybenzene
Trichloromethylanisole

<u>Ketones</u>

Hexachloroacetone, 30 ppb
Pentachloroacetone, 30 ppb
Tetrachloroacetone, 11 ppb

<u>Nitriles</u>

Trichloropropionitrile

<u>Nucleosides</u>

Chlorouridine, 2 ppb

<u>Phenols</u>

Chlorohydroxybenzophenone
Chloromethylphenol, 2 ppb
Chlorophenols, 0.5-2 ppb
Chlororesorcinol, 1 ppb
Tetrachlorophenol, 30 ppb
Trichlorophenol

<u>Purines</u>

Chloroaminopurine, 1 ppb
Chlorocaffeine, 2 ppb
Chloroxanthine, 2 ppb

<u>Pyrimidines</u>

Chlorouracil, 4 ppb

<u>Thiophenes</u>
Trichloroacetylthiophene

as sources of drinking water.

The World Health Organization (1975) has published a summary listing of over 1000 organic compounds found in wastewater, many of which have been found in domestic wastewater. Some of these compounds are toxic, and some of them have been identified as known or suspected carcinogens in a research report to Congress by the National Research Council of the National Academy of Sciences (1977).

The National Academy of Sciences (1980) reviewed 10 studies performed in the United States on cancer rates and water quality. Although it was not reported which studies included water that was partially derived from the indirect reuse of wastewater, the locations of the studies indicate that most of the cities were using water sources which were downstream of many effluent discharges. The report concluded that nine of the ten studies showed a number of associations, some of which were statistically significant, between indirectly characterized water quality and cancer rates (incidence of mortality). One, the Los Angeles County study, reported no associations, but this study appeared to have greater limitations than any of the others. It further concluded that "Cancer rates at several sites were positively associated with water quality in one or another study, but no site consistently predominated. The bladder, stomach, large intestine, and rectum, which were cancer sites identified in a number of geographic areas, warrant further study."

A report by Richard P. Arber Associates, Inc. (1985) concluded that the results of several bioassay studies indicated mutagenicity in organic fractions present in highly-treated wastewater. Microbiological in vitro studies on reclaimed water that have been reviewed by a panel of the National Academy of Sciences (1982) have all demonstrated a positive mutagenic effect in studies on reclaimed water. In many cases, techniques for analytical chemistry were unable to define the sources of mutagenicity, and complex relationships with an array of chemicals were suspected.

Some studies have been directed at the nonvolatile byproducts of water chlorination. Several studies summarized by Kool et al (1982) indicate that some nonvolatile fractions are mutagenic in Salmonella bacteria and in mammalian cells in vitro, cytotoxic, and carcinogenic in mice studies. Some of the organic halogens having known or suspected health effects that have been found in wastewater are chlorobenzene, trihalomethanes, trichloroethane, trichloroethylene, dichlorobenzene, and 1,1,2,2-tetrachloroethane [Okun et al (1985)].

Nellor et al (1984) found that several health-significant organic compounds in filtered and disinfected secondary effluent used for groundwater recharge were detected in well waters. Several reclaimed water (and storm water and imported water) organic residues were shown to be positive as potential mutagens using the Ames test, which is a short-term toxicity test using Salmonella bacteria as the test organism. In addition, the reclaimed water exceeded some of the EPA Water Quality Criteria and contained organohalides and epoxides, which contain many compounds considered to be potent mutagens.

The list of organic compounds identified in wastewater tends to be dominated by the simpler low molecular weight compounds that are more amenable to identification and quantification with presently available analytical techniques. All of the compounds that have been identified make up only a small fraction of the total organic carbon. Christman (1983) suggests that in the case of chlorinated organic species, nonvolatile chlorinated organics not normally measured with existing analytical techniques may be of greatest significance. Jolley et al (1982a) have reported a variety of nonvolatile mutagens in a number of domestic wastewater effluents. Most of the attention with respect to chlorinated organic species has been focused on the volatile compounds, particularly the trihalomethanes, which are discussed in greater detail in the following section.

The National Academy of Sciences (1982a) Panel on Quality Criteria for Water Reuse indicated that a focus on individual compounds is not an adequate basis for assessing health impacts in drinking water. The limitations associated with identifying all of the compounds which may be present and the requirement for assessing health impacts of complex mixtures of compounds are cited as problems which are not completely addressed with conventional toxicological approaches. Synergistic effects are of particular concern and are feared to be important, since two or more compounds which by themselves may present minimal health effects may, in combination, produce a significant health effect. Consequently, many uncertainties persist and simplistic approaches that rely on identification of specific compounds and associated health do not yield an adequate basis for evaluation. Therefore, assessments of measures for protection of drinking water sources must include consideration of controlling wastewater discharges to minimize human exposure to contaminants.

Trihalomethanes

Trihalomethane (THM) compounds are derivatives of methane where 3 of the 4 hydrogen atoms have been replaced by either chlorine, bromine, or iodine. Although ten possible trihalomethane compounds exist, only five have been detected after chlorination of drinking water: chloroform ($CHCl_3$); bromoform ($CHBr_3$); bromodichloromethane ($CHCl_2Br$); dibromochloromethane ($CHClBr_2$); and dichloroiodomethane ($CHCl_2I$). Apparently, the iodinated compound occurs infrequently. The California Department of Health Services has set a maximum contaminant level (MCL) of 100 ug/l for trihalomethanes in drinking water. Two major reasons for setting this MCL are the ubiquity of measurable concentrations of THM compounds in chlorinated water supplies and the concern for toxicity and mutagenicity of chlorinated organic compounds. More specifically, the MCL was based on animal studies which were used to make risk assessments. The risk estimates associated with the total trihalomethane MCL of 100 ug/l predict an incremental risk of three to four excess cancers per 10,000 population consuming 2 liters of water containing 100 ug/l of chloroform daily for 70 years.

Health Effects of THMs

In any assessment of risks from by-products of wastewater chlorination, the public health risk from consumption of trace quantities of the trihalomethanes must be weighed against the benefits for public health in reducing microbiological contamination. Most of the health effects data developed to date relate to chloroform, and very little information is available concerning the health effects of the other trihalomethanes. People are exposed to chloroform and other trihalomethanes from air, drinking water, and food. Symons et al (1981) reported that chloroform has been shown to be rapidly absorbed on oral and intraperitoneal administration and to be subsequently metabolized to carbon dioxide, chloride ion, phosgene, and other unidentified metabolites. The metabolic profile of chloroform in animal species such as mice and rats is qualitatively similar to that in humans.

Mammalian responses to chloroform exposure include the following: central nervous system depression, hepatoxicity, nephrotoxicity, teratogenicity, and carcinogenicity. These responses have been demonstrated in mammals after oral and inhalation exposures to high levels of chloroform ranging from 30 to 350 mg/kg of body weight, with the intensity of response being dependent upon the dosage [Symons et al (1981)]. There is some evidence that chloroform is a promotor, rather than an initiator of animal carcinogenesis [Pereira et al (1982), Deml and Oesterle (1985)]. Initiation is defined as an irreversible manifestation of genetic damage, where promotion is the process whereby initiated cells are stimulated to evolve into cancerous cells.

Although less toxicologic information is available for the brominated THMs, mutagenicity and carcinogenicity have been detected in some test systems. A recent analyses of data by Crouch et al (1983) indicated that, in most cases tested, the brominated analogues are more potent carcinogens than their chlorinated counterparts. Morimoto and Koizumi (1983) found that bromoform was 100 times more potent than chloroform in inducing sister chromatid exchanges in human lymphocytes in vitro. In cultured lymphocytes, the potency of the trihalomethanes toward induction of chromosome aberrations was $CHBR_3$ > $CHClBr_2$ > $CHCl_2Br$ > $CHCl_3$. In general, bromomethanes have a greater mutagenic potential than chloromethanes.

Under the Guidelines for Chemical Carcinogen Risk Assessments and Their Scientific Rationale (State of California, Department of Health Services, 1985), "positive evidence for carcinogenicity from properly conducted bioassays in two species of animals, or two properly conducted bioassays in the same species carried out at separate times in the same laboratory, or preferably in two separate laboratories, is considered by DHS and by IARC as sufficient evidence for carcinogenicity in animals. This, in turn, is considered as sufficient evidence for potential human carcinogenicity."

To date, only chloroform can be considered to be an animal carcinogen using the criteria outlined in the above-mentioned guidelines. Chlorodibromomethane was associated with a significant increase in hepatocellular carcinoma in mice, but no evidence for a significant carcinogenic response was found in rats. Bromodichloromethane was associated with a significant increase in female rat hepatic neoplastic nodules, but a determination of the carcinogenic potential in mice and confirmation of the response in rats awaits publication of the results of the ongoing National Toxicity Program (NTP) chronic toxicity and carcinogenesis bioassay. Bromoform is currently under NTP bioassay and there are currently no data from which to derive a carcinogenicity risk estimate. Finally, there are no data in the open literature from which to derive appropriate interspecies scaling factors, except in the case of chloroform.

Significant positive correlations have been reported between the concentrations of chloroform or other chlorinated organics in drinking water and cancers of the large intestine, rectum, and bladder [Kuzma et al (1977), Cantor et al (1978), Hogan et al (1979), Young et al (1981), Gottlieb et al (1981)]. Case-control and cohort appproaches have shown a positive correlation between chloroform exposure and human gastrointestinal and bladder cancers [Wilkins and Comstock (1981)]. To date, a correlation between chloroform exposure and cancer incidence in humans has not been clearly established.

While human epidemiological evidence is inconclusive, Symons et al (1981) reported that several studies have found positive associations between THMs and some cancer sites. Retrospective studies have investigated the relationship between cancer mortality or morbidity and drinking water variables. Because of various limitations in the epidemiologic methods, difficulties with the water quality data, and problems with the individual studies, the present evidence does not lead to a firm conclusion that an association exists between contaminants in drinking water and cancer mortality or morbidity. Therefore, causal relationships cannot be proven on the basis of results from epidemiological studies. However, when viewed collectively, the epidemiological studies provide sufficient evidence for maintaining the hypothesis that a health risk may be occurring and that the positive relationships may be reflecting a causal association between constituents of drinking water and cancer mortality.

A report by Crump and Guess (1980) commissioned by the President's Council on Environmental Quality concluded that recent case-control studies "strengthened the evidence for an association between rectal, colon, and bladder cancer and drinking water quality provided by the earlier studies reviewed by the NAS committee." According to the report, these studies found the cancer risk for the target organs associated with chlorinated water to be 1.1-2.0 times higher than the cancer risk in individuals consuming unchlorinated water.

THM Concentration in Wastewater

The concentration and species of trihalomethanes formed in wastewater are highly variable, depending mainly on the pH, temperature, chlorine dose, contact time, and concentrations of precursors and bromine. For example, one recent study by Nellor et al (1984) in Los Angeles County found that chloroform was the most prevalent THM present after chlorination, ranging from 4.9 to 84 ug/l for dechlorinated tertiary-treated effluent. Bromodichloromethane, dibromochloromethane, and bromoform were also found in the treated wastewaters, but generally at lower concentrations than chloroform (Table 10).

A more detailed study of chlorinated wastewater from nine San Francisco Bay area wastewater treatment plants by Richard (1980a) indicated that relatively high THM concentrations were formed, ranging from 60 to 400 ug/l, with an average of 140 ug/l. Several of the wastewater treatment plants are known to receive large quantities of industrial wastes, such as petrochemical wastes, dry cleaning wastes, and electrochemical industry wastes. Therefore, the high THM concentrations at some of the plants may have been due to contamination from industrial wastes rather than chlorination reactions.

Almost all of the unchlorinated and chlorinated samples examined during the study by Richard (1980a) contained bromine-substituted THM compounds in higher concentrations than chloroform, indicating saline water infiltration. Normal wastewater disinfection was practiced during the study, e.g., a chlorine dosage of approximately 20 mg/l resulting in 5-10 mg/l combined chlorine residuals after one hour reaction time. Omitting one sample taken during a chlorine spill, the increase in THM concentration compared to unchlorinated samples was from 1.2 to 145-fold. THM concentrations in chlorinated treated wastewaters reported by Glaze et al (1973), Bellar et al (1974), Baird et al (1979), and Oliver and Lawrence (1979) are lower than those observed by Richard (1980a), generally 50 ug/l or less, when combined chlorine residuals were used.

The study by Baird et al (1979), which employed combined residual chlorination, indicated that 5 to 6-fold increases in chloroform concentration occurred during chlorination of filtered secondary effluent. The chloroform concentrations after disinfection ranged from 0.5 to 9.1 ug/l. Low dosages of alum and polymer (5 ppm and 0.06 ppm, respectively) were added prior to filtration, and the chlorine residual was 5-10 mg/l after a contact time of 2 hours. This treatment scheme was designed to meet an effluent total coliform requirement of 2.2/100 ml. The same treatment scheme with a nitrified effluent and a free chlorine residual of 4 mg/l after 2 hours of contact time resulted in a 330-fold increase in chlorine concentration during chlorination - from 0.6 to 198 ug/l. The authors stated that the 6-fold increase during combined residual chlorination exceeds that cited by some other investigators and suggested that it may be indicative of short-lived existence of free residual chlorine. Haloforms and chloramines may be formed under these conditions in competing fast reactions between chlorine and either ammonia or haloform precursors. They concluded that the most important factor controlling THM increases seems to be the amount of free chlorine residual available during disinfection.

Richard (1980a) indicated that there was nearly a linear relationship between chlorine dose and THM formation in wastewater at the sewage treatment plants he studied in the San Francisco Bay area. This assumes that the concentration of organic precursors is not limiting. Trussell and

Table 10

Trihalomethane Concentrations in Treated Wastewater
[Nellor et al (1984)]

	Trihalomethane Concentration in ug/l			
	$CHCl_3$	$CHCl_2Br$	$CHClBr_2$	$CHBr_3$
San Jose Creek WRP				
2^o-$UnCl_2$	2.6, 4.6	<0.1, <0.1	<0.1, 0.3	<0.1, 0.6
2^o-Cl_2	7.7, 36.5	<0.1, <0.3	<0.1, <0.2	<0.1, 0.2
3^o-$DeCl_2$	7.1 - 16	0.1 - 1.7	<0.1 - 0.5	<0.1 - 4.8
Whittier Narrows WRP				
2^o-$UnCl_2$	3.2	<0.1	0.2	7.8
2^o-Cl_2	8.8	1.6	0.8	0.4
3-$DeCl_2$	4.9 - 84	0.6 - 1.8	0.6 - 1.2	0.2 - 0.8
Pomona WRP				
3^o-$DeCl_2$	5.8, 8.2	2.1, 2.9	1.5, 2.1	0.3, 1.4
L.A./Glendale WRP				
3^o-Cl_2	5.1 - 7.9	3.2 - 7.6	2.9 - 6.6	0.9 - 3.8

Note: 2^o-$UnCl_2$ = undisinfected secondary effluent; 2^o-Cl_2 = disinfected secondary effluent; 3^o-Cl_2 = disinfected tertiary effluent; and 3^o-$DeCl_2$ = chlorinated/dechlorinated tertiary effluent.

Umphres (1978) evaluated the effect of chlorine dosage on THM formation. Different amounts of chlorine (0-80 mg/l) were added to a synthetic water containing 10 mg/l of humic acid and 1 mg/l of NH_3Cl (as NH_3). The chlorine contact time was 2 hours. There was a small region at the beginning of the curve where THMs were inefficiently formed while the NH_3 was being oxidized. This was followed by a region of rapid THM development. After a dosage of about 20 mg/l a substantial chlorine residual was developed, but THM development was curtailed. Although this study was not done with wastewater, it did indicate that some THMs were formed at low chlorine dosages before breakpoint chlorination occurred and a free residual was formed. This supports the premise that the short-lived existence of free chlorine may cause THMs to be formed even though the measured residual may all be in the combined form.

It is important to note that the chlorine-organic reactions that form trihalomethanes are time-dependent and that THM formation can continue in chlorinated waters for extensive time periods. It is not surprising, then, that dechlorinated effluents have been shown to produce much lower levels of THMs than effluents that have not been dechlorinated. The wide range of THM concentrations reported by various investigators is likely the result of utilizing different chlorine contact times and, therefore, only represent the THM concentrations at the particular time of analysis. Hence, the widely differing results reported in the literature are not, in most cases, truly indicative of the ultimate potential for trihalomethane production.

The total trihalomethane formation potential (THMFP) is a laboratory test using very high chlorine concentrations to determine the ultimate potential that a water has for producing THMs upon disinfection. The THMFP is much higher than the actual amount of THMs formed during usual disinfection practices, but does give a measure of the potential of the organic matter present to produce THMs. As an example, in one study by McCarty et al (1982) at Water Factory 21 in Orange County, the trihalomethane concentration of the influent wastewater, which is secondary-treated effluent, was an average of 3.5 ug/l, while the THMFP averaged 296 ug/l. Similarly, after extensive tertiary treatment, including lime coagulation, air stripping, granular activated carbon, chlorination, and reverse osmosis, the average THM concentration was 1.8 ug/l, while the THMFP was 94 ug/l.

Very high maximum THM formation has been observed by Richard (1980a) in East Bay MUD primary and secondary effluents that were chlorinated such that a free residual was maintained for seven days. Under those conditions, the THMFP was found to be 550 ug/l and 1000 ug/l in the primary and secondary effluents, respectively. That same study determined the THMFP to be 235 ug/l for Las Gallinas trickling filter effluent and 475 for Richmond activated sludge effluent.

White (1978) described the results of a comprehensive study conducted on effluent from the Rancho Cordova sewage treatment plant. An activated sludge effluent that contained 20 ug/l of ammonia nitrogen and was chlorinated for disinfection purposes (3-4 mg/l residual after 30 minutes) formed 4 ug/l of chloroform in 4 hours, whereas the same effluent subjected to breakpoint chlorination with chlorine dosages of 200 mg/l or more and resulting in free chlorine residuals of 7-12 mg/l yielded 88 ug/l of chloroform in 2 minutes and 123 ug/l of chloroform in 5 minutes. When the same activated sludge effluent was nitrified, filtered, and disinfected with a chlorine dose of 11 mg/l, approximately 12 ug/l of chloroform was formed. The 3-fold increase over that found in activated sludge effluent was attributed to the formation of free chlorine residuals in the highly polished effluent.

Effects on Receiving Waters

Organic constituents in wastewater discharged to receiving waters are a public health concern when receiving waters are subsequently used as a source of potable supply. Typically, water treatment facilities are designed to adequately remove microbiological contaminants, not trace organic constituents. Conventional water treatment does remove organics to some degree, e.g., trihalomethane formation potential is typically reduced by 50 percent. Because of the known organic contaminants present in treated wastewater and the unknown health significance of largely unidentified fractions, it is prudent to discourage discharges to waters ultimately used as drinking water. However, where this cannot be avoided, such discharges may be acceptable if the potential adverse health impacts of organic constituents can be mitigated by wastewater treatment, natural purification processes (in-stream degradation or conversion of organics, volatilization, etc.), or dilution.

It would appear that volatile chlorinated organic constituents are less of a concern than nonvolatile organics in treated wastewater. For example, because of the long distances usually encountered in California between the introduction of wastewater into watercourses and the subsequent withdrawal of water, THMs produced upon disinfection of wastewater are likely to be reduced in concentration by volatilization in surface streams and rivers prior to water treatment. Therefore, an argument could be made that it may actually be advantageous to chlorinate the wastewater in order to form volatile organics that will subsequently be expunged from the water, thereby reducing the concentration of precursors that could react with chlorine at downstream water treatment plants. Unreacted precursors, other nonchlorinated organic constituents, and nonvolatile chlorinated organic constituents are a concern, however, and must be considered in such discharge situations.

It is difficult to predict and evaluate the organic constituent input resulting directly from wastewater discharges. Natural waters themselves contain many organic constituents and THM precursors. For example, although there are many wastewater discharges into the Sacramento River, their impact on trihalomethane formation potential appears to be minimal. The results of two studies summarized by Okun et al (1985) indicated that the THMFP of the American River at Nimbus Dam was 62 and 65 ug/l, while it was 78 and 85 ug/l in the Sacramento River at Greene's Landing, which is below the Sacramento Regional County Sanitation District's 150 MGD discharge of secondary effluent. It is interesting to note that the THMFP of Delta water at Clifton Court is almost twice as high (137 and 171 ug/l) as in the Sacramento River at Greene's Landing, probably due in part to higher bromine levels in that water and the additional precursors picked up in varying amounts.

A similar study of the trihalomethane formation potential was conducted by the California Department of Water Resources (1985). A different procedure was used to determine the THMFP, which resulted in consistently higher values than the aforementioned studies, but the results indicated similar relative differences in THMFPs.

Nelson and Khalifa (1980) examined raw water samples at nine locations in the Sacramento River system (from Little Castle Creek to the Sacramento River at Sacramento), the Delta, and the aqueduct systems that transport water from the Delta. They found that THM concentrations in the raw water were generally low, i.e., less than 5 ug/l, except for two samples taken during the drought conditions of 1977 that were approximately 17 ug/l (central Delta at Rock Slough). The authors concluded that the Sacramento River system, into which several sewage treatment plants discharge their effluent, does not contain significant precursors until the water reaches the delta, where additional precursors were picked up in varying amounts. They also found no apparent relationship between COD or TOC and THM formation.

Hoehn et al (1977) studied the Occoquan watershed collection system in Virginia, which receives both agricultural and urban runoff as well as treated sewage from 11 sewage treatment plants. They concluded that chloroform concentrations in surface waters do not necessarily have their origin in chlorinated watewater discharges. The upstream control on Bull Run had a mean chloroform concentration of 2.2 ug/l, while at Bull Run 2.3 miles below the last of the 11 sewage treatment plant discharges the mean chloroform concentration was 3.2 ug/l. The dilution was approximately 10-fold. Chloroform accounted for 90 percent or more of the total trihalomethanes in every sample. $CHBrCl_2$ constituted the majority of the remainder, and $CHBr_3$ and $CHBr_2Cl$ were seldom detected.

REFERENCES

Baird, R., et al. (1979). "Analysis of Selected Trace Organics in Advanced Wastewater Treatment Systems." Water Research, 13, 493.

Bellar, T.A., Lichtenberg, J.J., and Kroner, R.C. (1974). "The Occurrence of Organohalides in Chlorinated Drinking Waters." Jour. AWWA, 66, 703.

California Department of Health and Cooper, R.C. (1975). "Wastewater Contaminants and Their Effect on Public Health." In: A State-of-the-Art Review of Health Aspects of Wastewater Reclamation for Groundwater Recharge. pp. 33-95, California Department of Water Resources, Sacramento, CA.

California Department of Water Resources (1985). "Water Quality at Three Potential Diversion Sites - American River, Sacramento River, and Clifton Court Forebay." Report Prepared by the Central District, Department of Water Resources, Sacramento, CA.

Cantor, K.P., et al. (1978). "Associations of Cancer Mortality with Halomethanes in Drinking Water." Jour. Nat. Cancer Inst., 61, 979-985.

Christman, R.F. (1983). "Contaminants in Drinking Water: What and How We Know." Paper presented at the AWWA Health Effects Seminar, Denver, CO.

Crouch, E.A.C., Wilson, R., and Zeise, L. (1983). "The Risks of Drinking Water." Water Res. Research, 19, 1359-1375.

Deml, E., and Oesterle, D. (1985). Dose-dependent Promoting Activity of Chloroform in Rat Liver Foci Bioassay." Cancer Lett., 29, 59-63.

Glaze, W.H., et al. (1973). "Analysis of Organic Materials in Wastewater Effluents After Chlorination." Jour. Chromat. Sci., 11, 580.

Gottlieb, M.S., Carr, J.K., and Morris, D.T. (1981). "Cancer and Drinking Water in Louisiana: Colon and Rectum." Int. J. Epidemiol., 10, 117-125.

Grady, C.P.L., et al. (1984). "Molecular Weight Distribution in Activated Sludge." Water Res., 18(2), 293-246.

Harrison, R.M., Perry, R., and Wellings, R.A. (1976). "Effect of Water Chlorination Upon Levels of Some PAH in Water." Environ. Sci. Technol., 10, 1151.

Hoehn, R.C., et al. (1977). "Trihalomethanes and Viruses in a Water Supply." Jour. Environ. Engrg. Div., ASCE, 103(EE5), 803.

Hogan, M.D., et al. (1979). "Association Between Chloroform Levels in Finished Drinking Water Supplies and Various Site-specific Cancer Mortality Rates." J. Environ. Pathol. Toxicol., 2, 873-887.

Hunter, J.V. (1971). "Origin of Organics from Artificial Contamination." In: S.D. Faust and J.V. Hunter (Eds.), Organic Compounds in Aquatic Environments. Marcel Dekker, Inc., New York, NY, p. 51.

Jolley, R.L. (1973). "Chlorination Effects on Organic Constituents in Effluents from Domestic Sanitary Sewage Treatment Plants." ORNL/TM-4290, Oak Ridge National Laboratory, Oak Ridge, TN.

Jolley, R.L. (1975). "Chlorine-Containing Organic Constituents in Sewage Effluents." Jour. WPCF, 47, 601-618.

Jolley, R.L., (Ed.) (1976). Proceedings of the Conference on the Environmental Impacts of Water Chlorination. Oak Ridge National Laboratory, Oak Ridge, TN.

Jolley, et al. (1982). "Micropollutants Produced by Disinfection of Wastewater Effluents." Water Sci. Technol., 14, 45-59.

Jolley, et al. (1982a). "Nonvolatile Organics in Disinfected Wastewater Effluents: Chemical Characterization and Mutagenicity." EPA-600/2-82-017, U.S. Environmental Protecton Agency, Washington, D.C.

Jolley, R.L. (1984). "Chemistry and Fate of Disinfectant By-Products." In: Wastewater Disinfection - The Pros and Cons, Preconference Workshop, 57th Annual WPCF Conference, New Orleans, LA.

Kool, H.J., et al. (1982). "Presence, Introduction and Removal of Mutagenic Activity During the Preparation of Drinking Water in the Netherlands." Environ. Health Persp., 46, 207-214.

Koczwara, M.K., Kirsch, E.J., and Grady, C.P.L. (1983). "Formation of Organic Chlorine in Activated Sludge Effluents." Water Res., 17(2), 1863-1883.

Kowal, N.E. (1981). "An Overview of Public Health Effects." Paper presented at the Workshop on Utilization of Municipal Wastewater and Sludge on Land, Denver, CO.

Kuzma, R.J., Kuzma, C.M., and Buncher, C.R. (1977). "Ohio Drinking Water Source and Cancer Rates." Am. J. Public Health, 67, 725-729.

Lauer, W.C., Rogers, S.E., and Roy, J.M. (1984). "Denver's Potable Reuse Project - Current Status." In: Proceedings of Water Reuse Symposium III, Vol. 1, pp. 316-336, AWWA Research Foundation, Denver, CO.

Lawrence, C.E., et al. (1984). "Trihalomethanes in Drinking Water and Human Colorectal Cancer." Jour. Nat. Cancer Inst., 72, 563-568.

Majeti, V.A., and Clark, C.S. (1982). "Health Risks of Organics in Land Application." Jour. Envir. Engrg. Div., ASCE, 107(EE2), 339-357.

Manka, J.M. et al. (1974). "Characterization of Organics in Secondary Effluents." Envir. Sci. & Technol., 8, 1017-1020.

McCarty, P.L., et al. (1982). "Advanced Treatment for Wastewater Reclamation at Water Factory 21." Technical Report No. 21, Department of Civil Engineering, Stanford University, Palo Alto, CA.

Morimoto, K., and Koizumi, A. (1983). "Trihalomethanes Induce Sister Chromatid Exchanges in Human Lymmphocytes in vitro and Mouse Bone Marrow Cells in vivo." Environ. Res., 32, 72-79.

National Academy of Sciences (1977). Drinking Water and Health, National Academy Press, Washington, D.C.

National Academy of Sciences (1980). Drinking Water and Health - Volume 3, National Academy Press, Washington, D.C.

National Academy of Sciences (1982). Drinking Water and Health - Volume 4, National Academy Press, Washington, D.C.

National Academy of Sciences (1982a). Quality Criteria for Water Reuse, National Academy Press, Washington, D.C.

National Academy of Sciences (1983). Drinking Water and Health - Volume 5, National Academy Press, Washington, D.C.

Nellor, M.H., Baird, R.B., and Smyth, J.R. (1984). "Health Effects Study-Final Report." County Sanitation Districts of Los Angeles County, Whittier, CA.

Nelson, S.J., and Khalifa, S. (1980). "Trihalomethane Levels in Public Water Supplies Drawn From the Sacramento River System." Jour. AWWA, 72(7), 423.

Okun, D.A., Harris, R.H., and Tardiff, R. (1985). "Water Quality Considerations in Source Selection." Report Prepared for the East Bay Municipal Utility District, Oakland, CA.

Oliver, B.G., and Lawrence, J. (1979). "Haloforms in Drinking Water: A Study of Precursors and Precursor Removal." Jour. AWWA, 71, 161.

Pereira, M.A., et al. (1982). ""Trihalomethanes as Initiators and Promoters of Carcinogenesis." Environ. Health Perspect., 46, 151-156.

Reinhard, M., Drevenkar, V., and Giger, W. (1976). "Effect of Aqueous Chlorination on the Aromatic Fraction of Diesel Fuel - Analysis by Computer-Assisted GC-MS." J. Chrom., 116, 43.

Richard, M.G. (1980). "Trihalomethane Formation from Chlorine-Humic Material Interactions in Water and Wastewater - A Literature Review." UCB/SERL Report No. 80-6, Sanitary Engineering Research Laboratory, University of California, Berkeley, CA.

Richard, M.G. (1980a). "Trihalomethane Formation from Humic Materials in Chlorinated Reclaimed Water." Ph.D. Dissertation, University of California, Berkeley, CA.

Richard P. Arber Associates, Inc. (1985). "Human Considerations of Water Supplies." Technical Memorandum No A1, Richard P. Arber Associates, Inc., Denver, CO.

Symons, J.M., et al. (1981). "Treatment Techniques for Controlling Trihalomethanes in Drinking water." EPA-600/2-81-156, U.S. Environmental Protection Agency, Cincinnati, OH.

Trussell, R.R., and Umphres, M.D. (1978). "The Formation of Trihalomethanes." Jour. AWWA, 70, 604.

Westcot, D.W., and Ayers, R.S. (1984). "Irrigation Water Quality Criteria." In: G.S. Pettygrove and T. Asano (Eds.), Irrigation with Reclaimed Municipal Wastewater - A Guidance Manual, pp.3-1 - 3-37, Report No. 84-1 wr, California State Water Resources Control Board, Sacramento, CA.

White, G.C. (1978). Disinfection of Water and Wastewater for Reuse. Van Nostrand Reinhold Co., New York, NY.

Wilkins, J.R. III, and Comstock, G.W. (1981). "Source of Drinking Water at Home Site-specific Cancer Incidence in Washington County, Maryland." Am. Jour Epidemiol., 14, 178-190.

World Health Organization (1975). "Health Effects Relating to Direct and Indirect Reuse of Wastewater for Human Consumption." Technical Paper Series No. 7, WHO International Reference Center for Community Water Supply, World Health Organization, Geneva.

Young, T.B., Kanarek, M.S., and Tsiatis, A.A. (1981). "Epidemiologic Study of Drinking Water Chlorination and Wisconsin Female Cancer Mortality." Jour. Nat. Cancer Inst., 67, 1191-1198.

Zemansky, G.M. (1973). "Removal of Trace Metals During Conventional Water Treatment." Ph.D. Thesis, University of Colorado, Department of Civil Engineering, Boulder, CO.

APPENDIX A

WASTEWATER DISINFECTION

PUBLIC HEALTH PERSPECTIVE

WASTEWATER DISINFECTION: PUBLIC HEALTH PERSPECTIVE

The review report and related information regarding the specific issues associated with wastewater disinfection were critically reviewed by a Health Policy Committee of public health authorities with expertise and long experience in environmental health, wastewater disinfection, waterborne disease, and pathogenic agents. The committee members then provided their recommendations on wastewater disinfection guidelines.

It should be understood that the guidelines currently in use take into account the beneficial uses of the receiving waters, amount of dilution, and other factors in determining the disinfection requirement to be recommended to regional water quality control boards. In some instances, such as ocean discharges which are remote from use areas, no disinfection may be recommended. In other situations involving recreational use of wastewater with little dilution, a very high degree of treatment and disinfection is recommended. For discharges to inland waters where dilution is available, the typical recommended disinfection requirement is a median total coliform MPN of 23/100 ml. This significantly differs from the former EPA criteria of a median fecal coliform MPN of 200/100 ml for disinfected secondary effluent.

The committee members strongly supported the currently employed California disinfection guidelines than the former EPA criteria for health protection. The members recommended that the report include a separate presentative which brought together pertinent information and focused on the needs for the current guidelines. This portion of the report is intended to carry out the recommendation. Several specific examples offered by the committee members have been incorporated herein, and they provided additions to the review report writeup.

There are a number of limitations associated with the collection, identification, and enumeration of pathogenic agents in wastewater and receiving waters which tend to underestimate the types, concentrations, and presence of the organisms. Also, as the review report documents, where more rigorous studies have been undertaken, increased incidence of illness has been detected. The sum total of recognized concerns and uncertainties associated with pathogens and waterborne disease support a conservative disinfection stand.

Significant factors involved in the general paucity of information regarding the presence of pathogenic agents include the cost and complexity of analytical test procedures and, in some cases, the lack of test methods. There are approximately 120 serologically distinct enteric viruses, including the enteroviruses, adenoviruses, reoviruses, rotoviruses, and Norwalk-type viruses. Others, such as coronaviruses, astroviruses, and "small round" viruses, also may be waterborne (Sobsey 1984). As pointed out in Standard Methods 1985, a variety of "tentative" concentration, isolation, and assay procedures may be needed for the identification and enumeration of various viruses. The lack of convenient test methods for several suspected waterborne viruses has resulted in little or no information regarding their presence and numbers in the aquatic environment.

Similarly, identification of pathogenic bacteria in the water environment is hampered by available test procedures. Shigella test methodology is limited in sensitivity and provides qualitative results. Standard Methods 1985, cites 1700 Salmonella serotypes (Sobsey 1984 puts the number at 2,200) which can infect humans. As cautioned in Standard Methods, "...a negative result by any of these (test) methods does not imply the absence of salmonellae...".

Several pathogenic agents under certain circumstances have been found to be viable but nonculturable by conventional test methods. These include Vibrio cholera, Salmonella enteritidis, enteropathogenic Escherichia coli, and Campylobacter jejuni. In one experiment, Rollins 1986, found greater than 10^6 viable Campylobacter jejuni organisms per milliliter by direct microscopic methods after incubation in a streamwater stationary microcosm incubation for ten days, despite the fact that the cells could not be cultured by spread plating. It was suggested that the phenomenon exhibited by Campylobacter jejuni may be a survival strategy undertaken by bacteria in response to adverse environmental conditions and may be fundamental in microbial ecology.

An investigation by McFeters et al. (1985), found that "injured" total coliform bacteria in three domestic water supply systems did not grow on the accepted media used in the analysis of drinking water. It was found that the conventional media would enumerate less than ten percent of the viable coliform bacteria population that could be grown on improved media. These factors of cost and analytical capability tend to result in erring on the low side with regard to the presence and density of pathogens and indicator organisms in water.

There have been relatively recent study findings which challenge former conventional health precepts. Volunteer feeding studies involving known doses of Salmonella types were carried out and reported in the early 1950s. The volunteers were healthy adult males, and dosages on the order of a million organisms were necessary to cause clinical illness in half the volunteers. The results of these and other feeding studies of pathogenic agents were generally cited to demonstrate the relatively large doses required for illness and the improbability that the concentrations involved would be present in surface waters. Later investigations, principally of food-borne outbreaks, have indicated that the infecting dose of certain pathogenic agents, including salmonella strains, is lower than the volunteer feeding studies had indicated. The newer data raise the question of the susceptibility of the general population ~~represented by the outbreaks as compared to~~ a selected volunteer test group. As was pointed out in the review report, due to life styles, medical advances, and other factors, there is an increasing portion of the population with defects in mechanical or physiologic barriers to infection and an increased number of immunocompromised individuals. Defective barriers may include the diminution of gastric secretions and competing microflora. Immunocompromised individuals include patients with liver disfunction, diabetes, ulcers, and underlying cancers. The elderly may have waning immunity, and the very young have incomplete immune system development.

Conditions enhancing present-day stress (crowding, noise, traffic, smog, and other physiological or environmental factors) have been cited as tending to reduce individual health to "a body waiting for disease to happen".

New types of pathogenic agents continue to emerge. Salmonella agana was isolated only twice in man prior to 1970. By 1976, it ranked third among the most frequently isolated serotypes from human sources, and the infecting dose is thought to be low.

Campylobacter jejuni was not included as a waterborne pathogenic bacteria in the California State Water Resources Control Board publication "Water Quality Criteria" of 1963. It has now been implicated in waterborne outbreaks in Asia, England, Europe, and the United States. A tentative test procedure was first included in "Standard Methods" in 1985. According to Sobsey 1984, Campylobacter jejuni and closely related organisms "...are now recognized as the most frequent bacterial agents of acute gastroenteritis in U. S." Similarly, Yersinia enterocoliticia has only recently been recognized as a potentially significant bacterial waterborne pathogen which has possibly been the cause of two outbreaks of gastroenteritis in the United States.

Little is known regarding the presence, density, or survival of either Campylobacter or Yersinia in wastewater or natural waters.

The significance of Giardia lamblia as a waterborne disease agent has been recognized fairly recently. One survey has shown Giardia lamblia to be the most common parasite of man in the United States infecting approximately 3.8 percent of the populace with prevalence ranging from 2 to 20 percent (Davies 1979). The organism, along with waterborne viruses, are principal reasons for consideration of a national requirement for complete water treatment of surface waters used as a source of domestic water supply. While Giardia lamblia is resistant to disinfection, the concentrations in wastewater can be reduced by effective wastewater disinfection.

Perhaps the most significant findings in support of effective wastewater disinfection are those associated with studies of recreational water use and shellfish cultivation in other areas of the country. The U. S. EPA detected an unexpected number of added cases involving "highly credible" gastrointestinal symptoms associated with swimming in waters of the east, midwest, and south, which met current bacteriological water quality objectives. At the current degree of protection afforded by an objective of 200 fecal coliform per 100 milliliters, it was calculated that the added GI illnesses were 8 per 1,000 swimmers in fresh water and 19 per 1,000 swimmers in marine water.

Pathogen densities were not studied, and the etiologic agent(s) were not determined; however, it was suspected that viral agents, perhaps Norwalk-type viruses and rotoviruses, may have been involved.

Instances of outbreaks of viruses and bacteria-caused illnesses implicating shellfish grown in areas of the east and south, which apparently met current bacteriological standards, are included in the review report. Also, the systematic collection of epidemiological data undertaken in New York State in

1982 documented an epidemic (103 outbreaks) of gastroenteritis associated with shellfish consumption.

The growing concern with shellfish-associated disease outbreaks has triggered an EPA study of shellfish growing waters and shellfish meats similar in nature to the recreational waters study.

Future EPA plans call for an investigation of bacteriological indicator systems and water consumption reflecting concern with the issues raised earlier here and in the review report regarding uncertainties with indicator organisms and pathogens in drinking water supplies.

Finally, the separation in time and distance of wastewater discharge and water use has diminished and will continue to do so. The institution of effective wastewater disinfection provides a barrier to waterborne disease agents prior to their release to the environment and minimizes impacts on water treatment plants, recreational areas, and shellfish growing waters.

As pointed out by Olivieri 1984, in summarizing his review of current conditions: "Waterborne disease is alive and well in the United States". The public health view in California is that the potential for waterborne disease exists, and effective wastewater disinfection continues to be needed for the protection and preservation of the public health.

REFERENCES

Davies, Robert B., and Chas. P. Hibler (1979). Animal Reservoirs and Cross-Species Transmission of Giardia In: Waterborne Transmission of Giardiasis. EPA-600/9-79-001.

Feachem et al. (1983). Sanitation and Disease -- Health Aspects of Excreta and Wastewater Management. World Bank, John Wiley & Sons, New York.

Olivieri, V. P. (1984). Epidemiology in Wastewater Disinfection -- the Pros and Cons. Proceedings Wastewater Disinfection Committee, Water Pollution Control Federation.

Rollins, D. M. and R. R. Colwell (1986). Viable But Nonculturable Stage of *Campylobacter jejuni* and Its Role in Survival in the Natural Aquatic Environment. Applied and Environmental Microbiology, Vol. 52, No. 3.

Sobsey, M. D. (1984). Direct Measurement of Pathogens and Its Role in Assessing Wastewater Disinfection Practices in Wastewater Disinfection -- the Pros and Cons. Proceedings Wastewater Disinfection Committee, Water Pollution Control Federation.

Standard Methods for the Examination of Water and Wastewater (1985). APHA, AWWA, WPCF.

APPENDIX B

WASTEWATER DISINFECTION ISSUES

AND RECOMMENDED POSITION

WASTEWATER DISINFECTION ISSUES
AND
RECOMMENDED POSITION

Specific issues were identified and were requested to be addressed in the review of the Uniform Guidelines for Sewage Disinfection by waste dischargers, regional water quality control boards, the State Water Resources Control Board, and others. Based on information developed in the review report, issue papers have been prepared and contain the recommended position of staff and the Policy Committee on each subject.

A. Issues Raised by the Sacramento Regional County Sanitation District

1. California disinfection requirements are more strict than EPA criteria.

2. California disinfection requirements are more strict than those used by other states.

3. California disinfection requirements result in increased safety hazards to employees and the public.

4. California disinfection requirements necessitates the addition of substantially more chlorine and sulfur dioxide than would be necessary to meet EPA or Basin Plan criteria.

5. California disinfection requirements result in additional formation of trihalomethanes (THMs) and total dissolved solids.

6. California disinfection requirements will result in a minimum of $4.33 million extra cost over a five-year period at Sacramento.

B. Issues Identified by the State Water Resources Control Board

1. Risks of disease associated with exposure to pathogens in receiving waters, as a function of type of exposure (e.g., drinking, with or without further treatment, swimming, consumption of fish or shellfish, etc.), the concentration of pathogens, and other relevant factors (e.g., drinking water usually receives further treatment; swimming is a seasonal use).

2. Relationship between concentrations of various indicator organisms and the concentration of pathogens in receiving waters (e.g., total coliform, fecal coliform, enterococci, etc.).

3. Relationship between concentrations of various indicator organisms and the concentration of pathogens in disinfected sewage effluent.

4. Relationship between discharges of disinfected effluent, changes in the distribution and concentration of pathogens in receiving water,

and changes in receiving-water health risks. The purpose of establishing this relationship is to determine what level of disinfection is necessary to achieve or protect bacteriologic water quality objectives pursuant to the scientific assessment of risk developed from the Department's statewide study.

5. Potential formation of carcinogens during the disinfection processes.

C. Issue Raised by the City of Santa Rosa

Under the concept of an indirect discharge, would requirements be modified, including dilution requirements?

D. Issues Generated From Other Sources

1. Consideration of seasonally modified disinfection requirements (cold weather, low use, high flow).

2. Inclusion of a maximum indicator organism limit in the uniform disinfection guidelines.

3. Selection of the indicator organism to be used to demonstrate disinfection effectiveness.

CONTENTS

ISSUE 1: Would the safety hazards associated with chlorine be reduced significantly if chlorine dosages were reduced at sewage treatment plants?

DISCUSSION: Exposure to liquid chlorine can result in severe skin or eye burns. However, the most common exposure is to gaseous chlorine, which is the normal chlorine state at atmospheric pressure and normal temperatures. The most important human exposure routes to gaseous chlorine are inhalation and eye and skin contact. The threshold of inhaled chlorine gas ranges from 0.02 to 2.0 parts per million (ppm). At concentrations of 1 to 2 ppm and higher, irritation becomes a problem, and above 4 ppm, it becomes intolerable. There is little dose-response correlation for workers chronically exposed to chlorine below 1 ppm, but chronic exposure can result in respiratory complaints, nausea, increased susceptibility to tuberculosis, and corrosion of the teeth. High concentrations of chlorine irritate the skin, producing burning, stinging, inflammation, achrodermatosis, shrivelling, blistering, and development of nodules.

Acute exposure to chlorine presents both acute and latent effects. At low concentrations, the acute effects are confined to mild irritation of the eyes and upper respiratory tract, which are eliminated shortly after exposure is stopped. As the chlorine concentration increases, the effects become more severe. There is dyspnoea, cyanosis, vomiting, headache, and a heightening of anxiety, but there is complete recovery with palliative treatment. There have been few fatalities following chlorine exposure. However, at significantly high concentrations, the chemical can cause shock, coma, respiratory arrest, and death.

The identification of risks associated with chlorine can be grouped according to production, transportation and handling, and use. The amount of chlorine used as a wastewater disinfectant is very small in comparison to the total amount of chlorine produced; therefore, if chlorine were totally eliminated as a wastewater disinfectant, the decrease in risk associated with the overall production of chlorine would appear to be minimal and was neglected in the review report.

The transportation hazards are difficult to assess because of the variety of transportation modes and lack of data. Department of Transportation data for the years 1971 to 1980 indicate only one reported commercial transportation accident involving fatalities. A 1979 major rail accident in Youngstown, Ohio resulted in 160 injuries and 8 deaths. Other than that accident, there were 93 accidents resulting in 236 injuries during that 10-year period.

On-site accident information for sewage treatment plants is not available. However, a 1979 summary of American Water Works Association data on accidents at water treatment plants indicated that four percent of the injuries sustained were due to contact with radiation and caustic, toxic, and noxious substances. Chlorine exposure would fall into this category. Assuming that almost all of the injuries in that category were due to exposure to chlorine and that the accident rate at sewage treatment plants is similar to that at water treatment plants, a conservative estimate would indicate that four percent of accidents at sewage treatment plants were caused by chlorine.

RECOMMENDATION: The reported injuries associated with the production, transportation, and handling of chlorine are low and would not be measurably decreased by reducing the amount of chlorine used at sewage treatment plants. Similarly, the risks associated with the use of chlorine at sewage treatment plants are low and would not be reduced significantly by lowering the amount of chlorine used for disinfection purposes. Chlorine would still be present and used at the plants, and it is unlikely that a measurable reduction in risk would be achieved. Consequently, the premise that the safety hazards associated with chlorine would be significantly reduced if chlorine dosages were reduced at sewage treatment plants is not valid and should not enter into the decision-making process regarding disinfection limits for health protection.

ISSUE 2: Assess the monetary costs associated with different levels of disinfection.

DISCUSSION: The costs to meet various levels of disinfection are mostly related to the need for additional chlorine to meet lower coliform limits and, in some cases, the need to provide filtration prior to disinfection to remove suspended solids. Filtration may be necessary to meet an effluent coliform requirement of 2.2/100 ml. Filtration removes chlorine-consuming suspended solids and reduces the turbidity to levels that make the disinfection process more effective. It is also recognized that higher chlorine residuals result in additional costs for sulfur dioxide necessary to dechlorinate the effluent prior to discharge.

The review report contains data on the capital costs associated with chlorine disinfection. Since wastewater treatment plants which discharge to receiving waters have disinfection facilities, with the exception of some of those that discharge to the ocean, capital costs to increase the level of disinfection would be those associated with the need for additional chlorinators and equipment. In some cases, enlargement of the chlorine contact chambers may be required to achieve effective disinfection through additional contact time.

Generally, the cost issue revolves around the level of disinfection and not whether or not disinfection should be required at all. Hence, operation and maintenance costs are the cruces of the issue. The operation and maintenance costs for disinfection include chlorine, labor, supplies, and power.

The costs, both capital and operation and maintenance, decrease as the size of the treatment facility increases. Costs are usually presented as unit costs, typically in cents per thousand gallons of wastewater treated. Total chlorination costs range from approximately 1.5 to 8 cents per 1,000 gallons, depending on the size of the treatment plant, for a chlorine dosage of 10 mg/1. The total costs for both chlorination and dechlorination range from approximately 3 to 12 cents per 1,000 gallons. This amounts to six to ten percent of the total cost of a secondary treatment facility.

Capital costs account for roughly half of the total disinfection unit cost. The position that increasing the chlorine dosage to meet low coliform limits greatly increases costs does not appear to be accurate. For example, data from the Pomona Water Reclamation Plant indicate that if both the chlorine and sulfur dioxide dosages were increased by 5.0 mg/1 (from an original chlorine dosage of 9.5 mg/1), the total unit cost would be increased by $16 per million gallons, which is 1.6 cents per 1,000 gallons. Similarly, a recent cost analysis of a proposed 30-MGD wastewater reclamation facility indicated that increasing the chlorine dosage from 11 mg/1 to 16 mg/1 would increase the chlorine cost from 1.4 to 2.0 cents per 1,000 gallons. These incremental increases are low relative to total operation and maintenance costs for the facilities. The cost would be higher where additional chlorinators, evaporators, meters, etc., were needed, but the cost would still be a small percentage of the total.

Several models have been identified in the review report which relate coliform bacteria density reduction to chlorine residual and contact time. These may be used to estimate the amount of additional chlorine needed, and the added cost, to achieve a coliform bacteria MPN of 23/100 ml versus a fecal coliform MPN of 200/100 ml.

One of the models is based on the disinfection of secondary effluent from three Los Angeles County treatment plants:

$$MPN/MPN_o = 27.22\ [1 + (CT \times R)]^{-2.27}$$

Where:

MPN = disinfected effluent MPN/100 ml

MPN_o = undisinfected effluent MPN/100 ml

CT = chlorine contact time in minutes

R = chlorine residual in mg/l

For making the estimate, the following assumptions have been employed:

200/100 ml fecal coliform = 1,000/100 ml total coliform

CT = 60 minutes

MPN_o = 500,000/100 ml total coliform

Then:

$$23/500{,}000 = 27.22\ (1 + 60R)^{-2.27}$$

R = 5.72 mg/l chlorine residual

$$1{,}000/500{,}000 = 26.22\ (1 + 60R)^{-2.27}$$

R = 1.08 mg/l chlorine residual

ΔCl = 4.64 mg/l to meet 23/100 ml total coliform instead of 200/100 ml fecal coliform

The cost for the added chlorine per million gallons per day of effluent flow can be computed assuming that the increase in chlorine dosage is within the capacity of existing chlorination equipment, and that the cost of chlorine is $0.10 per pound.

Then:

Pounds of chlorine per year per MGD flow =
1 x 8.33 x 4.64 x 365 = 14,110

Cost of chlorine per year per MGD flow =
0.1 x 14,110 = $1,411

A second model was developed based on total coliform bacteria density reduction in primary effluent in an ideal plug flow reactor or a stirred batch reactor:

$$MPN//MPN_o = [1 + 0.23 (CT \times R)]^{-3}$$

Then:

$$23/500{,}000 = [1 + 0.23 (60R)]^{-3}$$

$$R = 1.89$$

$$1{,}000/500{,}000 = [1 + 0.23 (60R)]^{-3}$$

$$R = 0.54$$

$$\Delta Cl = 1.35 \text{ mg/1}$$

Cost of chlorine per year per MGD flow = $410

The added chlorine and the cost may be understated because the chlorine contact conditions used to develop the models were closer to ideal than may be available at most full-scale plants. The cost difference using the two models is significant. The range of the two costs for meeting a more stringent coliform standard is 0.1 to 0.4 cents per 1,000 gallons treated.

Using a third formula: $MPN/MPN_o = (bCt)^{-a}$ and factors obtained in wastewater studies at Palo Alto, Dublin, and San Jose (EPA - Municipal Wastewater Disinfection), the cost of added chlorine to meet a coliform bacteria MPN of 23/100 ml instead of 200/100 ml of fecal coliform is 0.1 to 0.5 cents per 1,000 gallons treated.

If dechlorination is required, the total incremental cost for chlorine and sulfur dioxide would be approximately 2.4 times that shown for chlorination alone.

In some cases, it may be difficult to consistently meet a disinfection requirement of 2.2 coliforms/100 ml without filtration. Data suggest that filtration with chemical addition can be 15 to 30 percent of the total treatment cost; it may be higher or lower depending on the type and number of other unit processes at the wastewater treatment plant.

__RECOMMENDATION__: The added cost of chlorine to meet certain discharge requirements is minor in relation to total treatment costs. The small incremental cost is more than offset by the added level of health protection associated with higher levels of disinfection. Filtration may be needed to meet disinfection requirements in some cases where there is substantial human contact with the treated effluent. In light of the disease risk associated with improperly disinfected wastewater, this level of treatment is justifiable. Therefore, it is not unreasonable, from an economic standpoint, to require disinfection for the maintenance of specific coliform limits that have been shown to provide adequate health protection.

__ISSUE 3__: What indicator organisms should be used to determine disinfection effectiveness?

__DISCUSSION__: Two indicator organisms are currently used to demonstrate the effectiveness of sewage disinfection -- total coliform bacteria and fecal coliform bacteria. Based on the review report, a third group, enterococci, has characteristics which support its consideration.

Many states use a fecal coliform number to demonstrate effective disinfection. The use of the fecal coliform organism was based on the EPA definition of secondary treatment which formerly contained an effluent requirement of 200/100 ml fecal coliform bacteria. Fecal coliform bacteria are used as an indicator of the sanitary quality of fresh surface waters by most states, and the use of the fecal coliform group would provide consistency of effluent and receiving-water indicator organisms. The fecal coliform group is less resistant to disinfection by chlorination than is the total coliform group and is less numerous in wastewater.

The total coliform group is used in the current disinfection guidelines for all wastewater discharge situations. The group has been used in studies to determine the relative effectiveness of treatment processes and disinfection on indicator organisms and viruses. It is more numerous than fecal coliform bacteria in untreated wastewater, more resistant to chlorination, and thereby, more representative of certain viruses and other pathogenic agents which are resistant to chlorination.

Total coliform bacteria display a higher degree of regrowth than other indicator organisms and originate from nonfecal sources. These characteristics reduce the value of the total coliform group as an indicator of surface water quality, but not disinfection efficiency.

Enterococci are considered to be indicator organisms of choice for determining the sanitary quality of recreational waters by EPA because of an identified relationship of gastrointestinal illness in recreational water use and enterococci concentrations. EPA has encouraged states to adopt enterococci standards or objectives for surface waters (or __E__. __coli__ for fresh water) in place of the current coliform or fecal coliform standards. Because of this, there may be the advantage of consistency in the future in the use of enterococci to measure disinfection. Enterococci is more resistant to disinfection than total or fecal coliform; however, it is found at a much lower density (approximately two to three logs) than the other indicator organisms. There has been no experience with its use as a measure of disinfection effectiveness in California and little information exists on concentrations in receiving waters, particularly fresh waters.

The use of indicator organisms throughout the nation may be in a period of reconsideration and possible transition. As data is generated on enterococci, there may be justification for a shift to this group; however, the information to do so is not available at this time.

__RECOMMENDATION__: The total coliform group should continue to be used to measure and demonstrate sewage effluent disinfection effectiveness. Data should be collected on enterococci in effluents and receiving waters to serve as an information base for consideration of this group as an indicator of both disinfection effectiveness and receiving-water sanitary quality. The regional water quality control boards should be requested to include monitoring for enterococci (and/or *E. coli* in freshwater receiving waters) in waste discharge requirements.

ISSUE 4: For discharges to rivers, should the guidelines use the former EPA disinfection standard of 200/100 ml median fecal coliform or the 23/100 ml median total coliform bacteria which is currently recommended in many California discharge situations, or should another criteria be considered?

DISCUSSION: The California State DHS has recommended a median total coliform density of 23/100 ml for discharges to surface waters where dilution of 20:1 to 100:1 is provided. The same median coliform MPN is requested in effluents where there is greater than 100:1 dilution and the discharge can affect the quality of waters overlying shellfish growing areas. For discharges near marine water recreational areas and fresh waters where the dilution is greater than 100:1, a median total coliform of 230 or 240/100 ml is recommended. For situations where there is little or no dilution and significant water use, more restrictive disinfection requirements are recommended. The California wastewater disinfection criteria appropriately take into consideration dilution and receiving-water uses more than the disinfection standards for any other state because of the extremely wide variety of conditions and uses in receiving areas.

The total coliform median of 23/100 ml is the value most often questioned and compared to effluent standards used in other states. For reasons expressed here and in the coverage of other issues, the 23/100 ml median is appropriate for health protection.

The 23/100 ml median coliform density was selected in part (as opposed to another number of similar magnitude) because it is a commonly reported, most probable number when five fermentation tubes per dilution are used in the laboratory test. Generally, publicly owned treatment works are required to use five tubes per dilution in the bacteriological examination of wastewater. Even with five tubes per dilation, the precision of the results is not of a high order. For an MPN of 23/100 ml, the lower and upper 95 percent confidence limits (according to "Standard Methods") are 9.0/100 ml and 86/100 ml, respectively. In order to obtain an MPN of 23/100 ml, all five fermentation tubes which receive 10 ml portions of wastewater must give positive results, and the tubes which receive 1.0 and 0.1 ml portions of wastewater must give negative results. This would be a typical reactive because of the tenfold greater amount of wastewater innoculated in the first tubes. Raising or lowering the recommended disinfection median value could result in an anomalous figure which would either represent unusual test results or would be unreported. (For example, an MPN of 200/100 ml is not a reported result.) Similarly, 230 or 240/100 ml are commonly reported results.

The alternative membrane filter test is rarely used for disinfected wastewater because, before it can be used for bacteriological analysis of wastewater, a demonstration that the test gives comparable results to the multiple tube fermentation test may be required.

The EPA had initially included a geometric mean of 200/100 ml fecal coliform bacteria concentration in the standard federal definition of "secondary treatment" (mean of samples collected in a period of 30 consecutive days or 400/100 ml in 7 days). This standard was later eliminated based on concerns that this degree of disinfection, or disinfection _per se_, was _not_ needed in all situations and all seasons. As is shown in the review report, where an effluent disinfection standard has been adopted by other states, it is often a 200/100 ml median fecal coliform standard with variations depending upon the quality of the receiving waters, water uses, and the season. A maximum limit (or 90 percentile) is also common.

In disinfected wastewater, the total to fecal coliform ratio appeared to be roughly five to one. A median of 200/100 ml fecal coliform would approximate 1,000/100 ml total coliform median or more than 50 times greater than the 23/100 ml. It is pointed out in Issue No. 10 that higher levels of pathogenic agents may be associated with the higher coliform densities (200/100 ml fecal coliform), and risk of infection or disease may increase in uses of the receiving waters.

The EPA health effects studies of fresh recreational waters determined that detectable increases in "highly credible" gastrointestinal illness rates were associated with swimming in waters which contained low concentrations of the indicator organisms E. coli and enterococci. At a fecal coliform median of 200/100 ml in the receiving waters, it was calculated that the increased illness rate was 8/1,000 swimmers (or 800/100,000). By comparison, the total reported morbidity rate from all causes for salmonellosis and shigellosis in California was 16.5/100,000 and 15.8/100,000, respectively, in 1985. (It should be noted that the EPA results were from a careful study directed at detecting illness, and morbidity results of certain notifiable diseases may be underreported by a factor of 10 to 20.) Further, the EPA studies detected measurable health effects associated with enterococcus levels as low as 10/100 ml via a route in which only 10 to 50 ml of water are ingested (according to EPA).

Unfortunately, in the EPA study, no definitive information was presented on the bacteriological quality of the waste discharges which influenced the recreational water quality, although the two states involved had a 200 median fecal coliform limit in the receiving waters or in the waste discharge if the receiving waters exceeded 200/100 ml. In view of the lack of information on disinfection, dilution, etc., the responsible public health position is that the disinfection requirements in the states which were studied did not provide adequate health protection for a use in which very little water was ingested, and resulted in unexpectedly high illness rates.

The EPA has recommended an E. coli geometric mean not to exceed 126/100 ml or enterococci not to exceed 33/100 ml for fresh recreational waters. These densities of indicator organisms were calculated to approximate the degree of protection and illness rate of swimmers now associated with the 200/100 ml fecal coliform density in recreational waters. If there is a shift to E. coli or enterococci in receiving waters, it may be appropriate to consider the same indicators to measure disinfection effectiveness. At present, there is little information on which to propose disinfection criteria based on these organisms. The apparent greater resistance of enterococci to disinfection is encouraging, but the smaller densities prior to disinfection may pose a problem. Because of the lack of information, it is appropriate to consider the inclusion of enterococci and E. coli in monitoring programs of receiving waters and waste discharges so that comparative data can be obtained. Inasmuch as a fecal coliform standard is not recommended for either wastewater disinfection or receiving-water quality by EPA, it would not be appropriate to change to a fecal coliform requirement as a disinfection guideline.

RECOMMENDATION: Use of the former EPA standard of a fecal coliform median MPN of 200/100 ml is not appropriate as a measure of effective disinfection. Health protection afforded by the former standard is questionable and the fecal coliform group is no longer recommended as an indicator organism by EPA. It is recommended the total coliform group continue to be used to measure disinfection effectiveness because of long experience with its use, and the current recommended levels of disinfection based on total coliform densities appears appropriate.

__ISSUE 5__: Why are the disinfection requirements recommended by the California DHS more strict than those used in other states?

__DISCUSSION__: The disinfection requirements recommended by DHS range from "no disinfection" to tertiary effluent meeting a median total coliform MPN of 2.2/100 ml. For a discharge to an inland waterway, the recommended disinfection requirement is generally a total coliform median density of 23/100 ml. As the review report indicated, many other states require a median fecal coliform limit in wastewater effluents of 200/100 ml which is approximately equal to a total coliform limit of 1,000/100 ml.

According to EPA, many states impose more strict disinfection requirements under certain circumstances. Over 45 states have multilevel disinfection standards. At least 15 states require effluent disinfection levels of 14 fecal coliform per 100 ml for discharge into shellfish waters, 9 states have more strict discharge requirements than the 14/100 ml fecal coliform, and Maryland, since 1976, has required all treatment facility plants to provide for disinfection to total coliform levels of 3 MPN/100 ml, unless a special exemption is granted.

The EPA information indicates that strict disinfection requirements are applied in many states where there is a recognized need based on water uses. The California requirement for discharges to fresh surface water is more strict than other states apply; however, issue No. 9, which addresses relative health risks based on health effects studies, and Issue No. 10, which estimates health risk through dose-response models, indicate the disinfection requirement most commonly used in other states may allow significantly more detectable illnesses or infections than California disinfection criteria. A more strict standard provides improved health protection.

__RECOMMENDATION__: Maintain the more strict California disinfection recommendation for health protection from waterborne diseases.

__ISSUE 6__: Should the disinfection criteria include a maximum allowable indicator organism concentration as well as a median concentration? If so, what should the maximum number be?

__DISCUSSION__: The disinfection criteria had indicated that a maximum coliform bacteria limit may be designated to address the serious situation where little or no disinfection is provided for a limited period. Such a situation would not violate discharge requirements where only a median value is specified.

The regional water quality control boards have included maximum indicator organism concentrations or concentrations not to be exceeded in two consecutive samples in many waste discharge requirements. The available data indicate the following typical maximum values:

Total Coliform Bacteria
MPN/100 ml

Median	Maximum
2.2	23
23	500
50	700
230	5,000

According to Velz, C. J. (1951), the 99 percent upper limit based on the standard distribution slope for the bacteriological test when five tubes are used for each decimal dilution is:

Median	99% Upper Limit
2.2	8.2
23	86
50	190
240	890

The 99 percent limit might be exceeded once in 100 tests due to the variations inherent in the test. The numbers are far below the maximum numbers used by regional boards. With an allowance included for actual changes in effluent quality (σ log = 0.5) then:

Median	99% Upper Limit
2.2	32
23	335
50	728
240	3490

With the above assumptions, there would be 1 chance in 100 that the maximum would be exceeded if it is set approximately 20 times the median value. The regional boards are often using this multiplier to set the maximum value, and it appears to be a reasonable control level which would not penalize the waste discharger and which would detect periods of poor disinfection.

__RECOMMENDATION__: Support the use of a maximum coliform bacteria concentration in waste discharge requirements, which is approximately 20 times the required median density. Compilation of coliform bacteria violations at wastewater treatment plants would be desirable to document the suitability of the suggested maximum limit.

Velz, C. J. (1951), "Graphical Approach to Statistics:IV, Evaluation of Bacterial Density"; Water and Sewage Works.

<u>ISSUE 7</u>: Should the guidelines be modified to provide seasonal adjustments in effluent disinfection to reflect differences in receiving-water uses?

<u>DISCUSSION</u>: The current discharge requirements contain the following statement regarding discharges to freshwater streams and rivers: "For these discharge situations, it is particularly important to fully consider the individual circumstances so that adequate health protection is provided through the application of reasonable disinfection requirements. For example, it may be appropriate to reflect seasonal changes in recreational use, dilution at the use area, etc."

Several factors argue for consideration of seasonal differences in disinfection requirements on a case-by-case basis. In portions of the State such as Ventura, Los Angeles, and San Diego counties, the ocean beaches are well attended throughout the year, and swimming, wading, and surfing are consistently among the most popular activities (Department of Parks and Recreation, Ventura County Beach Study (1976)). Seasonal variations in disinfection requirements for discharges which may affect these waters would be inappropriate.

In northern California, water contact recreation may be reduced significantly during the winter months, although it does continue to occur. For example, water skiing in the Sacramento-San Joaquin is distributed as follows:

March	-	May	17%
June	-	August	58%
September	-	November	23%
December	-	February	2%

(From SRI Recreation and Fishery Values in the San Francisco Bay and Delta.)

Also, it has been estimated that at least 200 persons swim or surf in the bay and ocean off San Francisco, even on a stormy winter day (R. C. Cooper, Public Health Aspects of San Francisco Waste Discharge).

While recreational water use varies significantly with the season, it is also affected by the immediate weather conditions. In the winter, the weather may be generally cold and inhospitable; however, there may be balmy days during which water recreation will occur. Additionally, there may be various sporting activities, such as fishing, with seasonal variation.

In considering the possibility of a seasonal variation in disinfection, the following factors should be considered:

. Ambient air and water temperatures.

. Type of beneficial uses.

. Accessibility to downstream areas.

. Weather conditions.

- Other seasonal causes of contamination, e.g., storm runoff.
- Dilution.

Where there is even limited use of downstream waters, health protection would allow no relaxation in disinfection requirements.

A modification would not be appropriate where downstream uses include domestic water supply, where simple chlorination is the only treatment, where there is significant water contact recreation all year, or where shellfish growing waters may be influenced by the discharge.

RECOMMENDATION: Seasonal modifications in waste discharge disinfection should be limited to those situations where health protection will not be lessened. The local health officer may assist in the determination of the suitability of a seasonal modification.

ISSUE 8: Should the disinfection guidelines recognize the concept of an "indirect discharge" to receiving waters and recommend guidelines for such a situation? Should the guidelines require dilution where the receiving waters are used as a source of domestic water supply?

DISCUSSION: The current Uniform Guidelines for Sewage Disinfection do not recognize the concept of an indirect discharge of wastewater to receiving waters. Under the guidelines, either there is a discharge or the wastewater is confined to land. In natural land disposal situations, wastewater may percolate underground from disposal ponds and ultimately reach a waterway through horizontal movement, either with or without reaching groundwater; however, where there is aboveground movement of wastewater from a pond system, spray area, or leach field to a receiving water, this is considered to be a discharge for the purposes of the guidelines.

The guidelines propose that a "no discharge" recommendation should be made for proposed discharges to lakes, reservoirs, and freshwater streams used for domestic water supply where land disposal is physically and economically possible. For a freshwater stream, if it is not possible to prevent a discharge, a discharge of not more than five percent of the stream flow may be considered. The recommended discharge limitation applies only where a beneficial use of the water is domestic water supply. Where water-contact recreation and agricultural irrigation are beneficial uses and not domestic water supply, there is no minimum dilution requirement of the water discharge for health protection. This is consistent with California wastewater reclamation regulations which provide for recreation and irrigation uses of undiluted reclaimed wastewater which has undergone extensive treatment and disinfection.

The concept of an indirect discharge to receiving waters has been proposed by a California sewerage agency. Under this concept, an indirect discharge (or a discharge equivalent to an indirect discharge) would exist if:

A. The wastewater is changed in a manner that would cause it to lose its character as a wastewater or no longer be distinguishable as a wastewater and be indistinguishable from the receiving water;

B. The point of compliance for discharge precedes the indirect discharge step;

C. The effluent passes through a polishing treatment step which alters the physical, chemical, and biological parameters of the wastewater: and

D. The effluent is dispersed before entering the receiving water.

Other conditions might also be applied according to the agency to distinguish the indirect discharge situation from a direct discharge. It has been proposed by the sewerage agency that, under these conditions, there would not be a need for the five percent discharge limitation and it would not apply.

The purpose of the disinfection guidelines is to ensure that the public health will be protected for the uses made of the receiving waters. This is independent of the conditions proposed above for an indirect discharge. Health protection is not related to the comparative quality or character of the wastewater and the receiving wastes, as measured by such characteristics as BOD, nutrients, and solids content, and is not related to alteration of general waste parameters or the degree of dispersal of the discharge. These conditions may be significant for protection of beneficial uses such as esthetic enjoyment or preservation of aquatic resources, but not for health protection for recreation, domestic water supply, and agricultural use, which are the subjects of the guidelines. Consequently, the concept of an indirect discharge, as proposed, is not a factor for consideration or incorporation in the disinfection guidelines.

More directly, the issue is whether a limitation on the discharge to provide a specified dilution where domestic water use is involved is an appropriate health protection provision or whether wastewater treatment and management can replace all or part of the dilution provision. The review report presents information in several areas which relates to this issue.

The report pointed out that the presence of organic materials in treated domestic wastewater is highly variable and dependent upon the source of the sewage and the treatment it receives. Although studies have identified a wide range of specific organic constituents in treated wastewater at low levels, the organic fraction remains largely unidentified. Laboratory studies indicate that nonvolatile fractions of organics may be mutagenic in various hosts. While technology regarding trace organics has advanced substantially over the past few decades, uncertainties persist regarding the range of compounds, synergistic effects, significant of mixtures, and the total health significance of trace organic constituents. It should be noted that the increased knowledge regarding organic substances has been accompanied by an increased concern with longer term health effects. In this setting, dilution provides a positive means for a reduction in a potential uncertain health risk where the water is ingested.

It could be reasoned that a few surface waters in other states, which receive much higher percentages of sewage effluent, are used as sources of domestic water supply. It has been estimated, however, that the water supply of only 0.7 percent of the United States population is derived from waters containing more than five percent sewage effluent during average river flow conditions (EPA 1985).

With regard to the health concerns associated with waste-borne disease through pathogens, the particular waste disposal-water use situation is important. It was reported that disease outbreaks were more often associated with small systems serving seasonal users. A substantial percentage of illness cases (44 percent) was associated with untreated or inadequately treated water and with interruptions in treatment. Systems depending on simple chlorination or intermediate treatment were less reliable than those depending on complete water treatment. Consequently, less dependence on dilution would be acceptable with regard to health protection from waterborne disease agents where the domestic water supplies provide complete and reliable treatment. The significance of the degree of water treatment provided is reflected in the water quality criteria which have been recommended at the federal and state level. The maximum recommended bacteriological quality for a water system providing only chlorination was 50/100 ml total coliform, whereas the limit for a water system providing complete treatment was 20,000/100 ml. Also, the U. S. EPA is considering a national standard which would require pretreatment and filtration of all surface waters and groundwaters, which are influenced by surface waters, for health protection from microbiological disease agents.

RECOMMENDATION: The guidelines for sewage disinfection should not include the concept of an indirect discharge. Where the receiving waters are used as a source of domestic water supply (or where domestic water supply has been designated as a beneficial use of the receiving waters by the RWQCB), it is appropriate to include a minimum dilution requirement in the uniform disinfection guidelines.

ISSUE 9: Address the relative risks of disease associated with exposure to pathogens in receiving waters, as a function of type of exposure (e.g., drinking, with or without further treatment, swimming, consumption of fish or shellfish, etc.), the concentration of pathogens, and other relevant factors (e.g., drinking water usually receives further treatment; swimming is a seasonal use).

DISCUSSION: The review indicated that there is limited information available on the concentration of pathogens in surface water used for domestic water supply, swimming, or shellfish cultivation. There are no known current programs carried out to identify pathogen concentrations in surface waters. The available data is limited to special studies involving a few pathogenic agents.

Indicator organisms have been used for decades in place of pathogens to indicate, indirectly, the relative sanitary quality of surface waters. There is data on the relative concentrations of indicator organisms and pathogenic agents in untreated sewage; however, it is variable, and differential removals of various pathogenic agents and indicator organisms are provided by different treatment processes. Also, die-off times are different in receiving waters. Consequently, it is difficult to provide a pathogen-indicator ratio that would apply generally and which would serve as the basis for estimating a direct disease risk.

Currently, the record of California surface water quality based on indicator organisms is limited. There is no statewide surface water bacteriological monitoring program. Local agencies and utilities are the sources of most monitoring information.

Dose-response information is available for some pathogenic agents, and models have been developed to estimate the probability of disease or infection associated with the ingestion of pathogens. Inasmuch as no substantial information has been developed on pathogen concentrations in surface waters, an estimate must be made in order to use the risk models. The pathogen concentration estimate involves preliminary estimates of morbidity rates, number of organisms shed, (or estimates of concentrations in sewage), effects of treatment and disinfection, dilution, and die-away factors. The many factors involved open to challenge the estimates of disease risk for the end uses of water based on risk models. (Recognizing the many assumptions and limitations, a calculated estimate of disease risk associated with waste discharges is presented in Issue No. 10.)

The foregoing presents the limitations which exist in developing a rigorous response to this issue. In spite of the information gaps, an indication of relative risks and concerns associated with current water quality standards can be obtained from health-related investigations carried out over the past decade for the various water uses.

Based on studies of marine and fresh waters by EPA, an added risk of disease (gastroenteritis) has been associated with swimming in waters which are influenced by sewage discharges and which meet the currently accepted median fecal coliform content of 200/100 ml. The reports of the studies did not specifically characterize the waste discharges, but used such terms as: "combined effluents... treated to various degrees... chlorinated only in part", "chlorinated secondary effluent", and "adequately treated with chlorine". Consequently, it is not possible to relate sewage treatment and disinfection effectiveness to the water quality and illness rates. At a median fecal coliform density of 200/100 ml, the added illness rate for "highly credible" gastroenteritis was calculated to be 8/1,000 swimmers for freshwater recreation and 19/1,000 swimmers for marine water recreation. Criteria have been recommended in terms of enterococci (or E. coli in fresh water) which provide this degree of protection.

The extent of the added illness rate was not anticipated by EPA and may not be acceptable to health authorities. (See Issue No. 4 for a comparison to notifiable gastrointestinal illness rates.)

The current California disinfection guidelines would provide a reduced risk of illness. Based on a risk model technique, an estimate of the probability of disease or infection, when current California disinfection guidelines are met, suggests that there would be 1 to 2 added cases per 1,000 swimmers in freshwater areas, which are one-day flow time downstream from a waste discharge, as compared to the 8/1,000 swimmers at a 200/100 ml fecal coliform limit. The number of estimates and assumptions in such an exercise are acknowledged.

As the review report indicates, concern has been expressed by medical authorities and others regarding hepatitis A, Norwalk Agent, and other viral and bacterial pathogen-caused illnesses associated with the consumption of shellfish taken from "approved" growing areas. The strong possibility exists that shellfish from unapproved areas may have been involved in some of the outbreaks. New York State shellfish-associated illness findings in 1982 (103 outbreaks with 1,017 illnesses) and 1985 (59 outbreaks with 888 cases) indicated that a substantial number of illnesses may be detected through the rigorous use of established procedures for systematic collection of epidemiologic data which otherwise may have gone unreported.

At the current time, while most shellfish outbreaks have been associated with contaminated growing waters or improper food handling, there is evidence presented in the review report to indicate a need to study the adequacy of standards which are currently applied to shellfish and shellfish growing waters. Such a study has been undertaken by EPA, and until the study is completed, the actual risk associated with water quality cannot be estimated accurately. Anecdotal information in the review report suggests, however, that there may be a low but actual health risk from shellfish taken from areas at or ~~near the state standard for growing waters.~~

As the review report indicates, the available information on waterborne disease outbreaks associated with domestic water systems does not lend itself to detailed interpretation regarding the relative risk of disease associated with the quality of the water sources. Generally, the quality of the surface water source at the time of the outbreak is not known, and the causative agent has been found in the water supply in less than one percent of the cases. Often, the overall statistics on outbreaks does not separate out the systems depending on surface water sources from those depending on groundwater sources (or both).

In addition to the variables which have been discussed regarding the ratio of pathogens to indicator organisms in receiving waters, disease risk would also depend on the effectiveness and reliability of the water treatment process(es). There are data available that indicate that domestic water supply systems which provide complete water treatment (coagulation with a primary coagulant and rapid mix, flocculation, sedimentation, filtration, and disinfection) constitute a low-risk situation with regard to pathogenic agents in the water source. The review report indicates that complete water treatment facilities may afford a six-log reduction of coliform bacteria and, by inference from proposed water treatment regulations, possibly four-log removal of viruses and three-log removal of Giardia cysts.

Systems depending on simple chlorination of surface waters (or groundwaters directly affected by surface water quality) represent a relatively high-risk situation for two reasons.

Chlorination of a surface water has little or no effect on protozoan cysts or certain viruses. Based on limited data, the product of free chlorine residual and time (CT) needed for a 99 percent inactivation of <u>Giardia</u> cysts is at least 1,000 times that for 99 percent <u>E</u>. <u>coli</u> inactivation. (The Polio I requirement is about 50 times greater than <u>E</u>. <u>coli</u>.)

There is little and conflicting information available on Giardia and virus densities in wastewater, receiving waters, and disinfected domestic water, so that risk factor estimation is difficult; however, because of the concern for the potential occurrence of Giardia cysts and viruses, EPA has drafted a "Criteria of Filtration Rule" under which, with very strict exceptions, all public water systems would be required to provide complete treatment and disinfection for water from surface sources.

The second reason that simple chlorination systems represent a high risk factor is the treatment tends to be less reliable and more subject to interruptions than complete water treatment systems.

Based on the results of the recreational water studies by EPA, a significant number of illnesses would be expected to be associated with a lack of water treatment or an interruption in treatment where a surface water supply source is influenced by waste discharges and meets a median fecal coliform density of 200/100 ml. By extrapolating the illness incidence of the recreational water studies to reflect a consumption of one liter of water, over 80 illnesses per 1,000 exposed persons might occur if the water treatment were interrupted for one-half day. Because of the ineffectiveness of simple chlorination on virus and protozoan cyst inactivation, a lower but actual disease incidence might be expected within the exposed population when the disinfection system was operating.

Historically, water sources standards reflect the differences in health risks associated with water treatment. (Geometric mean density of 2,000/100 ml fecal coliform and 20,000/100 ml total coliform for complete treatment; 50/100 ml total coliform for simple chlorination.) The draft filtration requirements of EPA similarly reflect the health risk difference of complete treatment and a lesser degree of treatment.

In summary, receiving waters used for domestic water supply without complete water treatment represent a relatively high health risk and would require effective wastewater treatment, disinfection, and dilution to provide health protection.

No studies have been conducted on the relative health risk associated with the consumption of food crops irrigated with surface waters which receive waste discharges. The time between irrigation and harvesting, effects of desiccation on pathogen survival times, and other factors suggest that wastewaters disinfected to meet the requirements of other water uses (recreation, water supply source) would pose little risk of infection or illness. The exception would be the direct use of wastewater effluent on food crops that may be consumed unprocessed. A high degree of treatment and disinfection would be required for assured health protection.

<u>RECOMMENDATION</u>: The review of health effects information indicates that health concerns are being raised with the use of recreation waters, domestic source waters, and shellfish growing waters which meet current receiving-water standards and wastewater disinfection requirements imposed in other states. Based on the current record, there is a health risk which can be reduced with each of the uses through more effective wastewater disinfection. The more conservative disinfection criteria used in California should be retained unless further studies refute current findings.

ISSUE 10: Identify the relationships among concentrations of various indicator organisms and the concentrations of pathogens in receiving waters and disinfected sewage effluent. Also identify the relationships among discharges of disinfected effluent, changes in the distribution and concentration of pathogens in receiving waters, and changes in receiving-water health risks.

DISCUSSION: The issue combines several issues which were requested to be addressed by the State Water Resources Control Board.

As the review report documents, there is little field data available to relate pathogen concentrations to indicator organism concentrations in receiving waters. The high variability of indicator organism densities in fresh water which may be unrelated to sewage effluent, the large number of specific pathogenic agents, types, laboratory analytical difficulties, and prohibitive costs have precluded the development of such data.

In order to address the issue, available information on the densities of organisms in untreated and treated sewage and removal efficiencies was used to estimate the range of organisms through the treatment and disinfection processes to receiving waters. The results are displayed in Table 1. The tabulated information, summarized from data in the review report, assumes secondary treatment, disinfection to a total coliform density of 100/100 ml or less, and a discharge equal to 5 percent of the river flow.

In preparing such a table, the limitations and uncertainties regarding organism densities are compounded moving through the treatment and discharge processes. A range of organisms are subject to a range of treatment efficiencies, etc.; secondary activated sludge treatment may provide different removals than secondary trickling filtration; helminth ova may be virtually unaffected by disinfection, but may be partially removed by sedimentation in the contact chamber, etc. Further, the table does not include pathogens of concern for which very limited information is available such as *Campylobacter*, *Yersinia*, pathogenic *E. coli*; certain pathogenic viruses (hepatitis A, rotavirus, Norwalk agent); and parasitic agents such as *Ent. histolytica*.

The estimated mid-range concentrations of pathogens in the receiving waters were then used to estimate the infection probability (viruses and parasites) or disease probability (bacteria) that might result from incidental (100 ml) ingestion of water while swimming in waters which receive ether disinfected or undisinfected effluent. Also, infection or disease risk was estimated for an extreme case where the surface water is used for domestic water supply with only chlorination treatment and the chlorination system has been out of service for half a day.

In determining the risk potential, the midpoint or average of the range of organisms estimated to be in the receiving waters was used. No reduction allowance was made for die-away, predation, or sedimentation of organisms from the discharge to the point of use. (Depending on the organism, the concentration may diminish by 50 percent in 24 hours or less in fresh water.) The probability of infection for two enteroviruses for which dose-response data is known was computed to display the difference of risk potential that may be involved with different virus types. In evaluating relative risk, the average

Table 1

Estimated Concentrations of Indicator Organisms and Pathogens
Density Per 100 ml

Organism	Untreated Sewage	Secondary Effluent	Disinfected Effluent	Receiving Water[a] (Disinfected)	Receiving Water[a] (Undisinfected)
Total Coliform	10^7-10^8	10^5-10^6	10^1-10^2	0.5-5[b]	5.000-50,000
Fecal Coliform	10^6-10^7	10^4-1-^{5}[c]	2-20	0.1-1.0[b]	500-5,000
Enterococci[d]	10^4-10^5	10^3-10^4	0.1-1.0	0.005-0.05[b]	50-500
Shigella[e]	1-500	0-50	0-0.005	0-.00025	0-2
Salmonella[e]	10^2-10^4	10-5-000	0-0.5	0-.025	0-200
Giardia Cyst[f]	10-200[g]	5-100[h]	0.23-4.66[n]	0.0115-0.233	0.25-5
Helminth Ova[i]	1-100	0-30[h]	0-20[j]	0-1	0-2
Enteroviruses	1-1,000[k]	0-500[l]	0-5[m]	0-0.25	0-25

a Wastewater is five percent of river flow.
b Assumes no regrowth.
c Fecal coliform is about 30 to 70 percent of total coliform prior to disinfection.
d Estimated from limited enterococci and fecal streptococci data.
e Limited data.
f Table does not include Ent. histolytica cysts; no U. S. data found.
g Some estimates based on infection rate and daily emissions were greater (up to 2.4 x 10^5).
h Poor removal by A-S process; fair by T. F.
i Virtually no data available.
j Relatively unaffected by wastewater disinfection.
k One estimate was 10,000/100 ml which may have been an error (EPA 1986).
l Poor removal by T. F. process. Fair removal by A. S. process.
m Est. 0.5 log decrease for each log decrease in coliform, based on 0.33 log decrease of Polio I (Pomona Study), which appears to be more resistant to chloramine than other enteroviruses.
n Est. 0.33 log decrease for each log decrease in coliform based on G. muris reduction at c.t 400-900 (chloramine), which is similar to Polio I reduction at c.t 600-900 (EPA 1986).

of the two was used. The conservative dose-response data for *Salmonella typhi* was used instead of data compiled by Cooper et al. (1984) for *Salmonella spp.*, other than *Salmonella typhi* because it appeared more in line with historical ID50s for salmonella types. (The *Salmonella spp.* risk estimate was significantly influenced by the inclusion of a high response rate based on an epidemiological study of an outbreak at Riverside.)

The probability of infection or illness utilizes the dose-response information summarized by Cooper et al. (1984), and Hass (1983). The beta-distributed ineffectivity probability model was used to compute risk. Again, the risk for virus and Giardia is in terms of infection and not disease.

Risk of Infection/Disease-Factors Used

Risk probability model $P = 1-(1 + n/b)^{-a}$

P = Probability of an individual contracting infection or illness from a single exposure.

N = Number of organisms in exposure (ingested).

a and b = Parameters of distribution based on dose-response data.

Swimming -- 100 ml water ingested.

Drinking -- 1,000 ml water ingested.

	a	b
Enterovirus (ECHO 12)	1.3	75
Enterovirus (Polio I)	0.119	200
Giardia lamblia	0.18	11.6
Salmonella	0.21	5,531
Shigella	0.16	155

Number of Organisms Ingested (N)

	Undisinfected		Disinfected	
	Swimming	Drinking	Swimming	Drinking
Enteroviruses	12.5	125	0.125	1.25
Giardia lamblia	2.625	26.25	0.1224	1.224
Salmonella	100	1,000	0.0125	0.125
Shigella	1	10	.000125	.00125

Table 2

Risk of Infection/Disease

	Discharge Undisinfected	Discharge Disinfected
Swimming (100 ml Ingested)		
Enterovirus (EHO12)*	1.82×10^{-1}	2.16×10^{-3}
Enterovirus (Polio I)*	7.19×10^{-3}	7.43×10^{-5}
Giardia lamblia	3.61×10^{-2}	1.89×10^{-3}
Salmonella	3.76×10^{-3}	4.74×10^{-7}
Shigella	1.03×10^{-3}	1.29×10^{-7}
Drinking (1,000 ml Ingested)		
Enterovirus (EHO12)	7.21×10^{-1}	2.13×10^{-2}
Enterovirus (Polio I)	5.61×10^{-2}	7.41×10^{-4}
Giardia lamblia	1.92×10^{-1}	1.79×10^{-2}
Salmonella	3.43×10^{-2}	4.75×10^{-6}
Shigella	9.95×10^{-3}	1.29×10^{-6}

* The two enteroviruses are shown to indicate the difference in risk due to the difference in dose-response of the two viruses. The actual risk associated with exposure to N number of the entero-virus group may lie in-between.

The limited amount of data available and the assumptions required to estimate the probability of infection or illness are acknowledged and should be taken into account in reviewing the results.

The results indicate that the potential for Giardia and virus infection from the ingestion of water-receiving sewage effluent is greater than the disease potential from bacterial pathogens.

The risk of Giardia infection through the ingestion of a liter of water containing undisinfected effluent appears significant (almost a 20 percent risk). It should be noted that the dose-response feeding experiments for Giardia resulted in infection from low to high doses (up to 10^6 cysts); however, no significant illness symptoms resulted. (Only two volunteers were exposed to most doses, however.) Also, the Giardia infection rates, even for developed countries, has been estimated to be 1 percent of the population or more. Consequently, the calculated infection rate may be reasonably accurate.

Based on the relative risks (and assuming the exposure to many enterovirus types would result in a risk between that estimated for Polio I and ECHO12), it might be expected that the discharge of a well-disinfected effluent (total coliform MPN <100/100 ml) would reduce virus and Giardia infections in swimmers by 90 percent or more and would essentially eliminate the 4 to 5 bacterial illnesses per 1,000 swimmers that might be related to an undisinfected discharge. (The estimate of illnesses due to bacterial pathogens may be significantly understated inasmuch as only two pathogens were considered.)

Where consumption of a liter of surface water had occurred, effective disinfection would reduce the risk of viral or Giardia infection over 90 percent, and the chance of 40 bacterial illnesses per 1,000 persons would be virtually eliminated.

The State Water Resources Control Board requested a further step of refinement in the evaluation of wastewater disinfection, pathogen density, and disease risk. Essentially, the Board wanted to know the risks associated with different levels of disinfection. Such an estimate requires another group of assumptions and estimates, which may be subject to challenge or other interpretations.

For the purposes of this exercise, it was assumed that the waste discharge would be disinfected to a total coliform density of 100/100 ml and compared to a discharge disinfected to a median fecal coliform density of 200/100 ml. If a lower total coliform density was used, i.e., 23/100 ml, the risk difference would be greater. A 200/100 ml fecal coliform density in the disinfected wastewater was estimated to be equivalent to a median total coliform density of 1,000 to 2,000/100 ml (a value of 1,000/100 ml was used). Estimates for indicator organisms and pathogens in the effluent and receiving waters were made and are shown in Table 3 along with the risk estimates.

Table 3

Estimated Concentrations of
Indicator Organisms and Pathogens

	Density Per 100 ml			
	Disinfected Effluent		Receiving Waters	
	(100TC/100)	(200FC/100)	(100TC/100)	(200FC/100)
Total coliform	100	1,000	5	50
Fecal coliform	20	200	1	10
Enterococci	1.0	10	0.05	0.5
Shigella	0.005	0.05	0.00025	0.0025
Salmonella	0.5	5.0	0.025	0.25
Giardia cysts	4.66	10.03	0.233	0.5015
Helminth ova	20	20	1.0	1.0
Enteroviruses	5	15.81	0.25	0.7905

Risk of Infection/Disease

	Discharge Disinfected to 200FC/100 ml	Discharge Disinfected to 100TC/100 ml
Swimming (100 ml Ingested)		
Enterovirus (EHO12)	1.35×10^{-2}	4.32×10^{-3}
Enterovirus (Polio I)	4.69×10^{-4}	1.49×10^{-4}
Giardia lamblia	7.59×10^{-3}	3.57×10^{-3}
Salmonella	9.49×10^{-6}	9.49×10^{-7}
Shigella	2.58×10^{-6}	2.58×10^{-7}
Drinking (1,000 ml Ingested)		
Enterovirus (EHO12)	1.22×10^{-1}	4.17×10^{-2}
Enterovirus (Polio I)	4.60×10^{-3}	1.48×10^{-3}
Giardia lamblia	6.26×10^{-2}	3.24×10^{-2}
Salmonella	9.48×10^{-5}	9.48×10^{-6}
Shigella	2.58×10^{-5}	2.58×10^{-6}

The computed risk indicates that the chance of virus and Giardia infection while swimming or drinking would be approximately doubled if the disinfected effluent bacterial density (disinfection requirement) were to be changed from <u>less than 100</u> total coliform per 100 ml to 200 fecal coliform per 100 ml. The low-level bacterial illnesses would be increased tenfold.

CONCLUSION: In order to address the issue of the relationship of sewage effluent disinfection and disease risk, many assumptions and estimates were required. Consequently, the conclusions are in fairly general terms.

The disinfection of sewage effluent provides a significant barrier to waterborne disease transmission although all pathogenic agents, and disease risks are not eliminated, even with a high degree of disinfection.

The typical disinfection requirement applied in California for waste discharges to inland surface waters (median total coliform MPN of 23/100 ml) reduces the risk of infection or disease by half or more as compared to the former EPA disinfection standard of 200 fecal coliform per 100 ml.

The California requirement is appropriate and should be retained.

ISSUE 11: What is the relative health risk of THM formation versus pathogen destruction with sewage disinfection?

DISCUSSION: THM compounds are derivatives of methane where three or four hydrogen atoms have been replaced by either chlorine, bromine, or iodine. Although ten possible THM compounds exist, only five have been detected after chlorination of water: chloroform ($CHCl_3$), bromoform ($CHBr_3$), dichlorobromomethane ($CHCl_2Br$), dibromochloromethane ($CHClBr_2$), and dichloroiodomethane ($CHCl_2I$). Apparently, the iodinated compound occurs infrequently. The U. S. EPA and DHS have set a maximum contaminant level (MCL) of 100 ug/l for THMs in drinking water. The risk estimates associated with the total THM MCL are based on animal studies and predict an incremental risk of 3 to 4 excess cancers per 10,000 population consuming 2 liters of water containing 100 ug/l of THM daily for 70 years.

The concentration and species of THMs formed in wastewater are highly variable, depending mainly on pH, temperature, chlorine dose, contact time, and concentration of precursors, ammonia compounds, and bromine. Chloroform is generally the predominant species formed during wastewater disinfection, with lesser concentrations of bromodichloromethane, dibromochloromethane, and bromoform being formed. Bromoform may predominate if the effluent contains substantial quantities of saline water. Also, there will likely be a several-fold increase in THM formation in chlorinated effluent that is nitrified, since THMs are mainly formed during disinfection with free available chlorine.

Most wastewater treatment plant effluents contain ammonia, which results in combined chlorine residuals upon disinfection, thus restricting THM formation. Most research indicates that THM concentrations in chlorinated wastewaters are generally less than 50 ug/l, and in many cases, less than 10 ug/l, even with a 2-hour chlorine contact time. The exceptions to this range, i.e., higher THMs, appear to be locations where saline water infiltration has influenced wastewater quality or where a free chlorine residual may be involved. The chlorine-organic reactions that form THMs are time dependent and, hence, dechlorinated effluents produce lower levels of THMs than effluents that have not been dechlorinated.

The limited amount of information available regarding THM concentration in disinfected wastewater is highly variable, and it is difficult to predict the relative amount of THMs formed at different levels of disinfection. For example, a study by the Los Angeles County Sanitation Districts found that chloroform formation, after disinfection of filtered, secondary effluent, ranged from 0.5 to 9.1 ug/l after a 2-hour contact time with a chlorine residual of 5 to 10 mg/l. The effluent met a total coliform requirement of 2.2/100 ml. In contrast, data from the Sacramento Regional Wastewater Treatment Plant indicated that the chloroform concentration of chlorinated secondary effluent ranged from 6.4 to 15 ug/l. This plant has a chlorine contact time of approximately 30 minutes, a residual of 12 mg/l, and is required to meet a coliform limit of 23/100 ml.

One study indicated that trickling-filter, secondary effluent from the Dayton, Ohio sewage treatment plant, which contains a significant amount of industrial wastewater, had a chloroform concentration of 0.3 to 1.4 ug/l in the unchlorinated secondary effluent and 0.4 to 12 ug/l in the effluent after disinfection. The same report stated that unchlorinated secondary effluent from Cincinnati's Muddy Creek activated sludge secondary treatment plant, which treats, primarily, domestic wastewater, had a chloroform concentration of 0.1 to 0.7 ug/l in the unchlorinated effluent and 0.5 to 12 ug/l in the chlorinated effluent. The disinfection levels and contact times at these two treatment facilities were not reported.

The fate of THMs in receiving waters has largely been ignored in the literature. One river study in Virginia found that the mean chloroform concentration was 3.2 ug/l in the river water approximately 2 miles below the last of 11 sewage treatment plants, which was only 1.0 ug/l greater than at an upstream control point on the river. The discharges constituted ten percent of the river flow. One study reported that THM concentrations of water at various locations in the Sacramento River, delta, and aqueduct systems that transport water from the delta were generally low, i.e., less than 5 ug/l. The highest THM concentration measured during the study was 25 ug/l, which was recorded in 1977 during the drought. The authors of that study found no apparent relationship between COD or TOC and THM formation.

Information regarding THM concentration in other watercourses in California apparently have not been reported in the literature, but some information is available regarding THM formation potential (THMFP). Two studies indicated that the THMFP of the American River at Nimbus Dam was 62 and 65 ug/l, while it was 78 and 85 ug/l in the Sacramento River at Green's Landing, which is below the Sacramento Regional Sanitation District's 150 MGD discharge of secondary effluent. If these data are indicative of other discharge situations, it appears that a small percentage of the THMs found in receiving waters are attributable to municipal wastewater discharges. In addition to dilution, natural purification processes and in-stream volatilization may be factors in lowering the concentration of THMs originating in disinfected wastewater effluents.

Where receiving waters are used as sources of drinking water, it is difficult to predict the effect that sewage disinfection has on the ultimate drinking water quality regarding THM formation. Wastewater contains organic precursors that may be converted to THMs upon disinfection either at sewage treatment plants or at water treatment plants. Generally, low quantities of THMs are produced during sewage disinfection and, through dilution and other means, there is not likely to be a significant increase at a water supply intake due to the disinfected sewage input. On the other hand, if the wastewater effluent is not disinfected, the precursors would still be present in the water, and a greater potential of forming THMs at a downstream water treatment plant would exist than if the wastewater were disinfected. In any case, however, most water purveyors disinfect with monochloramine to prevent the formation of THMs, so it would appear that the THM levels in drinking water would not be significantly affected whether or not the wastewater effluent is disinfected. The authors of one study of THM concentration in the Sacramento River system, into which several sewage treatment plants discharge their effluent, found that the river water does not contain significant precursors until the water reaches the delta.

While the THM concentrations in disinfected wastewater are highly variable, the levels in effluent that has been chlorinated and dechlorinated are almost always much less than the drinking water MCL of 100 ug/l. Assuming a THM concentration of 10 ug/l in disinfected wastewater, based on available data, a dilution ratio of 20:1 in the receiving water, and no loss of volatile THM in the surface waters, the THM concentration added to the receiving water from the wastewater discharge after dilution would be 0.5 ug/l. Using EPA's risk assessment model, where 0.19 ug/l results in an incremental lifetime cancer risk of 1×10^{-6}, the cancer risk resulting from the THMs in the effluent would be 2.6×10^{-6} (.003/1,000). This risk number is also based on the assumption that all of the THMs discharged in the effluent remain in the receiving water and are present in the finished drinking water.

As can be seen from Issue No. 10, the most prevalent infection/disease risks associated with disinfecting to a fecal coliform limit of 200/100 ml are approximately twice as high as the risks associated with disinfecting to a total coliform level of 100/100 ml, using a risk model described in the review report. The total risk due to the formation of THMs during disinfection may be 0.003/1000 for a lifetime exposure. The portion of this risk due to meet a total coliform level of 23/100 ml versus a higher coliform density cannot be estimated because of the lack of data.

RECOMMENDATION: Disinfection of wastewater results in the formation of low levels of THMs, which are suspected carcinogens. The added THM formed by disinfecting effluent to meet the California recommended guidelines may be a small fraction of that formed in effluent disinfected at other levels. There is little documentation to support the argument that disinfection to low coliform levels, e.g., 23/100 ml, increases the overall health risk to the population.

APPENDIX C

TECHNICAL ADVISORY COMMITTEE AND COMMITTEE CHARGE

HEALTH POLICY COMMITTEE AND COMMITTEE CHARGE

WASTEWATER DISINFECTION GUIDELINES REVIEW
POLICY COMMITTEE

The Sanitary Engineering Branch, DHS, has been requested to conduct a thorough review and reappraisal of the guidelines which it uses to recommend sewage disinfection requirements to the regional water quality control boards. The guidelines were developed initially from 1972-76 based on information available at that time. The review request was made by the State Water Resources Control Board as a result of several challenges of disinfection requirements by waste dischargers. A number of issues to be addressed in the review were identified by the State and regional boards, the basic issue being the public health need for the degree of disinfection recommended by the Department.

A comprehensive review of information associated with sewage effluent disinfection has been completed and a report prepared. The report covers chlorination chemistry, cost and safety, health effects of microbiological contaminants, and health effects of organic chemical contaminants.

A technical advisory committee assisted in the development of information and in assessing the accuracy and completeness of the review report.

Based on the review report information, position papers have been developed for each of the issues that have been raised, and recommendations have been made. These will be used to substantiate changes or lack of changes in the uniform wastewater disinfection guidelines.

There is a need for a policy committee made up of public health authorities and experts which can advise the staff regarding the recommended positions on the issues that have been raised. The committee may also want to recommend changes or additions to the review report.

Charge to the Committee

1. Advise staff on the recommended position regarding each of the issues which have been identified.

2. Advise staff on recommended changes to the disinfection guidelines.

3. Advise staff on modifications to the review report.

Meetings

It is proposed to convene a meeting in Berkeley to receive the advice of the committee. A follow-up meeting will be held, if needed, to resolve conflicts or provide additional information. Contacts with individual committee members may be adequate in place of a second meeting.

All information will be submitted to the members sufficiently in advance of the first meeting to permit review.

1. Indicator organisms and pathogenic agents,
2. Waterborne disease,
3. Disinfection requirements and compliance,
4. Disinfection processes and efficiencies,
5. Treatment costs,
6. Safety considerations,
7. Seasonal disinfection considerations, and
8. Mineralization and carcinogen formation.

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WASTEWATER DISINFECTION GUIDELINES REVIEW
TECHNICAL ADVISORY COMMITTEE

DHS will conduct a comprehensive review of the guidelines which it uses to recommend disinfection requirements to the regional water quality control boards. The review has been undertaken at the request of the State Water Resources Control Board as a result of the challenge of a disinfection requirement based on the guidelines which were imposed by a regional board, and because of previous concerns with recommended disinfection requirements. Due to the statewide significance of this matter, a thorough reconsideration of the guidelines, including their technical validity, has been requested and has been undertaken.

An informal technical advisory committee has been proposed to assist the Department in its guidelines review.

<u>Charge to the Committee</u>

Advise and comment on the technical aspects of the review project, including:

1. Project scope and procedures,
2. Interim reports,
3. Progress, and
4. Findings.

Specifically advise staff on:

1. Technical information to be included in the review such as pertinent studies, reports, and other technical references;
2. Adequacy of coverage given to specific technical subjects; and
3. Validity of the technical information and findings in the review.

<u>Meetings</u>

It is proposed to convene a minimum of two meetings of the Technical Advisory Committee. The first will be directed at the proposed scope and content of the review project, and the second will be directed at the project report and its findings. In addition to the meetings, there will be substantial informal contacts and information exchange among the staff and committee members.

<u>Technical Subjects</u>

The Committee will be asked to advise the staff regarding various technical subjects, including:

APPENDIX D

PROPOSED WASTEWATER DISINFECTION GUIDELINES

UNIFORM GUIDELINES FOR WASTEWATER DISINFECTION

The Sanitary Engineering Branch, State DHS, has prepared guidelines for various wastewater discharge situations for health protection. The Branch was guided in this endeavor by a policy committee of public health professionals within the Department.

Discharge situations in California range from remote ocean discharges by means of submarine outfalls to discharges to dry stream beds which pass through residential and popular park and recreation areas. The concept employed in the guidelines appropriately incorporates consideration of the type and degree of disease exposure in the establishment of disinfection requirements. Both available dilution and the type of receiving-water use are considered in the bacteriological criteria.

GENERAL PROVISIONS

Determination of Median Total Coliform Numbers

The median total coliform bacteria number should be based on the last seven samples for which analyses have been completed. All coliform values represent total coliform.

Sampling Frequency

Where a median coliform MPN of 23/100 ml or 240/100 ml is required, bacteriological samples should be collected at least twice per week. Where a median of 2.2 is required, or a significant discharge is involved, samples should be collected daily.

Maximum Bacterial Limits

The basic disinfection criteria, in terms of median coliform bacteria numbers, designate the operating levels which should be achieved for the particular discharge situation. The designation of a median coliform bacteria requirement does not address the serious situation where little or no disinfection is provided for a limited period. A maximum coliform bacteria limit may be designated to provide for this; however, the maximum limit should be significantly higher than the median so that it will not be exceeded due to statistical variations in the coliform test or other factors. Consequently, it is recommended that the maximum coliform bacteria number should be the concentration which is approximately 20 times the median coliform bacteria number.

SPECIFIC GUIDELINES

<u>Category I</u>. <u>Proposed discharge is to</u>:

Lakes and reservoirs.

There should be no discharge of sewage effluent to a lake or reservoir used for domestic water supply.

For lakes and reservoirs used for recreation where year-round confinement to land is not possible, the wastewater should be:

A. Confined to land, except for wet weather periods during the nonrecreational season. When there is a discharge, the wastewater should be disinfected to a median MPN of 23/100 ml.

B. If effluent is discharged during periods of recreational use, the effluent must be an adequately disinfected, oxidized, coagulated, and filtered wastewater. The effluent shall be considered adequately disinfected if the median MPN of coliform organisms does not exceed 2.2/100 ml.

<u>Category II</u>. <u>Proposed discharge is to</u>:

Accessible drainage ways or ephemeral streams with little or no natural flow during all or part of the year.

Accessible drainage ways and ephemeral streams which received waste discharges are often attractive areas for planned or unplanned recreational activities involving water contact. Further, there is generally little dilution available during the summer recreational season. The recommended disinfection criteria are related to the degree of public exposure.

A. A Category II discharge occurs where there is no nearby habitation and limited use of the discharge area. These areas are generally not identified as having recreation as a beneficial use, and contact with the waste discharge is not encouraged. In these situations, access should be limited, if possible, and posting of the area may be appropriate if there is potential for recreational use.

   <u>Recommendation</u>: The effluent must have a median coliform MPN not exceeding 23/100 ml.

B. A Category II discharge occurs where there is residential development in the vicinity of the receiving area, and the RWQCB has <u>not</u> designated water-contact recreation as a beneficial use. There may be ready access to the discharge area; however, water-contact recreation is not encouraged. Posting of the area may be appropriate.

   <u>Recommendation</u>: The effluent must be an adequately disinfected, oxidized wastewater.

The wastewater shall be considered adequately disinfected if the median MPN of coliform organisms does not exceed 2.2/100 ml.

C. A Category II discharge occurs where the RWQCB has identified water contact recreation as a beneficial use of the receiving area and most of the following conditions are met:

1. The discharge occurs in, or upstream from, a residential area.

2. There is ready access to the discharged wastewater, and exclusion of the public is not a realistic alternative.

3. Historical attempts to post the stream to warn and exclude the public have been unsuccessful.

4. The recreation potential in the stream is high and justified because of weather, proximity to other recreation areas, etc.

5. There is public interest in recreational use of the water.

<u>Recommendation</u>: The effluent must be adequately disinfected, oxidized, coagulated, and filtered wastewater. The wastewater shall be considered to be adequately disinfected if the median MPN of the coliform organisms does not exceed 2.2/100 ml.

The Category II designations may not apply for discharges to agricultural drainage ways and sloughs which are remote or inaccessible. Such discharges may have less restrictive requirements depending on dilution and use.

<u>Category III</u>. <u>Proposed discharge is to</u>:

Freshwater streams and rivers.

There should be no discharge of sewage effluent to streams and rivers used for domestic water supply. The Department should recommend that no discharge be permitted to such water where land disposal or other options are physically or economically possible.

Where it is not possible to prevent a discharge to freshwater rivers and streams, the following disinfection recommendations apply:

Beneficial Use*	Ratio of Downstream Water Flow Effluent at Low Stream Flow***		
	<20:1	20:1 to 100:1	>100:1
Domestic water supply	No discharge	Median 23/100 ml ***	Median 240/100 ml ***
Swimming or other water contact	Use Category II.C ****	Median 23/100 ml	Median 240/100 ml
Agricultural use	Use Category II.C ****	Median 23/100 ml	Median 240/100 ml

* Beneficial use identified by RWQCB.

** The low flow is meant to be an average over a period of time, and not the instantaneous minimum low flow of the year.

*** If downstream domestic water supplies do not provide complete water treatment, a more strict disinfection requirement shall be recommended.

For discharge situations in this category, it may be appropriate to consider seasonal changes in use and dilution in recommending disinfection requirements. The recommended median MPN could be increased during the winter high-flow periods.

**** In situations where there is no dilution, the water reclamation criteria shall apply.

Category IV. Proposed discharge is to:

Saltwater recreation areas.*

Disinfection criteria for the protection of water-contact sports areas from discharges to ocean or bay waters are based on the degree of dilution which is available.

A. Ocean discharge is remote from recreational waters to the extent that bacteriological monitoring has documented that the discharge has no significant effect on recreational water quality.	No disinfection is required, provided that recreational waters meet Ocean Water Contact Sports Area Standards.

B. Discharge is near recreational areas and could influence recreational water quality. Dilution is $\geq$100:1.	Median MPN 240/100 ml.
C. Discharge is near recreational waters. Dilution is from 20:1 to 100:1.	Median MPN 23/100 ml.
D. Discharge is to or near recreational waters and there is little or no dilution (<20:1).	The effluent must be an adequately disinfected, oxidized, coagulated, filtered wastewater. Median coliform MPN 2.2/100 ml.

* Saltwater recreation areas such as lagoons and beaches situated at the mouth of ephemeral streams are subject to Category II requirements.

<u>Category V</u>. <u>Proposed discharge is to</u>:

Shellfish growing areas.

Shellfish growing areas in the vicinity of discharges constitute a particularly sensitive situation because of the ability of shellfish to concentrate contaminants. Outbreaks of hepatitis and other diseases transmitted by contaminated shellfish have been documented. In open coastal areas and some bays a high degree of dilution is available; however, in other bay systems, dilution is limited, and more restrictive disinfection requirements should be recommended.

A. <u>Ocean</u> discharge is remote from shellfish waters to the extent that bacteriological monitoring has documented that the discharge has no significant effect on shellfish-growing water quality.	No disinfection is required provided that shellfish waters meet 70/100 ml (or a fecal coliform density of14/100 ml). Median MPN 23/100 ml.
B. <u>Ocean</u> <u>or</u> <u>bay</u> discharge where a high degree of dilution* is provided; however, the discharge can affect the quality of waters overlying shellfish beds.	Median MPN 23/100 ml.

C. <u>Ocean</u> discharge is in the vicinity of a shellfish growing area.	Required "closed area" to assure a high degree of dilution.* Median MPN 23/100 ml.
D. <u>Bay</u> discharge where a high degree of dilution* cannot be assured by a "closed area".	Median MPN 2.2/100 ml and provision for maximum possible dilution and separation.

* A "high degree of dilution" means a dilution of 100 to 1 or greater.

A second barrier is the natural delay between the consumption of suspect water or shellfish, the occurrence and recognition of disease, and the investigation of water or shellfish quality. Routine monitoring may provide little, if any, information on bacteriological quality at the time of exposure, and a follow-up investigation may be too late.

Finally, the degree of public exposure, i.e., the duration and concentration of infectious agents, and the amount consumed, inhaled, or otherwise exposed to is rarely known.

1. Risk Assessment

An assessment of risk associated with virtually any activity is dependent on the amount and accuracy of information which is available for the assessment. When information gaps exist and indirect approaches are taken or assumptions are made, the validity and accuracy of the assessment may suffer or can be questioned.

With ideal conditions in place, an estimate of disease risk associated with recreation, consumption, and other uses of water would be a relatively straightforward process. This would require the following to be known:

- o Concentration of indicator organisms;
- o Ratio of pathogenic agent(s) to indicator organisms;
- o Disease incidence associated with the exposure to pathogenic agent(s);
- o Degree of exposure; and
- o Population susceptibility.

Other variables which may apply include the effectiveness of treatment processes on pathogens, relative accumulation or disappearance rates in the environment, and the effects of environmental factors.

There are information gaps and limitations in each of the listed areas.

- o No indicator organism or group has exhibited all the desirable characteristics of such an organism. The variety that have been used or proposed for use is evidence of their individual limitations.
- o There have been very few studies of actual pathogens <u>and</u> indicator organisms in the environment on which to base ratios. Also, analytical techniques are lacking for certain pathogens so that either no test result or only a qualitative result is available.

- For some organisms there is virtually no dose-response data (for Yersinosis, there is only a single, one-person feeding study of limited value). There is little agreement on the risk associated with low doses of microorganisms, and these are the doses most likely to occur.

- Amounts of water ingested may vary widely among individuals for the various uses.

- Susceptibility of a population to disease agents depends on factors such as age, general health, previous disease, etc.

2. Infective Dose and Dose Response

A basic factor involved in estimating the relationship of waterborne disease, and water quality is the effective dose or dose-response relationship, i.e., the relationship of the number of pathogens to the infection rate.

Initially, efforts were made to determine the "minimal infective dose" of various pathogens in order to estimate their significance and relative infectiousness. Often, the ID_{50} or infective dose which causes infection in 50 percent of the exposed subjects has been referred to as the minimal infective dose making it a somewhat misleading term. Early studies involved the feeding of known numbers of pathogens to volunteers to determine directly the least number which resulted in one infection. Direct comparison of the results of several studies of different pathogens is difficult because of the different number of volunteers, different susceptibility of the host, and limited number of test results.

Statistical procedures were devised to use the results of available dose-response tests to determine the probability of an individual infection. (See the Section "Estimates of Pathogen Concentration and Disease Risks" for a description of these procedures and their uses.)

The data presented here provide an indication of the relative health significance of various pathogenic agents. Two reservations apply. The terms "infection" and "disease" are not synonymous. An infection is the establishment and propagation of an agent in a host whether or not it results in detectable pathogenic effects. It may be demonstrated by antibody response, other subclinical effects, or the presence of the pathogens. Disease is a definite morbid process having a characteristic chain of symptoms, and its etiology may or may not be known. The difference may be substantial. For example, in determining the waterborne risk of disease from viruses, Pipes (1978) has estimated that one of every 100 infections may result in disease. Other estimates range from 1 to 10 to 1 in 1,000. Numerous feeding studies have been conducted to determine the "infective dose", and it is not always clear from summaries and secondary reports of the studies whether the response measured was infection or clinical disease. Also, the specific infection or illness rate may not be identified where general "infective dose levels" are presented. Insofar as possible, these limitations have been avoided in the following discussion. (A summary of dose-response information is included in EPA 1983.)

With regard to shigella, it has been concluded that as few as ten ingested organisms have caused illness in a significant percentage of volunteers (DuPont, H. L. 1973).

A summary of the results involving pathogenic bacteria is presented in Table 21 (Pipes 1978).

Table 21

Infective Doses of Enteric Pathogens

Enteric Pathogen Dose: Viable Cells	Subjects Infected/Total Tested								
	10^1	10^2	10^3	10^4	10^5	10^6	10^7	10^8	10^9
<u>Shigella dysenteriae</u>									
Strain M131	1/10	2/4	7/10	5/6					
Strain A-1		1/4		2/6					
<u>Shigella flexneri</u>									
Strain 2A		6/33	33/49	66/87	15/24				
Strain 2A				1/4	3/4	7/8	13/19	7/8	
<u>Salmonella typhi</u>									
Strain Quailes			0/14		32/116		16/32	7/8	
<u>Vibria cholerae</u>									
Strain Inaba									
With $NaHCO_3$				11/13		45/52		2/2	
No $NaHCO_3$				0/2		0/4	0/4	2/4	1/2
Enteropathogenic <u>E. coli</u>									
Strain 4608				0/5		0/5		4/8	

The following were the results of volunteer studies involving viruses:

Table 22

Virus	Dose	Route	No. Infected/ No. Inoculated	%
Polio I	2 PFU	Oral	2/3	67
Polio III	10 TCID	Gavage	2/3	67
Measles	1 TCID	Intranasal	8/35	24

NOTE: Polio I -- Antibody response
Polio III -- CPE from stool material
Measles -- Infection

The results of human volunteer studies reported in 1951 which determined the dose of various salmonella species and strains causing clinical disease in 50 percent of the subjects (healthy males) is presented in Table 23. Recent studies indicate that some salmonella serotypes (S. agona and others) have much lower infective doses.

Table 23

Dose of Various Species and Strains of Salmonella That Caused Disease in Human Volunteers (From McCullough and Eisele)

Salmonella Species/Strain	Dose at Which 50% or More Respond*
S. meleagridis I	50,000,000
S. meleagridis II	41,000,000
D. meleagridis III	>10,000,000
S. anatum I	860,000
S. anatum II	67,000,000
S. anatum III	4,700,000
S. newport	1,350,000
S. derby	15,000,000
S. bareilly	1,700,000
S. pullorum I	>1,795,000,000
S. pullorum II	>163,000,000
S. pullorum III	>1,295,000,000
S. pullorum IV	1,280,000,000

* Developed clinical disease.

A significant body of information has been added to the collection of infective dose data for enteric pathogens in the last few years. This data may have a major impact on the estimation of the health risks associated with enteric pathogens in wastewater since the new data tends to recognize lower infective doses.

The following minimum infective dose data has been gathered from several recent reports as well as some older sources dealing with infection of "normal" healthy adults (referenced):

Organism (Serotype or Strain)	Disease Associated	Minimum Dose for Infection	Minimum Dose for Disease	Reference
Campylobacter jejuni	Campylobacter enteritis		$5x10^2$	Robinson (1981)
Escherichia coli				
(0143:K?:H-)	*E. coli* enteritis	10^4	10^8	Dupont (1971)
(0111:B4)	*E. coli* enteritis		10^6 b	Dupont (1971)
(0144:K?:H-)	*E. coli* enteritis		10^6	Dupont (1971)
Entamoeba coli	Amebiasis	10^1		Rendtorff (1954)
Giardia lamblia	Giardiasis	10^1		Rendtorff (1954a)
Hepatitis virus	Hepatitis	$<10^1$	$<10^1$	Dolin (1971)
Salmonella				
S. agona	Salmonellosis	$<10^1$ c	Low c	Anon. (1971)
S. infantis	Salmonellosis	10^1	10^1	Silliker (1980)
S. schwarzengrund	Salmonellosis	$2x10^2$	$2x10^2$	George (1976)
S. cubana	Salmonellosis	$15x10^3$	$15x10^3$	Lang (1967)
S. eastbourne	Salmonellosis	10^2	10^2	D'Aoust (1975)
S. newport	Salmonellosis	$6x10^1$	$6x10^1$	Fontaine (1978)
S. typhi	Typhoid fever		10^4 b	George (1976)
Shigella				
S. dysenteriae (M131)	Shigellosis	10^1	10^1	Levine (1973)
S. flexneri (2a)	Shigellosis	10^2	10^2	Dupont (1972)
Vibrios				
V. cholerae (ogawa)	Cholera		10^3 c	Hornick (1971)
V. cholerae (inaba)	Cholera	10^3	10^4 c	Hornick (1971)
V. parahaemolyticus	Gastroenteritis		10^5	Sanyal (1974)

a 10 to 25 percent of human subjects infected.
b 75 to 100 percent of human subjects infected.
c Buffered stomach or ingested with food.

Data indicating infection without disease is important because those infected may become carriers and contaminate food or fomites. Data is lacking for some of the most recently recognized pathogens, including Aeromonas hydrophila, Listeria monocytogenes, Yersinia enterocolitica, and various helophilic Vibrio spp., other than V. parahaemoly ticus and V. cholerae. One must expect that the minimum infective dose for all of these agents can be lower in infants, the elderly, and in compromised or immunosuppressed individuals.

Cliver (1979) reported on an incident where healthy adults and children developed gastroenteritis after eating candy contaminated with less than 1 to 100 S. eastbourne organisms per 100 grams.

Overall, there is now an increased appreciation of the role of low concentrations of enteric pathogens in food and water outbreaks.

a. Factors Influencing Infective Doses and Dose Responses

Although infective dose studies on selected enteropathogenic strains provide useful information regarding the relative virulence of individual bacterial species and innoculum concentrations required to produce clinical symptomatology, values generated in such a manner must be viewed with some caution since data derived in this manner most likely represents an underestimate of the risk assessment to the population as a whole for a given microorganism. Infective dose data is most often obtained from seeding experiments (oral challenge) in healthy adult volunteers or laboratory animals and, as such, neglects several major subgroups of the general population who are at increased risk to developing serious-to-life-threatening disease via consumption of or contact with water or aquatic-related sources. These groups include those with defects in normal mechanical or physiologic barriers to infection and immunocompromised individuals.

Mechanical barriers, normally operative as major host defenses against infection via the gastrointestinal route, include the gastric secretion of the stomach, which usually lower the contents of this organ to a pH of 2. Such an acidic environment makes fluids entering this structure essentially sterile (Cantey 1985, Hill 1986). However, in a significant percentage of the population, this functional barrier is absent either due to hypo or achlorhydria, previous consumption of antacids, or persons who have had either partial or complete gastrectomies (Giannella 1973). This point is particularly highlighted by the fact that such individuals often succumb to infection of the bowel with microorganisms less commonly associated with producing disease in healthier persons such as Plesiomonas shigelloides (Roltson 1984, Martin 1986), Aeroomonas hydrophila (George), and Vibrio mimicus (Shandera 1983). In a similar fashion, parties receiving antimicrobials or who are malnourished may experience a change in their normal colonic microflora. Such alterations in the resident bacterial population, normally a barrier to gastrointestinal infection via competition for epithelial cell receptor sites and nutrients present in the gut, may enhance susceptibility to infection (Hentges 1986, Penn 1982). Other physiologic or mechanical barriers, which may play important rolls in regulating intestinal infection, but are presently poorly defined, include gastrointestinal mucins, peristaltic flow (motility), lyzozyme, hormones, and other factors released into the lumen (Cantey 1985, Smith 1985).

The most important group with increased risk to develop either gastrointestinal disease or disseminated infections from site are those with immunocompromised status. Persons included in this high-risk group are people with liver dysfunction (i.e., cirrhosis), diabetes, and those with a variety of hematologic or gastrointestinal malignancies. Individuals with liver disease, including alcoholic patients, suffer from a multiplicity of defects in their immune system with depressed complementmediated bactericidal activity, impaired mononuclear and polymorpho-nuclear phagocytic function, and diminished cell-mediated immunity (Raykovic 1985, Adams 1984). These individuals are more susceptible to a wide range of bacteria, including E. coli, streptococci, Pseudomonas aeruginosa, Aeromonas, Enterobacter, Salmonella, and Klebisella, and are prone to developing peritonitis and bacteremia. Similarly, diabetics with impaired blood flow often have chronic ulcers which may become infected, or they may present with more disseminated disease.

By far, the group with the highest susceptibility to infection within the immunocompromised category are those with underlying cancers (Luna 1985). Since modern surgical techniques and chemotherapy have either cured significant percentages of people with cancer or have dramatically prolonged their life expec-tancy, this group within the general population is increasing. These patients may either present with more severe and prolonged courses of diarrhea or, because of their immunologic deficit, may see traditional enteropathogenic organisms disseminate from their bowel with subsequent development of a bacteremia. Such infections have been repeatedly documented with a variety of agents including Campylobacter (Nachamkin 1984), Piesiomonas (Penn 1982), Aeromonas (Harris 1985), V. vulnificus (Curti 1985), V. alginolyticus (Bonner 1983, Janda 1986), V. cholera non-01 (Siegel 1982, Hughes 1978), V. hollisae (Lowry 1986), and others.

Finally, it should be mentioned that one must consider both the neonate and the elderly as immunocompromised populations since the former lacks complete maturation of their immune system while, in the latter instance, both cellular and humoral immunity are waning (Wilson 1986, Gardner 1980). Although these populations have not been studied as closely as, for example, cancer patients, it is highly likely that both groups run a higher risk to infection than does the normal (adult) population.

b. Dose Response

Dose-response information for various organisms is given in Table 24. This type of information (i.e., responses to several levels of organism doses) is needed to determine the probability of an individual contracting disease or infection when exposed to an identified dose of an organism.

Cooper (1984) presented information from the literature on dose-response for pathogenic agents. That report should be used for detailed information on dose-response. Presented here is an indication of the relative sensitivity to the disease agents.

- <u>Shigella</u> -- Ten organisms caused 10 percent illness; 100 to 5,000 organisms caused 18 to 70 percent illness.

- <u>Pathogenic E. coli</u> -- Greater than 10^4 organisms needed for illness; 10^8 to 10^9 organisms cause 60 to 100 percent illness.

- <u>Salmonella</u> -- Salmonella typhi in concentrations of 10^3/ml cause 1 to 9 percent response (illness).

- <u>Enteroviruses</u> -- As low as 2 PFU of Polio I virus caused infection as did one TCID-50 of Polio II.

- <u>Giardia lamblia</u> -- Ten cysts caused an infection in 1 of 1 volunteers. Over 100 cysts always resulted in 100 percent infection.

c. Risk Assessment Models

(1) <u>EPA Prospective Study Approach</u> -- The study plan undertaken in the development of criteria for recreational use of waters by EPA in 1983 and 1984 (discussed under Recreational Water) is one approach that can be taken to develop information on which to base risk assessment; however, there are some drawbacks.

The EPA prospective study approach involved the direct investigation of the relationship of illness caused by swimming to the density of indicator organisms in the water. Insofar as possible, independent variables are eliminated by the study conditions. Illness incidence of swimmers versus nonswimmers at the same beach are compared, follow-up illness information is carefully sought, "swimming" and "illness" is defined, etc.

Table 24

Data sets used for model comparison

Organism and dose	Responses: Positive	Responses: Negative	Disease or infection	Reference
Salmonella typhosa (Quailes)				
10^3	0	14	Disease	12
10^5	32	84		
10^7	16	16		
10^9	40	2		
Shigella dysenteriae 1				
10	1	9	Disease	13
200	2	2		
2,000	7	3		
10,000	5	1		
Poliovirus 1 (Sabin)				
7	0	1	Infection	14
16	0	2		
27	0	2		
42	0	1		
50	3	3		
55	1	2		
65	0	6		
80	1	0		
90	3	1		
160	3	0		
210	2	0		
280	1	0		
Echovirus 12				
10	6	26	Infection	15
30	2	5		
100	14	7		
Shigella flexneri 2A#				
180	6	27	Infection	16
5,000	33	16		
10,000	66	21		
100,000	15	19		
Shigella flexneri 2A##				
10^4	1	3	Infection	16
10^5	3	1		
10^6	7	1		
10^7	13	6		
10^8	7	1		
Poliovirus 1 (Sabin)				
3,162	28	69	Infection	17
31,600	42	54		
316,000	48	36		
Entamoeba coli				
1	1	7	Infection	18
10	3	7		
100	2	2		
Poliovirus 3 (Fox)				
1	3	7	Infection	19
1.5	3	6		
10	2	1		

Based on the study results, a relationship can be established between the indicator density and illness. In the case of the EPA recreational water studies, correlation coefficients were determined for illness rates and indicator bacteria, and regression equations were developed for the illness indicators which showed the closest fit. (From data presented later, the studies implied that national standards are allowing a disease rate of up to 19 illnesses per 1,000 swimmers.)

One of the potential drawbacks to such a study effort is the sizeable cost and the lack of assurance that a reasonable relationship will be forthcoming at the water quality levels employed. In this case, a relationship was found between certain indicator organism densities and disease rates. An expected deficiency is the lack of information on the agent(s) which may be responsible for the illness.

(2) Pathogen Dose-Response Models -- Under the EPA recreational water study approach, the relationship of the concentration of pathogenic agents to disease rate is not determined. However, where this relationship is to be used to determine the risk of infection or disease, two hypotheses have been proposed.

Under one, the host organism (person) is assumed to require a minimal infective dose in order to exhibit a response (illness or infection). If a population is exposed to a given dose, and if the minimal infectious dose is log-normally distributed, then the fraction of the population which exhibits a response can be expressed by a log-normal model.

The second hypothesis views infection or disease as a two-step process. First, the host ingests one or more organisms which are capable of causing disease. The organisms are reduced by the host's defenses, and only a fraction arrive at the target site where infection or disease can occur. The hypothesis asserts that only a single organism need survive until it can infect and, if organisms are distributed in water in a random manner, the probability that a certain number will be ingested can be computed, and the "single hit" risk can be determined from a simple exponential model. Finally, the model has been refined to consider the variability of the infectivity of an organism (or the susceptibility of the host) so that the observed slopes of the dose-response curves are more in line with the values predicted by the model. This has been termed the beta-distributed model. A more detailed discussion of the three dose-response models is presented by Hass (1983). Doseresponse information from the literature was used by

Hass to determine the best fit distribution parameters for the 3 models and the ID 50 (infections dose for 50 percent of the population) for a number of pathogens.

Of nine pathogen data sets, the beta distribution model was consistent with seven, the log-normal model fit five, and the simple exponential model fit three. There was a reasonable relationship among the predicted 50 percent infectious doses of the 3 models. These are within the range of where the actual dose response data was available. The predicted dose-response results begin to differ between the log-normal model and the beta distribution model at low dose levels, that is, the log-normal model predicts a lower risk due to an exposure to doses less than one organism. Based on the beta distribution model, the extrapolated risks to an individual due to a single exposure to 10^1 organisms was as follows:

S. dysenteriae I (disease)	5×10^{-2}
S. flexneri 2A (disease)	1.47×10^{-2}
ECHO virus 12 (infection)	1.73×10^{-1}
Polio Virus I (Minor) (infection)	1.5×10^{-1}
Polio Virus I (Lepow) (infection)	6×10^{-3}
Polio Virus III (infection)	4.1×10^{0}
Ent. coli (infection)	1.2×10^{0}

Hass concluded that, due to the "satisfactory fit of the beta model to such data, it is impossible to rule out the hypothesis that one organism, when ingested, can cause infection and/or disease in at least a portion of the exposed population..." and "...the use of the stochiastic type of model provides a conservative estimator of risk associated with low dose exposure in comparison with log-normal model."

Although the results of virus dose-response studies have been reported as early as the 1940s, one of the more definitive coverages was presented at a symposium in 1965 sponsored by the U. S. Public Health Services and was directed at virus diseases (Berg 1967).

Animal as well as human virus studies were reviewed and, based on the results of volunteer testing, Plotkin and Katz concluded that "...one infective dose of tissue culture is sufficient to infect man if it is placed in contact with susceptible cells." It should be noted that this was based on tests where viruses were introduced by various routes only one of which was the oral route. Beard, in the same publication, pointed out the wide variability that may be exhibited in a host's response -- from complete resistance to high susceptibility depending on the host and possibly on the virus.

Cooper et al. (1984), carried out an "Assessment of Risk Associated with Water-Related Infectious Agents". The assessment was directed at determining the percent of a small number of troops which might become ill when ingesting water that contained pathogenic agents. In order to gain a data base for the assessment, a thorough review of the literature was conducted for information on pathogenic agents -- bacteria, viruses, and parasites. In particular, information was sought on their occurrence and concentration in the environmental, dose-response relationships, and indicator organism-pathogen relationships.

The following information was used as input to a risk assessment model:

- o Number of random samples;
- o Water volume consumed;
- o Water treatment efficiencies;
- o Pathogen concentration and standard deviation;
- o Dose response equation;
- o Dose response coefficients (based on available dose response information); and
- o Number exposed.

The report presents the detailed procedure used to develop the risk curves in the assessment and is not repeated here. Briefly, the volume of water consumed, treatment, and pathogen concentration are initially calculated and, based on these factors, the dose is determined. Using one of four dose-response equations from the literature, the expected response is computed.

Table 25 summarizes the <u>relative</u> risk associated with some of the pathogens assessed in the report for developed countries.

Table 25

Relative Disease/Infection Risk For Pathogenic Agents
Developed Counties
(Number is the Percent or Less That Would
Be Ill 50 Percent of the Time)

Pathogen	Low Risk Water Treatment	High Risk (No Treatment)	Pathogen Conc./Liter (Geometric/Mean)
Shigella	8	94	100
V. cholera	0	0	32
Campylobacter	4	95	100
Salmonella	8	92	172
Yersinia	63	93	100
E. coli	29	50	2,000
Enterovirus	8	94	113
Ent. histolytica	3	94	13

Example: For shigella, there is a risk with water treatment that 8 percent or less of exposed persons would be ill 50 percent of the time with 100 org./liter in the water source.

High Risk Assumes: 0 percent treatment efficiency, 15 liters + 10 percent consumed. Low Risk Assumes: 99 to 99.999 percent treatment efficiency, 10 liters + 10 percent consumed.

Although the situation assessed is not translatable to normal waste discharge and domestic water supply conditions in California, the results underscore the importance of treatment reliability for domestic water supplies.

By using the Kerr and Butterfield relationship between typhoid morbidity and the ratio of total coliform to S. typhi, and an estimate that only 5 percent of illness due to salmonella and shigella is reported, it was estimated that 95 percent of the time, 5 percent or less of persons consuming treated water with a coliform density of 54/100 ml would contract salmonellosis and 5 percent or less would contract shigellosis with a coliform density of 159/100 ml. Recognizing the many estimates and assumptions involved in such an exercise, the result does suggest that there may be concern with ingestion of waters containing relatively low levels of indicator organisms.

A risk assessment study, conducted by Colorado University at Denver 1985 for EPA, considered the effect on human health of discharging effluent that had not been disinfected. Initially, two dose-response models from the literature for salmonella were employed (Mechales et al. 1972, and EPA 1979); however, the lack of agreement on the relative risk generated by the two models led to the use of a quantitative regression model developed by Cabelli (1981) for enterococci corrected by an assumed die-away of organisms to the point of use.

This approach assumes no disinfection, a travel time of two days to the point of recreation, and a die-away rate constant. Where disinfection is employed, the model would have to consider regrowth and, possibly, a much shorter travel period to reflect actual conditions and a conservative risk assessment.

Under the Denver model, with a 10 to 1 dilution of water to sewage and an enterococci concentration of 1,000/100 ml, the health risk would be 33 cases of gastrointestinal distress per 1,000 swimmers. If swimming took place immediately downstream from the discharge, the risk would be 47 per 1,000 swimmers.

Hutzler and Boyle (1980) suggest an approach to dose response which depends on a number of assumptions or estimates. They recommend a linear extrapolation from a known point on a dose response curve to zero. A factor of conservatism is provided by starting from the upper confidence limit of the response at an actual experimental dosage. While easy to apply, the approach does not consider dose distribution and other factors in dose response models which may more accurately reflect low dose responses.

Hepatitis A virus (HAV) was the pathogen of concern in their exercise. Inasmuch as no direct information on virus concentrations was available, the amount of infected feces fed to volunteers was plotted as the dose versus the percent of volunteers with clinical symptoms. The 95 percent upper confidence limit at a dosage of 0.1 grams of infected feces was then computed.

A number of estimates were made to arrive at the possible dose of HAV in the environment.

These included:

- o The number of persons excreting HAV annually based on reported cases, an estimate of unreported cases, and those having an infection but not the disease;

- o Fecal output per day and average wastewater production;
- o Sewage treatment and chlorination effects, dilution, and survival in the environment;
- o Water treatment effects; and
- o Volume of water ingested during swimming or drinking.

The risk under this approach is considered minimal -- 0.004 cases of hepatitis A per 100,000 persons per year for swimming and 0.06 cases per 100,000 persons per year through surface water domestic supply.

The relationship of various organisms and illness rates was plotted by Mechalas (Figures 5 and 6). The validity of the virus-disease relationship has been questioned because the dose-response data was derived from a variety of sources and dealt with a variety of viruses whose route of introduction was via the respiratory tract. This was the model which the Denver study initially attempted to use and then discarded.

The results indicate that there can be very significant differences in estimated risk depending on the approach taken, the available information, and the assumptions made.

3. Drinking Water

Information on waterborne disease in the United States associated with drinking water is available in summary fashion for periods of years dating back to 1920 (Lippy 1984, Craum 1981). In addition, the medical journals and other scientific periodicals contain the details of epidemiological studies carried out for individual waterborne outbreaks. The following are significant items associated with this route of disease.

a. Disease Outbreaks

From 1946 to 1980 (35 years) 672 outbreaks were reported involving 150,000 persons. Based on results during improved reporting procedures, there may have actually been 1,330 outbreaks involving 350,000 persons. Peak occurrences take place in June, July, and August, apparently due to noncommunity water systems serving camps, restaurants, and recreational areas. Outbreaks associated with community water systems averaged 506 cases of illness per outbreak. The reported death rate is one per year, nationally.

5

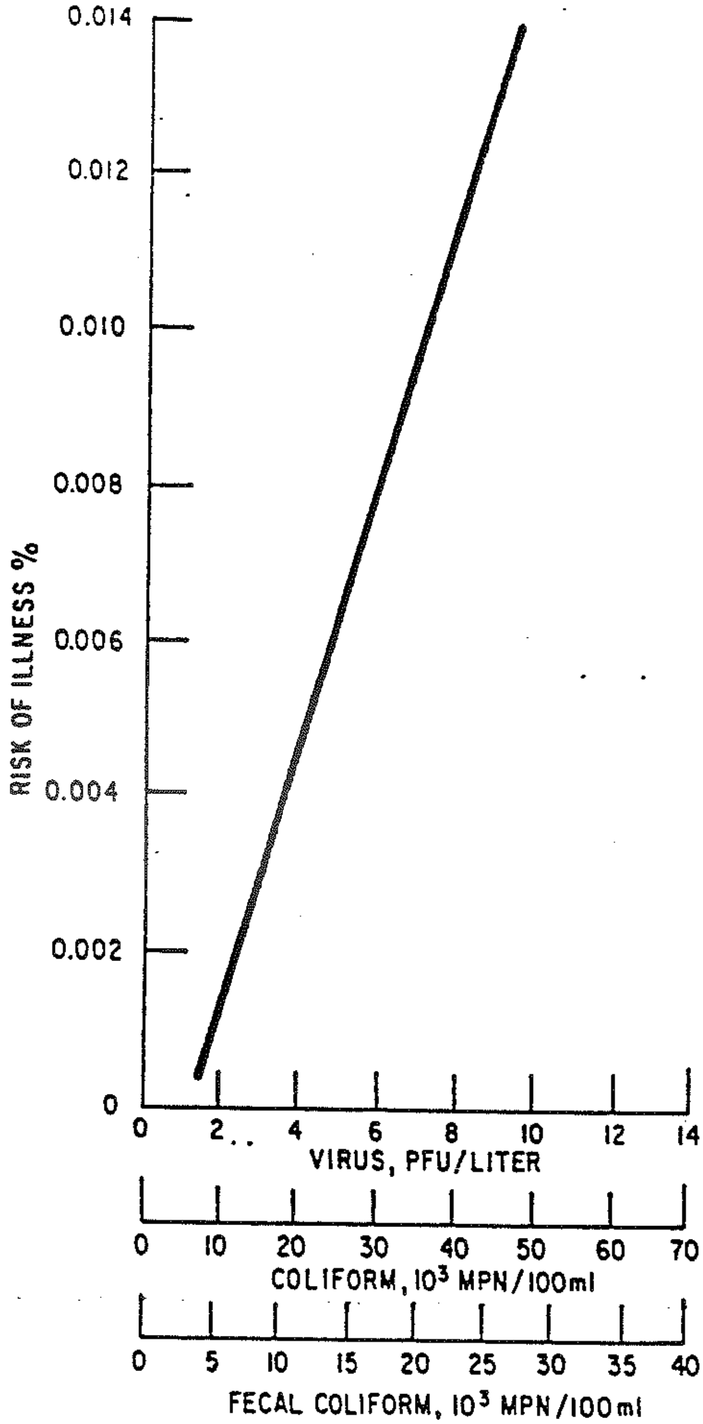

Fig. 5 Relationship Between Disease Risk and Viruses, Coliforms, and Fecal Coliforms

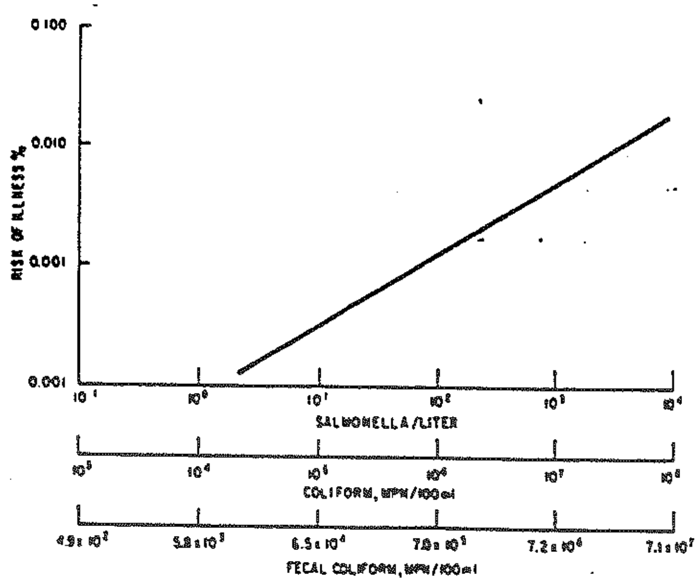

Fig. 6 **Relationship Between Disease Risk and *Salmonella* Coliforms, and Fecal Coliforms**

b. Disease Agents

Microbiological agents are rarely detected in water (less than one percent of the outbreaks) but are more often determined through stool or blood specimens. For more than half the reported outbreaks, no agent was identified due to the lag time between disease recognition and investigation and the limits of agent identification techniques.

The most commonly identified pathogen associated with surface water supply outbreaks was Giardia lamblia with 24 outbreaks from 1971-1978. Eighteen were caused by untreated surface water or surface water treated by simple chlorination. Three outbreaks resulted from ineffective filtration of surface water.

Parvovirus-like agents and possible other viruses not identified by common tissue culture techniques are suspected as etiologic agents in a portion of the outbreaks of unknown cause.

The following causative agents of waterborne disease (Table 26) have been associated with outbreaks in the United States (Lippy 1984):

Table 26

Agent	Outbreaks	Cases of Illness
Bacterial		
Campylobacter	2	3,800
Pasteurella	2	6
Leptospira	1	9
E. Coli	5	1,188
Shigella	61	13,089
Salmonella	75	18,590
TOTAL	146	36,682
Viral		
Parvovirus-like	10	3,147
Hepatitis	68	2,262
Polio	1	16
TOTAL	79	5,425
Parasite		
Entamoeba	6	79
Giardia	42	19,734
TOTAL	48	19,813
Unknown or Chemical	399	88,555

Cooper (1984) provided information on the attack rates associated with U. S. outbreaks associated with drinking water. The information is given in Table 27.

Table 27

U. S. Outbreaks

Organism	AR*
Shigella	437 720-920 185 180 681
Campylobacter	200 126-627 400 83
S. typhomurium	100
Salmonella	167
Yersinia	410
E. coli	380-825
Enteroviruses	
Polio ECHO	100 540

* AR = Attack Rate/1,000

c. Cause of Disease

An evaluation of data from 1946 to 1980 indicated 8.3 percent of outbreaks (4.7 percent of illness cases) were due to the use of contaminated, untreated surface water; and 27.2 percent of outbreaks (39 percent of illness cases) were due to inadequate or interrupted water treatment.

d. Quality of Waters During Disease Incidence

The following examples of specific disease outbreaks associated with domestic water supply indicate the difficulty of relating outbreaks to pathogen or indicator organism densities.

1. Bennington, VT -- 1976

   Approximately 3,000 illnesses of bacterial dysentery. Town water supply was implicated. Routine bacteriological testing failed to show positive coliform results.

2. Pico Rivera, CA -- 1971

   3,500 illnesses of gastroenteritis. "Heavy" fecal coliform bacteria contamination and salmonella contamination from an unknown source.

3. Comerio, PR -- 1976

   2,105 cases of shigellosis. Well water had total coliforms of 4,900/100 ml and fecal coliforms of 23/100 ml.

4. Ski Resort, VT

   750 cases of gastroenteritis. Well water had coliforms ranging from 1/100 ml to >16/100 ml.

e. EPA Water Quality -- Disease Study

The current status of the quality-disease relationship by the water supply route is indicated by the recent EPA statement (Federal Register 1986) that its <u>future</u> plans call for the examination of the relationship between bacterial indicator systems and health effects with water consumption.

In conducting studies of recreational water quality and health effects, however, EPA has generated information which might be translated to health effects and the quality of water used for domestic water supply.

EPA has related the incidence of gastrointestinal illness to swimming in recreational waters of different qualities. Reportedly, all of the waters met the quality standards for primary recreation. Approximately 6 illnesses per 1,000 swimmers were associated with fresh waters, which were influenced by waste discharges, and which fell within water quality standards of 200/100 ml fecal coliforms. (The illness rate presumably is associated with the incidental ingestion of water during swimming which has been estimated to range from 10 to 50 ml.) Domestic water systems employing simple chlorination, which use a similar quality of water as a source of supply, might be expected to have a much greater illness rate in the event of a treatment interruption due to the significant difference in water ingestion in the two uses.

f. Standards -- Source of Supply

A recommendation on the bacteriological quality of surface water used as a source of domestic water supply is contained in "Water Quality Criteria -- 1972". The document was prepared by the National Academy of Sciences and the National Academy of Engineering for the EPA (EPA 1972). The recommended criteria were based on studies of untreated water at intakes along the Ohio River and Missouri River in the early 1970s. It was specified that the water would receive the following treatment at a properly operated plant:

(1) Coagulation;

(2) Sedimentation;

(3) Rapid sand filtration; and

(4) Disinfection with chlorination.

Also, prechlorination as well as postchlorination may be required when the recommended quality limits are approached.

Recommended maximum bacteriological quality:

- Geometric mean density, fecal coliform 2,000/100 ml.
- Geometric mean density, total coliform 20,000/100 ml.

It should be noted that the water supplies used in the development of the source water recommendation serve large municipalities and provide complete water treatment. No reappraisal of bacteriological limits for source water quality has been conducted, and no health affects investigation was conducted to attempt to relate health to source water quality.

The U. S. Public Health Service had recommended a standard of 50 coliform/100 ml for surface water serving water supplies which received only chlorination. The DHS established a guide for the treatment of surface waters for domestic use (DHS 1973). Where only disinfection is provided, the water should originate from a watershed closed to general public use and on which all activities (cattle grazing, logging, etc.) are carefully controlled. The coliform bacteria content of the raw water should not exceed a median MPN of 50/100 ml over any 3-month period. Where filtration and chlorination are provided, the median MPN should not exceed 500/100 ml. Where complete water treatment is provided, the raw water quality should not exceed 20,000/100 ml coliform bacteria.

In assessing the need for mandatory filtration and disinfection of surface water, EPA (1985) pointed out that 17 percent of the total number of surface water systems which provided only disinfection had 35 percent of the disease outbreaks and 44 percent of the illnesses.

Water filtration with appropriate pretreatment generally removed 90 to 99.9 percent of bacteria, protozoan cysts, and viruses whereas filtration without pretreatment is ineffective in the removal of these organisms, and disinfection alone is only effective against bacterial agents.

g. Standards -- Treated Water Quality

A bacteriological standard employing the coliform group of bacteria was first adopted for the quality of water delivered to the consumer in 1914 in the federal regulations for interstate carriers. Essentially, the standard called for the absence of total coliform bacteria (mean density of 1/100 ml per month), and although there have been refinements in procedures, the concept and standard has remained the same. It is now applied to all public water supply systems serving more than 25 customers through state enforcement programs.

Several researchers have pointed out that for surface water supplies which receive chlorination but no filtration, the coliform bacteria indicator may not be adequate to signal the presence of Giardia cysts.

As a result of these concerns and the record of Giardiasis outbreaks associated with chlorinated surface water supplies, EPA has held a workshop to obtain advice, comments and recommendations on monitoring, filtration and disinfection needs which may lead to revised regulations.

The regulations may require effective filtration and disinfection for surface water supplies (EPA 1985).

In addition, problems expressed with the total coliform standard include:

(1) Test delay time of 24 to 96 hours;

(2) Insufficient test coverage for small systems;

(3) Need to employ other control measures (chlorine residuals, sanitary surveys); and

(4) The current total coliform bacteria test may enumerate less than ten percent of the viable coliform bacteria population present due to water disinfection resulting in injured but viable coliform.

A symposium sponsored by EPA in 1978 (EPA 1978) found general support for the coliform bacteria standard for treated water.

McFeters (EPA 1978) pointed out that for drinking water quality measurement, an indicator that is more encompassing is better than one that is more restrictive, and where this is a consideration, total coliform is better than E. coli.

The occurrence of disease outbreaks over several decades of recorded illness associated with water supplies were cited to suggest a possible disease risk although the outbreaks were most often associated with the period of noncompliance with the standard. The continued occurrence of diseases such as hepatitis, Giardiasis, and gastroenteritis suggested a need for further investigation of the current bacterial standard.

A virus standard of one detectable infectious unit per 100 gallons has been suggested, and a WHO Scientific Group concluded that no viruses should be detected in samples between 100 and 1,000 liters (WHO 1979).

4. Recreational Water

There has been a significant debate regarding the role of recreational water quality in the transmission of waterborne disease. This exposure route is the one which has received the greatest amount of planned study over the years.

a. Disease Outbreaks

As with drinking waters, the results of routine bacteriological samples have rarely indicated the quality of the recreational waters at the time of an outbreak. However, the record of disease outbreaks and the resultant follow-up investigations have revealed some general trends.

Dufour (1984), in summarizing swimming-associated outbreaks (see Table ~~28), noted that:~~

(1) There has been a marked decline in disease outbreaks associated with salmonella, and none have been reported after 1958. This was attributed to improved sewage treatment (although there are many countries where disinfection is not practiced).

Table 28

Outbreaks of Disease Associated With Swimming in Natural Waters

Year	Location	Etiologic Agent	Water Quality	No. Cases at Risk
1909	Walmer, England	Salmonella	U*	34/NG
1921	Connecticut	Salmonella	U*	6/NG
1932	New York	Salmonella	U*	51/NG
1942	California	Salmonella	U*	NG/NG
1947	Beccles, England	Salmonella	U*	9/NG
1958	Perth, Australia	Salmonella	U*	15/NG
1973	Vermont	Coxsackie B	U	21/33
1974	Niort, France	Coxsackie A_{16}	E. coli 50-1000/ 100 ml	5/NG
1974	S. Carolina	Hepatitis A	U*	14/30
1976	Iowa	Shigella	U	31/45
1982	Michigan	Norwalk Agent	U	126/NG

U -- Water quality not measured
* -- Suspected to be grossly polluted
NG -- Not given

(2) Later outbreaks have been virus-associated disease and one shigella outbreak. (The infectious dose of shigella reportedly may be as low as ten organisms.)

(3) No outbreaks of poliomyelitis have been confirmed, although this was a major concern before the advent of a vaccine, and the lack of an outbreak in the last few decades may be evidence of the effectiveness of the vaccine.

b. Health Effects Studies

The following are brief summaries of studies reported in the literature which were directed at relating illness to recreational water quality.

<u>1954 to 1956, England and Wales</u>

A study by Moore (1959) compared swimming experience of children with poliomyelitis with those that did not. No relationship of swimming to disease (poliomyelitis) was found. Very few cases of salmonellosis, which could be associated with swimming in polluted water, were found.

Cabelli (1983) has criticized the widely quoted conclusion by Moore, i.e., "There is little, if any, risk of enteric diseases from swimming in sewage-polluted waters unless aggregate fecal material is found therein, and, aesthetic considerations will limit beach usage long before there is a significant risk of swimming-associated enteric disease."

For the two diseases, Cabelli indicated that this may be accurate but the application to all enteric disease is questionable because of the limit of the test and uncertainty regarding "swimming", "pollution", and procedures used.

<u>1954 to 1959, England and Wales</u>

No evidence could be found that seaside residents had a higher rate of enteric disease than the nation as a whole (Committee, P.H., Lab Services 1959). The study was directed at typhoid and paratyphoid cases only where the organisms could be found in the water, and contained other requirements. With the above two studies, criticism could be leveled that sufficient criteria were not applied to provide a health-swimming assessment, and similarly, could be leveled that the criteria that were applied were extremely difficult to meet.

<u>1977 Madison, Wisconsin</u>

The study compared the swimming experiences of several hundred well and ill children and concluded that enteroviral disease was 3.4 times that in swimmers as in nonswimmers. The ratio was even higher among the very young. Other means of transmission, such as person to person, were not excluded.

<u>1948 to 1950, USPHS Inland and Marine Water Studies</u>

Studies to determine swimming-associated illness was conducted at Lake Michigan, Ohio River, and Long Island (Stevenson 1953). At the Ohio River location, gastrointestinal illness was more frequent among river swimmers when the median total coliform density was approximately 2,400/100 ml. A wider range of illness symptoms was detected among swimmers than nonswimmers at Chicago when the geometric mean density was 2,300/100 ml.

The study was criticized because of a use of a calendar which was filled out by the families, poor definition of "swimming", and the use of a swimming pool for the "clean water" group.

Later, in the mid-1960s, it was determined that the fecal coliform content of the river was approximately 18 percent of the total coliforms. This led to the recommended fecal coliform standard of 200/100 ml (employing a safety factor of 2x) which was not only criticized because of the study work, but also because of the lag time and assumption regarding the indicator organism ratio.

<u>1979 to 1982, EPA Fresh Water Studies</u>

EPA studies of freshwater swimming areas (Dufour 1984) indicated a relationship of "highly credible" gastrointestinal symptom rates associated with swimming and the enterococcus and E. coli indicator organisms in recreational waters. Study results indicated approximately a 6 per 1,000 illness rate for swimmers over nonswimmers at a geometric mean density of 77/100 ml for E. coli and 20/100 ml for enterococci. The log mean fecal coliform content for all areas was 115/100 ml.

<u>1972 to 1979, EPA Marine Water Studies</u>

EPA studies of marine water swimming areas (Cabelli 1983) indicated a relationship of "high credible" gastrointestinal symptom rates associated with the enterococci indicator organism in recreational water. Study results indicated that in waters meeting the criterion of 200 fecal coliform per 100 ml, risk levels of 15 gastrointestinal illnesses per 1,000 swimmers existed at an enterococci density of 16/100 ml.

The EPA studies have been criticized on a number of points by the Association of Metropolitan Sewerage Agencies (1984) including:

(1) There are anomalies in the epidemiological results in certain freshwater and saltwater studies.

(2) Failure to collect fecal coliform data for all cases.

(3) Pollution sources not defined for saltwater.

(4) MPN procedure instead of MF technique used for saltwater quality.

c. Development of Recreational Water Standards

As pointed out by Cabelli (1983), the history of recreational water quality guidelines or standards has proceeded from the attainable in terms of pollutant levels to detectable risk and is now entering the area of acceptable (or at least determinable) risk. Along the way, there have been highly respected scientists expressing either no concern with recreational water quality or the need for strict water quality standards. The general trend has been to suggest more strict water quality standards as more definitive information on illness and water quality has become available. Each study and recommended guideline or standards that has been put forward has received substantial criticism.

(1) California Saltwater Bathing Standard

The current California standard for ocean water recreational areas was established over 40 years ago. It provides that not more than 20 percent of the samples in any 30-day period shall have a coliform bacteria content of more than 1,000/100 ml, and no two consecutive samples shall exceed 10,000/100 ml. The standard was based in part on aesthetics. Where the standard was significantly exceeded, there often was sewage grease on the beach.

An extensive survey of water quality along the shoreline of Santa Monica Bay was carried out in 1942 (DHS 1943). The survey was conducted to determine the effect of waste discharges from the City of Los Angeles on water quality. At the time of the survey, the sewage received a minimum treatment consisting of screening through rotating drums screens having 1/16" x 2" openings.

Only 10 tons of 170 tons of suspended solids received daily ~~was removed, and very little grease was removed.~~ Bypassing of the screens occurred at least several days per month.

The data showed the water to be significantly polluted by the discharge, there were "sewage sleeks" in the water, and there was the presence of sewage grease deposits along the beach.

For a number of years a "standard" of 10 "E. coli" (coliform) per cc had been used prior to the Santa Monica Bay survey, and where the water quality greatly exceeded this standard, the survey indicated physical evidence of sewage contamination. In the survey report (DHS 1943) and a later memorandum (DHS 1949) the justification was given for using 1,000/100 ml coliform (or 10 "E. coli" per cc) as the standard for ocean recreation areas.

(a) The limit is about 500 times the pollution allowed by the U. S. Public Health Services in drinking water. Comparing relative ingestion of drinking water and water while recreating (2 to 3 cc), the two harmonize reasonably.

(b) There has been no well-defined epidemiology to indicate that water within the standard caused illness.

(c) The standard is at a level which should not confiscate recreational rights.

(d) Any less severe standard where there is raw or screened sewage would show "approved" areas within visible sleek fields of sewage and would appear to a layman as lacking in common sense and decency.

(e) Other states have adopted similar or more strict standards.

The standard allows an overrun of 20 percent of the samples in any 30-day period and provides that no two consecutive samples should exceed 10,000/100 ml coliform.

(2) Moore Conclusions

A retrospective (after-the-fact) epidemiological analysis of the risk of illness (principally poliomyelitis and secondarily, salmonellosis) carried out by Moore 1959, indicated no greater association of swimming in polluted waters among children ill with poliomyelitis than a non-swimming control group. Also, there were very few cases of salmonellosis where swimming in polluted waters might have been implicated. Based on the study findings, a broad and often-quoted conclusion was drawn that there is little, if any, risk of enteric disease from swimming in sewage-polluted waters unless aggregate fecal material is found therein, and that aesthetic consideration will limit beach usage long before there is a significant risk of swimming-associated enteric disease. The record of reported disease outbreaks associated with swimming in polluted waters would tend to support the conclusion for the two illnesses investigated. None of the few reported outbreaks of poliomyelitis have been associated with recreational waters, and only grossly polluted waters have been associated with salmonellosis.

The Moore study has been criticized because:

(a) The term "swimming" was not defined;

(b) The quality of the water was not determined;

(c) The diseases investigated may not be the ones of concern; and

(d) The lag between swimming and reporting was often protracted.

(3) Melnick Recommendation

Because the available information indicates that the infectious dose for viruses may approach one PFU, strict virus quality standards have been proposed for recreational waters. The recommendation most often cited is that of Melnick 1976, who recommended a maximum of 1 detectable infectious virus unit per 10 gallons of recreational water and 1 per 100 or 1,000 gallons of drinking water. Aside from the concern with the low infectious dose, the recommendation appears to be based in part on attainability inasmuch as Melnick concludes that we should "...accept the principle that the community is entitled to drinking water that is as free from virus as modern technology can provide." This viewpoint has been attacked as both unreasonable and unnecessary. A lack of clear documentation of the role of the water route in low level disease transmission has been cited in opposition to such proposed virus standards in the past.

(4) 1976 EPA Guideline

In 1976 the EPA recommended a guideline for recreational waters for a log mean fecal coliform content of 200/100 ml and not more than 10 percent to exceed 400/100 ml during a 30-day period. The standard was based on the inland water studies carried out by Stevenson which demonstrated a detectable health effect at levels of 2,300 to 2,400 coliforms per 100 ml. This was translated some years later to a fecal coliform level of 400/100 ml based on comparative total coliform/fecal coliform tests along the Ohio River. A safety factor was added to arrive at the 200/100 ml fecal coliform guideline.

Later, Geldreich (1970) found that when the fecal coliform content of fresh water exceeded 200/100 ml there was a significant increase in the isolation rate of salmonella.

This was concluded to provide evidence of support for the recreational water standard. The Stevenson studies have been criticized for a number of reasons:

(a) Swimming was not sufficiently defined:

(b) The control group of nonswimmers was not at the beach:

(c) Only one indicator, total coliform bacteria, was used; and

(d) The translation from total coliform to fecal coliform years after the study was questionable.

In spite of the deficiencies, the studies did indicate a relationship between recreational water quality and health and suggested an indicator organism level at which a detectable level of illness could be anticipated.

(5) EPA Marine Water Criteria

In 1983, the findings of a multiyear study of health effects associated with marine recreational water was reported by EPA (Cabelli 1983).

One conclusion of the study was: "...When the study design for the EPA program was being developed in 1969-70, it was thought that swimming in sewage-polluted waters would constitute a relatively minor route of transmission for gastrointestinal illness and that relatively high levels of pollution (as indicated by microbial indicator densities) would be required before gastrointestinal illness could be detected. These assumptions were made on the basis of existing notions and available information. Both these assumptions were incorrect."

The report concludes that swimming in "sewage polluted" waters constitutes a significant route of transmission for the illnesses obtained (gastrointestinal illness). It is important to note that <u>all</u> swimming areas that were studied met the current EPA criteria of 200 geometric mean fecal coliform per 100 ml, and the term "sewage polluted" is somewhat misleading.

The report recommends that the most responsible use of the criteria presented in the report is their translation into effluent guidelines governing the design, location, treatment, and disinfection requirements for sewerage facilities. The report, however, does not provide the information in terms of sewage effluent disinfection and receiving-water dilution/time factors which would assist development of such guidelines.

The study attempted to avoid the perceived deficiencies of the Stevenson work by defining the term "swimming", using nonswimmers at the beach as the control group, obtaining more precise information on health effects, and using multiple indicator organisms.

The study was carried out at beaches in New York, Boston, and Louisiana. Indicator organisms used in the study were:

Coliforms	C. perfringens
E. coli	P. aeruginos
Klebisella	V. parahaemolyticus
Enterococci	Salmonella
Fecal Coliform	Staphylococcus
A. hydrophila	Enterobacter-Citrobacter

The study concluded that enterococci and, to a lesser degree, E. coli correlated to gastrointestinal illness in swimmers better than the other indicators. Even at very low densities of the indicator organisms, as low as 10 per 100 ml, there were measurable health effects via a route in which only 10 to 50 ml of water is ingested. At densities of 70 and 10/100 ml, respectively, the rates for total and "highly credible" gastrointestinal symptoms among swimmers were twice those for nonswimmers and the effect level was projected to be about 1/100 ml. It was suggested that the etiologic agent(s) is present in sewage in large numbers, it is highly infective, and/or it survives sewage treatment, disinfection, and/or transport better than the indicator.

The findings indicated that an enterococcus standard of 3/100 ml would result in a gastrointestinal illness of 6 per 1,000 swimmers. This was initially proposed as the recommended criteria and was later changed to 35/100 ml which was related to 19 illnesses per 1,000 swimmers. This later figure was said to approximate the degree of protection now accepted with the currently used fecal coliform criteria. It is not clear that there has been discussion or agreement that this degree of protection is acceptable, rather, it would appear that it was unknown and unexpected.

(6) EPA Fresh Water Standard

EPA reported the findings of a study of health effects and fresh recreational water quality in 1984 (Dufour 1984). The study was similar to the marine recreational water study except that only three bacterial indicators were used to measure water quality -- E. coli, enterococci, and fecal coliform. A good correlation was observed between swimming-associated gastrointestinal symptoms and either E. coli or enterococci densities.

It was found that the illness rate for swimming in marine water was much higher than the illness rate for fresh water at comparable levels of indicator organisms. It was postulated that the etiologic agent might be a virus which survives over the same period whether in seawater or fresh water, whereas the survival time of the indicator organism, enterococci, is much shorter in a marine water environment. In order to provide similar health protection, separate criterion were, therefore, needed for the two types of water.

Initially, a recommended limit of 20 enterococci per 100 ml or 77 E. coli per 100 ml was proposed. This provided the same risk level of 6 illnesses per 1,000 swimmers that was initially provided in the marine standard of 3 enterococci per 100 ml. Later the recommended criteria was changed to 126/100 ml E. coli or 33/100 ml enterococci. Again, the reason for the change was to approximate the degree of protection currently accepted with the fecal coliform standard. The modified freshwater criteria relates to an illness rate of 8 per 1,000 swimmers; consequently, there is a different risk of illness associated with the two recommended criteria.

In a 5-year study of 10 beaches (2 seawater, 8 fresh water) in Washington State (Vasconcelos 1985), it was found that the enterococcus standards were much more difficult ones to meet than either a 1,000/100 ml total coliform or a 200/100 ml fecal coliform level.

5. Shellfish Growing Waters

One of the most sensitive water uses from the viewpoint of potential waterborne disease transmission is the use of waters for shellfish cultivation. This is because of the ability of shellfish (oysters, clams, and mussels) to concentrate contaminants, including pathogenic agents, as a result of their filter feeding process. An oyster may filter 1,500 liters of water each day in search of nutrients and may retain the contaminants from the filtered water.

Sewage pollution and the associated potential for disease has resulted in the closure of many major growing areas including: Boston Harbor (1907), San Francisco Bay (1930), Raritan Bay (1961), Narragansett Bay, and portions of Chesapeake Bay (Larkin 1982).

a. Disease Agents

Bacterial pathogens potentially hazardous to the shellfish consumer are:

E. coli	Pseudomonas sp.
Proteus sp.	Salmonella sp.
Shigella sp.	Yersinia sp.
Vibrio sp.	Campylobacter sp.

Shellfish-associated hepatitis outbreaks have demonstrated the potential for viral disease transmission. The most commonly occurring diseases associated with the consumption of contaminated shellfish in the United States have been typhoid fever from 1900 to 1950, gastroenteritis from 1940 to 1980, and infectious hepatitis from 1960 to 1980 (Larkin 1982). As many as 8.6 percent of reported hepatitis A cases in the United States are associated with shellfish contamination (Levin 1978). In a survey of hepatitis patients in a Boston hospital, it was concluded that consumption of shellfish was as important as person-to-person contact in disease transmission during a nonepidemic period (Koff 1967).

Elsewhere, in the Philippine Islands, India, and Thailand, cholera outbreaks have occurred through shellfish consumption.

b. Disease Outbreaks

The details of disease outbreaks provide insight into some of the complicating facets involved in assessing shellfish safety. Over 2,000 cases of viral gastroenteritis occurred in Australia attributed to the consumption of rock oysters harvested from polluted estuaries near Sydney. Norwalk Agent was identified as the causative organism. A two-day depuration period is now required and a panel of volunteers test-consume oyster samples before marketing (Murphy 1979). Findings of other studies suggest that the short depuration period may be inadequate.

An outbreak of hepatitis A (278 cases) in Louisiana in 1973 was related to similar outbreaks in Georgia and Texas. The outbreaks were attributed to contaminated oysters from an east Louisiana approved growing area. Contaminated river water had reached the area during a flood period. Harvesting of the oysters was prohibited until the overlying waters met the standard of 70/100 ml coliform bacteria. Oysters harvested four weeks <u>after</u> the standard was met were implicated in the outbreaks. It appears that the oysters had concentrated the viruses and had retained them in sufficient amounts for four or more weeks to cause the outbreaks (Goyal 1979). For this and other disease outbreaks associated with shellfish there is some concern that shellfish from a prohibited area may have been involved.

During 1982, outbreaks of gastroenteritis, associated with eating raw or steamed clams and oysters, reached epidemic proportions in New York State. Between May 1 and December 31, there were 103 well-documented outbreaks in which 1,017 persons became ill. Norwalk agent virus was implicated as a predominant etiologic agent (identified in five of seven outbreaks in which testing was done) (Morse 1986). Although spring storms and runoff may have contributed to some outbreaks, it is believed that others occurred during nonstorm periods and involved shellfish from approved areas. In 1985, the high shellfish-associated illness rate continued with 59 outbreaks resulting in 888 documented cases in New York State.

As a result of the New York findings, the New England Journal of Medicine cautioned consumers that "eating poorly-cooked shellfish is currently a high risk venture at best" and called for an investigation of standards for the depuration of enteric viruses (DuPont 1986).

c. Concentration and Elimination of Organisms

Laboratory and field studies have documented the ability of shellfish to concentrate organisms from lightly polluted water and cleanse themselves in clean water. There are several factors associated with this ability. The ability to concentrate and eliminate organisms declines with temperature because the rate of filter feeding declines. At a temperature of 2 to 12°C, depending on the type of shellfish, filtration shuts down (Metcalf 1968). Elimination of organisms occurs more rapidly in moving water. The survival of viruses in oysters is extended significantly (up to 42 days at 1 to 11°C) at low temperatures and longer if frozen. From several studies involving bacteria and viruses it was found that the rate of depuration is slower than the rate of accumulation.

The uptake of indicator organisms is rapid. Oysters placed in seawater with 10 E. coli per milliliter accumulated 100 per gm. in four hours. Studies of oysters in water containing polio virus showed at least a 10-fold concentration in the oysters within 3 hours. Elimination of the virus was a slower process (up to 48 hours, but >95 percent reduction took place within 8 hours) (Mitchell 1966).

In field studies shellfish have been found with 30 virus/100 gm in waters containing only 2 virus/ liter (Vaughn 1975). In a laboratory study using seawater seeded with S. typhimurium, ~~oysters were allowed to concentrate the salmonella for 48 hours.~~ After transfer to sterile water, the oysters contained low levels of salmonella after 42 days. The rate of salmonella elimination was much slower than rates reported for E. coli and enteroviruses (Janssen 1974).

d. Water Quality

There is little information which links illness rate, pathogen content in the shellfish, or water, and indicator organisms;

however, there is some limited information which suggests that current standards need careful review, and studies similar to those carried out for recreational waters are needed.

In a New Hampshire study of an estuary shellfish area, salmonella were isolated from water which met the recommended limit for approved growing waters of 70/100 ml total coliform and on two occasions salmonella were isolated from shellfish which met the standard of 230/100 gm. On occasion, salmonella were isolated from water and shellfish that contained no fecal coliforms (Slanetz 1968).

Human feeding experiments were carried out in Australia in 1978-79. Volunteers (1,390) consumed at least five oysters harvested from a sewage contaminated river and 52 became ill (nausea, vomiting, and diarrhea). Norwalk virus was the agent. All oysters had been depurated and met the 230/100 gm total coliform standard (Groham 1981). In studies involving depuration, there was a five-log decrease in E. coli while coliphage S13 (a small E. coli virus) decreased less than one-half log (Canzonier 1971).

The EPA has cited two reports which suggest the need for studies which define the relationship between water quality indicators and health effects related to eating shellfish. One was an investigation of two hepatitis A outbreaks associated with oysters taken from an approved shellfish growing area (Portnoy 1975). The other report described cases of gastroenteritis caused by Vibrio cholera in Florida attributed to oysters harvested from approved areas (Wilson 1981). The EPA has prepared a 3-year study proposal involving multiple water quality indicator organisms and feeding experiments to determine the water quality and health relationship.

e. Shellfish Standards

The following is the development of standards for shellfish growing waters:

1924 to 1925	Typhoid fever outbreak due to shellfish led to development of the National Shellfish Sanitation Program (NSSP) by the U. S. Public Health Service and coastal states.
1925	Growing water criteria of no "E. coli" (coliform bacteria) in 1 cc amounts of water was suggested by NSSP.
1936	Testing of the whole oyster was recommended.

1946 A coliform bacteria standard for approved shellfish growing waters of a median not to exceed 70/100 ml was published by NSSP.

1965 Standard was expanded by NSSP to include that not more than 10 percent exceed 230/100 ml for 5-tube dilutions or 330 ml for 3-tube dilutions.

1974 Based on numerous comparative studies of total and fecal coliform, a fecal coliform standard was set: median 14/100 ml and not more than 10 percent to exceed 43/100 ml for 5-tube dilutions or 49 for 3-tube dilutions.

Shellfish meat is approved as a product in commerce if the fecal coliform count does not exceed 230 MPN/100 grams of meat and a 35°C plate count does not exceed 500,000 per gram.

For "restricted waters" where shellfish must be relayed or depurated, the total coliform count will range between 70/100 ml and 700/100 ml with not more than 10 percent of samples exceeding 2,300/100 ml.

6. Agricultural Water

Most of the available information on health effects associated with the use of water for agricultural irrigation involves the direct application of sewage effluents on crops. In certain portions of southwest United States, where river and stream flows during the dry weather growing season mainly constitute effluent discharges, similar health concerns would apply.

With regard to agriculture, even greater concern from a worldwide health perspective has been the use of night soil and sewage sludge on crops which has received much greater coverage in the literature, but is not included in this review. From a standpoint of disease transmission under present-day conditions, which generally involve secondary treatment and disinfection of wastewater, it would appear that the potential for parasitic disease transmission is greater than bacterial disease transmission by the agricultural irrigation route. This is due in part to the lesser effectiveness of disinfection, and the longer survival times in soil of, progressively: bacteria, viruses, and helminth eggs.

The paucity of information in the literature on the transmission of viral disease may be due to several factors identified by Crook (1984):

- Limitation of detection methods;

- o Difficulty in disease studies:
- o Unreported illnesses; and
- o Illness delay and masking by person-to-person contact.

No structured studies of virus infection or illness by the agricultural irrigation route have been found.

a. Disease Outbreaks

Early disease outbreaks associated with effluent or polluted water irrigation involved typhoid fever and parasitic infections.

The following is a summary of typical reported disease outbreaks from Sepp (1971) where sewage or polluted water was involved.

Location	Disease	Cases	Cause	Date
Massachusetts	Typhoid Fever	63	Raw celery at hospital	1899
London	Typhoid Fever	110	Watercress	1903
California	Typhoid Fever	8	Vegetables	1919
Germany	Typhoid Fever	180	Vegetables	1948
Europe & South Africa	Beef Tapeworm	?	Beef	1958?

Human infection by the beef tapeworm from cattle grazing on "sewer farms" has occurred in Australia, France, and the United States (Sepp 1971). Other parasitic diseases have been transmitted by direct contact to field workers.

b. Survival of Organisms

A number of environmental factors can influence survival times for organisms on crops and soils. Generally, organisms will survive <u>longer</u>:

(1) In soils rather than on crops;

(2) In higher humidity;

(3) In soil with a high organic content;

(4) In cold temperatures; and

(5) Protected from sunlight.

From 65 studies reported in the literature, Sepp (1971) extracted the following summary of survival times for various organisms:

Table 29

Survival Times in Days

	Coliform	Streptococci	Salmonella	Amoeba Cysts	Ascaris Eggs
Grass and Clover	6-34	-	12-42	-	
Vegetables	35	-	5-40	3	27-35
Soil	38	38	15-46	-	2-3 yrs

Enteroviruses have been found to survive only four to six days on vegetables under dry conditions (Knonwalchuk 1975). Other studies have shown virus survival on vegetables to extend to three to five weeks (Larkin 1976, Tierney 1977) when the viruses were artificially inoculated into the applied wastewater, and up to six months in cold, saturated soil (Feachem 1981).

Other survival data showing the influence of crop type, temperature, and organism have been summarized from the data presented in Feachem 1983:

<u>Salmonella</u>

Root crops -- up to 53 days
Leafy vegetables -- up to 40 days
Berries -- up to 5 days
Orchard crops -- over 2 days

<u>Shigella</u>

Less than 7 days

<u>E. histolytica</u>

35^{o} C 1 day
25^{o} C 1 week
7^{o} C 1 month

c. Standards

A criteria for irrigation waters was recommended in 1974 (Water Quality Criteria) by advisory committees to EPA. It stated that irrigation waters below the fecal coliform density of 1,000/100 ml should contain sufficiently low concentrations of pathogenic microorganisms that no hazards to animals or man result from their use or from consumption of raw crops irrigated with such waters. For use of wastewater for irrigation, a fecal coliform standard for unrestricted irrigation water should be a maximum of 1,000/100 ml. For effluents used in agriculture, WHO recommend a fecal coliform count of less than 100/100 ml (WHO 1973).

A number of western states prohibit the use of sewage effluents on vegetables or crops that can be eaten raw. Others have set limits based on treatment and the types of crops. The European countries have a wide range of irrigation allowances and prohibitions and often specify a minimum time interval between cessation of irrigation and crop harvesting. This information is summarized by Sepp (1971).

The following examples of regulations or standards are directed at wastewater reuse and generally reflect a concern with water quality used in particular for food crop irrigation (Crook 1984).

(1) South Africa

(a) Chlorinated tertiary effluent for orchards, vineyards, and fodder crops.

(b) Disinfected wastewater with less than 1,000 coliforms per 100 ml in 80 percent of samples for processed food crops.

(c) Fruits requiring peeling are the only nonprocessed food irrigated with reclaimed water.

(2) Germany

(a) Secondary treatment and chlorination for irrigation of pasture.

(b) Cease irrigation four weeks before harvesting processed food crops.

(c) Early irrigation of potatoes and cereals are the only nonprocessed food crops irrigated with reclaimed water.

(3) Texas

(a) Undisinfected secondary effluent for pasture irrigation.

(b) Only processed food crops to be irrigated with reclaimed water.

(4) Florida

Disinfected secondary effluent for fodder crop irrigation.

(5) Arizona

(a) Secondary effluent for nonfood-crop irrigation.

(b) Unprocessed food crops require a geometric mean of 2.2/100 ml fecal coliform, a maximum of one turbidity unit, and not more than one virus PFU/40 liters.

(6) California

Wastewater treatment and quality criteria for crop irrigation in California.

Treatment Level	Limits	Types of Use
Primary		Surface irrigation of orchards and vineyards. Fodder, fiber, and seed crops.
Oxidation &	23/100 ml	Pasture for milking animals.
Disinfection	2.2/100 ml	Surface irrigation of food crops (no contact between water and edible portion of crop).
Oxidation, coagulation, clarification, filtration*, and disinfection	2.2/100 ml max. = 23/100 ml	

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* The turbidity of filtered effluent cannot exceed an average of 2 turbidity units during any 24-hour period.

CASE EXAMPLES

The following are examples of waste discharge situations in California. The intent here is to present the various issues that may arise in considering waste discharge requirements for different situations. The coverages here are very general and do not contain all the detailed information available regarding the discharge situation.

Santa Ana River

Table 30 presents the densities of several indicator organisms in the domestic wastewater effluents and in the Santa Ana River water at various points moving downstream. The water discharges currently receive either secondary treatment and disinfection or tertiary treatment and disinfection. All plants are under schedules imposed by the Regional Water Quality Control Board (RWQCB) to provide tertiary treatment (generally in-line coagulation, filtration, and disinfection) by July 1988. The Riverside plant is under order to upgrade its existing tertiary treatment system.

The samples were collected in the summer when the discharges make up a major portion of the river flow. The data suggest that:

1. There may be significant contributions to the bacteriological quality of the river other than the community waste discharges;

2. Regrowth or reactivation of indicator bacteria in the discharges may occur; or

3. A combination of factors may be taking place.

Water contact sports is a recognized beneficial use of the upper reaches of the Santa Ana River when there is uninterrupted flow. In many areas the river is only a few inches deep; however, it is attractive, accessible, and receives use.

Without the provision of tertiary treatment for the assured removal of viruses and other pathogenic agents, surface waters used for recreation and made up largely of effluent would be expected to pose a health risk. The risk which may be associated with the densities of indicator organisms is unclear. It may be very substantial if other sources of contamination are the principal cause, and something less if the densities are due, all or in part, to regrowth.

The proposed EPA freshwater recreational criteria were applied to the river water quality, and the results are shown in Table 31. Again, it is not known if the identified possible disease risk is accurate.

The provision of tertiary treatment in the form of coagulation-filtration and effective disinfection would meet the recommended health protection requirements for waste discharges in such a situation. If this is accomplished and any other potential sources of contamination are eliminated, it is not known what bacterial densities might then be found in the river and their significance.

Table 30

1985 (July Through September)
Santa Ana River

Station	Total Coliform	Fecal Coliform	Entero-cocci	E. Coli
	Log Mean MPN/100 ml			
San Bernardino STP	43	29	4	24
Colton STP	4	3	3	22
La Cadena Br.	274	186	6	384
Rialto STP No. 2	29	29	2	60
Rialto STP No. 3	23	14	5	43
Riverside Avenue Br.	14,477	6,076	71	601
Mission Avenue Br.	8,663	3,170	39	531
MWD Xing	9,821	803	77	722
Riverside STP	2	2	1	14
Hamner Avenue Br.	9,887	1,806	280	587
River Road Br.	16,891	1,365	156	629
Pardo Dam Gage	13,185	934	452	594
Chino STP No. 1	3	2	1	8
Chino STP No. 2	5	3	1	14
Green River Drive Br.	12,043	535	334	600
Highway 90 Br.	12,969	1,023	320	703

Table 31

Possible Disease Risk Associated With Recreation
Santa Ana River

	Entero-cocci Log Mean/ 100 ml	Disease/ 1,000 Swimmers	E. Coli Log Mean/ 100 ml	Disease/ 1,000 Swimmers.
La Codina Br.	6	13.6	384	36.0
Riverside Ave. Br.	71	23.7	601	37.8
Mission Ave. Br.	39	21.2	531	37.4
MWD Xing	77	24.0	722	38.6
Hamner Ave. Br.	280	29.3	587	37.8
Prado Dam	156	26.9	629	38.0
River Pond Br.	452	31.2	594	37.8
Green River Dr. Br.	334	30.0	600	37.8
Highway 90 Br.	320	29.8	703	38.5

Enterococci: $Y = 6.278 + 9.40 (\log x)$

E. coli: $Y = 11.74 + 9.397 (\log x)$

From Dufour 1984

Morro Bay

Morro Bay is a commercial shellfish growing and sports clamming area located on the coast midway between San Francisco and Los Angeles. There are several creeks which drain the watershed, some live-aboard boats, a large bird population, cattle operations, and two sewage treatment plants which may contribute pollution to the Bay. A new outfall was constructed in 1982 at the main sewage treatment plant which serves the City of Morro Bay and a sanitary district. It extends 4,400 feet offshore, whereas the effluent was previously discharged 700 feet from shore.

In early 1985, the EPA and the RWQCB granted a variance from secondary treatment requirements. Also, chlorination of the effluent was not practiced inasmuch as the extended outfall and resultant greater dilution might be expected to protect the growing waters. A bacteriological sampling program was carried out in September 1984 and January 1985. Water quality at a significant number of stations did not meet the standard of 70/100 ml (11 of 26 stations). Even fewer (4 of 26 stations) met the fecal coliform standard of 14/100 ml. Sixty percent of shellfish samples exceeded the NSSP fecal coliform standard of 230/100 grams. Results of shellfish samples through the years is shown in Table 32. The commercial shellfish operations were subsequently shutdown. The initial phase of the study did not separate out the relative effects of the possible pollution sources; however, in the second phase, the bacteriological results of ocean waters outside the bay and dye studies suggested the undisinfected discharge may affect bay water quality, and EPA has ordered the reinstitution of disinfection.

Table 32

Morro Bay Shellfish Meat Quality

	Median Bacteria MPN/100 Grams All Shellfish Samples Combined	
	Total Coliform	Fecal Coliform
1975 (packaged product)	33	14
1976 " "	330	45
1977	790	110
1978	1,100	330
1979 (Feb. study -- rain)	4,600	1,300
1979	2,300	93
1980	240	78
1981	360	20
1982	180	140
1984	800	280
1984 (rain)	3,900	1,300
1984 (Sept. & Jan. 1985)	490	410
1984 (rain)	3,900	1,300
1985	410	330
1985	230	20
1986 (May)	130	45
1986 (June)	1,300	490

Santa Rosa

During the summer, sewage effluent at Santa Rosa is confined to land (ponded and used for irrigation). From October 1 to May 15, an effluent discharge, not to exceed 1 percent of the Russian River flow, is allowed when the river flow exceeds 1,000 cfs under requirements imposed by the RWQCB. The discharge is to Laguna de Santa Rosa and then to the Russian River.

In February 1985 there was a discharge of 750 million gallons of secondary effluent and an accidental discharge of more than one million gallons of untreated wastewater when stream flows were low. Again, in January 1986, discharges occurred in excess of the one percent maximum allowed.

With the expected growth of Santa Rosa and the neighboring areas, concern has been expressed regarding possibly larger and more frequent discharges to the river from the existing facilities and the City has retained a consultant to develop options for the future. The legislature authorized special funds for conducting water quality monitoring of the river and this was carried out from September to November 1985, December 1985, and January 1986. In December and January there were discharges to the river in excess of the one percent limit.

There are numerous issues associated with the current situation at Santa Rosa involving health, environmental, esthetic, and economic concerns. The lower river is one of the highest used recreation areas in the State. Concern has been expressed with any possible residual effects (bottom deposits, growths, etc.) that the winter discharges may have on summer recreation. Concern has also been expressed regarding discharges in the future and their effects on recreation and the tourist business.

The water supply of many small communities along the lower portion of the river is derived from shallow wells which tap the river underflow. Turbidity in the well waters increase during storms when the river is turbid indicating a very porous river bed media. The water systems depend on simple chlorination for treatment.

Even in periods of no discharge, the total coliform bacteria content at stations along the lower river range from 100 to several thousand per 100 ml (see Table 33). The source water could be considered marginal for systems depending on simple chlorination.

Several export options are under consideration for a long-term solution as well as increased treatment with a discharge to the river. Under the river ~~discharge options, the discharge ratio is a critical factor. A discharge of up~~ to ten percent of the river flow could be necessary.

An "indirect discharge" is a concept proposed by the City's consultant to avoid discharge volume restrictions. It has been proposed that an indirect discharge (or a discharge equivalent to an indirect discharge) would exist if:

1. The wastewater was no longer distinguishable as a wastewater and was indistinguishable from the receiving water;

2. The point of compliance for discharge preceded the indirect discharge step;

3. The effluent passed through a polishing treatment step which altered its characteristics; and

4. The effluent was dispersed before entering the receiving waters.

Under the current disinfection guidelines, "no discharge" is recommended; however, a discharge of up to five percent of the stream flow may be considered if "no discharge" is not physically or economically practical.

Table 33

Russian River Water Quality
(MPN/100 ml)

No Discharge	Total Coliform	Fecal Coliform	Enterococci
Healdsburg (upstream)	21-540	4-23	2-21
Mark West Creek (near Russian River)	350->2,400	22-450	8-190
Cooks Beach (downstream)	17->2,400	<2-380	2-290
Johnson's Beach (downstream)	280->2,400	2-43	9-18
Vacation Beach (downstream)	180->2,400	2-49	9-16
Discharge 1-5%			
Healdsburg	430-290	22-95	7-100
Mark West Creek	1,600->2,400	920	70-1,500
Cooks Beach	>2,400	130-1,600	38-630
Johnson's Beach	>2,400	35->2,400	210-570
Vacation Beach	1,600	40-920	88-160

CALIFORNIA SURFACE WATER QUALITY

Several surface water quality monitoring programs were carried out in the 1960s and 1970s which included monitoring of bacteriological quality. The Department of Water Resources had a statewide monitoring network of surface waters and the results were reported annually. Monthly duplicate bacteriological results were presented. Bacteriological analyses were eliminated from the monitoring program several years ago. The State Water Resources Control Board monitors contaminants such as pesticides and PCBs in water, sediments, and aquatic life. The program does not include bacteriological analyses. The DHS had conducted river studies, usually under contract to other agencies, directed at the bacteriological quality of river waters. These studies were discontinued due to lack of funds approximately 15 years ago. Currently, there is no statewide plan or program for the collection and analysis of surface waters for bacteriological quality.

Individual waste dischargers may be required to conduct downstream monitoring of river water quality, but generally, this is directed at factors such as biochemical oxygen demand, temperature, dissolved oxygen, and analyses other than bacteriological analyses. Ocean dischargers generally monitor receiving waters for coliform bacteria in the area subject to influence by the discharge.

The larger water utilities may perform bacteriological analyses on their raw water supply. Usually, only coliform bacteria are analyzed on samples collected at the water intake.

Special studies may be carried out for various reasons along river reaches. Recently, bacteriological analyses have been conducted on samples from the Santa Ana River and the Russian River. Similarly, marine waters may be monitored by waste dischargers, local health departments, or others in shellfish growing areas and water contact sports areas. Studies have been carried out at most of the growing waters, in particular Morro Bay, and at the recreational waters of Newport Bay.

Because of the limited amount of information usually available and the limited types of analyses performed, the overall surface water data does not lend itself to significant interpretation. A sampling of the information and a general interpretation is presented here.

For the most part, the bacteriological monitoring of surface waters is limited to the analysis of one or two indicator organisms performed on samples collected on a rigid weekly or monthly schedule. There is usually no reference to environmental conditions which may influence the quality (waste discharge problems, storms, bypasses, etc.). The water supplier is the data gatherer; the environmental conditions may not be known.

American River

The bacteriological quality of the American River, as measured by the fecal coliform content at the City of Sacramento water intake, has generally averaged about 100 fecal coliform/100 ml during the summer months; however, the data are sparse for firm findings.

Sacramento River

The Sacramento River is subject to more influences on water quality than is the American River. These include domestic and agricultural waste discharges, and, below Sacramento, urban storm runoff. This is reflected in the higher bacteriological content than the American river and the wider seasonal changes in quality.

The available data include total and fecal coliform content at the City of Sacramento water intake (Sacramento plant), total coliform at Cache Slough, and total coliform at the South Bay Aqueduct which receives delta water by means of the California Aqueduct.

The Sacramento River plant water intake is at the confluence of the Sacramento River and the American River, thereby representing a mixture of the two. The fecal coliform content only occasionally exceeds 200/100 ml in dry weather periods. The total coliform content generally runs 10 times the fecal coliform content and normally ranges from 1,000 to 2,000/100 ml.

Cache Slough

At Cache Slough, a source of supply for Vallejo, the median total coliform bacteria content annually is roughly 1,200/100 ml.

Table 34

American River Plant (Raw Water)

	Fecal Coliform/100 ml					
	1980-81	1981-82	1982-83	1983-84	1984-85	Average
July	43	54	71	65	46	56
August	23	73	270	180	140	137
September	102	300	300	83	200	197
October	86	43	240	2,400	2,300	1,014
November	-	175	68	390	ND	211
December	-	58	470	72	360	240
January	69	280	630	59	60	220
February	76	100	160	50	120	101
March	1,040	160	210	20	85	303
April	71	270	75	25	38	96
May	300	18	2,200	20	65	521
June	88	29	93	53	85	70
Average	190	130	399	285	320	

Table 35

Sacramento River Plant (Raw Water)

The intake is near the confluence of the Sacramento River and American River thereby representing a mixture of the two.

	Fecal Coliform/100 ML				
	1980-81	1981-82	1982-83	1983-84	1984-85
July	20	68	44	56	68
August	35	90	300	310	70
September	42	33	140	600	130
October	34	41	280	390	130
November	42	75	330	-	210
December	100	-	-	-	-
January	16	-	-	110	87
February	-	-	1,900	160	260
March	-	-	500	48	1,500
April	240	-	150	31	56
May	49	12	340	46	26
June	28	52	29	47	250
Average	61	53	401	180	250

Table 36

Cache Slough at Vallejo Intake
1985

	Coliform Bacteria MPN/100 ml		
	Max.	Median	Min.
January	24,000	13,000	2,400
February	240	150	23
March	24,000	2,400	240
April	2,400	1,300	2,400
May	7,000	3,500	23
June	2,400	240	240
July	2,400	1,300	240
August	>2,400	700	62
September	2,400	240	240
October	7,000	2,400	700
November	2,400	1,500	23
December	240	62	

California Aqueduct

At the South Bay Aqueduct takeoff from the California Aqueduct, the bacteriological quality, as measured by total coliform, ranges approximately from 100 to 1,000/100 ml. At the southern end of the Aqueduct, untreated water samples ranged from >700 to <2.1 per 100 ml for total coliform in wet weather and dry weather, respectively, with fecal coliform concentrations often <2.1/100 ml.

Colorado River

Fecal coliform bacteria in samples collected at the Weymouth Treatment Plant influent (Metropolitan Water District) generally average less than 10/100 ml during the summer and fall and reach peaks often in excess of 100/100 ml during the winter and spring wet weather periods. (The sampling point is 200 miles by aqueduct from the river.)

According to the RWQCB (Region 7), the following is the fecal coliform density at stations along the river:

Fecal Coliform/100 ml

	Max	Med	Min
Nevada Border	20	17	5
Imperial Dam	13	8	8
Morales	200	205	17

Table 37

Santa Clara Valley Water District
South Bay Aqueduct Influent

Month	MPN Coliform/ 100 ml
May 1986	110
April 1986	185
March 1986	85
February 1986	700
January 1986	900
December 1985	150
November 1985	90
October 1985	250
September 1985	75
August 1985	85
July 1985	70
June 1985	175
May 1985	700
April 1985	90
March 1985	325
February 1985	75
January 1985	175

Table 38

BACTERIOLOGICAL EXAMINATIONS OF THE DISTRICT'S FILTRATION PLANT INFLUENTS
Multiple Fermentation Tube Method (MPN)*
Fiscal Year 1984-85

	WEYMOUTH PLANT INFLUENT			DIEMER PLANT INFLUENT			JENSEN PLANT INFLUENT			SKINNER PLANT INFLUENT			MILLS PLANT INFLUENT		
onth	Number of Samples	Percent of Samples Positive	Median MPN	Number of Samples	Percent of Samples Positive	Median MPN	Number of Samples	Percent of Samples Positive	Median MPN	Number of Samples	Percent of Samples Positive	Median MPN	Number of Samples	Percent of Samples Positive	Median MPN
y	15	100.0	460	17	100.0	43	22	45.5	<3	17	94.1	11	15	93.3	6
igust	18	94.4	1100	18	88.9	23	24	33.3	<3	18	100.0	11	18	88.9	8
ptember	15	100.0	1800	15	100.0	93	19	21.1	<3	15	100.0	20	12	91.7	6
:tober	19	94.7	39	18	88.9	23	23	26.1	<3	18	100.0	28	15	93.3	13
ovember	17	29.4	<3	17	29.4	<3	23	78.3	9	16	100.0	15	14	100.0	8
ecember	15	20.0	<2	16	81.2	<2.2	21	76.2	15	12	100.0	33	14	85.7	13
iuary	17	100.0	79	18	100.0	43	23	82.6	4	19	94.7	6	18	55.5	2
bruary	15	100.0	124	13	100.0	33	19	10.5	<3	16	81.2	6	13	0.0	<2.2
irch	15	100.0	110	16	100.0	43	21	47.6	<3	16	87.5	12	16	68.8	4
ril	17	88.2	33	18	77.8	12	21	42.9	<3	18	100.0	8	18	83.3	5
iy	17	100.0	17	17	94.1	6	22	36.4	<3	18	100.0	13	16	93.8	6
ne	15	26.7	<2	16	43.8	<3	19	36.8	<3	16	93.7	40	15	86.7	6
tal	195			199			257			199			184		
Average	16	79.5	59	17	83.4	23	21	45.5	<3	17	96.0	12	15	78.8	6

PN = Most Probable Number of coliform bacteria per 100 ml. sample.

Jensen Plant - State Project Water, west branch

Mills Plant - State Project Water, east branch

Weymouth and Diemer Plants, Colorado River Water

Table 39

Los Angeles Water and Power
Aqueduct Source Samples
1985

	Coliform MPN/100 ml	Fecal Coliform MPN/100 ml
January	240	<2.1
February	32	<2.1
March	4.8	<2.1
April	>700	>700
May	>700	<2.1
June	>700	4.8
July	240	32
August	<2.1	-
September	<2.1	<2.1
October	>700	4.8
November	>700	8.6

East Bay Municipal Utility District (EBMUD)

The EBMUD water is derived from carefully protected watersheds. In dry weather periods, the median total coliform bacteria content of raw water samples collected from various points in the raw water storage and transmission system is usually less than 2/100 ml. Rains may increase the content to 16/100 ml according to samples collected in 1984-85.

ESTIMATES OF PATHOGEN CONCENTRATIONS AND DISEASE RISKS

Several models have been reported in the literature for estimating the probability of an individual becoming infected or ill with a known dose of a pathogen where dose-response information is available. As described by Haas (1983) these include the log-normal model, the exponential model, and the beta-distributed infectivity probability model. In comparing the three models using available dose-response data, Haas found that the beta-distributed model was superior to the other two. In particular, it was preferred to the exponential model which tended to underestimate the disease probability at low-dose levels as compared to observed data.

The equation for the beta-distributed model is:

$$P = 1 - (1 + N/b)^{-a}$$

where P is the probability that a single individual exposed to a dose of N organisms will become infected, and a and b are computed parameters of distribution based on dose-response data.

In order to estimate disease probability for a pathogen, the number of organisms has to be known or estimated and dose-response information has to be available.

The U. S. EPA, through the results of field studies comparing indicator organisms to swimmer illness, has developed equations to relate the two. Under this approach, the type or density of the pathogenic agent(s) is not known. It does require information on the density of the indicator organisms -- enterococci and/or E. coli -- and this information is generally not available for California surface waters.

The equations for fresh waters are as follows:

$$Y = 6.278 + 9.40(\log X)$$

where X is the mean enterococci density per 100 ml and Y is the illness per 1,000 swimmers, and

$$Y = 11.74 + 9.397(\log X)$$

where X is the mean E. coli density per 100 ml.

The equation for marine water is:

$$Y = 0.20 + 12.17(\log X)$$

where X is the mean enterococci density per 100 ml.

Estimates of bacterial pathogens in receiving waters can be developed based on the following:

- Illness rate for possible waterborne disease.

- Factor to correct from reported illness to actual illness (estimated at 20 times reported illness).

- Kehr-Butterfield relationship for S. typhi holds for other pathogenic bacteria in both sewage and receiving waters.

Then:

$$Y = ar^n$$

where Y = pathogens per one million coliforms, r is the morbidity rate and a and n are constants of 3 and 0.46, respectively.

Estimation of Virus Disease Risk - Swimming

The following is an estimation of disease risk associated with swimming in receiving waters which receive a well-disinfected, secondary effluent (equivalent to 10^1 to 10^2/100 ml median total coliform density). It is based on an approach developed by Haas (1983).

Effect of Treatment and Disinfection

$$Nie = Nio(1 - fiz)(1 - fid)$$

where: Nie = effluent microorganism concentration

Nio = influent microorganism concentration

fiz = fraction of microorganisms removed by treatment (primary and secondary)

fid = fraction of microorganisms removed by disinfection

Effect of Dieoff

$$Nit = \frac{NieE(-kt)}{1 + d}$$

where: Nit is the downstream concentration of microorganisms at travel time t.

d = dilution ratio (receiving water to effluent)

Probability of Ingestion

$$Pit = (1 - \exp(-NitV))$$

where: Pit = probability that the ingestion of volume V results in the ingestion of an organism.

Probability of Disease or Infection

$$P = Pit \times Piz$$

where: Piz = the probability that once ingested, an organism will cause a disease or infection

Assumptions

Nio = concentration of viruses in wastewater = 5,000 per liter (any detectable virus can cause infection)

foi = 40 removal by primary treatment

fiz = 86% removal by activated sludge

fid = 99% removal by disinfection

v = 100 ml of water ingested while swimming

d = 0.95 - ratio of receiving water to effluent

t = 1 day

k = 0.5 based on coliform die-away ratio

Piz = 0.01 or one in a hundred probability

Then: Nie = 5,000 (0.6) (0.14) (0.99) = 4.2 virus/liter

$$Nit = 4.2\ e\ \frac{(-0.485)}{1.95} = 1.36,\ \text{day } 1.4 \text{ virus/liter}$$

$$Pit = 1 - \exp(1.4 \times 0.1) = 0.131$$

$P = 0.131 \times 0.01 = 1.3 \times 10^{-3}$ or 1 to 2 illnesses per 1,000 swimmers

The probability of infection can be estimated for specific virus at a concentration of 1.4/liter where:

	a	b
ECHO 12	1.3	75
Poliovirus I	0.119	200

$$P\ (\text{for ECHO}) = 1 - (1 + 0.14/75)^{-1.3} \quad 2.4 \times 10^{-3}$$

$$P\ (\text{for Polio I}) = 1 - (1 + 0.14/200)^{-0.119} \quad 8.3 \times 10^{-5}$$

Bacterial Disease Risk - Swimming

The estimate of possible waterborne bacterial disease is based on:

- o Illness rate for waterborne disease.
- o Factor to correct from reported illness to actual illness (reported illness x 20 = actual illness).
- o Kehr-Butterfield relationship for S. typhi holds for sewage and receiving waters and for other pathogenic bacteria.

Then: $Y = 3r^{0.46}$

where r is the morbidity rate and Y is the number of pathogens per one million coliforms.

Reported illness rate in California in 1985:

Salmonellosis = 16.5

Shigellosis = 15.8

Actual rate:

Salmonellosis = 330

Shigellosis = 316

Salmonella per 10^6 coliform = 3.330 = 43.2

Shigella per 10^6 coliform = 3.316 = 42.4

Assuming the receiving water contains 200 fecal coliform per 100 ml and fecal coliform is 10 percent of total coliform content. Total coliform is 2,000/100 ml and:

Salmonella/100 ml = 0.086

Shigella/100 ml = o.085

For salmonella dysenteria:

a = 0.5

b = 100

$P = 1 - (1 + 0.086/100)^{-0.5} = \underline{4.3 \times 10^{-4}}$

For shigella flexneri:

$a = 0.2$

$b = 2{,}000$

$P = 1 - (1 + 0.085/2{,}000)^{-0.2} = \underline{8.6 \times 10^{-6}}$

REFERENCES

Indicator Organisms

AWWA, APHA, WPCF (1985). Standard Methods for the Examination of Water and Wastewater, 16 Ed. APHA Printing Office, Washington, D.C.

Caplenas, N. R. et al. (1981). Source and Extent of Klebisella Pneumoniae in the Paper Industry; Applied and Environmental Microbiology, 42:779.

Collins, Harvey F., and Robert E. Selleck (1972). Process Kinetics of Wastewater Chlorination, SERL Report No. 72-5. Uiversity of California-Berkeley.

California State Department of Public Health (1968). Proceedings of the Symposium on Fecal Coliform Bacteria in Water and Wastewater.

Dufour, A. P. et al. (1976). Characteristics of Klebisella From Textile Finishing Plant Effluent; Journal Water Pollution Control Federation, 48:872.

EPA (1986). Design Manual; Municipal Wastewater Disinfection Water Engineering Research Laboratory, EPA/625/1-86/021.

Untreated Domestic Wastewater

American Society of Civil Engineers; Committee on Environmental Quality Management of the Sanitary Engineering Division (1970). Engineering Evaluation of Virus Hazard in Water, ASCE; Sanitation Engineering Division, 96(SA1) 111-161.

Bloom, H. H. et al. (1959). Identification of Enteroviruses in Sewage; Journal of Infectious Disease, 105.

Buras, N. (1976). Concentration of Enteric Viruses in Wastewater and Effluent: A Two Year Study. Water Research, 10, m 295-8.

Cabelli, V. J.; Dufour, A. P.; McCabe, L. J.; and Levin, M. A. (1983). A Marine Recreational Water Quality Criterion Consistent with Indicator Concepts and Risk Analysis; Journal Water Pollution Control Federation, Vol. 55, No. 10.

California State Department of Health Services (1968). Ibid.

Cooper, R. C. et al. (1984). Infectious Agent Risk Assessment Water Quality Project; EPA-600/1-84-004.

Engelbrecht, R. S. (1980). Microbiological Criteria and Standards for Reuse of Water; Draft Document.

Evans Jr., D. J.; and D. G. Evans (1983). Classification of pathogenic Escherichia coli according to serotype and the production of virulence factors with special reference to colonization factor antigens. Rev. Infectious Disease, 5:S692-701.

Feachem et al. (1983). Sanitation and Disease -- Health Aspects of Excreta and Wastewater Management; World Bank, John Wiley and Sons, New York.

Fujioka, R. S. and Loh, P. D. (1978). Recycling of Water for Irrigation: Persistence of Enteroviruses in Sewage Affluent and Natural Waters Receiving the Effluent. Water, Air, and Soil Pollution, 9.

Heyward, A. A. et al. (1979). Viruses Associated with Combined Sewer Overflows and Storm Water Overflows in the City of Seattle. Proc. of Annual Meeting; American Society of Microbiology, Washington.

Hickman-Brenner, F. W., D. J. Brenner, A. G. Stiegerwalt, M. Schreiber, S. D. Holmberg, L. M. Baldy, C. S. Lewis, N. M. Pickens, and J. J. Farmer III (1984). Vibrio fluvialis and Vibrio furnissii isolated from a stool sample of one patient. Journal Clinical Microbiology, 20125-127.

Holmberg, S. D., W. L. Schell, G. R. Fanning, I. K. Wachsmuth, F. W. Hickman-Brenner, P. A. Blake, D. J. Brenner, and J. J. Farmer III (1986). Aeromonas Intestinal Infections in the United States. Ann. Intern. Med. 105:683-689.

Holmberg, S. D., I. K. Wachsmuth, F. W. Hickman-Brenner, P. A. Blake, and J. J. Farmer III (1986a). Plesiomonas Enteric Infections in the United States. Ann. Intern. Med. 105:690-694.

Johnston, J. M., S. F. Becker, L. M. McFarland (1986). Gastroenteritis in Patients With Stool Isolates of Vibrio vulnificus. American Journal of Medicine 80:336-338.

Johnston, J. J., S. F. Becker, L. M. McFarland (1985). Vibrio vulnificus: Man and the Sea. Journal of American Medical Association, 253:25=8502853.

Karmali, M. A. (1985). Bacterial Diarrhea: An Update. Diagnostic Medicine. May 12-19.

Mechelas, J. M. et al. (1972). An Investigation Into Recreational Water Quality. Water Quality Data Book, Vol. 4, EPA 18040 DAZ 04/72.

McCoy, J. H. (1964). Salmonella in Crude Sewage Effluent and Sewage-Polluted Natural Waters. Advances in Water Pollution Research 1.

Morris Jr., J. G., R. Wilson, D. G. Hollis, R. E. Weaver, H. G. Miller, C. O. Tacket, F. W. Hickman, and P. A. Blake (1982). Illness Caused by Vibrio damsela and Vibrio hollisae.

Shadera, W. X., J. M. Johnston, B. R. Davis, and P. A. Blake (1983). Disease From Infection With Vibrio mimicus, A Newly Recognized Vibrio Species. Ann. Intern. Med. 99:169-171.

Strobel, G. A., (1968). Coliform-Fecal Coliform Bacteria in Tidal Waters: Journal Sanitary Engineering Division Proc., ASCE, 94 SA4.

Wang, W. L. L. and Dunlop, S. G. (1954). Animal Parasites in Sewage and Irrigation Water. Sewage and Industrial Wastes, 26.

Treated Wastewater

Atkin, E. W. et al. (1977). Health Hazards Associated with Wastewater Effluents and Sludge: Microbiological Considerations in Risk Assessment and Health Effects of Land Application of Municipal Wastewater and Sludges; University of Texas at San Antonio.

Buras, N. (1976). Ibid.

Cooper, R. C. et al. (1984). Infectious Agent Risk Assessment Water Quality Project. UCB/SEEHRL Report No. 84-4.

Department of Health Services (1968). Ibid.

Engelbrecht, R. S. and Lund, E. (1975). Biological Properties of Wastewater Sludge and Its Potential Health Risk. Proceedings of Conference, International Water Conservance Exhibition, Jonkoping, Sweden.

Engelbrecht, R. S. (1976). Removal and Inactivation of Enteric Viruses by Wastewater and Water Treatment Processes. Proceedings of Advanced Wastewater Treatment Seminar. Japan Research Group of Water Pollution, Tokyo, Japan.

Engelbrecht, R. S. (1980). Ibid.

Feachem, R. E. et al. (1980). Health Aspects of Excreta and Sullage Management -- A State-Of-The-Art Review. Appropriate Technology for Water Supply and Sanitation, 3; World Bank, Washington, D.C.

Feachem et al. (1983). Ibid.

Gerba, C. P. (1978). Characterization of Sewage Solid-Associated Viruses and Behavior in Natural Waters, Water Research, 12.

Glass, J. S. and O'Brien, R. T. (1980). Enterovirus and Coliphage Inactivation during Activated Sludge Treatment, Water Research, 14.

Malina, J. F. et al. (1974). Poliovirus Inactivation by Activated Sludge In: Virus Survival in Water and Wastewater Systems. University of Texas, Water Resources Research Center.

Moore, B. E. et al. (1974). Applications of Viral Concentration Techniques to Field Sampling, In: Virus Survival in Water and Wastewater Systems. University of Texas, Water Resources Research Center.

Napearstek, J. D. et al. (1976). Virus Removal in an Activated Sludge Plant, Water and Sewage Works, 123 R16 and R00.

Rao, V. C. et al. (1977). Virus Removal in Activated Sludge Sewage Treatment, Progress in Water Technology, 9, 113-127.

Sherman, V. R. et al. (1975). Virus Removals in Tricking Filter Plants, Water and Sewage Works, 122 R36-44.

U. S. Department of Interior (1979). Water Reuse and Recycling, Vol. 2. Evaluation of Treatment Technology, OWRT/RU-79-2.

Warner, H. P. et al. (1978). Wastewater Treatment for Reuse and Its Contributions to Water Supplies. EPA, Washington, D.C.

Regrowth

Department of Health Services (1968). Ibid.

Greenberg, A. E. (1971). Aftergrowth of Fecal Pollution Indicators. Unpublished Paper, Department of Health Services.

Shuval, H. I. et al. (1973). Regrowth of Coliforms and Fecal Coliforms in Chlorinated Wastewater Effluent. Water Research, 7.

Receiving Waters -- Fresh

Allen, M. J. (1978). Evaluating the Microbial Quality of Potable Waters. In: Evaluation of the Microbiology Standards for Drinking Waters. EPA-570/9-7-00C.

Borg, G. et al. (1978). Indicators of Virus in Water In: Indicators of Virus in Water and Food; Ed. Berg, G. Ann Arbor Science Publishers, Ann Arbor, Michigan.

Cooper, R. C. et al. (1984). Ibid.

Department of Health Services (1968). Ibid.

Feachem. R. G. et al. (1983). Ibid.

Geldreich, E. E. (1970). Applying Bacteriological Parameters to Recreational Water Quality Journal, AWWA, 61.

Receiving Waters -- Marine

Baross, J. A. et al. (1975). Survival of Human Enteric and Other Sewage Microorganisms Under Simulated Deep-Sea Conditions. Applied Microbiology, 30.

Cooper, R. C. et al. (1984). Ibid.

Edmond, T. D. et al. (1978). Detection of Enteroviruses Near Deep Marine Sewage Outfalls. Marine Pollution Bulletin, 9, 246-49.

Gerba, C. P. et al. (1979). Failure of Indicator Bacteria to Reflect the Occurrence of Enteroviruses in Marine Waters. American Journal of Public Health, 69.

Disease -- Various Routes

Cabelli, V. J. (1983). Public Health and Water Quality Significance of Viral Diseases Transmitted by Drinking Water and Recreational Water. Water Science Technology, 15.

Cooper, R. C. (1975). Public Health Aspects of San Francisco Waste Discharge; Report to San Francisco Board of Supervisors.

Infective Dose

Anon., (1977). Center for Disease Control. Drug-resistant *Salmonella agona* - New York. Morbidity Mortality Weekly Rep., 26:129-130, 135.

D. Aoust, J. Y.; D. J. Aris; P. Thisdele; A. Durante; N. Brisson; D. Dragon; G. Lachapelle; M. Johnston; and R. P. Laidley (1975). *Salmonella eastbourne* Outbreak Associated With Chocolate. Can Inst. Food Science Technology Journal, 8:181-184.

Dolin, R.; N. R. Blacklow; H. DuPont; S. Formal; R. F. Buscho; J. A. Kasel; R. P. Chames; R. Hornick; and R. M. Chanock (1971). Transmission of Acute Infectious Nonbacterial Gastroenteritis to Volunteers by Oral Administration of Stool Filtrates. Journal of Infectious Diseases, 123:307-312.

Dupont, H. L.; S. B. Formal; R. B. Hornick; M. J. Snyder; J. P. Libonati; D. G. Sheahan; E. H. LaBrec; and J. P. Kalas (1971). Pathogenesis of *Escherichia coli*, 285:1-9.

Dupont, H. L.; R. B. Hornick; M. J. Snyder; J. P. Libonati; S. B. Formal; and E. J. Gangarosa (1972). Immunity of Shigellosis II Protected Induced by Oral Live Vaccine or Primary Infection. Journal of Infectious Diseases, 125:12-16.

DuPont, H. L. (1973). Clinical Approach to Infectious Diarrhea, Medicine 52:265.

Fontaine, R. E.; S. Arnon; W. T. Martin; T. M. Vernon; E. J. Gangarosa; J. J. Farmer; A. B. Moran; J. H. Silliker; and D. L. Decker (1978). Raw Hamburger: An Interstate Common Source of Human Salmonellosis. American Journal of Epidemiology, 107:36-45.

George, R. H. (May 1976). Small Infective Doses of Salmonella. Lancet, vi:1130.

Hornick, R. B.; S. E. Greisman; T. E. Woodward; H. L. DuPont; A. T. Dawkins; and M. J. Snyder (1970). Typhoid Fever: Pathogenesis and Immunologic Control (1). New England Journal of Medicine, 283:686-691.

Hornick, R. B.; S. I. Music; R. Wenzel; R. Cash; J. P. Libonati; M. J. Snyder: and T. E. Woodward, (1971). The Broad Street Pump Revisited: Response of Volunteers to Ingested Cholera vibrios. Bull., New York Academy of Medicine, 47:1181-1191.

Lang, D. L.; L. J. Kunz; A. R. Martin; S. A. Schroeder; and L. A. Thomson (1967). Carmine as a Source of Nosocomial Salmonellosis. New England Journal of Medicine, 276, 829.

Levine, M. M.; H. L. Dupont; S. B. Formal; R. B. Hornick; A. Takevchi; E. J. Gangarosa; M. J. Snyder; and J. P. Libonati (1973). Pathogenesis of Shigella dysenteraise 1 (Shiga) Dysentery. Journal of Infectious Disease, 127:261-270.

Lippy, E. C. and Waltrip, S. C. (1984). Waterborne Disease outbreaks -- 1946 to 1980: A Thirty-Five-Year Perspective. Journal of American Water Works Association, February 1984.

McCullough, N. B. and Eisele, C. W. (1951). Experimental Human Salmonellosis I. Journal of Infectious Disease, 88, 278.

Rendtorff, R. C. (1954). The Experimental Transmission of Human Intestinal Protozoan Parasites I, Endamoeba coli Cysts Given in Capsules. American Journal of Epidemiology, 59:196-208.

Rendtorff, R. C. (1954a). The Experimental Transmission of Human Intestinal Protozoan Parasites II, Giardia lamblia Cysts Given in Capsules. American Journal.

Robinson, D. A. (1981). Infective Dose of Campylobacter jejuni in Milk. British Medical Journal, 282:1584.

Sanyal, S. C. and P. C. Sen (1974). Human Volunteer Study on the Pathogenicity of Vibrio Parahaemolyticus in International Symposium on Vibrio Parapaemolyticus. Saikon Publ. Co. Ltd., Tokyo, Japan.

Silliker, J. H. (1980). Status of Salmonella -- Ten Years Later. Journal Food Prot. 43:307-313.

Factors Influencing Infective Dose and Dose Response

Adams, H. G. and C. Jordan (1984). Infections In the Alcoholic. Medical Clinics North America, 68:179-200.

Bonner, J. R.; A. S. Coker; C. R. Berryman; and H. M. Pollock (1983). Spectrus of Virbio Infections In a Gulf Coast Community. Ann. Intern. Med. 99:464-469.

Cantey, J. R. (1985). Infectious Diarrhea: Pathogenesis and Risk Factors. American Journal of Medicine, 78:65-75.

Curtis, A. J.; J. H. Lin; and K. Szako (1985). Overwhelming Post-Splenectomy Infection With Plesiomonas shigelloides in a Patient Cured of Hodgkins Disease. American Journal of Clinical Pathology, 83:522-524.

Gardner, I. D. (1980). The Effect of Aging on Susceptibility to Infection. Rev. Infectious Disease, 2:801-810.

George, W. L.; M. M. Nakata; J. Thompson; and M. L. White. Aeromonas-Related Diarrhea in Adults. Archives of Internal Medicine, 145:2207-2211.

Giannella, R. A.; S. A. Broitman; and N. Zamcheck (1973). Influence of Gastric Acidity on Bacterial and Parasitic Infections. Ann. Internal Medicine, 78:271-276.

Harris, R. L.; V. Fainsten; L. Elting; R. L. Hopfer; and G. P. Bodey (1985). Bacteremia Caused by Zeromonas Species in Hospitalized Cancer Patients. Rev. Infectious Disease, 7:314-320.

Hentges, D. J. (1986). The Protective Function of the Indigenous Intestinal Flora. Ped. Infectious Disease, 5:S17-S20.

Hill, H. F. and F. A. Meier (1986). Host Defense Factors in the Gastro-intestinal Tract. Ped. Infectious Disease, 5:S144-S147.

Hughes, J. M.; D. G. Hollis; E. J. Gangarosa; and R. E. Weaver (1978). Non-cholera vibrio Infection in the United States: Clinical, Epidemiologic, and Laboratory Features. Ann. Internal Medicine, 88:602-606.

Janda, J. M.; R. Brenden; J. A. DeBenedetti: M. O. Constantino; and T. Robin (1986). Vibrio alginolyticus Bacteremia in an Immunocompromised Patient. Diagnostic Microbiol. Infectious Disease, 5:337-340.

Lowry, P. W.; L. M. McFarland; and H. K. Threefoot (1986). Vibrio hollisae Septicemmia After Consumption of Catfish. Journal Infectious Disease, 154:730-731.

Luna, M. A. (1985). Infection: The Threat Beyond Cancer. Diagnostic Medicine. June:26-35.

Martin, D. L.; and T. L. Gustafson (1985). Plesiomonas Gastroenteritis in Texas. Journal American Medical Association, 254:2063.

Nachamkin, I.; C. Stowell; D. Skalina; A. M. Jones; R. M. Roop; and R. M. Smibert (1984). Campylobacter laridis Causing Bacteremia in an Immuno-suppressed Patient. Ann. Internal Medicine, 101:55-57.

Penn, R. G.; D. K. Ciger; F. C. Knoopt; and L. C. Preheim (1982). Plesiomonas shigelloides Overgrowth in the Small Intestine. Journal Clinical Microbiology, 15:869-872.

Rajkovic, I. A.; and R. Williams (1985). Mechanisms of Abnormalities in Host Defenses Against Bacterial Infection in Liver Disease. Clinical Science 68:247-253.

Rolston, K. V. I.; and R. L. Hopfer (1984). Diarrhea Due to Plesiomonas shigelloids in Cancer Patients. Journal Clinical Microbiology, 20:597-598.

Shandera, W. X.; J. M. Johnston; B. R. Davis; and P. A. Blake. Disease From Infection With Vibrio mimicus, a Newly Recognized Vibrio Species. Ann. Internal Medicine, 99:169-171.

Siegel, M. I.; and A. I. Rodgers (1982). Fatal Non-01 Vibrio cholerae Septicemia in Chronic Lmphocytic Leukemia. Gastroenterology 83:1130-1131.

Smith, G. S.; and M. J. Blaser (1985). Fatalities Associated With Campylobacter jejuni Infections. Journal American Medical Association, 253:2873-2875.

Wilson, C. B. (1986). Immunologic Basis for Increased Susceptibility of the Neonate to Infection. Journal Pediatrics, 108:1-12.

Lippy, E. C. and Waltrip, S. C. (1984). Waterborne Disease Outbreaks -- 1946-1980: A Thirty-Five Year Perspective. Journal American Water Works Association, February, 1984.

McCullough, N. B. and Eisele, C. W. (1951). Experimental Human Salmonellosis I. Journal Infectious Disease, 88, 278.

Risk Assessment

Berg, G. (1967). Transmission of Viruses by the Water Route. Federation Water Pollution Control Admin., Interscience Publishers, New York.

Cabelli, V. J. (1981). Health Effects Criteria for Marine Recreational Waters. Health Effects Research Laboratory, U. S. Environmental Protection Agency. EPA-600/1-80-031.

Colorado University (1985). Risk Assessment of Wastewater Disinfection, EPA/600/2-85/037.

Cooper, R. C. et al. (1984). Ibid.

DuPont, H. L. (1973). Ibid.

Environmental Protection Agency (1979). An Approach for Comparing Health Risks of Wastewater Treatment Alternatives -- a Limited Comparison of Health Risks Between Slow Rate Land Treatment and Activated Sludge Treatment and Discharge, 430/9-79-009.

Environmental Protection Agency (1973). Microbial Health Considerations of Soil Disposal of Domestic Wastewater. EPA, 600/9-83-017.

Haas. C. N. (1983a). Estimation of Risk Due to Low Doses of Microorganisms: A Comparison of Alternative Methodologies. American Journal of Epidemiology, 118.

Hutzler. N. J. and Boyle. W. C. (1980). Wastewater Risk Assessment Journal of the Environmental Engineering Division. ASCE. October.

Michales, J. M. et al. (1972). Ibid.

Pipes. W. O. (1978). Water Quality and Health Significance of Bacterial Indicators of Pollution: Drexel University.

Drinking Water

Cooper, R. C. et al. (1984). Ibid.

Craun. G. F. (1981). Outbreaks of Waterborne Disease in the United States: 1971-1978. Journal American Water Works Association, July.

Department of Health Services (1973). Guide for Treatment of Surface Waters of Various Water Qualities for Domestic Use, Water Sanitation Section.

Environmental Protection Agency (1972). Committee on Water Criteria. National Academy of Sciences; National Academy of Engineering, EPA-R3-73-033.

Environmental Protection Agency (1978). Evaluation of the Microbiology Standards for Drinking Water, EPA-570/9-78-00C.

Environmental Protection Agency (1985). National Primary Drinking Water Regulations . . . Proposed Rule, 40 CFR Part 141; November 13, 1985.

Environmental Protection Agency (1985). Workshop on Filtration; Disinfection and Microbial Monitoring Workshop Summary.

Environmental Protection Agency (1986). Bacteriological Ambient Water Quality Criteria: Availability; Federal Register, March 7, 1986.

Lippy, E. C. and Waltrip, S. C. (1984). Waterborne Disease Outbreaks -- 1946-1980: A Thirty-Five Year Perspective. Journal American Water Works Association, February 1984.

World Health Organization (1979). Human Viruses in Water, Wastewater, and Soil, Tech. Report Ser. No. 639.

Recreational Water

Assoc. of Metro. Sewerage Agencies (1984). Comments on Water Quality Criteria. Bulletin No. TB 84-22.

Cabelli, V. J. et al. (1983). Health Effects Criteria for Marine Recreational Waters Health Effects Research Laboratory, EPA-600/1-80-031.

Committee on Bathing Beach Contamination of the Public Health Laboratory Service (1959). Sewage Contamination of Coastal Bathing Waters in England and Wales; Journal of Hygiene, 57.

Department of Health Services (1943). Report on a Pollution Survey of Santa Monica Bay Beaches in 1942; Bureau of Sanitary Engineering.

Department of Health Services (1949). Bacteriological Standard for Surf Bathing Water; Bureau of Sanitary Engineering.

D'Alessio, D. J. et al. (1981). A Study of the Proportions of Swimmers Among Well Controls and Children with Enterovirus-Like Illness Shedding or Not Shedding an Enterovirus. American Journal Epidemiology, 113.

Dufour, A. P. (1984). Ibid.

Environmental Protection Agency (1976). Quality Criteria for Water; Washington, D.C.

Geldreich, E. E. (1970). Ibid.

Melnick, J. L. (1976). "Viruses in Water." In: Viruses in Water American Public Health Association, Washington, D.C.

Moore, B. (1959). Sewage Contamination of Coastal Bathing Waters in England and Wales: A Bacteriological and Epidemiological Study; Journal of Hygiene, 54.

Stevenson, A. H. (1953). Studies of Bathing Water Quality and Health; American Journal Public Health Association, 43.

Vanconcelos, G. J. (1985). Microbiological Quality of Recreational Waters in the Northwest, Journal, WPCF, 57.

Shellfish Growing Waters

Canzonier, W. J. (1971). Accumulation and Elimination of Coliphage S-13 by the Hard Clam Mercenaria; Applied Microbiology, 21.

DuPont, H. L. (1986). Consumption of Raw Shellfish -- Is the Risk Now Unacceptable? New England Journal of Medicine, 314.

Goyal, S. M. (1979). Human Enteroviruses in Oysters and Their Overlying Waters; Applied and Environmental Microbiology, 37.

Grohman, G. S. et al. (1981). Norwalk Virus Gastroenteritis in Volunteers Consuming Depurated Oysters; Austr. Journal Exptl. Biol. and Medical Science, 59.

Janssen, W. A. (1974). Oysters: Retention and Excretion of Three Types of Human Waterborne Disease Bacteria. Health Laboratory Science, 11.

Koff, R. S. et al. (1967). Viral Hepatitis in a Group of Boston Hospitals: Importance of Exposure to Shellfish in a Nonepidemic Period. New England Journal of Medicine, 276.

Larkin E. P. and Hunt, D. A. (1982). Bivalve Mollusks: Control of Microbiological Contaminants. Bioscience, 32.

Levin, M. (1978). Fish and Shellfish Associated with Disease Outbreaks. Journal WPCF, 50.

Metcalf, T. G. (1978). Indicators of Viruses in Shellfish. In: Indicators of Viruses in Water and Food; Ed. G. Berg. Ann Arbor Science Publishers.

Mitchell, J. R. et al. (1966). Accumulation and Elimination of Poliovirus by the Eastern Oyster; American Journal of Epidemiology, 84.

Morse, D. L. et al. (1986). Widespread Outbreaks of Clam and Oyster-Associated gastroenteritis. New England Journal of Medicine, 314.

Murphy, A. M. et al. (1979). An Australian-Wide Waterborne Outbreak of Gastroenteritis from Oysters Caused by Norwalk Virus. Medical Journal of Australia, 2.

Portnoy, B. L. et al. (1975). Oyster Associated Hepatitis -- Failure of Shellfish Certification Programs to Prevent Outbreaks. JAMA, 233.

Slanetz, L. W. et al. (1968). Coliforms, Fecal Streptococci and Salmonella in Seawater and Shellfish; Health Laboratory Science, 5.

Vaughn, J. M. and Metcalf, T. G. (1975). Coliphages as Indicators of Enteric Viruses in Shellfish and Shellfish Raising Estuarine Waters; Water Research, 9.

Wilson, R. (1981). Non-O Group 1 Vibrio Cholera Gastroenteritis Associated with Eating Raw Oysters. American Journal Epidemiology, <114.

Agricultural Water

Crook, J. (1984). Ibid.

Environmental Protection Agency (1973). Water Quality Criteria EPA-R3-73-033.

Feachem, R.; Garelich H.; and Slade, J. (1981). Enteroviruses in the Environment. Tropical Diseases Bull., 78.

Feachem R. et al. (1983). Ibid.

Konowalchuk, J. and Speirs, J. I. (1975). Survival of Enteric Viruses on Fresh Vegetables; Journal of Milk and Food Technology, 38.

Larkin, E. P. et al. (1976). Persistence of Virus on Sewage-Irrigated Vegetables; Journal Eng. Engr. Division, ASCE, 102.

Rowan, W. B. (1964). Sewage Treatment and Shistosome Eggs: American Journal of Tropical Medicine and Hygiene, 13.

Sepp, E. (1971). The Use of Sewage for Irrigation, A Literature Review: California Department of Public Health.

Tierney, J. T. et al. (1977). Persistence of Poliovirus in Soil and on Vegetables Grown in Soil Previously Flooded with Inoculated Sewage Sludge or Effluent; Applied and Environmental Microbiology, 33.

Wang, W. L. L. and Dunlop, S. G. (1954). Ibid.

World Health Organization (1973). Reuse of Effluents Methods of Wastewater Treatment and Health Safeguards; Tech. Report Series No. 517, Geneva.

PART III

HEALTH EFFECTS

CHEMICAL CONTAMINANTS

TABLE OF CONTENTS

CHEMICAL CONSTITUENTS

The concentration of organic and inorganic constituents in wastewater is a concern where effluents are discharged into receiving waters that are subsequently utilized as potable water supplies. The episodes with mercury, selenium, cadmium, pesticides, and polychlorinated biphenyls, while not all involving the wastewater route, have demonstrated the serious and complex nature of the problem posed by chemical agents. Organic constituents are of particular concern, due to the potential for long-term health effects, e.g., cancer and birth defects, resulting from ingestion of water containing extremely low concentrations of certain constituents. Although the composition of trace organic chemical residues in wastewater is largely unknown and highly variable, a great deal of information is available on specific organic (and inorganic) constituents in different types of wastewater effluent, river water, and finished drinking water.

INORGANIC CONSTITUENTS

Unlike the somewhat ill-defined organic constituents present in wastewater which pose health concerns, the inorganic constituents of health significance are, for the most part, well-known. The quantities of inorganic constituents present in sewage effluents depend on several factors, but mainly on the type of waste, i.e., domestic or industrial, and the degree of treatment that the wastewater receives. Generally, the heavy metals and many of the other inorganic substances are readily removed by conventional sewage treatment processes. For example, an evaluation of ten full-scale wastewater treatment plants by Zemansky (1973) indicated that primary clarification removed 22-57% of the heavy metals, depending on the specific heavy metal, while the trickling filter and activated sludge processes (including clarification), removed 5-56% and 22-71%, respectively. Heavy metals have also been largely controlled by industrial source control via local and federal pretreatment requirements. Therefore, domestic effluents usually contain relatively low levels of heavy metals and other inorganic substances, as indicated in Tables 1, 2, and 3.

The U.S. Environmental Protection Agency has set effluent limitations on metals and a few other inorganics. As a result of lawsuits brought by public interest groups in 1976, EPA developed effluent guidelines for 65 classes of pollutants, including 126 "priority pollutants". Fifteen metals and other inorganics currently are classified as priority pollutants. The settlement of the lawsuit also required EPA to set pretreatment standards for 21 categories of industries to assure the protection of publicly owned sewage treatment works and the protection of the environment from contaminated sludge. The complete list of priority pollutants is shown in Table 4.

Table 1

Averaged Water Quality Data for Reclaimed Water Used for Groundwater Recharge in the Montebello Forebay
[Nellor et al (1984)]

	Reclaimed Water[1]		
	3^{o}-Cl_2/$DeCl_2$	2^{o}-Cl_2	2^{o}-$UnCl_2$
Turbidity, NTU	0.8	2.3	1.9
Color, CU	22.5	28.3	36.7
pH	7.23	7.73	7.52
Dissolved solids, mg/l	568	588	560
Total alkalinity, mg/l as $CaCO_3$	220	276	276
Chloride, mg/l	133	129	111
Sulfate, mg/l	113	102	102
Nitrate, mg/l as N	0.34	0.13	0.19
Nitrite, mg/l as N	0.40	0.52	0.50
Ammonia, mg/l as N	13.7	20.7	20.5
Phosphate, mg/l as PO_4	10.6	6.9	7.2
Fluoride, mg/l	0.65	0.66	0.60
Cyanide, mg/l	<0.01	<0.01	0.02
Total hardness, mg/l as $CaCO_3$	194	212	211
Arsenic, mg/l	0.003	0.004	0.003
Barium, mg/l	0.22	0.20	0.23
Cadmium, mg/l	<0.002	0.004	0.004
Chromium, mg/l	<0.01	<0.01	<0.01
Copper, mg/l	0.008	0.016	0.015
Iron, mg/l	<0.06	0.07	0.09
Lead, mg/l	<0.02	0.02	0.03
Manganese, mg/l	<0.01	0.02	0.02
Mercury, mg/l	<0.0001	<0.0001	<0.0001
Nickel, mg/l	0.05	0.07	0.07
Selenium, mg/l	<0.003	0.003	0.003
Silver, mg/l	0.001	<0.001	<0.001
Sodium, mg/l	105	126	116
Zinc, mg/l	0.03	0.05	0.05
TOC, mg/l as C	11.2	14.1	12.6
Bromide, mg/l	<0.30	0.32	0.31
Total coliform, MPN/100 ml	8	<8	$>5.0 \times 10^5$
Fecal coliform, MPN/100 ml	<2	<2	$>2.8 \times 10^4$
Fecal strep, MPN/100 ml	<2	<2	$>8.8 \times 10^2$
Total plate count/ml	194	83	$>2.9 \times 10^4$

[1] 3^{o}-Cl_2/$DeCl_2$ = chlorinated/dechlorinated tertiary effluent
2^{o}-Cl_2 = batch chlorinated secondary effluent
2^{o}-$UnCl_2$ = unchlorinated secondary effluent

Table 2

Trace Element Concentrations in Municipal Sewage Effluent From Selected Cities in California
[Westcot and Ayers (1984)]

Trace Element	City of Santa Rosa (mg/l)	Water Factory 21 (mg/l)	City of Fresno (mg/l)	Sacramento Regional Plant (mg/l)	Chino Basin MWD (mg/l)
Silver (Ag)	-	0.004	<0.001	0.004	<0.005
Arsenic (As)	0.003	0.002	0.002	0.0026	<0.001
Boron (B)	0.53	0.62	-	-	-
Barium (Ba)	-	0.082	0.005	-	<0.001
Beryllium (Be)	-	-	<0.001	<0.01	-
Cadmium (Cd)	0.006	0.009	<0.001	<0.01	0.006
Cobalt (Co)	<0.001[1]	-	-	-	<0.001
Chromium (Cr)	0.003	<u>0.204</u>	<0.001	0.015	0.01
Copper (Cu)	0.004	<u>0.291</u>	0.013	0.026	0.015
Iron (Fe)	0.21	0.19	-	-	<0.05
Mercury (Hg)	-	<0.001	0.0003	0.0006	<0.001
Manganese (Mn)	<u>0.068</u>[2]	0.038	-	-	0.01
Nickel (Ni)	0.04	-	0.030	0.08	0.032
Lead (Pb)	0.017	0.035	0.050	0.04	0.02
Selenium (Se)	0.001	0.007	0.003	0.0002	<0.001
Zinc (Zn)	0.06	0.308	0.041	0.06	0.022

[1]Values presented with a < sign signify that the element was present but at a concentraton below the level of detection, which is the value following the < sign.

[2]Those values underlined exceed the California Drinking Water Standards.

Table 3

Inorganic Contaminants in Raw and Treated Sewage
[California Department of Health and Cooper (1976)]

Constituent	Raw Sewage (mg/l)	Secondary Effluent[1] (mg/l)	Tertiary Effluent[2] (mg/l)	Tertiary Effluent[3] (mg/l)
Antimony		-	-	0.00044
Arsenic		0.00-0.01	0.03-0.3	0.005
Barium		0.0-<0.02	0.0-<1	-
Bicarbonate		200-450	250	-
Bromine		-	-	0.065
Cadmium	0.02-0.06	0.011-0.130	0.000-0.005	-
Calcium		70-110	80	-
Cesium		-	-	0.000006
Chloride		300-350	300-350	-
Chromium (+6)	0.05-3.80	0.09-0.19	0.00-0.04	0.0005
Cobalt	0.05	-	-	0.00033
Copper	0.1-1.30	0.09-0.39	0.02-0.30	0.0116
Iron	0.8-3.90	-	-	0.0003
Lead	0.2-0.60	0.00-0.05	0.00-0.04	-
Manganese	0.1-0.66	-	-	0.02
Magnesium		20-45	2	-
Mercury	0.0013-0.0680	<0.001-0.003	0.000-0.006	0.0005
Nitrate (as N)	0-2.10	<1	<1	-
Nitrite (as N)	0.02-0.230	<1	<1	-
Phosphate	3.6-20.4	20-45	<1	-
Potassium		20-35	20-35	-
Rubidium		-	-	0.010
Scandium		-	-	0.00044
Selenium		0.00-0.009	0.00-0.003	0.0005
Silver	0.05-0.60	0.00-0.01	0.00-0.01	0.0004
Sodium		240-260	240-260	-
Sulfate		270-350	270-350	-
Zinc	0.16-0.84	0.09-2.08	0.02-0.07	0.005

[1]Conventional Secondary Treatment - Trickling Filters
[2]Pilot Plant - Chemical Treatment, Sedimentation, Ammonia Stripping, Recarbonation, Mixed-Media Filtration, Activated Carbon Adsorption, and Chlorination
[3]South Lake Tahoe Advanced Wastewater Treatment Plant

Table 4

Priority Pollutants

Metals and Inorganics

Antimony
Arsenic
Asbestos
Beryllium
Cadmium
Chromium
Copper
Cyanides
Lead
Mercury
Nickel
Selenium
Silver
Thallium
Zinc

Pesticides

Acrolein
Aldrin
Chlordane
DDD
DDE
DDT
Dieldrin
Endosulfan & endosulfan sulfate
Endrin and endrin aldehyde
Heptachlor
Heptachlor epoxide
Hexachlorocyclohexane (α,β,δ isomers)
γ-Hexachlorocyclohexane (lindane)
Isophorone
TCDD
Toxaphene

PCBs and Related Compounds

Polychlorinated biphenyls (6 PCB arochlors)
2-Chloronaphthalene

Halogenated Aliphatics

Chloromethane (methyl chloride)
Dichloromethane (methylene chloride)
Trichloromethane (chloroform)
Tetrachloromethane (carbon tetrachloride)
Chloroethane (ethyl chloride)
1,1-Dichloroethane (ethylidine chloride)
1,2-Dichloroethane (ethylene dichloride)
1,1,1-Trichloroethane (methyl chloroform)
1,1,2-Trichloroethane
1,1,2,2-Tetrachloroethane
Hexachloroethane
Chloroethene (vinyl chloride)
1,1-Dichloroethane (vinylidine chloride)
1,2-Trans-dichloroethene
Trichloroethene
Tetrachloroethene (perchloroethylene)
1,2-Dichloropropane
1,3-Dichloropropene
Hexachlorobutadiene
Hexachlorocyclopentadiene
Bromomethane (methyl bromide)
Bromodichloromethane
Dibromochloromethane
Tribromomethane (bromoform)
[a]Dichlorodifluoromethane
[a]Trichlorofluoromethane

Ethers

[a]Bis(chloromethyl) ether
Bis(2-chloroethyl) ether
Bis(2-chloroisopropyl) ether
2-Chloroethyl vinyl ether
4-Chlorophenyl phenyl ether
4-bromophenyl phenyl ether
Bis(2-chloroxy) methane

Table 4 (Cont'd)

Priority Pollutants

Monocyclic Aromatics

Benzene
Chlorobenzene
1,2-Dichlorobenzene (o-dichlorobenzene)
1,3-Dichlorobenzene (m-dichlorobenzene)
1,4-Dichlorobenzene (p-dichlorobenzene)
1,2,4-Trichlorobenzene
Hexachlorobenzene
Ethylbenzene
Nitrobenzene
Toluene
2,4-Dinitrotoluene
2,6-Dinitrotoluene

Phenols and Cresols

Phenol
2-Chlorophenol
2,4-Dichlorophenol
2,4,6-Trichlorophenol
Pentachlorophenol
2-Nitrophenol
4-Nitrophenol
2,4-Dimethylphenol
2,4-Dinitrophenol
p-Chloro-m-cresol
4,6-Dinitro-p-cresol

Phthalate Esters

Dimethyl phthalate
Diethyl phthalate
Di-n-butyl phthalate
Di-n-octyl phthalate
Bis(2-ethylhexyl) phthalate
Butyl benzyl phthalate

Polycyclic Aromatics

Acenaphthene
Acenaphthylene
Anthracene
Benzo (a) anthracene
Benzo (b) fluoranthene
Benzo (k) fluoranthene
Benzo (ghi) perylene
Benzo (a) pyrene
Chrysene
Dibenzo (a,h) anthracene
Fluoranthene
Fluorene
Indeno (1,2,3-cd) pyrene
Naphthalene
Phenanthrene
Pyrene

Nitrosamines and Miscellaneous Compounds

Dimethyl nitrosamine
Diphenyl nitrosamine
Di-n-propyl nitrosamine
Benzidine
3,3-Dichlorobenzidine
1,2-Diphenylhydrazine (hydrazobenzene)
Acrylonitrile

EXHIBIT B

State of California **Department of Health Services**

M e m o r a n d u m

Date : August 18, 1992

To : Office of Drinking Water Management Staff

From : Office of Drinking Water

(RETYPED NOVEMBER 2000)

Subject: Uniform Guidelines for the Disinfection of Wastewater

In 1987, ODW adopted guidelines for development of recommendations to Regional Water Quality Control Boards regarding disinfection requirements for discharges to surface waters. Since that time, several things have occurred which make some of the provisions of that document no longer applicable.

One of the provisions of the guidelines called for a minimum dilution of treated wastewater in streams or rivers of 20 to 1. In other words, a maximum of 5% wastewater in the rivers. The reason for this limitation was to provide an extra measure of protection where downstream domestic water supplies may only receive chlorination or inadequate filtration. This concern has been addressed through the adoption and implementation of the new Surface Water Treatment Rule. With this rule in place, the concern for inadequate downstream treatment and the need for the dilution requirement no longer exists.

As you know, the Title 22 criteria regulations are being finalized and will be adopted in the near future. These regulations will address further concerns such as use of tertiary treated wastewater for unrestricted recreation. It is our intention to evaluate the need for continuing or modifying the disinfection guidelines as soon as the regulations are in place. In the meantime, the current guidelines should only be referred to for recreational situations. Any provisions in the guidelines relating to protection of downstream domestic water supplies should be disregarded including the current minimum dilution rates. All discharges of wastewater or reclaimed water which may affect domestic water supplies should be considered on a case by case basis taking all factors into account.

Peter A. Rogers, Chief

cc: Regional Water Quality
Control Boards

EXHIBIT B

(RETYPED NOVEMBER 2000)

UNIFORM GUIDELINES FOR WASTEWATER DISINFECTION

The Sanitary Engineering Branch, State DHS, has prepared guidelines for various wastewater discharge situations for health protection. The Branch was guided in this endeavor by a policy committee of public health professionals within the Department.

Discharge situations in California range from remote ocean discharges by means of submarine outfalls to discharges to dry stream beds which pass through residential and popular park and recreation areas. The concept employed in the guidelines appropriately incorporates consideration of the type and degree of disease exposure in the establishment of disinfection requirements. Both available dilution and the type of receiving-water use are considered in the bacteriological criteria.

GENERAL PROVISIONS

Determination of Median Total Coliform Numbers

The median total coliform bacteria number should be based on the last seven samples for which analyses have been completed. All coliform values represent total coliform.

Sampling Frequency

Where a median coliform MPN of 23/100 ml or 240/100 ml is required, bacteriological samples should be collected at least twice per week. Where a median of 2.2 is required, or a significant discharge is involved, samples should be collected daily.

Maximum Bacterial Limits

The basic disinfection criteria, in terms of median coliform bacteria numbers, designate the operating levels which should be achieved for the particular discharge situation. The designation of a median coliform bacteria requirement does not address the serious situation where little or no disinfection is provided for a limited period. A maximum coliform bacteria limit may be designated to provide for this, however, the maximum limit should be significantly higher than the median so that it will not be exceeded due to statistical variations in the coliform test or other factors. Consequently, it is recommended that the maximum coliform bacteria number should be the concentration which is approximately 20 times the median coliform bacteria number.

SPECIFIC GUIDELINES

Category I. Proposed discharge is to:

Lakes and reservoirs.

There should be no discharge of sewage effluent to a lake or reservoir used for domestic water supply.

For lakes and reservoirs used for recreation where year-round confinement to land is not possible, the wastewater should be:

A. Confined to land, except for wet weather periods during the nonrecreational season. When there is a discharge, the wastewater should be disinfected to a median MPN of 23/100 ml.

B. If effluent is discharged during periods of recreational use, the effluent must be an adequately disinfected, oxidized, coagulated, and filtered wastewater. The effluent shall be considered adequately disinfected if the median MPN of coliform organisms does not exceed 2.2/100 ml.

Category II. Proposed discharge is to:

Accessible drainage ways or ephemeral streams with little or no natural flow during all or part of the year.

Accessible drainage ways and ephemeral streams which received waste discharges are often attractive areas for planned or unplanned recreational activities involving water contact. Further, there is generally little dilution available during the summer recreational season. The recommended disinfection criteria are related to the degree of public exposure.

A. A Category II discharge occurs where there is no nearby habitation and limited use of the discharge area. These areas are generally not identified as having recreation as a beneficial use, and contact with the waste discharge is not encouraged. In these situations, access should be limited, if possible, and posting of the area may be appropriate if there is potential for recreational use.

   Recommendation: The effluent must have a median coliform MPN not exceeding 23/100 ml.

B. A Category II discharge occurs where there is residential development in the vicinity of the receiving area, and the RWQCB has not designated water-contact recreation as a beneficial use. There may be ready access to the discharge area; however, water-contact recreation is not encouraged. Posting of the area may be appropriate.

   Recommendation: The effluent must be an adequately disinfected, oxidized wastewater.

   The wastewater shall be considered adequately disinfected if the median MPN of coliform organisms does not exceed 2.2/100 ml.

C. A Category II discharge occurs where the RWQCB has identified water contact recreation as a beneficial use of the receiving area and most of the following conditions are met:

1. The discharge occurs in, or upstream from, a residential area.

2. There is ready access to the discharged wastewater, and exclusion of the public is not a realistic alternative.

3. Historical attempts to post the stream to warn and exclude the public have been unsuccessful.

4. The recreation potential in the stream is high and justified because of weather, proximity to other recreation areas, etc.

5. There is public interest in recreational use of the water.

<u>Recommendation</u>: The effluent must be adequately disinfected, oxidized, coagulated, and filtered wastewater. The wastewater shall be considered to be adequately disinfected if the median MPN of the coliform organisms does not exceed 2.2/100 ml.

The Category II designations may not apply for discharges to agricultural drainage ways and sloughs which are remote or inaccessible. Such discharges may have less restrictive requirements depending on dilution and use.

<u>Category III.</u> <u>Proposed discharge is to</u>:

Freshwater streams and rivers.

There should be no discharge of sewage effluent to streams and rivers used for domestic water supply. The Department should recommend that no discharge be permitted to such water where land disposal or other options are physically or economically possible.

Where it is not possible to prevent a discharge to freshwater rivers and streams, the following disinfection recommendations apply:

Beneficial Use*		Ratio of Downstream Water Flow Effluent at Low Stream Flow***	
	<20:1	20:1 to 100:1	>100:1
Domestic water supply	No discharge	Median 23/100 ml ***	Median 240/100 ml ***

Swimming or other water contact	Use Category II.C ****	Median 23/100 ml	Median 240/100 ml
Agricultural use	Use Category II.C ****	Median 23/100 ml	Median 240/100 ml

* Beneficial use identified by RWQCB.

** The low flow is meant to be an average over a period of time, and not the instantaneous minimum low flow of the year.

*** If downstream domestic water supplies do not provide complete water treatment, a more strict disinfection requirement shall be recommended.

For discharge situations in this category, it may be appropriate to consider seasonal changes in use and dilution in recommending disinfection requirements. The recommended median MPN could be increased during the winter high-flow periods.

**** In situations where there is no dilution, the water reclamation criteria shall apply.

<u>Category IV.</u> <u>Proposed discharge is to</u>:

Saltwater recreation areas.*

Disinfection criteria for the protection of water-contact sports areas from discharges to ocean or bay waters are based on the degree of dilution which is available.

A.	Ocean discharge is remote from recreational water to the extent that bacteriological monitoring has documented that the discharge has no significant effect on recreational water quality.	No disinfection is required, provided that recreational waters meet Ocean Water Contact Sports Area Standards.
B.	Discharge is near recreational areas and could influence recreational water quality. Dilution is $\geq$100:1.	Median MPN 240/100 ml.
C.	Discharge is near recreational waters. Dilution is from 20:1 to 100:1.	Median MPN 23/100 ml.
D.	Discharge is to or near recreational water and there is little or no dilution (<20:1).	The effluent must be an adequately disinfected, oxidized, coagulated, filtered wastewater. Median coliform MPN 2.2/100 ml.

* Saltwater recreation areas such as lagoons and beaches situated at the mouth of ephemeral streams are subject to Category II requirements.

Category V. Proposed discharge is to:

Shellfish growing areas.

Shellfish growing areas in the vicinity of discharges constitute a particularly sensitive situation because of the ability of shellfish to concentrate contaminants. Outbreaks of hepatitis and other diseases transmitted by contaminated shellfish have been documented. In open coastal areas and some bays a high degree of dilution is available; however, in other bay systems, dilution is limited, and more restrictive disinfection requirements should be recommended.

A.	Ocean discharge is remote from shellfish waters to the extent that bacteriological monitoring has documented that the discharge has no significant effect on shellfish-growing water quality.	No disinfection is required provided that shellfish waters meet 70/100 ml (or a fecal coliform density of 14/100 ml). Median MPN 23/100 ml.
B.	Ocean or bay discharge where a high degree of dilution* is provided; however, the discharge can affect the quality of water overlying shellfish beds.	Median MPN 23/100 ml.
C.	Ocean discharge is in the vicinity of a shellfish growing area.	Required "closed area" to assure a high degree of dilution.* Median MPN 23/100 ml.
D.	Bay discharge where a high degree of dilution* cannot be assured by a "closed area".	Median MPN 2.2/100 ml and provision for maximum possible dilution and separation.

* A "high degree of dilution" means a dilution of 100 to 1 or greater.

EXHIBIT C

Central Contra Costa Sanitary District
Protecting public health and the environment
5019 Imhoff Place, Martinez, CA 94553-4392

FAX: (925) 372-7892

JAMES M. KELLY
General Manager

KENTON L. ALM
Counsel for the District
(510) 808-2000

ELAINE R. BOEHME
Secretary of the District

December 28, 2011

Mr. Vince Christian
California Regional Water Quality Control Board
San Francisco Bay Region
1515 Clay Street, Suite 1400
Oakland, CA 94612

Dear Mr. Christian:

CENTRAL CONTRA COSTA SANITARY DISTRICT, RESPONSE TO COMMENTS ON THE TENTATIVE ORDER NO. R2-2011-XXXX, NPDES PERMIT NO. CA0037648

As you know, Central Contra Costa Sanitary District (hereafter "District") has requested renewal of its National Pollutant Discharge Elimination System (NPDES) Permit No. CA0037648 and has been actively participating in the Regional Water Board's permit renewal process.

On October 31, 2011, and November 1, 2011, the Regional Water Board received two comment letters on the proposed Tentative Order (TO) for the District's NPDES permit. The parties submitting comment letters were (1) the San Luis and Delta-Mendota Water Authority and the State Water Contractors (hereafter "Water Agencies") and (2) the San Francisco BayKeeper (hereafter "BayKeeper"). The subject comment letters contained assertions in support of requests for significant changes in the District's NPDES permit. The proposed changes by the Water Agencies would require the District to construct and operate costly new energy intensive treatment facilities. At a minimum, the requested changes by the Water Agencies and BayKeeper would require significant expenditures for research and monitoring.

On December 8, 2011, Bay Area Clean Water Agencies (BACWA) submitted a comment letter in response to the comments from the Water Agencies. The District supports the comments submitted by BACWA and has prepared this letter and its attachments to provide additional detail on some of the major assertions made in the subject comment letters. The District recognizes that the formal comment period for this TO closed on November 1, 2011, but respectfully request that these comments be entered into the record pursuant to Title 23 of California Code of Regulations, section 648.1(d).

The District understands the State and Regional Water Board's need to identify and evaluate nutrient-related problems, develop appropriate regulatory tools, and devise a long term nutrient management strategy for the San Francisco Bay Estuary. The District is supportive of the collaborative efforts that are underway with the Regional Water Board and other parties to develop a better understanding of the role of ammonium and other nutrients in the San Francisco Bay ecosystem. The District has taken a proactive and collaborative approach in working with San Francisco Estuary Institute (SFEI), the State Water Board, Regional Water Board and others on the San Francisco Bay Numeric Nutrient Endpoint (NNE) program, and also in working with various participants in the ongoing Suisun Bay study by funding aspects of the project in the 2010/2011 study season and continuing funding for the 2011/2012 second season study.

Unfortunately, the assertions made by the Water Agencies presuppose the outcomes of these ongoing efforts, rely on studies that have not been peer reviewed by Bay-Delta scientific experts, and greatly overstate the current understanding and level of certainty surrounding these issues. As a result, the District is obligated to respond to these assertions to establish a proper factual basis upon which to base permitting decisions that could cost the citizens of Contra Costa County from $70 to $150 million or more dollars in capital expenditures, based on preliminary studies conducted by our District.

This letter briefly summarizes the main assertions made in the comment letters submitted by the Water Agencies and BayKeeper, and describes the District's fundamental responses to those assertions. Detailed responses supported by expert advice from a team of scientists are contained in the attachments to this letter.

The following assertions pertaining to the District's discharge are made in the comment letter by the Water Agencies:

1. The discharge is causing or contributing to toxicity to aquatic organisms that are important to the Bay-Delta food web (page 8 #II – 1);
2. The discharge is adding ammonium loadings to Suisun Bay that are inhibiting diatom blooms and resulting in disruption of the Bay-Delta food web (page 9 # II - 2);
3. The discharge is changing nutrient ratios in the Delta, which is causing harmful effects in the Bay-Delta ecosystem (page 10 # II – 3);
4. Immediate actions to reduce nutrient loadings in the discharge would yield benefits to the ecosystem (page 10 # II – 4);
5. The proposed permit inappropriately assigns a mixing zone and dilution credit for ammonia (page 11 # III – A); and

6. The proposed permit fails to meet the requirements of the State and federal antidegradation policies (page 13 # III – B).

Assertions No. 1 through 4 are based on study results that have serious unresolved questions. In some instances, independent peer review has not been performed. In other instances, peer review has produced serious questions about the study results. In yet other instances, the study results are, in fact, untested hypotheses, which are being misrepresented as fact. In no case have these assertions been based on the use of adopted water quality objectives or water quality criteria developed or endorsed by the State of California or the USEPA.

For Assertion No. 1, the Water Agencies rely heavily on the results of a recently issued report by Dr. Swee Teh et al. to allege the existence of ammonia toxicity in Suisun Bay.[1] Serious questions exist regarding the key findings of that report, which has not been independently peer reviewed. Additionally, significant technical flaws exist in the Water Agencies' subsequent use of those findings to allege toxicity impacts in Suisun Bay. Assertions No. 2 through 4 are inconsistent with the findings of the team of highly esteemed coastal estuarine experts charged with evaluating the impacts of nutrients, including ammonium, on the San Francisco Bay Estuary as part of the development of NNEs.[2] The assertion that the District's discharge is disrupting the Delta food web by changing the nutrient balance in the estuary asserts hypothetical information as fact which has not been tested or accepted by San Francisco Bay scientific experts.

Assertion No. 5 is based on a flawed interpretation of both the San Francisco Bay Basin Plan and the detailed modeling studies performed by the District to establish a reasonable mixing zone and dilution credits. As described in the Tentative Order, the District's studies are consistent with numerous others that have been evaluated and approved by the Regional Water Board.

Assertion No. 6 is based on a legal interpretation of the antidegradation policies that is not consistent with State and Federal guidelines and policy precedents. In fact the Tentative Order is entirely consistent with the State and Federal policies.

Detailed responses to these assertions are included in the attached documents.

The following assertions and requests are made by the BayKeeper:

1. Effluent limits are needed for chlorine residual and settleable matter.

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[1] Teh, Swee; Flores, Ida; Kawaguchi, Michelle; Lesmeister, Sarah; and The Ching; *Full Life-Cycle Bioassay Approach to Assess Chronic Exposure of* <u>*Pseudodiaptomus forbesi*</u> *to Ammonia/Ammonium*, University of California at Davis; submitted to the State Water Resources Control Board pursuant to Agreement No. 06-447-300 (August 2011), (Teh et al., 2011).
[2] McKee, Lester; Sutula, Martha; Gilbreath, Alicia; Beagle, Julie; Gluchowski, David; Hunt, Jennifer; *Nutrient Numeric Endpoint Development for the San Francisco Bay Estuary: Literature Review and Data Gaps* (June 2011), (Hereinafter, McKee et al. 2011).

2. The permit should contain monitoring requirements for personal care products and should address sediment toxicity.

These suggested changes to the Tentative Order are also not appropriate, for reasons detailed in the attached documents.

In closing, it is important to recognize that no simple pollution prevention options exist for nutrient removal by POTWs. Preliminary studies of our treatment facilities indicate that removal of ammonia and other nutrients will require significant capital improvements. These studies also indicate that additional treatment can have significant environmental implications in terms of energy consumption and greenhouse gas emissions. We are continuing to fund studies to research evolving treatment technologies that may lead to less energy intensive methods of removing ammonia and nutrients in parallel with our contributions to studying the impact of ammonia and nutrients on the Suisun Bay. We continue to believe that such costly technological changes should not be undertaken without robust evidence that they are necessary and will provide benefits to the San Francisco Bay ecosystem commensurate with the economic and environmental costs.

This letter provides information that illustrates that the requested changes to the draft permit cannot be justified at this time, since the robust evidence needed to support such changes is not currently available. As such, the appropriate action by the Regional Water Board is the adoption of the draft permit as publicly noticed.

Our District mission is to protect the public health and the environment and we are committed to working with you to study the complex water ecosystem of the Suisun Bay to determine if changes in our operations or treatment technology are needed to protect it for future generations. Please contact me if you have any questions or comments regarding the content of this letter or any of the attachments.

Sincerely,

James M. Kelly

James M. Kelly, P.E.
General Manager

JMK/MPO/AEF/BTT:dp

Attachments:

1 – Responses to Comments on Tentative Order for NPDES permit for Central Contra Costa Sanitary District made by Water Agencies and San Francisco BayKeeper

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Appendix A - A Critical Review of: Full Life-Cycle Bioassay Approach to Assess Chronic Exposure of *Pseudodiaptomus forbesi* to Ammonia / Ammonium - Final Report. Dated August 31, 2011
Prepared by: Teh S, Flores I, Kawaguchi M, Lesmeister S, Teh C
Aquatic Toxicology Program, Department of Anatomy, Physiology, and Cell Biology, School of Veterinary Medicine, University of California Davis
This Critical Review Was Prepared By: Pacific EcoRisk, Inc.
2250 Cordelia Rd. Fairfield, CA 94534

Appendix B – Phytoplankton (Chlorophyll a) versus Ammonium Concentrations in Suisun Bay [1977 – 2010]

cc: CCCSD Board of Directors
Amy Chastain, Executive Director, Bay Area Clean Water Agencies
Ann E. Farrell, P.E., Deputy General Manager
Margaret P. Orr, P.E., Director of Plant Operations

Attachment 1

Responses to Comments on Tentative Order for NPDES permit for Central Contra Costa Sanitary District made by Water Agencies and San Francisco BayKeeper

December 27, 2011

On October 31, 2011 and November 1, 2011, the Regional Water Board received two comment letters on the proposed Tentative Order to renew the NPDES permit for Central Contra Costa Sanitary District (hereafter "District"). The parties submitting comment letters were (1) the San Luis and Delta Mendota Water Authority and the State Water Contractors (hereafter "Water Agencies") and (2) the San Francisco BayKeeper (hereafter "BayKeeper").

The comment letters from the Water Agencies and the BayKeeper raise a number of issues that are currently being studied and are unresolved in the scientific community. Issues are also raised regarding how certain technical and regulatory criteria, such as dilution and anti-degradation, are applied. In order to inform the dialogue on these issues, the District has prepared the following response.

Statement No. 1 –Excessive ammonium has been shown to be toxic to copepods. [10-31-11 Water Agencies letter page 8 # II -1]

Response: As detailed in the comment letter submitted by BACWA on December 8, 2011 and described further below, this statement is not fully supported by scientific data.

1. This statement relies on toxicity threshold values cited in a recent report prepared by Dr. Swee Teh et al. dated August, 2011.[1] The report summarizes research pertaining to one copepod species (*Pseudodiaptomus forbesi*) which is present in the Delta. Based on our review, this report has not been adequately peer reviewed and is not of sufficient quality to merit its use in a regulatory context. In fact, serious issues exist with the basic research, including the validity of toxicity threshold values derived from that research, the test methodology, and the reporting of methods and results. Examples of these issues are described in a memorandum prepared by

[1] Teh, Swee; Flores, Ida; Kawaguchi, Michelle; Lesmeister, Sarah; and The Ching; *Full Life-Cycle Bioassay Approach to Assess Chronic Exposure of* Pseudodiaptomus forbesi *to Ammonia/Ammonium*, University of California at Davis; submitted to the State Water Resources Control Board pursuant to Agreement No. 06-447-300 (August 2011). (Teh et al., 2011). Available at http://www.swrcb.ca.gov/rwqcb5/water_issues/delta_water_quality/ambient_ammonia_concentrations/tehetal_ammonium_exposure2011.pdf.

Pacific EcoRisk, Inc., which is included as Appendix A. The findings of this memorandum are summarized as follows:

"The reviewer is troubled by the absence of any discussion by Teh et al. regarding the variability in their test response data, either between tests or within tests (i.e., inter-replicate variability). Without such acknowledgement, it is left for the non-scientist to assume that the data as presented are definitive. Moreover, it raises the question of whether the data from this study are adequate (or 'ready') for use in regulatory decision-making. However, it is important to note that this critical review is not intended to negate Teh *et al.'s* observations that ammonia is toxic to naupliar, juvenile, and /or adult *P. forbesi* at elevated concentrations and that this toxicity is strongly influenced by pH. Indeed, the primary question of 'what are the effects of ammonia on *P. forbesi'* is relevant and Teh *et al.'s* study results certainly compel a more thorough examination of this. However, the problems associated with Teh *et al.'s* experimental methodology for Subtasks 3-3 and 3-4-1 and significant questions regarding the analysis of the resulting data do indicate that the quality of the work should preclude the resulting 'critical threshold' data ….. from being used for regulatory purposes."

2. Data on abundance of copepods in Suisun Bay does not support the allegation of reduced abundance or ammonia toxicity to copepods.

   Recent publications provide information that contradict the Water Agencies' comment letter regarding the impact of the District discharge on copepod abundance. For example, the Dr. Teh et al. report notes that the California Department of Fish and Game 2007 to 2009 20 mm survey for *P. forbesi* found that the abundance at station 711 (near Rio Vista) increased, despite the presence of higher levels of ammonium at this location than exist in Suisun Bay (mean ammonium concentration of 0.27 mg/L versus mean ammonium concentration 0.15 mg/L at Martinez (Station 405)).

   Additionally, the Interagency Ecological Program (IEP)'s Spring 2009 newsletter reported that *P. forbesi*, an introduced species first detected in 1988, "…has declined slightly since its introduction, [but] has remained relatively abundant in summer and fall compared to other copepods."[2] The Spring 2009 newsletter further noted that "[s]ummer abundance also increased slightly from 2007 to 2008, while fall abundance increased moderately and was the highest since 2002."[3] This evidence of increasing abundance of *P. forbesi* in Suisun Bay, despite the increased ammonia loadings and the increased ammonia ambient concentrations which are acknowledged for this period, is inconsistent with the allegation that ammonium toxicity is negatively impacting the abundance of this copepod in the Bay-Delta.

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[2] Interagency Ecological Program Newsletter, Vol 22., No. 2, (Spring 2009). p. 11 Available at http://www.water.ca.gov/iep/newsletters/2009/IEPNewsletterFINALSpring2009.pdf).

[3] *Id.*

3. Knowledge of the mechanistic linkages between various stressors and the Bay-Delta food web is lacking. The allegation that ammonium is an important stressor impacting the Bay-Delta food web is hypothetical; in fact, the Delta science community is well aware that ammonia is only one potential stressor out of a list of many known stressors affecting the Delta ecosystem. Much greater evidence exists that other stressors, including benthic grazing by invasive clams and changes in Delta flow regimes, have impacted the food web at a macroscopic level.

   In August, 2010, the State Water Resources Control Board (SWRCB) issued its recommendations to the State Legislature regarding the establishment of flow criteria for the Sacramento-San Joaquin Delta.[4] Those criteria called for a significant increase in Delta outflows over recent levels and, among other things, included "…flow criteria in the Delta to help protect fish from mortality in the central and southern Delta resulting from operations of the State and federal water export facilities."

   The following excerpts from the SWRCB's Delta flow criteria report highlight the importance of evaluating multiple factors when considering the health of the ecosystem:

   > "Flow is important to sustaining the ecological integrity of aquatic ecosystems, including the public trust resources that are the subject of this proceeding. Flow affects water quality, food resources, physical habitat, and biotic interactions. Alterations in the natural flow regime affect aquatic biodiversity and the structure and function of aquatic ecosystems."[pg 39]
   >
   > "The best available science suggests that current flows are not sufficient to protect public trust resources." [pg 2], and
   >
   > "The flow criteria identified in this report highlight the need…to develop an integrated set of solutions, to address ecosystem flow needs, including flow and non-flow measures….Although flow modification is an action that can be implemented in a relatively short time in order to improve the survival of desirable species and protection of public trust resources, public trust resource protection cannot be solely achieved through flows – habitat restoration is also needed." [pg 7]

   The SWRCB's Delta flow criteria report acknowledged that water quality issues, including ammonia and nutrients, should be evaluated and considered in the adaptive management of the Delta. However, both ammonia and nutrients were given lesser emphasis than flow in the SWRCB report, contrary to the content and implications of the Water Agencies comment letter.

[4] State Water Resource Control Board; *Development of Flow Criteria for the Sacramento-San Joaquin Delta Ecosystem*. (August 2010). Available at http://www.swrcb.ca.gov/waterrights/water_issues/programs/bay_delta/deltaflow/final_rpt.shtml

Statement No. 2 – The excess ammonium is inhibiting nitrogen uptake by diatoms and reducing diatom primary production in the Bay-Delta. [10-31-11 Water Agencies letter page 9 # II-2]

Response: The statement greatly overstates our knowledge regarding the existence and/or importance of ammonium effects on phytoplankton blooms in Suisun Bay or the Bay-Delta food web.

The importance of the inhibition effect is not well understood, particularly in the context of other factors (benthic grazing and light limitation) that are known to impact phytoplankton blooms in Suisun Bay. Numerous instances have been documented where ammonium levels below the inhibition threshold (0.056 mg/l) developed by Dr. Richard Dugdale and commonly cited by the water Agencies have not triggered phytoplankton blooms in Suisun Bay. Figures depicting these occurrences are provided in Appendix B. Research on this topic is ongoing in Suisun Bay, funded, in part, by the State and Federal water contractors and Central Contra Costa Sanitary District. Therefore, the parties making the definitive comments on the District's Tentative Order are well aware that the science is unsettled on the very points that they are alleging in the comment letter.

In the McKee et al. report prepared for the Regional Water Board by SFEI and the Southern California Coastal Water Research Project (SCCWRP), numerous statements are made which contradict the assertion that ammonium is commonly accepted as having a significant impact in San Francisco Bay. The report acknowledges the suggestion by Dr. Richard Dugdale and other researchers from the Romburg Tiburon Center that "ammonium inhibition could be one of the limiting factors that control primary productivity in the Bay."[5] However, the report goes on to state that the impacts of ammonium on diatom blooms is not well-understood, is just one of many factors known to affect productivity, and that additional work is needed to resolve this issue:

> "…the ecological importance of ammonium inhibition of spring diatom blooms is not well understood relative to factors known to control primary productivity…" [pg 147]
>
> "In SF Bay, the biomass associated with phytoplankton, measured as surface water chlorophyll *a* concentration, varies in space and time in response to nutrient availability from external loads and internal regeneration, grazing, stratification, water temperature, tidal energy, transparency, wind/wave energy, the availability of seed cysts, UV radiation effects on nitrate versus ammonium assimilation perhaps due to disruptions of enzyme pathways, differential uptake of nitrate and ammonium by larger versus smaller cells, inhibition of nitrate uptake by ammonium, predation by benthic invertebrates, and variations in the phase of the Pacific Decadal Oscillation and related changes to top down predation of benthic invertebrates."[pg 153]
>
> "…the effect of ammonium inhibition on phytoplankton productivity throughout the Bay has not been modeled vis-à-vis other contributing factors…the next logical step is to develop models

[5] *McKee et al. 2011*.

> that synthesize understanding of the relative importance of ammonium and urea versus other factors controlling phytoplankton assemblages."[pg 46]
>
> "Elevated ammonium concentrations have been suggested as a major mechanism by which spring diatom blooms appear to be suppressed in the North Bay and Lower Sacramento River…Despite this evidence, the ecological importance of ammonium inhibition of spring diatom blooms is not well understood relative to factors known to control primary productivity, particularly in other regions of the Bay where water column chlorophyll *a* appears to be increasing. Thus, *the linkage between ammonium concentrations and Bay beneficial uses is not at this time universally accepted*. San Francisco Bay Technical Advisory Team (TAT) members agree that additional data synthesis is required to better understand the role of ammonium in SF Bay."[pg 154]

It is important to note that members of the TAT responsible for scientific review of and input on the NNE document include Dr. James Cloern, a highly recognized expert in San Francisco Bay ecology and two members from the Romburg Tiburon Center, including Dr. Dugdale. The cited statements and recommendations of the NNE report should therefore be interpreted as current prevailing scientific opinion regarding the role of ammonium in Suisun Bay phytoplankton dynamics.

Statement No. 3 – Nutrient discharges into the Bay-Delta estuary are contributing to a shift in algal communities by changing the nutrient ratios to favor harmful, invasive species. [10-31-11 Water Agencies letter page 10 # II-3]

Response: This statement is largely based on two papers, funded by the State Water Contractors, that offer hypothetical arguments based on selective correlation analysis that, in part, have been rejected by the scientific community and otherwise have not been accepted as fact.

The Water Agencies allege that research by Dr. Patricia Glibert confirms that nutrient loadings from the District contribute to changes in nutrient ratios in Suisun Bay, and that those changed ratios explain adverse ecosystem changes in the Bay-Delta, including the precipitous decline of key fish species.[6] In fact, the cited work has not been accepted or endorsed by leading Bay-Delta scientists. For example, the San Francisco Bay NNE science team considered Dr. Glibert's 2010 paper, but neither endorsed it or adopted it as fact in the final McKee et al. 2011 NNE report.

It should also be noted that the work by Glibert in 2010, funded by the State Water Contractors, was criticized for its inappropriate use of statistical methods and other issues. In a peer-reviewed paper titled "Perils of Correlating CUSUM-transformed variables to infer ecological relationships (Breton et al...

[6] Glibert, Patricia; *Long-Term Changes in Nutrient Loading and Stoichiometry and Their Relationships with Changes in the Food Web and Dominant Pelagic Fish Species in the San Francisco Estuary, California*; Reviews in Fisheries Science, Vol. 18, Issue 2 (August 2010). (Glibert, 2010). Available at http://www.sfcwa.org/2011/05/20/sed-lobortis-tellus-vel-ligula-pretium-mollis/.

2006, Glibert 2010)"[7] the authors James Cloern, Alan Jassby, Jacob Carstense, William Bennett, Wim Kimmerer, Ralph MacNally, David Schoellhamer and Monika Winder stated the following:

- "Glibert (2010) concluded that recent large population declines of diatoms, copepods, and several species of fish were responses to a single factor – increased ammonium inputs from a municipal wastewater treatment plant."
- "Glibert's study…contradicts the overwhelming weight of evidence that population collapses of native fish…and their supporting food webs in the San Francisco Estuary are responses to multiple stressors including landscape change, water diversions, introductions of exotic species and changing turbidity."
- "…CUSUM transformation, as used by…Glibert (2010), violates the assumptions underlying regression techniques."
- "…CUSUM-transformed variables often have an apparent statistically significant correlation even when none exists…"
- "…Glibert (2010) inferred a strong negative association between delta smelt abundance and wastewater ammonium from regression of CUSUM-transformed time series. However, the…correlation… is not significant…"
  The Glibert 2010 work was also criticized as being incomplete for not having analyzed the importance of other factors, including export volumes, benthic grazing by invasive clams, major changes in the hydrologic regime in the Delta, and other stressors that are commonly recognized as major contributors to stress on the Delta ecosystem.

The recently released Glibert et al. 2011 paper[8]- funded in part by the State Water Contractors, the San Luis & Delta-Mendota Water Authority and Metropolitan Water District - has not yet been effectively scrutinized by the San Francisco Bay NNE science team or other Bay-Delta experts. On its face, the subject paper is not a definitive piece of work on the effect of nutrients on the Bay-Delta ecosystem. The paper instead offers ecological stoichiometric theory as a hypothetical framework for consideration and suggests that nutrient stoichiometry may be a significant driver influencing food webs in the Bay-Delta ecosystem. The paper asserts the potential validity of this theory based on extensive, albeit selective, correlation analysis. The paper relies, at least in part, on the statistical analysis from the Glibert 2010 paper that was so roundly criticized. The paper does not assert that it has developed conclusive scientific evidence for its theories applicable to the San Francisco Bay or Delta.

In fact, excerpts from the Glibert et al. 2011 paper state that "while compelling, the ecological stoichiometric model raises many questions that need further analysis in the San Francisco Estuary…"

[7] Cloern, J.E., A.D. Jassby, J. Carstensen, W.A. Bennett, W. Kimmerer, R. Mac Nally, D.H. Schoellhamer and M. Winder. 2011. *Perils of correlating CUSUM-transformed variables to infer ecological relationships (Breton et al. 2006, Glibert 2010)*. Limnology and Oceanography, in press.

[8] Glibert, Patricia; Fullerton, David; Burkholder, Joann; Cornwell, Jeffrey; Kana, Todd. *Ecological Stoichiometry, Biogeochemical Cycling, Invasive Species, and Aquatic Food Webs: San Francisco Estuary and Comparative Systems*. Reviews in Fisheries Science, Vol. 19, Issue 4 (October 2011). (Glibert et al., 2011).

and "…regulation of the food web by nutrient controls is directly testable…there is much that needs to be explored to test these relationships directly."[9]

In summary, the cited papers by Glibert offer theories that are strongly supported by the Water Agencies but that have not been accepted or endorsed by the Bay-Delta scientific community, the Delta Science Program or any other reputable scientific body. These theories, while interesting and perhaps worthy of further exploration, are not an appropriate basis for the imposition of very costly changes to municipal wastewater management in the San Francisco Bay region.

Statement No. 4 – Where implemented in impacted ecosystems, nutrient removal has improved the natural ecosystem and aquatic life. [10-31-11 Water Agencies letter page 10 # II-4]

Response: This statement pre-supposes the outcome of the San Francisco Bay NNE process and other efforts to address the issue of nutrient management in San Francisco Bay.

The Water Agencies allege that nutrient load reduction, as a general management action, will create various benefits to the Bay-Delta ecosystem. This overarching philosophy is offered as a rationale to support the imposition of restrictive effluent limits in the District's permit to force nitrogen load reductions. Such generalized statements are not borne out by the main body of scientific research on this topic. For instance, the following excerpts are taken from a 2010 report by Damann Anderson and Anthony Janicki[10] published by the Water Environment Research Foundation which investigates the complexity of nutrient management decision making:

> "…nutrient water quality impacts are typically waterbody specific, and thus waterbody specific assessments are necessary to develop appropriate nutrient numeric criteria…"
>
> "…determination of the causative agents for eutrophication impairment is not straightforward, but needs to be determined prior to developing management decisions…"
>
> "…all benefits and costs of available nutrient controls should be evaluated for all stakeholders prior to implementation…"

The following statement is particularly relevant to the San Francisco Bay-Delta estuary:

> "Nutrient load reductions to a waterbody that is light-limited…may show no change in resultant water quality. Hydrologic alterations affect residence time in a waterbody and can also confound the relationship between nutrient loading and water quality conditions."

The ongoing Numeric Nutrient Endpoint (NNE) development effort in the San Francisco Estuary being led by the State Water Board, Regional Water Board, the San Francisco Estuary Institute, and various

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[9] *Id* at 84.

[10] Anderson, D.L and A. Janicki. 2010. *Linking Receiving Water Impacts to Sources and to Water Quality Management Decisions: Using Nutrients as an Initial Case Study*, Prepared for Water Environment Research Foundation, WERF 3C10

stakeholders is a proper scientific and policy forum for the evaluation of complex nutrient issues. McKee et al. noted that "evidence is building that the historic resilience of San Francisco Bay to the harmful effects of nutrient enrichment is weakening."[11] The NNE effort is a systematic study to address the need for future nutrient management actions. That effort is ongoing and has yet to produce definitive recommendations regarding nutrient criteria or a nutrient management plan. An important component of the NNE framework in San Francisco Bay is the development of load-response models that can simulate the ecological response of the Estuary to nutrients and other important co-factors.

The following statements are made in the McKee et al. report:

> "Estuaries within California are highly variable in how they respond to nutrient loading due to differences in physiographic setting, salinity regime, frequency and timing of freshwater flows, magnitude of tidal forcing, sediment load, stratification, residence time, denitrification, etc."[Page 8]

With regard to San Francisco Bay, specifically, the report states:

> "…San Francisco Bay has long been recognized as an estuary in which phytoplankton biomass and pelagic primary productivity is not driven by simple nutrient limitation, due to a variety of co-factors that modulate primary producer response to nutrients…" [Page 70]

It is clear that the generalized allegations and associated permit demands by the Water Agencies should not deter or distract from the ongoing NNE effort as the proper vehicle for addressing nutrient management questions in San Francisco Bay and for determining whether nutrient load reductions will provide commensurate benefits.

Statement No. 5 - The Regional Board's application of a dilution factor is flawed and should be reconsidered. [10-31-11 Water Agencies comment letter page 11 # III - A]

Response: The Regional Water Board has followed established regulatory policies and procedures in evaluating the dilution characteristics of the District's discharge and in using that information in the derivation of effluent limits for ammonia in the Tentative Order.

The six specific comments made by the Water Agencies pertaining to the proposed dilution credits for ammonia in the District's Tentative Order are addressed below, on a point-by-point basis:

> *"The Public Water Agencies are concerned that the Regional Board staff has erred in its application of a dilution factor to set effluent limits for ammonium. As the Tentative Order acknowledges, the applicable Basin plan had Water Quality Objectives for un-ionized ammonia of 0.025 mg/L (annual median) and 0.16 mg/L (maximum) upstream of the Bay Bridge. Tentative Order, Attachment F at F-23. As the un-ionized component of the total ammonia is only small fraction of the total discharges, these are then converted to total ammonia objectives of 5.0*

[11] McKee et al. 2011 at page 161.

mg/L (acute) and 1.6 mg/L (chronic). Given that the MEC for ammonium in 30.2 mg/L, there unquestionably is a reasonable potential to exceed these objectives. However, the Tentative Order then proceeds to allow a substantial dilution for total ammonia to set the effluent limits relying on the "Mixing Zone Study." Yet, this would not appear to be appropriate for several reasons:"

1. "Regional Board staff acknowledges the inability to set a mixing zone."

The language referenced by the Water Agencies and included by Regional Water Board staff in the Fact Sheet of the Tentative Order (Section IV.C.4.b.ii.(2), page F-20) addresses the difficulties in setting a mixing zone for persistent, non-bioaccumulative pollutants. The language was provided as a justification for limiting the dilution credit for copper, cyanide, and bis(2-ethylhexyl)phthalate to 10:1 even though the District's discharges achieve a much greater dilution (43:1 based on average dry weather flow rates, 33:1 based on peak flow rates). The Regional Water Board staff did not state they were unable to determine a mixing zone, just that a conservative approach is warranted for such pollutants. The referenced language does not pertain to ammonia, which is neither persistent nor bioaccumulative.

2. "Regional Board granted a mixing zone for total ammonia even though they acknowledged inability to set a mixing zone. In addition, they are applying dilution credits for control of un-ionized ammonia, not ammonium which is the more serious constituent of concern."

As mentioned in the response to Issue #1, Regional Water Board staff did not state they were unable to set a mixing zone, just that a limited dilution credit/mixing zone is appropriate for non-bioaccumulative, persistent pollutants. A mixing zone (based on initial dilution) was granted for total ammonia because it is a non-bioaccumulative, non-persistent pollutant. The Basin Plan (Section 3.3.20) includes objectives for un-ionized ammonia (which are translated to total ammonia when setting effluent limits) to "protect against the chronic toxic effects of ammonia in the receiving waters" but acknowledges "in most instances, ammonia will be diluted or degraded to a nontoxic state fairly rapidly." No chronic effects are expected from ammonia and granting a mixing zone based on initial dilution will not impact water quality. The San Francisco Bay Basin Plan[12] does not include objectives for ammonium. The Water Agencies' concerns regarding ammonium are based on recent research on copepod toxicity by Teh et al. 2011 that has not been properly peer-reviewed, and unproven theories of phytoplankton inhibition by Dr. Richard Dugdale that have not been translated into water quality criteria or surrogates for such criteria. Additional work is needed to establish meaningful water quality thresholds before they can be used in the derivation of effluent limits.

3. "The Basin Plan cautions against application of dilution credits in light of various concerns, including the difficulty in measuring the discharge in a tidal zone."

[12] California Regional Water Quality Control Board, San Francisco Bay Region; *San Francisco Bay Basin (Region 2) Water Quality Control Plan* (December 31, 2010). (Basin Plan 2010).

The section of the Basin Plan referenced by the Water Agencies (Section 4.6.1.1.) includes the rationale for limiting dilution credits and not approving mixing zones that extend outside the zone of initial dilution. Regional Water Board followed this approach when determining effluent limits for total ammonia. A "Mixing Zone Study" prepared for the District in 2011 utilized the U.S. EPA-approved CORMIX model to delineate the shape of the discharge plume. [13] CORMIX is approved for use in assessing environmental impacts of regulatory mixing zones that result from continuous point source discharges[14]. Near-field mixing processes were modeled for the District that included buoyant jet mixing (receiving water currents and merging of individual port's plumes) and boundary interactions (sediment bed, water surface, and density gradient effects). The plume shape was conservatively delineated by the surface area containing one standard deviation (i.e., 68%) of the plume in a Gaussian distribution-shaped cross-section. Initial dilution was assumed to be complete when the plume's discharge momentum and buoyancy dissipate. Although turbulent diffusion subsequently dilutes the effluent plume even more, initial dilution is commonly applied for calculating effluent limitations. The edge of the regulatory mixing zone is delineated by this near-field region, a term used in CORMIX output to describe the zone of strong initial mixing where "near-field" processes occur.

4. "The "Mixing Zone Study" indicated the plume re-stratifies and dilution does not persist beyond the zone of initial dilution."

The comment from the Water Agencies is based on a misunderstanding of the applicability of the CORMIX plume model. The model results apply to near-field mixing associated with buoyant jet mixing (the jet from each diffuser port rising due to buoyancy, spreading due to turbulence, deflecting due to receiving water current effects, and merging with neighboring port's plumes) and boundary interactions (the Bay's sediment bed and water surface, density gradients in the water column). The "plume" refers to whatever remaining portion of the initial discharge can still be distinguished from the ambient receiving water at the point of interest.

Stating that dilution "does not persist beyond the zone of initial dilution" implies an impossible physical process of the plume reconstituting or reassembling itself. Dilution of a plume only works in one direction. While the initial, momentum-induced mixing essentially ends at the edge of the mixing zone, subsequent dilution occurs through the processes of dispersion and advection. Far-field hydrodynamic modeling can be applied to predict additional dilution beyond the edge of the mixing zone.

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[13] Larry Walker Associates; *Near-Field Mixing Zone and Dilution Analysis for the Central Contra Costa Sanitary District Outfall Diffuser to Suisun Bay*, (May 27, 2011). Prepared for the Central Contra Costa Sanitary District. (LWA 2011).

[14] Doneker, R.L., and G.H. Jirka; *CORMIX User Manual – A Hydrodynamic Mixing Zone Model and Decision Support System for Pollutant Discharges into Surface Waters* (2007). EPA-823-K-07-001.

In fact, the RMA-2 model was used in 2000 to determine dilution of the District's effluent away from the outfall[15] and in 2008 to estimate currents (speed and direction) at the District's diffuser.[16] RMA-2 is a generalized free surface hydrodynamic model that is used to compute two-dimensional depth-averaged velocity and water surface elevation. The RMA Bay-Delta model extends from the Golden Gate to the confluence of the American and Sacramento Rivers and to Vernalis on the San Joaquin River. A 15-minute tidal boundary time series is applied at the Golden. Time series of daily average inflow boundary conditions are applied for the Sacramento River, San Joaquin River, Yolo Bypass, San Joaquin River, Cosumnes River, Mokelumne River, miscellaneous eastside flows which include Calaveras River and other minor flows, and Napa River. Delta exports applied in the model include the State Water Project, the Central Valley Project, Contra Costa exports and North Bay Aqueduct intake at Barker Slough. The model provides a detailed spatially-varying and time-varying description of tidal directions and velocities. Dynamic boundary conditions assure accurate computation of net Delta outflows as they vary with inflow conditions and exports. Because RMA-2 is a depth-averaged model, density driven flows are not included. Estuarine conditions are otherwise well-represented.

Extreme low Delta flow conditions (average net Delta outflow of 2,300 cubic feet per seconds (cfs) occurring October 1-31, 1977) were simulated in 2000 along with the District's permitted average dry weather discharge flow rate (53.8 mgd). A conservative tracer was used to represent District's discharges. The model produced output every 15 minutes during the simulation, capturing the effects of ingoing/outgoing tides, delta and riverine flows, and discharges from wastewater treatment plants throughout the San Francisco Bay. Minimum dilution near the outfall was observed approximately 10 days after tracer release.

The 2000 tracer simulation results indicated that the District's discharge plume is highly diluted in the Bay with a minimum dilution of 200:1 near the outfall (outside the zone of initial dilution). The minimum 200:1 dilution occurs at slack after flood tide and at slack after ebb tide. The dilute plume then oscillates with the tide along the southern shore of Carquinez Strait and Suisun Bay. Results of the tracer simulation illustrate that under periods of low Delta outflow, the discharge plume may extend upstream some distance (at 500:1 dilution), but will not reach the City of Antioch. Under high Delta outflow conditions the extent of the discharge plume is much smaller and the dilution is much greater as shown in the following figure (Figure 28 excerpted from RMA 2000).

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[15] Resource Management Associates, Inc; *Water Quality Impacts of Central Contra Costa Sanitary District Discharge on San Francisco Bay* (August 2000), prepared for Larry Walker Associates and Central Contra Costa Sanitary District. (RMA 2000).

[16] Resource Management Associates, Inc. *Numerical Modeling of Central Contra Costa Sanitary District Discharge in San Francisco Bay – Technical Summary Report* (June 2008), prepared for Larry Walker Associates and Central Contra Costa Sanitary District. (RMA 2008).

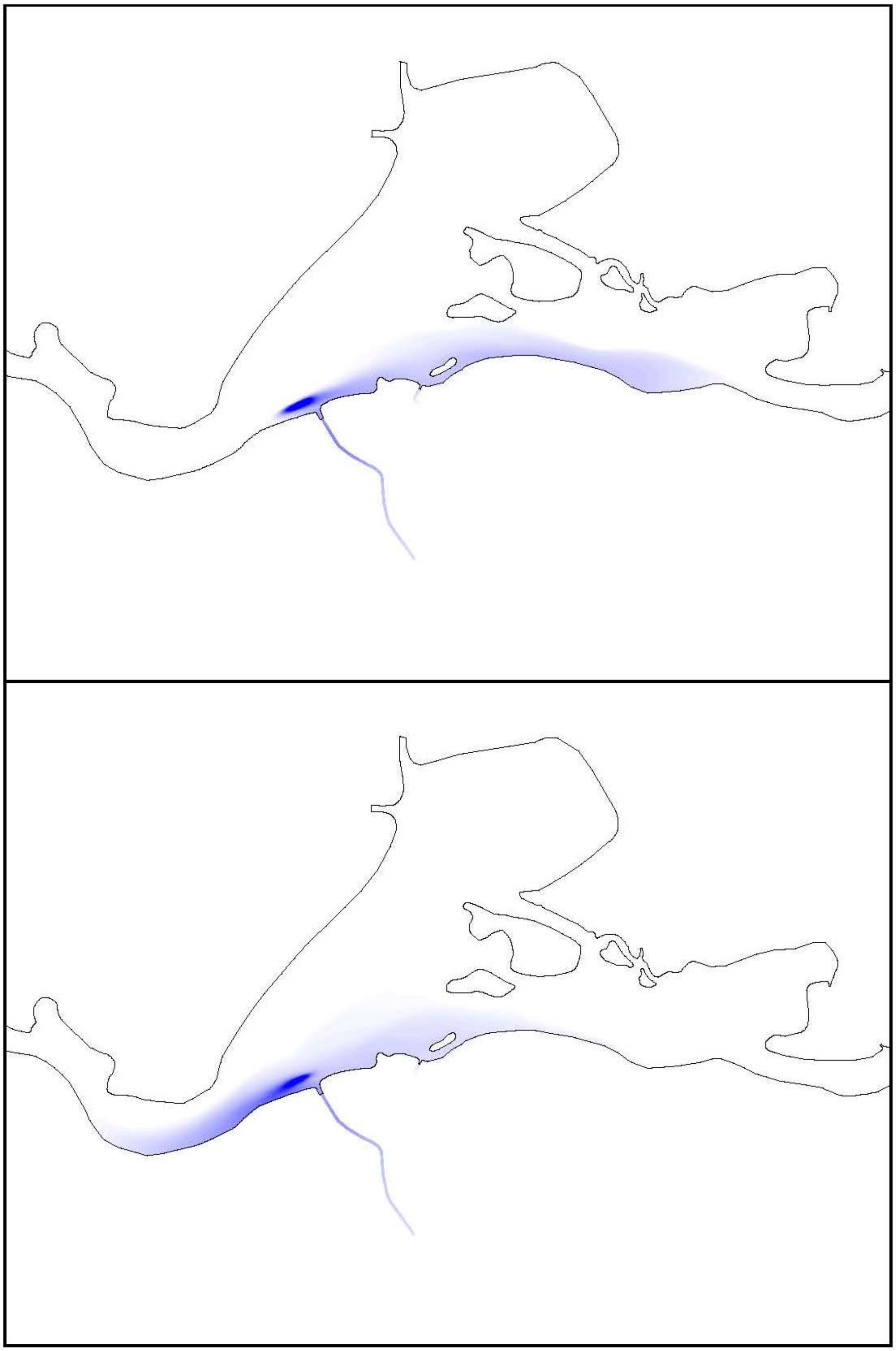

Figure 28. Color contours of tracer concentration at slack after flood tide (a), and slack after ebb tide (b). Color scale ranges from 200:1 dilution (darkest blue) to 500:1 dilution (lightest blue). (RMA 2000)

5. *"The Basin Plan cautions against use of mixing zone models in estuarine environments because it is difficult to estimate the effects of re-entrainment. Also, all discharge plume models are limited because they do not account for transport due to tidal currents."*

The comment from the Water Agencies is based on a misunderstanding of the applicability of CORMIX and the conservative estimation of plume dilution relative to the potential effects of re-entrainment of effluent reducing the effective dilution. As opposed to relying on a dye study which is done at one point in time, a calibrated simulation model can evaluate a broad range of conditions. Although CORMIX results delineate the effluent plume defining the edge of the mixing zone under steady-state conditions, the average of median speeds during ebb and flood tides over a simulation period representing low (10th percentile) net Delta outflows was applied.

For a sense of perspective, Figure 1 (excerpted from LWA 2011) and shown on the following page can be used to portray the spatial scales of tidal currents and the mixing zone:

- The diffuser is at an average depth of 24 feet.
- The 115-ft long diffuser is smaller than the dot in the figure identifying the diffuser's location in Suisun Bay.
- Near-field mixing is complete at a distance of 125 feet from the diffuser centerline.
- In the half-hour before or after slack tide, when currents are weakest and reversing direction, Suisun Bay water moves an average of 800 feet away from the diffuser, over six times the length of the mixing zone.
- The distance that Suisun Bay water moves away from the diffuser over a tidal cycle is typically over five miles—beyond the frame of Figure 1.

Three other lines of evidence can be used to estimate the effects of re-entrainment on dilution of the District's effluent:

- Because the tidal current does not return in exactly the opposite direction in Suisun Bay, additional dispersion of the remaining reversing plume occurs over each tidal cycle. Even during the hour surrounding slack tide, the plume's centerline typically will be offset by more than the length of the diffuser upon its return.
- A dye study[17] performed for the District in 1970 found dilutions around 200:1 in boils directly over the outfall, increasing several fold away from that point.
- The Bay-wide RMA-2 hydrodynamic model (described above in response to Issue # 4) used in 2000 to simulate extremely conservative conditions (slack tides and extreme low Delta outflow conditions) indicated dilution was greater than 200:1 near the outfall.

Re-entrainment of effluent diluted 200-fold would be a trivial factor in the dilution analysis, as confirmed by both the dye study and the hydrodynamic modeling.

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[17] Brown and Caldwell Consulting Engineers; *Report on Continuous Dye Release Study Suisun Bay Outfall* (August 1970), prepared for Central Contra Costa Sanitary District.

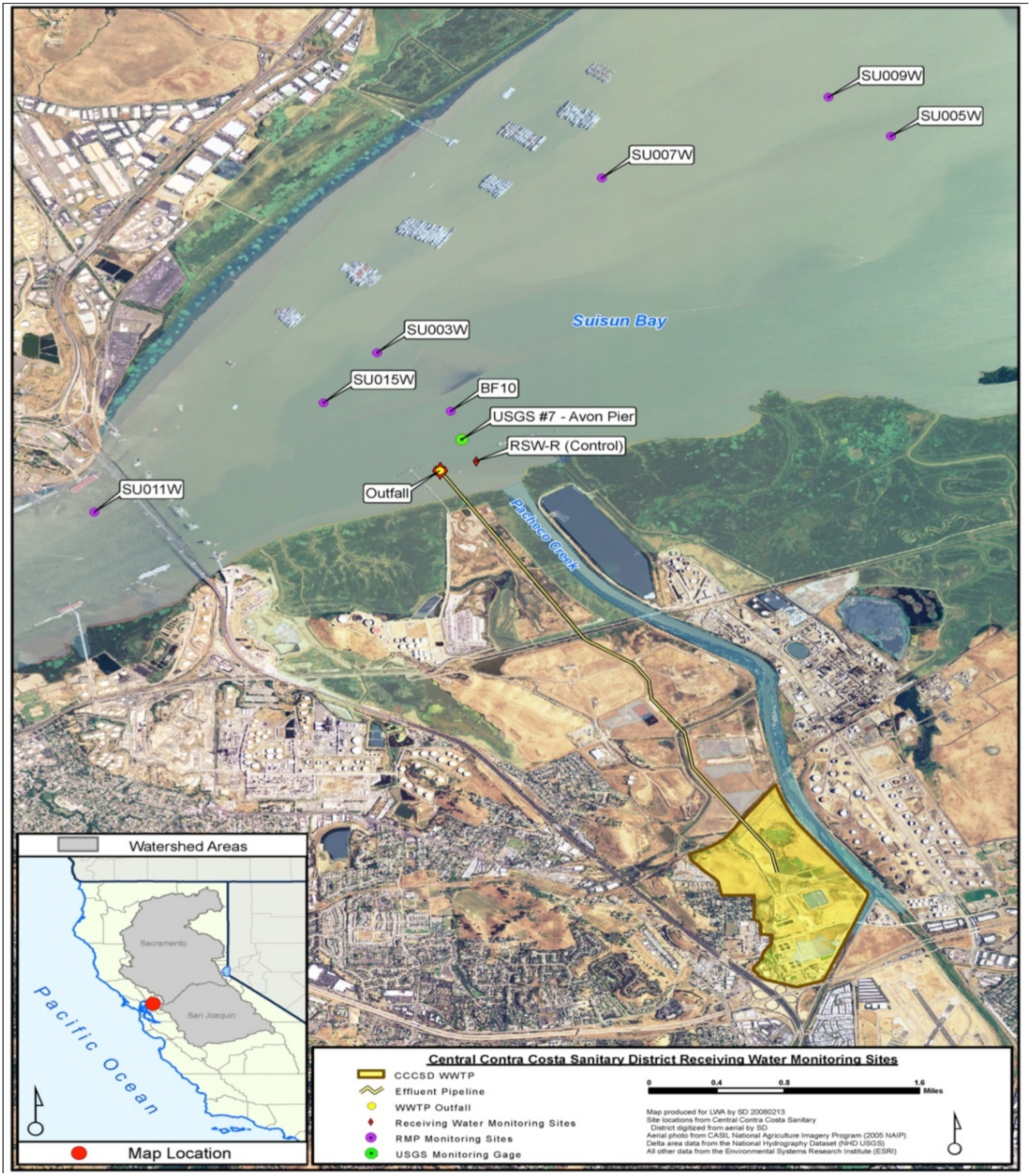

Figure 1. CCCSD outfall study area. Nearby monitoring stations are identified. (LWA 2011)

6. “The Basin Plan indicates that a conservative approach must be used when calculating effluent limits. Ignoring research on diatom inhibition and copepod toxicity due to ammonium is not a conservative approach.”

See the response provided to Statements No. 1 and 2 earlier in this memo. The issues cited are currently under study and are unresolved in the scientific community.

Statement No. 6 – The Regional Board’s analysis of Anti-degradation with regard to ammonia is contrary to established principles of law. [1-31-11 Water Agencies comment letter page 13 # III - B]

Response: This statement is not supported by State and federal regulations or policies, nor is it supported by the most recent State Water Board decisions or case law on this topic, all of which indicate that the Regional Water Board is not obligated to perform a new antidegradation analysis in the issuance of the Tentative Order or in the adoption of that order as a new NPDES permit.

The Tentative Order does not allow a lowering of water quality in comparison to the existing permit. The proposed permit does not increase the permitted discharge flow. The proposed permit places limits on ammonia that did not exist previously. Therefore, the permit is more restrictive than the previous permit with regard to ammonia discharges, and does not allow a lowering of water quality.

According to State Water Board guidance, where the Regional Board “has no reason to believe that existing water quality will be reduced due to the proposed action, no antidegradation analysis is required.”[18] According to U.S. Environmental Protection Agency (Region 9) guidance, “The first step in an antidegradation analysis is to determine whether or not the proposed action will lower water quality…If the action will not lower water quality, no further analysis is needed and EPA considers 40 CFR 131.12 to be satisfied.”[19]

The Water Agencies’ arguments are based on the incorrect premise that the baseline water quality used to determine whether the Tentative Order will result in a reduction in water quality is the wastewater treatment plant’s current discharge rather than the permitted discharge. While this position is supported by the State Water Board’s decision in *In re Matter of the Petition of Citizens for a Better Environment et al.*, Order No. 90-5 (Oct. 4, 1990) (“Order WQ 90-5”), subsequent authorities have declined to follow the State Water Board’s interpretation in Order WQ 90-5 and have concluded that the appropriate baseline is the permitted discharge. Order WQ 90-5 involved the issuance of NPDES permits to three sewage treatment plants that discharge to the San Francisco Bay south of the Dumbarton Bridge (“South Bay”). At the time the permits were issued, the South Bay had been listed as impaired for toxic pollutants but no Total Maximum Daily Loads (TMDLs) had been developed. The permits

[18] State Water Resources Control Board, *Antidegradation Policy Implementation for NPDES Permitting, Administrative Procedures Update 90-004* (July 2, 1990) at pg. 2.

[19] U.S. Environmental Protection Agency (Region 9), *Guidance on Implementing the Antidegradation Provisions of 40 CFR 131.12* (June 3, 1987) at pg. 3.

therefore contained interim performance –based effluent limits for toxic pollutants and interim mass loading limits for toxic pollutants that were 33-80% higher than actual loading. The petitioner argued that these interim limits violated state and federal antidegradation policies.

The State Board determined that issuance of the South Bay permits triggered the state and federal antidegradation policies because the permits allowed an increase in the volume of the discharges as well as an increase in the mass emissions of toxic pollutants over current levels. The Board concluded that because the interim limits for toxics were based on performance, they did not necessarily ensure protection of existing in-stream beneficial uses as required by the first part of the federal antidegradation policy. The Board further concluded that because the South Bay was impaired for toxic pollutants, the mass loading limits should be based on the dischargers' best performance since 1975 (the effective date of the federal antidegradation policy), i.e., their current performance as of 1990. The State Water Board directed the Regional Water Board to calculate the limits by "multiplying the 1989 annual mean effluent concentration by the 1985-1988 annual average flow."

However, in *Own Motion Review of the Petition of Communities for a Better Environment et al.*, Order No. 90-09 (Oct. 21, 1999) ("Order WQ 99-09"), which involved a challenge to the 1998 South Bay permits, the State Water Board used the permitted discharge, not the current discharge as the water quality baseline. The 1998 permits contained mass loading limits for copper and nickel based on the average flow data from 1985-1988 and average concentration data from 1989 as instructed by the State Board in WQ 90-5. Environmental groups asserted that the limits violated the antidegradation policies because they were much higher than the mass of copper and nickel actually discharged by the South Bay dischargers over the last five years.

The State Water Board upheld the copper and nickel mass limits in the 1998 permit despite the fact that current performance was better than in 1990. It found that the limits for nickel in the 1998 permit did not trigger antidegradation requirements because "the mass limits are unchanged from the 1993 permits. The 1998 permits do not authorize an increase in mass emissions over the 1993 permit limits."

Similarly, in *San Francisco Baykeeper v. State Water Resources Control Board,* 2003 Cal. App. Unpub. LEXIS 5290 (May 28, 2003), the court held that interim mass limits for mercury that are higher than the actual mass of mercury being discharged do not violate antidegradation policies. That case involved challenges to the City of Petaluma's and Fairfield Suisun Sanitary District's 1998 NPDES permits. When the permits were issued, the receiving waters were listed as impaired for mercury but a TMDL had not yet been developed. The permits contained interim mass limits for mercury based on treatment plant performance but higher than the actual mass of mercury being discharged due to the dischargers' reclamation programs. Environmental groups, relying on WQ Order 90-5, argued that the limits violate antidegradation policies because they exceed actual loading. The Court rejected the plaintiffs' arguments, concluding

> "State Board's interpretation of Tier 1 of the antidegradation policy evolved in the years following Order WQ 90-5, and we accord greater weight to its more recent construction of federal policy [Order WQ 99-09]. There is no contention

> here that the 1998 Petaluma and FSSD permits authorize an increase in the mass of mercury over mass allowed by the 1990 permits. We conclude that Regional Board was not obliged to set mass limits at the current mercury loading levels."

The State Water Board Guidance is consistent with the conclusions in Order WQ 99-09 and *San Francisco Baykeeper* that the appropriate baseline is the permitted discharge not the current discharge. The State Water Board Guidance states that baseline water quality is

> "…the best quality of the receiving water that has existed since 1968 when considering Resolution No. 68-16 [state antidegradation policy], or since 1975 under the federal policy, unless subsequent lowering was due to regulatory action consistent with State and federal antidegradation policies. If poorer water quality was permitted, *the most recent water quality resulting from the permitted action is the baseline water quality* to be considered in any antidegradation analysis. State Board Guidance at p. 4 (emphasis added)."

Third, the Tentative Order will not result in a reduction of water quality as compared to its current permit. The Tentative Order does not authorize an increase in the volume of the discharge. Both the current permit and the Tentative Order permit the District to discharge up to 53.8 mgd. The Regional Water Board approved an increase in the volume of the District's discharge from 45 mgd to 53.8 mgd without conditions ten years ago after the District conducted an antidegradation study, which concluded that such an increase was consistent with state and federal antidegradation studies. Nor does the Tentative Order authorize a substantial increase in the mass emission of a pollutant. To the contrary, the Tentative Order is more restrictive than the current permit with respect to ammonia. The District's current permit does not contain any limits on ammonia while the Tentative Order imposes effluent limitations on total ammonia. While the Water Agencies assert that the Tentative Order will reduce water quality because it will permit the District to discharge more ammonia than the District is currently discharging, as discussed above, current performance is not the correct baseline. Therefore, because the Tentative Order does not authorize an increase in the volume of the discharge or an increase in the mass emissions of a pollutant as compared to the current permit, the antidegradation policies are not triggered and the Regional Water Board was not required to conduct an antidegradation analysis or make any antidegradation findings.

Statement No. 7 – The Draft Permit Must Include Effluent Limitations for Residual Chlorine and Settleable Matter. [11-1-11 San Francisco BayKeeper comment letter page 1 # 1].

Response: Effluent limits for settleable matter and chlorine residual should not be included in the District's NPDES permit.

Effluent produced at the District's wastewater treatment plant and discharged to Suisun Bay receives primary and secondary treatment and is disinfected with ultraviolet light. There is no chlorine used and no chlorine residual produced with ultraviolet (UV) disinfection, so effluent limitations for chlorine residual at the District's facility are unnecessary.

The Regional Water Board adopted a Basin Plan Amendment on January 21, 2004 to update water quality objectives and NPDES permit implementation provisions.[20] The amendment clarified that effluent limitations for settleable matter do not apply to either secondary or advanced sewage treatment facilities and should not be included in future permits for those facilities. As stated in the staff report that accompanied the subject Basin Plan amendment,[21] "For secondary and advanced treatment systems, the equivalent limitation is suspended solids, pursuant to 40 CFR 133.102. Application of both suspended solids and settleable matter effluent limitations to secondary and advanced sewage treatment facilities is not only redundant, but also does not afford better protection of beneficial uses. Settleable matter is not a relevant indicator of adverse effects of secondary and advanced treated sewage on receiving waters. It is a technology-based effluent limit for only primary treatment and was mistakenly applied to secondary and advanced treatment plants." The NPDES permit issued to the District includes effluent limitations for total suspended solids as required to monitor performance of secondary treatment facilities.

Statement No. 8 – The Draft Permit Must Conduct a Complete Reasonable Potential Analysis that Fully Addresses Pharmaceuticals, Chemicals from Personal Care Products, and Sediment Toxicity. [11-1-11 San Francisco BayKeeper comment letter page 1 # 2].

Response: Information does not exist to allow the performance of a reasonable potential analysis for the parameters in question.

Pharmaceuticals and Personal Care Products

As prescribed in the SIP, the RPA process is applicable only to priority pollutant criteria and objectives established by the U.S. EPA and the State of California.[22] Under the RPA process, water quality based effluent limits must be implemented if effluent quality exceeds the criteria/objectives (Trigger 1) or the ambient receiving water quality exceeds the criteria/objectives and the pollutant was detected in the effluent (Trigger 2). Utilization of Trigger 3 (sometimes referred to as best professional judgment) to establish effluent limits must be supported by scientific facts (e.g., presence in effluent and receiving waters) and evidence of impacts to beneficial uses. However, because no criteria or objectives have been adopted for pharmaceuticals and personal care products (PPCPs), none of these evaluations can be conducted. Trigger 3 has not been utilized for PPCPs because water quality thresholds have not been established, there is no process for determining if these constituents are impacting beneficial uses at ambient levels, and no monitoring has been completed to link levels in effluent to levels of concern in the ecosystem.

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[20] Order No. R2-2004-0003.

[21] San Francisco Bay Regional Water Quality Control Board, *Staff Report – Proposed Amendment to the Water Quality Control Plan for the San Francisco Bay Basin (Basin Plan) Updating Water Quality Objectives and Implementation Language* (December 19, 2003), page 32.

[22] SWRCB (2005), page 3.

The most recent recommendations for PPCP monitoring were developed by a Blue Ribbon Panel of scientific advisors convened by the State Water Resources Control Board in 2009. The panel was convened by the State Water Board to "provide guidance for developing monitoring programs that assess potential Constituents of Emerging Concern (CEC) threats from various water recycling practices, including indirect potable reuse via surface spreading; indirect potable reuse via subsurface injection into a drinking water aquifer; and urban landscape irrigation." [23] The panel's monitoring recommendations were released in 2010 and included a discrete list of surrogate compounds and performance based indicators related to human health exposures during recycled water applications. Additional research was suggested by the panel to develop and validate analytical methods for those compounds, prioritize compounds for monitoring, and determine environmental impacts under different exposures to recycled water or wastewater. At this point in time, there is no process or thresholds for identifying constituents of concern or for assessing impacts to aquatic ecosystems.

Sediment Toxicity

A Statewide plan for assessing sediment toxicity and its impacts on water quality was adopted by the State Water Resources Control Board in 2008.[24] The U.S. Environmental Protection Agency approved the plan on August 25, 2009, allowing full implementation in California. Part 1 of the Water Quality Control Plan pertains only to sub-tidal, surficial sediments of enclosed bays and estuaries. Applied to the San Francisco Bay Estuary, the assessment procedures are clearly defined only for polyhaline conditions (i.e., Central San Francisco Bay, portions of San Pablo Bay), not for the mesohaline (moderately saline, South Bay/Suisun Bay) or oligohaline (freshwater) areas.

Part 1 establishes narrative sediment quality objectives (SQOs) to protect aquatic life and human health in sediments. The approved plan includes only the process for evaluating compliance with the aquatic life SQO (Phase I). The aquatic life SQO is implemented using a Multiple Lines of Evidence Approach (MLOE) that integrates three types of measurement/assessment tools. The lines of evidence include (1) sediment toxicity (laboratory exposure of invertebrates to surficial sediments), (2) benthic community condition (measure of composition, abundance, diversity of aquatic species inhabiting the surficial sediments), and (3) sediment chemistry (measurement of the concentration of chemicals of concern in surficial sediments). The human health SQO is currently implemented on a case-by-case basis by the Regional Water Boards, based on monitoring and assessments performed by various state agencies.

Integrating lines of evidence following the SQO approach will yield an assessment level that may range from "unimpacted" to "clearly impacted" for each monitored site. A finding of "unimpacted" and "likely unimpacted" will be considered in compliance with receiving water limits. The findings of "likely impacted" or "clearly impacted" will be considered degraded when evaluating waters for placement on

[23] *Recommendations of a Science Advisory Panel, Monitoring for Chemicals of Emerging Concern (CECs) in Recycled Water* (June 25, 2010). Panel convened by the State Water Resources Control Board.

[24] State Water Resources Control Board, *Water Quality Control Plan for Enclosed Bays and Estuaries - Part 1 Sediment Quality* (September 16, 2008). Adopted as Resolution No. 2008-0070.

the Section 303(d) list and evaluating compliance with the aquatic life SQO. A finding of "possibly impacted" requires additional study to determine if a site is truly degraded.

Part I was amended in April 2011 to add a narrative SQO that protects resident finfish and wildlife from the detrimental effects caused by exposure to pollutants in sediments, a process for implementing the narrative objectives, new definitions in the glossary in support of the proposed narrative objectives, and corrections for omissions and typographical errors. State Water Board staff is currently drafting policy to implement Phase II amendments. Phase II will address the methodologies needed to interpret and implement SQOs to protect benthic communities from direct exposure in mesohaline areas of San Francisco Bay and the freshwater Delta, and protect human health from consumption of fish and shellfish.

If a Regional Water Board determines that a discharge of toxic pollutants has the potential to cause or contribute to an exceedance of the SQOs, the objectives may be applied as receiving water limits in NPDES permits. Effluent limits will only be developed if specific pollutant(s) causing degradation have been identified, if a discharge is clearly linked to those identified pollutants, and if an estimated reduction in pollutant loading from the discharge in question will improve sediment quality. In the San Francisco Bay Area, the Regional Water Board is in the process of compiling information on sediment quality and the health of sediment biota. As a result, wastewater dischargers are required to participate in the Regional Monitoring Program (RMP) to obtain further information on sediment quality and benthic populations. As this information is acquired and assessed, the Regional Water Board will decide if receiving water limits are needed in NPDES permits or if the RMP should conduct stressor identifications in specific areas of the Bay. Stressor identifications involve tests to confirm pollutant related impacts, identify specific pollutants, and (as needed) identify the pollutant sources. Sites designated as "clearly impacted" or "likely impacted" are the highest priority for stressor identification studies in accordance with the policy adopted by the State Water Board.

The requirements and findings in the District's Tentative Order are consistent with the SQO policy and the current state of the science in San Francisco Bay. The multiple lines of evidence approach for assessing compliance with SQOs was first applied to samples collected in 2008. However, these results have not yet been peer-reviewed or released for regulatory decision-making. In addition, and as mentioned above, the SQO techniques are to-date applicable only to polyhaline conditions, not the mesohaline conditions that exist in Suisun Bay. Until SQO assessments are conducted on samples collected in Suisun Bay and it is determined if sediments are impacted, specific monitoring by a discharger is not required. Continued participation by the District in the RMP is the appropriate approach for determining SQO compliance and addressing concerns regarding sediment toxicity in Suisun Bay.

Appendices:

Appendix A: A Critical Review of "Full Life-Cycle Bioassay Approach to Assess Chronic Exposure of *Pseudodiaptomus forbesi* to Ammonia/Ammonium - Final Report. Teh et al., August 31, 2011." Prepared by Pacific Ecorisk, Inc., December, 2011.

Appendix B: Phytoplankton (chlorophyll a) versus ammonium concentrations in Suisun Bay (1977-2010).

Appendix A

FINDINGS REPORT

From A Critical Review of:

Full Life-Cycle Bioassay Approach to Assess Chronic Exposure of *Pseudodiaptomus forbesi* to Ammonia/Ammonium - Final Report

Dated August 31, 2011

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Submittal Date:
December 26, 2011

1. INTRODUCTION

On behalf of the Central Contra Costa Sanitary District (CCCSD), Larry Walker Associates has contracted Pacific EcoRisk, Inc. (PER) to perform a critical review of the "Final Report: *Full Life-Cycle Bioassay Approach to Assess Chronic Exposure of Pseudodiaptomus forbesi to Ammonia/Ammonium*" authored by Teh S, Flores I, Kawaguchi M, Lesmeister S, and Teh C (dated August 31, 2011).). As requested by CCCSD, the primary focus of this review were the experiments described as Subtasks 3-3 and 3-4-1 in the Teh *et al*. report. Additional comments on study methodology and data analysis were developed and can be provided to interested parties on request as evidence that additional study is needed.

2. COMMENTS ON SUB-TASK 3-3 (CHRONIC [31-DAY] LIFE CYCLE TOXICITY TESTING)

Comment #1. Teh *et al*.'s analysis of the number of nauplii and number of juveniles produced during the chronic (31-day) exposure is believed to be flawed at a very fundamental level. It is apparent in Teh et al.'s derivation of 'mean number of nauplii, juveniles, and adult *P. forbesi* produced per female' (in Teh *et al*.'s Table 11) and in the 'sum total number of nauplii, juvenile, and adult *P. forbesi* produced' (in Teh *et al*.'s Appendix III table) that they summed the counts of nauplii and juveniles that were counted on the progressive 2-3 day intervals (the raw data for these counts were provided in Teh *et al*.'s Appendix I) as if each new progressive count was of new individuals that had not been counted on the previous count day. So when 17 nauplii were counted in Control replicate A on Day 5 of the test, and 20 nauplii were counted on Day 7, and 17 were counted on Day 10, and so on, Teh *et al*. summed these up as if they were different nauplii that had been produced during the progressive 'count days'.

This would be correct had the nauplii and juveniles that were counted on each 'count day' been removed from the original replicate container and transferred to a new replicate container such that any nauplii or juveniles observed and counted in the original replicate containers on subsequent days would have been new organisms separate and distinct from the organisms that had been counted during the previous count day(s). Note that this approach would have created a logistical challenge, with a doubling of the number of experimental replicate beakers on Day 3 of the test (going from the original n=20 to n=40), a tripling of the beakers on Day 5 (n=60), a quadrupling of beakers on Day 7 (n=80), and so on and so on. This would then be compounded as nauplii that had transformed into juveniles would again need to be transferred to new replicates so as to allow observation of new juveniles produced by the remaining nauplii. The number of necessary beakers rapidly becomes logistically improbable.

However, it is not believed that this is what happened. Unfortunately, their report's inadequate description of test methodology is not explicit on this. However, it can be deduced from the nature of the study that the neonates were left in place in each replicate, as these were the source of the subsequent juveniles, which were similarly left in place to serve as the source for the subsequent adults. This was confirmed by inquiry made with one of the other authors of the report (M Kawaguchi, pers. comm.). As a result, when 20 nauplii were counted in Control replicate A on Day 7, some (if not most) of these organism were the very same organisms that had been counted on the earlier Day 5 count, and the nauplii that were counted on Day 10 were some of the same as had been counted on Days 7 and Day 5.

This conclusion is also supported by the following observations made for closely-related congener *Pseudodiaptomus annandalei* (Golez et al. 2004):

1. hatching of the first brood of nauplii occurs within 24-hrs of spawning;
2. females produced new ovisacs at ~ 1 day intervals, again with hatching occurring within that 24-hrs;
3. "females that were isolated from males produced only two clutches of viable eggs". Additional ovisacs were produced (making it appear that the female is reproductive), but the "succeeding clutches of eggs were aborted or shed off within 48 hrs and never hatched out".

Of course, the reproductive biology of *P. forbesi* may differ from that of the congener *P. annandalei*; however, in the absence of contradictory empirical evidence, Occam's razor would dictate otherwise.

We are left to conclude that **<u>Teh *et al*.'s reported results for 'total number' and 'mean number per female' for the nauplii and juveniles are incorrect, and that their analyses of that data are similarly incorrect</u>**.

Interestingly, in Teh *et al*.'s analyses of the 'total number' and 'mean number per female' of adults produced during the study, the number of adults counted on each progressive 'count day' were **NOT** summed in similar fashion, with Teh *et al*. instead evaluating on the count data from a single 'count day' (Day 31).

<u>Comment #2</u>. While it is believed that Teh *et al*.'s count data are incorrect, let us assume for a moment that they are in fact correct. The organism counts using Teh *et al*.'s summation method are summarized in Table 1 below. When their juvenile count data are analyzed using CETIS (a statistical software specifically designed to analyze aquatic toxicity data), the NOEC and LOEC are shown to be 0.79 mg/L TAN and 1.62 mg/L TAN (Table 2 below), NOT the lower concentrations reported by Teh *et al*.

It should noted that CETIS is the statistical software most commonly used by toxicity testing labs to analyze toxicity test data, and is believed to be the statistical software used at the UC Davis Aquatic Toxicology Lab; indeed, Teh *et al*. used CETIS to analyze their Subtask 3-4-1 and Subtask 3-4-2 experimental data as evidenced in Appendices IV and V of their report.

It should also be noted that our assessment of problems with Teh *et al*.'s statistical analyses should not be interpreted as indicating that there was no effect resulting from the ammonia, but simply that the experimental data do not support any differences that were observed as being statistically significant.

Table 1. Production on *Pseudodiaptomus forbesi* nauplii, juveniles, and adults (from Appendix I in Teh *et al*. report)

Test Treatment (mg/L TAN)	Test Replicate	Total # of *Pseudodiaptomus forbesi* Life Stage Counted			
		Nauplii[A]	Juveniles[A]	Adults[A] (counts made only on Day 31)	Adults[B] (counts made as for nauplii & juveniles)
Control	A	86	38	11	93
	B	100	73	26	178
	C	68	45	7	122
	D	75	52	3	52
0.36	A	60	27	0	1
	B	62	57	3	36
	C	83	79	18	167
	D	71	43	7	77
0.79	A	24	48	10	77
	B	64	31	4	45
	C	41	17	1	17
	D	52	22	8	77
1.62	A	47	1	0	0
	B	32	0	0	0
	C	46	14	5	28
	D	54	23	19	108
3.23	A	15	1	1	4
	B	39	1	1	6
	C	42	18	13	83
	D	30	13	5	34

A - For the nauplii and juveniles, Teh et al. summed the progressive counts on successive days as separate individuals; as explained in our review, this is believed to be erroneous, and is inconsistent with the counts of the "produced" adults which consist of the number of adults that were alive on Day 31 of the test.

B - Counts of "produced" adults using the summation of the progressive counts on successive days as separate individuals (as used by Teh et al. for the nauplii and juveniles); as explained in our review, this is believed to be erroneous.

Table 2. Comparative analyses of juvenile and adult production in the 31-day test (from CETIS analysis of juvenile data using Teh et al. summation method)				
Statistical Endpoint	Juveniles		Adults	
	Teh et al. Analyses	CETIS Analyses	Teh et al. Analyses	CETIS Analyses
NOEC =	0.36 mg/L TAN	0.79 mg/L TAN	**<0.36 mg/L TAN**	**3.23 mg/L TAN**
LOEC =	0.79 mg/L TAN	1.62 mg/L TAN	**0.36 mg/L TAN**	**>3.23 mg/L TAN**
Chronic Value =	1.13 mg/L TAN	1.13 mg/L TAN	<0.36 mg/L TAN	>3.23 mg/L TAN

Chronic Value = geometric mean of NOEC and LOEC.

Comment #3. Teh *et al*.'s apparently erroneous statistical analysis of the adult data is even more significant (Table 2). Teh *et al*. reported that the NOEC and LOEC for adults were <0.36 mg/L TAN and 0.36 mg/L TAN, respectively. However, their inter-replicate variability for that endpoint is so high (CVs ranged from 70% to 150%) that even qualitative evaluation suggests otherwise. CETIS analysis indicates that the NOEC and LOEC are 3.23 mg/L TAN and >3.23 mg/L TAN.

Again, it should be noted that our assessment of problems with Teh *et al*.'s statistical analyses should not be interpreted as indicating that there was no effect resulting from the ammonia, but simply that the experimental data do not support any differences that were observed as being statistically significant. Certainly, the NOECs and LOECs resulting from this experiment should not be considered suitable for use in a regulatory framework.

3. COMMENTS ON SUBTASK 3-4-1 (EFFECTS OF AMMONIA ON NAUPLII PRODUCTION OVER 3 DAYS)

Comment #4. In this test, Teh *et al*. exposed individual gravid female copepods to TAN concentrations of 0 (control treatment), 0.38, and 0.79 mg/L for 3 days after which the number of nauplii produced were counted. The results of this test have been summarized in the Table 3 below.

From data reported in Teh et al.'s Table 12 and Appendix V:

Table 3. Effects of ammonia on *P. forbesi* production of nauplii over 3 days (Teh *et al*. Subtask 3-4-1).

TAN Concentration (mg/L)	Mean # of Nauplii per Female
Control	7.6
0.38	5.5
0.79	5.4

The results from this test are somewhat troubling in that, while technically monotonically increasing as the ammonia concentration increases, no apparent concentration-response relationship is observed between the 0.38 mg/L treatment and the 0.79 mg/L treatment. One would expect that as the TAN concentration increases from 0.38 mg/L (a presumably toxic concentration) to 0.79 mg/L (a two-fold greater concentration), there should be an increase in the toxic response – this is a fundamental paradigm of toxicology.
We have already seen in the data evaluations presented above that there is variability in toxic responses made by these organisms. Indeed, in some cases, the variability has been so extreme as to preclude a meaningful statistical analysis (as in the case of the adult data from the 31-day test). The absence of the expected concentration-response in the current test (Table 3) suggests that variability in organism response is occurring (the CV was 48% in the 0.38 mg/L treatment) such that the treatment means may be deviating from the true population mean (in statistical terms, this is referred to as a "false positive" or a "false negative").

In the present case, it is impossible to determine which of the two test responses is deviating most from the true population mean response. However, it is worth noting that:

1. there were two replicates at the 0.38 mg/L treatment that had 10 nauplii (the highest number observed in ANY replicate) whereas there was only one replicate at the control treatment that had 10 nauplii, and
2. the CV at the 0.38 mg/L treatment was 48%, which was markedly higher than at the Control or 0.78 mg/L treatment.

This is suggestive that the variability at the 0.38 mg/L treatment was elevated and may have resulted in a false positive, such that the observed mean response of 5.5 nauplii per female was lower than the true population mean. If correct, then the conclusion(s) drawn from the test data may not reflect true conditions, and the true LOEC could be 0.79 mg/L, and not 0.38 mg/L. At a minimum, the absence of the expected concentration-response should cast enough uncertainty on the test results as to make them inappropriate for regulatory decision-making.

Comment #5. It is fortunate that multiple sets of test data from the study allow comparison of results between tests; for instance, the results of Subtask 3-4-1 can be compared to those generated in the earlier Subtask 3-3 (31-day) test in which gravid females were exposed to varying concentrations of TAN and counts of nauplii produced after 3 days were counted, but were also counted after 5 days and 7 days (recall that counts made on progressive count days are not believed to be all new organisms). The Subtask 3-3 data are summarized in Table 4 below, along with the data from Task 3-4-1.

If one were to "cherry-pick" the Day 3 data and exclude the additional data, then Teh *et al*.'s conclusion for the Subtask 3-4-1 might stand. However, by extending the observation period beyond 3 days, it becomes evident that not only is there no reduction in nauplii production at 0.36 mg/L TAN, but nauplii production actually appears to be *increased* relative to the control treatment (the maximum mean # of nauplii on Day 5 at the 0.36 mg/L TAN treatment is **31% greater** than the highest mean # of nauplii produced in the Control treatment on any of the count days). Furthermore, CETIS analysis indicates that there were no statistically significant reductions in nauplii production at the 0.36 mg/L (Table 5). Even if we use the count summation used by Teh et al., by extending the counts beyond 3 days, it becomes apparent that there is no statistically significant difference between the response at 0.36 mg/L TAN and the Control treatment. This certainly creates a very significant uncertainty over the results of the Subtask 3-4-1 test of the effects of ammonia on nauplii production over 3 days.

It could be argued that this phenomenon is the result of ammonia having caused a delay in egg hatching, and the 31-day data are certainly suggestive of that. However, the only way to address that would have been to have some information from the scientific literature on the egg gestation period for this species, coupled with testing being performed under the current test conditions using females with egg sacs of the same age.

Table 4. Effects of ammonia on *P. forbesi* nauplii produced over 3 and 5 days.

Teh *et al.* Study Task	TAN Treatment (mg/L)	Mean Number of Nauplii per Female		
		Day 3	Day 5	Sum through Day 5 (Day 3 + Day 5)[A]
Subtask 3-4-1	Control	**7.6**	not counted	not counted
	0.38	**5.5**	not counted	not counted
	0.79	**5.4**	not counted	not counted
Subtask 3-3	Control-A	5.67	6.67	12.33
	Control-B	6.67	6.67	13.33
	Control-C	5	5	10
	Control-D	5	5	10
	treatment mean	**5.6**	**5.8**	**11.4**
	0.36-A	3	5	8
	0.36-B	2.33	8.33	10.67
	0.36-C	3.33	8.33	11.67
	0.36-D	3.33	3.33	6.67
	treatment mean	**3.0**	**6.3**	**9.3**
	0.79-A	0.33	1.67	2
	0.79-B	6.67	3.33	10
	0.79-C	2.67	2.67	5.33
	0.79-D	6.67	4	10.67
	treatment mean	**4.1**	**2.9**	**7.0**

A – These counts are made using method of Teh *et al.*, which assumes that the progressive counts on successive days are separate individuals; as explained in our review, this is believed to be erroneous.

Table 5. Comparison of nauplii production test results (all results expressed as mg/L TAN) (from CETIS analysis of data)

Statistical Endpoint	**Subtask 3-4-1**	**Subtask 3-3**				
	Day 3	Day 3	Day 5	Day 3 + Day 5[A]	Total (31 days)[A]	Total (31 days)[B]
NOEC =	<0.38	3.23	0.36	0.36	0.36	0.79
LOEC =	0.38	>3.23	0.79	0.79	0.79	1.62
Chronic Value =	<0.38	>3.23	0.53	0.53	0.53	1.13

Chronic Value = geometric mean of NOEC and LOEC.

A – These counts are made using method of Teh *et al.*, which assumes that the progressive counts on successive days are separate individuals; as explained in PER's review, this is believed to be erroneous.

B – These counts are made using what is believed to be the best remaining method: identifying the maximum number of nauplii observed on any given day for each replicate (this assumes that the individuals were left in the replicate beakers and were counted again and again on progressive days [i.e. repeated measures]).

4. FINAL COMMENT

The reviewer is troubled by the absence of any discussion by Teh et al. regarding the variability in their test response data, either between tests or within tests (i.e., inter-replicate variability). Without such acknowledgement, it is left for the non-scientist to assume that the data as presented are definitive. Moreover, it raises the question of whether the data from this study are adequate (or ‘ready’) for use in regulatory decision-making. However, it is important to note that this critical review is not intended to negate Teh *et al*.’s general observations that ammonia is toxic to naupliar, juvenile, and/or adult *P. forbesi* at elevated concentrations and that this toxicity is strongly influenced by pH. Indeed, the primary question of ‘what are the effects of ammonia on *P. forbesi*’ is relevant and Teh *et al*.’s study results certainly compel a more thorough examination of this. However, the problems associated with Teh et al.’s experimental methodology for Subtasks 3-3 and 3-4-1 and significant questions regarding the analysis of the resulting data do indicate that the quality of the work should preclude the resulting ‘critical threshold’ data (i.e., NOECs, LOECs, and point estimates [e.g., ECx, LCx, and ICx values]) from being used for regulatory purposes.

References Cited:

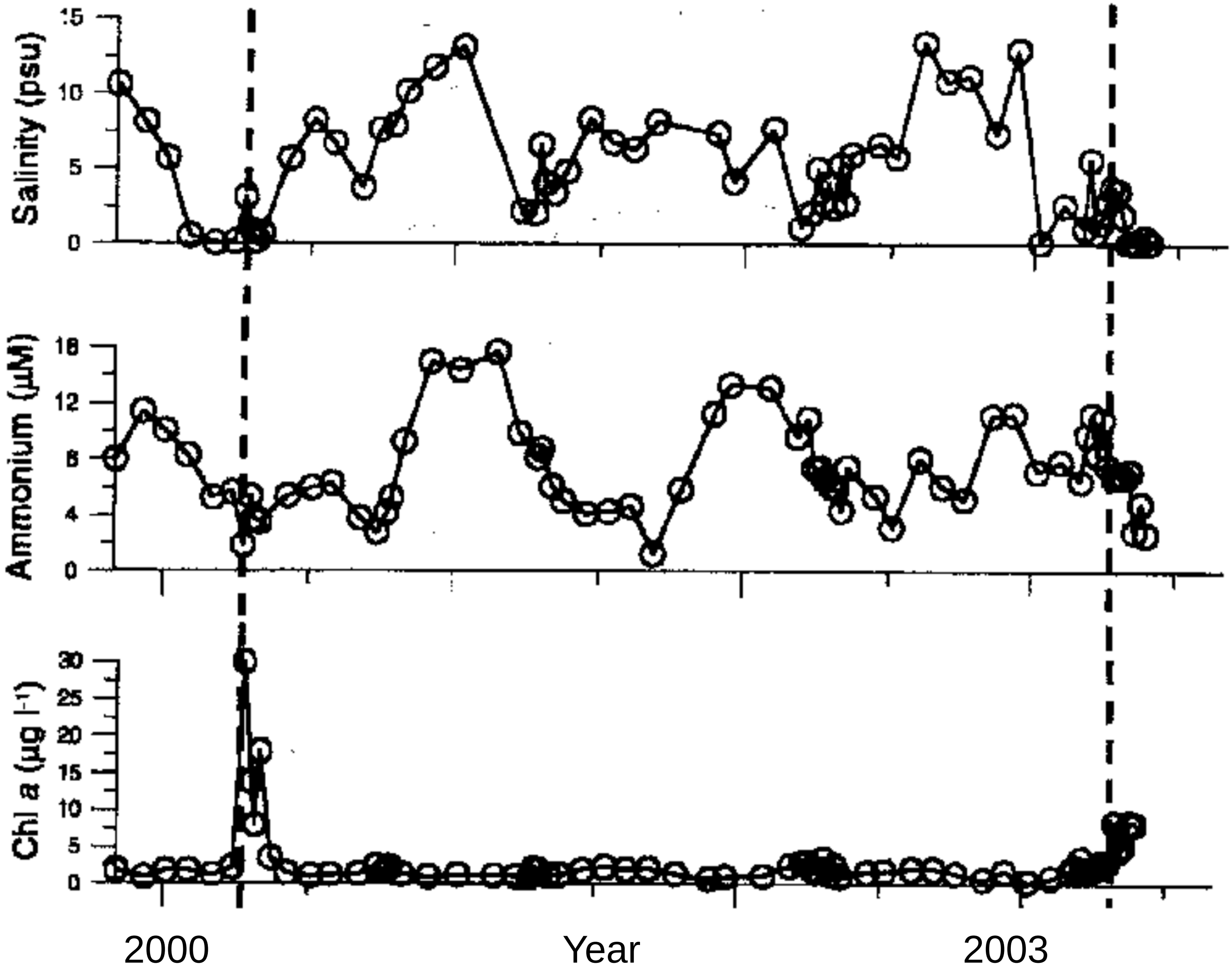
Golez MSN, Takahashi T, Ishimaru T, Ohnoa A (2004) Post-embryonic development and reproduction of *Diaptomus annandalei* (Copepoda: Calanoida). Plankton Biology & Ecology 51(1):15-25.

Appendix B

Phytoplankton (Chlorophyll a) versus Ammonium Concentrations in Suisun Bay [1977-2010]

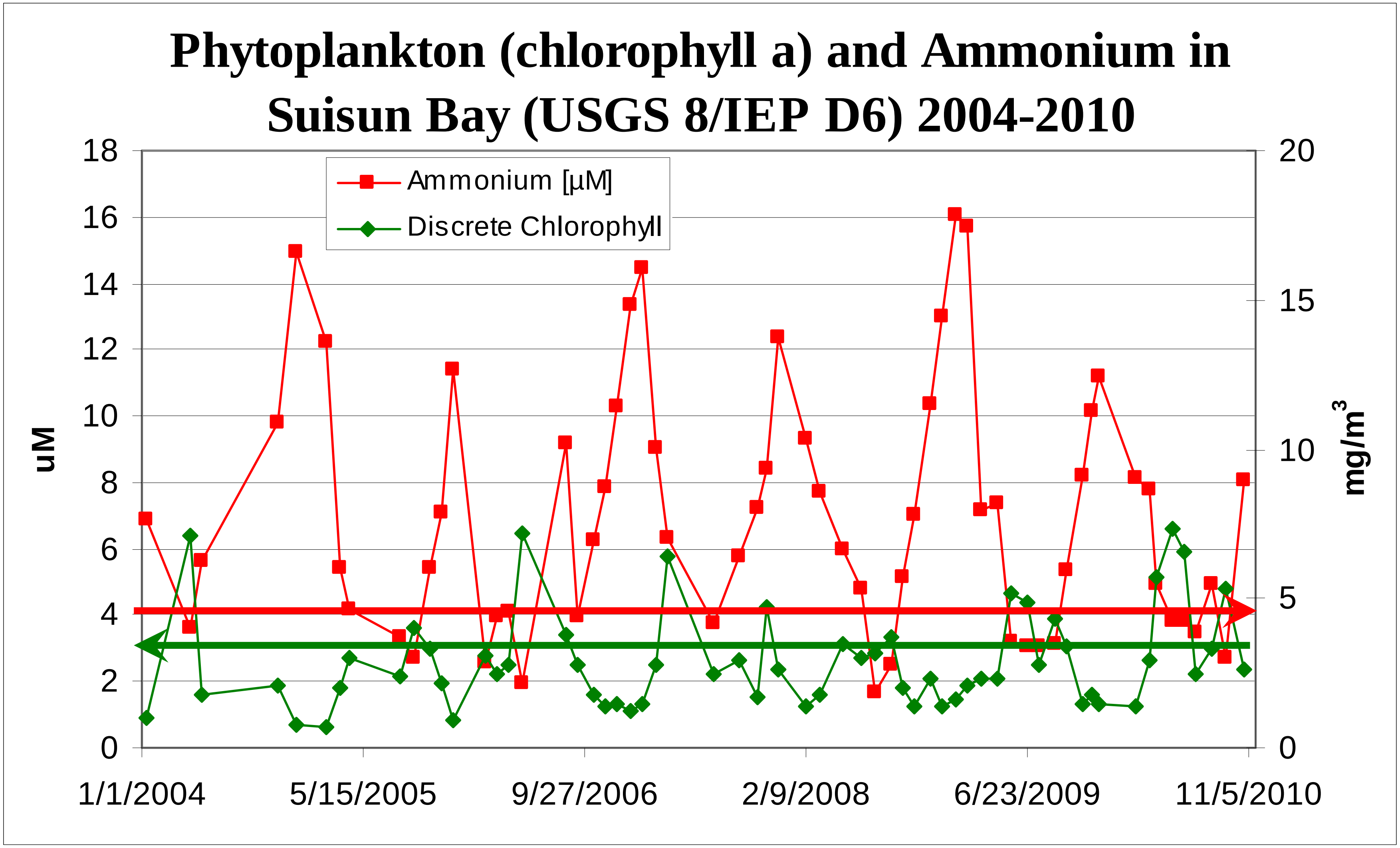
Dr. Dugdale et al., hypothesis that ammonia-N has to drop below 4 μ moles for phytoplankton bloom to occur is not fully supported by ambient data.

(1) In four years of monthly monitoring in Suisun Bay (November 1999–August 2003) by Dr. Dugdale et al., ammonia-N dropped below 4 μ moles 5 times, yet bloom occurred only one time.



Salinity (psu)
15
10
5
0
Ammonium (µM)
16
12
8
4
0
Chl a (µg l⁻¹)
30
25
20
15
10
5
0
2000
Year
2003

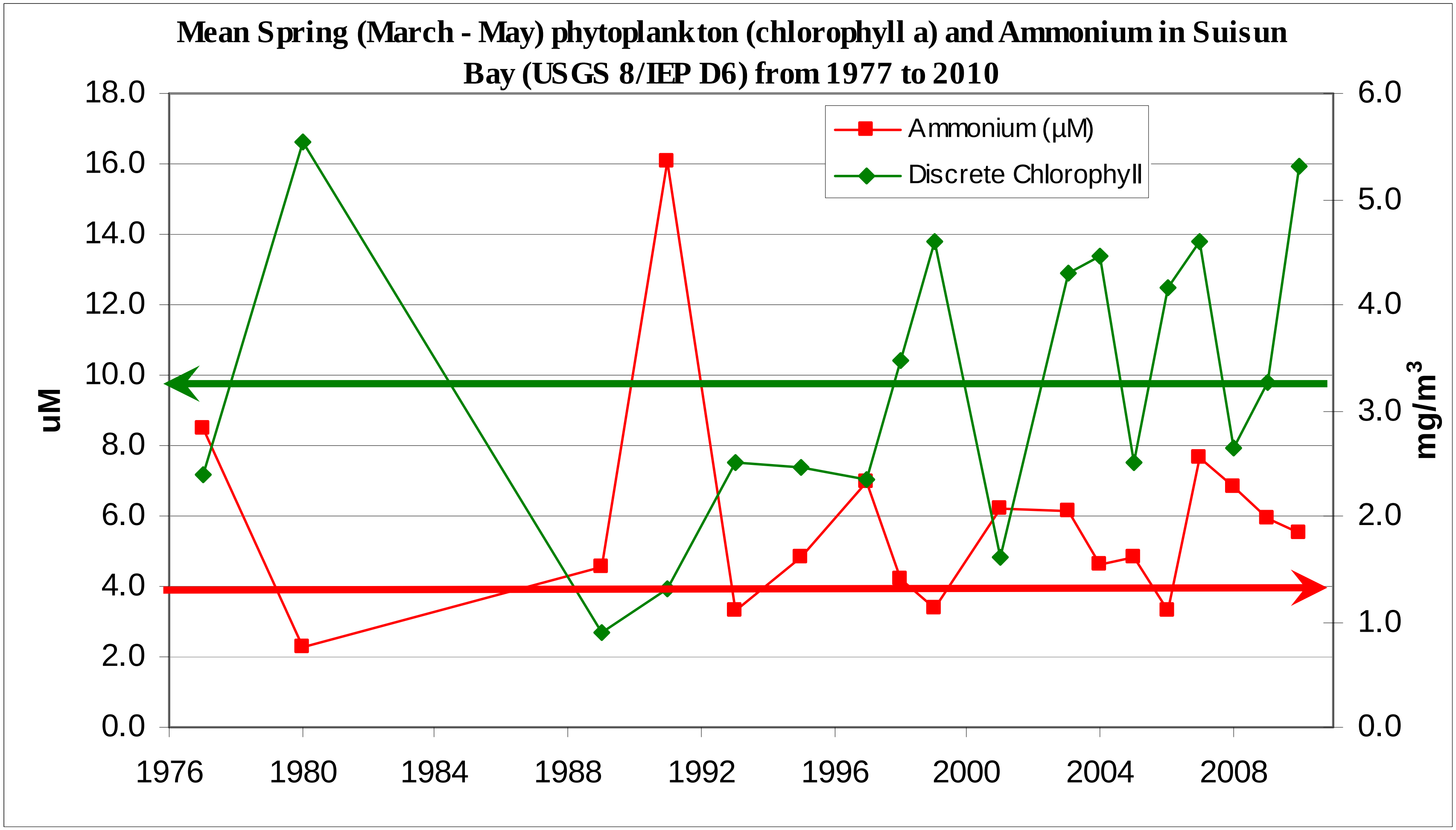
(2) From 2004 to 2010 ammonia dropped below 4 μ moles 20 times, yet there were only 5 blooms.



Phytoplankton (chlorophyll a) and Ammonium in Suisun Bay (USGS 8/IEP D6) 2004-2010
Ammonium [µM]
Discrete Chlorophyll
uM
mg/m3
0
2
4
6
8
10
12
14
16
18
0
5
10
15
20
1/1/2004
5/15/2005
9/27/2006
2/9/2008
6/23/2009
11/5/2010

(3) From 1977 to 2010, the spring mean chlorophyll reached above average only 3 out of 9 times (33%) when ammonia was below 4 μ moles.

This suggests that there are other factors more important than ammonia for increased chlorophyll production.



Mean Spring (March - May) phytoplankton (chlorophyll a) and Ammonium in Suisun Bay (USGS 8/IEP D6) from 1977 to 2010
Ammonium (µM)
Discrete Chlorophyll
uM
mg/m3
18.0
16.0
14.0
12.0
10.0
8.0
6.0
4.0
2.0
0.0
6.0
5.0
4.0
3.0
2.0
1.0
0.0
1976
1980
1984
1988
1992
1996
2000
2004
2008

EXHIBIT D

San Luis & Delta-Mendota Water Authority

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October 31, 2011

Mr. Vince Christian
California Regional Water Quality Control Board
San Francisco Bay Region
1515 Clay Street, Suite 1400
Oakland, CA 94612

Dear Mr. Christian:

Re: Comments on Tentative Order No. R2-2011-XXXX (NPDES No. CA0037648) for the Central Contra Costa Sanitary District Wastewater Treatment Plant

The San Luis & Delta-Mendota Water Authority ("Authority")[1] and the State Water Contractors, Inc. ("SWC")[2] (collectively, "Public Water Agencies") respectfully submit the attached

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[1] The Authority is a joint powers authority, established under California's Joint Exercise of Powers Act. (Gov. Code, § 6500 et seq.). The Authority is comprised of 29 member agencies, 27 of which hold contractual rights to water from the federal Central Valley Project (CVP). The Authority member agencies have historically received up to 3,100,000 acre-feet annually of CVP water for the irrigation of highly productive farm land primarily along the San Joaquin Valley's Westside, for municipal and industrial uses, including within California' Silicon Valley, and for publicly and privately managed wetlands situated in the Pacific Flyway. The areas served by the Authority's member agencies span portions of seven counties encompassing about 3,300 square miles, an area roughly the size of Rhode Island and Delaware combined. The Authority's members are: Banta-Carbona Irrigation District; Broadview Water District; Byron Bethany Irrigation District (CVPSA); Central California Irrigation District; City of Tracy; Columbia Canal Company (a Friend); Del Puerto Water District; Eagle Field Water District; Firebaugh Canal Water District; Fresno Slough Water District; Grassland Water District; Henry Miller Reclamation District #2131; James Irrigation District; Laguna Water District; Mercy Springs Water District; Oro Loma Water District; Pacheco Water District; Pajaro Valley Water Management Agency; Panoche Water District; Patterson Irrigation District; Pleasant Valley Water District; Reclamation District 1606; San Benito County Water District; San Luis Water District; Santa Clara Valley Water District; Tranquillity Irrigation District; Turner Island Water District; West Side Irrigation District; West Stanislaus Irrigation District; Westlands Water District.

[2] The SWC organization is a nonprofit mutual benefit corporation that represents and protects the common interests of its 27 member public agencies in the vital water supplies provided by California's State Water Project ("SWP"). Each of the member agencies of the State Contractors holds a contract with the California Department of Water Resources ("DWR") to receive water supplies from the SWP. Collectively, the SWC members deliver water to more than 25 million residents throughout the state and more than 750,000 acres of agricultural lands. SWP water is served from the San Francisco Bay Area, to the San Joaquin Valley and the Central Coast, to Southern California. The SWC's members are: Alameda County Flood Control and Water Conservation District Zone 7; Alameda County Water District; Antelope Valley-East Kern Water Agency; Casitas Municipal Water District; Castaic Lake

comments on the National Pollutant Discharge Elimination System Tentative Permit for the Central Contra Costa Sanitary District's ("CCCSD") Wastewater Treatment Plant. The Public Water Agencies thank the San Francisco Regional Water Quality Control Board ("Regional Board") for the opportunity to present this input and our views on the Tentative Permit.

The Public Water Agencies also respectfully request designated party status for themselves and their members at the hearing regarding the Tentative Permit. For the reasons we outline in our attached comments, the Public Water Agencies are harmed by CCCSD's discharges of treated sewage into Suisun Bay, including by the impacts caused by the discharges on the ecology, aquatic life and habitat in the Bay-Delta. These impacts have contributed to severe water restrictions that have resulted in human hardship, irretrievable resource losses, and economic and environmental harms to the Public Water Agencies, our members and the communities that they serve. Accordingly, the Public Water Agencies meet the statutory requirements for designated party status and ask the Regional Board to grant our request. The Public Water Agencies have a direct interest in the Tentative Permit.

Thank you for this opportunity to participate in the CCCSD discharge proceeding. We request that further notices and correspondence concerning this proceeding be directed to the individuals identified on the attached Public Water Agencies contact list.

Sincerely,

Daniel Nelson
Executive Director
San Luis & Delta-Mendota Water Authority

Terry Erlewine
General Manager
State Water Contractors

Enclosures

Water Agency; Central Coastal Water Authority; City of Yuba City; Coachella Valley Water District; County of Kings; Crestline-Lake Arrowhead Water Agency; Desert Water Agency; Dudley Ridge Water District; Empire-West Side Irrigation District; Kern County Water Agency; Littlerock Creek Irrigation District; Metropolitan Water District of Southern California; Mojave Water Agency; Napa County Flood Control and Water Conservation District; Oak Flat Water District; Palmdale Water District; San Bernardino Valley Municipal Water District; San Gabriel Valley Municipal Water District; San Gorgonio Pass Water Agency; San Luis Obispo County Flood Control & Water Conservation District; Santa Clara Valley Water District; Solano County Water Agency; Tulare Lake Basin Water Storage District.

Public Water Agencies' Comments on the Tentative NPDES Permit Renewal for the Central Contra Costa Sanitary District Wastewater Treatment Plant October 31, 2011

The State Water Contractors, Inc. ("SWC") and the San Luis & Delta-Mendota Water Authority ("SLDMWA" or "Authority") (collectively, "Public Water Agencies") appreciate the opportunity to comment on the tentative renewal of the National Pollutant Discharge Elimination System permit ("Tentative Permit") for the Central Contra Costa Sanitary District's ("CCCSD") Wastewater Treatment Plant ("Treatment Plant").

The CCCSD Treatment Plant collects and discharges, on average, 40 million gallons per day of treated sewage into Suisun Bay, a tidal estuary within the defined critical habitat for threatened and endangered aquatic species. Included in the daily discharge are thousands of pounds of "nutrients," in the form of ammonium (or "ammonia as nitrogen") that CCCSD does not remove or otherwise treat at its Treatment Plant.

Suisun Bay lies at the confluence of the Sacramento and San Joaquin Rivers, forming the western tip of the Sacramento-San Joaquin River Delta. To its west, Suisun Bay drains to the Carquinez Strait, which connects to San Pablo Bay, a northern extension of San Francisco Bay. The greater San Francisco Bay/Sacramento-San Joaquin River Delta estuary system is referred to as the Bay-Delta estuary, the largest estuary on the United States' pacific coast.

The Public Water Agencies have a significant interest in how the Regional Board regulates CCCSD's discharge because the members of the Public Water Agencies receive water through the California State Water Project ("SWP") and the federal Central Valley Project ("CVP"). These projects collect and store water in large reservoirs in northern California for use throughout the State. After water is released from reservoirs, the water flows to the Delta. From there, water is pumped for use by more than 2 million acres of prime farmland and some 25 million Californians living in two-thirds of the state's households.

It is well documented that water quality and aquatic resources within the Bay-Delta estuary are under stress. The estuary and many of its tributaries are listed as impaired, and the populations of both pelagic and anadromous fish have suffered serious decline in recent years. To date, regulators have largely responded to the decline by regulating the SWP and CVP and restricting the water available to the members of the Public Water Agencies. These restrictions have had a direct and severe adverse impact on the ability of the members of the Water Agencies to serve the people who depend on SWP and CVP water.

Unfortunately, while the focus on water users has resulted in great hardship, it has not led to real improvements in the delta smelt, salmon or other aquatic life of the Bay-Delta. To the Public Water Agencies, this has not been a surprise. Federal and state agencies have long recognized that nutrient loadings seriously impacts water quality and aquatic life.[3] Although it has long

[3] According to U.S. EPA: "Nutrient pollution, especially from nitrogen and phosphorus, has consistently ranked as one of the top causes of degradation in some U.S. waters for more than a decade. Excess nitrogen and phosphorus lead to significant water quality problems including harmful algal blooms, hypoxia and declines in wildlife and

been thought that the Bay Delta Estuary was not vulnerable to nutrient impacts, that is no longer the case.

Indeed, writings by Regional Board staff have acknowledged the scientific evidence that establish the nexus between nutrient discharges and impacts on aquatic life. On June 4, 2010, the Regional Board submitted a letter to the Central Valley Regional Water Quality Control Board citing published studies that document the impacts of ammonium in Suisun Bay and urging the Central Valley Regional Board to take all necessary actions to ensure beneficial uses in Suisun Bay are fully protected.[4] Further, a work plan co-authored by a Regional Board Senior Scientist, Karen Taberski, states that "there is evidence that primary productivity is inhibited in Suisun Bay, and that NH_4 [ammonium] may be causing that inhibition." *See* Taberski, Dugdale, et al., SWAMP Monitoring Plan 2011-2012, *San Francisco Bay Region Work Plan, Monitoring Spring Phytoplankton Bloom Progression in Suisun Bay* at 1 (Dec. 2010) ("Work Plan").[5] The Work Plan recognizes that a "potentially important source of NH_4 to Suisun Bay is the Central Contra Costa Wastewater Treatment Plant" and that "nutrient concentrations, including NH_4, are higher in Suisun Bay compared to other" nearby bay systems. *Id.* at 2.

The Public Water Agencies fully support the Work Plan, which will directly assess CCCSD's relative contribution to the inhibition in primary productivity, as compared to the "dominant source of ammonium" in the Bay-Delta, the Sacramento Regional Wastewater Treatment Plant ("SRWTP"). Work Plan at 2. According to the Work Plan, the SRWTP's discharges 90 percent of the ammonium in the Bay-Delta. *Id.* Indeed, SRWTP's massive discharge is certainly the predominant source of ammonium that impairs beneficial uses in receiving waters from the Sacramento River at Freeport all the way through Suisun Bay, as the Central Valley Regional Board has found and as we have outlined in filings before the Central Valley and State Boards. Relative to the Sacramento Regional WWTP, CCCSD's discharge contributes a smaller share of the total ammonium loadings and appears, at least at this point, to primarily impact western Suisun Bay.

However, despite the clear differences between the two discharges, the CCCSD discharge of thousands of pounds per day of ammonium is significant. Hence, as detailed here and in the materials here provided for the Regional Board's review, the Public Water Agencies submit that there is already overwhelming scientific literature and research, grounded in sound science, to demonstrate that the ongoing discharge of nutrients from both Sacramento Regional *and* CCCSD are major stressors that are contributing to the decline of the food web that supports aquatic life throughout the Bay-Delta. The record is more than clear that CCCSD's ammonium discharge has the potential to exceed water quality standards and to impair designated beneficial uses of receiving waters.

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wildlife habitat. Excesses have also been linked to higher amounts of chemicals that make people sick." http://water.epa.gov/scitech/swguidance/standards/criteria/nutrients/

[4] San Francisco Bay Regional Water Quality Control Board letter from Bruce H. Wolfe, Executive Officer, to Kathy Harder, Central Valley Regional Water Quality Control Board re Comments on "Issue Paper – Aquatic Life and Wildlife Preservation Related Issues – Proposed NPDES Permit Renewal for Sacramento Regional County Sanitation District Sacramento Regional Wastewater Treatment Plant". June 4, 2010.

[5] http://www.waterboards.ca.gov/water_issues/programs/swamp/docs/workplans/1112rb2wp.pdf

As such, the Public Water Agencies urge the Regional Board as follows:

1. The Regional Board should revise the Tentative Permit to expeditiously provide for nitrification of the discharge to remove ammonium. Further interim limits should be set that restrict the discharge while treatment is designed and built. In addition, as part of the ongoing studies, the Regional Board and CCCSD should evaluate whether denitrification should also be required.

2. In the alternative, if the Regional Board is convinced that further study is needed before requiring nutrient removal, the Water Agencies urge the Regional Board to expedite (consistent with good scientific practice) the completion of the ongoing studies, but defer issuing this Permit until that work is completed and published, so the Regional Board may consider those data and analyses.

3. Lastly, if the Regional Board determines it must proceed with a permit now, the Water Agencies urge the Regional Board to include a detailed framework in the final permit that includes (a) a schedule for promptly completing the studies, with assured funding by CCCSD, (b) a clear procedure for reconsideration of the ammonium issue, with full public participation in the process, and (c) interim limits consistent with the actual maximum concentrations of ammonium in CCCSD discharges.

I. The Tentative Permit Does Not Address the Significant Uncontrolled Discharge of Ammonia-Nitrogen From the CCCSD Wastewater Treatment Plant

The Public Water Agencies' concern with the Regional Board's Tentative Permit is grounded in one irrefutable fact well known to the Regional Board: on average the CCCSD discharges approximately 7,000 pounds of untreated ammonium in its wastewater every day. *See* Taberski, et al., *supra.*[6] This daily loading into the Bay-Delta estuary system is pouring more than *2,500,000 pounds* of untreated nutrients directly into Suisun Bay every year.[7]

Moreover, if the scope of the discharge were not enough, CCCSD discharges directly into the critical habitat of endangered and threatened species. As the Regional Board is surely aware, the Bay-Delta estuary provides critical habitat for at least five species listed under the federal Endangered Species Act, including the delta smelt[8] and spring-run Chinook salmon (threatened), the winter-run Chinook salmon (endangered), and the fall- and late fall-run Chinook salmon (species of concern).[9] Given the expansive view of federal and state agencies of the need to

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[6] The actual discharge may be considerably higher, as the permitted flow rate is 53.8 million gallons per day and the 99th percentile daily effluent flow rate is 70.3 million gallons per day. Tentative Order, Attachment F at F-19.

[7] 3.5 tons x 2000 lbs. x 365 day = 2,555,000 lbs./year.

[8] The U.S. Fish and Wildlife Service listed the delta smelt as a threatened species in 1993 and designated critical habitat for the smelt in 1994. 58 Fed. Reg. 12854 (March 5, 1993); 59 Fed. Reg. 65256 (Dec. 19, 1994). A threatened species is "any species which is likely to become an endangered species within the foreseeable future throughout all or a significant portion of its range." 16 U.S.C. § 1532(20). Endangered species are those which are "in danger of extinction throughout all or a significant portion of its range." 16 U.S.C. § 1532(6).

protect listed species, one would expect efforts would be made to address the ammonium in CCCSD's discharge.

Yet, the Regional Board has proposed a permit that does not limit the ammonium in the CCCSD's discharge. In contrast, many other municipal wastewater treatment plants in central California that discharge into waters that feed into the Bay-Delta estuary have stepped up and made the investments (or been required to make the investments) needed to install treatment technology that would remove ammonium. Thus, if required to bring its treatment up to date, CCCSD would not be singled out to invest in new or unproven technology. These plants are listed in Table 1.[10]

Indeed, not only is CCCSD not required to treat and remove ammonium, the Tentative Permit allows CCCSD to discharge *more* ammonium than available data indicate it has been discharging. Thus, while the Tentative Permit now places effluent limits on ammonium, the proposed limits would allow CCCSD to discharge up to 10,000 pounds per day or more. Indeed, the monthly average effluent limitation is more than *twice* the maximum effluent concentration ("MEC") that has been observed – and the daily limit is almost three times higher than the MEC. *Compare* Tentative Order, Attachment F at F-17 (MEC of 30.2 mg/L) *with* Tentative Order at 9 (effluent limitations on Total Ammonia as N – average monthly limit 65 mg/L, maximum daily 84 mg/L).

II. CCCSD's Significant Uncontrolled Discharge of Ammonium May Adversely Affect Beneficial Uses of Waters of this State and the United States

In the Tentative Permit, the Regional Board staff has not discussed the substantial available evidence linking nutrient discharges to serious impacts on aquatic life. The Public Water Agencies submit that the full body of research and scientific literature already available demonstrates that full ammonium removal should be applied to CCCSD. At a minimum, the Public Water Agencies strongly urge the Regional Board to consider carefully that evidence before deciding whether to allow the continued untreated discharge of tons of "nutrients" into Suisun Bay every day.

In fact, the overwhelming data and scientific literature demonstrate the millions of pounds of nutrients discharged by CCCSD directly to a critical habitat for threatened and endangered species is likely causing toxic effects on aquatic species and contributing to the altering of the aquatic food web—the foundation of the entire Bay Delta ecosystem. These impacts to the beneficial uses of waters of this State and the United States require a far more vigorous response by the Board than in the Tentative Permit.

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[9] *See* U.S. Fish & Wildlife Service Sacramento Fish & Wildlife Office Species Account, available at www.fws.gov/sacramento/es/animal_spp_acct/delta_smelt.pdf; Updated Status of Federally Listed ESUs of West Coast Salmon and Steelhead, available at www.nwr.noaa.gov/Publications/Biological-Status-Reviews/loader.cfm?csModule=security/getfile&pageid=21346

[10] *See also* West Yost Associates, Wastewater Control Measures Study (March 2011), available at http://www.swrcb.ca.gov/rwqcb5/water_issues/drinking_water_policy/dwp_wastewtr_cntrl_meas_stdy.pdf. This report, prepared for the Central Valley Regional Board, lists 26 treatment plants that are currently achieving nutrient removal and nine additional plants that are required by current NPDES permits to achieve this standard of treatment.

That untreated nutrients cause serious impacts on aquatic life is not a novel proposition, as detailed here and in the enclosed Technical Memorandum collecting and summarizing the recent nutrient research. *See* Water Agencies' Technical Memorandum (November 1, 2011) (Attachment 1). Indeed, among other work, the Memorandum highlights the most recent work done by Dr. Patricia Glibert, et al.[11] Dr. Glibert's latest paper analyzes comparable eco-systems and demonstrates that the fact that nutrient loadings materially impact the food web is well established by stoichiometric analysis of data from systems across the United States and around the world. Thus, while the research in the Work Plan that is now being conducted in Suisun Bay will surely advance the body of knowledge, the existing literature provides ample support for the Regional Board to take action now to restrict the discharge of nutrients.

Indeed, in addition to the literature and research outlined in the Technical Memorandum, the Public Water Agencies and their members have previously provided comments in other proceedings which further detail how ammonium is harming aquatic species in the Bay-Delta estuary and altering the aquatic food web. *See* Water Agencies' Response to Discharger's Petition For Review, In the Matter of the Sacramento Regional County Sanitation District's Petition for Review SWRCB/OCC File Nos. A-2144(a) and A 2144(b) (Consolidated) (May 4 and 6, 2011); San Luis & Delta-Mendota Water Authority and State Water Contractors Comments on EPA Advanced Notice of Proposed Rulemaking Regarding Water Quality Challenges in the San Francisco Bay/Sacramento-San Joaquin Delta Estuary, Docket No. EPA-R09-OW-210-0976, 76 Federal Register 9709, February 22, 2011 (April 21, 2011); Westlands Water District's Comments on EPA Advanced Notice of Proposed Rulemaking Regarding the Water Quality Challenges in the San Francisco Bay/Sacramento-San Joaquin Delta Estuary 76 Federal Register 9709 (February 22, 2011) Docket Number: EPA-R09-OW-2010-0976 (April 25, 2011);[12] Proposed NPDES Permit Renewal and TSO, Sacramento Regional County Sanitation District, Water Agencies' Testimony before Central Valley Regional Water Quality Control Board Meeting (December 9, 2010) (Water Agencies' Testimony); Comments of the Water Agencies on the Tentative Waste Discharge Requirements Renewal for the Sacramento Regional County Sanitation District Sacramento Regional Wastewater Treatment Plant (Oct. 8. 2010);[13] Comments of Westlands Water District (Westlands) and the San Luis & Delta-Mendota

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[11] Dr. Glibert is an aquatic ecologist and nutrient bio-geochemist with over 30 years of experience working on issues related to nutrient loading, nutrient ratios, eutrophication, changes in trophic dynamics, harmful algae, and management implications of nutrients loading all over the world. She has a Ph.D. from Harvard University and was awarded an honorary doctorate degree from Linnaeus University, Sweden earlier this year. She has studied and published widely on nutrients and food web dynamics in systems covering phytoplankton nutrient uptake and photosynthesis, nutrient excretion by zooplankton, harmful algal physiology, nutrient preferential use by phytoplankton taxa, eutrophication, and global nutrient modeling. Her field investigations span the globe – including the Chesapeake Bay, Long Island Sound, Florida Bay, Australia, Brazil, the Baltic Sea, East China Sea, Kuwait Bay, Gulf of Oman, and Hong Kong coastal waters, as well as many other sites, including San Francisco Bay Delta. She serves as the co-chair of the U.S. National HAB Committee, chair of the committee on eutrophication for the international GEOHAB Programme, and co-chair of the international SCOR/LOICZ Working Group on HABs and Eutrophication. She has consulted with the governments of Kuwait and Oman on issues related to nutrient pollution, served as an independent advisor to the Chinese Academy of Sciences on their studies of eutrophication, served on numerous panels and advisory boards related to nutrient management for the federal government and the states of Florida and Maryland.

[12] http://www.regulations.gov/#!documentDetail;D=EPA-R09-OW-2010-0976-0037

[13] http://www.swrcb.ca.gov/centralvalley/board_decisions/tentative_orders/1012/sac_regional/srcsd_com_wateragencies.pdf

Water Authority (Authority) on Tentative Waste Discharge Requirements Renewal (NPDES Permit No. CA0077682) for Sacramento Regional County Sanitation District, Sacramento Regional Wastewater Treatment Plant (Oct. 8, 2010);[14] San Luis Delta-Mendota Water Authority and State Water Contractors Comments on Draft Report Titled "Nutrient Concentrations and Biological Effects in the Sacramento-San Joaquin Delta" (June 14, 2010);[15] Water Agencies' Comments on Aquatic Life and Wildlife Preservation Issues Concerning the Sacramento Regional Wastewater Treatment Plant NPDES Permit Renewal (June 1, 2010).[16] The Public Water Agencies hereby incorporate by reference the arguments, analysis, data and scientific literature catalogued in those comments.

Among other things, the research outlines four basic scientific propositions:

1. Excessive ammonium has been shown to be toxic to copepods

Recent studies indicate that ammonium at concentrations present in the Bay-Delta estuary and Suisun Bay is acutely toxic to copepods central to the food web that supports aquatic life in the Bay-Delta estuary, including the endangered delta smelt. *See* Technical Memorandum at 7. Specifically, Dr. Swee Teh (and colleagues) at the University of California at Davis[17] have done a variety of studies of the effects of ammonium on the native *Eurytemora affinis* and *Pseudodiaptomus forbesi.*[18] Dr. Teh found ten percent mortality occurred in invertebrate species exposed to ammonia concentrations present in the Sacramento River using a 96-hour

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[14] http://www.swrcb.ca.gov/centralvalley/board_decisions/tentative_orders/1012/sac_regional/srcsd_com_westlands.pdf

[15] San Luis & Delta Mendota Water Authority and State Water Contractors letter to Dr. Chris Foe, Central Valley Regional Water Quality Control Board re Comments on Draft Report Titled "Nutrient Concentrations and Biological Effects in the Sacramento-San Joaquin Delta". June 14, 2010.

[16] Water Agencies letter to Ms. Kathleen Harder, Central Valley Regional Water Quality Control Board re Comments on Aquatic Life and Wildlife Preservation Issues Concerning the Sacramento Regional Wastewater Treatment Plant NPDES Permit Renewal. June 1, 2010.

[17] Dr. Teh is a Ph.D in Comparative Pathology and a Research Toxicologist and Pathologist in the Department of Anatomy, Physiology, and Cell Biology at the University of California - Davis. He serves as the Interim Director of the Aquatic Toxicology Laboratory at the UC-Davis School of Veterinary Medicine, and is a UC-Davis Faculty Member for the Graduate Group in Ecology, the Center for Aquatic Biology and Aquaculture, the Center for Health and the Environment, and the John Muir Institute of Environment. Dr. Teh conducted his work under the auspices of the Central Valley Regional Water Quality Control Board.

[18] The relevant research and related writings include Dr. Teh's presentation at the Ammonia Summit at Central Valley Regional Water Board http://www.waterboards.ca.gov/centralvalley/water_issues/delta_water_quality/ambient_ammonia_concentrations/index.shtml (August 18-19, 2009) ("Teh Presentation") (also provided with these comments as an attachment to the Declaration of Dr. Swee Teh (May 4, 2011) ("Teh Decl."); Werner, et al., Pelagic Organism Decline (POD): Acute and Chronic Invertebrate and Fish Toxicity Testing in the Sacramento-San Joaquin Delta 2008-2010, Final Report Submitted to the California Department of Water Resources (July 24, 2010), (http://www.science.calwater.ca.gov/pdf/workshops/POD/Werner%20et%20al_2010_POD2008-2010_Final%20Report.pdf) (also at Teh Decl. Exhibit 3); Full Life-Cycle Bioassay Approach to Assess Chronic Exposure of P. forbesi to Ammonia/Ammonium to the Delta Pelagic Organism Decline Contaminants Work Team (July 6, 2010) Teh Decl. Exhibit 4; Letter from S. Teh to C. Foe (November 10, 2010) Teh Decl. Exhibit 5; S. Teh, et al., Final Report, Full Life-Cycle Bioassay Approach to Assess Chronic Exposure of *Pseudodiaptomus forbesi* to Ammonia/Ammonium – Submitted to C. Foe and M. Gowdy (March 4, 2011) Teh Decl. Exhibit 6.

toxicity test.[19] Dr. Teh has likewise conducted life cycle tests to assess the impacts of different concentrations of ammonium on the ability of the copepod to reproduce and thrive. Dr. Teh found that total ammonia impacted adult *P. forbesi* reproduction at concentrations greater than or equal to 0.79 mg L^{-1}, while nauplii and juveniles are affected at concentrations as low as 0.36 mg L^{-1}.[20] Dr. Teh repeated the life cycle testing and confirmed his results, which he provided to the Central Valley Regional Board.[21]

The toxic effect of total ammonia is a major stressor on aquatic life that has a pervasive impact across the Bay-Delta estuary. In fact, CCCSD's own data show that ammonium levels in the receiving water in the vicinity of the Treatment Plant discharge are at levels sufficient to be toxic to copepods 15% of the time. *See* Exhibit 1.

2. The excess ammonium is inhibiting nitrogen uptake by diatoms and reducing diatom primary production in the Bay-Delta.

In addition to toxic effects, the ammonium loadings are disrupting the food supply by inhibiting nitrogen uptake by diatoms in the Bay-Delta estuary. The phytoplankton that form the base of the food web are essential to a healthy aquatic ecosystem. Primary consumers, including copepods (such as *P. forbesi*) rely on that primary production by phytoplankton as their main source of food, which, in turn, serve as food source for other aquatic life. In recent research, Dr. Richard Dugdale and others have found that excessive ammonium from treatment plant discharges is inhibiting nitrogen uptake by diatoms and contributing to reduced diatom production in the Bay-Delta.[22] *See* Technical Memorandum at 1 and Work Plan at 1-3. Indeed, as the Work Plan acknowledges, Dr. Dugdale has found that at an ammonium concentration of 4 μmol L^{-1}, nitrate uptake is fully inhibited. Work Plan at 2.[23] This level of ammonium is exceeded a majority of the time in western Suisun Bay near Martinez. *See* Exhibit 2. In fact, the ammonium levels in the receiving water in the vicinity of the Treatment Plant discharge exceed

[19] Werner, et al., *supra;* Teh Presentation, *supra.* Dr. Teh was unfairly criticized that his initial testing did not apply a representative average pH. This criticism was not valid, as Dr. Teh's first test was within the range found in the River 20 percent of the time. Nonetheless, Dr. Teh repeated his analysis and again observed that comparable toxic effects occurred at a pH of 7.8. Teh, S. et al., August 31, 2011 Final Report to C. Foe, *supra.*

[20] Teh, S. Full Life-Cycle Bioassay Approach, *supra* (Teh Decl. Exhibit 4).

[21] Teh, S. et al., Final Report to C. Foe, *supra (*August 31, 2011 Report).

[22] *See e.g.,* Parker, A.E., A.M. Marchi, J. Drexel-Davidson, R.C. Dugdale, and F.P. Wilkerson. "Effect of ammonium and wastewater effluent on riverine phytoplankton in the Sacramento River, CA. Final Report to the State Water Resources Control Board; Wilkerson, F.P., R.C. Dugdale, V.E. Hogue and A. Marchi, 2006. Phytoplankton blooms and nitrogen productivity in San Francisco Bay, Estuaries and Coasts 29(3): 401-416. ; Dugdale, R.C., F.P. Wilkerson, V.E. Hogue and A. Marchi. 2007. The Role of ammonium and nitrate in spring bloom development in San Francisco Bay. *Estuarine, Coast and Shelf Science* 73: 17-29 ; Sommer, T., C. Armor, R. Baxter, R. Bruer, L. Brown, M. Chotkowski, S. Culberson, F. Feyrer, M. Gingras, B. Herbold, W. Kimmerer, A. Mueller-Solger, M. Nobriga and K. Souza. 2007. The Collapse of Pelagic Fishes in the Upper San Francisco Estuary, *Fisheries* 32(6):270-277; Glibert, P. 2010a. "Long-term changes in nutrient loading and stoichiometry and their relationships with changes in the food web and dominant pelagic fish species in the San Francisco Estuary, California," *Reviews in Fisheries Science.* 18(2):211 – 232.

[23] Note that even below that level, Dr. Dugdale has observed inhibitory effects, as have others, *see* Technical Memorandum at 1 (researchers "describe the threshold for inhibition of nitrate uptake at ammonium levels of approximately 1 μmol L^{-1}).but that the complete inhibition has been observed at ammonium concentrations of 4 μmol L^{-1}.

the threshold level 87% of the time. *See* Exhibit 1. Further, in the receiving water in the vicinity of the discharge point, 14% of the data show ammonium concentrations of *40* μmol L^{-1} or more, *10 times* above the inhibition threshold. *See* Exhibit 1.

While the additional research by Ms. Taberski and Dr. Dugdale outlined in the Work Plan will surely provide additional useful information to supplement the body of knowledge of how ammonium inhibits productivity, the existing data amply document the effects of nutrient discharges like those from CCCSD sufficient to require nutrient removal. At a minimum, as noted, the Regional Board should consider carefully these recent studies, before deciding whether or not to allow CCCSD to continue to discharge tons of nutrients into Suisun Bay.

3. Nutrient discharges into the Bay-Delta estuary are contributing to a shift in algal communities by changing the nutrient ratios to favor harmful, invasive species.

Further, the research of Dr. Glibert and others demonstrates that excessive ammonium discharges have adversely impacted aquatic life in the Bay-Delta by increasing the ratio of nitrogen to phosphorus in the receiving water which triggers impacts to the food web on which aquatic life depends. Increasing ammonium discharges, particularly when phosphorus discharges have been declining, degrades water quality by changing the ratio between dissolved inorganic nitrogen and phosphorus, as well as the total nitrogen to total phosphorus ratio. These ratios are known to have profound influences on food webs.[24] Dr. Glibert's research strongly suggests that changes in delta smelt and several other fish species' abundance are ultimately related to changes in ammonium load from wastewater discharges. Dr. Glibert concluded that "[r]emediation of pelagic fish populations should be centered on reduction of nitrogen loads and reestablishment of balanced nutrient ratios delivered from point source discharges."[25] *See* Technical Memorandum at 3-4.

4. Where implemented in impacted ecosystems, nutrient removal has improved the natural ecosystem and aquatic life.

Requiring nitrification and denitrification of wastewater treatment plant effluent would help restore the health of the ecosystem and aquatic life in the Bay-Delta estuary. Again, the literature is clear that requiring nutrient removal on wastewater treatment plants has proven to be effective at reversing the harmful effects of previously un/undertreated discharges and restoring native eco-systems. As just one example that is discussed in Dr. Glibert's work, nutrient removal at the Blue Plains treatment plant in Washington D.C. has reduced the invasive species and begun to restore the native vegetation in the river. Once nutrient removal was implemented at Blue Plains in the 1990s, within several years, the abundance of the invasive *Hydrilla* began to decline and the abundance of native grasses increased. There are many other examples in other systems. *See* Technical Memorandum at 4-5.

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[24] Sterner, R.W. and J.J. Elser. 2002. Ecological stoichiometry: The biology of elements from molecules to the biosphere. Princeton University Press, Princeton, N.J. Sterner and Elser (2002), state that, "Stoichiometry can either constrain trophic cascades by diminishing the chances of success of key species, or be a critical aspect of spectacular trophic cascades with large shifts in primary producer species and major shifts in ecosystem nutrient cycling."

[25] Glibert, P. 2010a.

To reiterate: The Public Water Agencies submit the existing literature amply documents the effects of nutrient discharges like those from CCCSD sufficient to require treatment. At a minimum, before issuing any permit to CCCSD, the Regional Board should consider carefully these studies, as the Central Valley Regional Board did in deciding to impose full nutrient removal on the Sacramento Regional WWTP.

III. The Regional Board's Consideration of Ammonium in the Tentative Order is Incomplete and Contrary to Law

The Regional Board considers ammonium (referred to as total ammonia as N) essentially in two ways. First, the Regional Board evaluates whether the ammonium in CCCSD's discharge has the reasonable potential to exceed a water quality objective and if so, whether a water quality based effluent limit is required. Second, after setting the limits, the Regional Board determined that the anti-backsliding requirements are met, because no previous permit included any limits. Neither analysis appears to be correct.

A. The Regional Board's application of a dilution factor is flawed and should be reconsidered

The Public Water Agencies are concerned that the Regional Board staff has erred in its application of a dilution factor to set effluent limits for ammonium. As the Tentative Order acknowledges, the applicable Basin Plan has Water Quality Objectives for un-ionized ammonia of 0.025 mg/L (annual median) and 0.16 mg/L (maximum) upstream of the Bay Bridge. Tentative Order, Attachment F at F-23. As the un-ionized component of total ammonia is only a small fraction of the total discharge, these are then converted to total ammonia objectives of 5.0 mg/L (acute) and 1.6 mg/L (chronic). Given that the MEC for ammonium is 30.2 mg/L, there unquestionably is a reasonable potential to exceed these objectives. However, the Tentative Order then proceeds to allow a substantial dilution for total ammonia to set the effluent limits by relying on the "Mixing Zone Study." Yet, this would not appear to be appropriate for several reasons:

<u>First,</u> the staff acknowledges the inability to set a Mixing Zone.

> Because of the complex hydrology of San Francisco Bay, a mixing zone has not been established. There are uncertainties in accurately determining an appropriate mixing zone. The models used to predict dilution have not considered the three dimensional nature of San Francisco Bay currents resulting from the interaction of tidal flushes and seasonal fresh water outflows. Being heavier and colder than fresh water, ocean salt water enters San Francisco Bay on a twice-daily tidal cycle, generally beneath the warmer fresh water that flows seaward. When these waters mix and interact, complex circulation patterns occur due to the varying densities of the fresh and ocean waters. ***The complex patterns occur throughout San Francisco Bay, but are most prevalent in the San Pablo, Carquinez Straight, and Suisun Bay areas.*** The locations of this mixing and interaction change, depending on the strength of

> each tide. Additionally, sediment loads from the Central Valley change on a long-term basis, affecting the depth of different parts of San Francisco Bay, resulting in alteration of flow patterns, mixing, and dilution at the outfall.

Tentative Order, Attachment F at F-20 (emphasis added). In fact, it emphasizes that the complexities are greatest in vicinity of the location of the of the CCCSD discharge. Given that, it would be wholly illogical for the Regional Board to then go ahead and apply a full dilution factor for ammonia to the CCCSD discharge and establish limits substantially greater than the maximum concentration observed.

Nonetheless, second, the staff goes on and proposes that dilution be applied to ammonia. In doing so, the staff asserts as follows:

> In granting dilution for ammonia, the Regional Water Board considered that ammonia is not a persistent pollutant and the Basin Plan states, "In most instances, ammonia will be diluted or degraded to a nontoxic state fairly rapidly." As such, there is unlikely to be cumulative toxicity effects associated with discharges containing elevated concentrations of ammonia. Therefore, granting dilution credits based on actual initial dilution is protective of water quality.

Tentative Order, Attachment F at F-20. However, the Basin Plan reference to the dilution of ammonia would appear to be referring to the "un-ionized" fraction of ammonia, not the ammonium that is the primary component of concern in the CCCSD discharge. Basin Plan, §3.3.20 at 3-7. Consistent with the general assessment that the "complex patterns" near the discharge point are not appropriate, the same approach ought to be applied to ammonium.

Third, the Basin Plan cautions *against* application of dilution in light of various concerns, including the difficulty in measuring the discharge in a tidal zone, Basin Plan §4.6.1.1 at 4-18, precisely where the CCCSD discharge is located. It further states that it would "consider inclusion of an effluent limitation greater than that calculated from water quality objectives when the increase in concentration is caused by implementation of significant water reclamation or water reuse programs at the facility; the increase in the effluent limitations does not result in an increase in the mass loading; and the water quality objectives will not be exceeded outside the zone of initial dilution." Basin Plan §4.6.1.1 at 4-18. But no such findings or analyses are done here.

In fact, fourth, the "Mixing Zone" study on which the staff rely recognizes that while there is initial dilution above the diffuser, that dilution *does not persist* beyond the zone of initial dilution. According to the study: "Under both chronic and acute conditions, the plume becomes vertically fully mixed over the diffuser, ***but re-stratifies later and is not mixed in the far field.***" *See* Near-field Mixing Zone and Dilution Analysis for the Central Contra Costa Sanitary District Outfall Diffuser to San Pablo Bay at 12 (Nov. 10, 2010) ("Mixing Zone Study"). Indeed, the Tentative Order expressly recognizes that the study only "presents the findings regarding the initial dilution of the discharge at the outfall." Attachment F at F-18. It makes no determination, as the Basin Plan directs, that "the water quality objectives will not be exceeded outside the zone of initial dilution." Attachment F at F-18.

Further, fifth, the Basin Plan also cautions against relying on modeling when evaluating a discharge in an estuarine environment because models are limited to the initial dilution analysis. This include EPA models, like the one used by CCCSD's consultant. Mixing Zone Study (uses EPA approved CORMIX model). Specifically, "the direction of waste transport varies over the course of the tidal cycle, so it is difficult to determine the fraction of new water versus recirculated water mixing with the discharge. U.S. EPA has developed several models of initial dilution for discharge plumes, ***but none takes into account transport due to tidal currents."*** Basin Plan §4.6.1.1 at 4-18.

Finally, sixth, regardless, the analysis of ammonia and dilution are entirely divorced from the overwhelming body of literature and data outlined in and provided with these comments. In fact, as outlined, the data demonstrate that the concentration of ammonium (or total ammonia as N) is consistently exceeding both the toxicity level for copepods and the inhibitory threshold for primary productivity. That suggests the proposed dilution is not the "conservative approach to calculating effluent limitations" required by the Basin Plan. Basin Plan §4.6.1.1 at 4-18 Instead, those data must be considered carefully and fully by the Regional Board before deciding that the tons of "nutrients" poured into the critical habitat for endangered and threatened species will simply be "diluted" away.

B. The Regional Board's analysis of Anti-degradation with regard to ammonia is contrary to established principles of law

California's Antidegradation Policy is summarized by a 1990 Administrative Procedures Update ("APU") from the State Board, which was meant to "provide guidance for the Regional Boards for implementing State Board Resolution No. 68-16 . . . and the Federal Antidegradation Policy, as set forth in 40 C.F.R. § 131.12." (APU 90-04, (July 1, 1990) at p. 1.) As such, the APU is designed to help the Regional Boards implement both federal policy (40 C.F.R. § 131.12) and the State Board's Antidegradation Policy (Resolution No. 68-16).

For high quality waters, Resolution 68-16 mandates that the water quality must be maintained—unless the Discharger can prove that lowering the water quality: (1) will provide "maximum benefit" to the state; (2) will not impair present or anticipated beneficial uses of the receiving water; and (3) will not violated water quality objectives. Additionally, discharges which increase the volume or concentration of waste in high quality waters must comply with discharge limits based on the "best practicable treatment or control" ("BPTC") which ensures that no pollution or nuisance will occur and that the highest water quality will be maintained.

If approved, the Tentative Permit would violate state and federal Antidegradation Policy by allowing degradation of receiving waters due to ammonium discharge. The Tentative Permit provides an ammonia discharge limit that is higher than the existing and historic ammonia discharge and which allows increased concentration of ammonia in the discharge. And the CCCSD is asking the Regional Board to issue a final discharge permit allowing the Treatment Plant to physically increase its discharge of secondarily treated sewage to up to 53.8 mgd, up nearly 34 percent from the existing baseline discharge of approximately 40 mgd. As a result, if the CCCSD's requested Permit were granted, CCCSD would physically increase the discharge of pollutants, like total ammonia nitrogen, into the Bay-Delta estuary—critical habitat for listed fish species and the largest single source of fresh water supply in all California.

Before the Regional Board can issue, reissue, amend, or revise such a discharge permit, however, federal and state Antidegradation Policy require the Regional Board to determine that permit conditions result in BPTC and to determine whether any water quality degradation that will result is permissible when balanced against the benefit to the public from issuing the permit. Here, the Tentative Permit makes no findings with respect to BPTC and the balancing of water quality degradation against any public benefit from allowing degradation. The Tentative Permit discloses zero analysis connecting facts in the record to any express conclusion that allowing the ongoing and increased discharge of ammonium and other pollutants complies with Antidegradation Policy.

To the extent that CCCSD might contend that some aspect of the required Antidegradation Policy analysis is addressed in some unspecified, prior California Environmental Quality Act ("CEQA") documentation, it is important to understand that substantially changed circumstances and significant new information prevent reliance on any prior CEQA review to support the Tentative Permit. The pelagic organism decline and scientific evidence that ammonia discharges harm the Bay-Delta foodweb would prevent reliance on any prior CEQA document to help support the Antidegradation Policy analysis and compliance determination that is required before CCCSD's new discharge permit may be approved.

The Regional Board must work with CCCSD to complete a legally adequate Antidgradation Policy analysis, and findings, and then circulate a revised Tentative Permit whose terms and Fact Sheet demonstrate Antidegradation Policy compliance, including ammonium effluent limits that achieve BPTC. Failure to do so will result in approval of an unlawful permit that degrades receiving water quality, violates water quality objectives, impairs designated beneficial uses—all in violation of state and federal water quality protection law.

IV. The Regional Board Should Take Affirmative Steps to Address the Ammonium in the CCCSD Discharge

A. The Regional Board should require CCCSD to install nitrification treatment

In view of the scientific evidence, the Regional Board should require CCCSD to reduce the nutrients in its discharge to acceptable levels. The Regional Board should set final effluent limits that are achievable with full nitrification treatment, as well as a reasonable schedule for designing and building the treatment system. Further, daily and monthly interim effluent limits on ammonium (ammonia as N) should be set that reflect the real maximum concentrations that have been observed in the discharge, with a modest margin for compliance.

There are well established technologies available to CCCSD to remove nutrients, as evidenced by the many other municipalities in California and across the country that have implemented ammonium removal through the "nitrification" of the wastewater. *See* discussion, *supra*.

Unquestionably, this is a feasible technology that has previously been determined to satisfy BPTC under California law.[26]

B. The Regional Board should defer issuing the Tentative Permit until pending studies on the effects of nutrients in CCCSD's discharge are completed

In the alternative, if the Regional Board remains convinced that further study must be completed before addressing nutrients in CCCSD's discharge, the Public Water Agencies urge the Regional Board to expedite (consistent with good scientific practice) the ongoing Work Plan, but defer issuing a final permit until that work is completed and published. In that way, the Regional Board may consider those data and analyses before issuing a new permit.

Given the stated focus of the Work Plan and the recognized concerns about the CCCSD discharge on the productivity in the Bay Delta estuary, proceeding to finalize the permit without the benefit of the latest data is unreasonable. Specifically, among other objectives, the study is designed to further assess whether "high NH_4 concentrations in Suisun Bay correlate with low primary production." Work Plan, Attachment at 2. It would be prudent to complete this work before granting CCCSD another five years to discharge millions of tons of nutrients into a critical habitat for threatened and endangered species. Indeed, part of the study is focused directly on CCCSD's contribution. As the Work Plan acknowledges, the data gathered to date has found that "an additional NH_4 signal was detected in the western part of Suisun that may play a role in controlling phytoplankton blooms in Suisun Bay." Work Plan at 4. One of the objectives of the research, therefore, is to determine if the CCCSD and its discharge of "7,000 lbs/day of ammonia to western Suisun Bay" are the source of that observed ammonium – as logically is currently "presumed to be" the case. *Id.; see also* Work Plan, Attachment at 2-9 (defining "Monitoring Objectives and Questions").

C. Alternatively, if the Regional Board is intent on finalizing a permit, the final permit should at a minimum be revised to address ammonium more effectively

Lastly, if the Regional Board determines it must proceed with a permit now and is not prepared to require full nitrification, then the Public Water Agencies urge the Regional Board to include a detailed framework in the final permit to address ammonium. The framework should include three components:

First: The Regional Board should make specific findings in the permit regarding its concern that the ammonium in CCCSD's discharge may be contributing to impacts in Suisun Bay and that therefore it is in the process of implementing studies to evaluate those concerns. The permit should then include a schedule for promptly completing the Surface Water Ambient Monitoring Program sampling and associated studies outlined in the Work Plan, with assured funding of that work by CCCSD as a condition for receiving the new permit.

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[26] A number of municipal sanitations districts have also been required to install "denitrification" treatment which follows nitrification to further treat the wastewater by removing the nitrates from the discharge. In the case of Sacramento Regional, the Water Agencies believe the evidence strongly supported the Central Valley Board's decision to require that additional treatment given the available data concerning that discharge. Here, as the Regional Board develops additional data regarding CCCSD's discharge, it should consider whether denitrification should also be included.

<u>Second:</u> The Regional Board should set a clear procedure for reconsideration of the ammonium issue, with full public participation in the process, after the studies are completed and the data are published. The Regional Board should include deadlines to ensure the ammonium limits are reconsidered no later than 12 months after the Regional Board issues a final permit.

<u>Third:</u> As outlined for the recommended interim limits, the Regional Board should set effluent limits consistent with the actual maximum concentrations of ammonium in the CCCSD discharge, with a modest margin for compliance. With the maximum observed concentration of ammonium according to the Regional Board in the range of 30 mg/L, there is no rational basis in the record to set limits of 65 mg/L (monthly) and 84 mg/L (daily maximum).

Table 1. Treatment Requirements for Select Wastewater Treatment Plants That Discharge Directly or Indirectly to the Bay-Delta Estuary.

Discharger	**Permitted Average Dry Weather Flow (mgd)**	**Treatment Requirements**
		Nitrification or Nitrification + Denitrification
Sacramento Regional WWTP	181	Yes
Stockton	55	Yes
Central Contra Costa Sanitary District	*53.8*	*No*
Fairfield	17.5	Yes
Manteca	17.5	Yes
Delta Diablo	16.5	No
Tracy	16	Yes
Vallejo	15.5	No
Vacaville Easterly WWTP	15	Yes
Woodland	10.4	Yes
Lodi	8.5	Yes
Davis	7.5	Yes
Mountain House	5.4	Yes
Benicia	4.5	No
Galt	4.5	Yes

Technical Memorandum

Summary of Nutrient Impacts

There is a large body of literature documenting the significant impacts of increased loading and changing forms, concentrations, and ratios of nitrogen and phosphorus both within the San Francisco Bay/Sacramento-San Joaquin Delta Estuary (Bay-Delta) and globally to the food web form and function. The form of nutrients matters. Wilkerson, *et al.* (2006) and Dugdale, *et al.* (2007) show that "bloom levels of chlorophyll are evident only when nitrate uptake occurs and that nitrate uptake only takes place at lower ambient ammonium concentrations." They conclude that ammonium concentrations greater than 4 μmol L^{-1} inhibit nitrate uptake by diatoms and thus suppress bloom formation. This level of ammonium is exceeded a majority of the time in the Sacramento River and Suisun Bay.

In enclosure experiments with samples from Central Bay, Suisun Bay and the Sacramento River at Rio Vista, Wilkerson *et al.* (in preparation) observed "a gradient of decreasing phytoplankton physiological rates in the upstream direction as far as Rio Vista." Algal biomass accumulation was delayed in enclosures from Suisun Bay and was not observed within 96 hours in enclosures from Rio Vista. Also supporting this finding, Parker, *et al.* (in review) observed a 55% decline in primary production in the Sacramento River below the Sacramento Regional Wastewater Treatment Plant compared to production above the Treatment Plant's outfall. Parker, *et al.* (in review) conclude that "[t]he quantitative reduction in primary productivity and nitrogen uptake at various points in the river was predictable and strongly related with NH_4 concentrations."

These observations of ammonium suppression are not new or unique to the Bay-Delta. There is a large body of scientific research describing ammonium suppression of algae productivity, which was first observed as far back as the 1930s (Ludwig, 1938; Harvey, 1953). Some of the early field demonstrations of this phenomenon were by MacIsaac and Dugdale (1969, 1972), followed by research in the Chesapeake Bay by McCarthy, *et al.* (1975). Lomas and Glibert (1999a) describe the threshold for inhibition of nitrate uptake at ammonium levels of approximately 1 μmol L^{-1}. Ammonium suppression of nitrate uptake when both nutrients are in ample supply should not be confused with the preferential use of ammonium by phytoplankton when nitrogen is limiting. Under the latter conditions, phytoplankton will use ammonium preferentially because it requires less energy than nitrate. Under the former conditions, the cells must cope with an excess; and in doing so, their metabolism is altered away from an ability to assimilate nitrate. Total primary productivity is suppressed as a result. This is particularly problematic for the Bay-Delta as it is already a comparatively low producing estuary (Jassby *et al.*, 2002). Laboratory experiments suggest that Delta-wide chl-a levels are now low enough to limit zooplankton abundance (Müller -Solger *et al.*, 2002).

Nutrient form also affects phytoplankton species composition. Cyanobacteria have been shown to preferentially use chemically reduced forms of nitrogen over nitrate in many studies.

Chemically reduced nitrogen not only includes ammonium, but also urea and dissolved organic nitrogen. This evidence comes from:

- Measurements of enzyme activities in the cells – enzymes that process these forms of nitrogen. Cyanobacteria have been shown to have some of the highest measured rates of urease, for example, relative to all phytoplankton species tested, and among cyanobacteria, *Microcystis* rates are the highest (Solomon et al., 2010).

- Directly determined rates of nitrogen uptake using isotope tracer techniques. These rates show that cyanobacteria use reduced nitrogen forms and, in many cases, avoid the chemically oxidized forms (Glibert *et al.,* 2004).

- Direct growth studies. These studies based on growth measurements in the laboratory demonstrate that growth rates of *Microcystis* can be significantly higher on urea than on nitrate (Berman and Chava, 1999). Meyer, *et al.* (2009) state: "Compared to NO_3- and N_2 (via fixation) as N sources, NH_4+ produces the highest growth and primary production rates for *Microcystis aeruginosa* and other cyanobacteria (*Aphanizomenon flos-aquae* and *Anabaena flos-aquae*) in laboratory studies [citations removed]" (Meyer *et al.,* 2009).

Moreover, retrospective analysis of the data in the Bay-Delta system further demonstrates that at very high ammonium concentrations (*i.e.*, > 200 $\mu g\ L^{-1}$), phytoplankton functional groups such as flagellates, cryptophytes and diatoms are outcompeted by cyanobacteria (Glibert, P., Univ. of Maryland. Personal communication). Thus, even though *Microcystis* may have a broad capability for using different forms of nitrogen to support their physiological demands for nitrogen, they have a greater capacity to take up and metabolize reduced forms of nitrogen compared to other functional groups and may have higher growth rates under reduced nitrogen compared to nitrate and thus may outcompete other phytoplankton groups at very high ammonium levels. Lehman et al. (2010) concedes: "Recent increases in ammonium concentration in the western delta may give a competitive advantage to *Microcystis* which rapidly assimilates ammonium over nitrate."

The physiological literature strongly supports the concept that different algal communities use different forms of nitrogen. Diatoms generally have a preference for nitrate; dinoflagellates and cyanobacteria generally prefer more chemically reduced forms of nitrogen (ammonium, urea, organic nitrogen) (Berg, *et al.*, 2001; Glibert, et al., 2004, 2006; Brown, 2009). It has long been recognized that diatoms may have a nutritional requirement for, and under some circumstances even a preference for, nitrate (Lomas and Glibert, 1999a; 1999b). Moreover, diatoms often show no evidence of nitrate uptake saturation under very high nitrate conditions (Collos et al. 1992, 1997), in contrast to the generally accepted saturating uptake kinetic relationships that are used to describe the relationship between nutrients and uptake rate. Thus, cyanobacteria may grow particularly well on ammonium while their competitors, such as diatoms, do not.

The shift in algal community composition in the Bay-Delta has been far more extensive than just the recent increase in annual blooms of *Microcystis.* The Delta's algal species composition has shifted from diatoms to smaller and lower quality food species such as flagellates, cryptophytes and cyanobacteria (Lehman, 2000; Lehman *et al.*, 2005; Lehman et al., 2010; Jassby *et al.*, 2002; Sommer *et al.*, 2007; Glibert, 2010; Glibert et al., 2011; Winder and Jassby, 2010) and to invasive macrophytes such as *Egeria densa* (Sommer, *et al.*, 2007; Nobriga et al., 2005; Glibert et al., 2011). Jassby (2008) states:

> *A decrease in percentage of diatom biovolume occurred during 1975–1989, caused by both a decrease in diatoms and an increase in green algae, cyanobacteria, and flagellate species biovolume (Kimmerer 2005; Lehman 1996), i.e., probably in the direction of declining nutritional value per unit biomass. In principle, the total nutritional value of a community could decrease even as its biomass increases. Moreover, changes in size, shape, and motility of species comprising the phytoplankton community could also affect their availability as food particles for crustacean zooplankton and other consumers.*

In addition, the ratios of nitrogen to phosphorus are known to have profound influences on food webs. Sterner and Elser (2002) state: "[s]toichiometry can either constrain trophic cascades by diminishing the chances of success of key species, or be a critical aspect of spectacular trophic cascades with large shifts in primary producer species and major shifts in ecosystem nutrient cycling."

The N:P ratio has long been shown to influence phytoplankton community composition and the presence - or absence - of native species and vegetation, as extensive studies have repeatedly demonstrated in systems around the world including: Hong Kong, Tunisia, Germany, Florida, Spain, Korea, Japan, and Washington D.C. (Chesapeake Bay), to name just a few. The Potomac River (Chesapeake Bay) was invaded by submerged aquatic vegetation, *Hydrilla,* and clams, *Corbicula*, when the N:P ratio of effluent from the large Blue Plains sewage treatment facility increased after phosphorus was reduced in the 1980s (Ruhl and Rybicki 2010). In Spain's Ebro River estuary, *Hydrilla* and *Corbicula* invaded shortly after phosphorus was removed from effluent (Ibanez *et al.* 2008). In Tolo Harbor, Hong Kong, nutrient loading, particularly phosphorus loading, increased due to population increases in the late 1980's. The result was that a distinct shift from diatoms to dinoflagellates was observed in the harbor, coincident with a decrease in the N:P ratio (Hodgkiss and Ho 1997; Hodgkiss 2001). Once the phosphorous was removed from the sewage effluent that was being discharged into the harbor and stoichiometric proportions were re-established, there was a resurgence of diatoms and a decrease in dinoflagellates (Lam and Ho 1989). In Tunisian aquaculture lagoons, dinoflagellates have been shown to develop seasonally when N:P ratios decrease (Romdhane, et al. 1998). Comparable results have been observed in systems in Germany (Radach *et al.*, 1990) and along the coast of Florida (Glibert *et al.*, 2004; Heil *et al.*, 2007).

N:P ratios have also been shown to influence zooplankton community composition. Norwegian studies monitored lakes for many years and found that different zooplankton tend to dominate under different N:P ratios, due to the different phosphorus content of different species found in the lake (Hessen 1997). Hessen (1997), for example, showed that a shift from calanoid copepods to *Daphnia* tracked N:P; calanoid copepods retain proportionately more N, while *Daphnia* are proportionately more P rich. Studies from experimental whole lake ecosystems found that zooplankton size, composition and growth rates changed as the N:P ratio varied (*e.g.*, Schindler 1974, Sterner and Elser 2002).

There has been a measureable change in the N:P ratio in the Bay-Delta, an increase in total N loading, a decrease in total P loading, and a change in the dominant form of nitrogen from nitrate to ammonium (Glibert, 2010). In a retrospective analysis of 30 years of data from the Bay Delta, Glibert (2010; Glibert et al., 2011) found that the variation in these nutrient concentrations and ratios is highly correlated to variations in the base of the food web, primarily the composition of phytoplankton, to variations in the composition of zooplankton, to variations in the abundance of invasive clams, and to variations in the abundance of several fish species.

Winder and Jassby (2010) provide additional documentation of the shift that has occurred in the phytoplankton and zooplankton community.

The shift in the phytoplankton community has ripple effects through the food web. Cloern and Dufford (2005) state, "[t]he efficiency of energy transfer from phytoplankton to consumers and ultimate production at upper trophic levels vary with algal species composition: diatom-dominated marine upwelling systems sustain 50 times more fish biomass per unit of phytoplankton biomass than cyanobacteria-dominated lakes [citations removed]." Slaughter and Kimmerer (2010) provide further support. They observed lower reproductive rates and lower growth rates of the copepod, *Acartia* sp. in the low salinity zone compared to taxa in other areas of the estuary and conclude that "[t]he combination of low primary production, and the long and inefficient food web have likely contributed to the declines of pelagic fish."

There is also a growing body of literature documenting improvements in ecosystem functions in systems where nutrient loading is reduced. Reducing nutrient loading in the Chesapeake Bay, Tampa Bay, and coastal areas of Denmark has proven to be effective at reversing the harmful effects of previously undertreated discharges and restoring the native systems. For example, within several years of increasing nutrient removal at the Blue Plains treatment plant in Washington DC, N:P ratios in the Potomac River declined, the abundance of the invasive *Hydrilla verticillata* and *Corbicula fluminea* began to decline (Figure 1), and the abundance of native grasses increased (Ruhl and Rybicki 2010).

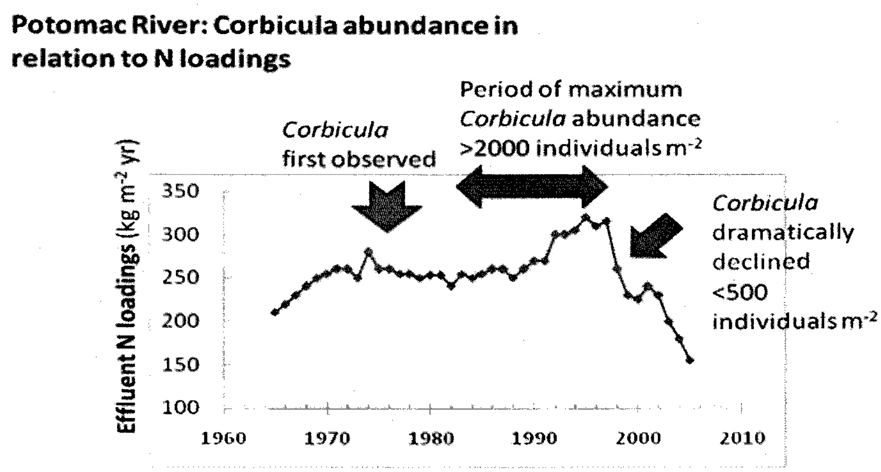

Figure 1. Comparative relationships for the Potomac River showing the change in effluent N loading and the relative abundance of the invasive clam, *Corbicula fluminea* clams. Data derived from Dresler and Cory (1980), Jaworski *et al.* (2007), and Cummins *et al.* (2010).

Tampa Bay provides another important example. Eutrophication problems in the Bay were severe in the 1970s, with N loads approximating 24 tons per day, about half of which was due to point source effluent (Greening and Janicki 2006). Several years after nitrogen and phosphorus reductions were achieved, native seagrass began to increase. Lower nutrient discharges also had positive effects on the coastal waters around the island of Funen, Denmark (Rask *et al.* 1999). Since the mid 1980s, there has been a roughly 50% reduction in the loading of N and P in the region due to point source reductions. Again, native grasses returned and low oxygen problems were reversed.

Moreover, there is recent evidence that diatom blooms can be restored in the Bay-Delta if ammonium loading were reduced. In Suisun Bay a diatom bloom reached chlorophyll concentrations of 30 $\mu g\ L^{-1}$ during spring 2000 when ammonium concentrations declined to 1.9 $\mu mol\ L^{-1}$ (Wilkerson *et al.* 2006). Similarly, chlorophyll concentrations in Suisun Bay reached 35 $\mu g\ L^{-1}$ during spring 2010 when ammonium concentrations declined to 0.5 $\mu mol\ L^{-1}$ (Dugdale *et al.*, 2011). These blooms are comparable to spring chlorophyll levels from 1969-1977 (Ball and Arthur, 1979) when ammonium concentrations were 1.8 $\mu mol\ L^{-1}$ during summer and 4.0 $\mu mol\ L^{-1}$ during winter (Cloern and Cheng, 1981). If clam abundance declines, as has occurred in San Pablo Bay and South San Francisco Bay (Cloern *et al.*, 2007), chlorophyll levels may also be restored during summer in Suisun Bay if ammonium loading were reduced.

Additionally, as Glibert (2010) reported, "[s]upporting the idea that correct balance of nutrients is important for restoration of delta smelt and other pelagic fish, there is a small but apparently successful subpopulation of delta smelt in a restored habitat, Liberty Island. Liberty Island is outside the immediate influence of Sacramento River nutrients. It has abundant diatoms among a mixed phytoplankton assemblage, as well as lower NH_4 levels and higher ratios of NO_3:NH_4 than the main Sacramento River [citations removed]."

The recent increase in *Microcystis* bloom frequency and size can also be explained by changes in Delta nutrients. Based on stable isotope analyses of particulate organic matter and nitrate, Kendall (2010) observed that ammonium, not nitrate, is the dominant source of nitrogen utilized by *Microcystis* at the Antioch and Mildred Island sites in the summer 2007 and 2008.

Nutrients affect more than *Microcystis* growth; nutrients may also affect its production of toxins. In Daechung Reservoir, Korea, researchers found that toxicity was related not only to an increase in N in the water, but to the cellular N content as well (Oh, *et al.* 2000). A very recent report by van de Waal (2010) demonstrated in chemostat experiments that under high CO_2 and high N conditions, microcystin production was enhanced in *Microcystis*. Similar relationships were reported for a field survey of the Hirosawa-no-ike fish pond in Kyoto, Japan, where the strongest correlations with microcystin were high concentrations of NO_3 and NH_4 and the seasonal peaks in *Microcystis* blooms were associated with extremely high N:P ratios (Ha *et al.* 2009). Thus, not only is *Microcystis* abundance enhanced under high N:P, but its toxicity is as well (Oh, *et al.* 2000).

Glibert *et al.* (2011) provides further support for the hypothesis that nutrient form and ratio is driving food web composition in the Delta. Using several different statistical approaches, Glibert *et al.* (2011) evaluated the relationships between approximately thirty different aquatic species and various nutrient ratios and found significant correlations for a majority of them. After comparing trends in the Bay-Delta estuary to those in Lake Washington, Potomac River, Hudson River and several European lakes and estuaries, they state,

> *Moreover, the physiology of the resident organisms and biogeochemical pathways lends support to the premise that similar trophic structure, including the appearance of* Microcystis, *in many of these systems has resulted from similar nutrient dynamics, biogeochemistry and food web interactions that resulted, in turn, from changes in stoichiometry and the relative abilities of different types of organisms to either sequester nutrients and/or to tolerate nutrients that are in excess (e.g., NH_4^+).*

They suggest that, "[r]eductions in N (especially NH_4^+) will allow organisms, from diatoms to fish, that cannot withstand high NH_4^+ (and/or that are outcompeted by NH_4^+ -tolerant organisms, such as various harmful dinoflagellates and cyanobacteria), to compete."

Glibert *et al.* (2011) found, "[f]or all organisms, with the exception of *Acartia*, for which strong correlations were observed with X2 (Table 9), *i.e.*, *Eurytemora*, *Pseudodiaptomus*, *Daphnia*, *Bosmina*, *Corbula*, *Crangon*, longfin smelt, splittail, striped bass, starry founder, crappie, sunfish and largemouth bass, equal or more significant correlations were observed with nutrients or nutrient ratios." This analysis determined pairwise relationships between biological parameters and nutrients and/or nutrient ratios using both the original data and data that were adjusted for autocorrelation. Glibert *et al.* (2011) also found that total phosphorus "explained at least as much of the variability in delta smelt as did the [Feyrer *et al.*, 2010] habitat index (Table 4), and dinoflagellate abundance explained even more (Table 6)." Unlike the X2 relationships whose mechanisms of effect are largely unknown, the nutrient relationships have a strong mechanistic explanation in ecological stoichiometry and stable state principles.

Ammonia Toxicity

Studies have been conducted by scientists at UC Davis investigating the effects of ammonia to the calanoid copepod *Pseudodiaptomus forbesi* using a full-life cycle bioassay approach. *P. forbesi* is an important food organism for the young of many fish species in the Bay-Delta including delta smelt and longfin smelt, two State listed species. Evidence of the toxic effects of ammonia on *P. forbesi* comes from life cycle tests conducted by Teh *et al.* (2011). Teh *et al.* (2011) found that total ammonia nitrogen at 0.36 ± 0.01 mg L^{-1} significantly affects the recruitment of new adult copepods and total ammonia nitrogen at 0.38 ± 0.01 mg L^{-1} significantly affects the number of newborn nauplii surviving to 3 days old.

Clam Invasion

There is no denying that the overbite clam has had a significant impact on the ecosystem since it took hold in the mid-1980s. Kimmerer (2002) and Kimmerer *et al.* (2009) found that many of the relationships between spring X2 and abundance changed in the mid-1980s, presumably due to the invasion by the overbite clam, *Corbula amurensis.* Phytoplankton biomass also declined significantly due to grazing pressure from the invasive clams. There is some scientific debate regarding the ability, or lack thereof, to manage clam populations by increasing freshwater outflows. However, this strategy fails to account for the potential consequences of an increased distribution in the freshwater clam, *Corbicula fluminea*, if freshwater flow is used to try to push the distribution of the brackish water clams further west of the Delta.

In addition, Glibert *et al.* (2011) found that "the change after 1987 also corresponds with the change in nutrient loading. X2 is strongly correlated with PO_4^{3-}, TP and NH_4^{+}." Glibert (2010) suggested that changes in nutrients created the environment in which these clams could dominate. Glibert (2010) found a strong correlation between the CUSUM trends in clam abundance and ammonium concentration and in the ratio of inorganic nitrogen to inorganic phosphorus (DIN:DIP).

Glibert *et al.* (2011) provides further support for nutrient effects on clam abundance as well as on the abundance of other invasive organisms such as non-native centrarchids and non-native invasive weeds. Using several statistical approaches, Glibert *et al.* (2011) found "a strong long-term correlation between water-column DIN:TP ratios (and DIN:PO_4^{3-} ratios) and abundance of the clam, *Corbula*…there is also a strong long-term positive relationship between pH and *Corbula* abundance." They explain,

> *Changes in external nutrient loads can drive changes in internal ecosystem biogeochemistry and, in turn, trophodynamics. This analysis suggests that increasing*

> *dominance over time of macrophytes, clams, and* Microcystis *along with more omnivorous fish that are fueled by a benthic food web, are not a result of stochastic events (random invasions) but, rather, are related to a cascade of changes in biogeochemistry resulting from changes in nutrient loading over time as a major driver. This analysis supports the premise that reductions in P loading from external sources drive aquatic systems toward increased importance of sediment dynamics, and toward the sediments as a major source of P. The food webs that are supported are different from those supported when the water column is the major source of P; they are benthic-dominated. Macrophytes such as* Egeria *and phytoplankton such as* Microcystis *are physiologically well adapted to these altered nutrient and pH regimes. The communities of bivalves and fish change accordingly.* (Glibert *et al.*, 2011, pp. 389-399)

As discussed previously, and in more detail in Glibert *et al.* (2011), numerous examples exist where nutrient reductions in other ecosystems has led to the restoration of native sea grasses and to declines in invasive bivalve populations.

REFERENCES

1. Ball, M.D. and J. F. Arthur. 1979. Planktonic chlorophyll dynamics in the Northern San Francisco Bay and Delta. Pacific Division of the American Association for the Advancement of Science c/o California Academy of Sciences Golden Gate Park San Francisco, California 94118.

2. Berg, G.M., P.M. Glibert, N.O.G. Jorgensen, M. Balode and I. Purina. 2001. Variability in inorganic and organic nitrogen uptake associated with riverine nutrient input in the Gulf of Riga, Baltic Sea. Estuaries, 24: 176-186

3. Berman, T and S. Chava, 1999. "Algal growth on organic compounds as nitrogen sources." Journal of Plankton Research, 21:1423-1437.

4. Brown, T. 2009. Phytoplankton community composition: The rise of the flagellates. IEP Newsletter. 22(3):20-28.

5. Cloern, J.E and R.T. Cheng. 1981. Simulation model of Skeletonema costatum population dynamics in Northern San Francisco Bay, California. Estuarine, Coastal and Shelf Science. 12:83-100

6. Cloern, J.E. and R. Dufford. 2005. Phytoplankton community ecology: principles applied in San Francisco Bay. Marine Ecology Progress Series, 285:11-28.

7. Cloern, J. E., A.D. Jassby, J. K. Thompson, and K. A. Hieb. 2007. A cold phase of the East Pacific triggers new phytoplankton blooms in San Francisco Bay. PNAS. 104(47): 18561–18565.

8. Collos, Y., M. Y. SIDDIQI, M. Y. WANG, A. D. M. GLASS, AND P. J.HARRISON. 1992. Nitrate uptake kinetics by two marine diatoms using the radioactive tracer 13N. J. Exp. Mar. Biol. Ecol.163: 251–260.

9. Collos, Y., A. VAQUER, B. BIBENT, G. SLAWYK, N. GARCIA, AND P. SOUCHU. 1997. Variability in nitrate uptake kinetics of phytoplankton communities in a Mediterranean coastal lagoon. Estuarine Coastal Shelf Sci. 44: 369–375.

10. Cummins, J., C. Buchanan, C. Haywood, H. Moltz, A.Griggs, R. C. Jones, R. Kraus, N. Hitt, and R. V. Bumgardner. 2010. Potomac Basin Large River Environmental Flow Needs. ICPRB Report 10- 3. Interstate Commission on the Potomac River Basin.

11. Dresler, P. V. and R. L. Cory. 1980. The Asiatic clam, Corbicula fluminea (Müller), in the tidal Potomac River, Maryland. Estuaries, 3: 150-152).

12. Dugdale, R.C., F. P. Wilkerson, V. E. Hogue and A. Marchi. 2007. The role of ammonium and nitrate in spring bloom development in San Francisco Bay. Estuarine, Coastal and Shelf Science, 73: 17-29.

13. Dugdale, R., F. Wilkerson, A. Parker. 2011. Brief report in response to selected issues raised by Sacramento Regional County Sanitation District in petition for review of discharge permit issued by the Central Valley Regional Water Quality Control Board. May 4, 2011.

14. Glibert, P., 2010. Long-term changes in nutrient loading and stoichiometry and their relationships with changes in the food web and dominant pelagic fish species in the San Francisco Estuary, California. Reviews in Fisheries Science. 18(2):211-232.

15. Glibert, P., C.A. Heil, D. Hollander, M. Revilla, A. Hoare, J. Alexander, S. Murasko. 2004. "Evidence for dissolved organic nitrogen and phosphorous uptake during a cyanobacterial bloom in Florida bay." Mar Ecol Prog Ser, 280:73-83.

16. Glibert, P.M., J. Harrison, C.A. Heil, and S. Seitzinger. 2006. Escalating worldwide use of urea – a global change contributing to coastal eutrophication. Biogeochem., 77: 441-463.

17. Glibert, P.M., D. Fullerton, J.M. Burkholder, J.C. Cornwell, and T.M. Kana. 2011. Ecological stoichiometry, biogeochemical cycling, invasive species, and aquatic food webs: San Francisco Estuary and comparative systems. Reviews in Fisheries Science, 19(4): 358-417.

18. Glibert, P., Univ. of Maryland. Personal communication.

19. Greening, H. and A.Janicki. 2006. Toward reversal of eutrophic conditions in a subtropical estuary: Water quality and seagrass response to nitrogen loading reductions in Tamp Bay, Florida, USA. Environ. Mgt. 38(2):163-178.

20. Ha, J.H., T. Hidaka, and H. Tsuno. 2009. Quantification of toxic Microcystis and evaluation of its dominance ratio in blooms using real-time PCR. Envir. Sci. Technol. 43: 812-818.

21. Harvey, H.W. 1953, Synthesis of Organic Nitrogen and Chlorophyll by Nitzschia Closterium. J. Mar. Biol. Res. Assoc. U.K. 31:477-487.

22. Heil, C.A., M. Revilla, P.M. Glibert and S. Murasko. 2007. Nutrient quality drives phytoplankton community composition on the West Florida Shelf. Limnology Oceanogr. 52: 1067-1078.

23. Hessen, D.O.. 1997. Stoichiometry in food webs – Lotka revisted. Oikos 79: 195-200.

24. Hodgkiss, I.J. and K.C. Ho. 1997. Are changes in N:P ratios in coastal waters the key to increased ref tide blooms?. Hydrobiologia. 352:141-147.

25. Hodgkiss, I.J. 2001. The N:P ratio revisited. In: K.C. Ho and Z.D. Wang (Eds.), Prevention and Management of Harmful Algal Blooms in the South China Sea. School of Science and Technology, Open University of Hong Kong.

26. Ibanez, C., N. Prat, C. Duran, M. Pardos, A. Munne, R. Andreu, N. Caiola, N. Cid, H. Hampel, R. Sanchez, and R. Trobajo. 2008. Changes in dissolved nutrients in the lower Ebro river: Causes and consequences. Limnetica. 27(1):131-142.

27. Jassby, A.D., J.E. Cloern and B.E. Cole. 2002. Annual primary production: Patterns and mechanisms of change in a nutrient-rich tidal ecosystem. Limnol. Oceanogr., 47(3): 698–712.

28. Jassby, A., "Phytoplankton in the Upper San Francisco Estuary: recent biomass trends, their causes, and their trophic significance." San Francisco Estuary and Watershed Science. 6(1): Article 2, February 2008.

29. Jaworski, N., W. Romano, and C. Buchanan. 2007. The Potomac River Basin and its estuary: Landscape loadings and water quality trends 1895-2005.

30. Kendall, C. 2010. Use of stable isotopes for evaluating environmental conditions associated with Microcystis blooms in the Delta. Oral Presentation at the 6th Biennial Bay-Delta Science Conference, Sacramento, CA, September 27-29, 2010.

31. Kimmerer, W.J. 2002 Physical, biological, and management responses to variable freshwater flow into the San Francisco Estuary. Estuaries. 25(6):1275-1290.

32. Kimmerer, W.J., E.S. Gross, and M.L. MacWilliams. 2009. Is the Response of Estuarine Nekton to Freshwater Flow in the San Francisco Estuary explained by Variation in Habitat Volume? Estuaries and Coasts, DOI 10.1007/s12237-008-9124-x.

33. Lam, C. W. Y. and K. C. Ho. 1989. Red tides in Tolo Harbour, Hong Kong, p. 49–52. In T. Okaichi, D. M.Anderson, and T. Nemoto (eds.), Red Tides: Biology, Environmental Science and Toxicology. Elsevier, New York.

34. Lehman, P. W. 2000. The influence of climate on phytoplankton community biomass in San Francisco Bay Estuary. Limnol. Oceanogr. 45: 580–590

35. Lehman, P. W., G. Boyer, C. Hall, S. Waller and K. Gehrts. 2005. Distribution and toxicity of a new colonial Microcystis aeruginosa bloom in the San Francisco Bay Estuary, California. Hydrobiologia, 541:87-99

36. Lehman, P.W., S.J. Teh, G.L. Boyer, M.L. Nobriga, E. Bass, and C. Hogle. 2010. Initial impacts of Microcystis aeruginosa blooms on the aquatic food web in the San Francisco Estuary. Hydrobiologia, 637:229-248

37. Lomas, M.W. and P.M. Glibert. 1999a. Interactions between NH+4 and NO3 uptake and assimilation: comparison of diatoms and dinoflagellates at several growth temperatures. Marine Biology, 133:541-551.

38. Lomas, M.W. and P.M. Glibert. 1999b. Temperature regulation of nitrate uptake: A novel hypothesis about nitrate uptake and reduction in cool-water diatoms. Limnol Oceanogr, 44:556-572.

39. Ludwig, C.A. 1938. The availability of different forms of nitrogen to a green alga (Chlorella) Am.J.Bot. 25:448-458.

40. MacIsaac, J.J. and R.C. Dugdale , 1969. The kinetics of nitrate and ammonium uptake by natural populations of marine phytoplankton. Deep-Sea Res. 16:45-67.

41. MacIsaac, J.J. and R.C. Dugdale, 1972. Interactions of light and inorganic nitrogen controlling nitrogen uptake in the sea. Deep-Sea Res. 19:209-232.

42. McCarthy, J.J., W.R. Taylor and J.L. Taft, 1975. The dynamics of nitrogen and phosphorous cycling in the open water of the Chesapeake Way. In: T.M. Church (ed.) Marine Chemistry in the Coastal Environment. American Chemical Society Symposium Series 18. Washington D.C., pp. 664-681.

43. Meyer, J.S., P.J. Mulholland, H.W. Paerl, and A.K. Ward. 2009. "A framework for research addressing the role of ammonia/ammonium in the Sacramento-San Joaquin Delta and the San Francisco Bay Estuary ecosystem." Report to CalFed Science Program.

44. Müller-Solger, A., A.D. Jassby and D.C. Müller-Navarra. 2002. Nutritional quality of food resources for zooplankton (Daphnia) in a tidal freshwater system (Sacramento-San Joaquin River Delta). Limnol Oceanogr, 47(5):1468-1476.

45. Nobriga, M.L., F. Feyrer, R.D. Baxter, and M. Chotkowski. 2005. Fish community ecology in an altered river delta: spatial patterns in species composition, life history strategies, and biomass. Estuaries, 28(5):776-785.

46. Oh, H-M., S.J. Lee, M-H. Jang and B-D. Yoon. 2000. Microcystin production by Microcystis aeruginosa in a phosphorus-limited chemostat. Appl. Envir. Microbiol. 66: 176-179.

47. Parker, A.E., R.C. Dugdale, and F.P. Wilkerson. (in review). Biochemical processing of anthropogenic ammonium in river and estuarine water columns. Marine Pollution Bulletin.

48. Radach, G., J. Berg, and E. Hagameir. 1990. Long-term changes of the annual cycles of meteorological, hydrographic nutrient and phytoplankton time series at Helgoland and at LV Elbe 1 in the German Bight. Continental Shelf Research 10:305– 328.

49. Rask, N., S. E. Pedersen, and M. H. Jensen. 1999. Response to lowered nutrient discharges in the coastal waters around the island of Funen, Denmark. Hydrobiologia 393: 69–81

50. Romdhane, M.S., H.C. Eilertsen, O.K.D. Yahia, and Y.N.D. Daly. 1998. Toxic dinoflagellate blooms in Tunisian lagoons: causes and consequences for aquaculture. In: Harmful Algae Edited by B.Reguera, J.Blanco, M.L.Fern'andez & T.Wyatt, Xunta de Galicia and Intergovernmental Oceanographic Commission of UNESCO, pp. 80–83

51. Ruhl, H.A. and N.B. Rybicki. 2010. Long-term reductions in anthropogenic nutrients link to improvements in Chesapeake Bay habitat. www.pnas.org/cgi/doi/10.1073/pnas.1003590107.

52. Schindler, D. W. 1974. Eutrophication and Recovery in Experimental Lakes: Implications for Lake Management. Science. 184(4139):897-899.

53. Slaughter, A. and W. Kimmerer. 2010. Abundance, composition, feeding, and reproductive rates of key copepodsspecies in the food-limited Low Salinity Zone of the San Francisco Estuary. Poster Presentation at the 6th Biennial Bay-Delta Science Conference, Sacramento, CA, September 27-29, 2010.

54. Solomon, C.M., J.L. Collier, G.M. Berg and P.M. Glibert. 2010. Role of urea in microbial metabolism in aquatic systems: a biochemical and molecular review. Aquatic Microbial Ecology, 59: 67-88.

55. Sommer, T., C. Armor, R. Baxter, R. Breuer, L. Brown, M. Chotkowski, S. Culberson, F. Feyrer, M. Gingras, B. Herbold, W. Kimmerer, A. Mueller-Solger, M. Nobriga and K.

Souza. 2007. The collapse of pelagic fishes in the upper San Francisco Estuary. Fisheries, 32(6):270-277.

56. Sterner, R.W. and J.J. Elser. 2002. Ecological stoichiometry: The biology of elements from molecules to the biosphere. Princeton University Press, Princeton, N.J.

57. Teh, S., I. Flores, M. Kawaguchi, S. Lesmeister and C. Teh. 2001. Full Life-Cycle Bioassay Approach to Assess Chronic Exposure of *Pseudodiaptomus forbesi* to Ammonia/Ammonium. Report to Chris Foe and Mark Gowdy at the State Water Board.

58. Wilkerson, F.P, R.C. Dugdale, V.E. Hogue and A. Marchi. 2006. Phytoplankton blooms and nitrogen productivity in San Francisco Bay. Estuaries and Coasts, 29(3): 401–416.

59. Winder, M. and A.D. Jassby. 2010. Shifts in zooplankton community structure: Implications for food web processes in the Upper San Francisco Estuary. Estuaries and Coasts. DOI 10.1007/s12237-010-9342-x

Exhibit 1. This exhibit presents an evaluation of Central Contra Costa Sanitary District (CCCSD) receiving water data in the vicinity of the CCCSD wastewater treatment plant discharge for the 2006-2010 time period. The frequency of observations of receiving water ammonium concentrations in specified ranges is presented in Figure A. An evaluation of the percent of receiving water samples that exceed ammonium concentrations known to cause impacts to diatoms and zooplankton are presented in Table A.

Figure A

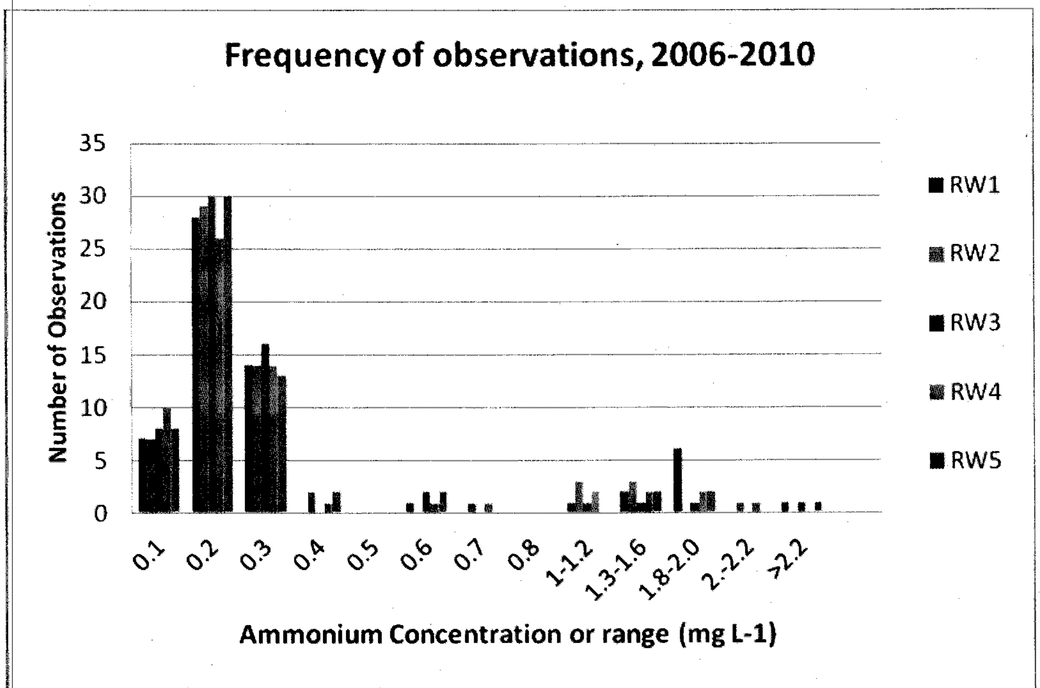

Table A

Criterion	**Ammonium level**	**Percent of samples that exceeded this level from 2006-2010 in CCCSD Receiving Water samples**
Dugdale inhibition of diatoms	0.06 mg L-1 or 4 µMol-N	87%
Swee Teh toxicity to zooplankton	0.36 mg L-1 or 25.7 µMol-N	15%
10-x greater than the Dugdale inhibition level	0.6 mg L-1 or 40 µMol-N	14%

Exhibit 2. This exhibit presents historical ammonium concentration data collected from 1975 to 2010 by the Environmental Monitoring Program of the Interagency Ecological Program for the Bay-Delta Estuary at a monitoring location in western Suisun Bay near Martinez (monitoring Station D6). The ammonium concentration of 0.056 mg L^{-1} (equivalent to 4μmol L^{-1}) is indicated on the graph. This ammonium concentration has been found to inhibit nitrogen uptake by diatoms and contribute to reduced diatom production in the Bay-Delta estuary.

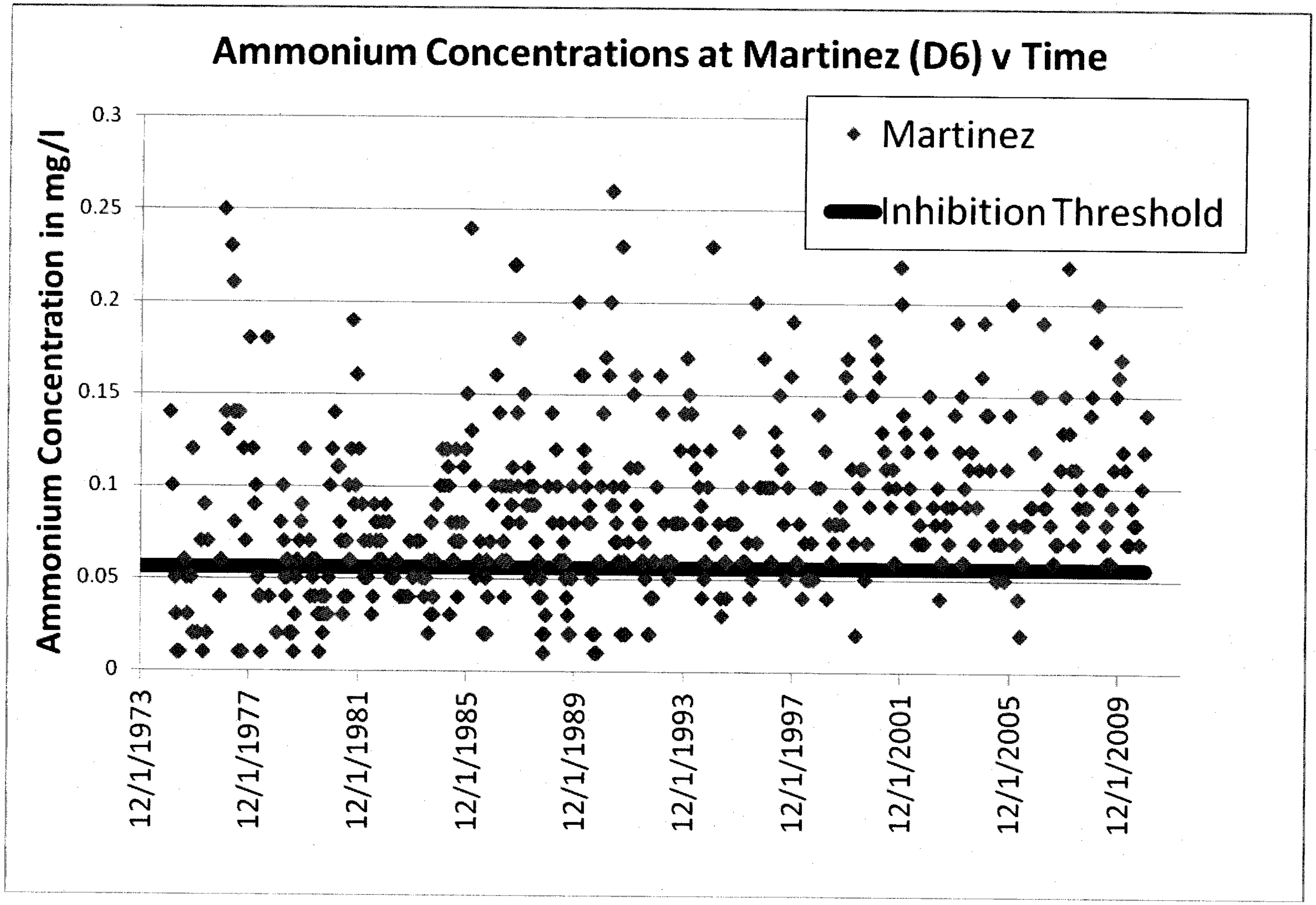

EXHIBIT E

December 8, 2011

Mr. Vince Christian
California Regional Water Quality Control Board
San Francisco Bay Region
1515 Clay Street, Suite 1400
Oakland, CA 94612

RE: Tentative Order for the Central Contra Costa Sanitary District, No. R2-2011-XXXX, NPDES Permit No. CA0037648.

Dear Mr. Christian,

The Bay Area Clean Water Agencies (BACWA) submits these comments on the tentative order (TO) for the Central Contra Costa Sanitary District (CCCSD) National Pollutant Discharge Elimination System (NPDES) Permit (Order No. R2-2011-XXXX, NPDES Permit No. CA0037648). BACWA is a joint powers agency, formed under California Government Code section 6500 *et seq.*; our members own and operate publicly-owned treatment works (POTWs) and sanitary sewer systems that provide sanitary services to over 6.5 million people in the nine-county San Francisco Bay Area. BACWA members are public agencies, governed by elected officials and managed by professionals charged with protecting the environment and public health.

BACWA recognizes that the formal comment period for this TO closed on November 1, but respectfully request that these comments be entered into the record pursuant to Title 23 of California Code of Regulations, section 648.1(d). The TO publicly noticed by the San Francisco Bay Regional Water Quality Control Board (Regional Water Board) did not raise issues warranting submittal of comments on behalf of BACWA and our member agencies. The District and BACWA were supportive of contents of the noticed TO and the direction of collaborative efforts underway with the Regional Water Board to better understand nutrient science and management needs. After the close of the comment period, however, BACWA became aware of comments submitted by the San Luis & Delta-Mendota Water Authority and the State Water Contractors (Water Agencies) that warrant our response because the issues raised have potentially significant implications for Bay Area POTWs.[1] We understand that the Regional Water Board may be considering imposing additional requirements on BACWA members as a result of the Water Agencies' comments. This turn of events concerns us greatly; we strongly recommend our comments be taken under consideration and that the TO be adopted as originally issued.

Specifically, the comments submitted by the Water Agencies cite recent studies on the potential impacts of ammonia discharges on Suisun Bay water quality. The letters request that, based on these studies, the District be required to nitrify their effluent or, in the alternative, undertake a very substantial research effort to determine whether the District is harming beneficial uses. The assertions made by the Water Agencies, if considered in discharge permitting, could have an impact on multiple BACWA members because some of the studies relied upon suggest that fully treated municipal wastewater is fundamentally

[1] Letter from the San Luis & Delta-Mendota Water Authority and the State Water Contractors regarding the Tentative NPDES Permit Renewal for the Central Contra Costa Sanitary District Wastewater Treatment Plant, dated October 31, 2011.

EXHIBIT E

affecting the Bay-Delta food web. These assertions, however, are based on limited study results that are either preliminary or that other experts have questioned because a clear cause and effect relationship has not been demonstrated. Scientific peer review of these studies has not been completed, nor has a rulemaking process been undertaken to adopt revised state, or site specific, water quality standards. It is therefore premature to consider these studies as the basis for any changes to the TO at this time. Our comments identify some of the limitations of the research conducted to date and illustrate the complexity of both the Bay-Delta system and the scientific questions raised.

As is described in more detail below, BACWA and our member agencies have taken a proactive and collaborative approach to working with the Regional Water Board in understanding the possible effects of ammonium and other nutrients in San Francisco Bay. We understand the Regional Water Board's need to develop sound nutrient criteria and an implementation strategy. We emphasize, though, that this should be done through a joint fact-finding process with broad stakeholder involvement in order to minimize wasting resources through litigation and adversarial science. We hope that the Regional Water Board will carefully review the evidence provided and continue to work with BACWA, regional scientists, and other stakeholders in a coordinated and transparent way.

I. The District and BACWA Are Committed to Resolving Existing Uncertainties about Nutrient Impacts to San Francisco Bay Beneficial Uses.

The question of whether nutrient discharges from POTWs have the potential to impact beneficial uses is of paramount importance to BACWA and our members. The public agencies that own and operate POTWs must balance their mandate to protect environmental and public health with their responsibility to their communities to ensure that these services are reasonable and affordable.

No simple "end of pipe" or pollution prevention nutrient removal options exist for POTWs. Removal of ammonia from municipal wastewater is typically done biologically by nitrification, wherein ammonia (NH_3/NH_4^+) is oxidized to nitrite (NO_2^-), and then nitrite is oxidized to nitrate (NO_3^-). This oxygen intensive process requires capital improvements to allow for greater aeration and increased solids retention time, as well as chemical feed to provide alkalinity to support nitrification and to accommodate increased solids loading on the clarifiers. These changes to the treatment process are not only financially costly, but have significant environmental implications in terms of energy consumption and greenhouse gas emissions.

While the expense of these infrastructure and operational changes will depend on a particular agency's current treatment train and land availability, they are very significant. Such changes cannot and should not be undertaken without robust evidence that they are necessary and will provide benefits to the Bay commensurate with the economic and environmental costs. For these reasons, BACWA is committed to ensuring that the potential impacts of nutrient discharges are well understood and documented, and that any possible control measures implemented will result in the desired improvements to water quality.

BACWA has expressed this commitment through our support of the multiple nutrient-related initiatives underway in the region. We understand the State Water Board and Regional Water Boards' need to establish a nutrient monitoring program, better quantify loadings from various sources, develop load-response and other models, and develop Numeric Nutrient Endpoints (NNEs). BACWA has collaborated with Regional Water Board staff to further these efforts by providing funding to the San Francisco Estuary Institute (SFEI) for the June 29, 2011 Nutrient Science and Management Workshop and to synthesize nutrient loading data in the South San Francisco Bay. Additionally, our member agencies have and will continue to provide nutrient effluent data above and beyond current monitoring requirements. We are also currently working with SFEI and Regional Water Board staff to determine how BACWA may provide longer-term financial support for efforts to more clearly identify and resolve key scientific and management questions related to potential nutrient impacts.

Our work also extends to the Suisun Bay Work Plan referenced in the Water Agencies' comment letter. We first became aware of this Surface Water Ambient Monitoring Program (SWAMP) funded effort in the fall of 2010 and shortly thereafter began participating in workgroup meetings. BACWA provided financial support for water chemistry analyses of samples; CCCSD provided in-kind services as well as funding to add a monitoring station and expand the analyte lists. Only one of the two years of work contemplated in the Work Plan has been completed. Data from this first year are not yet available and have not undergone any form of peer-review. BACWA hopes that, once the study concludes next year, the results of this and related projects will be presented in a public forum with opportunities for review and discourse.

II. Regulation Based on the Assertions in the Water Agencies' Comments is Inappropriate Because the Impacts of Ammonium on Suisun Bay are Controvertible.

In their comment letters on the District TO, the Water Agencies make numerous allegations regarding the certainty of the effect of the District's discharges of ammonium to Suisun Bay. These claims are inconsistent with the findings of the team of highly esteemed coastal estuarine experts charged with evaluating the impacts of nutrients, including ammonium, on the San Francisco Bay Estuary as part of the development of NNEs.[2] The Water Agencies also rely heavily on the results of a recently issued report by Dr. Swee Teh to allege the existence of ammonia toxicity in Suisun Bay.[3] Serious questions exist regarding the key findings of that report, which has not been independently peer reviewed, and the subsequent use of those findings to demonstrate toxicity associated with the District's discharge. The Water Agencies also assert that these discharges are disrupting the Delta food web by changing the nutrient balance in the estuary. Such an allegation pre-supposes knowledge regarding the impact of nutrients in the San Francisco Estuary which does not currently exist, and asserts information as fact which has clearly not yet been resolved by San Francisco Bay scientific experts.

In the following discussion, facts are presented regarding the current state of knowledge of the effects of ammonium, specifically, and nutrients, in general, on San Francisco Bay. This information illustrates that the requested changes to the draft permit are without merit and demonstrates that the appropriate action by the Regional Water Board at this time is the adoption of the draft permit as publicly noticed.

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[2] McKee, Lester; Sutula, Martha; Gilbreath, Alicia; Beagle, Julie; Gluchowski, David; Hunt, Jennifer; *Nutrient Numeric Endpoint Development for the San Francisco Bay Estuary: Literature Review and Data Gaps* (June 2011). (Hereinafter, McKee et al. 2011). Available at http://www.swrcb.ca.gov/rwqcb5/water_issues/delta_water_quality/ambient_ammonia_concentrations/tehetal_ammonium_exposure2011.pdf.

[3] Teh, Swee; Flores, Ida; Kawaguchi, Michelle; Lesmeister, Sarah; and The Ching; *Full Life-Cycle Bioassay Approach to Assess Chronic Exposure of* Pseudodiaptomus forbesi *to Ammonia/Ammonium*, University of California at Davis; submitted to the State Water Resources Control Board pursuant to Agreement No. 06-447-300 (August 2011). (Teh et al., 2011). Available at http://www.swrcb.ca.gov/rwqcb5/water_issues/delta_water_quality/ambient_ammonia_concentrations/tehetal_ammonium_exposure2011.pdf.

A. Response to the allegation that ammonium levels in Suisun Bay are toxic to copepods and are linked to the District discharge.

Comments by the Water Agencies claim that the District's discharge is contributing to the toxicity of two copepods: *Pseudodiapotomus forbesi* (*P. forbesi*) and *Eurytemora affinis*. This is based on the recent research of Dr. Swee Teh at the University of California at Davis on only one of those species, *P. forbesi* (hereinafter, Teh et al. study). The comments also allege that toxicity to these copepods is significantly impacting the food web that supports aquatic life in the Bay-Delta estuary.

1. *Serious questions exist regarding reliance on the results of Dr. Teh's research in making changes to the District permit.*

The report issued by Dr. Teh presents results from studies which are described by the author as a "developmental protocol with a non-standard species."[4] Given the unique and "developmental" nature of the test species and testing methods used, independent scientific review is needed to establish the validity of the study results. In fact, the report has not undergone formal peer review and has not been widely circulated within the scientific community to garner even informal peer review.

Review to date by other expert toxicologists has raised a serious question regarding the methodology used in data interpretation to derive the most significant results in the report. In addition, questions regarding the test methods and test results cast further doubt on the key conclusions contained in the report. These questions, enumerated below, highlight the need for independent peer review of the final report, and replication of the study itself, prior to applying in a regulatory context.

<u>Data Interpretation</u>

Independent analysis of the raw data for adult *P. forbesi* obtained from the 31-day life cycle testing performed in the Teh et al. study indicates that the test treatment at 0.36 mg/l is not statistically different from the control. This significantly changes a primary conclusion in the Teh et al. report, and essentially eliminates the concern about adult copepod toxicity in Suisun Bay. A revised chronic toxicity threshold therefore would be calculated as 0.53 mg/l (based on numbers of nauplii and juveniles in the 31-day reproduction test), rather than the reported 0.36 mg/l. As described in greater detail below, ambient levels of ammonium do not reach such threshold levels in Suisun Bay for periods long enough to cause chronic toxicity.

In addition to this significant concern, the following questions exist regarding test methods and test results. These issues warrant consideration by independent peer reviewers of the study to assess many aspects of this study, including even whether the 0.53 mg/l threshold value is valid.

<u>Test Methods</u>

A new test methodology was developed by Teh et al. to perform the study in question on *P. forbesi*. This raises inherent questions, owing to the lack of a track record for the method and the absence of a

[4] Letter from Dr. Swee Teh, University of California at Davis, to Ms. Linda Dorn, Environmental Program Manager for the Sacramento County Regional Sanitation District, dated August 31, 2011.

standardized methodology to rely upon. A few of the issues relating to these novel test methods are provided below.

- Due to budget constraints, the Teh et al. study did not include use of reference toxicity tests, which is a relatively standard procedure in toxicity testing to confirm the sensitivity of the test organisms. Dr. Teh has stated in writing that he would propose to use reference toxicity testing in future studies and that he agrees it would be appropriate to do so.[5]

- The Teh et al. study was performed at a nominal salinity of two parts per thousand (ppt), bringing into question the applicability of these results in more saline environments. At the District's discharge points, the salinity varies from several to twenty ppt depending on Delta outflows and climatic conditions. Dr. Teh agrees that future studies should address salinity/conductivity tolerance questions.[6]

- Teh et al. selected a control survival threshold of 80% by which to judge the acceptability of a given test. This control survival threshold is relatively low and raises the possibility that unhealthy or substandard test organisms could have biased the results.

- Teh et al. used the initial measured concentrations of ammonium as the basis for reporting test results. Ammonium levels can increase over the test period. This approach, therefore, tends to overstate the toxic effect of a particular concentration because of the possibility that the test organisms were, in fact, exposed to higher concentrations than those reported.

<u>Test Results</u>

A major concern with the chronic toxicity test results summarized in the report is the high variability between replicates at the same concentration. Such variability would arguably be the basis for invalidation of the results. Teh et al., however, have relied on these data, without qualification, in concluding that an ammonium concentration of 0.36 mg/l was chronically toxic to *P. forbesi*.

2. *Information regarding the increasing abundance of these copepods (P. forbesi) in Suisun Bay contradicts the allegations made and highlights the uncertainties in our understanding of the Delta food web.*

Recent publications provide information that contradicts allegations and inferences contained in the Water Agencies' comment letter regarding the impact of the District discharge on copepod abundance. For example, the Dr. Teh et al. report notes that the California Department of Fish and Game 2007 to 2009 20 mm survey for *P. forbesi* found that the abundance at station 711 (near Rio Vista) increased, despite the presence of higher levels of ammonium at this location than exist in Suisun Bay (mean ammonium concentration of 0.27 mg/l versus mean ammonium concentration 0.15 mg/l at Martinez (Station 405)).

Additionally, the Interagency Ecological Program (IEP)'s Spring 2009 newsletter reported that *P. forbesi*, an introduced species first detected in 1988, "…has declined slightly since its introduction, [but] has

[5] *Id* at 3.
[6] *Id* at 6.

remained relatively abundant in summer and fall compared to other copepods."[7] The newsletter further noted that "[s]ummer abundance also increased slightly from 2007 to 2008, while fall abundance increased moderately and was the highest since 2002."[8] This evidence of increasing abundance of *P. forbesi* in Suisun Bay, despite the increased ammonia loadings and the increased ammonia ambient concentrations which are acknowledged for this period, is inconsistent with the allegation that ammonium toxicity is negatively impacting the abundance of this copepod in the Bay-Delta.

3. *The allegation that toxicity is occurring in Suisun Bay is based on an erroneous interpretation of available ambient ammonium data in Suisun Bay*.

The Water Agencies have used a data set that is not representative of ambient conditions in Suisun Bay to allege the existence of ammonium toxicity. Additionally, the methods used by the Water Agencies to assess the existence of chronic toxicity near the District's discharge are flawed and do not support the allegations made regarding impacts to copepod species or the Delta food web.

The United States Geological Survey (USGS) data set at Station 8 (IEP Station D6), in the vicinity of the CCCSD outfall, indicates that ambient ammonia levels through 2010 never exceeded 0.25 mg/l. According to data collected from 2006 through 2009 by the University of California at Davis, mean ammonia-N concentrations at Martinez (Station 405), when ammonium discharges to the Bay-Delta system were the highest on record, were only 0.15 mg/L ± 0.01.[9] The mean plus two standard deviation (SD) values (the approximate 95th percentile value) of the Davis data was only 0.35 mg/l. The NNE Literature Review reports that the mean ammonium concentrations in Suisun Bay from 1999 through the present were even lower: 0.11 mg/l.[10]

Some of the data cited in the Water Agencies' comment letter was obtained at the CCCSD outfall, in the immediate vicinity of the discharge plume, during periods of slack tide when mixing was at a minimum. Over the course of the normal tidal cycle, the mixing and advection that occurs at the District's outfall rapidly disperses the plume and reduces such concentrations to ambient levels. The high values cited by Water Agencies as problematic were localized, short term (instantaneous) data values which are not appropriate measures of chronic toxicity, and are certainly not representative of ambient conditions in Suisun Bay. Moreover, the preponderance of the values observed at the CCCSD outfall was less than the District's laboratory detection limit of 0.3 mg/l.

These data suggest that the levels of ammonium in Suisun Bay, including those in the vicinity of CCCSD's discharge, do not reach threshold values over averaging periods needed to create chronic toxicity to *P. forbesi*, including either the 0.53 mg/l threshold value described above that is based on a revised interpretation of the Teh et al results, or the Teh et al. value of 0.36 mg/l cited in the Water Agencies' letter.

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[7] Interagency Ecological Program Newsletter, Vol 22., No. 2(Spring 2009), p. 11 (available at http://www.water.ca.gov/iep/newsletters/2009/IEPNewsletter_FINALSpring2009.pdf).
[8] *Id.*
[9] Werner, Inge; Markiewicz, Dan; Deanovic, Linda; Connon, Richard; Beggel, Sebastian; Teh, Swee; Stillway, Marie; Reece; Charissa; *Pelagic Organism Decline (POD): Acute and Chronic Invertebrate and Fish Toxicity Testing in the Sacramento-San Joaquin Delta 2008-2010, Final Report*; submitted to the California Department of Water Resources by the University of California at Davis Aquatic Toxicology Laboratory (July 24, 2010), p. 20. Available at http://science.calwater.ca.gov/pdf/workshops/POD/Werner%20et%20al_2010_POD2008-2010_Final%20Report.pdf.
[10] McKee et al. 2011 at 148.

B. Response to allegation that ammonium linked to the District discharges is inhibiting diatom primary production in Suisun Bay.

The Water Agencies also state that the District is contributing to ammonium levels in Suisun Bay that exceed the "Dugdale threshold" and that these levels are causing a significant disruption in the Bay-Delta food web due to the inhibition of spring blooms of phytoplankton in Suisun Bay.

1. The best available scientific understanding of SF Bay experts is that the importance of the suggested "inhibition effect" is uncertain.

In the June 2011 report prepared for the Regional Water Board by SFEI and the Southern California Coastal Water Research Project (SCCWRP), numerous statements are made which contradict the assertion that ammonium is commonly accepted as having a significant impact in San Francisco Bay. The report, *Nutrient Numeric Endpoint Development for the San Francisco Bay Estuary: Literature Review and Data Gaps Analysis,* acknowledges the suggestion by Dr. Richard Dugdale and other researchers from the Romburg Tiburon Center that "ammonium inhibition could be one of the limiting factors that control primary productivity in the Bay."[11] However, the report goes on to state that the impacts of ammonium on diatom blooms is not well-understood, is just one of many factors known to affect productivity, and that additional work is needed to resolve this issue:

> "…the ecological importance of ammonium inhibition of spring diatom blooms is not well understood relative to factors known to control primary productivity…"[12]
>
> "In SF Bay, the biomass associated with phytoplankton, measured as surface water chlorophyll *a* concentration, varies in space and time in response to nutrient availability from external loads and internal regeneration, grazing, stratification, water temperature, tidal energy, transparency, wind/wave energy, the availability of seed cysts, UV radiation effects on nitrate versus ammonium assimilation perhaps due to disruptions of enzyme pathways, differential uptake of nitrate and ammonium by larger versus smaller cells, inhibition of nitrate uptake by ammonium, predation by benthic invertebrates, and variations in the phase of the Pacific Decadal Oscillation and related changes to top down predation of benthic invertebrates."[13]
>
> "…the effect of ammonium inhibition on phytoplankton productivity throughout the Bay has not been modeled vis-a-vis other contributing factors…the next logical step is to develop models that synthesize understanding of the relative importance of ammonium and urea versus other factors controlling phytoplankton assemblages."[14]
>
> "Elevated ammonium concentrations have been suggested as a major mechanism by which spring diatom blooms appear to be suppressed in the North Bay and Lower Sacramento River…Despite this evidence, the ecological importance of ammonium inhibition of spring diatom blooms is not well understood relative to factors known to control primary productivity, particularly in other

[11] *Id* at 147.
[12] *Id* at 153.
[13] *Id* at 46 (internal citations omitted).
[14] *Id* at 154.

> regions of the Bay where water column chlorophyll *a* appears to be increasing. Thus, *the linkage between ammonium concentrations and Bay beneficial uses is not at this time universally accepted*. San Francisco Bay Technical Advisory Team (TAT) members agree that additional data synthesis is required to better understand the role of ammonium in SF Bay."[15]

It is important to note that members of the TAT responsible for scientific review of and input on the NNE document include James Cloern, a highly recognized expert in San Francisco Bay ecology and two members from the Romburg Tiburon Center, including Dr. Dugdale. The cited statements and recommendations of the NNE report should therefore be interpreted as current prevailing scientific opinion.

2. *The State Water Contractors are participating in Suisun Bay studies to address whether ammonium is inhibiting spring phytoplankton blooms*

Corroboration that the effect of ammonium in San Francisco Bay is "unsettled science" is reflected in the fact that studies are ongoing to determine the role of various factors, including ammonium, on the frequency and magnitude of phytoplankton blooms in Suisun Bay. One of the comment letter authors, the State Water Contractors, are participants in these studies and are therefore well aware of the ongoing uncertainties that exist regarding the validity and significance of the "Dugdale effect." The Final SWAMP Work plan for FY 2010-2011 and 2011-2012 for Monitoring Spring Phytoplankton Bloom Progression in Suisun Bay explicitly states that "[t]he main purpose of this study is to…determine if there is inhibition, and, if so, to determine what is causing the inhibition."[16] *It would be premature to impose the NPDES permit requirements requested by the Water Agencies given the uncertainties regarding the existence or importance of the effect of ammonium in Suisun Bay and the fact that studies are currently ongoing to reduce these uncertainties.*

C. Response to allegation that the District nutrient loadings are changing nutrient ratios in Suisun Bay, resulting in a harmful shift in algal communities and other adverse ecological impacts.

The Water Agencies also suggest that research by Dr. Patricia Glibert confirms that nutrient loadings from the District contribute to changes in nutrient ratios in Suisun Bay, and that those changed ratios explain adverse ecosystem changes in the Bay-Delta, including the precipitous decline of key fish species.[17] In fact, the cited work has not been accepted or endorsed by leading Bay-Delta scientists. For example, the San Francisco Bay NNE science team considered Dr. Glibert's 2010 paper, but neither endorsed it or adopted it as fact in the final report.

It should also be noted that the work by Glibert in 2010, funded by the State Water Contractors, was criticized for its inappropriate use of statistical methods and other issues. In a peer-reviewed paper titled "Perils of Correlating CUSUM-transformed variables to infer ecological relationships (Breton et al..

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[15] *Id* at 155 (emphasis added).
[16] Final SWAMP Workplan at 1.
[17] Glibert, Patricia; *Long-Term Changes in Nutrient Loading and Stoichiometry and Their Relationships with Changes in the Food Web and Dominant Pelagic Fish Species in the San Francisco Estuary, California*; Reviews in Fisheries Science, Vol. 18, Issue 2 (August 2010). (Hereinafter, Glibert et al., 2010). Available at http://www.sfcwa.org/2011/05/20/sed-lobortis-tellus-vel-ligula-pretium-mollis/.

2006, Glibert 2010)"[18] authors James Cloern, Alan Jassby, Jacob Carstense, William Bennett, Wim Kimmerer, Ralph MacNally, David Schoellhamer and Monika Winder stated the following:

- "Glibert (2010) concluded that recent large population declines of diatoms, copepods, and several species of fish were responses to a single factor – increased ammonium inputs from a municipal wastewater treatment plant."
- "Glibert's study…contradicts the overwhelming weight of evidence that population collapses of native fish…and their supporting food webs in the San Francisco Estuary are responses to multiple stressors including landscape change, water diversions, introductions of exotic species and changing turbidity."
- "…CUSUM transformation, as used by…Glibert (2010), violates the assumptions underlying regression techniques."
- "…CUSUM-transformed variables often have an apparent statistically significant correlation even when none exists…"
- "…Glibert (2010) inferred a strong negative association between delta smelt abundance and wastewater ammonium from regression of CUSUM-transformed time series. However, the…correlation… is not significant…"

The Glibert 2010 work was also criticized as being incomplete for not having analyzed the importance of other factors, including export volumes, benthic grazing by invasive clams, major changes in the hydrologic regime in the Delta, and other stressors that are commonly recognized as major contributors to stress on the Delta ecosystem.

The recently released Glibert et al. 2011 paper - funded in part by the State Water Contractors, the San Luis & Delta-Mendota Water Authority and Metropolitan Water District - has not yet been effectively scrutinized by the San Francisco Bay NNE science team or other Bay-Delta experts. On its face, the subject paper is not a definitive piece of work on the effect of nutrients on the Bay-Delta ecosystem. The paper instead offers ecological stoichiometric theory as a hypothetical framework for consideration and suggests that nutrient stoichiometry may be a significant driver influencing food webs in the Bay-Delta ecosystem. The paper asserts the potential validity of this theory based on extensive, albeit selective, correlation analysis. The paper relies, at least in part, on the statistical analysis from the Glibert et al. 2010 paper that was so roundly criticized. The paper does not assert that it has developed conclusive scientific evidence for its theories applicable to the San Francisco Bay or Delta.

In fact, conclusory excerpts from the Glibert et al. 2011 paper state that "while compelling, the ecological stoichiometric model raises many questions that need further analysis in the San Francisco Estuary…" and "…regulation of the food web by nutrient controls is directly testable…there is much that needs to be explored to test these relationships directly."[19]

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[18] Cloern, J.E., A.D. Jassby, J. Carstensen, W.A. Bennett, W. Kimmerer, R. Mac Nally, D.H. Schoellhamer and M. Winder. 2011. *Perils of correlating CUSUM-transformed variables to infer ecological relationships (Breton et al. 2006, Glibert 2010)*. Limnology and Oceanography, in press.
[19] Glibert et al., 2011, at 84.

In summary, the cited papers by Glibert offer theories that are strongly supported by the Water Agencies but that have not been accepted or endorsed by the Bay-Delta scientific community, the Delta Science Program or any other reputable scientific body. This theory, while interesting and perhaps worthy of further exploration, is not an appropriate basis for the imposition of very costly changes to municipal wastewater management in the San Francisco Bay region.

III. The Regional Water Board Should Adopt the TO Without Changes.

It is a widely acknowledged that that San Francisco Bay-Delta is a complex ecosystem affected by myriad natural and anthropogenic factors. This is clearly evidenced by the wide range of factors that have been identified as potentially contributing to the decline in populations of Bay-Delta pelagic fish, including: Delta flows, turbidity, water diversions, habitat loss, introduced species, salinity, contaminants (including ammonium), and large-scale climatic shifts. Teasing out the relative effects of each of these factors has been and will continue to be challenging and require extensive resources.

As described above, the evidence relied upon by the Water Agencies' in their comment letter is far from conclusive. It is unreasonable and inappropriate at this time, therefore, to impose new permit requirements on the District. The imposition of nutrient limits based on nitrification is of great concern not just because of the inconclusiveness of the research done to date, but because this action would essentially circumvent the collaborative NNE process currently underway. The purpose of the NNE process is to develop nutrient water quality objectives. This rulemaking process is a transparent one with opportunity for stakeholder review and input. In contrast, this permit adoption process is an adjudicatory one with limited stakeholder involvement and little time for review of the bases for the requirements. Imposing limits in this permit, however, would have the effect of setting new de facto water quality objectives for ammonium in Suisun Bay.

We strongly urge the Regional Water Board to adopt the TO without any additional changes. BACWA will continue to work with staff, regional scientists, public agencies and the private sector to identify and fill data gaps related to this and other nutrient issues. The proper mechanism for resolving these scientifically, politically, and socio-economically difficult issues is a joint fact-finding process with transparency and broad stakeholder involvement that results in the establishment of water quality objectives that can then be implemented via permit and other management measures. This approach will reduce the likelihood of litigation and adversarial science and ensure that management options, should they become necessary, are carefully considered and well supported by science that is accepted by independent Bay-Delta experts.

Sincerely,

Amy Chastain

Amy Chastain
Executive Director

Enclosed:

Letter from Dr. Swee Teh, University of California at Davis, to Ms. Linda Dorn, Environmental Program Manager for the Sacramento County Regional Sanitation District, dated August 31, 2011.

Glibert, Patricia; *Long-Term Changes in Nutrient Loading and Stoichiometry and Their Relationships with Changes in the Food Web and Dominant Pelagic Fish Species in the San Francisco Estuary, California*; Reviews in Fisheries Science, Vol. 18, Issue 2 (August 2010).

Cloern, J.E., A.D. Jassby, J. Carstensen, W.A. Bennett, W. Kimmerer, R. Mac Nally, D.H. Schoellhamer and M. Winder. 2011. *Perils of correlating CUSUM-transformed variables to infer ecological relationships (Breton et al. 2006, Glibert 2010)*. Limnology and Oceanography, in press.

EXHIBIT F

California Regional Water Quality Control Board
San Francisco Bay Region

RESPONSE TO WRITTEN COMMENTS
on October 2011 Tentative Order for
Central Contra Costa Sanitary District Wastewater Treatment Plant
5019 Imhoff Place, Martinez, Contra Costa County

The Regional Water Board received written comments from the following parties on a tentative order distributed in October 2011 for public comment:

1. Central Contra Cost Sanitary District
2. San Luis & Delta-Mendota Water Authority and State Water Contractors
3. San Francisco Baykeeper

This response to their comments summarizes each comment in *italics* followed by the Regional Water Board staff response. For the full content and context of each comment, refer to the comment letters.

In addition, we identify below staff-initiated changes to the tentative order. These changes modify Attachment G, Regional Standard Provisions, and Monitoring and Reporting Requirements, to be consistent with current circumstances.

CENTRAL CONTRA COSTA SANITARY DISTRICT

District Comment 1: ***The District requests several revisions to the facility information for accuracy and clarity.*** *The District requests several specific revisions to the tentative order in underline/strikeout format to improve accuracy and clarity.*

Response: We agree and revised the tentative order to reflect them.

District Comment 2: ***The District requests clarification of the minimum dilution requirements.*** *The District requests revisions to Prohibition III.B and Fact Sheet sections IV.A.2, section IV.A.4, and section IV.C.4.b regarding the minimum initial dilution requirement. The District thinks the revisions are more consistent with the prohibition's derivation and written so that the provision cannot be interpreted to apply under all possible conditions.*

Response: We agree and revised the tentative order. However, we did not revise it exactly as suggested. Instead, we revised Prohibition III.B to clarify that the dilution ratio refers to the nominal dilution at the outfall and added the following sentence:

> Compliance shall be achieved by proper operation and maintenance of the discharge outfall to ensure that it (or its replacement, in whole or in part) is in good working

> order and is consistent with, or can achieve better mixing than, that described in the Fact Sheet (Attachment F). The Discharger shall address measures taken to ensure this in its application for permit reissuance.

This revision is consistent with the text we are now using in other permits under development. It requires that the outfall diffuser be maintained so as to ensure that the dilution assumptions underlying the permit's requirements remain valid. Accordingly, we revised Fact Sheet section IV.A.2 to explain that we used a dilution credit of 44:1 in the calculation of one or more water quality-based effluent limitations, based on available information about the dilution at the outfall, and that this prohibition is necessary to ensure that our assumptions remain valid. We also revised Fact Sheet section II.A.4 to add a more detailed description of the diffuser.

We revised Fact Sheet sections IV.A.4 and IV.C.4.b consistent with the District's suggestions.

District Comment 3: ***The District requests that the narrative chronic toxicity effluent limitation more accurately reflect the appropriate Basin Plan language.*** *The District requests that the narrative chronic toxicity effluent limit be revised because language is more stringent than what is required in the Basin Plan. The Basin Plan states that "there shall be no chronic toxicity in ambient waters." The language included in the tentative order indicated that there was to be no chronic toxicity in the effluent, which did not allow for any dilution at the outfall.*

Response: We agree and revised the tentative order to more accurately reflect the appropriate Basin Plan language.

District Comment 4: ***The District requests removal the requirement to measure pH, temperature, and ammonia concurrently in both effluent and receiving water.*** *The District requests removing Footnote 8 from Table E-3, which had required that ammonia samples be collected concurrently with effluent and receiving water monitoring for temperature and pH. The Regional Monitoring Program (RMP) is responsible for receiving water monitoring, and it is impractical to coordinate sampling timing between organizations. Also, the footnote is unnecessary because effluent pH and temperature are reported daily and will certainly be available for the one day per month that the 24-hour composite ammonia sample is collected.*

Response: We agree and revised the tentative order.

District Comment 5: ***The District requests that detailed chronic toxicity test information be required to be retained on site, but not reported.*** *The District questions the value of providing the requested level of detail about chronic toxicity tests (in addition to the results) in PDF format with electronic self-monitoring reports and suggests that it would be sufficient to instead retain those records onsite and available for review.*

Response: We disagree. We review this background information regarding the chronic toxicity tests, including the detailed information about how the tests were performed, to ensure permit compliance. Without this information, we cannot always understand the results in their appropriate context.

District Comment 6: ***The District requests Table E-4 be revised to eliminate the requirement to collect multiple grab samples for pretreatment monitoring and a reduction in monitoring frequencies for volatile organic compounds (VOCs) and base/neutrals and acids extractable organic compounds (BNAs).***

Table E-4 contains monitoring requirements for pretreatment and biosolids. The District requests that the Regional Water Board reduce the monitoring frequencies for VOCs and BNAs from quarterly to semi-annually based on its evaluation of historical data, which were rarely measured above detection levels.

The District also requests that it be allowed to continue collecting a single grab sample for certain constituents in place of multiple grab samples equally spaced over a 24-hour period. The District thinks that multiple grab samples would not provide any additional benefit, and it is not a practical use of staff resources.

Response: We agree and revised the tentative order.

District Comment 7: ***The District requests a correction to the rationale for including copper effluent limits.*** *The District notes that the justification for establishing copper effluent limitations was in error because the maximum effluent concentration (12 µg/L does) not exceed the governing water quality objective (14 µg/L).*

Response: We agree and revised the tentative order. Our revision cites the specific Basin Plan provision that requires the copper limit.

District Comments 8 and 9: ***The District requests revisions for clarity.*** *The District requests to eliminate the word "minimum" from the BOD and TSS removal requirements in Table 6 and to refer to Appendix A for a definition of "RP" in Provision VI.C.3.c(1).*

Response: We disagree. We retained the word "minimum" to clarify that BOD and TSS removal is a minimum limit, whereas all other limits in Table 6 are maxima. We did not refer to the definition of "RP" in Attachment A because doing so is unnecessary and, as it is, the wording of Provision VI.C.3.c(1) is taken directly from the State Implementation Policy.

District Comments 10 — 16: ***The District notes typographical errors.*** *The District noted error in Attachment B, MRP section VIII.C.1, Fact Sheet section IV.A.3, Table F-7, Fact Sheet section IV.C.4.c(4)(c), Fact Sheet section IV.C.4.c(4)(c), Fact Sheet section VI.E, and Fact Sheet section VI.E.*

Response: We revised the tentative order.

SAN LUIS & DELTA-MENDOTA WATER AUTHORITY AND STATE WATER CONTRACTORS (Water Agencies)

Water Agencies Introductory Comments: ***The Water Agencies request designated party status.*** *The Water Agencies request designated party status, claiming they have a direct interest in the tentative order. They also summarize the state of knowledge regarding potential ammonia impacts on Suisun Bay and offer three remedies they hope the Water Board will consider. These remedies are discussed further in Water Agencies Comments IV.A, IV.B, and IV.C.*

Response: We are not formally designating parties for this tentative order because doing so will not limit or enhance any party's rights under these proceedings. Designating parties is normally unnecessary for NPDES hearings; it is more common for enforcement hearings. A designated party has the right to submit evidence, is allowed to cross-examine during hearings, and is subject to the same time limits during hearings as other parties, such as the discharger (in contrast, "interested" parties can only make policy comments and cannot offer evidence). However, our standard NPDES hearing practices provide all these rights anyway, even without formal designation. The Water Agencies have had the same opportunity to submit written comments and evidence as all other parties, will be offered the same amount of time for oral comments at the hearing, and will have the same ability to cross-examine (if they so choose) as everyone else.

Regarding the three remedies the Water Agencies ask the Water Board to consider, see our responses to Water Agencies Comments IV.A, IV.B, and IV.C, below.

Water Agencies Comment I: ***The tentative order does not address ammonium discharges.*** *The Water Agencies contend that the tentative order does not address ammonium discharges and object to its water quality-based ammonia effluent limits being higher than the plant's current ammonia discharge concentrations. The Water Agencies contend that, because some other wastewater treatment plants remove ammonia, requiring this plant to remove ammonia would not require a new or unproven technology.*

Response: We agree that the tentative order did not explicitly address ammonium discharges. We revised the tentative order (Fact Sheet section IV.C.7) to include findings related to ammonium. The ammonia limits proposed in the tentative order were based only on the Basin Plan's un-ionized ammonia objective. These limits were based on water quality requirements, not the actual discharge or current performance.

We agree that technology for additional ammonia removal is available. However, U.S. EPA's technology-based requirements for municipal wastewater treatment plants (i.e., the secondary treatment standards) do not require ammonia removal. We believe more information is needed before imposing an ammonium limit that requires additional treatment. We are working with the Bay Area Clean Water Agencies (BACWA) to obtain this information (see Exhibit 1 for some of our correspondence with BACWA). In the interim, we revised the tentative order (Table 7 and

Fact Sheet section IV.C.7) to incorporate a mass-based effluent limit that reflects existing ammonium treatment performance (see response to Water Agencies Comment IV.C).

Water Agencies Comment II: ***Uncontrolled ammonium discharges could adversely affect beneficial uses.*** *The Water Agencies call on the Water Board to review available scientific information and contend that available information points to the need for ammonia removal. Essentially, the Water Agencies contend that there is reasonable potential for the discharge to cause or contribute to violations of water quality standards; therefore, effluent limits are necessary to address ammonium discharges. The Water Agencies note that (1) excessive ammonium is toxic to copepods, (2) excess ammonium inhibits nitrogen uptake by diatoms and reduces diatom primary production, (3) nutrient discharges affect algal communities by changing nutrient ratios to favor harmful species, and (4) nutrient removal at wastewater treatment plants improves ecosystems and aquatic life where implemented.*

Response: We agree that available scientific information provides cause for concern. We do not agree that existing information is sufficient to require additional ammonia removal from the District's discharge at this time. Available information may not be as conclusive as the Water Agencies suggest. The copepod ammonium toxicity is not an issue for Suisun Bay because the ammonia concentrations observed in Suisun Bay are well below the low observed effect concentration derived in the studies. The potential for ammonium from the District's discharge to inhibit phytoplankton productivity in Suisun Bay exists, but needs to be evaluated in the context of other possible factors that could also affect productivity. Finally, scientists disagree about whether changing nutrient ratios are harming Suisun Bay algal communities.

More information is needed to understand the relative contributions of the various Suisun Bay ammonia sources to Suisun Bay ammonia concentrations and their impacts (for example, tidal action likely affects various ammonia sources differently). While the Suisun Bay ammonia load from the upstream Sacramento County Regional plant is similar in magnitude to the District's load, the District's discharge is located at the western end of Suisun Bay, close to where Suisun Bay flows into the Carquinez Strait and San Pablo Bay. It is likely that the District's actual contribution to Suisun Bay effects is much smaller than that of the Sacramento County Regional plant because a much larger portion of the District's ammonia flows out of Suisun Bay soon after discharge.

Efforts are underway to obtain the information we need to better evaluate ammonia's potential water quality effects throughout our region. Our approach is consistent with SIP section 1.3, step 8, which indicates that when available data are insufficient to complete a reasonable potential analysis, monitoring should be required as necessary to determine whether an effluent limit is appropriate. We are working with the Bay Area Clean Water Agencies (BACWA) and other stakeholders to complete necessary studies and engage in joint fact finding, We plan to complete this work in time for the District's next permit reissuance (see Exhibit 1).

Water Agencies Comment III.A: ***The tentative order should not provide a dilution credit for ammonium.***

The Water Agencies contend that the tentative order is flawed in providing a dilution credit for ammonium. They make six points:

1. *They assert that the tentative order says a mixing zone cannot be evaluated due to the complex hydrology of San Francisco Bay, and further assert that it is illogical to provide full dilution credit such that calculated limits are less stringent than current performance.*
2. *They point out that, when the Basin Plan states, "…ammonia will be diluted or degraded to a nontoxic state fairly rapidly," it refers to un-ionized ammonia, not ammonium.*
3. *They note that the Basin Plan cautions against providing a dilution credit for a discharge to a tidal zone. They then object to there being no finding justifying proposed effluent limits greater than those calculated from water quality objectives.*
4. *They mention that the dilution study indicates that mixing does not persist in the far field, beyond the zone of initial dilution. Therefore, the study only presents findings for initial dilution.*
5. *They say the Basin Plan cautions against relying on models because they only estimate initial dilution. None accounts for tidal currents.*
6. *They argue that the tentative order does not address ammonium concentrations found to be toxic to copepods and to inhibit diatom productivity; therefore, the tentative order is insufficiently protective.*

Response: We disagree that the ammonia dilution credit in the tentative order is unjustified. The Water Agencies' concerns regarding the ammonia dilution credit appear to be misplaced since, as the Water Agencies correctly point out, the tentative order's ammonia limits are based solely on the Basin Plan's un-ionized ammonia objective. The proposed limits do not address ammonium concerns and were not intended to do so. Dilution credit may be appropriate for un-ionized ammonia, but it may or may not be appropriate for ammonium. See our response to Water Agencies Comment IV.C regarding revisions to the tentative order that address ammonium. Our responses to the Water Agencies' six points are as follows:

1. The tentative order does *not* say a mixing zone cannot be evaluated. It describes the challenges in establishing a mixing zone and estimating dilution, and, considering these challenges, it justifies limiting dilution credits to reflect only initial dilution or even more so in some circumstances. The proposed dilution credits are based on the properties of the outfall and conservatively account for uncertainties regarding mixing within receiving waters.

2. We agree that the quoted Basin Plan text regarding ammonia dilution and degradation pertains to un-ionized ammonia. Since the tentative order only addressed un-ionized ammonia, it was consistent with this portion of the Basin Plan. Nevertheless, we note that un-ionized ammonia and ammonium always exist together in equilibrium; as one form degrades, so does the other.

3. The Basin Plan describes challenges related to estimating dilution for discharges to tidal waters; it does not prohibit doing so. Moreover, it does not prohibit limiting dilution credits

to reflect only initial dilution, as the tentative order does. By not accounting for dilution by tidal action, the proposed ammonia dilution credit is more conservative than it would otherwise be. Contrary to the Water Agencies' comment, the ammonia limits in the tentative order are based on un-ionized ammonia water quality objectives; therefore, no special findings are necessary. Providing a dilution credit does not mean a resulting limit is not based on water quality objectives. The Basin Plan allows higher limits than those based on water quality objectives if justified to encourage water recycling, but that is not the case here.

4. We agree that the dilution study only presents findings for initial dilution. It does not account for far-field mixing. Therefore, the proposed dilution credit conservatively reflects only initial dilution. It does not, as the Water Agencies imply, reflect any additional far-field dilution.

5. The Basin Plan describes challenges related to modeling discharges to tidal waters. It does not prohibit far-field dilution modeling that incorporates tidal mixing. However, since such models are not readily available, most dilution studies (including the one cited in this tentative order) are limited to initial dilution, which is more conservative.

6. We agree that Suisun Bay ammonium concentrations provide cause for concern and that the tentative order did not address these concerns. We address them in our responses to Water Agencies Comments IV.A, IV.B, and IV.C, below, and in the revised tentative order (Table 7 and Fact Sheet sections VI.C.1.c and IV.C.7).

Water Agencies Comment III.B: ***The tentative order does not comply with anti-backsliding and antidegradation policies.***

The Water Agencies object to the tentative order's conclusion that, because the previous permit did not contain ammonia limits, the proposed new limits comply with anti-backsliding requirements. The Water Agencies also object to the tentative order's conclusion that the proposed ammonia limits comply with antidegradation requirements.

The Water Agencies cite antidegradation policies that apply when allowing waste flows or concentrations to increase in high quality waters and require effluent limits based on best practicable treatment or control (BPTC). The Water Agencies claim the tentative order would increase Suisun Bay ammonia concentrations because its limits are higher than the maximum observed effluent concentration. They also claim it would allow a 30 percent increase over existing discharge flows, thus increasing ammonia loads.

The Water Agencies warn against relying on prior California Environmental Quality Act (CEQA) documentation to comply with antidegradation policies. They call for revising the antidegradation analysis and recirculating the tentative order for public comment.

Response: We disagree. The tentative order complies with anti-backsliding and antidegradation requirements. Anti-backsliding requirements relate to changing effluent limits from one permit to the next. Reissued permits may not contain less stringent effluent limits than those in the permits they replace, except under specific circumstances. Because the previous permit did not contain ammonia effluent limits, this tentative order could not possibly contain less stringent ammonia effluent limits. Therefore, it complies with anti-backsliding requirements.

Antidegradation requirements relate to changes in receiving water quality. Water Board actions, such as issuing permits, cannot result in water quality degradation, except under specific circumstances. The baseline water quality condition for comparison purposes is the water quality that reflects all past regulatory and permitting actions approved in accordance with antidegradation policies. In this case, the ammonia baseline is the condition that reflects the previous permit, which the Water Board issued in accordance with antidegradation policies. When compared to the previous permit, the tentative order could not possibly degrade Suisun Bay water quality with respect to ammonia. Contrary to the Water Agencies' claim, the tentative order does not authorize any increase in effluent flow or ammonia concentrations beyond those the previous permit allowed. The Water Agencies incorrectly compare the permitted flow to the actual existing flow. The permitted flow in the tentative order is the same as it was in the previous permit. The Water Agencies also incorrectly compare the proposed ammonia limits to actual effluent concentrations. The previous permit contained no ammonia effluent limit. Therefore, by imposing an ammonia limit for the first time, the tentative order is more stringent than the previous permit and could only improve water quality, not degrade it. No CEQA document is necessary to support this conclusion. For these reasons, no findings justifying degradation are necessary, and there is no need to recirculate the tentative order for further comment.

Antidegradation policy set forth in State Water Board Resolution 68-16 requires that effluent limits be based on best practicable treatment or control (BPTC) to ensure that pollution or nuisance will not occur and that the highest water quality consistent with maximum benefit to the people of California will be maintained. U.S. EPA specifies technology-based limitations for municipal wastewater treatment plants. These "secondary treatment standards" do not require ammonia removal, and other municipal treatment plants in our region that discharge into deep water do not routinely treat to a higher standard. Nevertheless, the plant's existing treatment is clearly practicable; therefore, we revised the tentative order (Table 7 and Fact Sheet section IV.C.7) to include new performance-based ammonia effluent limits.

Water Agencies Comment IV.A: ***The Water Board should reduce ammonium discharges by requiring nitrification.***

The Water Agencies ask the Water Board to set final effluent limits that require nitrification (and possibly denitrification) and provide a compliance schedule for designing and building the additional treatment. They call for interim limits based on the maximum observed ammonia discharge concentration. The Water Agencies assert that, because other municipal wastewater treatment plants provide nitrification, feasible technologies are practicable and must be required as BPTC pursuant to antidegradation policies.

Response: We disagree. Although we may require some plants to provide nitrification (and possibly denitrification) in the future, requiring such treatment would be a big step and should be undertaken only after gaining a better understanding of the water quality benefits. Nitrification is also costly and consumes substantial energy, resulting in significant air emissions and other environmental impacts. While we agree that there are good reasons to be concerned about Suisun Bay ammonium concentrations, we do not believe available information is yet sufficient to require nitrification by the District (see response to Water Agencies Comment II). We are working with BACWA to obtain sufficient information (see Exhibit 1).

As discussed in our response to Water Agencies Comment III.B, we do not believe antidegradation policies necessarily require nitrification as BPTC at all municipal wastewater treatment plants. U.S. EPA's technology-based limitations for municipal wastewater treatment plants do not require ammonia removal. Nevertheless, we revised the tentative order (Table 7 and Fact Sheet section IV.D) to include new performance-based ammonia effluent limits.

Water Agencies Comment IV.B: ***The Water Board should defer permit reissuance until pending studies are completed.***

The Water Agencies suggest, as an alternative to requiring nitrification now, delaying permit reissuance until pending studies are completed and ammonia effluent limits may be established with more certainty.

Response: We disagree. NPDES permits are to be reissued every five years. The existing permit expires on March 31, 2012. It will take several more years to complete the studies necessary to develop ammonia limits that account for ammonium impacts. We are working with BACWA to obtain this information (see Exhibit 1), but the studies will take time. Postponing adoption until they are completed serves no purpose. This is unnecessary because the five-year permit term ensures that the entire permit will be reconsidered within about five years.

Water Agencies Comment IV.C: ***The Water Board should more effectively address ammonium.***

The Water Agencies' offer a third suggestion for reducing ammonium discharges if the Water Board decides to reissue the permit on time and is not yet prepared to require nitrification. In this case, the Water Agencies urge the Water Board to adopt findings acknowledging that ammonium could be harming Suisun Bay and describing studies underway to address these concerns. They ask the Water Board to establish a schedule for completing the studies and to ensure the funding necessary to complete them. They also ask the Water Board to commit to reconsidering the ammonia issue within 12 months and provide opportunities for public participation. Finally, they ask the Water Board to impose effluent limits based on actual treatment performance.

Response: We agree, mostly. We revised the tentative order (Fact Sheet section IV.C.7) to include findings related to ammonium. We also revised the tentative order (section VI.C.1.c) to allow the Water Board to reopen the permit and reconsider ammonium issues when more information is available. Prior to any Water Board action on this matter, it would provide opportunities for public participation, as it does with any permit reissuance or amendment. We did not, however, commit the Water Board to reopen the permit within 12 months. In our view, more time will be needed to complete necessary studies (note the schedule set forth in our correspondence with BACWA, Exhibit 1).

We revised the tentative order (Table 7 and Fact Sheet section IV.C.7) to incorporate a new performance-based ammonia effluent limit because the plant's existing treatment is clearly practicable. To avoid inadvertently imposing a disincentive for water conservation and recycling, the new limit is mass-based, calculated by multiplying average ammonia concentration by the plant's designed flow capacity.

SAN FRANCISCO BAYKEEPER

Baykeeper Comment 1: ***The tentative order must include effluent limitations for residual chlorine and settleable matter.***

Baykeeper says the Basin Plan requires all NPDES permits for wastewater treatment facilities protect beneficial uses by limiting residual chlorine discharges to 0.0 mg/L and settleable matter discharges to 0.2 ml/1-hour per day and 0.1 ml/1-hour per on average over 3 days (Basin Plan Table 4-2).

Response: We disagree. Residual chlorine effluent limits are unnecessary because the District does not use chlorine to disinfect its wastewater. It uses ultra-violet radiation. Likewise, settleable matter effluent limits are unnecessary because Basin Plan Table 4-2 does not require them. In the past, Basin Plan Table 4-2 required settleable matter limits for all treatment facilities, but this requirement, found in footnote d, has since been removed for settleable matter. The settleable matter limits had become outdated for municipal wastewater treatment plants since they were historically used to evaluate primary treatment. All municipal wastewater treatment plants must now meet more stringent secondary treatment standards.

Baykeeper Comment 2: ***The tentative order must conduct a reasonable potential analysis that addresses pharmaceuticals, chemicals from personal care products, and sediment toxicity.***

Baykeeper says the tentative order's reasonable potential analysis ignores several pollutants likely in the wastewater, such as antibiotics, contraceptives, various medicines, nanoparticles from sunscreen, and chemical fragrances. It says these substances may cause ecological and human harm, noting, for example, that triclosan (the active ingredient in many antibacterial products) has been detected in San Francisco Bay, is toxic to aquatic organisms, and bioaccumulates within the food web. At a minimum, Baykeeper requests monitoring for pharmaceuticals and personal care products.

Baykeeper also says the tentative order should evaluate the potential for sediments to impair Suisun Bay. According to the State Water Board's "Water Quality Control Plan for Enclosed Bays and Estuaries—Part 1, Sediment Quality", sediments are not to have pollutants that harm benthic communities, wildlife, resident finfish, or human health. Baykeeper faults the tentative order for dismissing the sediment quality objectives because "there is no evidence directly linking compromised sediment conditions to the discharges subject to this Order." Instead, *Baykeeper says the tentative order should require sediment pollutant monitoring to evaluate the need for additional limits during the next permit cycle.*

Response: We did not revise the tentative order. We are unaware of promulgated water quality standards that would allow us to perform a reasonable potential analysis for the compounds Baykeeper suggested. While we share some of Baykeeper's concern that some of the compounds have been detected in the Bay, there is insufficient information to specifically determine if the levels detected are causing actual problems, or how to translate a potential problem into a numeric limit.

Until sufficient information is available, the tentative order, like nearly all other permits in this region, would require compliance with the Basin Plan's toxicity objective through acute and chronic toxicity testing and compliance with limitations if appropriate. Toxicity tests would measure unregulated pollutants, such as personal care products and pharmaceuticals, or pollutants with synergistic effects, in the discharges. Both of these tests are conducted on the most sensitive species available and serve as indicators for protecting all other aquatic life. Including mortality, the chronic toxicity tests specifically measure sublethal impacts, such as changes in reproduction or growth, from these unregulated compounds.

That said, we are working with the San Francisco Estuary Institute (SFEI) to better understand personal care products and pharmaceuticals, and to identify any that we should target for further monitoring. For example, SFEI measured triclosan at detectable concentrations in San Francisco Bay, but found the concentrations to be less than the known toxicity threshold for this pollutant. Moving forward, SFEI has a workgroup through the Regional Monitoring Program (RMP) that is addressing emerging contaminants and is expected to produce a report on next steps in spring 2012. The RMP has and is currently funded in large part by all San Francisco Bay dischargers.

Regarding sediment monitoring, the tentative order would require the District to participate in the RMP. Through this effort, additional sediment toxicity data are being collected that will allow us to revisit whether the discharge may be impacting sediment quality. The State Water Board's *Water Quality Control Plan for Enclosed Bays and Estuaries—Part 1, Sediment Quality* requires a multiple lines of evidence approach (toxicity, chemistry, and benthos) to determine impairment. For San Francisco Bay sites identified as impacted, SFEI is working on how to conduct a stressor analysis to determine the causal factors behind toxicity. This is a necessary step before we can conduct a linkage analysis to identify sources of sediment toxicity. Given the complex nature of assessing sediment quality, we believe it is most effective to require all San Francisco Bay dischargers to support the RMP as opposed to requiring individual dischargers to attempt this complex and costly work by themselves.

STAFF INITIATED CHANGES

In addition to making minor formatting and typographical edits, Water Board staff made the following revisions to Attachment E, Monitoring and Reporting Program. These changes modify Attachment G, Regional Standard Provisions, and Monitoring and Reporting Requirements, to be consistent with current circumstances. The revisions to Attachment G are shown in underline/~~strikeout~~ format.

We revised MRP section VIII.A as follows:

A. General Monitoring and Reporting Requirements

The Discharger shall comply with all Federal Standard Provisions (Attachment D) and Regional Standard Provisions (Attachment G) related to monitoring, reporting, and recordkeeping, with modifications shown in section VIII.D below.

We added MRP section VIII.D as follows:

D. Modifications to Attachment G

1. Attachment G sections V.C.1.f and V.C.1.g are revised as follows, and section V.C.1.h (Reporting data in electronic format) is deleted.

f. Annual self-monitoring report requirements

By the date specified in the MRP, the Discharger shall submit an annual report to the Regional Water Board covering the previous calendar year. The report shall contain the following:

1) Annual compliance summary table of treatment plant performance, including documentation of any blending events (this summary table is not required if the Discharger has submitted the year's monitoring results to CIWQS in electronic reporting format by EDF/CDF upload or manual entry);

2) [Subsection V.C.1.f.2 is unchanged from Attachment G.]

3) Both tabular and graphical summaries of the monitoring data for the previous year if parameters are monitored at a frequency of monthly or greater (this item is not required if the Discharger has submitted the year's monitoring results to CIWQS in electronic reporting format by EDF/CDF upload or manual entry);

[Subsections V.C.1.f.4 through V.C.1.f.7 are unchanged from Attachment G.]

g. Report submittal

The Discharger shall submit SMRs addressed as follows, unless the Discharger submits SMRs electronically to CIWQS:

California Regional Water Quality Control Board
San Francisco Bay Region

1515 Clay Street, Suite 1400
Oakland, CA 94612
Attn: NPDES Wastewater Division

h. Reporting data in electronic format – Deleted

2. Attachment G sections V.E.2, V.E.2.a, and V.E.2.c are revised as follows, and sections V.E.2.b (24-hour Certification) and V.E.2.d (Communication Protocol) are deleted.

2. Unauthorized Discharges from Municipal Wastewater Treatment Plants[1]

The following requirements apply to municipal wastewater treatment plants that experience an unauthorized discharge at their treatment facilities and ~~are consistent with and~~ supersede requirements imposed on the Discharger by the Executive Officer by letter of May 1, 2008~~, issued pursuant to California Water Code Section 13383~~.

a. Two (2)-Hour Notification

For any unauthorized discharges that ~~result in a discharge to~~ enter a drainage channel or a surface water, the Discharger shall, as soon as possible, but not later than two (2) hours after becoming aware of the discharge, notify the ~~State Office of~~ California Emergency ~~Services~~ Management Agency (CalEMA currently 800-852-7550), the local health officers or directors of environmental health with jurisdiction over the affected water bodies, and the Regional Water Board. ~~The~~ Timely notification by the Discharger to CalEMA also satisfies notification to the Regional Water Board~~'s online reporting system at~~ www.wbers.net~~, and~~. Notification shall include the following:

[Subsections V.E.2.a.1 through V.E.2.a.6 are unchanged from Attachment G.]

b. 24-hour Certification – Deleted

c. 5-day Written Report

Within five business days, the Discharger shall submit a written report~~, via the Regional Water Board's online reporting system at www.wbers.net,~~ that includes, in addition to the information required above, the following:

[Subsections V.E.2.c.1 through V.E.2.c.7 are unchanged from Attachment G.]

d. Communication Protocol – Deleted

[1] California Code of Regulations, Title 23, Section 2250(b), defines an unauthorized discharge to be a discharge, not regulated by waste discharge requirements, of treated, partially treated, or untreated wastewater resulting from the intentional or unintentional diversion of wastewater from a collection, treatment or disposal system.

Exhibit 1

Matthew Rodriquez
Secretary for Environmental Protection

California Regional Water Quality Control Board
San Francisco Bay Region

1515 Clay Street, Suite 1400, Oakland, California 94612
(510) 622-2300 • FAX (510) 622-2460
http://www.waterboards.ca.gov/sanfranciscobay

Edmund G. Brown Jr.
Governor

January 24, 2012
CIWQS Place IDs 213875, 219552, and 270006

Bay Area Clean Water Agencies
Attn: Amy Chastain, Executive Director
P.O. Box 24055
Oakland, California 94623

Dear Ms. Chastain:

SUBJECT: Water Board Support for Nutrient Strategy Development and Implementation

We support the proposal by the Aquatic Science Center and the San Francisco Estuary Institute, *Nutrient Strategy Development and Implementation: A proposal to BACWA and the San Francisco Bay Regional Water Quality Control Board* (revised January 18, 2012), and we appreciate the Bay Area Clean Water Agencies' (BACWA's) support as well. The potential impacts of nutrient discharges on San Francisco Bay water quality in general, and ammonium discharges on Suisun Bay water quality in particular, are of increasing concern and not well understood. We believe this proposal will allow the Water Board and BACWA to take necessary next steps to inform future decisions, and we look forward to working closely with BACWA as we move forward.

Consistent with the proposal, our goal is to collect sufficient information for sound decision-making by February 2016, prior to considering the Central Contra Costa Sanitary District's NPDES permit for reissuance. We understand that the path to reach our goal is uncertain; therefore, we look forward to approaching these studies adaptively and seeking input from scientific experts and others with a stake in San Francisco Bay water quality. We recognize that the scope of future work will depend on the outcomes of our initial efforts. Therefore, we support the overall schedule set forth in the proposal, with the understanding that it could change.

We believe the proposal, which BACWA has agreed to fund, at least initially, provides the certainty we need to ensure that this important work will proceed without delay. Naturally, we assume that BACWA will fund future efforts as roughly outlined in the proposal. If not, we reserve our right and responsibility to require appropriate dischargers to submit necessary information pursuant to California Water Code § 13267. To ensure that adequate progress on the proposal continues through the coming years, we request that routine progress reports be submitted at least annually.

Please use Naomi Feger, the Water Board's Chief of Planning, as BACWA's contact for carrying out the proposal and for the nutrient strategy in general. She is reachable at NFeger@waterboards.ca.gov or 510-622-2328.

Sincerely,

Digitally signed by Bruce Wolfe
Date: 2012.01.24 15:33:15 -08'00'

Bruce H. Wolfe
Executive Officer

Cc: James Kelly, Central Contra Costa Sanitary District
Gary Darling, Delta Diablo Sanitary District
Ronald Matheson, Vallejo Sanitation and Flood Control District
David Senn, Aquatic Science Center / San Francisco Estuary Institute

January 18, 2012

Mr. Bruce Wolfe
California Regional Water Quality Control Board
San Francisco Bay Region
1515 Clay Street, Suite 1400
Oakland, CA 94612

RE: BACWA Support for Development and Implementation of a Nutrient Science and Management Strategy

Dear Mr. Wolfe:

I am writing on behalf of the Bay Area Clean Water Agencies (BACWA) to inform you that, on December 19, the BACWA Executive Board approved funding for a work to help advance the understanding of potential nutrient-related impacts on San Francisco Bay water quality and to develop numeric nutrient endpoints (NNEs). The attached proposal was developed by the San Francisco Estuary Institute/Aquatic Science Center (SFEI/ACS) as requested by BACWA, and in coordination with San Francisco Bay Regional Water Quality Control Board (Regional Water Board) staff.

The proposal consists of three work elements; identified as high priority projects by SFEI/ASC, the Regional Water Board and BACWA:

- **Work Element #1. Coordination of Nutrient Strategy Development and Implementation**

This element will enable refinement of the preliminary Nutrient Strategy already developed by the Regional Water Board and based on the NNE Literature Review and Data Gaps analysis. It will also provide resources to develop more detailed work plans and funding proposals for parts of the Strategy that are currently only captured in general terms. An important component of this element will be the convention of stakeholder and technical advisory groups to assist with refining goals, developing work plans, identifying priority projects, and strategizing about funding mechanisms.

- **Work Element #2. Box Models and Budgets, Suisun Bay and South Bay: Hypothesis testing and sensitivity analysis**

This element will build upon conceptual model work funded by the Regional Monitoring Program (RMP) for completion in 2012 and the phytoplankton assessment framework funded by the Water Boards in 2012. The outcomes will be numeric box models for Suisun Bay and the South Bay, which are critical links between the conceptual models and future three dimensional dynamic simulation models. These models will assist with generating and testing hypotheses; quantifying the relative importance of processes; and performing sensitivity analysis to help identify critical data gaps. In addition, these models will be used to evaluate biological response under future scenarios in these Bay segments (e.g., decreased nutrient loads, changes in flow from the Delta, changes in nitrogen speciation due to potential addition of a nitrification step at wastewater treatment plants, continued decreases in suspended sediment loads, etc.).

- **Work Element #3: Suisun Bay Study Plan Development**

Questions about the potential relationship between ammonia, nutrients, and other stressors in the Suisun Bay and the decline of protected pelagic fish species have been raised by this Regional Water Board, the Central Valley Regional Water Board, and stakeholders (including agencies that rely on water from the Central Valley Water Project). These issues are also identified in the NNE Literature Review and the current version of the Nutrient Strategy as requiring further investigation. This element is a logical next step towards resolving existing uncertainties. Specific deliverables include: (1) a draft study plan summarizing existing knowledge related to beneficial use impairment from ionized ammonia, describing ongoing studies, and identifying additional studies to address knowledge gaps; (2) a technical report on the life cycles of diatoms and copepods suspected of being adversely affected by ammonia, nutrients and other stressors in the system; and (3) a Suisun Bay model that builds upon work element two (described above) and that is consistent with the Bay-Delta wide modeling plan being developed within the nutrient strategy.

This work will be carried out by the San Francisco Estuary Institute and Aquatic Science Center (SFEI/ASC), in cooperation with the Regional Water Board, BACWA, other regional stakeholders, and regional scientists. It supplements and complements work previously undertaken by SFEI/ASC and funded by BACWA, including the June 2011 Nutrient RMP Workshop and South Bay nutrient load refinement. It is also consistent with work in progress related to the Surface Water Ambient Monitoring Program-funded Suisun Bay Work Plan which BACWA and the Central Contra Costa Sanitary District have supported and the Regional Monitoring Program's 2012 nutrient studies recently approved by the Steering and Technical Review Committees.

In early 2012, we anticipate working with SFEI/ASC and Regional Water Board representatives to develop the details necessary to implement this $350,000 proposal. We believe that this effort is a good start towards a cohesive plan that will provide strong scientific and technical bases for future scientific studies and models that can then be used understand nutrient-related impacts on the Bay and ultimately guide management decisions to further protect the San Francisco Bay Estuary.

Sincerely,

Amy Chastain
Executive Director

Attachment: Nutrient Strategy Development and Implementation: A proposal to BACWA and the San Francisco Bay Regional Water Quality Control Board

CC: Tom Mumley, SFBRWQCB
Naomi Feger, SFBRWQCB
Lila Tang, SFBRWQCB
Bill Johnson, SFBRWQCB
Vince Christian, SFBRWQCB
David Senn, SFEI/ASC
Gary Darling, Delta Diablo Sanitary District
Amanda Roa, Delta Diablo Sanitary District
BACWA Executive Board

Nutrient Strategy Development and Implementation: A proposal to BACWA and the San Francisco Bay Regional Water Quality Control Board

David Senn, PhD.
Aquatic Science Center and
San Francisco Estuary Institute

December 19, 2011, revised
January 18, 2012.

1. Introduction

San Francisco Bay has long been recognized as a nutrient-enriched estuary, but one that has historically proven resilient to the harmful effects of nutrient enrichment, such as excessive phytoplankton blooms and hypoxia. However, evidence is building that, since the late 1990s, the historic resilience of the Bay to the harmful effects of nutrient enrichment is weakening, as shown through significant increases in phytoplankton biomass (e.g., Cloern et al., 2007) and through hypothesized linkages between elevated ammonium and decreased diatom primary productivity rates (e.g., Dugdale et al. 2007).

Concurrently, the State Water Resources Control Board (State Board) has begun developing numeric objectives for nutrients in estuaries, and has adopted the Nutrient Numeric Endpoint (NNE) framework for this work. The NNE framework utilizes biological indicators as endpoints combined with load-response modeling to determine nutrient loads to estuaries that are protective of beneficial uses. The California Regional Water Quality Control Board, San Francisco Region, (Regional Water Board) is using the NNE approach to develop nutrient objectives for the San Francisco Bay. An early product of that effort was a literature review (McKee et al., 2011) that identifies candidate biological indicators for the Bay and important science and data gaps that need to be addressed along the path to setting nutrient objectives.

In response to the apparent changes in the Bay's resilience to nutrient loading and recognizing the need for nutrient objectives, Regional Water Board staff and various Bay stakeholders have begun the process of developing a Nutrient Strategy. An initial draft strategy was developed in 2011, with a main goal of laying out a well-reasoned and cost-effective program to generate the scientific understanding needed to fully support major management decisions related to nutrients. The draft strategy has four main work elements: i) defining the problem; ii) monitoring program development and implementation; iii) developing a nutrient assessment framework; iv) developing a modeling strategy that can be used to assess potential impacts of various management actions.

Within the framework of the Regional Water Board and BACWA's cooperative effort on nutrients in San Francisco Bay, this proposal requests funds to support on-going nutrient strategy development, and to begin work on two sets of high priority projects. The proposal consists of three work elements: Work Element #1: Coordination of Nutrient Strategy Development and Implementation; Work Element #2: Box Models and Budgets, Suisun Bay and South Bay: Hypothesis testing and sensitivity analysis; and Work Element #3: Suisun Bay Study Plan Development. This work will be carried out by the SFEI/ASC in collaboration with the Southern California Coastal Water Research Project (SCCWRP), and in cooperation with the Regional Water Board, BACWA, other regional stakeholders, and regional scientists.

2. Work Element 1: Coordination of Nutrient Strategy Development and Implementation.

Numerous organizations are either funding or actively engaged in nutrient-related work in the Bay-Delta Estuary (e.g., the Regional Water Board, Central Valley Regional Water Board, Interagency Ecological Program (IEP), United States Geographical Survey (USGS), Romberg Tiburon Center, State and Federal Contractors Water Agency (SFCWA), Delta Stewardship Council (DSC), BACWA, Bay Area Stormwater Management Agencies Association (BASMAA), and the California Department of Water Resources). However, there is limited coordination between these efforts, and no overarching set of science or management goals, or a cohesive science plan, across Bay segments and into the Delta. A nutrient science and management strategy for the Bay-Delta Estuary is urgently needed so that work is carried out in a coherent, complementary, and prioritized fashion.

This work element focuses on coordinating the on-going development of an Estuary-wide nutrient strategy that has broad-base support among the stakeholder and regulatory/management communities. For jurisdictional reasons, an initial focus will be placed on the Bay-proper (west of and including Suisun Bay), however upstream factors that strongly influence Suisun and downgradient Bay segments (e.g., nutrient loads, flow) will obviously remain major considerations.

Work Process and Deliverables

The draft nutrient strategy (September 2011) will serve as the baseline document for further strategy development. The major deliverable of Work Element #1 will be a revised draft nutrient strategy that describes management questions and goals, major work elements that address the questions/goals, sub-tasks within each work element, and detailed work plans or scopes of work. To the extent possible, a draft funding strategy will also be developed. An additional deliverable will be a Bay-Delta nutrient website that will serve as a clearinghouse for information (news, downloadable reports, links to related websites, calendar), and a related listserv.

SFEI/ASC and will work with the Regional Water Board, BACWA and other stakeholders on the process for developing the strategy documents, including the organizational structure/process for decision-making. SFEI/ASC staff will meet with scientists with expertise in this area to refine goals and develop the detailed scientific work plans. Smaller meetings, or sub-committee meetings, with program managers (e.g., IEP, DSC) and stakeholders groups are also envisioned to identify priority projects and strategize about ways to fund work. To inform the website development, an informal survey of anticipated users will be conducted in the first quarter of 2012 to identify content that will be most useful. SFEI/ASC will work with the Regional Water Board, as requested, to ensure that documents - including agendas, meeting minutes and reports - will be made available to Regional Water Board staff in a timely fashion so they can be posted on the agency's website.

Deliverables will include the draft nutrient strategy, which will be completed by November 2012. The website and listserv will be on-line by April 2012.

3. Work Element 2: Box Models and Budgets, Suisun Bay and South Bay: Hypothesis testing and sensitivity analysis.

Nutrient dynamics and biological response to nutrient loads in San Francisco Bay are highly complex and highly variable in both space and time. The published literature suggests that the accumulation of phytoplankton biomass in the Bay is strongly limited by tidal mixing, grazing pressure by invasive clams, and light limitation from high turbidity. In addition, it is hypothesized that, in the North Bay, elevated levels of ammonium actually inhibit diatom primary productivity and contribute to relatively low levels of phytoplankton biomass there (Dugdale et al., 2007). However, the relative importance of these processes on the food web is poorly understood, and it is thus difficult to determine what are the main drivers, and in turn anticipate changes in biological response to future changes in the system. Further, to date, there has been limited quantitative assessment of nutrient budgets in Bay segments, and such budgets are needed to, for example, assess the importance of internal sources and sinks (mineralization of organic matter in the sediments and nitrogen and phosphorous flux to the water column, denitrification at the sediment water interface).

A major component of the NNE framework is the use of "load-response" models to translate numeric endpoints into sustainable load estimates. A recent report sponsored by BACWA (BACWA 2011) recommended a staged approach to nutrient and water quality model development in the Bay, beginning with conceptual model development, and followed by numeric models of increasing complexity, as needed. The San Francisco Bay Regional Monitoring Program (RMP) has funded a nutrient conceptual model project that began in January 2012. This project will develop spatially and temporally explicit conceptual models for individual Bay segments. It will also evaluate scenarios for future changes to key drivers/factors that influence biological responses to nutrient loads, prioritize scenarios that could be investigated through future modeling efforts, and additional scientific investigations to address critical knowledge gaps. A critical intermediate step between conceptual models and three-dimensional (3D) dynamic simulation models is the development of numeric box models for individual Bay segments.

This work element proposes to develop box models for Suisun Bay and South Bay for the purposes of generating and testing hypotheses; quantifying the relative importance of different processes; performing sensitivity analysis to help identify critical data gaps and will support evaluation of proposed indicators as part of the

NNE. In addition, these models will be used to evaluate biological responses under future scenarios in these Bay segments (e.g., decreased nutrient loads, changes in flow from the Delta, changes in nitrogen speciation due to potential addition of a nitrification step at wastewater treatment plants, continued decreases in suspended sediment loads, etc.). Work will begin by convening a Model Evaluation Group (MEG) to develop a modeling plan. Work Element #2 will consider models that are already under development by USGS and the San Francisco State University's Romberg Tiburon Center for Environmental Studies (SFSU-RTC), and those researchers will be integrally involved as project advisors (USGS: J. Cloern, J. Kuwabara; SFSU-RTC: R. Dugdale, A. Parker). This effort will also include consideration of research/models available nationwide. This is a two year project, beginning in the third quarter of 2012.

Work Process and Deliverables

This study will be designed around addressing questions with clear management implications and with direct linkages to beneficial uses. Although secondary producers will be included in the model (e.g., zooplankton, clams), this is not intended to be a food web model or assess changes to the food web. Instead, secondary producers have been added as top-down controls on the accumulation of phytoplankton biomass. The main endpoints of interest for this work will be phytoplankton biomass and dissolved oxygen, and using mass balance constraints to gain a better understanding of internal processes and loads. Some examples of the modeling goals and approach are described below. However, a key task of the MEG will be to identify the main questions to be addressed through the modeling work, approaches for incorporating key processes into the model, and the appropriate model platform. A draft technical report produced by the MEG will be reviewed by a stakeholder advisory group, and stakeholder comments will be addressed in a final version.

For South Bay, key questions to be addressed through modeling work may include: What conditions would generate water-quality impairment such as hypoxia or chlorophyll concentration beyond a threshold, or significant shifts in the algal community composition given today's nutrient loading? How much must nutrient inputs be reduced to prevent these impairments? How would the Bay respond to a steady increase in air temperature or reduced freshwater inflow and flushing? How would it respond to a steady increase in water transparency?

For Suisun Bay the questions would be somewhat different. Relevant and testable questions include: How relatively important are ammonium, nutrients, and other stressors; light limitation; and clam grazing as controls on annual primary production or algal community composition? How would phytoplankton biomass in Suisun Bay respond to reductions in ammonium loading and reductions in nitrogen loading? How would phytoplankton biomass respond to changes in the speciation of nitrogen loaded to the system (i.e., nitrate instead of ammonium, due to nitrification in POTWs)? Under the above scenarios, what is the relative importance of freshwater flow (e.g. wet versus dry year and current water supply scenarios versus planned changes in flows). For both sets of models, best estimates of current and future loads will be needed. Thus, results from another RMP-funded study that will quantify loads to individual Bay segments will serve as important input. Planned changes at the Sacramento County Regional Wastewater Treatment Plant will explicitly be taken into account for future scenarios in Suisun Bay. The relative importance of loads from the Central Contra Costa Sanitary District (CCCSD), the Delta Diablo Sanitation District (DDSD), and other discharges will also be evaluated under those conditions.

The model for South Bay will be developed as a two-layer box model to allow for vertical stratification and bloom formation in the surface stratified layer. Suisun Bay will be set up as a two-compartment horizontal model to represent the shallow and deep waters. In both models, water column boxes will be coupled to sediment process boxes. Dynamic components will include: a small number of phytoplankton life forms (e.g., diatoms, flagellates, cyanobacteria), ammonium, nitrate+nitrite, zooplankton, light attenuation coefficient, and dissolved oxygen. Key processes will include: vertical mixing (wind and tides); horizontal transport across boundaries; nutrient inputs; benthic regeneration; nitrification-denitrification; pelagic regeneration; ammonium and nitrate uptake; growth rate as a function of internal nitrogen pools, light, temperature; zooplankton grazing; benthic grazing; photosynthesis; pelagic respiration; benthic respiration; and air-water oxygen exchange.

Researchers at USGS and SFSU-RTC have initial models that already include many of these parameters. The focus of this project will be on continued model development, as well as model calibration and validation, and on future scenario analysis. If time and funding permit during the two-year project, the core model features developed through the box model work will be incorporated into a coarse spatially explicit box model of the whole Bay (e.g., the Uncles-Peterson model). This would be a logical next step, building toward implementation of a full 3D model.

It is anticipated that this project will be carried out by a PhD-level scientist with a strong background in both modeling and biogeochemistry. This scientist will be hired for a two-year position and will be based at SFEI, but work closely with researchers at USGS and RTC for model development. Deliverables will include regular project updates (6 month intervals), as well as draft and final reports. This project will continue until approximately June 2014. While BACWA is only being asked to fund this work though 2012, BACWA has expressed a commitment to ensure that funding is made available for subsequent years.

4. Work Element #3: Suisun Bay Study Plan Development

Several nutrient-related issues in Suisun Bay have recently been brought to the forefront by Regional Water Board staff due to time-sensitive considerations around the reissuance of the CCCSD discharge permit. The issues are related to:

- the extent to which loads from CCCSD and DDSD, and other discharges, affect ammonia/ammonium concentrations in different parts of Suisun Bay and over different seasons;
- the impact of other ammonium sources on ammonia/ammonium concentrations (e.g., including sediment flux.);
- how temporal variations in Suisun Bay ammonium concentrations relates to diatom and copepod life cycles, driven by concerns over potential ammonium inhibition of diatom primary production and ammonium toxicity to copepods.

The potential role that elevated ammonium levels play in beneficial use impairment in Bay-Delta Estuary were explored during a 2009 CALFED Bay-Delta workshop and a subsequent Central Valley Regional Water Quality Control Board Ammonia Summit. The CALFED Bay-Delta workshop resulted in a research framework that proposed a number of research topics.[1]

Several of the issues raised by the San Francisco and Central Valley Regional Water Boards, and identified in the Ammonia Summit summary, specific to Suisun Bay, are currently being explored in studies carried out through a Regional Water Board-coordinated effort, in collaboration with the SFCWA, SFSU-RTC, BACWA, and CCCSD. Additional relevant studies are also being carried out through the IEP as part of the Bay-Delta program. Some of these studies are ongoing, and for others final syntheses have not yet been completed.

In addition, resolving scientific issues related to ammonium in Suisun Bay has been envisioned as an important component of the nutrient strategy. As such, a synthesis of the science related to ammonium, nutrients and other stressors on beneficial uses in Suisun Bay are planned within the context of Work Element #1 and an RMP-funded conceptual model and problem definition study for the Bay. However, the focus and level of detail required exclusively for Suisun Bay extends beyond the intended scopes of those projects and will require additional resources.

[1] A Framework for Research Addressing the Role of Ammonia/Ammonium in the Sacramento-San Joaquin Delta and the San Francisco Bay Estuary Ecosystem, available at http://www.swrcb.ca.gov/centralvalley/water_issues/delta_water_quality/ambient_ammonia_concentrations/ammonia_mem.pdf.

Work Process and Deliverables

Three components and deliverables are proposed to address time-sensitive questions related to ammonium studies in Suisun Bay. Work on these deliverables will commence in 2012, but will likely extend beyond 2012, as discussed further below. See Section 5 for schedule deliverable due dates. One of the goals of these studies is to enable Regional Water Board staff to make decisions related to permit reissuance by February 2016.

The first deliverable will be a study plan that summarizes existing knowledge related to beneficial use impairment from ionized ammonia in Suisun Bay, describes recently completed or on-going studies - including the Regional Water Board's ongoing Suisun Bay study - and identifies additional studies needed to address persistent knowledge gaps. As noted above, some of this work falls generally within the scope of Work Element #1 and the conceptual model studies being sponsored by the RMP, and this deliverable is meant to complement and augment those efforts. The study plan will propose a schedule for completion of additional studies identified and a schedule, dependent on identifying future funding, for completion of the deliverables identified in the work elements discussed below. It is anticipated that the study plan will be revisited annually as part of annual progress report submittal and adjustments will be recommended to Regional Water Board staff. The study plan may also be adaptively managed as an element of the overall nutrient strategy.

A second deliverable under Work Element #3 will be a technical report that:
i) describes the life cycles of the diatoms and copepods shown, or suspected, to be adversely impacted by ammonia (e.g., feeding, reproduction, salinity tolerances, population dynamics, etc.); ii) summarizes available information regarding the potential impacts of ammonia on these biological resources (e.g., toxicity and other adverse effects); iii) explores the potential role of nutrients and other stressors in the system; iv) identifies remaining critical information gaps.

A third work product will be a Suisun Bay model, provided that funding is available beyond 2012. The efforts in work element #2 should identify the key questions that need to be addressed through modeling (e.g., contribution of CCCSD and DDSD, and other discharges, to ammonium concentrations at various points in space and time within Suisun Bay, nutrient dynamics, and linkages to biological endpoints), the modeling approaches/platforms needed to address these questions, and data/research needs for calibrating and validating the model. The understanding gained from box-modeling work described in Work Element #2 will contribute significantly to developing a Suisun Bay specific model for ammonium, as will the RMP-sponsored conceptual model. The model for Suisun Bay should ideally be consistent with the Bay-Delta wide modeling plan.

Due to the complex nature of these questions, the myriad researchers addressing them, and the fact that highly relevant studies are being carried out in Spring 2012 (with sample analysis and data interpretation ongoing through Fall 2012), developing these work products is expected to be an iterative process. Work will commence on Work Element #3 in 2012, and will include activities such as literature review, meetings with regional scientists and workgroups carrying out studies. In addition to the annual progress reports, other progress updates will be provided as necessary. Work on some of these deliverables will extend beyond 2012, with the focus, scope, budget, and funding to be refined through discussions with Regional Water Board, stakeholders, and technical experts as part of the development of the study plan.

5. Schedule and Budget

The budget and schedule below are estimates and will be refined as needed. This funding is anticipated to be sufficient for all 2012 deliverables, although it is expected that additional costs will arise in 2012 associated with coordinating strategy development and implementation. These include convening technical teams (travel, honarium or per diem), holding stakeholder meetings (e.g., lunch, refreshments), funding experts to contribute to the writing of technical reports, and regional travel expenses for SFEI staff to attend meetings.

Funding and schedules for work completed beyond 2012 will be agreed upon by the Regional Water Board, BACWA and other stakeholders.

	Dates[2]	**2012 Funding**
Work Element #1: Nutrient Strategy		$200,000
Revised Nutrient Strategy	Nov. 2012	
Website	Apr. 2012	
Listserv	Apr. 2012	
Coordination & Reporting[3]	NA	
Work Element #2: Suisun and South Bay Box Models		$75,000[4]
Box model study plan	Jun. 2012	
Model data compilation	Nov. 2012	
Final model technical report[4]	Jun. 2014	
Work Element #3		$75,000
Draft Suisun Bay study	Sep. 2012	
Suisun Bay study plan	Dec. 2012	

[2] Dates are estimates and are contingent on funding.
[3] Includes coordination for Elements #2 and 3 and reporting on the status to the Regional Water Board and stakeholders.
[4] $75k supports the first six months of effort for this two-year task, which is 25% of its anticipated total cost through June 2014.

Ammonium impacts technical report	Jun. 2013
Suisun Bay model	Feb. 2014
Final synthesis report	Feb. 2016

Total **$350,000**

EXHIBIT G

Matthew Rodriguez
Secretary for Environmental Protection

California Regional Water Quality Control Board

San Francisco Bay Region

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Edmund G. Brown, Jr.
Governor

ORDER NO. R2-2012-0016
NPDES NO. CA0037648

The following discharger and discharges from the discharge point identified below are subject to waste discharge requirements set forth in this Order.

Table 1. Discharger Information

Discharger	Central Contra Costa Sanitary District
Name of Facility	Central Contra Costa Sanitary District Wastewater Treatment Plant and its associated wastewater collection system
CIWQS Place Number	213875
Facility Address	5019 Imhoff Place, Martinez, CA 94553 Contra Costa County
The U.S. Environmental Protection Agency (USEPA) and the Regional Water Quality Control Board have classified this discharge as a major discharge.	

Table 2. Discharge Location

Discharge Point	Effluent Description	Discharge Point Latitude	Discharge Point Longitude	Receiving Water
001	Secondary Treated Municipal Wastewater	38° 02′ 44″ N	122° 05′ 55″ W	Suisun Bay

Table 3. Administrative Information

This Order was adopted by the Regional Water Quality Control Board on:	February 8, 2012
This Order shall become effective on:	April 1, 2012
This Order shall expire on:	March 31, 2017
The Discharger shall file a Report of Waste Discharge in accordance with Title 23, California Code of Regulations, as application for re-issuance of waste discharge requirements no later than:	September 30, 2016

I, Bruce H. Wolfe, Executive Officer, do hereby certify that this Order with all attachments is a full, true, and correct copy of an Order adopted by the California Regional Water Quality Control Board, San Francisco Bay Region, on the date indicated above.

Bruce H. Wolfe, Executive Officer

Contents

Tables

Attachments

I. FACILITY INFORMATION

The following Discharger is subject to the waste discharge requirements set forth in this Order:

Table 4. Facility Information

Discharger	Central Contra Costa Sanitary District
Name of Facility	Central Contra Costa Sanitary District Wastewater Treatment Plant and its associated wastewater collection system
Facility Address	5019 Imhoff Place, Martinez CA 94553 Contra Costa County
CIWQS Place Number	213875
CIWQS Party Number	220151
Facility Contact, Title, and Phone	Margaret P. Orr P.E., Director of Plant Operations, 925-228-9500
Mailing Address	5019 Imhoff Place, Martinez, CA 94553
Type of Facility	Publicly Owned Treatment Works
Facility Design Flow	53.8 million gallons per day (MGD) (average dry weather flow) 250 MGD (peak wet weather influent design flow)
Service Area	Danville, Lafayette, Martinez, Moraga, Orinda, Pleasant Hill, San Ramon, Walnut Creek, Concord, Clayton, and adjacent unincorporated areas, including Alamo, Blackhawk, Clyde, and Pacheco
Service Population	455,000

II. FINDINGS

The California Regional Water Quality Control Board, San Francisco Bay Region (hereinafter the Regional Water Board), finds:

A. Background. Central Contra Costa Sanitary District (hereinafter the Discharger) is currently discharging under Order No. R2-2007-0008 (CIWQS Regulatory Measure No. 319679), National Pollutant Discharge Elimination System (NPDES) Permit No. CA0037648. The Discharger submitted a Report of Waste Discharge dated June 1, 2011, and applied for an NPDES permit reissuance to discharge treated wastewater from its Wastewater Treatment Plant to waters of the State and the United States. The discharge is also regulated under Regional Water Board Order No. R2-2007-0077 (NPDES Permit No. CA0038849), as amended by Order No. R2-2011-0012, which superseded all requirements on mercury and polychlorinated biphenyls (PCBs) from wastewater discharges. This Order does not affect the mercury and PCBs permit.

For the purposes of this Order, references to the "discharger" or "permittee" in applicable federal and State laws, regulations, plans, or policy are held to be equivalent to references to the Discharger herein.

B. Facility Description and Discharge Location

1. **Facility Description.** The Discharger owns and operates the Central Contra Costa Sanitary District Wastewater Treatment Plant (hereinafter the Plant) and its associated wastewater collection system (hereinafter collectively the Facility). The Plant, located north of Concord and east of Martinez, (see Attachment B) provides secondary treatment of domestic,

commercial, and industrial wastewater for Danville, Lafayette, Martinez, Moraga, Orinda, Pleasant Hill, San Ramon, Walnut Creek, Concord, Clayton, and adjacent unincorporated areas, including Alamo, Blackhawk, Clyde, and Pacheco. The population of the service area is approximately 455,000. From April 2007 through December 2010, the maximum daily influent flow rate was 141 MGD and the average daily flow rate was 38.7 MGD. Both rates are well within the permitted 53.8 MGD average dry weather flow and 250 MGD peak wet weather design flow. Twenty-two (22) significant industrial users also discharge to the Facility and these discharges are regulated by the Facility's pretreatment program.

2. **Collection System.** The Discharger's wastewater collection system includes approximately 1,500 miles of pipeline, ranging from 6 to 102 inches in diameter, and 16 wastewater pumping stations. This collection system is part of the Facility covered by this Order. The City of Concord, separate from the Discharger, owns and maintains the collection system within most of Concord's city limits and the City of Clayton.

3. **Treatment Description.** Treatment processes consist of screening, grit removal, primary sedimentation, secondary biological treatment, secondary clarification and ultraviolet (UV) disinfection. These steps are shown in the process flow diagram in Attachment C.

4. **Discharge Point.** Secondary-treated wastewater is discharged at Discharge Point 001 to Suisun Bay about 3.5 miles from the Facility via a submerged outfall equipped with a multiport diffuser. The location of the outfall diffuser is approximately 1600 feet offshore at an average depth of approximately 24 feet. The diffuser is 6 feet in diameter and imbedded 4 feet into the sediment. The diffuser is oriented nearly perpendicular to the shoreline. It consists of 11 upward-facing ports separated 11.5 feet on center, for a total length of 115 feet.

   The Plant has holding basins for temporary storage of wet weather flows, with a combined volume of 170 million gallons. These basins are used to store excess wastewater after primary treatment when inflow exceeds the Plant's secondary treatment capacity. When flows subside, the stored wastewater is routed back to the headworks for full treatment.

5. **Recycled Water.** In 2010, the Discharger diverted approximately 600 million gallons of UV-disinfected effluent from its outfall to its Recycling Plant where the effluent was tertiary treated through sand/anthracite filtration and chlorine disinfection. This recycled water volume represents about 4% of the total wastewater treated. Recycled water is stored in a covered seven million gallon reservoir prior to distribution. Recycled water customers include landscape irrigators, corporation yards, private soil farms, concrete recycling and batch plants, and the county animal shelter. Recycled water activities are regulated under Regional Water Board Order No. 96-011.

6. **Biosolids Management.** Secondary sludge is thickened via dissolved air flotation, combined with primary sludge and lime, dewatered by centrifuges, and incinerated onsite. Ash is hauled by a contractor to an offsite recycling facility and used as a soil amendment. If Facility incinerators are inoperable, biosolids are diverted to sludge storage facilities and then hauled to local landfills or to the East Bay Municipal Utility District for treatment and disposal.

7. **Stormwater Discharge.** The Discharger is not required to be covered under the State Water Resources Control Board's (State Water Board) statewide industrial stormwater NPDES permit (NPDES General Permit No. CAS000001). All stormwater flows in contact with equipment or wastewater at the Plant and the pump stations serving the Plant are collected and directed to the headworks for treatment.

Attachment B provides a map of the area around the Plant. Attachment C provides a flow schematic of the Plant.

C. **Legal Authorities.** This Order is issued pursuant to Clean Water Act (CWA) section 402 and implements regulations adopted by the USEPA and chapter 5.5, division 7 of the California Water Code (CWC), commencing with section 13370. It serves as an NPDES permit for point source discharges from the Facility to surface waters. This Order also serves as Waste Discharge Requirements (WDRs) pursuant to CWC article 4, chapter 4, division 7 (commencing with section 13260).

D. **Background and Rationale for Requirements.** The Regional Water Board developed the requirements in this Order based on information submitted as part of the application, through monitoring and reporting programs, and other available information. The Fact Sheet (Attachment F), which contains background information and rationale for requirements of the Order, is hereby incorporated into this Order and constitutes part of the findings for this Order. Attachments A through E, G, and H, are also incorporated into this Order.

E. **California Environmental Quality Act (CEQA).** Under CWC section 13389, this action to adopt an NPDES permit is exempt from Chapter 3 of CEQA.

F. **Technology-Based Effluent Limitations.** CWA section 301(b) and NPDES regulations at Title 40 of the Code of Federal Regulations section 122.44 (40 CFR 122.44) require that permits include conditions meeting applicable technology-based requirements at minimum, and any more stringent effluent limitations necessary to meet applicable water quality standards. The discharge authorized by this Order must meet minimum federal technology-based requirements based on Secondary Treatment Standards at 40 CFR 133. Further discussion of the technology-based effluent limitations is included in the Fact Sheet (Attachment F).

G. **Water Quality-Based Effluent Limitations (WQBELs).** CWA section 301(b) and NPDES regulations at 40 CFR 122.44(d) require that permits include limitations more stringent than applicable federal technology-based requirements where necessary to achieve applicable water quality standards. NPDES regulations at 40 CFR 122.44(d)(1)(i) mandate that permits include effluent limitations for all pollutants that are or may be discharged at levels that have the reasonable potential to cause or contribute to an exceedance of a water quality standard, including numeric and narrative objectives within a standard. Where reasonable potential has been established for a pollutant, but there is no numeric criterion or objective for the pollutant, WQBELs must be established using (1) USEPA criteria guidance under CWA section 304(a), supplemented where necessary by other relevant information; (2) an indicator parameter for the pollutant of concern; or (3) a calculated numeric water quality criterion (WQC), such as a proposed state criterion or policy interpreting the state's narrative criterion, supplemented with other relevant information, as provided in 40 CFR 122.44(d)(1)(vi).

H. Water Quality Control Plan. *The Water Quality Control Plan for the San Francisco Bay Basin* (hereinafter the Basin Plan) is the Regional Water Board's master water quality control planning document. It designates beneficial uses and water quality objectives (WQOs) for waters of the State, including surface and groundwater. It also includes implementation programs to achieve WQOs. The Basin Plan was duly adopted by the Regional Water Board and approved by the State Water Board, the Office of Administrative Law, and USEPA. Requirements of this Order implement the Basin Plan.

Basin Plan beneficial uses for Suisun Bay are listed in the table below.

Table 5. Basin Plan Beneficial Uses

Receiving Water Name	Beneficial Uses
Suisun Bay	Industrial Service Supply (IND) Industrial Process Supply (PROC) Commercial, and Sport Fishing (COMM) Estuarine Habitat (EST) Fish Migration (MIGR) Preservation of Rare and Endangered Species (RARE) Fish Spawning (SPWN) Wildlife Habitat (WILD) Water Contact Recreation (REC1) Non-Contact Water Recreation (REC2) Navigation (NAV)

The State Water Board's *Water Quality Control Plan for Enclosed Bays and Estuaries—Part 1,Sediment Quality* became effective on August 25, 2009. This plan supersedes other narrative sediment quality objectives and establishes new sediment quality objectives and related implementation provisions for specifically defined sediments in most bays and estuaries.

I. National Toxics Rule (NTR) and California Toxics Rule (CTR). USEPA adopted the NTR on December 22, 1992, and later amended it on May 4, 1995, and November 9, 1999. About 40 criteria in the NTR apply in California. On May 18, 2000, USEPA adopted the CTR. The CTR promulgated new toxics criteria for California and, in addition, incorporated the previously adopted NTR criteria that applied in the State. The CTR was amended on February 13, 2001. These rules contain WQC for priority pollutants.

J. State Implementation Policy. On March 2, 2000, the State Water Board adopted the *Policy for Implementation of Toxics Standards for Inland Surface Waters, Enclosed Bays, and Estuaries of California* (hereinafter the State Implementation Policy [SIP]). The SIP became effective on April 28, 2000, with respect to the priority pollutant criteria promulgated through the NTR and to the priority pollutant objectives established in the Basin Plan. The SIP became effective on May 18, 2000, with respect to the priority pollutant criteria USEPA promulgated through the CTR. The State Water Board adopted amendments to the SIP on February 24, 2005, that became effective on July 13, 2005. The SIP establishes implementation provisions for priority pollutant criteria and objectives and provisions for chronic toxicity control. Requirements of this Order implement the SIP.

K. Alaska Rule. On March 30, 2000, USEPA revised its regulation that specifies when new and revised state and tribal water quality standards become effective for CWA purposes. [65 Fed. Reg. 24641 (April 27, 2000), codified at 40 CFR 131.21]. Under the revised regulation (also known as the Alaska Rule), new and revised standards submitted to USEPA after May 30, 2000, must be approved by USEPA before being used for CWA purposes. The final rule also provides that standards already in effect and submitted to USEPA by May 30, 2000, may be used for CWA purposes, whether or not approved by USEPA.

L. Stringency of Requirements for Individual Pollutants. This Order contains both technology-based and WQBELs for individual pollutants. The technology-based effluent limitations consist of restrictions on carbonaceous biochemical oxygen demand (CBOD), total suspended solids (TSS), and pH. These technology-based limitations are discussed further in the Fact Sheet (Attachment F). This Order's technology-based pollutant restrictions on CBOD, TSS, and pH implement the minimum applicable federal technology-based requirements and are more stringent than the minimum federal technology-based requirements only as necessary to meet water quality standards.

WQBELs have been derived to implement WQOs that protect beneficial uses. Both the beneficial uses and the WQOs have been approved pursuant to federal law and are the applicable federal water quality standards. To the extent that toxic pollutant WQBELs were derived from the CTR, the CTR is the applicable standard pursuant to 40 CFR 131.38. On May 18, 2000, USEPA approved the procedures for calculating individual WQBELs for priority pollutants based on the SIP. Most beneficial uses and WQOs contained in the Basin Plan were approved under State law and submitted to USEPA. Any WQOs and beneficial uses submitted to USEPA prior to May 30, 2000, but not approved by USEPA before that date, are nonetheless "applicable water quality standards for the purposes of the CWA" pursuant to 40 CFR 131.21(c)(1).

M. Antidegradation Policy. NPDES regulations at 40 CFR 131.12 require that state water quality standards include an antidegradation policy consistent with the federal policy. The State Water Board established California's antidegradation policy in State Water Board Resolution No. 68-16, which incorporates the federal antidegradation policy where the federal policy applies under federal law and requires that existing quality of waters be maintained unless degradation is justified based on specific findings. The Basin Plan implements, and incorporates by reference, both the State and federal antidegradation policies.

N. Anti-Backsliding Requirements. CWA sections 402(o)(2) and 303(d)(4) and NPDES regulations at 40 CFR 122.44(l) prohibit backsliding in NPDES permits. These anti-backsliding provisions require effluent limitations in a reissued permit to be as stringent as those in the previous permit, with some exceptions where limitations may be relaxed.

O. Endangered Species Act. This Order does not authorize any act that results in the taking of a threatened or endangered species or any act that is now prohibited, or becomes prohibited in the future, under either the California Endangered Species Act (Fish and Game Code sections 2050 to 2097) or the federal Endangered Species Act (16 U.S.C.A. sections 1531 to 1544). This Order requires compliance with effluent limits, receiving water limits, and other requirements to protect the beneficial uses of waters of the State. The Discharger is responsible for meeting all requirements of applicable State and federal law pertaining to threatened and endangered species.

P. **Monitoring and Reporting.** NPDES regulations at 40 CFR 122.48 require that all NPDES permits specify requirements for recording and reporting monitoring results. CWC sections 13267 and 13383 authorize the Regional Water Board to require technical and monitoring reports. The Monitoring and Reporting Program (MRP, Attachment E) establishes monitoring and reporting requirements to implement federal and State requirements.

Q. **Standard and Special Provisions.** Attachment D contains Federal Standard Provisions that apply to all NPDES permits in accordance with 40 CFR 122.41, and additional conditions applicable to specified categories of permits in accordance with 40 CFR 122.42. The Discharger must comply with all standard provisions and with those additional conditions that apply under 40 CFR 122.42. The Discharger must also comply with the Regional Standard Provisions provided in Attachment G. The Regional Water Board has also included in this Order special provisions applicable to the Discharger. The attached Fact Sheet (Attachment F) provides rationales for the special provisions.

R. **Provisions and Requirements Implementing State Law.** None of the requirements in this Order are included to implement State law only.

S. **Notification of Interested Parties.** The Regional Water Board notified the Discharger and interested agencies and persons of its intent to prescribe WDRs for the discharge and provided them with an opportunity to submit written comments and recommendations. The Fact Sheet (Attachment F) provides details of the notification.

T. **Consideration of Public Comment.** The Regional Water Board, in a public meeting, heard and considered all comments pertaining to the discharge. The Fact Sheet (Attachment F) provides details of the public hearing.

IT IS HEREBY ORDERED, that this Order supersedes Order No. R2-2007-0008, except for enforcement purposes, and, in order to meet the provisions contained in CWC Division 7 (commencing with section 13000) and regulations adopted thereunder, and the provisions of the federal Clean Water Act (CWA) and regulations and guidelines adopted thereunder, the Discharger shall comply with the requirements in this Order.

III. DISCHARGE PROHIBITIONS

A. Discharge of treated wastewater at a location or in a manner different from that described in this Order is prohibited.

B. Discharge at any point at which the treated wastewater does not receive an initial dilution of at least 44:1 (nominal) is prohibited. Compliance shall be achieved by proper operation and maintenance of the discharge outfall to ensure that it (or its replacement, in whole or in part) is in good working order and is consistent with, or can achieve better mixing, than that described in the Fact Sheet (Attachment F). The Discharger shall address measures taken to ensure this in its application for permit reissuance.

C. The bypass of untreated or partially treated wastewater to waters of the United States is prohibited, except as provided for in the conditions stated in Subsections I.G of Attachment D of this Order.

D. The average dry weather effluent flow, measured at monitoring station EFF-001 as described in the attached MRP (Attachment E), shall not exceed 53.8 MGD. Actual average dry weather flow shall be determined for compliance with this prohibition over three consecutive dry weather months each year.

E. Any sanitary sewer overflow that results in a discharge of untreated or partially treated wastewater to waters of the United States is prohibited.

IV. EFFLUENT LIMITATIONS AND DISCHARGE SPECIFICATIONS

A. Effluent Limitations for Conventional and Non-Conventional Pollutants

The Discharger shall maintain compliance with the effluent limitations contained in Table 6 at Discharge Point 001, with compliance measured at Monitoring Location EFF-001, as described in the attached MRP (Attachment E).

Table 6. Conventional and Non-Conventional Pollutant Effluent Limitations

Parameter	Units	Effluent Limitations				
		Average Monthly	Average Weekly	Maximum Daily	Instantaneous Minimum	Instantaneous Maximum
Carbonaceous BOD 5-day @ 20°C (BOD_5)	mg/L	25	40	---	---	---
Total Suspended Solids (TSS)	mg/L	30	45	---	---	---
BOD and TSS percent removal [1]	%	85 minimum	---	---	---	---
Oil and Grease	mg/L	10	---	20	---	---
pH[2]	s.u	---	---	---	6.0	9.0
Enterococcus Bacteria	colonies per 100 mL	35[3]	---	---	---	---

Unit Abbreviations:

mg/L = milligrams per liter
s.u. = standard units
mL = milliliters

Footnotes to Table 6:

[1] 85 Percent Removal. The arithmetic mean of $CBOD_5$ at 20°C and TSS, by concentration, for effluent samples collected in each calendar month shall not exceed 15 percent of the arithmetic mean of the respective values, by concentration, for influent samples collected at INF-001 as described in the MRP (Attachment E) at approximately the same times during the same period.

[2] pH. If the Discharger monitors pH continuously, pursuant to 40 CFR 401.17, the Discharger shall be in compliance with the pH limitation specified herein provided that both of the following conditions are satisfied: (i) the total time during which the pH values are outside the required range of pH values shall not exceed 7 hours and 26 minutes in any calendar month; and (ii) no individual excursion from the range of pH values shall exceed 60 minutes.

[3] Enterococcus Bacteria. The monthly geometric mean shall not exceed 35 colonies per 100 mL.

B. Toxic Substances Effluent Limitations

The Discharger shall maintain compliance with the effluent limitations contained in Table 7 at Discharge Point 001, with compliance determined at Monitoring Location EFF-001, as described in the attached MRP (Attachment E).

Table 7. Toxic Pollutant Effluent Limitations

Constituent	Units	Effluent Limitations[1,2]	
		Average Monthly	Maximum Daily
Copper	µg/L	89	120
Cyanide	µg/L	22	39
Dioxin-TEQ	µg/L	1.4×10^{-8}	2.8×10^{-8}
Acrylonitrile	µg/L	6.3	13
Bis(2-ethylhexyl)phthalate	µg/L	55	170
Total Ammonia, as N	mg/L	65	84
Total Ammonia, as N	kg/day	5500	

Unit Abbreviations:

µg/L = micrograms per liter
mg/L = milligrams per liter
kg/day = kilograms per day

Footnotes to Table 7:

[1] Limitations apply to the average concentration of all samples collected during the averaging period (daily = 24-hour period; monthly = calendar month).

[2] All limitations for metals are expressed as total recoverable metals.

C. Whole Effluent Toxicity

1. Whole Effluent Acute Toxicity

a. Representative samples of the effluent at Discharge Point 001, with compliance measured at EFF-001 as described in the MRP (Attachment E), shall meet the following limits for acute toxicity. Bioassays shall be conducted in compliance with MRP section V.A (Attachment E.)

(1) An eleven (11) – sample median value of not less than 90 percent survival; and
(2) An eleven (11) – sample 90th percentile value of not less than 70 percent survival.

b. These acute toxicity limitations are further defined as follows:

(1) **11-sample median**. A bioassay test showing survival of less than 90 percent represents a violation of this effluent limit, if five or more of the past ten or less bioassay tests show less than 90 percent survival.

(2) **11-sample 90th percentile**. A bioassay test showing survival of less than 70 percent represents a violation of this effluent limit, if one or more of the past ten or less bioassay tests show less than 70 percent survival.

c. Bioassays shall be performed using the most up-to-date USEPA protocols and species as specified in MRP section V.A. Bioassays shall be conducted in compliance with *Methods for Measuring the Acute Toxicity of Effluents and Receiving Water to Freshwater and Marine Organisms*, currently 5th Edition (EPA-821-R-02-012), with exceptions granted by the Executive Officer, with exceptions granted by the Executive Officer and the Environmental Laboratory Accreditation Program (ELAP) upon the Discharger's request with justification.

2. Whole Effluent Chronic Toxicity

The discharge shall not contain chronic toxicity at a level that would cause or contribute to toxicity in the receiving water. Chronic toxicity is a detrimental biological effect of growth rate, reproduction, fertilization success, larval development, or any other relevant measure of the health of an organism population or community. Compliance with this limit shall be determined by analysis of indicator organisms and toxicity tests measured at EFF-001 as described in the MRP.

V. RECEIVING WATER LIMITATIONS

The discharges shall not cause the following in the receiving water:

A. The discharge of waste shall not cause the following conditions to exist in waters of the State at any place:

1. Floating, suspended, or deposited macroscopic particulate matter or foams;

2. Bottom deposits or aquatic growths to the extent that such deposits or growths cause nuisance or adversely affect beneficial uses;

3. Alteration of temperature, turbidity, or apparent color beyond present natural background levels;

4. Visible, floating, suspended, or deposited oil or other products of petroleum origin; and

5. Toxic or other deleterious substances to be present in concentrations or quantities that cause deleterious effects on wildlife, waterfowl, or other aquatic biota, or that render any of these unfit for human consumption, either at levels created in the receiving waters or as a result of biological concentration.

B. The discharge of waste shall not cause the following limits to be exceeded in waters of the State at any place within 1 foot of the water surface:

1. Dissolved Oxygen — 7.0 mg/L, minimum

Furthermore, the median dissolved oxygen concentration for any three consecutive months shall not be less than 80% of the dissolved oxygen content at saturation. When natural factors cause concentrations less than that specified above, the discharge shall not cause further reduction in ambient dissolved oxygen concentrations.

2.	Dissolved Sulfide	Natural background levels
3.	pH	The pH shall not be depressed below 6.5 or raised above 8.5. The discharge shall not cause changes greater than 0.5 pH units in normal ambient pH levels.
4.	Nutrients	Waters shall not contain biostimulatory substances in concentrations that promote aquatic growths to the extent that such growths cause nuisance or adversely affect beneficial uses.

C. The discharge shall not cause a violation of any particular water quality standard for receiving waters adopted by the Regional Water Board or the State Water Board as required by the CWA and regulations adopted thereunder. If more stringent applicable water quality standards are promulgated or approved pursuant to CWA section 303, or amendments thereto, the Regional Water Board may revise and modify this Order in accordance with such more stringent standards.

VI. PROVISIONS

A. Standard Provisions

1. **Federal Standard Provisions.** The Discharger shall comply with Federal Standard Provisions included in Attachment D of this Order.

2. **Regional Standard Provisions.** The Discharger shall comply with all applicable items of the Regional Standard Provisions, and Monitoring and Reporting Requirements (Supplement to Attachment D) for NPDES Wastewater Discharge Permits (Attachment G), including amendments thereto.

B. MRP Requirements

The Discharger shall comply with the MRP (Attachment E), and future revisions thereto, including applicable sampling and reporting requirements in the standard provisions listed in Provision VI.A above.

C. Special Provisions

1. Reopener Provisions

The Regional Water Board may modify or reopen this Order prior to its expiration date in any of the following circumstances as allowed by law:

a. If present or future investigations demonstrate that the discharges governed by this Order have or will have a reasonable potential to cause or contribute to, or will cease to have, adverse impacts on water quality or beneficial uses of the receiving waters.

b. If new or revised WQOs or total maximum daily loads (TMDLs) come into effect for the San Francisco Bay Estuary and contiguous water bodies (whether Statewide, regional, or site-specific). In such cases, effluent limitations in this Order will be modified as necessary to reflect updated WQOs and waste load allocations in TMDLs. Adoption of effluent limitations contained in this Order is not intended to restrict in any way future modifications based on legally adopted WQOs or TMDLs, or as otherwise permitted under federal regulations governing NPDES permit modifications.

c. If data, results, or other information developed in connection with translator, dilution, or other water quality studies (such as, but not limited to, studies related to Suisun Bay ammonium effects, including, but not limited to, studies conducted pursuant to Provision VI.C.5.c) provide a basis for determining that a permit condition, such as but not limited to ammonia effluent limitations, should be modified.

d. If State Water Board precedential decisions, new policies, new laws, or new regulations on chronic toxicity or total chlorine residual become available.

e. If an administrative or judicial decision on a separate NPDES permit or WDRs addresses requirements similar to this discharge.

f. If the Discharger requests adjustments in effluent limits due to the implementation of a stormwater diversion pursuant to the Municipal Regional Stormwater Permit (Order No. R2-2009-0074), for redirecting dry weather and first flush discharges from the storm drain system to the sanitary sewer system as a stormwater pollutant control strategy.

g. Or as otherwise authorized by law.

The Discharger may request permit modification based on any of the circumstances described above. With any such request, the Discharger shall include an antidegradation and antibacksliding analysis.

2. Effluent Characterization Study and Report

a. Study Elements

The Discharger shall collect representative samples of the discharge at EFF-001, as defined MRP (Attachment E), at least once per year.

The samples shall be analyzed for the priority pollutants listed in Table C of the Regional Standard Provisions (Attachment G), except for those priority pollutants with effluent limitations where monitoring is already required by the MRP. Compliance with this requirement shall be achieved in accordance with the specifications of Regional Standard Provisions (Attachment G) sections III.A.1 and III.A.2.

The Discharger shall evaluate on an annual basis if concentrations of any of these priority pollutants significantly increase over past performance. The Discharger shall investigate the cause of such increase. The investigation may include, but need not be limited to, an increase in monitoring frequency, monitoring of internal process streams, or monitoring of influent sources. The Discharger shall establish remedial measures addressing any increase resulting in reasonable potential to cause or contribute to an excursion above applicable water quality objectives. This requirement may be satisfied through identification of the constituent as a "pollutant of concern" in the Discharger's Pollutant Minimization Program, described in Provision VI.C.3.

b. Reporting Requirements

(1) Routine Reporting

The Discharger shall, within 30 days of receipt of analytical results, report in the transmittal letter for the appropriate monthly self-monitoring report the following:

(a) Indication that a sample or samples for this characterization study was or were collected; and

(b) Identity of priority pollutants detected above their applicable water quality criteria (see Fact Sheet [Attachment F] Table F-8 for the criteria), together with the detected concentrations of those pollutants.

(2) Annual Reporting

The Discharger shall provide a summary of the annual data evaluation and source investigation in the annual self-monitoring report.

(3) Final Report

The Discharger shall submit a final report that presents all these data to the Regional Water Board no later than 180 days prior to the Order expiration date. The final report shall be submitted with the application for permit reissuance.

3. Best Management Practices and Pollutant Minimization Program

a. The Discharger shall continue to improve, in a manner acceptable to the Executive Officer, its existing Pollutant Minimization Program to promote minimization of pollutant loadings to the treatment plant and therefore to the receiving waters.

b. The Discharger shall submit an annual report, acceptable to the Executive Officer, no later than February 28 of each calendar year. Each annual report shall include at least the following information:

(1) *A brief description of the treatment plant, treatment plant processes and service area.*

(2) *A discussion of the current pollutants of concern.* Periodically, the Discharger shall analyze its own situation to determine which pollutants are currently a

problem and which pollutants may be potential future problems. This discussion shall include the reasons for choosing the pollutants.

(3) *Identification of sources for the pollutants of concern.* This discussion shall include how the Discharger intends to estimate and identify sources of the pollutants. The Discharger shall also identify sources or potential sources not directly within the ability or authority of the Discharger to control, such as pollutants in the potable water supply and air deposition.

(4) *Identification of tasks to reduce the sources of the pollutants of concern.* This discussion shall identify and prioritize tasks to address the Discharger's pollutants of concern. The Discharger may implement the tasks by itself or participate in group, regional, or national tasks that will address its pollutants of concern. The Discharger is strongly encouraged to participate in group, regional, or national tasks that will address its pollutants of concern whenever it is efficient and appropriate to do so. A time line shall be included for the implementation of each task.

(5) *Outreach to employees.* The Discharger shall inform employees about the pollutants of concern, potential sources, and how they might be able to help reduce the discharge of these pollutants of concern into the treatment facilities. The Discharger may provide a forum for employees to provide input.

(6) *Continuation of Public Outreach Program.* The Discharger shall prepare a public outreach program to communicate pollution prevention to its service area. Outreach may include participation in existing community events such as county fairs, initiating new community events such as displays and contests during Pollution Prevention Week, conducting school outreach programs, conducting plant tours, and providing public information in newspaper articles or advertisements, radio or television stories or spots, newsletters, utility bill inserts, and web site. Information shall be specific to the target audiences. The Discharger shall coordinate with other agencies as appropriate.

(7) *Discussion of criteria used to measure Pollutant Minimization Program and task effectiveness.* The Discharger shall establish criteria to evaluate the effectiveness of its Pollutant Minimization Program. This section shall discuss the specific criteria used to measure the effectiveness of each of the tasks in sections VI.C.3.b.(3), (4), (5), and (6).

(8) *Documentation of efforts and progress.* This discussion shall detail all of the Discharger's Pollutant Minimization Program activities during the reporting year.

(9) *Evaluation of Pollutant Minimization Program and task effectiveness.* This Discharger shall use the criteria established in section VI.C.3.b.(7) to evaluate the Program's and tasks' effectiveness.

(10) *Identification of specific tasks and time schedules for future efforts.* Based on the evaluation, the Discharger shall detail how it intends to continue or change its tasks in order to more effectively reduce the amount of pollutants to the treatment plant, and subsequently in its effluent.

c. Pollutant Minimization Program for Pollutants with Effluent Limitations

The Discharger shall develop and conduct a Pollutant Minimization Program as further described below when there is evidence that a priority pollutant is present in the effluent above an effluent limitation (e.g., sample results reported as DNQ when the effluent limitation is less than the MDL, sample results from analytical methods more sensitive than those methods required by this Order, presence of whole effluent toxicity, health advisories for fish consumption, results of benthic or aquatic organism tissue sampling) and either:

(1) A sample result is reported as DNQ and the effluent limitation is less than the RL; or

(2) A sample result is reported as ND and the effluent limitation is less than the MDL, using SIP definitions.

d. Pollutant Minimization Program Submittals for Pollutants with Effluent Limitations

If triggered by the reasons in section VI.C.3.c, above, the Discharger's Pollutant Minimization Program shall include, but not be limited to, the following actions and submittals acceptable to the Regional Water Board:

(1) Annual review and semi-annual monitoring of potential sources of the reportable priority pollutants, which may include fish tissue monitoring and other bio-uptake sampling, or alternative measures approved by the Executive Officer when it is demonstrated that source monitoring is unlikely to produce useful analytical data;

(2) Quarterly monitoring for the reportable priority pollutants in the influent to the wastewater treatment system, or an alternative measures approved by the Executive Officer, when it is demonstrated that influent monitoring is unlikely to produce useful analytical data;

(3) Submittal of a control strategy designed to proceed toward the goal of maintaining concentrations of the reportable priority pollutants in the effluent at or below the effluent limitation;

(4) Implementation of appropriate cost-effective control measures for the reportable priority pollutants, consistent with the control strategy; and

(5) Annual report required by section VI.C.3.b above, shall specifically address the following items:

(a) All Pollutant Minimization Program monitoring results for the previous year;

(b) List of potential sources of the reportable priority pollutants;

(c) Summary of all actions undertaken pursuant to the control strategy; and

(d) Description of actions to be taken in the following year.

4. Special Provisions for POTWs

a. Pretreatment Program

The Discharger shall implement and enforce its approved pretreatment program in accordance with federal Pretreatment Regulations (40 CFR 403), pretreatment standards promulgated under CWA sections 307(b), 307(c), and 307(d), pretreatment requirements specified under 40 CFR 122.44(j), and the requirements in Attachment H, "Pretreatment Requirements." The Discharger's responsibilities include, but are not limited to:

(1) Enforcement of National Pretreatment Standards of 40 CFR 403.5 and 403.6;

(2) Implementation of its pretreatment program in accordance with legal authorities, policies, procedures, and financial provisions described in the National Pretreatment Program (40 CFR 403).

(3) Submission of reports to the State Water Board and the Regional Water Board as described in Attachment H, "Pretreatment Requirements."

(4) Evaluation of the need to revise local limits under 40 CFR 403.5(c)(1) and, within 180 days after the effective date of this Order, submission of a report acceptable to the Executive Officer describing the changes, with a plan and schedule for implementation. To ensure no significant increase in copper discharges, and thus compliance with antidegradation requirements, the Discharger shall not consider eliminating or relaxing local limits for copper in this evaluation.

b. Biosolids Management Practices

(1) All biosolids shall be disposed of, managed or reused in a municipal solid waste landfill, through land application, as a Class A compost, through a waste to energy facility, or other recognized and approved technology, disposed of in a sludge-only landfill or fired in a sewage sludge incinerator in accordance with 40 CFR Part 503.

(2) Biosolids treatment, storage and disposal or reuse shall not create a nuisance, such as objectionable odors or flies, or result in groundwater contamination.

(3) The biosolids treatment and storage site shall have facilities adequate to divert surface runoff from adjacent areas, to protect boundaries of the site from erosion,

and to prevent any conditions that would cause drainage from the materials in the temporary storage site. Adequate protection is defined as protection from at least a 100-year storm and protection from the highest possible tidal stage that may occur.

(4) Biosolids disposed of in a municipal solid waste landfill shall meet the requirements of 40 CFR Part 258. In the annual Self-Monitoring Report, the Discharger shall include the amount of biosolids disposed and the landfill to which it was sent.

(5) This Order does not authorize permanent on-site biosolids storage or disposal. A Report of Waste Discharge shall be filed and the site brought into compliance with all applicable regulations prior to commencement of any such activity.

c. Sanitary Sewer Overflows and Sewer System Management Plan

The Discharger's collection system is part of the Facility that is subject to this Order. As such, the Discharger shall properly operate and maintain its collection system (Attachment D, Standard Provisions - Permit Compliance, subsection I.D). The Discharger shall report any noncompliance (Attachment D, Standard Provision - Reporting, sections V.E.1 and V.E.2) and mitigate any discharge from the Discharger's collection system in violation of this Order (Attachment D, Standard Provisions - Permit Compliance, section I.C).

The General Waste Discharge Requirements for Wastewater Collection Agencies, State Water Board Order No. 2006-0003 DWQ (General Collection System WDRs), has requirements for operation and maintenance of collection systems and for reporting and mitigating sanitary sewer overflows. While the Discharger must comply with both the General Collection System WDRs and this Order, the General Collection System WDRs more clearly and specifically stipulates requirements for operation and maintenance and for reporting and mitigating sanitary sewer overflows.

Implementation of the General Collection System WDRs requirements for proper operation and maintenance and mitigation of spills will satisfy the corresponding federal NPDES requirements specified in Attachment D (as supplemented by Attachment G) of this Order. Following notification and reporting requirements in the General Collection System WDRs will satisfy NPDES reporting requirements specified in Attachment D (as supplemented by Attachment G) of the Order for sewage spills from the collection system upstream of the Plant boundaries. Attachments D and G of this Order specify reporting requirements for unauthorized discharges from anywhere within the Plant downstream of the Plant boundaries.

5. Other Special Provisions

a. Copper Action Plan

The Discharger shall implement pretreatment, source control, and pollution prevention for copper in accordance with the following tasks and time schedule.

Table 8. Copper Action Plan

Task	Compliance Date
1. **Review Potential Copper Sources** The Discharger shall submit an inventory of potential copper sources to the treatment plant.	March 1, 2012
2. **Implement Copper Control Program** The Discharger shall submit a plan for and begin implementation of a program to reduce copper sources identified in Task 1. The plan shall consists, at a minimum, of the following elements: a. Provide education and outreach to the public (e.g., focus on proper pool and spa maintenance and plumbers' roles in reducing corrosion.) b. If corrosion is determined to be a significant copper source, work cooperatively with local water purveyors to reduce and control water corrosivity, as appropriate, and ensure that local plumbing contractors implement best management practices to reduce corrosion in pipes. c. Educate plumbers, designers, and maintenance contractors for pools and spas to encourage best management practices that minimize copper discharges.	With the annual pollution prevention report due February 28, 2013.
3. **Implement Additional Measures** If the Regional Water Board notifies the Discharger that the three-year rolling mean copper concentration of the receiving water exceeds 2.8 μg/L, then within 90 days of the notification, the Discharger shall evaluate the effluent copper concentration trend, and if it is increasing, develop and begin implementation of additional measures to control copper discharges. The Discharger shall report on the progress and effectiveness of action taken together with a schedule for actions to be taken in the next 12 months.	With the annual pollution prevention report due February 28 following 90 days after notification.
4. **Undertake Studies to Reduce Copper Pollutant Impact Uncertainties** The Discharger shall submit an updated study plan and schedule to conduct or cause to be conducted technical studies to investigate possible copper sediment toxicity and technical studies to investigate sublethal effects on salmonids. Specifically, the Discharger shall include the manner in which the above will be accomplished and describe the studies to be performed with an implementation schedule. To satisfy this requirement, the Discharger may collaborate and conduct these studies as a group.	Study Plan already submitted by Bay Area Clean Water Agencies satisfies this requirement
5. **Report Status of Copper Control Program** The Discharger shall submit an annual report documenting copper control program implementation and addressing the effectiveness of the actions taken including any additional copper controls required by Task 3 above, together with a schedule for actions to be taken in the next 12 months. Additionally, the Discharger shall report the findings and results of the studies completed, planned, or in progress under Task 4. Regarding Task 4 studies dischargers may collaborate and provide this information in a single report to satisfy this requirement for an entire group.	With annual pollution prevention report due February 28 each year, commencing February 28, 2014.

b. Cyanide Action Plan

The Discharger shall implement monitoring and surveillance, pretreatment, source control and pollution prevention for cyanide in accordance with the following tasks and time schedule.

Table 9. Cyanide Action Plan

Task	Compliance Date
1. Review Potential Cyanide Sources The Discharger shall submit an inventory of potential cyanide sources to the treatment plant. If no cyanide sources are identified, Tasks 2 and 3 are not required, unless the Discharger receives a request to discharge detectable levels of cyanide to the sewer. If so, the Discharger shall notify the Executive Officer and implement Tasks 2 and 3.	Completed 2008
2. Implement Cyanide Control Program The Discharger shall submit a plan and begin implementation of a program to minimize cyanide discharges to its treatment plant consisting, at a minimum, of the following elements: a. Inspect each potential source to assess the need to include that contributing source in the control program. b. Inspect contributing sources included in the control program annually. Inspection elements may be based on USEPA guidance, such as Industrial User Inspection and Sampling Manual for POTWs (EPA 831-B-94-01). c. Develop and distribute educational materials to contributing sources and potential contributing sources regarding the need to prevent cyanide discharges. d. Prepare an emergency monitoring and response plan to be implemented if a significant cyanide discharge occurs.	With annual pollution prevention report due February 28, 2012
3. Implement Additional Cyanide Control Measures If the Regional Water Board notifies the Discharger that ambient monitoring shows cyanide concentrations are 1.0 μg/L or higher in the main body of San Francisco Bay, then within 90 days of the notification, the Discharger shall commence actions to identify and abate cyanide sources responsible for the elevated ambient concentrations, and shall report on the progress and effectiveness of actions taken, together with a schedule for actions to be taken in the next 12 months.	With next annual pollution prevention report due February 28 (at least 90 days following notification)
4. Report Status of Cyanide Control Program The Discharger shall submit an annual report documenting cyanide control program implementation and addressing the effectiveness of actions taken, including any additional cyanide controls required by Task 3, above, together with a schedule for actions to be taken in the next 12 months.	With annual pollution prevention report due February 28 each year

c. Nutrient Discharge Work Plan, Studies, and Reports

i. **Draft Work Plan.** By June 1, 2012, the Discharger shall submit to the Regional Water Board a draft work plan to conduct the studies listed in item c.iii, below, to evaluate further the effects on Suisun Bay of ammonia, ammonium, and other nutrients in its discharge. The Discharger may complete these studies itself or in conjunction with others, including but not limited to the State and Federal

Contractors Water Agency, the State Water Contractors, and the San Luis & Delta-Mendota Water Authority (collectively, "Water Contractors"); the Bay Area Clean Water Agencies; and the Regional Water Board. The draft work plan shall call for the studies to be completed no later than September 1, 2014.

The draft work plan shall delineate a process to disseminate study results for stakeholder review. The Discharger shall distribute the draft work plan to stakeholders, including but not limited to the Water Contractors.

ii. **Final Work Plan.** By August 1, 2012, the Discharger shall submit a final work plan that incorporates Executive Officer feedback on the draft work plan.

iii. **Work Plan Elements.** The work plan shall include schedules and commitments to fund the following:

(a) Surface Water Ambient Monitoring Program sampling and associated studies set forth in *San Francisco Bay Region Work Plan, Monitoring Spring Phytoplankton Bloom Progression in Suisun Bay* (Taberski, Dugdale, et al., SWAMP Monitoring Plan 2011-2012, December 2010). The Discharger shall commit technical expertise, laboratory support, and funding for the studies. Specifically, the Discharger shall fund an additional sample site to characterize the San Joaquin River delta input, analyze samples for nutrients and metals, and fund analysis for pesticides.

(b) Collection of representative effluent samples sufficient to characterize nutrient forms, concentrations, and loads. The data to be obtained shall include the form and ratios of nitrogen and phosphorus, including organic and inorganic nitrogen and phosphorus. (Regional Water Board staff intends to obtain such information soon from most wastewater dischargers in the Region.)

(c) Collaborative study of the Discharger's contribution to ammonium concentrations in Suisun Bay and related toxicity to copepods in the context of Suisun Bay. These studies shall include, to the extent possible, an evaluation of acute toxicity to copepod larvae (nauplii) and full life cycle toxicity. The study shall use a methodology acceptable to the Executive Officer.

(d) Collaborative studies evaluating the role of ammonia and ammonium in primary productivity and zooplankton abundance, the significance of nutrient ratios, nutrient fate and transport, and the role of sediment biogeochemistry in nutrient fluxes. Such studies would include, for example, a determination whether sampling locations adequately characterize the potential impact of the Discharger's discharge and those studies committed to by the Bay Area Clean Water Agencies to be conducted by the Aquatic Science Center and the San Francisco Estuary Institute (Chastain, Bay Area Clean Water Agencies, "Nutrient Strategy Development and Implementation: A proposal to

BACWA and the San Francisco Bay Regional Water Quality Control Board," January 18, 2012).

iv. **Final Report.** The Discharger shall implement the final work plan described in item c.ii, above, and, by November 1, 2014, submit a final report acceptable to the Executive Officer regarding the results of the studies completed pursuant to the final work plan.

d. Facility Plan and Site Characterization

i. **Work Plan.** By July 1, 2012, the Discharger shall submit a work plan to evaluate alternative treatment technologies to remove ammonia from its discharge, including nitrification technologies. The evaluation shall include facility planning for a range of potential ammonia effluent limits and pilot scale systems analyses. The Discharger shall evaluate the suitability of the Facility and property owned or controlled by the Discharger to provide land necessary for ammonia treatment and removal. As part of this evaluation, the Discharger shall conduct sampling to characterize sufficiently the portion of its property where materials previously placed for disposal would have to be managed to develop a nitrification treatment train.

ii. **Report.** By February 28, 2014, the Discharger shall provide a report acceptable to the Executive Officer containing the conclusions of the studies completed pursuant to item d.i, above.

VII.COMPLIANCE DETERMINATION

Compliance with effluent limitations for priority pollutants shall be determined using sample reporting protocols defined in Attachment A—Definitions, the MRP (Attachment E), Fact Sheet section VI, and the Regional Standard Provisions (Attachment G). For purposes of reporting and administrative enforcement by the Regional and State Water Boards, the Discharger shall be deemed out of compliance with effluent limitations if the concentration of the priority pollutant in the monitoring sample is greater than the effluent limitation and greater than or equal to the reporting level (RL).

ATTACHMENT A – DEFINITIONS

Arithmetic Mean (μ)
Also called the average, is the sum of measured values divided by the number of samples. For ambient water concentrations, the arithmetic mean is calculated as follows:

Arithmetic mean = $\mu = \Sigma x / n$ where: Σx is the sum of the measured ambient water concentrations, and n is the number of samples.

Average Monthly Effluent Limitation (AMEL)
The highest allowable average of daily discharges over a calendar month, calculated as the sum of all daily discharges measured during a calendar month divided by the number of daily discharges measured during that month.

Average Weekly Effluent Limitation (AWEL)
The highest allowable average of daily discharges over a calendar week (Sunday through Saturday), calculated as the sum of all daily discharges measured during a calendar week divided by the number of daily discharges measured during that week.

Bioaccumulative
Those substances taken up by an organism from its surrounding medium through gill membranes, epithelial tissue, or from food and subsequently concentrated and retained in the body of the organism.

Carcinogenic
Carcinogenic pollutants are substances that are known to cause cancer in living organisms.

Coefficient of Variation (*CV*)
CV is a measure of the data variability and is calculated as the estimated standard deviation divided by the arithmetic mean of the observed values.

Daily Discharge
Daily Discharge is defined as either: (1) the total mass of the constituent discharged over the calendar day (12:00 am through 11:59 pm) or any 24-hour period that reasonably represents a calendar day for purposes of sampling (as specified in this Order), for a constituent with limitations expressed in units of mass or; (2) the unweighted arithmetic mean measurement of the constituent over the day for a constituent with limitations expressed in other units of measurement (e.g., concentration).

The daily discharge may be determined by the analytical results of a composite sample taken over the course of one day (a calendar day or other 24-hour period defined as a day) or by the arithmetic mean of analytical results from one or more grab samples taken over the course of the day.

For composite sampling, if 1 day is defined as a 24-hour period other than a calendar day, the analytical result for the 24-hour period will be considered as the result for the calendar day in which the 24-hour period ends.

Detected, but Not Quantified (DNQ)
DNQ are those sample results less than the RL, but greater than or equal to the laboratory's MDL.

Dilution Credit
Dilution Credit is the amount of dilution granted to a discharge in the calculation of a water quality-based effluent limitation, based on the allowance of a specified mixing zone. It is calculated from the dilution ratio or determined through conducting a mixing zone study or modeling of the discharge and receiving water.

Effluent Concentration Allowance (ECA)
ECA is a value derived from the water quality criterion/objective, dilution credit, and ambient background concentration that is used, in conjunction with the coefficient of variation for the effluent monitoring data, to calculate a long-term average (LTA) discharge concentration. The ECA has the same meaning as waste load allocation (WLA) as used in USEPA guidance (Technical Support Document For Water Quality-based Toxics Control, March 1991, second printing, EPA/505/2-90-001).

Enclosed Bays
Enclosed Bays means indentations along the coast that enclose an area of oceanic water within distinct headlands or harbor works. Enclosed bays include all bays where the narrowest distance between the headlands or outermost harbor works is less than 75 percent of the greatest dimension of the enclosed portion of San Francisco Bay. Enclosed bays include, but are not limited to, Humboldt Bay, Bodega Harbor, Tomales Bay, Drake's Estero, San Francisco Bay, Morro Bay, Los Angeles-Long Beach Harbor, Upper and Lower Newport Bay, Mission Bay, and San Diego Bay. Enclosed bays do not include inland surface waters or ocean waters.

Estimated Chemical Concentration
The estimated chemical concentration that results from the confirmed detection of the substance by the analytical method below the ML value.

Estuaries
Estuaries means waters, including coastal lagoons, located at the mouths of streams that serve as areas of mixing for fresh and ocean waters. Coastal lagoons and mouths of streams that are temporarily separated from the ocean by sandbars shall be considered estuaries. Estuarine waters shall be considered to extend from a bay or the open ocean to a point upstream where there is no significant mixing of fresh water and seawater. Estuarine waters include, but are not limited to, the Sacramento-San Joaquin Delta, as defined in California Water Code section 12220, Suisun Bay, Carquinez Strait downstream to the Carquinez Bridge, and appropriate areas of the Smith, Mad, Eel, Noyo, Russian, Klamath, San Diego, and Otay rivers. Estuaries do not include inland surface waters or ocean waters.

Inland Surface Waters
All surface waters of the State that do not include the ocean, enclosed bays, or estuaries.

Instantaneous Maximum Effluent Limitation
The highest allowable value for any single grab sample or aliquot (i.e., each grab sample or aliquot is independently compared to the instantaneous maximum limitation).

Instantaneous Minimum Effluent Limitation
The lowest allowable value for any single grab sample or aliquot (i.e., each grab sample or aliquot is independently compared to the instantaneous minimum limitation).

Maximum Daily Effluent Limitation (MDEL)
The highest allowable daily discharge of a pollutant, over a calendar day (or 24-hour period). For pollutants with limitations expressed in units of mass, the daily discharge is calculated as the total mass of the pollutant discharged over the day. For pollutants with limitations expressed in other units of measurement, the daily discharge is calculated as the arithmetic mean measurement of the pollutant over the day.

Median
The middle measurement in a set of data. The median of a set of data is found by first arranging the measurements in order of magnitude (either increasing or decreasing order). If the number of measurements (*n*) is odd, then the median = $X_{(n+1)/2}$. If *n* is even, then the median = $(X_{n/2} + X_{(n/2)+1})/2$ (i.e., the midpoint between the *n*/2 and *n*/2+1).

Method Detection Limit (MDL)
MDL is the minimum concentration of a substance that can be measured and reported with 99 percent confidence that the analyte concentration is greater than zero, as defined in title 40 of the Code of Federal Regulations (40 CFR), Part 136, Attachment B, revised as of July 3, 1999.

Minimum Level (ML)
ML is the concentration at which the entire analytical system must give a recognizable signal and acceptable calibration point. The ML is the concentration in a sample that is equivalent to the concentration of the lowest calibration standard analyzed by a specific analytical procedure, assuming that all the method specified sample weights, volumes, and processing steps have been followed.

Mixing Zone
Mixing Zone is a limited volume of receiving water that is allocated for mixing with a wastewater discharge where water quality criteria can be exceeded without causing adverse effects to the overall water body.

Not Detected (ND)
Sample results less than the laboratory's MDL.

Ocean Waters
The territorial marine waters of the State as defined by California law to the extent these waters are outside of enclosed bays, estuaries, and coastal lagoons. Discharges to ocean waters are regulated in accordance with the State Water Board's California Ocean Plan.

Persistent Pollutants
Persistent pollutants are substances for which degradation or decomposition in the environment is nonexistent or very slow.

Pollutant Minimization Program (PMP)
PMP means waste minimization and pollution prevention actions that include, but are not limited to, product substitution, waste stream recycling, alternative waste management methods, and education of the public and businesses. The goal of the PMP shall be to reduce all potential sources of a priority pollutant(s) through pollutant minimization (control) strategies, including pollution prevention measures as appropriate, to maintain the effluent concentration at or below the water quality-based effluent limitation. Pollution prevention measures may be particularly appropriate for persistent bioaccumulative priority pollutants where there is evidence that beneficial uses are being impacted. The Regional Water

Board may consider cost effectiveness when establishing the requirements of a PMP. The completion and implementation of a Pollution Prevention Plan, if required pursuant to California Water Code section 13263.3(d), shall be considered to fulfill the PMP requirements.

Pollution Prevention
Pollution Prevention means any action that causes a net reduction in the use or generation of a hazardous substance or other pollutant that is discharged into water and includes, but is not limited to, input change, operational improvement, production process change, and product reformulation (as defined in California Water Code section 13263.3). Pollution prevention does not include actions that merely shift a pollutant in wastewater from one environmental medium to another environmental medium, unless clear environmental benefits of such an approach are identified to the satisfaction of the State or Regional Water Board.

Reporting Level (RL)
RL is the ML (and its associated analytical method) chosen by the Discharger for reporting and compliance determination from the MLs included in this Order. The MLs included in this Order correspond to approved analytical methods for reporting a sample result that are selected by the Regional Water Board either from Appendix 4 of the SIP in accordance with section 2.4.2 of the SIP or established in accordance with section 2.4.3 of the SIP. The ML is based on the proper application of method-based analytical procedures for sample preparation and the absence of any matrix interferences. Other factors may be applied to the ML depending on the specific sample preparation steps employed. For example, the treatment typically applied in cases where there are matrix-effects is to dilute the sample or sample aliquot by a factor of ten. In such cases, this additional factor must be applied to the ML in the computation of the RL.

Satellite Collection System
The portion, if any, of a sanitary sewer system owned or operated by a different public agency than the agency that owns and operates the wastewater treatment facility that a sanitary sewer system is tributary to.

Source of Drinking Water
Any water designated as municipal or domestic supply (MUN) in a Regional Water Board Basin Plan.

Standard Deviation (σ)
Standard Deviation is a measure of variability that is calculated as follows:

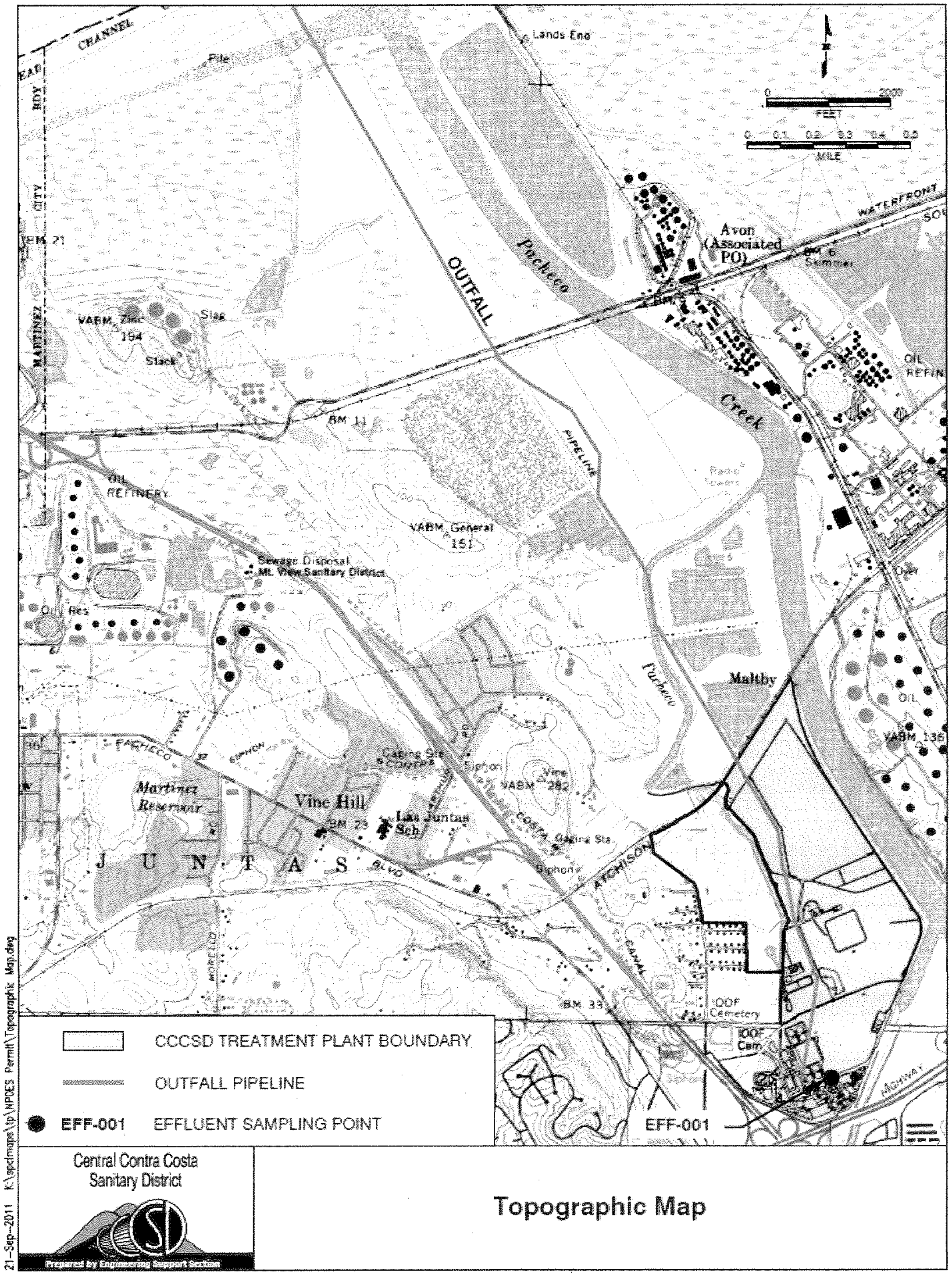
$$\sigma = (\sum[(x - \mu)^2]/(n - 1))^{0.5}$$

where:
x is the observed value;
μ is the arithmetic mean of the observed values; and
n is the number of samples.

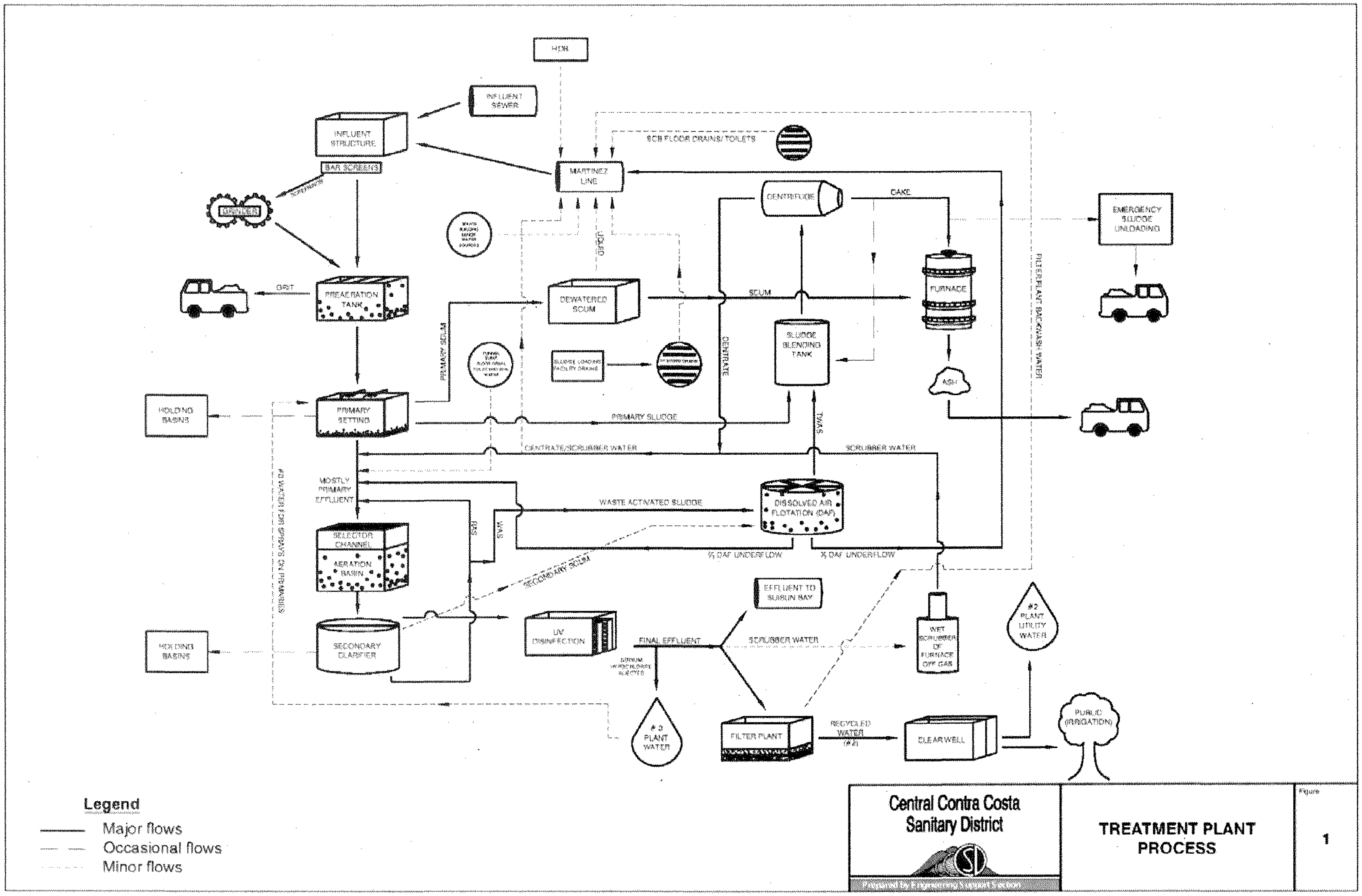
Toxicity Reduction Evaluation (TRE)
TRE is a study conducted in a step-wise process designed to identify the causative agents of effluent or ambient toxicity, isolate the sources of toxicity, evaluate the effectiveness of toxicity control options, and then confirm the reduction in toxicity. The first steps of the TRE consist of the collection of data relevant to the toxicity, including additional toxicity testing, and an evaluation of facility operations and maintenance practices, and best management practices. A Toxicity Identification Evaluation (TIE) may be required as part of the TRE, if appropriate. (A TIE is a set of procedures to identify the specific

chemical(s) responsible for toxicity. These procedures are performed in three phases (characterization, identification, and confirmation) using aquatic organism toxicity tests.)

ATTACHMENT B – FACILITY MAP



ATTACHMENT C – PROCESS FLOW DIAGRAM



ATTACHMENT D –STANDARD PROVISIONS

I. STANDARD PROVISIONS – PERMIT COMPLIANCE

A. Duty to Comply

1. The Discharger must comply with all of the conditions of this Order. Any noncompliance constitutes a violation of the Clean Water Act (CWA) and the California Water Code and is grounds for enforcement action, for permit termination, revocation and reissuance, or modification; or denial of a permit renewal application. (40 CFR 122.41(a)).

2. The Discharger shall comply with effluent standards or prohibitions established under section 307(a) of the CWA for toxic pollutants and with standards for sewage sludge use or disposal established under section 405(d) of the CWA within the time provided in the regulations that establish these standards or prohibitions, even if this Order has not yet been modified to incorporate the requirement. (40 CFR 122.41(a)(1)).

B. Need to Halt or Reduce Activity Not a Defense

It shall not be a defense for a Discharger in an enforcement action that it would have been necessary to halt or reduce the permitted activity in order to maintain compliance with the conditions of this Order. (40 CFR 122.41(c).)

C. Duty to Mitigate

The Discharger shall take all reasonable steps to minimize or prevent any discharge or sludge use or disposal in violation of this Order that has a reasonable likelihood of adversely affecting human health or the environment. (40 CFR 122.41(d).)

D. Proper Operation and Maintenance

The Discharger shall at all times properly operate and maintain all facilities and systems of treatment and control (and related appurtenances) which are installed or used by the Discharger to achieve compliance with the conditions of this Order. Proper operation and maintenance also includes adequate laboratory controls and appropriate quality assurance procedures. This provision requires the operation of backup or auxiliary facilities or similar systems that are installed by a Discharger only when necessary to achieve compliance with the conditions of this Order (40 CFR 122.41(e)).

E. Property Rights

1. This Order does not convey any property rights of any sort or any exclusive privileges. (40 CFR 122.41(g).)

2. The issuance of this Order does not authorize any injury to persons or property or invasion of other private rights, or any infringement of state or local law or regulations. (40 CFR 122.5(c).)

F. Inspection and Entry

The Discharger shall allow the Regional Water Board, State Water Board, USEPA and/or their authorized representatives (including an authorized contractor acting as their representative), upon the presentation of credentials and other documents, as may be required by law, to (40 CFR 122.41(i); Wat. Code, § 13383):

1. Enter upon the Discharger's premises where a regulated facility or activity is located or conducted, or where records are kept under the conditions of this Order (40 CFR 122.41(i)(1));

2. Have access to and copy, at reasonable times, any records that must be kept under the conditions of this Order (40 CFR 122.41(i)(2));

3. Inspect and photograph, at reasonable times, any facilities, equipment (including monitoring and control equipment), practices, or operations regulated or required under this Order (40 CFR 122.41(i)(3)); and

4. Sample or monitor, at reasonable times, for the purposes of assuring Order compliance or as otherwise authorized by the CWA or the Water Code, any substances or parameters at any location. (40 CFR 122.41(i)(4).)

G. Bypass

1. Definitions

   a. "Bypass" means the intentional diversion of waste streams from any portion of a treatment facility. (40 CFR 122.41(m)(1)(i).)

   b. "Severe property damage" means substantial physical damage to property, damage to the treatment facilities, which causes them to become inoperable, or substantial and permanent loss of natural resources that can reasonably be expected to occur in the absence of a bypass. Severe property damage does not mean economic loss caused by delays in production. (40 CFR 122.41(m)(1)(ii).)

2. Bypass not exceeding limitations. The Discharger may allow any bypass to occur which does not cause exceedances of effluent limitations, but only if it is for essential maintenance to assure efficient operation. These bypasses are not subject to the provisions listed in Standard Provisions – Permit Compliance I.G.3, I.G.4, and I.G.5 below. (40 CFR 122.41(m)(2).)

3. Prohibition of bypass. Bypass is prohibited, and the Regional Water Board may take enforcement action against a Discharger for bypass, unless (40 CFR 122.41(m)(4)(i)):

   a. Bypass was unavoidable to prevent loss of life, personal injury, or severe property damage (40 CFR 122.41(m)(4)(i)(A));

   b. There were no feasible alternatives to the bypass, such as the use of auxiliary treatment facilities, retention of untreated wastes, or maintenance during normal periods of equipment downtime. This condition is not satisfied if adequate back-up equipment should have been installed in the exercise of reasonable engineering judgment to prevent a bypass that occurred during normal periods of equipment downtime or preventive maintenance (40 CFR 122.41(m)(4)(i)(B)); and

c. The Discharger submitted notice to the Regional Water Board as required under Standard Provisions – Permit Compliance I.G.5 below. (40 CFR 122.41(m)(4)(i)(C).)

4. The Regional Water Board may approve an anticipated bypass, after considering its adverse effects, if the Regional Water Board determines that it will meet the three conditions listed in Standard Provisions – Permit Compliance I.G.3 above. (40 CFR 122.41(m)(4)(ii).)

5. Notice

   a. Anticipated bypass. If the Discharger knows in advance of the need for a bypass, it shall submit a notice, if possible at least 10 days before the date of the bypass. (40 CFR 122.41(m)(3)(i).)

   b. Unanticipated bypass. The Discharger shall submit notice of an unanticipated bypass as required in Standard Provisions - Reporting V.E below (24-hour notice). (40 CFR 122.41(m)(3)(ii).)

H. Upset

Upset means an exceptional incident in which there is unintentional and temporary noncompliance with technology-based permit effluent limitations because of factors beyond the reasonable control of the Discharger. An upset does not include noncompliance to the extent caused by operational error, improperly designed treatment facilities, inadequate treatment facilities, lack of preventive maintenance, or careless or improper operation. (40 CFR 122.41(n)(1).)

1. Effect of an upset. An upset constitutes an affirmative defense to an action brought for noncompliance with such technology-based permit effluent limitations if the requirements of Standard Provisions – Permit Compliance I.H.2 below are met. No determination made during administrative review of claims that noncompliance was caused by upset, and before an action for noncompliance, is final administrative action subject to judicial review. (40 CFR 122.41(n)(2).).

2. Conditions necessary for a demonstration of upset. A Discharger who wishes to establish the affirmative defense of upset shall demonstrate, through properly signed, contemporaneous operating logs or other relevant evidence that (40 CFR 122.41(n)(3)):

   a. An upset occurred and that the Discharger can identify the cause(s) of the upset (40 CFR 122.41(n)(3)(i));

   b. The permitted facility was, at the time, being properly operated (40 CFR 122.41(n)(3)(ii));

   c. The Discharger submitted notice of the upset as required in Standard Provisions – Reporting V.E.2.b below (24-hour notice) (40 CFR 122.41(n)(3)(iii)); and

   d. The Discharger complied with any remedial measures required under Standard Provisions – Permit Compliance I.C above. (40 CFR 122.41(n)(3)(iv).)

3. Burden of proof. In any enforcement proceeding, the Discharger seeking to establish the occurrence of an upset has the burden of proof. (40 CFR 122.41(n)(4).)

II. STANDARD PROVISIONS – PERMIT ACTION

A. General

This Order may be modified, revoked and reissued, or terminated for cause. The filing of a request by the Discharger for modification, revocation and reissuance, or termination, or a notification of planned changes or anticipated noncompliance does not stay any Order condition. (40 CFR 122.41(f).)

B. Duty to Reapply

If the Discharger wishes to continue an activity regulated by this Order after the expiration date of this Order, the Discharger must apply for and obtain a new permit. (40 CFR 122.41(b).)

C. Transfers

This Order is not transferable to any person except after notice to the Regional Water Board. The Regional Water Board may require modification or revocation and reissuance of this Order to change the name of the Discharger and incorporate such other requirements as may be necessary under the CWA and the Water Code. (40 CFR 122.41(l)(3); 122.61.)

III. STANDARD PROVISIONS – MONITORING

A. Samples and measurements taken for the purpose of monitoring shall be representative of the monitored activity. (40 CFR 122.41(j)(1).)

B. Monitoring results must be conducted according to test procedures under Part 136 or, in the case of sludge use or disposal, approved under Part 136 unless otherwise specified in Part 503 unless other test procedures have been specified in this Order. (40 CFR 122.41(j)(4); 122.44(i)(1)(iv).)

IV. STANDARD PROVISIONS – RECORDS

A. Except for records of monitoring information required by this Order related to the Discharger's sewage sludge use and disposal activities, which shall be retained for a period of at least five years (or longer as required by Part 503), the Discharger shall retain records of all monitoring information, including all calibration and maintenance records and all original strip chart recordings for continuous monitoring instrumentation, copies of all reports required by this Order, and records of all data used to complete the application for this Order, for a period of at least three (3) years from the date of the sample, measurement, report or application. This period may be extended by request of the Regional Water Board Executive Officer at any time. (40 CFR 122.41(j)(2).)

B. Records of monitoring information shall include:

1. The date, exact place, and time of sampling or measurements (40 CFR 122.41(j)(3)(i));

2. The individual(s) who performed the sampling or measurements (40 CFR 122.41(j)(3)(ii));

3. The date(s) analyses were performed (40 CFR 122.41(j)(3)(iii));

4. The individual(s) who performed the analyses (40 CFR 122.41(j)(3)(iv));

5. The analytical techniques or methods used (40 CFR 122.41(j)(3)(v)); and

6. The results of such analyses. (40 CFR 122.41(j)(3)(vi).)

C. Claims of confidentiality for the following information will be denied (40 CFR 122.7(b)):

1. The name and address of any permit applicant or Discharger (40 CFR 122.7(b)(1)); and

2. Permit applications and attachments, permits and effluent data. (40 CFR 122.7(b)(2).)

V. STANDARD PROVISIONS – REPORTING

A. Duty to Provide Information

The Discharger shall furnish to the Regional Water Board, State Water Board, or USEPA within a reasonable time, any information which the Regional Water Board, State Water Board, or USEPA may request to determine whether cause exists for modifying, revoking and reissuing, or terminating this Order or to determine compliance with this Order. Upon request, the Discharger shall also furnish to the Regional Water Board, State Water Board, or USEPA copies of records required to be kept by this Order. (40 CFR 122.41(h); Wat. Code, § 13267)

B. Signatory and Certification Requirements

1. All applications, reports, or information submitted to the Regional Water Board, State Water Board, and/or USEPA shall be signed and certified in accordance with Standard Provisions – Reporting V.B.2, V.B.3, V.B.4, and V.B.5 below. (40 CFR 122.41(k))

2. All permit applications shall be signed by either a principal executive officer or ranking elected official. For purposes of this provision, a principal executive officer of a federal agency includes: (i) the chief executive officer of the agency, or (ii) a senior executive officer having responsibility for the overall operations of a principal geographic unit of the agency (e.g., Regional Administrators of USEPA). (40 CFR 122.22(a)(3)).

3. All reports required by this Order and other information requested by the Regional Water Board, State Water Board, or USEPA shall be signed by a person described in Standard Provisions – Reporting V.B.2 above, or by a duly authorized representative of that person. A person is a duly authorized representative only if:

    a. The authorization is made in writing by a person described in Standard Provisions – Reporting V.B.2 above (40 CFR 122.22(b)(1));

    b. The authorization specifies either an individual or a position having responsibility for the overall operation of the regulated facility or activity such as the position of plant manager, operator of a well or a well field, superintendent, position of equivalent responsibility, or an individual or position having overall responsibility for environmental matters for the company. (A duly authorized representative may thus be either a named individual or any individual occupying a named position.) (40 CFR 122.22(b)(2)); and

c. The written authorization is submitted to the Regional Water Board and State Water Board. (40 CFR 122.22(b)(3))

4. If an authorization under Standard Provisions – Reporting V.B.3 above is no longer accurate because a different individual or position has responsibility for the overall operation of the facility, a new authorization satisfying the requirements of Standard Provisions – Reporting V.B.3 above must be submitted to the Regional Water Board and State Water Board prior to or together with any reports, information, or applications, to be signed by an authorized representative. (40 CFR 122.22(c))

5. Any person signing a document under Standard Provisions – Reporting V.B.2 or V.B.3 above shall make the following certification:

   "I certify under penalty of law that this document and all attachments were prepared under my direction or supervision in accordance with a system designed to assure that qualified personnel properly gather and evaluate the information submitted. Based on my inquiry of the person or persons who manage the system or those persons directly responsible for gathering the information, the information submitted is, to the best of my knowledge and belief, true, accurate, and complete. I am aware that there are significant penalties for submitting false information, including the possibility of fine and imprisonment for knowing violations." (40 CFR 122.22(d))

C. Monitoring Reports

1. Monitoring results shall be reported at the intervals specified in the Monitoring and Reporting Program (Attachment E) in this Order. (40 CFR 122.22(l)(4))

2. Monitoring results must be reported on a Discharge Monitoring Report (DMR) form or forms provided or specified by the Regional Water Board or State Water Board for reporting results of monitoring of sludge use or disposal practices. (40 CFR 122.41(l)(4)(i))

3. If the Discharger monitors any pollutant more frequently than required by this Order using test procedures approved under Part 136 or, in the case of sludge use or disposal, approved under Part 136 unless otherwise specified in Part 503, or as specified in this Order, the results of this monitoring shall be included in the calculation and reporting of the data submitted in the DMR or sludge reporting form specified by the Regional Water Board. (40 CFR 122.41(l)(4)(ii))

4. Calculations for all limitations, which require averaging of measurements, shall utilize an arithmetic mean unless otherwise specified in this Order. (40 CFR 122.41(l)(4)(iii))

D. Compliance Schedules

Reports of compliance or noncompliance with, or any progress reports on, interim and final requirements contained in any compliance schedule of this Order, shall be submitted no later than 14 days following each schedule date. (40 CFR 122.41(l)(5))

E. Twenty-Four Hour Reporting

1. The Discharger shall report any noncompliance that may endanger health or the environment. Any information shall be provided orally within 24 hours from the time the Discharger becomes aware of the circumstances. A written submission shall also be provided within five (5) days of the time the Discharger becomes aware of the circumstances. The written submission shall contain a description of the noncompliance and its cause; the period of noncompliance, including exact dates and times, and if the noncompliance has not been corrected, the anticipated time it is expected to continue; and steps taken or planned to reduce, eliminate, and prevent reoccurrence of the noncompliance. (40 CFR 122.41(l)(6)(i).)

2. The following shall be included as information that must be reported within 24 hours under this paragraph (40 CFR 122.41(l)(6)(ii)):

   a. Any unanticipated bypass that exceeds any effluent limitation in this Order. (40 CFR 122.41(l)(6)(ii)(A).)

   b. Any upset that exceeds any effluent limitation in this Order. (40 CFR 122.41(l)(6)(ii)(B).)

3. The Regional Water Board may waive the above-required written report under this provision on a case-by-case basis if an oral report has been received within 24 hours. (40 CFR 122.41(l)(6)(iii).)

F. Planned Changes

The Discharger shall give notice to the Regional Water Board as soon as possible of any planned physical alterations or additions to the permitted facility. Notice is required under this provision only when (40 CFR 122.41(l)(1)):

1. The alteration or addition to a permitted facility may meet one of the criteria for determining whether a facility is a new source in section 122.29(b) (40 CFR 122.41(l)(1)(i)); or

2. The alteration or addition could significantly change the nature or increase the quantity of pollutants discharged. This notification applies to pollutants that are not subject to effluent limitations in this Order. (40 CFR 122.41(l)(1)(ii).)

3. The alteration or addition results in a significant change in the Discharger's sludge use or disposal practices, and such alteration, addition, or change may justify the application of permit conditions that are different from or absent in the existing permit, including notification of additional use or disposal sites not reported during the permit application process or not reported pursuant to an approved land application plan. (40 CFR 122.41(l)(1)(iii).)

G. Anticipated Noncompliance

The Discharger shall give advance notice to the Regional Water Board or State Water Board of any planned changes in the permitted facility or activity that may result in noncompliance with General Order requirements. (40 CFR 122.41(l)(2).)

H. Other Noncompliance

The Discharger shall report all instances of noncompliance not reported under Standard Provisions – Reporting V.C, V.D, and V.E above at the time monitoring reports are submitted. The reports shall contain the information listed in Standard Provision – Reporting V.E above. (40 CFR 122.41(l)(7).)

I. Other Information

When the Discharger becomes aware that it failed to submit any relevant facts in a permit application, or submitted incorrect information in a permit application or in any report to the Regional Water Board, State Water Board, or USEPA, the Discharger shall promptly submit such facts or information. (40 CFR 122.41(l)(8).)

VI. STANDARD PROVISIONS – ENFORCEMENT

A. The Regional Water Board is authorized to enforce the terms of this Order under several provisions of the Water Code, including, but not limited to, sections 13385, 13386, and 13387.

VII. ADDITIONAL PROVISIONS – NOTIFICATION LEVELS

A. Publicly-Owned Treatment Works (POTWs)

All POTWs shall provide adequate notice to the Regional Water Board of the following (40 CFR 122.42(b)):

1. Any new introduction of pollutants into the POTW from an indirect discharger that would be subject to sections 301 or 306 of the CWA if it were directly discharging those pollutants (40 CFR 122.42(b)(1)); and

2. Any substantial change in the volume or character of pollutants being introduced into that POTW by a source introducing pollutants into the POTW at the time of adoption of this Order. (40 CFR 122.42(b)(2).)

3. Adequate notice shall include information on the quality and quantity of effluent introduced into the POTW as well as any anticipated impact of the change on the quantity or quality of effluent to be discharged from the POTW. (40 CFR 122.42(b)(3).)

ATTACHMENT E – MONITORING AND REPORTING PROGRAM

Contents

Tables

ATTACHMENT E – MONITORING AND REPORTING PROGRAM (MRP)

National Pollutant Discharge Elimination System (NPDES) regulations at 40 CFR 122.48 require that all NPDES permits specify monitoring and reporting requirements. California Water Code (CWC) sections 13267 and 13383 also authorize the Regional Water Quality Control Board (hereinafter the Regional Water Board) to require technical and monitoring reports. This MRP establishes monitoring and reporting requirements that implement the federal and State regulations.

I. GENERAL MONITORING PROVISIONS

A. The Discharger shall comply with this MRP. The Executive Officer may amend this MRP pursuant to 40 CFR 122.62, 122.63, and 124.5. If any discrepancies exist between this MRP and the Regional Standard Provisions (Attachment G), this MRP prevails.

B. The Discharger shall conduct all monitoring in accordance with Attachment D, section III, as supplemented by Attachment G of this Order. Equivalent test methods must be more sensitive than those specified in 40 CFR 136 and must be specified in the permit.

II. MONITORING LOCATIONS

The Discharger shall establish the following monitoring locations to demonstrate compliance with the effluent limitations, discharge specifications, and other requirements in this Order.

Table E-1. Monitoring Station Locations

Type of Sampling Location	Monitoring Location Name	Monitoring Location Description
Influent	INF-001	At any point in the treatment facility headworks at which all waste tributary to that plant is present and preceding any phase of treatment. Recycle streams from internal treatment plant processes may be included in the flow for this sampling station.
Effluent	EFF-001	At any point in the treatment facility between the point of discharge and the point at which all flow tributary to the outfall is present.
Biosolids	BIO-001	Sludge monitoring in the treatment facility.

III. INFLUENT MONITORING REQUIREMENTS

The Discharger shall monitor influent to the Plant at Monitoring Location INF-001 as follows:

Table E-2. Influent Monitoring

Parameter	Units	Sample Type	Minimum Sampling Frequency
Flow[1]	MGD	Continuous	Continuous/D
$CBOD_5$	mg/L	C-24	2/Week
TSS	mg/L	C-24	2/Week

Legend for Table E-2
Unit Abbreviations:

MG	= million gallons
MGD	= million gallons per day
mg/L	= milligrams per liter

Sample Type:
C-24 = 24-hour composite
Sampling Frequency:
Continuous/D = measured continuously, and recorded and reported daily
2/Week = Two times per week

Footnote for Table E-2

[1] Flow Monitoring. Flow shall be monitored continuously, and the following information shall be reported in self-monitoring reports for each month:
- Daily average flow (MGD)
- Total daily flow volume (MG)
- Monthly average flow (MGD)
- Total monthly flow volume (MG)
- Maximum and minimum daily average flow rates (MGD) and time of occurrence

IV. EFFLUENT MONITORING REQUIREMENTS

The Discharger shall monitor discharges of treated wastewater from the Plant at EFF-001 as follows.

Table E-3. Effluent Monitoring

Parameter	Units	Sample Type	Minimum Sampling Frequency
Flow[1]	MGD	Continuous	Continuous/D
$CBOD_5$	mg/L	C-24	2/Week
TSS	mg/L	C-24	2/Week
CBOD and TSS % Removal[2]	%	Calculate	1/Month
Oil and Grease[3]	mg/L	Grab	2/Year
pH [4]	s.u.	Grab	1/Day or Continuous/D
Dissolved Oxygen	mg/L	Grab	1/Day
Dissolved Sulfides[5]	mg/L	Grab	1/Day
Temperature	°C	Grab	1/Day
Enterococcus Bacteria	Colonies/100 mL	Grab	2/Week
Acute Toxicity[6]	% Survival	Flow through	1/Month
Chronic Toxicity[7]	TUc	C-24	Quarterly
Ammonia	mg/L as N	C-24	1/Month
Copper	µg/L	C-24	1/Month
Cyanide	µg/L	Grab	1/Month
Dioxin-TEQ	µg/L	Grab	2/Year
Acrylonitrile	µg/L	Grab	2/Year
Bis(2-ethylhexyl)phthalate	µg/L	Grab	2/Year

Legend to Table E-3:

Unit Abbreviations:
MG = million gallons
MGD = million gallons per day
s.u. = standard units
mg/L = milligrams per liter
mg/L as N = milligrams per liter as nitrogen
MPN/100 mL = most probable number per 100 milliliters
°F = degrees Fahrenheit
TUc = chronic toxicity units
µg/L = micrograms per liter
Sample Type:
C-24 = 24-hour composite

Sampling Frequency:

Continuous/D	= measured continuously, and recorded and reported daily
1/Day	= Once per day
2/Week	= Two times per week
3/Week	= Three times per week
5/Week	= Five times per week
1/Month	= Once per month
1/2 Months	= Once every two months
1/Year	= Once per year
2/Year	= Twice per year

Footnotes to Table E-3:

[1] Flow Monitoring. Flow shall be monitored continuously, and the following information shall be reported in self-monitoring reports for each month:
- Daily average flow (MGD)
- Total daily flow volume (MG)
- Monthly average flow (MGD)
- Total monthly flow volume (MG)
- Maximum and minimum daily average flow rates (MGD) and time of occurrence

[2] CBOD and TSS % Removal. The percent removal for CBOD and TSS shall be reported for each calendar month in accordance with Effluent Limitation IV.A.1. Samples for CBOD and TSS shall be collected simultaneously with influent samples.

[3] Oil and Grease. Each oil and grease sampling and analysis event shall be conducted in accordance with Standard Methods 21st Ed.

[4] pH. If pH is monitored continuously, the minimum and maximum pH values for each day shall be reported in monthly Self-Monitoring Reports (SMRs).

[5] Dissolved Sulfides. Measured when dissolved oxygen concentration is less than 2.0 mg/L.

[6] Acute toxicity. Acute bioassay tests shall be performed in accordance with section V.A of this MRP.

[7] Chronic toxicity. Critical life stage toxicity tests shall be performed and reported in accordance with the Chronic Toxicity Requirements of specified in section V.B of this MRP.

V. WHOLE EFFLUENT TOXICITY TESTING REQUIREMENTS

The Discharger shall monitor whole effluent acute and chronic toxicity at EFF-001 as follows:

A. Whole Effluent Acute Toxicity

1. Compliance with the acute toxicity effluent limitations of this Order shall be evaluated by measuring survival of test organisms exposed to 96-hour continuous flow-through bioassays at Monitoring Location EFF-001.

2. Test organisms shall be fathead minnow (*Pimephales promelas*) or rainbow trout (*Oncorhynchus mykiss*) unless the Executive Officer specifies otherwise in writing.

3. All bioassays shall be performed according to the most up-to-date protocols in 40 CFR 136, currently *Methods for Measuring the Acute Toxicity of Effluents and Receiving Water to Freshwater and Marine Organisms,* 5th Edition.

4. If specific identifiable substances in the discharge can be demonstrated by the Discharger as being rapidly rendered harmless upon discharge to the receiving water, compliance with the acute toxicity limit may be determined after the test samples are adjusted to remove the influence of those substances. Written approval from the Executive Officer must be obtained to authorize such an adjustment.

5. The sample may be taken from final secondary effluent prior to disinfection. Monitoring of the bioassay water shall include, on a daily basis, the following parameters: pH, dissolved oxygen, ammonia (if toxicity is observed), temperature, hardness, and alkalinity. These

results shall be reported. If a violation of acute toxicity requirements occurs, the bioassay test shall be repeated with new fish as soon as practical and shall be repeated until a test fish survival rate of 90 percent or greater is observed. If the control fish survival rate is less than 90 percent, the bioassay test shall be restarted with new fish and shall continue as soon as practical until an acceptable test is completed (i.e., control fish survival rate is 90 percent or greater).

B. Whole Effluent Chronic Toxicity

1. Chronic Toxicity Monitoring Requirements

a. **Sampling.** The Discharger shall collect 24-hour composite samples of the effluent at monitoring location EFF-001, for critical life stage toxicity testing as indicated below. For toxicity tests requiring renewals, 24-hour composite samples collected on consecutive days are required.

b. **Test Species.** The test species shall be either *Selenastrum capricornutum* (green algae) or *Americamysis bahia* (mysid shrimp). The Discharger shall conduct a screening chronic toxicity test as described in Appendix E-1 following any significant change in the nature of the effluent or prior to application for permit renewal. The most sensitive species shall be used thereafter for routine chronic toxicity monitoring. The Executive Officer may authorize a change to another test species if the Discharger's chronic toxicity screening data suggest that another test species is more sensitive to the discharge.

c. **Frequency.** The frequency of routine and accelerated chronic toxicity monitoring shall be as specified below:

(1) Undertake routine monitoring *quarterly*.

(2) Accelerate monitoring to *monthly* after exceeding a three-sample median of 10 TU_c[1] or a single sample maximum of 20 TU_c. The Executive Officer may specify a different frequency for accelerated monitoring based on the TU_c results.

(3) Return to routine monitoring if accelerated monitoring does not exceed either trigger in (2), above.

(4) If accelerated monitoring confirms consistent toxicity in excess of either trigger in (2), above, continue accelerated monitoring and initiate toxicity reduction evaluation (TRE) procedures in accordance with section B.3, below.

(5) Return to routine monitoring after implementing appropriate elements of the TRE, and either the toxicity drops below both triggers in (2), above, or, based on the TRE results, the Executive Officer authorizes a return to routine monitoring.

Monitoring conducted pursuant to a TRE effort shall satisfy the requirements for routine and accelerated monitoring while the TRE investigation is underway.

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1 A TU_c equals 100 divided by the no observable effect level (NOEL). The NOEL is determined from IC_{25}, EC_{25}, or NOEC values. These terms, their usage, and other chronic toxicity monitoring program requirements are defined in the MRP (Attachment E).

d. **Methodology.** Sample collection, handling, and preservation shall be in accordance with USEPA protocols. In addition, bioassays shall be conducted in compliance with the most recently promulgated test methods, as shown in Appendix E-1. These are *Short-Term Methods for Estimating the Chronic Toxicity of Effluents and Receiving Waters to Marine and Estuarine Organisms,* currently fourth Edition (EPA-821-R-02-013), with exceptions granted the Discharger in writing by the Executive Officer and the Environmental Laboratory Accreditation Program (ELAP). If specific identifiable substances in the discharge can be demonstrated by the Discharger as being rapidly rendered harmless upon discharge to the receiving water, compliance with the chronic toxicity limit may be determined after the test samples are adjusted to remove the influence of those substances. Written approval from the Executive Officer must be obtained to authorize such an adjustment.

e. **Dilution Series.** The Discharger shall conduct tests with a control and five effluent concentrations (including 100% effluent) using a dilution factor ≥ 0.5. Test sample pH in each dilution in the series may be controlled to the level of the effluent sample as received prior to being salted up.

2. **Chronic Toxicity Reporting Requirements**

a. **Routine Reporting.** Toxicity test results for the current reporting period shall include, at a minimum, for each test:

(1) Sample date

(2) Test initiation date

(3) Test species

(4) End point values for each dilution (e.g., number of young, growth rate, percent survival)

(5) No Observable Effect Level (NOEL) values in percent effluent. The NOEL shall equal to the IC_{25} or EC_{25} (see Appendix E-1). If the IC_{25} or EC_{25} cannot be statistically determined, the NOEL shall equal to the No Observable Effect Concentration (NOEC) derived using hypothesis testing. The NOEC is the maximum percent effluent concentration that causes no observable effect on test organisms based on a critical life stage toxicity test.

(6) IC_{15}, IC_{25}, IC_{40}, and IC_{50} values (or EC_{15}, EC_{25} ... etc.) as percent effluent

(7) TUc values (TU_c = 100/NOEL).

(8) Mean percent mortality (±s.d.) after 96 hours in 100% effluent (if applicable)

(9) NOEC and LOEC values for reference toxicant tests

(10) IC_{50} or EC_{50} values for reference toxicant tests

(11) Available water quality measurements for each test (pH, dissolved oxygen, temperature, conductivity, hardness, salinity, ammonia)

b. **Compliance Summary.** The results of the chronic toxicity testing shall be provided in the self-monitoring report as TUc's.

3. **Chronic Toxicity Reduction Evaluation (TRE)**

a. The Discharger shall prepare a generic TRE work plan within 90 days of the effective date of this Order to be ready to respond to toxicity events. The Discharger shall review and update the work plan as necessary so that it remains current and applicable to the discharge and discharge facilities.

b. Within 30 days of exceeding either chronic toxicity trigger, the Discharger shall submit to the Regional Water Board a TRE work plan, which shall be the generic work plan revised as appropriate for this toxicity event after consideration of available discharge data.

c. Within 30 days of the date of completion of the accelerated monitoring tests observed to exceed either trigger, the Discharger shall initiate a TRE in accordance with a TRE work plan that incorporates any and all comments from the Executive Officer.

d. The TRE shall be specific to the discharge and be in accordance with current technical guidance and reference materials, including USEPA guidance materials. The TRE shall be conducted as a tiered evaluation process, such as summarized below:

(1) Tier 1 consists of basic data collection (routine and accelerated monitoring).

(2) Tier 2 consists of evaluation of optimization of the treatment process, including operation practices and in-plant process chemicals.

(3) Tier 3 consists of a toxicity identification evaluation (TIE).

(4) Tier 4 consists of evaluation of options for additional effluent treatment processes.

(5) Tier 5 consists of evaluation of options for modifications of in-plant treatment processes.

(6) Tier 6 consists of implementation of selected toxicity control measures, and follow-up monitoring and confirmation of implementation success.

e. The TRE may be ended at any stage if monitoring finds there is no longer consistent toxicity (complying with requirements of Provision IV.C.2 of the Order).

f. The objective of the TIE shall be to identify the substance or combination of substances causing the observed toxicity. All reasonable efforts using currently available TIE methodologies shall be employed.

g. As toxic substances are identified or characterized, the Discharger shall continue the TRE by determining the sources and evaluating alternative strategies for reducing or eliminating the substances from the discharge. All reasonable steps shall be taken to reduce toxicity to levels consistent with chronic toxicity evaluation parameters.

h. Many recommended TRE elements parallel required or recommended efforts of source control, pollution prevention, and storm water control programs. TRE efforts should be coordinated with such efforts. To prevent duplication of efforts, evidence of complying with requirements or recommended efforts of such programs may be acceptable to comply with TRE requirements.

i. The Regional Water Board recognizes that chronic toxicity may be episodic and identification of causes of and reduction of sources of chronic toxicity may not be successful in all cases. Consideration of enforcement action by the Regional Water Board will be based in part on the Discharger's actions and efforts to identify and control or reduce sources of consistent toxicity.

VI. RECEIVING WATER MONITORING REQUIREMENTS

The Discharger shall continue to participate in the RMP, which involves collection of data on pollutants and toxicity in San Francisco Bay water, sediment, and biota. The Discharger's participation and support of the RMP is the basis for not including receiving water monitoring requirements in this permit.

VII. PRETREATMENT AND BIOSOLIDS MONITORING REQUIREMENTS

The Discharger shall comply with the pretreatment requirements specified below for influent (at Monitoring Location INF-001), effluent (at Monitoring Location EFF-001), and biosolids monitoring (at Monitoring Location BIO-001). The Discharger shall report summaries of analytical results in annual and semi-annual pretreatment reports in accordance with Attachment H.

Table E-4. Pretreatment and Biosolids Monitoring Requirements

Constituents	Sampling Frequency			Sample Type[4]	
	Influent INF-001	Effluent EFF-001[3]	Biosolids BIO-001	INF-001 and EFF-001	Biosolids BIO-001
VOC	2/Year	2/Year	2/Year	Grab	Grab [4c]
BNA	2/Year	2/Year	2/Year	Grab	Grab [4c]
Organophosphorus Pesticides	2/Year	2/Year	2/Year	24-hr Composite[4a]	Grab [4c]
Metals[1]	1/Month	1/Month	2/Year	24-hr Composite[4a]	Grab [4c]
Hexavalent Chromium[2]	1/Month	1/Month	2/Year	Grab	Grab [4c]
Mercury	1/Month	1/Month	2/Year	24-hr Composite[4a,4b]	Grab [4c]
Cyanide	1/Month	1/Month	2/Year	Grab	Grab [4c]

Legend for Table E-4:

Constituents:
VOC volatile organic compounds
BNA base/neutrals and acids extractable organic compounds

Sampling Frequency:
1/month once per month
2/year twice per year

Footnotes for Table E-4:

[1] The metals are arsenic, cadmium, copper, lead, nickel, silver, zinc, and selenium.
[2] The Discharger may elect to report total chromium instead of hexavalent chromium. Sample collection for total chromium measurements shall be 24-hour composite sampling.
[3] Effluent monitoring conducted in accordance with Table E-3 can be used to satisfy these pretreatment monitoring requirements.

[4] Sample types:

a. 24-hour composite samples may be made up discrete grab samples and may be combined (volumetrically flow-weighted) prior to analysis, or they may be mathematically flow-weighted. If an automatic compositor is used, 24-hour composite samples must be obtained through flow-proportioned composite sampling.

b. Automatic compositors are allowed for mercury if either (1) the compositing equipment (hoses and containers) comply with ultraclean specifications, or (2) appropriate equipment blank samples demonstrate that the compositing equipment has not contaminated the sample.

c. The biosolids sample shall be a composite of the biosolids to be disposed. Biosolids collection and monitoring shall comply with the requirements specified in Attachment H, Appendix H-4. The Discharger shall also comply with the biosolids monitoring requirements of 40 CFR 503.

VIII. REPORTING REQUIREMENTS

A. General Monitoring and Reporting Requirements

The Discharger shall comply with all Federal Standard Provisions (Attachment D) and Regional Standard Provisions (Attachment G) related to monitoring, reporting, and recordkeeping, with modifications shown in VIII.D below.

B. Self Monitoring Reports (SMRs)

1. **SMR Format.** The Discharger shall electronically submit SMRs using the State Water Board's California Integrated Water Quality System (CIWQS) Program Web site (http://www.waterboards.ca.gov/ciwqs/index.html). The CIWQS website will provide additional directions for SMR submittals in the event of a service interruption for electronic submittal.

2. **SMR Due Dates and Contents.** The Discharger shall submit SMRs by the due dates, and with the contents, specified below:

   a. **Monthly SMRs** — Monthly SMRs shall be due 30 days after the end of each calendar month, covering that calendar month. The monthly SMR shall contain the applicable items described in sections V.B and V.C of both Attachments D and G of this Order. See Provision VI.C.6.b (Effluent Characterization Study and Report) of this Order for information that must also be reported with the monthly SMR.

   b. **Annual SMR** — Annual SMRs shall be due February 1 each year, covering the previous calendar year. The annual SMR shall contain the items described in sections V.C.1.f .(2), V.C.1.f .(6) as applicable, and V.C.1.f .(7) of the Regional Standard Provisions (Attachment G). Information described in the other subsections of V.C.1.f of Attachment G is not required. See also Provision VI.C.2.b(2) (Effluent Characterization Study and Report) for requirements to submit reports with the annual SMR.

   c. **Additional Specifications for Submitting SMRs to CIWQS** — The Discharger shall submit analytical results and other information using one of the following methods:

Table E-5. SMR Reporting for CIWQS

Parameter	Method of Reporting	
	EDF/CDF data upload or manual entry	Attached File
All parameters identified in influent, effluent, and receiving water monitoring tables (except	Required for All Results	

Dissolved Oxygen and Temperature)		
Dissolved Oxygen Temperature	Required for Monthly Maximum and Minimum Results Only [1]	Discharger may use this method for all results or keep records
Cyanide Arsenic Cadmium Chromium Copper Lead Mercury Nickel Selenium Silver Zinc Dioxins and Furans (by U.S. EPA Method 1613)	Required for All Results [2]	
Antimony Berylium Thallium Pollutants by U.S. EPA Methods 601, 602, 608, 610, 614, 624, and 625	Not Required (unless identified in influent, effluent, or receiving water monitoring tables), But Encouraged [1]	Discharger may use this method and submit results with application for permit reissuance, unless data submitted by CDF/EDF upload
Analytical Method	Not Required (Discharger may select "data unavailable") [1]	
Collection Time Analysis Time	Not Required (Discharger may select "0:00") [1]	

Footnotes for Table E-5:

[1] The Discharger shall continue to monitor at the minimum frequency specified in the monitoring tables, keep records of the measurements, and make the records available upon request.

[2] These parameters require EDF/CDF data upload or manual entry regardless of whether monitoring is required by this MRP or other provisions of this Order (except for biosolids, sludge, or ash provisions).

3. **Monitoring Periods.** Monitoring periods for all required monitoring shall be completed as set forth in the table below:

Table E-6. Monitoring Periods and Reporting Schedule

Sampling Frequency	Monitoring Period Begins On…	Monitoring Period
Continuous	Permit effective date	All
1/Day	Permit effective date	(Midnight through 11:59 PM) or any 24-hour period that reasonably represents a calendar day for purposes of sampling.
2/Week 4/Week 5/Week	Permit effective date	Sunday through Saturday
1/Month	Permit effective date	First day of calendar month through last day of calendar month
1/2 Months	Permit effective date	First day of calendar month through last day of next calendar month
1/Year	Permit effective date	January 1through December 31
2/Year	Permit effective date	Once during the wet season (typically November 1 – April 30) and once during the dry season (typically May 1 through October 31)

4. **ML and MDL Reporting.** The Discharger shall report with each sample result the Reporting Level (RL) and Method Detection Limit (MDL) as determined by the procedure in 40 CFR 136. The Discharger shall report the results of analytical determinations for the presence of chemical constituents in a sample using the following reporting protocols:

    a. Sample results greater than or equal to the RL shall be reported as measured by the laboratory (i.e., the measured chemical concentration in the sample).

    b. Sample results less than the RL, but greater than or equal to the laboratory's MDL, shall be reported as "Detected, but Not Quantified," or DNQ. The estimated chemical concentration of the sample shall also be reported. For purposes of data collection, the laboratory shall write the estimated chemical concentration next to DNQ as well as the words "Estimated Concentration" (may be shortened to "Est. Conc."). The laboratory may, if such information is available, include numerical estimates of the data quality for the reported result. Numerical estimates of data quality may be percent accuracy (+/- a percentage of the reported value), numerical ranges (low to high), or any other means the laboratory considers appropriate.

    c. Sample results less than the laboratory's MDL shall be reported as "Not Detected" or ND.

    d. The Discharger shall instruct laboratories to establish calibration standards so that the minimum level (ML) value (or its equivalent if there is differential treatment of samples relative to calibration standards) is the lowest calibration standard. At no time is the Discharger to use analytical data derived from extrapolation beyond the lowest point of the calibration curve.

C. Discharge Monitoring Reports

1. As described in section VIII.B.1 above, at any time during the term of this permit, the State or Regional Water Board may notify the Discharger to electronically submit SMRs that will satisfy federal requirements for submittal of Discharge Monitoring Reports (DMRs.) Until such notification is given, the Discharger shall submit DMRs in accordance with the requirements described below.

2. Once notified by the State or Regional Water Board, the Discharger shall submit hard copy DMRs. DMRs must be signed and certified as required by the Standard Provisions (Attachment D). The Discharger shall submit the original DMR and one copy of the DMR to one of the addresses listed below:

Standard Mail	FedEx/UPS/Other Private Carriers
State Water Resources Control Board Division of Water Quality c/o DMR Processing Center PO Box 100 Sacramento, CA 95812-1000	State Water Resources Control Board Division of Water Quality c/o DMR Processing Center 1001 I Street, 15th Floor Sacramento, CA 95814

3. All discharge monitoring results must be reported on the official USEPA pre-printed DMR forms (EPA Form 3320-1). Forms that are self-generated will not be accepted unless they follow the exact same format of EPA Form 3320-1.

D. Modifications to Attachment G

1. Attachment G sections V.C.1.f and V.C.1.g are revised as follows, and section V.C.1.h (Reporting data in electronic format) is deleted.

f. Annual self-monitoring report requirements

By the date specified in the MRP, the Discharger shall submit an annual report to the Regional Water Board covering the previous calendar year. The report shall contain the following:

1) Annual compliance summary table of treatment plant performance, including documentation of any blending events (this summary table is not required if the Discharger has submitted the year's monitoring results to CIWQS in electronic reporting format by EDF/CDF upload or manual entry);

2) Comprehensive discussion of treatment plant performance and compliance with the permit (This discussion shall include any corrective actions taken or planned, such as changes to facility equipment or operation practices that may be needed to achieve compliance, and any other actions taken or planned that are intended to improve performance and reliability of the Discharger's wastewater collection, treatment, or disposal practices.);

3) Both tabular and graphical summaries of the monitoring data for the previous year if parameters are monitored at a frequency of monthly or greater (this item is not required if the Discharger has submitted the year's monitoring results to CIWQS in electronic reporting format by EDF/CDF upload or manual entry);

4) List of approved analyses, including the following:

(i) List of analyses for which the Discharger is certified;

(ii) List of analyses performed for the Discharger by a separate certified laboratory (copies of reports signed by the laboratory director of that laboratory shall not be submitted but be retained onsite); and

(iii) List of "waived" analyses, as approved;

5) Plan view drawing or map showing the Discharger's facility, flow routing, and sampling and observation station locations;

6) Results of annual facility inspection to verify that all elements of the SWPP Plan are accurate and up to date (only required if the Discharger does not route all storm water to the headworks of its wastewater treatment plant); and

7) Results of facility report reviews (The Discharger shall regularly review, revise, and update, as necessary, the O&M Manual, the Contingency Plan, the Spill Prevention Plan, and Wastewater Facilities Status Report so that these documents remain useful and relevant to

current practices. At a minimum, reviews shall be conducted annually. The Discharger shall include, in each Annual Report, a description or summary of review and evaluation procedures, recommended or planned actions, and an estimated time schedule for implementing these actions. The Discharger shall complete changes to these documents to ensure they are up-to-date.).

g. Report submittal

The Discharger shall submit SMRs addressed as follows, unless the Discharger submits SMRs electronically to CIWQS:

California Regional Water Quality Control Board
San Francisco Bay Region
1515 Clay Street, Suite 1400
Oakland, CA 94612
Attn: NPDES Wastewater Division

2. Attachment G sections V.E.2, V.E.2.a, and V.E.2.c are revised as follows, and sections V.E.2.b (24-hour Certification) and V.E.2.d (Communication Protocol) are deleted.

2. Unauthorized Discharges from Municipal Wastewater Treatment Plants[2]

The following requirements apply to municipal wastewater treatment plants that experience an unauthorized discharge at their treatment facilities and supersede requirements imposed on the Discharger by the Executive Officer by letter of May 1, 2008.

a. Two (2)-Hour Notification

For any unauthorized discharges that enter a drainage channel or a surface water, the Discharger shall, as soon as possible, but not later than two (2) hours after becoming aware of the discharge, notify the California Emergency Management Agency (CalEMA currently 800-852-7550), the local health officers or directors of environmental health with jurisdiction over the affected water bodies, and the Regional Water Board. Timely notification by the Discharger to CalEMA also satisfies notification to the Regional Water Board. Notification shall include the following:

1) Incident description and cause;

2) Location of threatened or involved waterway(s) or storm drains;

3) Date and time the unauthorized discharge started;

4) Estimated quantity and duration of the unauthorized discharge (to the extent known), and the estimated amount recovered;

[2] California Code of Regulations, Title 23, Section 2250(b), defines an unauthorized discharge to be a discharge, not regulated by waste discharge requirements, of treated, partially treated, or untreated wastewater resulting from the intentional or unintentional diversion of wastewater from a collection, treatment or disposal system.

5) Level of treatment prior to discharge (e.g., raw wastewater, primary treated, undisinfected secondary treated, and so on); and

6) Identity of the person reporting the unauthorized discharge.

b. 24-hour Certification – Deleted

c. 5-day Written Report

Within five business days, the Discharger shall submit a written report that includes, in addition to the information required above, the following:

1) Methods used to delineate the geographical extent of the unauthorized discharge within receiving waters;

2) Efforts implemented to minimize public exposure to the unauthorized discharge;

3) Visual observations of the impacts (if any) noted in the receiving waters (e.g., fish kill, discoloration of water) and the extent of sampling if conducted;

4) Corrective measures taken to minimize the impact of the unauthorized discharge;

5) Measures to be taken to minimize the chances of a similar unauthorized discharge occurring in the future;

6) Summary of Spill Prevention Plan or O&M Manual modifications to be made, if necessary, to minimize the chances of future unauthorized discharges; and

7) Quantity and duration of the unauthorized discharge, and the amount recovered.

d. Communication Protocol – Deleted

APPENDIX E-1
CHRONIC TOXICITY
DEFINITION OF TERMS AND SCREENING PHASE REQUIREMENTS

I. Definition of Terms

A. No observed effect level (NOEL) for compliance determination is equal to IC_{25} or EC_{25}. If the IC_{25} or EC_{25} cannot be statistically determined, the NOEL shall be equal to the NOEC derived using hypothesis testing.

B. Effective concentration (EC) is a point estimate of the toxicant concentration that would cause an adverse effect on a quantal, "all or nothing," response (such as death, immobilization, or serious incapacitation) in a given percent of the test organisms. If the effect is death or immobility, the term lethal concentration (LC) may be used. EC values may be calculated using point estimation techniques such as probit, logit, and Spearman-Karber. EC_{25} is the concentration of toxicant (in percent effluent) that causes a response in 25 percent of the test organisms.

C. Inhibition concentration (IC) is a point estimate of the toxicant concentration that would cause a given percent reduction in a nonlethal, nonquantal biological measurement, such as growth. For example, an IC_{25} is the estimated concentration of toxicant that would cause a 25 percent reduction in average young per female or growth. IC values may be calculated using a linear interpolation method such as USEPA's Bootstrap Procedure.

D. No observed effect concentration (NOEC) is the highest tested concentration of an effluent or a toxicant at which no adverse effects are observed on the aquatic test organisms at a specific time of observation. It is determined using hypothesis testing.

II. Chronic Toxicity Screening Phase Requirements

A. The Discharger shall perform screening phase monitoring:

1. Subsequent to any significant change in the nature of the effluent discharged through changes in sources or treatment, except those changes resulting from reductions in pollutant concentrations attributable to source control efforts, or

2. Prior to permit reissuance. Screening phase monitoring data shall be included in the NPDES permit application for reissuance. The information shall be as recent as possible, but may be based on screening phase monitoring conducted within 5 years before the permit expiration date.

B. Design of the screening phase shall, at a minimum, consist of the following elements:

1. Use of test species specified in Appendix E-2, attached, and use of the protocols referenced in those tables, or as approved by the Executive Officer.

2. Two stages:
   a. Stage 1 shall consist of a minimum of one battery of tests conducted concurrently. Selection of the type of test species and minimum number of tests shall be based on Appendix E-2 (attached).

b. Stage 2 shall consist of a minimum of two test batteries conducted at a monthly frequency using the three most sensitive species based on the Stage 1 test results and as approved by the Executive Officer.

3. Appropriate controls.

4. Concurrent reference toxicant tests.

5. Dilution series of 100%, 50%, 25%, 12.5%, 6.25%, and 0 %, where "%" is percent effluent as discharged, or as otherwise approved the Executive Officer.

C. The Discharger shall submit a screening phase proposal acceptable to the Executive Officer. The proposal shall address each of the elements listed above. If within 30 days, the Executive Officer does not comment, the Discharger shall commence with screening phase monitoring.

APPENDIX E-2
SUMMARY OF TOXICITY TEST SPECIES REQUIREMENTS

Table AE-1. Critical Life Stage Toxicity Tests for Estuarine Waters

Species	(Scientific Name)	Effect	Test Duration	Reference
Alga	*(Skeletonema costatum)* *(Thalassiosira pseudonana)*	Growth rate	4 days	1
Red alga	*(Champia parvula)*	Number of cystocarps	7–9 days	3
Giant kelp	*(Macrocystis pyrifera)*	Percent germination; germ tube length	48 hours	2
Abalone	*(Haliotis rufescens)*	Abnormal shell development	48 hours	2
Oyster Mussel	*(Crassostrea gigas)* *(Mytilus edulis)*	Abnormal shell development; percent survival	48 hours	2
Echinoderms - Urchins Sand dollar	*(Strongylocentrotus purpuratus, S. franciscanus)* *(Dendraster excentricus)*	Percent fertilization	1 hour	2
Shrimp	*(Americamysis bahia)*	Percent survival; growth	7 days	3
Shrimp	*(Holmesimysis costata)*	Percent survival; growth	7 days	2
Topsmelt	*(Atherinops affinis)*	Percent survival; growth	7 days	2
Silversides	*(Menidia beryllina)*	Larval growth rate; percent survival	7 days	3

Toxicity Test References:

1. American Society for Testing Materials (ASTM). 1990. Standard Guide for Conducting Static 96-Hour Toxicity Tests with Microalgae. Procedure E 1218-90. ASTM, Philadelphia, PA.

2. Short-term Methods for Estimating the Chronic Toxicity of Effluent and Receiving Waters to West Coast Marine and Estuarine Organisms. EPA/600/R-95/136. August 1995.

3. Short-term Methods for Estimating the Chronic Toxicity of Effluent and Receiving Waters to Marine and Estuarine Organisms. EPA/600/4-90/003. July 1994.

Table AE-2. Critical Life Stage Toxicity Tests for Fresh Waters

Species	(Scientific Name)	Effect	Test Duration	Reference
Fathead minnow	*(Pimephales promelas)*	Survival; growth rate	7 days	4
Water flea	*(Ceriodaphnia dubia)*	Survival; number of young	7 days	4
Alga	*(Selenastrum capricornutum)*	Final cell density	4 days	4

Toxicity Test Reference:

4. Short-term Methods for Estimating the Chronic Toxicity of Effluents and Receiving Waters to Freshwater Organisms, fourth Edition Chronic manual (EPA-821-R-02-013, October 2002).

Table AE-3. Toxicity Test Requirements for Stage One Screening Phase

Requirements	Receiving Water Characteristics		
	Discharges to Coast	Discharges to San Francisco Bay[1]	
	Ocean	Marine/Estuarine	Freshwater
Taxonomic diversity	1 plant 1 invertebrate 1 fish	1 plant 1 invertebrate 1 fish	1 plant 1 invertebrate 1 fish
Number of tests of each salinity type: Freshwater[2] Marine/Estuarine	0 4	1 or 2 3 or 4	3 0
Total number of tests	4	5	3

[1] (a) Marine refers to receiving water salinities greater than 1 part per thousand (ppt) at least 95 percent of the time during a normal water year.
(b) Freshwater refers to receiving water with salinities less than 1 ppt at least 95 percent of the time during a normal water year.
(b) Estuarine refers to receiving water salinities that fall between those of marine and freshwater, as described above.

[2] The freshwater species may be substituted with marine species if:
(a) The salinity of the effluent is above 1 ppt greater than 95 percent of the time, or
(b) The ionic strength (TDS or conductivity) of the effluent at the test concentration used to determine compliance is documented to be toxic to the test species.

ATTACHMENT F - FACT SHEET

Contents

Tables

ATTACHMENT F – FACT SHEET

As described in section II of this Order, this Fact Sheet includes the legal requirements and technical rationale that serve as the basis for the requirements of this Order.

This Order has been prepared under a standardized format to accommodate a broad range of discharge requirements for dischargers in California. Only those sections or subsections of this Order that are specifically identified as "not applicable" have been determined not to apply to this Discharger. Sections or subsections of this Order not specifically identified as "not applicable" fully apply to this Discharger.

I. PERMIT INFORMATION

The following table summarizes administrative information related to the Central Contra Costa Wastewater Treatment Plant (Plant):

Table F-1. Facility Information

WDID	2 071008001
CIWQS Place ID	213875
Discharger	Central Contra Costa Sanitary District
Name of Facility	Central Contra Costa Sanitary District Wastewater Treatment Plant and its associated wastewater collection system
Facility Address	5019 Imhoff Place, Martinez, CA 94553
	Contra Costa County
Facility Contact, Title, Phone	Margaret P. Orr, P.E., Director of Plant Operations, (925) 228-9500
Authorized Person to Sign and Submit Reports	Same as above
Mailing Address	5019 Imhoff Place, Martinez, CA 94553
Billing Address	Same as Mailing Address
Type of Facility	Publicly Owned Treatment Works (POTW)
Major or Minor Facility	Major
Threat to Water Quality	1
Complexity	B
Pretreatment Program	Yes
Reclamation Requirements	Regional Water Board Order No. 96-011
Mercury and PCBs Discharge Requirements	Regional Water Board Order No. R2-2007-0077
Facility Permitted Flow	53.8 million gallons per day (MGD) (average daily dry weather flow)
Facility Design Flow	53.8 MGD (average dry weather flow) 250 MGD (peak wet weather influent design flow)
Watershed	Suisun
Receiving Water	Suisun Bay
Receiving Water Type	Estuarine
Service Area	Danville, Lafayette, Martinez, Moraga, Orinda, Pleasant Hill, San Ramon, Walnut Creek, Concord, Clayton, and adjacent unincorporated areas, including Alamo, Blackhawk, Clyde, and Pacheco
Service Area Population	455,000

A. Central Contra Costa Sanitary District (hereinafter the Discharger) is the owner and operator of the Plant, a Publicly Owned Treatment Works, and its associated sewage collection system (collectively, the Facility). The Plant provides secondary treatment of wastewater collected from its service area and discharges it to Suisun Bay.

For the purposes of this Order, references to the "discharger" or "permittee" in applicable federal and state laws, regulations, plans, or policy are held to be equivalent to references to the Discharger herein.

B. Discharge of treated wastewater from the Plant to Suisun Bay, a water of the State and the United States, is currently regulated by Order No. R2-2007-0008 (NPDES Permit No. CA0037648), which was adopted on January 23, 2007, became effective on April 1, 2007, and expires on March 31, 2012.

C. The Discharger filed a Report of Waste Discharge (ROWD) and submitted a complete application for renewal of its waste discharge requirements (WDRs) and NPDES permit dated June 1, 2011.

II. FACILITY DESCRIPTION

A. Description of Wastewater and Biosolids Treatment

1. **Facility Description.** The Discharger owns and operates the Central Contra Costa Sanitary District Wastewater Treatment Plant (hereinafter the Plant) and its associated wastewater collection system (hereinafter collectively the Facility). The Plant, located north of Concord and east of Martinez, (See Attachment B) provides secondary treatment of domestic, commercial, and industrial wastewater for Danville, Lafayette, Martinez, Moraga, Orinda, Pleasant Hill, San Ramon, Walnut Creek, Concord, Clayton, and adjacent unincorporated areas, including Alamo, Blackhawk, Clyde, and Pacheco. The population of the service area is approximately 455,000. From April 2007 through December 2010, the maximum daily influent flow rate was 141 MGD and the average daily flow rate was 38.7 MGD. Both rates are well within the permitted 53.8 MGD average dry weather flow and 250 MGD peak wet weather influent design flow. Twenty-two (22) significant industrial users also discharge to the Facility, and these discharges are regulated by the Facility's pretreatment program.

2. **Collection System.** The Discharger's wastewater collection system includes approximately 1,500 miles of pipeline, ranging from 6 to 102 inches in diameter, and 16 wastewater pumping stations. The City of Concord separately maintains the collection system within most of Concord's city limits and the City of Clayton.

3. **Treatment Description.** Treatment processes consist of screening, grit removal, primary sedimentation, secondary biological treatment, secondary clarification and ultraviolet (UV) disinfection. These steps are shown in the process flow diagram in Attachment C.

4. **Discharge Point.** Secondary-treated wastewater is discharged at Discharge Point 001 to Suisun Bay about 3.5 miles from the Facility via a submerged outfall equipped with a multiport diffuser. The location of the outfall diffuser is approximately 1600 feet offshore at an average depth of approximately 24 feet. The diffuser is 6 feet in diameter and imbedded 4 feet into the sediment. The diffuser is oriented nearly perpendicular to the shoreline. It consists of 11 upward-facing ports separated 11.5 feet on center, for a total length of 115 feet.

The Plant has holding basins for temporary storage of wet weather flows, with a combined volume of 170 million gallons. These basins are used to store excess wastewater after primary treatment when inflow exceeds the Plant's secondary treatment capacity. When flows subside, the stored wastewater is routed back to the headworks for full treatment.

5. **Recycled Water.** In 2010, the Discharger diverted approximately 600 million gallons of UV-disinfected effluent from the outfall to the Recycling Plant for tertiary treatment through sand/anthracite filtration and chlorine disinfection. This recycled water volume represents about 4% of the total wastewater treated. Recycled water is stored in a covered seven million gallon reservoir prior to distribution. Recycled water customers include landscape irrigators, corporation yards, private soil farms, concrete recycling and batch plants, and the county animal shelter. Recycled water activities are regulated under Regional Water Board Order No. 96-011.

6. **Biosolids Management.** Secondary sludge is thickened via dissolved air flotation, combined with primary sludge and lime, dewatered by centrifuges, and incinerated on-site. Ash is hauled by a contractor to an off-site recycling facility and used as a soil amendment. If Facility incinerators are inoperable, biosolids may be hauled to local landfills for disposal or to an East Bay Municipal Utility District site for treatment prior to disposal.

7. **Stormwater Discharge.** The Discharger is not required to be covered under the State Water Board's statewide industrial stormwater NPDES permit (NPDES General Permit No. CAS000001). All stormwater flows in contact with equipment or sewage at the Plant and the pump stations serving the Plant are collected and directed to the headworks for treatment.

8. **Outfall Pipe Maintenance.** About every 5 to 10 years, during the dry season, the Discharger drains and inspects its 3.5-mile long, 72-inch reinforced concrete outfall pipe to verify the alignment and assess the physical integrity of the pipe joint seals. During this time, fully-treated effluent is diverted to a holding basin and then discharged to Walnut Creek from a concrete weir at the holding basin. This maintenance project was last done in 2003, and it took 18 weeks to dewater the outfall, inspect it, repair the damaged joints, and return it to service. The Discharger has informed the Regional Water Board that an inspection (and any necessary repairs identified as a result) will have to be completed again during this permit cycle to ensure the integrity of the outfall. The fully treated effluent will be discharged to Walnut Creek via a new concrete weir structure at the holding basin. The Discharger expects that the diversion time will be similar to the last event, although it could vary depending on the extent of repairs needed. This bypass is necessary for unavoidable maintenance and is subject to Federal Standard Provisions, section I.G (Attachment D).

B. Discharge Point and Receiving Waters

The location of the discharge point and the receiving waters are shown below:

Table F-2. Outfall Locations

Discharge Point	Effluent Description	Discharge Point Latitude	Discharge Point Longitude	Receiving Water
001	Secondary Treated Municipal Wastewater	38° 02′ 44″ N	122° 05′ 55″ W	Suisun Bay

Suisun Bay is located within the Suisun watershed. Suisun Bay is a tidally influenced, estuarine waterbody. The discharge to Suisun Bay is a deep water discharge and receives a minimum of 10:1 initial dilution.

C. Summary of Existing Requirements and Self-Monitoring Report Data

Effluent limitations applicable to Discharge Point 001 contained in the previous Order (Order No. R2-2007-0008) and representative monitoring data from the term of the previous permit are presented below.

Table F-3. Historic Effluent Limitations and Monitoring Data for Conventional and Non-Conventional Pollutants

Parameter	Units	Effluent Limitations			Monitoring Data (From 04/07- 02/11)
		Monthly Average	Weekly Average	Daily Maximum	Highest Daily Discharge
5-day Carbonaceous Biological Oxygen Demand ($CBOD_5$)	mg/L	25	40	50	27
Total Suspended Solids (TSS)	mg/L	30	45	60	20
Oil and Grease	mg/L	10	---	20	4.4
pH	s.u.	6.0 – 9.0 at all times			6.8 – 8.0
Enterococcus Bacteria	Colonies/ 100 mL	35[1]	---	---	2400

Legend to Table F-3:
Unit Abbreviations:

mg/L = milligrams per liter
s.u. = standard units
mL = = milliliters

Footnotes to Table F-3:
< = Non-Detect
[1] The enterococci limitation is expressed as a monthly geometric mean.

Table F-4. Historic Effluent Limitations and Monitoring Data for Toxic Pollutants

Parameter	Units	Effluent Limitations		Monitoring Data (From 04/07 – 02/11)
		Monthly Average	Daily Maximum	Highest Daily
Copper	µg/L	14	20	12
Lead	µg/L	3.5	8.2	1.1
Cyanide	µg/L	20	45	6.7
Acrylonitrile	µg/L	6.3	13	1.1
Dioxin-TEQ	µg/L	1.4×10^{-8}	2.8×10^{-8}	1.2×10^{-9}

Legend to Table F-4:
Unit Abbreviations:

µg/L = micrograms per liter

D. Compliance Summary

1. **Compliance with Numeric Effluent Limits.** The Discharger has not exceeded any effluent limitation during the previous permit term.

2. **Compliance with Previous Permit Provisions.** The Discharger has completed all special activities required by the previous permit provisions.

E. Planned Changes

No changes are planned during this Order's term.

III. APPLICABLE PLANS, POLICIES, AND REGULATIONS

This Order's requirements are based on the requirements and authorities described in this section.

A. Legal Authorities

This Order is issued pursuant to federal Clean Water Act (CWA) section 402 and implementing regulations adopted by the USEPA and chapter 5.5, division 7, of the California Water Code (CWC), commencing with section 13370. It serves as an NPDES permit for point source discharges from the Facility to surface waters. This Order also serves as waste discharge requirements (WDRs) pursuant to article 4, chapter 4, division 7 of the CWC (commencing with section 13260).

B. California Environmental Quality Act (CEQA)

Under CWC section 13389, this action to issue an NPDES permit is exempt from Chapter 3 of CEQA.

C. State and Federal Regulations, Policies, and Plans

1. **Water Quality Control Plan.** *The Water Quality Control Plan for the San Francisco Bay Basin* (hereinafter the Basin Plan) is the Regional Water Board's master water quality control planning document. It designates beneficial uses and water quality objectives (WQOs) for waters of the State, including surface and groundwater. It also includes implementation programs to achieve WQOs. The Basin Plan was duly adopted by the Regional Water Board and approved by the State Water Resources Control Board (State Water Board), the Office of Administrative Law, and USEPA. Requirements of this Order implement the Basin Plan.

   The Basin Plan implements State Water Board Resolution No. 88-63, which establishes State policy that all waters, with certain exceptions, should be considered suitable or potentially suitable for municipal or domestic supply. Because of the marine influence on Suisun Bay, total dissolved solids levels exceed 3,000 mg/L and thereby meet an exception to State Water Board Resolution No. 88-63. The MUN designation therefore does not apply to the receiving water. The Basin Plan beneficial uses of Suisun Bay are listed below.

Table F-5. Basin Plan Beneficial Uses

Receiving Water Name	Beneficial Uses
Suisun Bay	Industrial Service Supply (IND) Industrial Process Supply (PROC) Commercial, and Sport Fishing (COMM) Estuarine Habitat (EST) Fish Migration (MIGR) Preservation of Rare and Endangered Species (RARE) Fish Spawning (SPWN) Wildlife Habitat (WILD) Water Contact Recreation (REC1) Non-Contact Water Recreation (REC2) Navigation (NAV)

The State Water Board's *Water Quality Control Plan for Enclosed Bays and Estuaries—Part 1, Sediment Quality* became effective on August 25, 2009. This plan supersedes other narrative sediment quality objectives and establishes new sediment quality objectives and related implementation provisions for specifically defined sediments in most bays and estuaries.

2. **National Toxics Rule (NTR) and California Toxics Rule (CTR).** USEPA adopted the NTR on December 22, 1992, and amended it on May 4, 1995, and November 9, 1999. About 40 criteria in the NTR and apply in California. On May 18, 2000, USEPA adopted the CTR. The CTR promulgated new toxics criteria for California and, in addition, incorporated the previously adopted NTR criteria that applied in the State. The CTR was amended on February 13, 2001. These rules contain water quality criteria (WQC) for priority toxic pollutants.

3. **State Implementation Policy.** On March 2, 2000, the State Water Board adopted the *Policy for Implementation of Toxics Standards for Inland Surface Waters, Enclosed Bays, and Estuaries of California* (hereinafter the State Implementation Policy [SIP]). The SIP became effective on April 28, 2000, with respect to the priority pollutant criteria promulgated through the NTR and to the WQOs established in the Basin Plan. The SIP became effective on May 18, 2000, with respect to the priority pollutant criteria promulgated through the CTR. The State Water Board adopted amendments to the SIP on February 24, 2005 that became effective on July 13, 2005. The SIP establishes implementation provisions for priority pollutant criteria and objectives and provisions for chronic toxicity control. Requirements of this Order implement the SIP.

4. **Alaska Rule.** On March 30, 2000, USEPA revised its regulation that specifies when new and revised state and tribal water quality standards (WQS) become effective for CWA purposes [65 Fed. Reg. 24641 (April 27, 2000), codified at 40 CFR 131.21]. Under the revised regulation (also known as the Alaska Rule), new and revised standards submitted to USEPA after May 30, 2000, must be approved by USEPA before being used for CWA purposes. The final rule also provides that standards already in effect and submitted to USEPA by May 30, 2000, may be used for CWA purposes, whether or not approved by USEPA.

5. **Antidegradation Policy.** 40 CFR 131.12 requires that state WQS include an antidegradation policy consistent with the federal policy. The State Water Board established California's

antidegradation policy in State Water Board Resolution 68-16, which incorporates the federal antidegradation policy where the federal policy applies under federal law and requires that existing quality of waters be maintained unless degradation is justified based on specific findings. The Regional Water Board's Basin Plan implements, and incorporates by reference, both the State and federal antidegradation policies.

6. **Anti-Backsliding Requirements.** CWA sections 402(o)(2) and 303(d)(4) and 40 CFR 122.44(l) prohibit backsliding in NPDES permits. These anti-backsliding provisions require that effluent limitations in a reissued permit must be as stringent as those in the previous permit, with some exceptions in which limitations may be relaxed.

D. Impaired Water Bodies on CWA 303(d) List

In November 2006, pursuant to CWA section 303(d), USEPA approved a revised list of impaired water bodies prepared pursuant to CWA section 303(d), which requires identification of specific water bodies where it is expected that water quality standards will not be met after implementation of technology-based effluent limitations on point sources. In November 2010, USEPA partially approved an updated 303(d) list. Where it has not done so already, the Regional Water Board plans to adopt Total Maximum Daily Loads (TMDLs) for pollutants on the 303(d) list. TMDLs establish wasteload allocations for point sources and load allocations for non-point sources, and are established to achieve the water quality standards for the impaired waterbodies. The SIP requires that final effluent limitations for all 303(d)-listed pollutants be consistent with TMDLs and associated wasteload allocations.

Suisun Bay is listed as an impaired waterbody. The pollutants impairing Suisun Bay are chlordane, DDT, dieldrin, exotic species, dioxins and furans, mercury, nickel, PCBs, and selenium. On February 12, 2008, the USEPA approved a TMDL for mercury in the San Francisco Bay. On March 29, 2010, the USEPA approved a TMDL for PCBs in San Francisco Bay. The TMDLs for mercury and PCBs are incorporated into the Basin Plan and apply to this discharge. Mercury and PCBs discharges from the Facility are regulated by Regional Water Board Order No. R2-2007- 0077 as amended by Regional Water Board Order No. R2-2011-0012.

IV. RATIONALE FOR EFFLUENT LIMITATIONS AND DISCHARGE SPECIFICATIONS

The CWA requires point source dischargers to control the amount of conventional, non-conventional, and toxic pollutants that are discharged into waters of the United States. The control of pollutants discharged is established through effluent limitations and other requirements in NPDES permits. There are two principal bases for effluent limitations in the NPDES regulations: 40 CFR section 122.44(a) requires that permits include applicable technology-based limitations and standards; and section 122.44(d) requires that permits include water quality-based effluent limitations (WQBELs) to attain and maintain applicable numeric and narrative WQC to protect the beneficial uses of the receiving water.

Several specific factors affecting the development of limitations and requirements in this Order are discussed as follows:

A. Discharge Prohibitions

1. **Discharge Prohibition III.A (No discharge other than that described in this Order):** This prohibition is based on 40 CFR 122.21(a), "Duty to Apply," and CWC section 13260, which requires filing an application and Report of Waste Discharge before a discharge can occur. Discharges not described in the permit application and Report of Waste Discharge, and subsequently in this Order, are prohibited.

2. **Discharge Prohibition III.B (No discharge receiving less than 44:1 dilution):** This Order allows a dilution credit of 44:1 in the calculation of one or more water quality-based effluent limitations, based on information of dilution achieved by the Discharger's current outfall. Thus, this prohibition is necessary to ensure that the assumptions used to derive the dilution credit remain substantially the same so that the limitations are protective of water quality.

3. **Discharge Prohibition III.C (Bypass or overflow of untreated or partially treated wastewaters to waters of the U.S. is prohibited, except as provided for in section I.G of Attachment D):** This prohibition is based on 40 CFR 122.41(m). See Federal Standard Provisions, Attachment D, section G.

4. **Discharge Prohibition III.D (Average dry weather flow not to exceed permitted dry weather flow):** This prohibition is based on the design treatment capacity of the Facility treatment system. The permitted average dry weather flow rate is 53.8 MGD. Exceedance of the Plant's average dry weather flow could result in lowering the reliability of achieving compliance with water quality requirements.

5. **Discharge Prohibition III.E (No sanitary sewer overflows):** Basin Plan Discharge Prohibition 15 (Table 4-1) and the CWA prohibit the discharge of wastewater to surface waters except as authorized under an NPDES permit. Publicly owned treatment works must achieve secondary treatment at a minimum and any more stringent limitations necessary to meet water quality standards [33 U.S.C. § 1311 (b)(1)(B and C)]. A sanitary sewer overflow that results in the discharge of raw sewage, or wastewater not meeting this Order's effluent limitations, to surface waters is therefore prohibited under the CWA and the Basin Plan.

B. Conventional and Non-Conventional Pollutant Limitations

1. Scope and Authority

CWA section 301(b) and 40 CFR 122.44 require that permits include conditions meeting technology-based requirements at a minimum, and any more stringent effluent limitations necessary to meet applicable water quality standards. The discharge authorized by this Order must meet the minimum federal technology-based requirements based on Secondary Treatment Standards at 40 CFR 133, which are summarized below. The 30-day average percent removal for BOD_5 (or $CBOD_5$) and TSS, by concentration, is not to be less than 85 percent. The Basin Plan contains additional requirements for certain pollutants.

Table F-6. Secondary Treatment Requirements

Parameters	Monthly Average	Weekly Average
BOD_5	30 mg/L	45 mg/L
$CBOD_5$ [1]	25 mg/L	40 mg/L

TSS	30 mg/L	45 mg/L
pH	6.0 – 9.0 standard units	

Footnotes for Table F-6:
[1] At the option of the permitting authority, $CBOD_5$ effluent limitations may be substituted for BOD_5 limitations.

2. Effluent Limitations for Conventional and Non-conventional Pollutants

a. **$CBOD_5$ and TSS.** The effluent limitations for $CBOD_5$ and TSS, including the 85 percent removal requirement, are required by the secondary treatment standards requirements.

b. **Oil and Grease.** Basin Plan Table 4-2 requires the oil and grease effluent limitations in this Order.

c. **pH.** Secondary treatment regulations and Basin Plan Table 4-2 require the pH limitation in this Order for deep water discharges.

d. **Enterococcus Bacteria.** The enterococcus bacteria effluent limitations are based on Basin Plan Table 4-2A.

C. Water Quality-Based Effluent Limitations (WQBELs) for Toxic Substances

WQBELs have been derived for toxic pollutants to implement WQOs that protect beneficial uses. Both the beneficial uses and the WQOs have been approved pursuant to federal law. The procedures for calculating individual WQBELs are based on the SIP and the Basin Plan. Most Basin Plan beneficial uses and WQOs were approved under State law and submitted to and approved by USEPA prior to May 30, 2000. Any WQOs and beneficial uses submitted to USEPA prior to May 30, 2000, but not approved by USEPA before that date, are nonetheless "applicable water quality standards for purposes of the [Clean Water] Act" pursuant to 40 CFR 131.21(c)(1). Collectively, this Order's restrictions on individual pollutants are no more stringent than those required by CWA water quality standards.

1. Scope and Authority

a. NPDES regulations at 40 CFR 122.44(d)(1)(i) mandate that permits include effluent limitations for all pollutants that are or may be discharged at levels that have reasonable potential to cause or contribute to an excursion of a water quality standard, including numeric and narrative objectives within a standard. As specified in 40 CFR 122.44(d)(1)(i), permits are required to include WQBELs for all pollutants "which the Director determines are or may be discharged at a level which will cause, have the reasonable potential to cause, or contribute to an excursion above any state water quality standard."

The process for determining "reasonable potential" and calculating WQBELs when necessary is intended to protect the designated beneficial uses of the receiving water as specified in the Basin Plan, and achieve applicable WQOs contained in the CTR, NTR, and other state plans and policies.

b. NPDES regulations and the SIP provide the basis to establish Maximum Daily Effluent Limitations (MDELs).

(1) NPDES Regulations. NPDES regulations at 40 CFR 122.45(d) state, "For continuous discharges all permit effluent limitations, standards, and prohibitions, including those necessary to achieve water quality standards, shall *unless impracticable* be stated as MDELs and average monthly discharge limitations (AMELs) for all discharges other than publicly owned treatment works."

(2) SIP. SIP section 1.4 requires WQBELs to be expressed as MDELs and AMELs.

c. MDELs are used in this Order to protect against acute water quality effects. The MDELs are necessary for preventing fish kills or mortality to aquatic organisms.

2. Beneficial Uses and WQOs

The WQOs applicable to the receiving water for this discharge are from the Basin Plan; the CTR, established by USEPA at 40 CFR 131.38; and the NTR, established by USEPA at 40 CFR 131.36. Some pollutants have WQOs established by more than one of these sources.

a. Basin Plan. The Basin Plan specifies numeric WQOs for 10 priority toxic pollutants, as well as narrative WQOs for toxicity and bioaccumulation in order to protect beneficial uses. The pollutants for which the Basin Plan specifies numeric objectives are arsenic, cadmium, chromium (VI), copper in marine and freshwater, lead, mercury, nickel, silver, zinc, and cyanide. The narrative toxicity objective states, "All waters shall be maintained free of toxic substances in concentrations that are lethal to or that produce other detrimental responses in aquatic organisms." The bioaccumulation objective states, "Controllable water quality factors shall not cause a detrimental increase in concentrations of toxic substances found in bottom sediments or aquatic life. Effects on aquatic organisms, wildlife, and human health will be considered." Effluent limitations and provisions contained in this Order are designed to implement these objectives, based on available information.

b. CTR. The CTR specifies numeric aquatic life criteria for 23 priority toxic pollutants and numeric human health criteria for 57 priority toxic pollutants. These criteria apply to all inland surface waters and enclosed bays and estuaries of the San Francisco Bay Region, although Basin Plan Tables 3-3 and 3-4 include numeric objectives for certain of these priority toxic pollutants that supersede CTR criteria (except in the South Bay south of the Dumbarton Bridge). Human health criteria are further identified as for "water and organisms" and for "organisms only." The CTR criteria applicable to "organisms only" apply to the receiving water because it is not a source of drinking water.

c. NTR. The NTR establishes numeric aquatic life criteria for selenium and numeric human health criteria for 33 toxic organic pollutants for waters of San Francisco Bay upstream to and including Suisun Bay and the Sacramento River-San Joaquin River Delta.

d. Sediment Quality Objectives. The *Water Quality Control Plan for Enclosed Bays and Estuaries – Part 1, Sediment Quality* contains a narrative WQO, "Pollutants in sediments shall not be present in quantities that, alone or in combination, are toxic to benthic communities in bays and estuaries of California." This WQO is to be implemented by integrating three lines of evidence: sediment toxicity, benthic community condition, and sediment chemistry. The policy requires that if the Regional Water Board determines that a discharge has reasonable potential to cause or contribute to an exceedance of this WQO, it is to impose the WQO as a receiving water limit.

e. **Basin Plan Receiving Water Salinity Policy.** The Basin Plan (like the CTR and the NTR) states that the salinity characteristics (i.e., freshwater vs. saltwater) of the receiving water are to be considered in determining the applicable WQOs. Freshwater criteria apply to discharges to waters with salinities equal to or less than one part per thousand (ppt) at least 95 percent of the time. Saltwater criteria apply to discharges to waters with salinities equal to or greater than 10 ppt at least 95 percent of the time in a normal water year. For discharges to water with salinities between these two categories, or tidally influenced freshwaters that support estuarine beneficial uses, the WQOs are the lower of the salt or freshwater WQOs (the latter calculated based on ambient hardness) for each substance.

   The receiving water for discharge from the facility is Suisun Bay, an estuarine water body based on salinity data collected by the San Francisco Estuary Institute (SFEI) Regional Monitoring Program (RMP). Historically, the RMP conducted sampling at 26 locations throughout the San Francisco Bay region. In 2002, the system was redesigned to incorporate random sampling in place of the 26 established locations. Salinity data collected from March 1993 to August 2001 at the Pacheco Creek (BF10) station and additional random sampling at various locations within Suisun Bay collected from July 2002 to July 2008 indicate that the salinity was less than 1 ppt in 29 percent of the samples and greater than 10 ppt in 18 percent of the samples in Suisun Bay. The waters of Suisun Bay are therefore classified as estuarine, and the reasonable potential analysis and effluent limitations in this Order are based on the more stringent of the fresh and saltwater WQOs.

f. **Receiving Water Hardness.** Ambient hardness data collected at the Pacheco Creek (BF10) RMP station from February 1995 to August 2001 and additional random sampling at various locations within Suisun Bay collected from August 2003 to August 2006 were used to calculate freshwater WQOs that are hardness dependent. To calculate the WQOs for hardness dependent metals, the data set was censored to cap hardness values above 400 mg/L as $CaCO_3$ at 400 mg/L. The resulting data set of 19 values was used to calculate an adjusted geometric mean, which is the value that 30 percent of the measurements fall below. The calculated hardness value was 146 mg/L as $CaCO_3$.

g. **Site-Specific Metals Translators.** NPDES regulations at 40 CFR 122.45(c) require that effluent limitations for metals be expressed as total recoverable metal. Since applicable WQOs for metals are typically expressed as dissolved metal, translators must be used to convert metals concentrations from dissolved to total recoverable and vice versa. The CTR includes default translators; however, site-specific conditions, such as water temperature, pH, suspended solids, and organic carbon greatly affect the form of metal (dissolved, non-filterable, or otherwise) present in the water and therefore available to cause toxicity. In general, the dissolved form of the metal is more available and more toxic to aquatic life than non-filterable forms. Site-specific translators can be developed to account for site-specific conditions, thereby preventing exceedingly stringent or under protective WQOs. For deep water discharges north of Dumbarton Bridge, the Basin Plan translators for copper are 0.38 (AMEL) and 0.66 (MDEL).

3. Determining the Need for WQBELs

Assessing whether a pollutant has reasonable potential to exceed a WQO in the water body is the fundamental step in determining whether or not a WQBEL is required.

a. Reasonable Potential Methodology

For priority pollutants and most other toxic pollutants, the reasonable potential Analysis (RPA) identifies the observed maximum effluent concentration (MEC) for each pollutant based on effluent concentration data. There are three triggers in determining reasonable potential according to SIP section 1.3.

(1) The first trigger (Trigger 1) is activated if the MEC is greater than or equal to the lowest applicable WQO (MEC $\geq$ WQO), which has been adjusted, if appropriate, for pH, hardness, and translator data. If the MEC is greater than or equal to the adjusted WQO, then that pollutant has reasonable potential, and a WQBEL is required.

(2) The second trigger (Trigger 2) is activated if the observed maximum ambient background concentration (B) is greater than the adjusted WQO (B > WQO), and the pollutant is detected in any of the effluent samples.

(3) The third trigger (Trigger 3) is activated if a review of other information determines that a WQBEL is required to protect beneficial uses, even though both MEC and B are less than the WQO.

b. Effluent Data

The Discharger's priority pollutant data and the nature of the discharge were analyzed to determine if the discharge has reasonable potential. The RPA is based on effluent monitoring data collected by the Discharger from April 2007 through January 2011 for most inorganic pollutants, and from May 2007 to December 2010 for most organic pollutants.

c. Ambient Background Data

The SIP states that, for calculating WQBELs, ambient background concentrations are either the observed maximum ambient water column concentrations or, for objectives intended to protect human health from carcinogenic effects, the arithmetic mean of observed ambient water concentrations. Ambient background concentrations are the observed maximum detected water column concentrations for aquatic life protection.

On May 15, 2003, a group of San Francisco Bay Region dischargers known as the Bay Area Clean Water Agencies, or BACWA, submitted a collaborative receiving water study, entitled the *San Francisco Bay Ambient Water Monitoring Interim Report* (2003). This study includes monitoring results from sampling events in 2002 and 2003 for the remaining priority pollutants not monitored by the RMP. This study included the Yerba Buena Island RMP station. Additional data were provided from the BACWA *Ambient Water Monitoring: Final CTR Sampling Update* report, dated June 15, 2004.

For priority pollutants, the RPA was conducted and WQBELs were calculated using RMP data from 1993 through 2009 at the Yerba Buena Island RMP station (BC10), and additional data from the BACWA receiving water study. For ammonia, the RPA was conducted and WQBELs were calculated using receiving water data collected by the Discharger at six monitoring locations between April 2007 and January 2011.

d. RPA for Toxic Pollutants

The MECs, most stringent applicable WQO, and background concentrations used in the RPA are presented in the following table, along with the RPA results (yes or no) for each pollutant. Reasonable potential was not determined for all pollutants because there are not applicable WQOs for all pollutants, and monitoring data are not available for others. Based on a review of the effluent data collected during the previous permit term from April 2007 through January 2011, the pollutants that exhibit reasonable potential at Discharge Point 001 are cyanide, acrylonitrile, bis(2-ethylhexyl)phthalate, and total ammonia by Trigger 1; and copper and dioxin-TEQ by Trigger 3.

Table F-7. Reasonable Potential Analysis Summary

CTR #	Priority Pollutants	Governing WQO (µg/L)	MEC or Minimum DL [1][2] (µg/L)	Maximum Background or Minimum DL [1][2] (µg/L)	RPA Results [3]
1	Antimony	4300	0.55	1.8	No
2	Arsenic	36	1.89	2.46	No
3	Beryllium	No Criteria	0.03	0.215	Ud
4	Cadmium	1.5	0.11	0.1268	No
5a	Chromium (III)	282	2.5	Not Available	No
5b	Chromium (VI)	11	2.5	4.4	No
6	Copper	5.9	12	2.55	Yes [4]
7	Lead	5.2	1.1	0.8040	No
8	Mercury (303(d) listed)[4]	---	---	0.0086	---
9	Nickel (303d listed)	30	2.65	3.73	No
10	Selenium (303(d) listed)	5.0	1.27	0.39	No
11	Silver	2.2	0.8	0.052	No
12	Thallium	6.3	< 0.01	0.21	No
13	Zinc	86	54.3	5.1	No
14	Cyanide	2.9	6.7	< 0.4	Yes
15	Asbestos	No Criteria	--	Not Available	Ud
16	2,3,7,8-TCDD (303(d) listed)	1.4E-08	< 7.5E-07	Not Available	No
	Dioxin TEQ (303(d) listed)	1.4E-08	1.2E-09	7.10E-09	Yes
17	Acrolein	780	<0.40	< 0.5	No
18	Acrylonitrile	0.66	1.1	0.03	Yes
19	Benzene	71	< 0.20	0.05	No
20	Bromoform	360	0.2	0.5	No
21	Carbon Tetrachloride	4.4	< 0.29	0.06	No
22	Chlorobenzene	21000	0.2	0.5	No
23	Chlorodibromomethane	34	0.3	0.05	No
24	Chloroethane	No Criteria	< 0.20	0.5	Ud
25	2-Chloroethylvinyl ether	No Criteria	< 0.20	0.5	Ud
26	Chloroform	No Criteria	0.8	0.5	Ud
27	Dichlorobromomethane	46	< 0.25	0.05	No
28	1,1-Dichloroethane	No Criteria	< 0.20	0.05	Ud
29	1,2-Dichloroethane	99	0.2	0.04	No
30	1,1-Dichloroethylene	3.2	< 0.28	< 0.5	No

CTR #	Priority Pollutants	Governing WQO (µg/L)	MEC or Minimum DL [1][2] (µg/L)	Maximum Background or Minimum DL [1][2] (µg/L)	RPA Results [3]
31	1,2-Dichloropropane	39	< 0.20	< 0.05	No
32	1,3-Dichloropropylene	1700	< 0.20	< 0.5	No
33	Ethylbenzene	29000	< 0.20	< 0.5	No
34	Methyl Bromide	4000	4.4	< 0.5	No
35	Methyl Chloride	No Criteria	1.2	< 0.5	Ud
36	Methylene Chloride	1600	< 0.30	22	No
37	1,1,2,2-Tetrachloroethane	11	0.2	< 0.05	No
38	Tetrachloroethylene	8.9	< 0.40	< 0.05	No
39	Toluene	200000	5.3	< 0.3	No
40	1,2-Trans-Dichloroethylene	140000	< 0.20	< 0.5	No
41	1,1,1-Trichloroethane	No Criteria	< 0.25	< 0.5	Ud
42	1,1,2-Trichloroethane	42	< 0.20	< 0.05	No
43	Trichloroethylene	81	< 0.07	< 0.5	No
44	Vinyl Chloride	525	< 0.25	< 0.5	No
45	2-Chlorophenol	400	< 0.10	< 1.2	No
46	2,4-Dichlorophenol	790	0.3	< 1.3	No
47	2,4-Dimethylphenol	2300	< 0.8	< 1.3	No
48	2-Methyl- 4,6-Dinitrophenol	765	< 0.2	< 1.2	No
49	2,4-Dinitrophenol	14000	< 0.4	< 0.7	No
50	2-Nitrophenol	No Criteria	< 0.1	< 1.3	Ud
51	4-Nitrophenol	No Criteria	< 0.5	< 1.6	Ud
52	3-Methyl 4-Chlorophenol	No Criteria	3.4	< 1.1	Ud
53	Pentachlorophenol	7.9	< 0.6	< 1	No
54	Phenol	4600000	1.9	< 1.3	No
55	2,4,6-Trichlorophenol	6.5	< 0.4	< 1.3	No
56	Acenaphthene	2700	< 0.030	0.0019	No
57	Acenaphthylene	No Criteria	0.21	0.0013	Ud
58	Anthracene	110000	< 0.030	0.0006	No
59	Benzidine	0.00054	< 4.1	< 0.0015	No
60	Benzo(a)Anthracene	0.049	< 0.020	0.0053	No
61	Benzo(a)Pyrene	0.049	< 0.020	0.00029	No
62	Benzo(b)Fluoranthene	0.049	< 0.020	0.0046	No
63	Benzo(ghi)Perylene	No Criteria	< 0.020	0.0027	Ud
64	Benzo(k)Fluoranthene	0.049	< 0.020	0.0015	No
65	Bis(2-Chloroethoxy)Methane	No Criteria	< 0.30	< 0.3	Ud
66	Bis(2-Chloroethyl)Ether	1.4	< 0.10	< 0.3	No
67	Bis(2-Chloroisopropyl)Ether	170000	< 0.10	Not Available	No
68	Bis(2-Ethylhexyl)Phthalate	5.9	21.9	< 0.5	Yes
69	4-Bromophenyl Phenyl Ether	No Criteria	< 0.1	< 0.23	Ud
70	Butylbenzyl Phthalate	5200	0.8	< 0.52	No
71	2-Chloronaphthalene	4300	< 0.2	< 0.3	No
72	4-Chlorophenyl Phenyl Ether	No Criteria	< 0.1	< 0.3	Ud
73	Chrysene	0.049	< 0.02	0.0024	No
74	Dibenzo(a,h)Anthracene	0.049	< 0.03	0.00064	No
75	1,2-Dichlorobenzene	17000	0.3	< 0.8	No
76	1,3-Dichlorobenzene	2600	0.2	< 0.8	No
77	1,4-Dichlorobenzene	2600	0.3	< 0.8	No
78	3,3 Dichlorobenzidine	0.077	< 0.3	< 0.001	No
79	Diethyl Phthalate	120000	0.7	< 0.24	No
80	Dimethyl Phthalate	2900000	< 0.1	< 0.24	No
81	Di-n-Butyl Phthalate	12000	0.5	< 0.5	No
82	2,4-Dinitrotoluene	9.1	< 0.1	< 0.27	No
83	2,6-Dinitrotoluene	No Criteria	< 0.1	< 0.29	Ud

CTR #	Priority Pollutants	Governing WQO (µg/L)	MEC or Minimum DL [1][2] (µg/L)	Maximum Background or Minimum DL [1][2] (µg/L)	RPA Results [3]
84	Di-n-Octyl Phthalate	No Criteria	< 0.1	< 0.38	Ud
85	1,2-Diphenylhydrazine	0.54	< 0.1	0.27	No
86	Fluoranthene	370	1.12	0.29	No
87	Fluorene	14000	0.14	0.38	No
88	Hexachlorobenzene	0.00077	< 0.1	0.0000202	No
89	Hexachlorobutadiene	50	< 0.2	< 0.3	No
90	Hexachlorocyclopentadiene	17000	< 2.6	< 0.31	No
91	Hexachloroethane	8.9	< 0.2	< 0.2	No
92	Indeno(1,2,3-cd)Pyrene	0.049	< 0.02	0.004	No
93	Isophorone	600	< 0.2	< 0.3	No
94	Naphthalene	No Criteria	1.3	0.0023	Ud
95	Nitrobenzene	1900	< 0.3	< 0.25	No
96	N-Nitrosodimethylamine	8.1	< 0.4	< 0.3	No
97	N-Nitrosodi-n-Propylamine	1.4	< 0.3	< 0.001	No
98	N-Nitrosodiphenylamine	16	< 0.1	< 0.001	No
99	Phenanthrene	No Criteria	< 0.02	0.0061	Ud
100	Pyrene	11000	0.05	0.0051	No
101	1,2,4-Trichlorobenzene	No Criteria	< 0.2	< 0.3	Ud
102	Aldrin	0.00014	< 0.002	4.04E-06	No
103	Alpha-BHC	0.013	< 0.002	0.000413	No
104	Beta-BHC	0.046	< 0.002	0.0007034	No
105	Gamma-BHC	0.063	0.02	0.000042	No
106	Delta-BHC	No Criteria	< 0.002	0.00018	Ud
107	Chlordane (303(d) listed)	0.00059	< 0.003	0.000066	No
108	4,4'-DDT (303(d) listed)	0.00059	< 0.003	0.000693	No
109	4,4'-DDE (linked to DDT)	0.00059	< 0.003	0.000313	No
110	4,4'-DDD	0.00084	< 0.003	0.000264	No
111	Dieldrin (303d listed)	0.00014	< 0.002	0.000031	No
112	Alpha-Endosulfan	0.0087	< 0.003	0.000069	No
113	beta-Endolsulfan	0.0087	< 0.003	0.0000819	No
114	Endosulfan Sulfate	240	< 0.002	0.000036	No
115	Endrin	0.0023	< 0.002	Not Available	No
116	Endrin Aldehyde	0.81	< 0.002	0.000019	No
117	Heptachlor	0.00021	< 0.003	0.00002458	No
118	Heptachlor Epoxide	0.00011	< 0.002	0.000413	No
119-125	PCBs sum (303(d) listed)	---	Not Available	Not Available	---
126	Toxaphene	0.0002	< 0.19	Not Available	No
	Tributylin	0.0074	Not Available	< 0.001	No
	Total PAHs	15	1.3	0.26	No
	Total Ammonia [4]	1.6 mg/L	30.2 mg/L	2.4 mg/L	Yes

Footnotes to Table F-7:

[1] The Maximum Effluent Concentration (MEC) and maximum background concentration are the actual detected concentrations unless preceded by a "<" sign, in which case the value shown is the minimum detection level (DL).

[2] The MEC or maximum background concentration is "Not Available" when there are no monitoring data for the constituent.

[3] RPA Results = Yes, if MEC > WQC, B > WQC and MEC is detected, or Trigger 3;
= No, if MEC and B are < WQC or all effluent data are undetected;
= Undetermined (Ud), if no criteria have been promulgated or there are insufficient data.

[4] Copper has reasonable potential by trigger 3 pursuant to Basin Plan Section 7.2.

[5] Units for Total Ammonia are milligrams per liter.

e. **Constituents with limited data.** In some cases, reasonable potential cannot be determined because effluent data are limited, or ambient background concentrations are unavailable. The Discharger will continue to monitor for these constituents in the effluent using analytical methods that provide the best feasible detection limits. When additional data become available, further RPA will be conducted to determine whether numeric effluent limitations are necessary.

f. **Pollutants with No Reasonable Potential.** WQBELs are not included in this Order for constituents that do not demonstrate reasonable potential; however, monitoring for those pollutants is still required. If concentrations of these constituents are found to have increased significantly, the Discharger will be required to investigate the sources of the increases. Remedial measures are required if the increases pose a threat to receiving water quality.

g. **RPA for Sediment Quality Objective.** Pollutants in some receiving water sediments may be present in quantities that alone or in combination are toxic to benthic communities. Efforts are underway to identify stressors causing such conditions. However, to date there is no evidence directly linking compromised sediment conditions to the discharges subject to this Order; therefore the Regional Water Board cannot draw a conclusion about reasonable potential for the discharges to cause or contribute to exceedances of the sediment quality objectives. Nevertheless, the Discharger continues to participate in the RMP, which monitors San Francisco Bay sediment and seeks to identify stressors responsible for degraded sediment quality. Thus far, the monitoring has provided only limited information about potential stressors and sediment transport. The Regional Water Board is exploring options for obtaining additional information that may inform future RPAs.

4. **WQBEL Calculations**

a. **Pollutants with Reasonable Potential.** WQBELs were developed for the toxic and priority pollutants determined to have reasonable potential to cause or contribute to exceedances of the WQOs. The WQBELs were calculated based on WQOs and the procedures specified in SIP section 1.4. The WQOs used for each pollutant with reasonable potential are discussed below.

b. **Dilution Credit.** The SIP allows dilution credits for completely-mixed discharges, and under certain circumstances for incompletely-mixed discharges. The Discharger submitted a Near-field Mixing Zone and Dilution Analysis for the Central Contra Costa Sanitary District Outfall Diffuser to San Pablo Bay, dated May 27, 2011. The report presents the findings regarding the initial dilution of the discharge at the outfall.

The near-field dilution was estimated using the USEPA-supported CORMIX modeling package. The study used the average dry-weather flow rate to calculate a chronic dilution ratio and the 99th percentile daily flow rate to calculate an acute dilution ratio.

The study found that near-field mixing is complete at 125 feet from the diffuser center line. Initial dilutions estimated by CORMIX are:

44:1(D=43) at the permitted average dry weather flow rate (53.8 MGD), representing chronic conditions; and

34:1 (D=33) at the 99th percentile daily effluent flow rate (70.3 MGD), representing acute discharge conditions.

i. **Bioaccumulative Pollutants:** For certain bioaccumulative pollutants, dilution credit is significantly restricted or denied. This determination is based on available data on concentrations of these pollutants in aquatic organisms, sediment, and the water column. Specifically, these pollutants include chlordane, DDT, dieldrin, dioxin compounds, furan compounds, mercury, PCBs, and dioxin-like PCBs, which all appear on the CWA section 303(d) list for Suisun Bay because they impair beneficial uses. The following factors suggest insufficient assimilative capacity in San Francisco Bay for these pollutants.

   Tissue samples taken from fish in San Francisco Bay show the presence of these pollutants at concentrations greater than screening levels (*Contaminant Concentrations in Fish from San Francisco Bay*, May 1997). The results of a 1994 San Francisco Bay pilot study, presented in *Contaminated Levels in Fish Tissue from San Francisco Bay* (Regional Water Board, 1994) also showed elevated levels of chemical contaminants in fish tissues. The Office of Environmental Health and Hazard Assessment completed a preliminary review of the data in the 1994 report and in December 1994 issued an interim consumption advisory covering certain fish species in San Francisco Bay due to the levels of some of these pollutants, including dioxins and pesticides (e.g. DDT). This advisory is still in effect. Therefore, dilution credits are denied for bioaccumulative pollutants on the 303(d) list for which there is lack of data on sources and significant uncertainty about how different sources of these pollutants contribute to bioaccumulation.

ii. **Non-Bioaccumulative Pollutants:** For non-bioaccumulative pollutants (except ammonia), a conservative dilution allowance of 10:1 (D = 9) has been assigned. The 10:1 dilution allowance is consistent with the previous permit and is based, in part, on Basin Plan Prohibition 1 (Table 4-1), which prohibits discharges with less than 10:1 dilution. SIP section 1.4.2 allows for limiting the dilution credit:

   (1) A far-field background station is appropriate because San Francisco Bay is a very complex estuarine system with highly variable and seasonal upstream freshwater inflows and diurnal tidal saltwater inputs. SIP section 1.4.3 allows background conditions to be determined on a discharge-by-discharge or water body-by-water body basis. A water body-by-water body basis approach is taken here due to inherent uncertainties in characterizing ambient background conditions in a complex estuarine system on a discharge-by-discharge basis. The Yerba Buena Island RMP monitoring station, relative to other RMP stations, fits SIP guidance criteria for establishing background conditions. The SIP requires that background water quality data be representative of the ambient receiving water that will mix with the discharge. Water quality data from the Yerba Buena Island monitoring station is representative of the water that will mix with the discharge.

   (2) Because of the complex hydrology of San Francisco Bay, a mixing zone has not been established. There are uncertainties in accurately determining an appropriate mixing zone. The models used to predict dilution have not considered the three dimensional nature of San Francisco Bay currents resulting from the interaction of

tidal flushes and seasonal fresh water outflows. Being heavier and colder than fresh water, ocean salt water enters San Francisco Bay on a twice-daily tidal cycle, generally beneath the warmer fresh water that flows seaward. When these waters mix and interact, complex circulation patterns occur due to the varying densities of the fresh and ocean waters. The complex patterns occur throughout San Francisco Bay, but are most prevalent in the San Pablo, Carquinez Straight, and Suisun Bay areas. The locations of this mixing and interaction change, depending on the strength of each tide. Additionally, sediment loads from the Central Valley change on a long-term basis, affecting the depth of different parts of San Francisco Bay, resulting in alteration of flow patterns, mixing, and dilution at the outfall.

(3) For ammonia, a non-bioaccumulative and non-persistent pollutant, a minimum initial dilution of 44:1 (D = 43) was used to represent chronic conditions (based on the Mixing Zone Study described above), and 34:1 (D=33) was used to represent acute conditions. In granting dilution for ammonia, the Regional Water Board considered that ammonia is not a persistent pollutant and the Basin Plan states, "In most instances, ammonia will be diluted or degraded to a nontoxic state fairly rapidly." As such, there is unlikely to be cumulative toxicity effects associated with discharges containing elevated concentrations of ammonia. Therefore, granting dilution credits based on actual initial dilution is protective of water quality.

c. Development of WQBELs for Specific Pollutants

(1) Copper

(a) **WQOs.** The Basin Plan contains chronic and acute marine WQOs for copper of 6.0 micrograms per liter (μg/L) and 9.4 μg/L, respectively, expressed as dissolved metal (site-specific objectives for San Francisco Bay). These WQOs were converted to total recoverable metal using the site-specific translators of 0.38 (chronic) and 0.66 (acute), as described in section IV.C.2.g, above. The resulting acute WQO is 14 μg/L and chronic WQO is 16 μg/L.

(b) **RPA Results.** This Order establishes effluent limitations for copper because of reasonable potential by Trigger 3, consistent with Basin Plan section 7.2

(c) **WQBELs**. WQBELs for copper, calculated according to SIP procedures with an effluent data coefficient of variation (CV) of 0.21 and a dilution credit of D = 9 (dilution ratio = 10:1), are an AMEL of 89 μg/L and an MDEL of 120 μg/L.

(d) **Anti-backsliding.** The copper limits in this Order are less stringent than those the previous order because they were calculated based on SSOs. CWA section 303(d)(4)(B) allows effluent limits to be revised for water bodies that meet water quality standards if such revisions are consistent with antidegradation policies. Suisun Bay meets its copper WQOs and the SSOs were designed to be protective of beneficial uses. Furthermore, the Basin Plan requires copper action plans for all discharges to Suisun Bay. Therefore, Suisun Bay will not be degraded by copper discharges, antidegradation policies have been met, and revised copper limits are appropriate.

(2) Cyanide

(a) WQOs. The Basin Plan contains chronic and acute marine WQOs for cyanide of 2.9 μg/L and 9.4 μg/L, respectively (site-specific objectives for San Francisco Bay).

(b) RPA Results. This Order establishes effluent limitations for cyanide because the MEC (6.7 μg/L) exceeds the governing WQO (2.9 μg/L), demonstrating reasonable potential by Trigger 1.

(c) WQBELs. WQBELs for cyanide, calculated according to SIP procedures with an effluent data CV of 0.47 and a dilution credit of D = 9 (dilution ratio = 10:1), are an AMEL of 22 μg/L and an MDEL of 39 μg/L.

(d) Anti-backsliding. The cyanide limits in this Order are less stringent than those the previous order because they were calculated based on SSOs. CWA section 303(d)(4)(B) allows effluent limits to be revised for water bodies that meet water quality standards if such revisions are consistent with antidegradation policies. Suisun Bay meets its cyanide WQOs and the SSOs were designed to be protective of beneficial uses. Furthermore, the Basin Plan requires cyanide action plans for all discharges to Suisun Bay. Therefore, Suisun Bay will not be degraded by cyanide discharges, antidegradation policies have been met, and revised cyanide limits are appropriate.

(3) Dioxin – TEQ

(a) WQO. The Basin Plan narrative WQO for bioaccumulative substances states, "Many pollutants can accumulate on particulates, in sediments, or bioaccumulate in fish and other aquatic organisms. Controllable water quality factors shall not cause a detrimental increase in concentrations of toxic substances found in bottom sediments or aquatic life. Effects on aquatic organisms, wildlife, and human health will be considered."

Because it is the consensus of the scientific community that dioxins and furans associate with particulates, accumulate in sediments, and bioaccumulate in the fatty tissue of fish and other organisms, the Basin Plan's narrative bioaccumulation WQO is applicable to these pollutants. Elevated levels of dioxins and furans in fish tissue in San Francisco Bay demonstrate that the narrative bioaccumulation WQO is not being met. USEPA has therefore included Suisun Bay as impaired by dioxin and furan compounds in the current 303(d) listing of receiving waters, where water quality objectives are not being met after imposition of applicable technology-based requirements.

The CTR establishes a numeric WQO for 2,3,7,8-tetrachlorinated dibenzo-p-dioxin (2,3,7,8-TCDD) of 1.4×10^{-8} μg/L for the protection of human health, when aquatic organisms are consumed. When the CTR was promulgated, USEPA stated its support of the regulation of other dioxin and dioxin-like compounds through the use of toxicity equivalencies (TEQs) in NPDES permits. For California waters, USEPA stated specifically, "if the discharge of dioxin or dioxin-like compounds has reasonable potential to cause or contribute to a violation of a narrative criterion, numeric WQBELs for dioxin or dioxin-like compounds should be included in NPDES permits and should be expressed using a TEQ scheme" [65 Fed. Reg. 31682, 31695 (2000)].

This Order uses a TEQ scheme based on a set of toxicity equivalency factors (TEFs) the World Health Organization (WHO) developed in 1998, and a set of bioaccumulation equivalency factors (BEFs) USEPA developed for the Great Lakes region (40 CFR132, Appendix F) to convert the concentration of any congener of dioxin or furan into an equivalent concentration of 2,3,7,8-TCDD. The CTR criterion is used as a criterion for dioxin-TEQ because dioxin-TEQ represents a toxicity weighted concentration equivalent to 2,3,7,8-TCDD, thus translating the narrative bioaccumulation objective into a numeric criterion appropriate for the RPA.

To determine if the discharge of dioxin or dioxin-like compounds has reasonable potential to cause or contribute to a violation of the Basin Plan's narrative bioaccumulation WQO, TEFs and BEFs were used to express the measured concentrations of 16 dioxin congeners in effluent and background samples as 2,3,7,8-TCDD. These "equivalent" concentrations were then compared to the CTR numeric criterion for 2,3,7,8-TCDD (1.4×10^{-8} μg/L). Although the 1998 WHO scheme includes TEFs for dioxin-like PCBs, they are not included in this Order's TEQ scheme. The CTR has established a specific water quality standard for PCBs, and dioxin-like PCBs are included in the analysis of total PCBs.

(b) **RPA Results.** Dioxin-TEQ has been detected in the effluent and the receiving waters are listed as impaired due to dioxin and furan bioaccumulations within the food web. Because the dioxin-TEQ in the discharge could cause or contribute to an exceedance of the Basin Plan's bioaccumulation WQO, there is reasonable potential based on Trigger 3.

(c) **WQBELs.** WQBELs for dioxin-TEQ, calculated according to SIP procedures with a default CV of 0.6 and no dilution credit, are an AMEL of 1.4×10^{-8} μg/L and an MDEL of 2.8×10^{-8} μg/L.

(d) **Anti-backsliding.** Antibacksliding requirements are satisfied because the limits for dioxin-TEQ are the same as the limits in the previous order.

(4) Acrylonitrile

(a) **WQO.** The CTR contains a human health WQO for acrylonitrile of 0.66 μg/L.

(b) **RPA Results.** This Order establishes effluent limitations for acrylonitrile because the MEC (1.1 μg/L) exceeds the WQO (0.66 μg/L), demonstrating reasonable potential by Trigger 1.

(c) **WQBELs.** WQBELs for acrylonitrile, calculated according to SIP procedures with a CV of 0.7 and a dilution credit of D = 9 (dilution ratio = 10:1), are an AMEL of 6.3 μg/L and an MDEL of 14 μg/L. However, the previous order contained an AMEL of 6.3 μg/L and an MDEL of 13 μg/L. The 13 μg/L MDEL is retained from the previous order.

(d) **Antibacksliding.** Antibacksliding requirements are satisfied because the limits for acrylonitrile are the same as the limits in the previous order.

(5) Bis(2-ethylhexyl)phthalate

(a) **WQO.** The CTR contains a human health WQO for bis(2-ehylhexyl)phthalate of 5.9 μg/L.

(b) **RPA Results.** This Order establishes effluent limitations for bis(2-ehylhexyl)phthalate because the MEC (22 μg/L) exceeds the WQO for this pollutant, demonstrating reasonable potential by Trigger 1.

(c) **WQBELs.** WQBELs for bis(2-ethylhexyl)phthalate, calculated according to SIP procedures with a CV of 2.6 and a dilution credit of D = 9 (dilution ratio = 10:1), are an AMEL of 55 μg/L and an MDEL of 170 μg/L.

(d) **Antibacksliding.** Antibacksliding requirements are satisfied because the previous order did not include effluent limitations for bis(2-ethylhexyl)phthalate.

(6) Total Ammonia

(a) **WQOs.** The Basin Plan contains WQOs for un-ionized ammonia of 0.025 mg/L as an annual median and 0.16 mg/L as a maximum upstream of the San Francisco Bay Bridge. These WQOs were translated from un-ionized ammonia concentrations to equivalent total ammonia concentrations (as nitrogen) since: (1) sampling and laboratory methods are not available to analyze for un-ionized ammonia; and (2) the fraction of total ammonia that exists in the toxic un-ionized form depends on the pH, salinity, and temperature of the receiving water.

To translate the Basin Plan un-ionized ammonia objectives, pH, salinity and temperature data were used from six receiving water monitoring stations collected by the Discharger between April 2007 and January 2011. The un-ionized fraction of total ammonia is calculated as follows:

For salinity > 10 ppt: fraction of $NH_3 = \dfrac{1}{1+10^{(pK-pH)}}$

Where:

$$pK = 9.245 + 0.116(I) + 0.0324\,(298 - T) + \frac{0.0415(P)}{(T)}$$

$$I = \text{Molal ionic strength of saltwater} = \frac{19.9273(S)}{(1{,}000 - 1.005109(S))}$$

S = Salinity (parts per thousand)

T = Temperature in degrees Kelvin

P = Pressure (one atmosphere)

The 90th percentile and median un-ionized ammonia fractions were then used to express the daily maximum and the annual average un-ionized objectives as acute and chronic total ammonia objectives, respectively. This approach is consistent with USEPA guidance on translating dissolved metal WQOs to total recoverable metal WQOs (USEPA, 1996, *The Metals Translator: Guidance for Calculating a Total Recoverable Limit from a Dissolved Criterion*, EPA Publication 823-B-96-007.)

The equivalent total ammonia acute and chronic WQOs are 5.0 mg/L and 1.6 mg/L, respectively.

(b) RPA Results. Basin Plan section 4.5.5.2 indicates that WQBELs are to be calculated according to the SIP. Basin Plan section 3.3.20 refers to ammonia as a toxic pollutant. Therefore, the SIP methodology was used to perform the RPA and to calculate effluent limitations for ammonia. This Order establishes effluent limitations for total ammonia because the MEC of 30.2 mg/L (as nitrogen) exceeds the most stringent applicable translated WQO for this pollutant, demonstrating reasonable potential by Trigger 1.

(c) WQBELs. Total ammonia WQBELs were calculated according to SIP procedures using both acute and chronic conditions, and the more stringent (chronic) results were chosen. The effluent data CV was 0.13 and the chronic dilution credit was D=43 (dilution ratio = 44:1). The resulting WQBELs are 84 mg/L (MDEL) and 65 mg/L (AMEL).

Statistical adjustments were made to the total ammonia WQBEL calculations because:

- the Basin Plan's chronic WQO for un-ionized ammonia is based on an annual median instead of the typical 4-day average; and

- the SIP assumes a 4-day average concentration and a monthly sampling frequency of 4 days per month to calculate effluent limitations based on chronic criteria, whereas a 365-day average and a monitoring frequency of 30 days per month (the maximum daily sampling frequency in a month since the averaging period for the chronic criteria is longer than 30 days) were used.

These statistical adjustments are supported by USEPA's *Water Quality Criteria; Notice of Availability; 1999 Update of Ambient Water Quality Criteria for Ammonia,* published on December 22, 1999, in the Federal Register. Following the SIP methodology, the maximum ambient background total ammonia concentration (2.4 mg/L) was used to calculate effluent limitations based on the acute criterion, and the median background total ammonia concentration

(0.15 mg/L) to calculate effluent limitations based on the chronic criterion. Because the Basin Plan's chronic un-ionized ammonia objective is an annual median, the median background concentration is more representative of ambient conditions than a daily maximum.

(d) **Anti-backsliding.** Anti-backsliding requirements are satisfied because the previous permit did not include WQBELs for total ammonia.

e. **Effluent Limit Calculations**

The following table shows the WQBEL calculations for copper, cyanide, dioxin-TEQ, acrylonitrile, bis(2-ethylhexyl)phthalate, and total ammonia.

Table F-8. WQBEL Calculations

Pollutant	Copper	Cyanide	Dioxin-TEQ	Acrylonitrile	Bis(2-ethylhexyl) phthalate	Ammonia (acute)	Ammonia (chronic)
Units	µg/L	µg/L	µg/L	µg/L	µg/L	mg/L-N	mg/L-N
Basis and Criteria Type	BP SSOs	BP SSOs	BP narrative	CTR - HH	CTR - HH	BP aquatic life	BP aquatic life
Criteria – Acute	3.9	9.4	---	---	---	5.0	
Criteria – Chronic	2.5	2.9	---	---	---	---	1.6
HH criteria	---	220000	1.4E-08	0.66	5.9	---	---
Water Effects Ratio	2.4	1	1	1	1	1	1
Lowest WQO	5.9	2.9	1.4E-08	0.66	5.9	5.0	1.6
Site Specific Translator - MDEL	0.66	---	---	---	---	---	---
Site Specific Translator – AMEL	0.38	---	---	---	---	---	---
Dilution Factor (D)	9	9	0	9	9	33	43
No. of samples per month	4	4	4	4	4	4	30
Aquatic life analysis required?	Y	Y	N	N	N	Y	Y
HH analysis required?	N	Y	Y	Y	Y	N	N
Applicable Acute WQO	14	9.4	---	---	---	5.0	---
Applicable Chronic WQO	16	2.9	---	---	---	---	1.6
Background	2.6	0.4	7.1E-08	0.03	0.50	2.4	0.15
Is the pollutant on the 303(d) list?	N	N	Y	N	N	N	N
ECA acute	119	90.4	---	---	---	91	---
ECA chronic	135	25.4	---	---	---	---	64
ECA human health	---	2199996	1.4E-08	6.3	55	---	---
No. of data points <10, or at least 80% non-detect	N	N	Y	N	N	N	N
Average effluent concentration	7.5	2.7	---	0.4	2.2	23	23
Standard Deviation	1.6	1.3	---	0.3	5.7	2.91	2.91
CV calculated	0.21	0.47	---	0.7	2.6	0.13	0.13
CV selected	0.21	0.47	0.6	0.7	2.6	0.13	0.13
ECA acute mult99	0.63	0.39				0.75	
ECA chronic mult99	0.79	0.60					0.98
LTA acute	75.0	35.3				68	
LTA chronic	106.4	15.2					63
Minimum LTA	75.0	15.2				68	63

AMEL mult95	1.2	1.4	1.6	1.7	3.1	1.11	1.04
MDEL mult95	1.6	2.6	3.1	3.6	10	1.34	1.34
AMEL aquatic life	88.6	21.7				126	65
MDEL aquatic life	119	39.0				152	84
MDEL/AMEL multiplier	1.34	1.8	2.01	2.16	3.2	1.20	1.3
AMEL human health		2199996		6.3	55		
MDEL human health		3952332		14	174		
Final limit - AMEL	**89**	**22**	**1.4E-08**	**6.3**	**55**		**65**
Final limit - MDEL	**120**	**39**	**2.8E-08**	**13**	**170**		**84**

5. Whole Effluent Acute Toxicity

This Order includes effluent limitations for whole effluent acute toxicity based on Basin Plan Table 4-3. All bioassays are to be performed according to the USEPA approved method in 40 CFR 136, currently *Methods for Measuring the Acute Toxicity of Effluents and Receiving Waters to Freshwater and Marine Organisms*, 5th Edition. The approved test species currently specified in the Monitoring and Reporting Program (Attachment E) is the fathead minnow.

6. Whole Effluent Chronic Toxicity

a. **Toxicity Objective.** Basin Plan section 3.3.18 states, "There shall be no chronic toxicity in ambient waters. Chronic toxicity is a detrimental biological effect on growth rate, reproduction, fertilization success, larval development, population abundance, community composition, or any other relevant measure of the health of an organism, population, or community."

b. **Reasonable Potential Analysis.** The previous permit included chronic toxicity triggers of a single sample maximum of 20 TUc and a 3-sample median of 10 TUc, which would trigger accelerated chronic toxicity testing if exceeded. The Discharger conducted chronic toxicity testing every two months during the previous permit term using *Haliotis rufescens* and/or *Americamysis bahia*. Chronic toxicity testing results from April 2007 through January 2011 indicate the maximum single sample result was 19.6 TUc, and the maximum 3-sample median was 14.7 TUc. From July to December 2009, the Discharger exceeded the 3-sample median trigger several times, which triggered accelerated monitoring and a TIE. The TIE indicated that the cause of the toxicity was ammonia. Since then, with permission from Regional Water Board staff, the Discharger has been filtering its chronic toxicity samples through a Zeolite filter to remove ammonia. The Discharger has not exceeded chronic toxicity trigger levels since December 2009.

c. **Permit Requirements.** Chronic toxicity requirements are based on the narrative Basin Plan toxicity objective and are unchanged from the previous order.

d. **Screening Phase Study and Monitoring Requirement.** The Discharger is required to conduct a chronic toxicity screening phase study, as described in MRP Appendix E-1 (Attachment E) prior to permit issuance. The Discharger's July 19, 2011, chronic toxicity screening study indicated that *Selenastrum capricornutum* (green algae) or *Americamysis bahia* (mysid shrimp) were equally the most sensitive species. The MRP specifies that either species may be used for chronic toxicity testing during the permit term. The accelerated monitoring trigger levels are consistent with the previous permit and Table 4-6 of the Basin Plan.

7. Ammonia Mass Limit

This Order seeks to maintain existing ammonia treatment performance to avoid possible ammonium-related degradation of receiving water quality. In water, ammonia exists in two forms: un-ionized ammonia (NH_3) and ammonium (ionized ammonia, NH_4^+). Together, these forms are referred to as "total ammonia." The relative proportion between the two forms depends on pH, temperature, and salinity. The Basin Plan contains WQOs for un-ionized ammonia of 0.025 mg/L as an annual median and 0.16 mg/L as a maximum upstream of the San Francisco Bay Bridge, but there are no numeric WQOs for ammonium. The total ammonia WQBELs described in section IV.C.4.c.6, above, implement only the un-ionized ammonia WQOs.

Recent studies indicate that ammonium may affect Suisun Bay through at least two mechanisms: effects on diatoms and effects on copepods. Diatoms are single-cell algae that significantly contribute to primary production in Suisun Bay (the base of the food web). Copepods are important secondary producers, providing food for many fish. The potential impacts of Suisun Bay ammonium are of increasing concern but not well understood. Suisun Bay is very complex hydrologically, chemically, and biologically, and these complexities make it difficult to determine the severity of any possible impacts. There is also insufficient information to weigh the relative contribution of the Discharger's ammonium discharges to those of other sources. While the Discharger is responsible for the largest ammonia load discharged directly to Suisun Bay, there are also many other sources, both local and upstream.

Studies are necessary to determine the potential extent of any possible ammonium impacts and to develop ammonium limitations, if necessary, that protect beneficial uses. These studies are currently in progress. In the meantime, this Order seeks to maintain current treatment performance and avoid any possible degradation of receiving water quality related to ammonium by establishing a performance-based limit of 5500 kg/day of total ammonia as a monthly average. This limit was calculated by multiplying the 95th percentile ammonia concentration (27.2 mg/L) by the permitted dry weather flow (53.8 MGD) and a unit conversion factor of 3.785. The 95th percentile concentration was calculated by transforming the ammonia data to obtain a normal distribution (using the square of the ammonia concentrations). Historical data indicates the Discharger can comply with this limit.

8. Anti-backsliding and Antidegradation

Effluent limitations in this Order that are less stringent than those in the previous permit or are not retained from the previous permit comply with anti-backsliding and antidegradation requirements for the reasons explained below:

- This Order does not retain the daily maximum effluent limits from the previous permit for $CBOD_5$ and TSS. These limits are inconsistent with federal secondary treatment standards and Table 4-2 of the Basin Plan. It is also inconsistent with 40 CFR 122.45(d) that excludes maximum daily limits for publically owned treatment works unless impracticable. The previous permit did not provide a rationale for these limits other than that they were retained from the permit before that one. Removal of daily maximum limits for $CBOD_5$ and TSS is exempt from antibacksliding pursuant to Clean Water Act 402(o)(2)(ii) to correct a technical or legal mistake in a technology-based limitation. Compliance with anti-degradation is assured by retaining the same weekly and monthly technology-based limits as the previous permit.

- This Order does not retain the mercury effluent limit in the previous permit because mercury discharges to San Francisco Bay are now regulated by Regional Water Board Order No. R2-2007-0077, which is a watershed permit that implements the San Francisco

Bay Mercury TMDL. Order No. R2-2007-0077 complied with anti-backsliding and antidegradation requirements.

- The previous permit contained effluent limitations for lead; however, the RPA shows that the discharge no longer demonstrates reasonable potential for this pollutant to cause or contribute to exceedances of the applicable WQOs. This Order, therefore, does not retain these limitations. Elimination of these limitations is consistent with State Water Board Order No. WQ 2001-16. Receiving water quality will not be degraded because the Discharger will maintain its current level of treatment.

- This Order contains copper and cyanide limits based on SSOs that were developed from new site-specific information for Suisun Bay and are less than those in the previous permit. However, CWA section 303(d)(4)(B) allows effluent limits to be revised for water bodies that meet water quality standards if such revisions are consistent with antidegradation policies. Suisun Bay meets its copper and cyanide WQOs and the SSOs were designed to be protective of beneficial uses. Furthermore, the Basin Plan requires copper and cyanide action plans for all discharges to Suisun Bay. Therefore, Suisun Bay will not be degraded by copper and cyanide discharges, antidegradation policies have been met, and revised copper and cyanide limits are appropriate.

- The previous permit contained a mass loading limitation for dioxin-TEQ in addition to the concentration-based limitation. The permit retains the concentration-based limitation but not the mass-based loading limitation. Because the concentration-based limitations are the same as those in the previous permit, and because the permit does not allow an increase in the permitted flow rate from the Facility, removal of the mass-based loading limits will not allow any increase in discharges of dioxin-TEQ from the Facility. Receiving water quality will not be degraded because the Discharger will maintain its current level of treatment.

V. RATIONALE FOR RECEIVING WATER LIMITATIONS

Receiving water limitations V.A.1 and V.A.2 are based on the narrative and numeric objectives contained in Basin Plan Chapter 3. Receiving water limitation V.A.3 is retained from the previous permit and requires compliance with federal and State water quality standards.

VI. RATIONALE FOR MONITORING AND REPORTING REQUIREMENTS

The principal purposes of a monitoring program are to:

- Document compliance with waste discharge requirements and prohibitions established by the Regional Water Board,

- Facilitate self-policing by the Discharger in the prevention and abatement of pollution arising from waste discharge,

- Develop or assist in the development of limitations, discharge prohibitions, national standards of performance, pretreatment and toxicity standards, and other standards, and

- Prepare water and wastewater quality inventories.

The MRP is a standard requirement in almost all NPDES permits issued by the Regional Water Board, including this Order. It contains definitions of terms and sets out requirements for reporting routine monitoring data in accordance with NPDES regulations, the CWC, and State and Regional Water Board policies. The MRP also defines the sampling stations and frequency, the pollutants to be monitored, and additional reporting requirements. Pollutants to be monitored include all parameters for which effluent limitations are specified. Monitoring for additional constituents, for which no effluent limitations are established, is also required to provide data for future completion of RPAs.

The following provides the rationale for the monitoring and reporting requirements contained in the MRP for this facility:

A. Influent Monitoring

Influent monitoring requirements at INF-001 for $CBOD_5$ and TSS are unchanged from the previous permit to allow determination of compliance with this Order's 85% removal requirement. Flow monitoring is also retained to evaluate compliance with Prohibition III.D (average dry weather flow).

B. Effluent Monitoring

The MRP retains most effluent monitoring requirements at Monitoring Location EFF-001 from the previous permit. Changes in effluent monitoring are summarized as follows:

- The MRP retains routine monitoring for the toxic pollutants with effluent limitations (copper, cyanide, dioxin-TEQ, and acrylonitrile.) Monitoring for all other priority toxic pollutants is required to characterize the discharge pursuant to characterization study required by Provision VI.C.2.

- Routine effluent monitoring for bis(2-ethylhexyl)phthalate is established to determine compliance with the newly established effluent limitations.

- The MRP does not retain explicit monitoring requirements from the previous permit for EFF-002, EFF-003, EFF-004, and EFF-005 because additional monitoring at these locations are not necessary to assess permit compliance.

C. Whole Effluent Toxicity Testing Requirements

1. **Acute Toxicity.** Monthly 96-hour bioassay testing is required to demonstrate compliance with the effluent limitation for acute toxicity. The MRP requires the use of either fathead minnow or rainbow trout as the bioassay test species.

2. **Chronic Toxicity.** This Order establishes the requirement for the Discharger to conduct chronic toxicity testing quarterly to ensure the discharge has acceptable levels of chronic toxicity. The Discharger conducted an effluent toxicity screening study during the previous permit term, which determined that *Selenastrum capricornutum* (green algae) and *Americamysis bahia* (mysid shrimp) were equally the most sensitive species. The permit therefore requires the use of either species as the testing species for chronic toxicity. The Discharger shall re-screen in accordance

with MRP Appendix E-1 (Attachment E) after any significant change in the nature of the effluent or prior to 180 days prior to the expiration of this Order.

D. Receiving Water Monitoring

The Discharger is not required to collect receiving water information as long as it continues to support the RMP program.

E. Pretreatment and Biosolids Monitoring

This Order specifies pretreatment and biosolids monitoring requirements to ensure compliance with pretreatment and biosolids regulations. The previous permit did not contain specific pretreatment and biosolids monitoring, but the Discharger continued to monitor biosolids anyway for the same pretreatment and biosolids parameters it had monitored before the previous permit. Composites made up of discrete grabs for several parameters are necessary because of the potential loss of the constituents during automatic compositing. Hexavalent chromium is chemically unstable. It, cyanide, and BNAs are also somewhat volatile. For these same reasons, discrete analyses are also necessary since constituents are subject to loss during compositing at the laboratory.

VII. RATIONALE FOR PROVISIONS

A. Standard Provisions (Provision VI.A)

Standard Provisions, which in accordance with 40 CFR 122.41and 122.42 apply to all NPDES discharges and must be included in every NPDES permit, are provided in Attachments D of this Order. NPDES regulations at 40 CFR 122.41(a)(1) and (b) through (n) establish conditions that apply to all state-issued NPDES permits. These conditions must be incorporated into the permits either expressly or by reference. NPDES regulations at 40 CFR 123.25(a)(12) allow the state to omit or modify conditions to impose more stringent requirements. The Regional Standard Provisions (Attachment G) supplement the Federal Standard Provisions. In accordance with 40 CFR 123.25, this Order omits federal conditions that address enforcement authority specified in 40 CFR 122.41(j)(5) and (k)(2) because the CWC enforcement authority is more stringent. In lieu of these conditions, this Order incorporates by reference CWC section 13387(e).

B. MRP Requirements (Provision VI.B)

The Discharger is required to monitor the permitted discharge in order to evaluate compliance with permit conditions. Monitoring requirements are contained in the MRP (Attachment E), Federal Standard Provisions (Attachment D), and Regional Standard Provisions (Attachment G). This provision requires compliance with these documents and is authorized by 40 CFR 122.41(h) and (j), and CWC sections 13267 and 13383.

C. Special Provisions (Provision VI.C)

1. Reopener Provisions

These provisions are based on 40 CFR 122.63 and allow modification of this Order and its effluent limitations as necessary in response to updated WQOs, regulations, or other new relevant information that may be established in the future and other circumstances allowed by law. Regional Water Board staff intends to reassess the appropriateness of the total ammonia effluent limitations in Table 7 of the Order by April 1, 2015. The permit may be reopened at any time under the circumstances set forth in Provision VI.C.1 of the Order.

2. Effluent Characterization Study and Report

This Order does not include effluent limitations for priority pollutants that do not demonstrate reasonable potential, but this provision requires the Discharger to continue monitoring for these pollutants as described in the Regional Standard Provisions (Attachment G) and as specified in the MRP (Attachment E). If concentrations of these constituents increase significantly, the Discharger must investigate the source of the increases and establish remedial measures if the increases result in reasonable potential to cause or contribute to an excursion above the applicable WQO. This requirement may be satisfied through identification of the constituent as a "pollutant of concern" in the Dischargers' Pollutant Minimization Program, described in Provision VI.C.3 of the Order. This provision is based on the SIP.

3. Best Management Practices and Pollutant Minimization Program

This provision for a Pollutant Minimization Program is based on Basin Plan Chapter 4 (section 4.13.2) and SIP Chapter 2 (section 2.4.5).

4. Special Provisions for POTWs

a. **Pretreatment Program.** This provision is based on 40 CFR 403 (General Pretreatment Regulations for Existing and New Sources of Pollution) and is retained from the previous permit. The Discharger implements a pretreatment program due to the nature and volume of industrial influent to the Plant.

b. **Biosolids Management Practices.** This provision is based on Basin Plan Chapter 4, section 4.17, and 40 CFR Parts 257 and 503, and is retained from the previous permit.

c. **Sanitary Sewer Overflows and Sewer System Management Plan.** This provision is to explain the Order's requirements as they relate to the Discharger's collection system, and to promote consistency with the State Water Board-adopted General Collection System WDRs (General Order, Order No. 2006-0003-DWQ).

The General Order requires public agencies that own or operate sanitary sewer systems with greater than one mile of pipes or sewer lines to enroll for coverage under the General Order. The General Order requires agencies to develop sanitary sewer management plans and report all sanitary sewer overflows, among other requirements and prohibitions. Furthermore, the General Order contains requirements for operation and maintenance of collection systems and for reporting and mitigating sanitary sewer overflows. Inasmuch that the Discharger's collection system is part of the system that is subject to this Order, certain standard provisions apply as specified in Provision VI.C.5. The Discharger must comply with both the General Order and this Order. The Discharger and public agencies that are discharging wastewater to the Facility were required to obtain enrollment for regulation under the General Order by December 1, 2006. The State

Water Board amended the General Order (No. WQ 2008-0002-EXEC) on February 20, 2008 to strengthen the notification and reporting requirements for sanitary sewer overflows.

5. **Other Special Provisions**

a. **Copper Action Plan.** This provision is based on Basin Plan section 7.2.1.2. It is necessary to ensure that use of copper site-specific objectives is consistent with antidegradation policies.

b. **Cyanide Action Plan.** This provision is based on Basin Plan section 4.7.2.2. It is necessary to ensure that use of cyanide site-specific objectives is consistent with antidegradation policies.

c. **Nutrient Discharge Work Plan, Studies, and Reports.** This provision is intended to ensure that sufficient information is available in a timely manner to conduct reasonable potential analyses for ammonia and ammonium, and if necessary to revise the water quality-based effluent limits in this Order. This provision is authorized by CWC section 13267.

d. **Facility Plan and Site Characterization.** This provision is intended to obtain information regarding the Discharger's ability to remove ammonia from the discharge and is authorized by CWC section 13267.

VIII. PUBLIC PARTICIPATION

The Regional Water Board is considering the issuance of WDRs that will serve as an NPDES permit for the Facility. As a step in the WDRs adoption process, Regional Water Board staff has developed tentative WDRs. The Regional Water Board encourages public participation in the WDR adoption process.

A. Notification of Interested Parties

The Regional Water Board has notified the Discharger and interested agencies and persons of its intent to prescribe WDRs for the discharge and has provided them with an opportunity to submit written comments and recommendations. Notification was provided through the Contra Costa Times.

B. Written Comments

Staff determinations are tentative. Interested persons are invited to submit written comments concerning these tentative WDRs. Comments must be submitted either in person or by mail to the Executive Office at the Regional Water Board at the address provided on the cover page of this Order, to the attention of Vince Christian.

To receive full consideration and a written response, written comments must be received at the Regional Water Board offices by 5:00 p.m. on November 1, 2011.

C. Public Hearing

The Regional Water Board will hold a public hearing on the tentative WDRs during its regular meeting at the following date and time, and at the following location:

Date:	February 8, 2012
Time:	9:00 am
Location:	Elihu Harris State Office Building 1515 Clay Street, 1st Floor Auditorium Oakland, CA 94612
Contact:	Vince Christian, (510) 622-2336, email VChristian@waterboards.ca.gov

Interested persons are invited to attend. At the public hearing, the Regional Water Board will hear testimony, if any, pertinent to the discharge, WDRs, and permit. Oral testimony will be heard; however, for accuracy of the record, important testimony should be in writing.

Dates and venues may change. The Regional Water Board web address is http://www.waterboards.ca.gov/sanfranciscobay, where one can access the current agenda for changes in dates and locations.

D. Waste Discharge Requirements Petitions

Any aggrieved person may petition the State Water Board to review the decision of the Regional Water Board regarding the final WDRs. The petition must be submitted within 30 days of the Regional Water Board's action to the following address:

State Water Resources Control Board
Office of Chief Counsel
P.O. Box 100, 1001 I Street
Sacramento, CA 95812-0100

E. Information and Copying

The Report of Waste Discharge, related documents, tentative effluent limitations, and special provisions, comments received, and other information are on file and may be inspected at the address above at any time between 9:00 a.m. and 5:00 p.m., Monday through Friday. Copying of documents may be arranged by calling 510-622-2300.

F. Register of Interested Persons

Any person interested in being placed on the mailing list for information regarding the WDRs and NPDES permit should contact the Regional Water Board, reference this facility, and provide a name, address, and phone number.

G. Additional Information

Requests for additional information or questions regarding this order should be directed to Vince Christian at 510-622-2336 or e-mail at VChristian@waterboards.ca.gov.

CALIFORNIA REGIONAL WATER QUALITY CONTROL BOARD
SAN FRANCISCO BAY REGION

ATTACHMENT G
REGIONAL STANDARD PROVISIONS, AND MONITORING AND REPORTING REQUIREMENTS
(SUPPLEMENT TO ATTACHMENT D)

For

NPDES WASTEWATER DISCHARGE PERMITS

March 2010

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CALIFORNIA REGIONAL WATER QUALITY CONTROL BOARD
SAN FRANCISCO BAY REGION

REGIONAL STANDARD PROVISIONS, AND MONITORING AND REPORTING REQUIREMENTS (SUPPLEMENT TO ATTACHMENT D)

FOR

NPDES WASTEWATER DISCHARGE PERMITS

APPLICABILITY

This document applies to dischargers covered by a National Pollutant Discharge Elimination System (NPDES) permit. This document does not apply to Municipal Separate Storm Sewer System (MS4) NPDES permits.

The purpose of this document is to supplement the requirements of Attachment D, Standard Provisions. The requirements in this supplemental document are designed to ensure permit compliance through preventative planning, monitoring, recordkeeping, and reporting. In addition, this document requires proper characterization of issues as they arise, and timely and full responses to problems encountered. To provide clarity on which sections of Attachment D this document supplements, this document is arranged in the same format as Attachment D.

I. STANDARD PROVISIONS - PERMIT COMPLIANCE

A. Duty to Comply – Not Supplemented

B. Need to Halt or Reduce Activity Not a Defense – Not Supplemented

C. Duty to Mitigate – This supplements I.C. of Standard Provisions (Attachment D)

1. **Contingency Plan** - The Discharger shall maintain a Contingency Plan as originally required by Regional Water Board Resolution 74-10 and as prudent in accordance with current municipal facility emergency planning. The Contingency Plan shall describe procedures to ensure that existing facilities remain in, or are rapidly returned to, operation in the event of a process failure or emergency incident, such as employee strike, strike by suppliers of chemicals or maintenance services, power outage, vandalism, earthquake, or fire. The Discharger may combine the Contingency Plan and Spill Prevention Plan into one document. Discharge in violation of the permit where the Discharger has failed to develop and implement a Contingency Plan as described below will be the basis for considering the discharge a willful and negligent violation of the permit pursuant to California Water Code Section 13387. The Contingency Plan shall, at a minimum, contain the provisions of a. through g. below.

   a. Provision of personnel for continued operation and maintenance of sewerage facilities during employee strikes or strikes against contractors providing services.

   b. Maintenance of adequate chemicals or other supplies and spare parts necessary for continued operations of sewerage facilities.

c. Provisions of emergency standby power.

d. Protection against vandalism.

e. Expeditious action to repair failures of, or damage to, equipment and sewer lines.

f. Report of spills and discharges of untreated or inadequately treated wastes, including measures taken to clean up the effects of such discharges.

g. Programs for maintenance, replacement, and surveillance of physical condition of equipment, facilities, and sewer lines.

2. **Spill Prevention Plan** - The Discharger shall maintain a Spill Prevention Plan to prevent accidental discharges and minimize the effects of such events. The Spill Prevention Plan shall:

a. Identify the possible sources of accidental discharge, untreated or partially treated waste bypass, and polluted drainage;

b. Evaluate the effectiveness of present facilities and procedures, and state when they became operational; and

c. Predict the effectiveness of the proposed facilities and procedures, and provide an implementation schedule containing interim and final dates when they will be constructed, implemented, or operational.

This Regional Water Board, after review of the Contingency and Spill Prevention Plans or their updated revisions, may establish conditions it deems necessary to control accidental discharges and to minimize the effects of such events. Such conditions may be incorporated as part of the permit upon notice to the Discharger.

D. Proper Operation & Maintenance – This supplements I.D of Standard Provisions (Attachment D)

1. **Operation and Maintenance (O&M) Manual** - The Discharger shall maintain an O&M Manual to provide the plant and regulatory personnel with a source of information describing all equipment, recommended operational strategies, process control monitoring, and maintenance activities. To remain a useful and relevant document, the O&M Manual shall be kept updated to reflect significant changes in treatment facility equipment and operational practices. The O&M Manual shall be maintained in usable condition and be available for reference and use by all relevant personnel and Regional Water Board staff.

2. **Wastewater Facilities Status Report** - The Discharger shall regularly review, revise, or update, as necessary, its Wastewater Facilities Status Report. This report shall document how the Discharger operates and maintains its wastewater collection, treatment, and disposal facilities to ensure that all facilities are adequately staffed, supervised, financed, operated, maintained, repaired, and upgraded as necessary to provide adequate and reliable transport, treatment, and disposal of all wastewater from both existing and planned future wastewater sources under the Discharger's service responsibilities.

3. **Proper Supervision and Operation of Publicly Owned Treatment Works (POTWs)** - POTWs shall be supervised and operated by persons possessing certificates of appropriate grade pursuant to Division 4, Chapter 14, Title 23 of the California Code of Regulations.

E. Property Rights – Not Supplemented

F. Inspection and Entry – Not Supplemented

G. Bypass – Not Supplemented

H. Upset – Not Supplemented

I. Other – This section is an addition to Standard Provisions (Attachment D)

1. Neither the treatment nor the discharge of pollutants shall create pollution, contamination, or nuisance as defined by California Water Code Section 13050.

2. Collection, treatment, storage, and disposal systems shall be operated in a manner that precludes public contact with wastewater, except in cases where excluding the public is infeasible, such as private property. If public contact with wastewater could reasonably occur on public property, warning signs shall be posted.

3. If the Discharger submits a timely and complete Report of Waste Discharge for permit reissuance, this permit continues in force and effect until a new permit is issued or the Regional Water Board rescinds the permit.

J. Storm Water – This section is an addition to Standard Provisions (Attachment D)

These provisions apply to facilities that do not direct all storm water flows from the facility to the wastewater treatment plant headworks.

1. Storm Water Pollution Prevention Plan (SWPP Plan)

The SWPP Plan shall be designed in accordance with good engineering practices and shall address the following objectives:

a. To identify pollutant sources that may affect the quality of storm water discharges; and

b. To identify, assign, and implement control measures and management practices to reduce pollutants in storm water discharges.

The SWPP Plan may be combined with the existing Spill Prevention Plan as required in accordance with Section C.2. The SWPP Plan shall be retained on-site and made available upon request of a representative of the Regional Water Board.

2. Source Identification

The SWPP Plan shall provide a description of potential sources that may be expected to add significant quantities of pollutants to storm water discharges, or may result in non-storm water discharges from the facility. The SWPP Plan shall include, at a minimum, the following items:

a. A topographical map (or other acceptable map if a topographical map is unavailable), extending one-quarter mile beyond the property boundaries of the facility, showing the wastewater treatment facility process areas, surface water bodies (including springs and wells), and discharge point(s) where the facility's storm water discharges to a municipal storm drain system or other points of discharge to waters of the State. The requirements of this paragraph may be included in the site map required under the following paragraph if appropriate.

b. A site map showing the following:

1) Storm water conveyance, drainage, and discharge structures;

2) An outline of the storm water drainage areas for each storm water discharge point;

3) Paved areas and buildings;

4) Areas of actual or potential pollutant contact with storm water or release to storm water, including but not limited to outdoor storage and process areas; material loading, unloading, and access areas; and waste treatment, storage, and disposal areas;

5) Location of existing storm water structural control measures (i.e., berms, coverings, etc.);

6) Surface water locations, including springs and wetlands; and

7) Vehicle service areas.

c. A narrative description of the following:

1) Wastewater treatment process activity areas;

2) Materials, equipment, and vehicle management practices employed to minimize contact of significant materials of concern with storm water discharges;

3) Material storage, loading, unloading, and access areas;

4) Existing structural and non-structural control measures (if any) to reduce pollutants in storm water discharges; and

5) Methods of on-site storage and disposal of significant materials.

d. A list of pollutants that have a reasonable potential to be present in storm water discharges in significant quantities.

3. Storm Water Management Controls

The SWPP Plan shall describe the storm water management controls appropriate for the facility and a time schedule for fully implementing such controls. The appropriateness and priorities of controls in the SWPP Plan shall reflect identified potential sources of pollutants. The description of storm water management controls to be implemented shall include, as appropriate:

a. Storm water pollution prevention personnel

Identify specific individuals (and job titles) that are responsible for developing, implementing, and reviewing the SWPP Plan.

b. Good housekeeping

Good housekeeping requires the maintenance of clean, orderly facility areas that discharge storm water. Material handling areas shall be inspected and cleaned to reduce the potential for pollutants to enter the storm drain conveyance system.

c. Spill prevention and response

Identify areas where significant materials can spill into or otherwise enter storm water conveyance systems and their accompanying drainage points. Specific material handling procedures, storage requirements, and cleanup equipment and procedures shall be identified, as appropriate. The necessary equipment to implement a cleanup shall be available, and personnel shall be trained in proper response, containment, and cleanup of spills. Internal reporting procedures for spills of significant materials shall be established.

d. Source control

Source controls include, for example, elimination or reduction of the use of toxic pollutants, covering of pollutant source areas, sweeping of paved areas, containment of potential pollutants, labeling of all storm drain inlets with "No Dumping" signs, isolation or separation of industrial and non-industrial pollutant sources so that runoff from these areas does not mix, etc.

e. Storm water management practices

Storm water management practices are practices other than those that control the sources of pollutants. Such practices include treatment or conveyance structures, such as drop inlets, channels, retention and detention basins, treatment vaults, infiltration galleries, filters, oil/water separators, etc. Based on assessment of the potential of various sources to contribute pollutants to storm water discharges in significant quantities, additional storm water management practices to remove pollutants from storm water discharges shall be implemented and design criteria shall be described.

f. Sediment and erosion control

Measures to minimize erosion around the storm water drainage and discharge points, such as riprap, revegetation, slope stabilization, etc., shall be described.

g. Employee training

Employee training programs shall inform all personnel responsible for implementing the SWPP Plan. Training shall address spill response, good housekeeping, and material management practices. New employee and refresher training schedules shall be identified.

h. Inspections

All inspections shall be done by trained personnel. Material handling areas shall be inspected for evidence of, or the potential for, pollutants entering storm water discharges. A tracking or follow up procedure shall be used to ensure appropriate response has been taken in response to an inspection. Inspections and maintenance activities shall be documented and recorded. Inspection records shall be retained for five years.

i. Records

A tracking and follow-up procedure shall be described to ensure that adequate response and corrective actions have been taken in response to inspections.

4. Annual Verification of SWPP Plan

An annual facility inspection shall be conducted to verify that all elements of the SWPP Plan are accurate and up-to-date. The results of this review shall be reported in the Annual Report to the Regional Water Board described in Section V.C.f.

K. Biosolids Management – This section is an addition to Standard Provisions (Attachment D)

Biosolids must meet the following requirements prior to land application. The Discharger must either demonstrate compliance or, if it sends the biosolids to another party for further treatment or distribution, must give the recipient the information necessary to ensure compliance.

1. Exceptional quality biosolids meet the pollutant concentration limits in Table III of 40 CFR Part 503.13, Class A pathogen limits, and one of the vector attraction reduction requirements in 503.33(b)(1)-(b)(8). Such biosolids do not have to be tracked further for compliance with general requirements (503.12) and management practices (503.14).

2. Biosolids used for agricultural land, forest, or reclamation shall meet the pollutant limits in Table I (ceiling concentrations) and Table II or Table III (cumulative loadings or pollutant concentration limits) of 503.13. They shall also meet the general requirements (503.12) and management practices (503.14) (if not exceptional quality biosolids) for Class A or Class B pathogen levels with associated access restrictions (503.32) and one of the 10 vector attraction reduction requirements in 503.33(b)(1)-(b)(10).

3. Biosolids used for lawn or home gardens must meet exceptional quality biosolids limits.

4. Biosolids sold or given away in a bag or other container must meet the pollutant limits in either Table III or Table IV (pollutant concentration limits or annual pollutant loading rate limits) of 503.13. If Table IV is used, a label or information sheet must be attached to the biosolids packing that explains Table IV (see 503.14). The biosolids must also meet the Class A pathogen limits and one of the vector attraction reduction requirements in 503.33(b)(1)-(b)(8).

II. STANDARD PROVISIONS – PERMIT ACTION – Not Supplemented

III. STANDARD PROVISIONS – MONITORING

A. Sampling and Analyses – This section is a supplement to III.A and III.B of Standard Provisions (Attachment D)

1. Use of Certified Laboratories

Water and waste analyses shall be performed by a laboratory certified for these analyses in accordance with California Water Code Section 13176.

2. Use of Appropriate Minimum Levels

Table C lists the suggested analytical methods for the 126 priority pollutants and other toxic pollutants that should be used, unless a particular method or minimum level (ML) is required in the MRP.

For priority pollutant monitoring, when there is more than one ML value for a given substance, the Discharger may select any one of the analytical methods cited in Table C for compliance determination, or any other method described in 40 CFR part 136 or approved by USEPA (such as the

1600 series) if authorized by the Regional Water Board. However, the ML must be below the effluent limitation and water quality objective. If no ML value is below the effluent limitation and water quality objective, then the method must achieve an ML no greater than the lowest ML value indicated in Table C. All monitoring instruments and equipment shall be properly calibrated and maintained to ensure accuracy of measurements.

3. Frequency of Monitoring

The minimum schedule of sampling analysis is specified in the MRP portion of the permit.

a. Timing of Sample Collection

1) The Discharger shall collect samples of influent on varying days selected at random and shall not include any plant recirculation or other sidestream wastes, unless otherwise stipulated by the MRP.

2) The Discharger shall collect samples of effluent on days coincident with influent sampling unless otherwise stipulated by the MRP or the Executive Officer. The Executive Officer may approve an alternative sampling plan if it is demonstrated to be representative of plant discharge flow and in compliance with all other permit requirements.

3) The Discharger shall collect grab samples of effluent during periods of day-time maximum peak effluent flows (or peak flows through secondary treatment units for facilities that recycle effluent flows).

4) Effluent sampling for conventional pollutants shall occur on at least one day of any multiple-day bioassay test the MRP requires. During the course of the test, on at least one day, the Discharger shall collect and retain samples of the discharge. In the event a bioassay test does not comply with permit limits, the Discharger shall analyze these retained samples for pollutants that could be toxic to aquatic life and for which it has effluent limits.

i. The Discharger shall perform bioassay tests on final effluent samples; when chlorine is used for disinfection, bioassay tests shall be performed on effluent after chlorination-dechlorination; and

ii. The Discharger shall analyze for total ammonia nitrogen and calculate the amount of un-ionized ammonia whenever test results fail to meet the percent survival specified in the permit.

b. Conditions Triggering Accelerated Monitoring

1) If the results from two consecutive samples of a constituent monitored in a 30-day period exceed the monthly average limit for any parameter (or if the required sampling frequency is once per month and the monthly sample exceeds the monthly average limit), the Discharger shall, within 24 hours after the results are received, increase its sampling frequency to daily until the results from the additional sampling show that the parameter is in compliance with the monthly average limit.

2) If any maximum daily limit is exceeded, the Discharger shall increase its sampling frequency to daily within 24 hours after the results are received that indicate the exceedance of the maximum daily limit until two samples collected on consecutive days show compliance with the maximum daily limit.

3) If final or intermediate results of an acute bioassay test indicate a violation or threatened violation (e.g., the percentage of surviving test organisms of any single acute bioassay test is less than 70 percent), the Discharger shall initiate a new test as soon as practical, and the Discharger shall investigate the cause of the mortalities and report its findings in the next self monitoring report (SMR).

4) The Discharger shall calibrate chlorine residual analyzers against grab samples as frequently as necessary to maintain accurate control and reliable operation. If an effluent violation is detected, the Discharger shall collect grab samples at least every 30 minutes until compliance with the limit is achieved, unless the Discharger monitors chlorine residual continuously. In such cases, the Discharger shall continue to conduct continuous monitoring as required by its permit.

5) When a bypass occurs (except one subject to provision III.A.3.b.6 below), the Discharger shall monitor flows and collect samples on a daily basis for all constituents at affected discharge points that have effluent limits for the duration of the bypass (including acute toxicity using static renewals), except chronic toxicity, unless otherwise stipulated by the MRP.

6) Unless otherwise stipulated by the MRP, when a bypass approved pursuant to Attachment D, Standard Provisions, Sections I.G.2 or I.G.4, occurs, the Discharger shall monitor flows and, using appropriate procedures as specified in the MRP, collect and retain samples for affected discharge points on a daily basis for the duration of the bypass. The Discharger shall analyze for total suspended solids (TSS) using 24-hour composites (or more frequent increments) and for bacteria indicators with effluent limits using grab samples. If TSS exceeds 45 mg/L in any composite sample, the Discharger shall also analyze the retained samples for that discharge for all other constituents that have effluent limits, except oil and grease, mercury, dioxin-TEQ, and acute and chronic toxicity. Additionally, at least once each year, the Discharger shall analyze the retained samples for one approved bypass discharge event for all other constituents that have effluent limits, except oil and grease, mercury, dioxin-TEQ, and acute and chronic toxicity. This monitoring shall be in addition to the minimum monitoring specified in the MRP.

c. Storm Water Monitoring

The requirements of this section only apply to facilities that are not covered by an NPDES permit for storm water discharges and where not all site storm drainage from process areas (i.e., areas of the treatment facility where chemicals or wastewater could come in contact with storm water) is directed to the headworks. For storm water not directed to the headworks during the wet season (October 1 to April 30), the Discharger shall:

1) Conduct visual observations of the storm water discharge locations during daylight hours at least once per month during a storm event that produces significant storm water discharge to observe the presence of floating and suspended materials, oil and grease, discoloration, turbidity, and odor, etc.

2) Measure (or estimate) the total volume of storm water discharge, collect grab samples of storm water discharge from at least two storm events that produce significant storm water discharge, and analyze the samples for oil and grease, pH, TSS, and specific conductance.

The grab samples shall be taken during the first 30 minutes of the discharge. If collection of the grab samples during the first 30 minutes is impracticable, grab samples may be taken during the first hour of the discharge, and the Discharger shall explain in the Annual Report why the grab sample(s) could not be taken in the first 30 minutes.

3) Testing for the presence of non-storm water discharges shall be conducted no less than twice during the dry season (May 1 to September 30) at all storm water discharge locations. Tests may include visual observations of flows, stains, sludges, odors, and other abnormal conditions; dye tests; TV line surveys; or analysis and validation of accurate piping schematics. Records shall be maintained describing the method used, date of testing, locations observed, and test results.

4) Samples shall be collected from all locations where storm water is discharged. Samples shall represent the quality and quantity of storm water discharged from the facility. If a facility discharges storm water at multiple locations, the Discharger may sample a reduced number of locations if it establishes and documents through the monitoring program that storm water discharges from different locations are substantially identical.

5) Records of all storm water monitoring information and copies of all reports required by the permit shall be retained for a period of at least three years from the date of sample, observation, or report.

d. Receiving Water Monitoring

The requirements of this section only apply when the MRP requires receiving water sampling.

1) Receiving water samples shall be collected on days coincident with effluent sampling for conventional pollutants.

2) Receiving water samples shall be collected at each station on each sampling day during the period within one hour following low slack water. Where sampling during lower slack water is impractical, sampling shall be performed during higher slack water. Samples shall be collected within the discharge plume and down current of the discharge point so as to be representative, unless otherwise stipulated in the MRP.

3) Samples shall be collected within one foot of the surface of the receiving water, unless otherwise stipulated in the MRP.

B. Biosolids Monitoring – This section supplements III.B of Standard Provisions (Attachment D)

When biosolids are sent to a landfill, sent to a surface disposal site, or applied to land as a soil amendment, they must be monitored as follows:

1. Biosolids Monitoring Frequency

Biosolids disposal must be monitored at the following frequency:

Metric tons biosolids/365 days	**Frequency**
0-290	Once per year
290-1500	Quarterly
1500-15,000	Six times per year
Over 15,000	Once per month

(Metric tons are on a dry weight basis)

2. Biosolids Pollutants to Monitor

Biosolids shall be monitored for the following constituents:

Land Application: arsenic, cadmium, copper, mercury, molybdenum, nickel, lead, selenium, and zinc

Municipal Landfill: Paint filter test (pursuant to 40 CFR 258)

Biosolids-only Landfill or Surface Disposal Site (if no liner and leachate system): arsenic, chromium, and nickel

C. Standard Observations – This section is an addition to III of Standard Provisions (Attachment D)

1. Receiving Water Observations

The requirements of this section only apply when the MRP requires standard observations of the receiving water. Standard observations shall include the following:

a. *Floating and suspended materials* (e.g., oil, grease, algae, and other macroscopic particulate matter): presence or absence, source, and size of affected area.

b. *Discoloration and turbidity*: description of color, source, and size of affected area.

c. *Odor*: presence or absence, characterization, source, distance of travel, and wind direction.

d. *Beneficial water use*: presence of water-associated waterfowl or wildlife, fisherpeople, and other recreational activities in the vicinity of each sampling station.

e. *Hydrographic condition*: time and height of corrected high and low tides (corrected to nearest National Oceanic and Atmospheric Administration location for the sampling date and time of sample collection).

f. *Weather conditions*:

1) Air temperature; and

2) Total precipitation during the five days prior to observation.

2. Wastewater Effluent Observations

The requirements of this section only apply when the MRP requires wastewater effluent standard observations. Standard observations shall include the following:

a. *Floating and suspended material of wastewater origin* (e.g., oil, grease, algae, and other macroscopic particulate matter): presence or absence.

b. *Odor*: presence or absence, characterization, source, distance of travel, and wind direction.

3. Beach and Shoreline Observations

The requirements of this section only apply when the MRP requires beach and shoreline standard observations. Standard observations shall include the following:

a. *Material of wastewater origin*: presence or absence, description of material, estimated size of affected area, and source.

b. *Beneficial use*: estimate number of people participating in recreational water contact, non-water contact, or fishing activities.

4. Land Retention or Disposal Area Observations

The requirements of this section only apply to facilities with on-site surface impoundments or disposal areas that are in use. This section applies to both liquid and solid wastes, whether confined or unconfined. The Discharger shall conduct the following for each impoundment:

a. Determine the amount of freeboard at the lowest point of dikes confining liquid wastes.

b. Report evidence of leaching liquid from area of confinement and estimated size of affected area. Show affected area on a sketch and volume of flow (e.g., gallons per minute [gpm]).

c. Regarding odor, describe presence or absence, characterization, source, distance of travel, and wind direction.

d. Estimate number of waterfowl and other water-associated birds in the disposal area and vicinity.

5. Periphery of Waste Treatment and/or Disposal Facilities Observations

The requirements of this section only apply when the MRP specifies periphery standard observations. Standard observations shall include the following:

a. *Odor*: presence or absence, characterization, source, and distance of travel.

b. *Weather conditions*: wind direction and estimated velocity.

IV. STANDARD PROVISIONS – RECORDS

A. Records to be Maintained – This supplements IV.A of Standard Provisions (Attachment D)

The Discharger shall maintain records in a manner and at a location (e.g., wastewater treatment plant or Discharger offices) such that the records are accessible to Regional Water Board staff. The minimum period of retention specified in Section IV, Records, of the Federal Standard Provisions shall be extended during the course of any unresolved litigation regarding the subject discharge, or when requested by the Regional Water Board or Regional Administrator of USEPA, Region IX.

A copy of the permit shall be maintained at the discharge facility and be available at all times to operating personnel.

B. Records of monitoring information shall include – This supplements IV.B of Standard Provision (Attachment D)

1. Analytical Information

Records shall include analytical method detection limits, minimum levels, reporting levels, and related quantification parameters.

2. Flow Monitoring Data

For all required flow monitoring (e.g., influent and effluent flows), the additional records shall include the following, unless otherwise stipulated by the MRP:

a. Total volume for each day; and

b. Maximum, minimum, and average daily flows for each calendar month.

3. Wastewater Treatment Process Solids

a. For each treatment unit process that involves solids removal from the wastewater stream, records shall include the following:

1) Total volume or mass of solids removed from each collection unit (e.g., grit, skimmings, undigested biosolids, or combination) for each calendar month or other time period as appropriate, but not to exceed annually; and

2) Final disposition of such solids (e.g., landfill, other subsequent treatment unit).

b. For final dewatered biosolids from the treatment plant as a whole, records shall include the following:

1) Total volume or mass of dewatered biosolids for each calendar month;

2) Solids content of the dewatered biosolids; and

3) Final disposition of dewatered biosolids (disposal location and disposal method).

4. Disinfection Process

For the disinfection process, these additional records shall be maintained documenting process operation and performance:

a. For bacteriological analyses:

1) Wastewater flow rate at the time of sample collection; and

2) Required statistical parameters for cumulative bacterial values (e.g., moving median or geometric mean for the number of samples or sampling period identified in this Order).

b. For the chlorination process, when chlorine is used for disinfection, at least daily average values for the following:

1) Chlorine residual of treated wastewater as it enters the contact basin (mg/L);

2) Chlorine dosage (kg/day); and

3) Dechlorination chemical dosage (kg/day).

5. Treatment Process Bypasses

A chronological log of all treatment process bypasses, including wet weather blending, shall include the following:

a. Identification of the treatment process bypassed;

b. Dates and times of bypass beginning and end;

c. Total bypass duration;

d. Estimated total bypass volume; and

e. Description of, or reference to other reports describing, the bypass event, the cause, the corrective actions taken (except for wet weather blending that is in compliance with permit conditions), and any additional monitoring conducted.

6. Treatment Facility Overflows

This section applies to records for overflows at the treatment facility. This includes the headworks and all units and appurtenances downstream. The Discharger shall retain a chronological log of overflows at the treatment facility and records supporting the information provided in section V.E.2.

C. Claims of Confidentiality – Not Supplemented

V. STANDARD PROVISIONS – REPORTING

A. Duty to Provide Information – Not Supplemented

B. Signatory and Certification Requirements – Not Supplemented

C. Monitoring Reports – This section supplements V.C of Standard Provisions (Attachment D)

1. Self Monitoring Reports

For each reporting period established in the MRP, the Discharger shall submit an SMR to the Regional Water Board in accordance with the requirements listed in this document and at the frequency the MRP specifies. The purpose of the SMR is to document treatment performance, effluent quality, and compliance with the waste discharge requirements of this Order.

a. Transmittal letter

Each SMR shall be submitted with a transmittal letter. This letter shall include the following:

1) Identification of all violations of effluent limits or other waste discharge requirements found during the reporting period;

2) Details regarding violations: parameters, magnitude, test results, frequency, and dates;

3) Causes of violations;

4) Discussion of corrective actions taken or planned to resolve violations and prevent recurrences, and dates or time schedule of action implementation (if previous reports have been submitted that address corrective actions, reference to the earlier reports is satisfactory);

5) Data invalidation (Data should not be submitted in an SMR if it does not meet quality assurance/quality control standards. However, if the Discharger wishes to invalidate any measurement after it was submitted in an SMR, a letter shall identify the measurement suspected to be invalid and state the Discharger's intent to submit, within 60 days, a formal request to invalidate the measurement. This request shall include the original measurement in question, the reason for invalidating the measurement, all relevant documentation that supports invalidation [e.g., laboratory sheet, log entry, test results, etc.], and discussion of the corrective actions taken or planned [with a time schedule for completion] to prevent recurrence of the sampling or measurement problem.);

6) If the Discharger blends, the letter shall describe the duration of blending events and certify whether blended effluent was in compliance with the conditions for blending; and

7) Signature (The transmittal letter shall be signed according to Section V.B of this Order, Attachment D – Standard Provisions.).

b. Compliance evaluation summary

Each report shall include a compliance evaluation summary. This summary shall include each parameter for which the permit specifies effluent limits, the number of samples taken during the monitoring period, and the number of samples that exceed applicable effluent limits.

c. Results of analyses and observations

1) Tabulations of all required analyses and observations, including parameter, date, time, sample station, type of sample, test result, method detection limit, method minimum level, and method reporting level, if applicable, signed by the laboratory director or other responsible official.

2) When determining compliance with an average monthly effluent limitation and more than one sample result is available in a month, the Discharger shall compute the arithmetic mean unless the data set contains one or more reported determinations of detected but not quantified (DNQ) or nondetect (ND). In those cases, the Discharger shall compute the median in place of the arithmetic mean in accordance with the following procedure:

i. The data set shall be ranked from low to high, reported ND determinations lowest, DNQ determinations next, followed by quantified values (if any). The order of the individual ND or DNQ determinations is unimportant.

ii. The median value of the data set shall be determined. If the data set has an odd number of data points, then the median is the middle value. If the data set has an even number of data points, then the median is the average of the two values around the middle unless one or both of the points are ND or DNQ, in which case the median value shall be the lower of the two data points where DNQ is lower than a value and ND is lower than DNQ.

If a sample result, or the arithmetic mean or median of multiple sample results, is below the reporting limit, and there is evidence that the priority pollutant is present in the effluent above an effluent limitation and the Discharger conducts a Pollutant Minimization Program, the Discharger shall not be deemed out of compliance.

3) Dioxin-TEQ Reporting: The Discharger shall report for each dioxin and furan congener the analytical results of effluent monitoring, including the quantifiable limit (reporting level), the method detection limit, and the measured concentration. The Discharger shall report all measured values of individual congeners, including data qualifiers. When calculating dioxin-TEQ, the Discharger shall set congener concentrations below the minimum levels (ML) to zero. The Discharger shall calculate and report dioxin-TEQs using the following formula, where the MLs, toxicity equivalency factors (TEFs), and bioaccumulation equivalency factors (BEFs) are as provided in Table A:

$$\text{Dioxin-TEQ} = \Sigma\,(C_x \times TEF_x \times BEF_x)$$

where: C_x = measured or estimated concentration of congener x
TEF_x = toxicity equivalency factor for congener x
BEFx = bioaccumulation equivalency factor for congener x

Table A

Minimum Levels, Toxicity Equivalency Factors, and Bioaccumulation Equivalency Factors

Dioxin or Furan Congener	Minimum Level (pg/L)	1998 Toxicity Equivalency Factor (TEF)	Bioaccumulation Equivalency Factor (BEF)
2,3,7,8-TCDD	10	1.0	1.0
1,2,3,7,8-PeCDD	50	1.0	0.9
1,2,3,4,7,8-HxCDD	50	0.1	0.3
1,2,3,6,7,8-HxCDD	50	0.1	0.1
1,2,3,7,8,9-HxCDD	50	0.1	0.1
1,2,3,4,6,7,8-HpCDD	50	0.01	0.05
OCDD	100	0.0001	0.01
2,3,7,8-TCDF	10	0.1	0.8
1,2,3,7,8-PeCDF	50	0.05	0.2

2,3,4,7,8-PeCDF	50	0.5	1.6
1,2,3,4,7,8-HxCDF	50	0.1	0.08
1,2,3,6,7,8-HxCDF	50	0.1	0.2
1,2,3,7,8,9-HxCDF	50	0.1	0.6
2,3,4,6,7,8-HxCDF	50	0.1	0.7
1,2,3,4,6,7,8-HpCDF	50	0.01	0.01
1,2,3,4,7,8,9-HpCDF	50	0.01	0.4
OCDF	100	0.0001	0.02

d. Data reporting for results not yet available

The Discharger shall make all reasonable efforts to obtain analytical data for required parameter sampling in a timely manner. Certain analyses require additional time to complete analytical processes and report results. For cases where required monitoring parameters require additional time to complete analytical processes and reports, and results are not available in time to be included in the SMR for the subject monitoring period, the Discharger shall describe such circumstances in the SMR and include the data for these parameters and relevant discussions of any observed exceedances in the next SMR due after the results are available.

e. Flow data

The Discharger shall provide flow data tabulation pursuant to Section IV.B.2.

f. Annual self monitoring report requirements

By the date specified in the MRP, the Discharger shall submit an annual report to the Regional Water Board covering the previous calendar year. The report shall contain the following:

1) Annual compliance summary table of treatment plant performance, including documentation of any blending events;

2) Comprehensive discussion of treatment plant performance and compliance with the permit (This discussion shall include any corrective actions taken or planned, such as changes to facility equipment or operation practices that may be needed to achieve compliance, and any other actions taken or planned that are intended to improve performance and reliability of the Discharger's wastewater collection, treatment, or disposal practices.);

3) Both tabular and graphical summaries of the monitoring data for the previous year if parameters are monitored at a frequency of monthly or greater;

4) List of approved analyses, including the following:

(i) List of analyses for which the Discharger is certified;

(ii) List of analyses performed for the Discharger by a separate certified laboratory (copies of reports signed by the laboratory director of that laboratory shall not be submitted but be retained onsite); and

(iii) List of "waived" analyses, as approved;

8) Plan view drawing or map showing the Discharger's facility, flow routing, and sampling and observation station locations;

9) Results of annual facility inspection to verify that all elements of the SWPP Plan are accurate and up to date (only required if the Discharger does not route all storm water to the headworks of its wastewater treatment plant); and

10) Results of facility report reviews (The Discharger shall regularly review, revise, and update, as necessary, the O&M Manual, the Contingency Plan, the Spill Prevention Plan, and Wastewater Facilities Status Report so that these documents remain useful and relevant to current practices. At a minimum, reviews shall be conducted annually. The Discharger shall include, in each Annual Report, a description or summary of review and evaluation procedures, recommended or planned actions, and an estimated time schedule for implementing these actions. The Discharger shall complete changes to these documents to ensure they are up-to-date.).

g. Report submittal

The Discharger shall submit SMRs to:

California Regional Water Quality Control Board
San Francisco Bay Region
1515 Clay Street, Suite 1400
Oakland, CA 94612
Attn: NPDES Wastewater Division

h. Reporting data in electronic format

The Discharger has the option to submit all monitoring results in an electronic reporting format approved by the Executive Officer. If the Discharger chooses to submit SMRs electronically, the following shall apply:

1) *Reporting Method*: The Discharger shall submit SMRs electronically via a process approved by the Executive Officer (see, for example, the letter dated December 17, 1999, "Official Implementation of Electronic Reporting System [ERS]" and the progress report letter dated December 17, 2000).

2) *Monthly or Quarterly Reporting Requirements*: For each reporting period (monthly or quarterly as specified in the MRP), the Discharger shall submit an electronic SMR to the Regional Water Board in accordance with the provisions of Section V.C.1.a-e, except for requirements under Section V.C.1.c(1) where ERS does not have fields for dischargers to input certain information (e.g., sample time). However, until USEPA approves the electronic signature or other signature technologies, Dischargers that use ERS shall submit a hard copy of the original transmittal letter, an ERS printout of the data sheet, and a violation report (a receipt of the electronic transmittal shall be retained by the Discharger). This electronic SMR submittal suffices for the signed tabulations specified under Section V.C.1.c(1).

3) *Annual Reporting Requirements*: Dischargers who have submitted data using the ERS for at least one calendar year are exempt from submitting the portion of the annual report required under Section V.C.1.f(1) and (3).

D. Compliance Schedules – Not supplemented

E. Twenty-Four Hour Reporting – This section supplements V.E of Standard Provision (Attachment D)

1. Spill of Oil or Other Hazardous Material Reports

a. Within 24 hours of becoming aware of a spill of oil or other hazardous material that is not contained onsite and completely cleaned up, the Discharger shall report by telephone to the Regional Water Board at (510) 622-2369.

b. The Discharger shall also report such spills to the State Office of Emergency Services [telephone (800) 852-7550] only when the spills are in accordance with applicable reporting quantities for hazardous materials.

c. The Discharger shall submit a written report to the Regional Water Board within five working days following telephone notification unless directed otherwise by Regional Water Board staff. A report submitted electronically is acceptable. The written report shall include the following:

1) Date and time of spill, and duration if known;

2) Location of spill (street address or description of location);

3) Nature of material spilled;

4) Quantity of material involved;

5) Receiving water body affected, if any;

6) Cause of spill;

7) Estimated size of affected area;

8) Observed impacts to receiving waters (e.g., oil sheen, fish kill, water discoloration);

9) Corrective actions taken to contain, minimize, or clean up the spill;

10) Future corrective actions planned to be taken to prevent recurrence, and schedule of implementation; and

11) Persons or agencies notified.

2. Unauthorized Discharges from Municipal Wastewater Treatment Plants[1]

The following requirements apply to municipal wastewater treatment plants that experience an unauthorized discharge at their treatment facilities and are consistent with and supercede

[1] California Code of Regulations, Title 23, Section 2250(b), defines an unauthorized discharge to be a discharge, not regulated by waste discharge requirements, of treated, partially treated, or untreated wastewater resulting from the intentional or unintentional diversion of wastewater from a collection, treatment or disposal system.

requirements imposed on the Discharger by the Executive Officer by letter of May 1, 2008, issued pursuant to California Water Code Section 13383.

a. Two (2)-Hour Notification

For any unauthorized discharges that result in a discharge to a drainage channel or a surface water, the Discharger shall, as soon as possible, but not later than two (2) hours after becoming aware of the discharge, notify the State Office of Emergency Services (telephone 800-852-7550), the local health officers or directors of environmental health with jurisdiction over the affected water bodies, and the Regional Water Board. The notification to the Regional Water Board shall be via the Regional Water Board's online reporting system at www.wbers.net, and shall include the following:

1) Incident description and cause;

2) Location of threatened or involved waterway(s) or storm drains;

3) Date and time the unauthorized discharge started;

4) Estimated quantity and duration of the unauthorized discharge (to the extent known), and the estimated amount recovered;

5) Level of treatment prior to discharge (e.g., raw wastewater, primary treated, undisinfected secondary treated, and so on); and

6) Identity of the person reporting the unauthorized discharge.

b. 24-hour Certification

Within 24 hours, the Discharger shall certify to the Regional Water Board, at www.wbers.net, that the State Office of Emergency Services and the local health officers or directors of environmental health with jurisdiction over the affected water bodies have been notified of the unauthorized discharge.

c. 5-Day Written Report

Within five business days, the Discharger shall submit a written report, via the Regional Water Board's online reporting system at www.wbers.net, that includes, in addition to the information required above, the following:

1) Methods used to delineate the geographical extent of the unauthorized discharge within receiving waters;

2) Efforts implemented to minimize public exposure to the unauthorized discharge;

3) Visual observations of the impacts (if any) noted in the receiving waters (e.g., fish kill, discoloration of water) and the extent of sampling if conducted;

4) Corrective measures taken to minimize the impact of the unauthorized discharge;

5) Measures to be taken to minimize the chances of a similar unauthorized discharge occurring in the future;

6) Summary of Spill Prevention Plan or O&M Manual modifications to be made, if necessary, to minimize the chances of future unauthorized discharges; and

7) Quantity and duration of the unauthorized discharge, and the amount recovered.

d. Communication Protocol

To clarify the multiple levels of notification, certification, and reporting, the current communication requirements for unauthorized discharges from municipal wastewater treatment plants are summarized in Table B that follows.

Table B

Summary of Communication Requirements for Unauthorized Discharges[1] from Municipal Wastewater Treatment Plants

Discharger is required to:	Agency Receiving Information	Time frame	Method for Contact
1. Notify	California Emergency Management Agency (Cal EMA)	As soon as possible, but not later than **2 hours** after becoming aware of the unauthorized discharge.	Telephone – (800) 852-7550 (obtain a control number from Cal EMA)
	Local health department	As soon as possible, but not later than **2 hours** after becoming aware of the unauthorized discharge.	Depends on local health department
	Regional Water Board	As soon as possible, but not later than **2 hours** after becoming aware of the unauthorized discharge.	Electronic[2] www.wbers.net
2. Certify	Regional Water Board	As soon as possible, but not later than **24 hours** after becoming aware of the unauthorized discharge.	Electronic[3] www.wbers.net

[1] California Code of Regulations, Title 23, Section 2250(b), defines an unauthorized discharge to be a discharge, not regulated by waste discharge requirements, of treated, partially treated, or untreated wastewater resulting from the intentional or unintentional diversion of wastewater from a collection, treatment or disposal system.

[2] In the event that the Discharger is unable to provide online notification within 2 hours of becoming aware of an unauthorized discharge, it shall phone the Regional Water Board's spill hotline at (510) 622-2369 and convey the same information contained in the notification form. In addition, within 3 business days of becoming aware of the unauthorized discharge, the Discharger shall enter the notification information into the Regional Water Board's online system in electronic format.

[3] In most instances, the 2-hour notification will also satisfy 24-hour certification requirements. This is because the notification form includes fields for documenting that OES and the local health department have been contacted. In other words, if the Discharger is able to complete all the fields in the notification form within 2 hours, certification requirements are also satisfied. In the event that the Discharger is unable to provide online certification within 24 hours of becoming aware of an unauthorized discharge, it shall phone the Regional Water Board's spill hotline at (510) 622-2369 and convey the same information contained in the certification form. In addition, within 3 business days of becoming aware of the unauthorized discharge, the Discharger shall enter the certification information into the Regional Water Board's online system in electronic format.

3. Report	Regional Water Board	Within **5 business days** of becoming aware of the unauthorized discharge.	Electronic[4] www.wbers.net

F. Planned Changes – Not supplemented

G. Anticipated Noncompliance – Not supplemented

H. Other Noncompliance – Not supplemented

I. Other Information – Not supplemented

VI. STANDARD PROVISIONS – ENFORCEMENT – Not Supplemented

VII. ADDITIONAL PROVISIONS – NOTIFICATION LEVELS – Not Supplemented

VIII. DEFINITIONS – This section is an addition to Standard Provisions (Attachment D)

More definitions can be found in Attachment A of this NPDES Permit.

1. Arithmetic Calculations

a. Geometric mean is the antilog of the log mean or the back-transformed mean of the logarithmically transformed variables, which is equivalent to the multiplication of the antilogarithms. The geometric mean can be calculated with either of the following equations:

$$\text{Geometric Mean} = Anti\log\left(\frac{1}{N}\sum_{i=1}^{N} Log(C_i)\right)$$

or

$$\text{Geometric Mean} = (C_1 * C_2 * \ldots * C_N)^{1/N}$$

Where "N" is the number of data points for the period analyzed and "C" is the concentration for each of the "N" data points.

b. Mass emission rate is obtained from the following calculation for any calendar day:

$$\text{Mass emission rate (lb/day)} = \frac{8.345}{N}\sum_{i=1}^{N} Q_i C_i$$

$$\text{Mass emission rate (kg/day)} = \frac{3.785}{N}\sum_{i=1}^{N} Q_i C_i$$

[4] If the Discharger cannot satisfy the 5-day reporting requirements via the Regional Water Board's online reporting system, it shall submit a written report (preferably electronically in pdf) to the appropriate Regional Water Board case manager. In cases where the Discharger cannot satisfy the 5-day reporting requirements via the online reporting system, it must still complete the Regional Water Board's online reporting requirements within 15 calendar days of becoming aware of the unauthorized discharge.

In which "N" is the number of samples analyzed in any calendar day and "Q_i" and "C_i" are the flow rate (MGD) and the constituent concentration (mg/L) associated with each of the "N" grab samples that may be taken in any calendar day. If a composite sample is taken, "C_i" is the concentration measured in the composite sample and "Q_i" is the average flow rate occurring during the period over which the samples are composited. The daily concentration of a constituent measured over any calendar day shall be determined from the flow-weighted average of the same constituent in the combined waste streams as follows:

$$C_d = \text{Average daily concentration} = \frac{1}{Q_t}\sum_{i=1}^{N} Q_i C_i$$

In which "N" is the number of component waste streams and "Q" and "C" are the flow rate (MGD) and the constituent concentration (mg/L) associated with each of the "N" waste streams. "Q_t" is the total flow rate of the combined waste streams.

c. Maximum allowable mass emission rate, whether for a 24-hour, weekly 7-day, monthly 30-day, or 6-month period, is a limitation expressed as a daily rate determined with the formulas in the paragraph above, using the effluent concentration limit specified in the permit for the period and the specified allowable flow.

d. POTW removal efficiency is the ratio of pollutants removed by the treatment facilities to pollutants entering the treatment facilities (expressed as a percentage). The Discharger shall determine removal efficiencies using monthly averages (by calendar month unless otherwise specified) of pollutant concentration of influent and effluent samples collected at about the same time and using the following equation (or its equivalent):

Removal Efficiency (%) = 100 × [1-(Effluent Concentration/Influent Concentration)]

2. Biosolids means the solids, semi-liquid suspensions of solids, residues, screenings, grit, scum, and precipitates separated from or created in wastewater by the unit processes of a treatment system. It also includes, but is not limited to, all supernatant, filtrate, centrate, decantate, and thickener overflow and underflow in the solids handling parts of the wastewater treatment system.

3. Blending is the practice of recombining wastewater that has been biologically treated with wastewater that has bypassed around biological treatment units.

4. Bottom sediment sample is (1) a separate grab sample taken at each sampling station for the determination of selected physical-chemical parameters, or (2) four grab samples collected from different locations in the immediate vicinity of a sampling station while the boat is anchored and analyzed separately for macroinvertebrates.

5. Composite sample is a sample composed of individual grab samples collected manually or by an automatic sampling device on the basis of time or flow as specified in the MRP. For flow-based composites, the proportion of each grab sample included in the composite sample shall be within plus or minus five percent (+/-5%) of the representative flow rate of the waste stream being measured at the time of grab sample collection. Alternatively, equal volume grab samples may be individually analyzed with the flow-weighted average calculated by averaging flow-weighted ratios of each grab sample analytical result. Grab samples comprising time-based composite samples shall be collected at intervals not greater than those specified in the MRP. The quantity of each grab sample comprising a time-based composite sample shall be a set of flow proportional volumes as specified in the MRP. If a particular time-based or flow-based composite sampling protocol is not specified in the MRP, the Discharger shall determine and

implement the most representative sampling protocol for the given parameter subject to Executive Officer approval.

6. Depth-integrated sample is defined as a water or waste sample collected by allowing a sampling device to fill during a vertical traverse in the waste or receiving water body being sampled. The Discharger shall collect depth-integrated samples in such a manner that the collected sample will be representative of the waste or water body at that sampling point.

7. Flow sample is an accurate measurement of the average daily flow volume using a properly calibrated and maintained flow measuring device.

8. Grab sample is an individual sample collected in a short period of time not exceeding 15 minutes. Grab samples represent only the condition that exists at the time the wastewater is collected.

9. Initial dilution is the process that results in the rapid and irreversible turbulent mixing of wastewater with receiving water around the point of discharge.

10. Overflow is the intentional or unintentional spilling or forcing out of untreated or partially treated wastes from a transport system (e.g., through manholes, at pump stations, and at collection points) upstream from the treatment plant headworks or from any part of a treatment plant facility.

11. Priority pollutants are those constituents referred to in 40 CFR Part 122 as promulgated in the Federal Register, Vol. 65, No. 97, Thursday, May 18, 2000, also known as the California Toxics Rule, the presence or discharge of which could reasonably be expected to interfere with maintaining designated uses.

12. Storm water means storm water runoff, snow melt runoff, and surface runoff and drainage. It excludes infiltration and runoff from agricultural land.

13. Toxic pollutant means any pollutant listed as toxic under federal Clean Water Act section 307(a)(1) or under 40 CFR 401.15.

14. Untreated waste is raw wastewater.

15. Waste, waste discharge, discharge of waste, and discharge are used interchangeably in the permit. The requirements of the permit apply to the entire volume of water, and the material therein, that is disposed of to surface and ground waters of the State of California.

Table C

List of Monitoring Parameters and Analytical Methods

CTR No.	Pollutant/Parameter	Analytical Method[5]	Minimum Levels[6] (µg/l)											
			GC	GCMS	LC	Color	FAA	GFAA	ICP	ICP MS	SPGFAA	HYD RIDE	CVAA	DCP
1.	Antimony	204.2					10	5	50	0.5	5	0.5		1000
2.	Arsenic	206.3				20		2	10	2	2	1		1000
3.	Beryllium						20	0.5	2	0.5	1			1000
4.	Cadmium	200 or 213					10	0.5	10	0.25	0.5			1000
5a.	Chromium (III)	SM 3500												
5b.	Chromium (VI)	SM 3500				10	5							1000
	Chromium (total)[7]	SM 3500					50	2	10	0.5	1			1000
6.	Copper	200.9					25	5	10	0.5	2			1000
7.	Lead	200.9					20	5	5	0.5	2			10,000
8.	Mercury	1631 (note)[8]												
9.	Nickel	249.2					50	5	20	1	5			1000
10.	Selenium	200.8 or SM 3114B or C						5	10	2	5	1		1000
11.	Silver	272.2					10	1	10	0.25	2			1000
12.	Thallium	279.2					10	2	10	1	5			1000
13.	Zinc	200 or 289					20		20	1	10			
14.	Cyanide	SM 4500 CN^- C or I				5								
15.	Asbestos (only required for dischargers to MUN waters)[9]	0100.2 [10]												
16.	2,3,7,8-TCDD and 17 congeners (Dioxin)	1613												
17.	Acrolein	603	2.0	5										
18.	Acrylonitrile	603	2.0	2										
19.	Benzene	602	0.5	2										
33.	Ethylbenzene	602	0.5	2										
39.	Toluene	602	0.5	2										
20.	Bromoform	601	0.5	2										
21.	Carbon Tetrachloride	601	0.5	2										
22.	Chlorobenzene	601	0.5	2										
23.	Chlorodibromomethane	601	0.5	2										
24.	Chloroethane	601	0.5	2										

[5] The suggested method is the USEPA Method unless otherwise specified (SM = Standard Methods). The Discharger may use another USEPA-approved or recognized method if that method has a level of quantification below the applicable water quality objective. Where no method is suggested, the Discharger has the discretion to use any standard method.

[6] Minimum levels are from the *State Implementation Policy*. They are the concentration of the lowest calibration standard for that technique based on a survey of contract laboratories. Laboratory techniques are defined as follows: GC = Gas Chromatography; GCMS = Gas Chromatography/Mass Spectrometry; LC = High Pressure Liquid Chromatography; Color = Colorimetric; FAA = Flame Atomic Absorption; GFAA = Graphite Furnace Atomic Absorption; ICP = Inductively Coupled Plasma; ICPMS = Inductively Coupled Plasma/Mass Spectrometry; SPGFAA = Stabilized Platform Graphite Furnace Atomic Absorption (i.e., USEPA 200.9); Hydride = Gaseous Hydride Atomic Absorption; CVAA = Cold Vapor Atomic Absorption; DCP = Direct Current Plasma.

[7] Analysis for total chromium may be substituted for analysis of chromium (III) and chromium (VI) if the concentration measured is below the lowest hexavalent chromium criterion (11 ug/l).

[8] The Discharger shall use ultra-clean sampling (USEPA Method 1669) and ultra-clean analytical methods (USEPA Method 1631) for mercury monitoring. The minimum level for mercury is 2 ng/l (or 0.002 ug/l).

[9] MUN = Municipal and Domestic Supply. This designation, if applicable, is in the Findings of the permit.

[10] *Determination of Asbestos Structures over 10 [micrometers] in Length in Drinking Water Using MCE Filters*, USEPA 600/R-94-134, June 1994.

CTR No.	Pollutant/Parameter	Analytical Method[5]	Minimum Levels[6] (μg/l)											
			GC	GCMS	LC	Color	FAA	GFAA	ICP	ICP MS	SPGFAA	HYD RIDE	CVAA	DCP
25.	2-Chloroethylvinyl Ether	601	1	1										
26.	Chloroform	601	0.5	2										
75.	1,2-Dichlorobenzene	601	0.5	2										
76.	1,3-Dichlorobenzene	601	0.5	2										
77.	1,4-Dichlorobenzene	601	0.5	2										
27.	Dichlorobromomethane	601	0.5	2										
28.	1,1-Dichloroethane	601	0.5	1										
29.	1,2-Dichloroethane	601	0.5	2										
30.	1,1-Dichloroethylene or 1,1-Dichloroethene	601	0.5	2										
31.	1,2-Dichloropropane	601	0.5	1										
32.	1,3-Dichloropropylene or 1,3-Dichloropropene	601	0.5	2										
34.	Methyl Bromide or Bromomethane	601	1.0	2										
35.	Methyl Chloride or Chloromethane	601	0.5	2										
36.	Methylene Chloride or Dichlorormethane	601	0.5	2										
37.	1,1,2,2-Tetrachloroethane	601	0.5	1										
38.	Tetrachloroethylene	601	0.5	2										
40.	1,2-Trans-Dichloroethylene	601	0.5	1										
41.	1,1,1-Trichloroethane	601	0.5	2										
42.	1,1,2-Trichloroethane	601	0.5	2										
43.	Trichloroethene	601	0.5	2										
44.	Vinyl Chloride	601	0.5	2										
45.	2-Chlorophenol	604	2	5										
46.	2,4-Dichlorophenol	604	1	5										
47.	2,4-Dimethylphenol	604	1	2										
48.	2-Methyl-4,6-Dinitrophenol or Dinitro-2-methylphenol	604	10	5										
49.	2,4-Dinitrophenol	604	5	5										
50.	2-Nitrophenol	604		10										
51.	4-Nitrophenol	604	5	10										
52.	3-Methyl-4-Chlorophenol	604	5	1										
53.	Pentachlorophenol	604	1	5										
54.	Phenol	604	1	1		50								
55.	2,4,6-Trichlorophenol	604	10	10										
56.	Acenaphthene	610 HPLC	1	1	0.5									
57.	Acenaphthylene	610 HPLC		10	0.2									
58.	Anthracene	610 HPLC		10	2									
60.	Benzo(a)Anthracene or 1,2 Benzanthracene	610 HPLC	10	5										
61.	Benzo(a)Pyrene	610 HPLC		10	2									
62.	Benzo(b)Fluoranthene or 3,4 Benzofluoranthene	610 HPLC		10	10									
63.	Benzo(ghi)Perylene	610 HPLC		5	0.1									
64.	Benzo(k)Fluoranthene	610 HPLC		10	2									
74.	Dibenzo(a,h)Anthracene	610 HPLC		10	0.1									
86.	Fluoranthene	610 HPLC	10	1	0.05									
87.	Fluorene	610 HPLC		10	0.1									
92.	Indeno(1,2,3-cd) Pyrene	610 HPLC		10	0.05									
100.	Pyrene	610 HPLC		10	0.05									
68.	Bis(2-Ethylhexyl)Phthalate	606 or 625	10	5										

CTR No.	Pollutant/Parameter	Analytical Method[5]	Minimum Levels[6] (μg/l)											
			GC	GCMS	LC	Color	FAA	GFAA	ICP	ICP MS	SPGFAA	HYD RIDE	CVAA	DCP
70.	Butylbenzyl Phthalate	606 or 625	10	10										
79.	Diethyl Phthalate	606 or 625	10	2										
80.	Dimethyl Phthalate	606 or 625	10	2										
81.	Di-n-Butyl Phthalate	606 or 625		10										
84.	Di-n-Octyl Phthalate	606 or 625		10										
59.	Benzidine	625		5										
65.	Bis(2-Chloroethoxy)Methane	625		5										
66.	Bis(2-Chloroethyl)Ether	625	10	1										
67.	Bis(2-Chloroisopropyl)Ether	625	10	2										
69.	4-Bromophenyl Phenyl Ether	625	10	5										
71.	2-Chloronaphthalene	625		10										
72.	4-Chlorophenyl Phenyl Ether	625		5										
73.	Chrysene	625		10	5									
78.	3,3'-Dichlorobenzidine	625		5										
82.	2,4-Dinitrotoluene	625	10	5										
83.	2,6-Dinitrotoluene	625		5										
85.	1,2-Diphenylhydrazine (note)[11]	625		1										
88.	Hexachlorobenzene	625	5	1										
89.	Hexachlorobutadiene	625	5	1										
90.	Hexachlorocyclopentadiene	625	5	5										
91.	Hexachloroethane	625	5	1										
93.	Isophorone	625	10	1										
94.	Naphthalene	625	10	1	0.2									
95.	Nitrobenzene	625	10	1										
96.	N-Nitrosodimethylamine	625	10	5										
97.	N-Nitrosodi-n-Propylamine	625	10	5										
98.	N-Nitrosodiphenylamine	625	10	1										
99.	Phenanthrene	625		5	0.05									
101.	1,2,4-Trichlorobenzene	625	1	5										
102.	Aldrin	608	0.005											
103.	α-BHC	608	0.01											
104.	β-BHC	608	0.005											
105.	γ-BHC (Lindane)	608	0.02											
106.	δ-BHC	608	0.005											
107.	Chlordane	608	0.1											
108.	4,4'-DDT	608	0.01											
109.	4,4'-DDE	608	0.05											
110.	4,4'-DDD	608	0.05											
111.	Dieldrin	608	0.01											
112.	Endosulfan (alpha)	608	0.02											
113.	Endosulfan (beta)	608	0.01											
114.	Endosulfan Sulfate	608	0.05											
115.	Endrin	608	0.01											
116.	Endrin Aldehyde	608	0.01											
117.	Heptachlor	608	0.01											
118.	Heptachlor Epoxide	608	0.01											
119-	PCBs: Aroclors 1016, 1221,	608	0.5											

[11] Measurement for 1,2-Diphenylhydrazine may use azobenzene as a screen: if azobenzene is measured at >1 ug/l, then the Discharger shall analyze for 1,2-Diphenylhydrazine.

CTR No.	Pollutant/Parameter	Analytical Method[5]	Minimum Levels[6] (µg/l)											
			GC	GCMS	LC	Color	FAA	GFAA	ICP	ICP MS	SPGFAA	HYD RIDE	CVAA	DCP
125	1232, 1242, 1248, 1254, 1260													
126.	Toxaphene	608	0.5											

ATTACHMENT H – PRETREATMENT REQUIREMENTS

CALIFORNIA REGIONAL WATER QUALITY CONTROL
BOARD
SAN FRANCISCO BAY REGION

ATTACHMENT H
PRETREATMENT PROGRAM PROVISIONS

For

NPDES POTW WASTEWATER DISCHARGE PERMITS

March 2011

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Attachment H: Pretreatment Program Provisions

1. The Discharger shall be responsible and liable for the performance of all Control Authority pretreatment requirements contained in 40 CFR 403, including any regulatory revisions to Part 403. Where a Part 403 revision is promulgated after the effective date of the Discharger's permit and places mandatory actions upon the Discharger as Control Authority but does not specify a timetable for completion of the actions, the Discharger shall complete the required actions within six months from the issuance date of this permit or six months from the effective date of the Part 403 revisions, whichever comes later.

   (If the Discharger cannot complete the required actions within the above six-month period due to the need to process local adoption of sewer use ordinance modifications or other substantial pretreatment program modifications, the Discharger shall notify the Executive Officer in writing at least 60 days prior to the six-month deadline. The written notification shall include a summary of completed required actions, an explanation for why the six month deadline cannot be met, and a proposed timeframe to complete the rest of the required actions as soon as practical but not later than within twelve months of the issuance date of this permit or twelve months of the effective date of the Part 403 revisions, whichever comes later. The Executive Officer will notify the Discharger in writing within 30 days of receiving the request if the extension is not approved.)

   The United States Environmental Protection Agency (USEPA), the State and/or other appropriate parties may initiate enforcement action against a nondomestic user for noncompliance with applicable standards and requirements as provided in the Clean Water Act (Act).

2. The Discharger shall enforce the requirements promulgated under Sections 307(b), 307(c), 307(d) and 402(b) of the Act with timely, appropriate and effective enforcement actions. The Discharger shall cause nondomestic users subject to Federal Categorical Standards to achieve compliance no later than the date specified in those requirements or, in the case of a new nondomestic user, upon commencement of the discharge.

3. The Discharger shall perform the pretreatment functions as required in 40 CFR 403 and amendments or modifications thereto including, but not limited to:

   A) Implement the necessary legal authorities to fully implement the pretreatment regulations as provided in 40 CFR 403.8(f)(1);

   B) Implement the programmatic functions as provided in 40 CFR 403.8(f)(2);

   C) Publish an annual list of nondomestic users in significant noncompliance as provided per 40 CFR 403.8(f)(2)(viii);

   D) Provide for the requisite funding and personnel to implement the pretreatment program as provided in 40 CFR 403.8(f)(3); and

   E) Enforce the national pretreatment standards for prohibited discharges and categorical standards as provided in 40 CFR 403.5 and 403.6, respectively.

4. The Discharger shall submit annually a report to USEPA Region 9, the State Water Board and the Regional Water Board describing its pretreatment program activities over the previous calendar year. In the event that the Discharger is not in compliance with any conditions or requirements of the Pretreatment Program, the Discharger shall also include the reasons for noncompliance and a plan and schedule for achieving compliance. The report shall contain, but is not limited to, the information specified in Appendix H-1 entitled, "Requirements for Pretreatment Annual Reports." The annual report is due each year on February 28.

5. The Discharger shall submit a pretreatment semiannual report to USEPA Region 9, the State Water Board and the Regional Water Board describing the status of its significant industrial users (SIUs). The report shall contain, but is not limited to, information specified in Appendix H-2 entitled, "Requirements for Pretreatment Semiannual Reports." The semiannual report is due July 31 for the period January through June. The information for the period July through December of each year shall be included in the Annual Report identified in Appendix H-1. The Executive Officer may exempt the Discharger from the semiannual reporting requirements on a case by case basis subject to State Water Board and USEPA's comment and approval.

6. The Discharger shall conduct the monitoring of its treatment plant's influent, effluent, and sludge (biosolids) as described in Appendix H-4 entitled, "Requirements for Influent, Effluent and Sludge (Biosolids) Monitoring." (The term "biosolids," as used in this Attachment, shall have the same meaning as wastewater treatment plant "sludge" and will be used from this point forward.) The Discharger shall evaluate the results of the sampling and analysis during the preparation of the semiannual and annual reports to identify any trends. Signing the certification statement used to transmit the reports shall be deemed to certify the Discharger has completed this data evaluation. A tabulation of the data shall be included in the pretreatment annual report as specified in Appendix H-4. The Executive Officer may require more or less frequent monitoring on a case by case basis.

APPENDIX H-1

REQUIREMENTS FOR PRETREATMENT ANNUAL REPORTS

The Pretreatment Annual Report is due each year on February 28 and shall contain activities conducted during the previous calendar year. The purpose of the Annual Report is to:

- Describe the status of the Discharger's pretreatment program; and
- Report on the effectiveness of the program, as determined by comparing the results of the preceding year's program implementation.

The report shall contain, at a minimum, the following information:

1) **Cover Sheet**

The cover sheet shall include:

A) The name(s) and National Pollutant Discharge Elimination Discharge System (NPDES) permit number(s) of the Discharger(s) that is part of the Pretreatment Program;

B) The name, address and telephone number of a pretreatment contact person;

C) The period covered in the report;

D) A statement of truthfulness; and

E) The dated signature of a principal executive officer, ranking elected official, or other duly authorized employee who is responsible for overall operation of the Publicly Owned Treatment Works (POTW) (40 CFR 403.12(m)).

2) **Introduction**

This section shall include:

A) Any pertinent background information related to the Discharger and/or the nondomestic user base of the area;

B) List of applicable interagency agreements used to implement the Discharger's pretreatment program (e.g., Memoranda of Understanding (MOU) with satellite sanitary sewer collection systems); and

C) A status summary of the tasks required by a Pretreatment Compliance Inspection (PCI), Pretreatment Compliance Audit (PCA), Cleanup and Abatement Order (CAO), or other pretreatment-related enforcement actions required by the Regional Water Board or the USEPA. A more detailed discussion can be referenced and included in the section entitled, "Program Changes," if needed.

3) **Definitions**

This section shall include a list of key terms and their definitions that the Discharger uses to describe or characterize elements of its pretreatment program, or the Discharger may provide a reference to its website if the applicable definitions are available on-line.

4) **Discussion of Upset, Interference and Pass Through**

This section shall include a discussion of Upset, Interference or Pass Through incidents, if any, at the Discharger's treatment plant(s) that the Discharger knows of or suspects were caused by nondomestic user discharges. Each incident shall be described, at a minimum, consisting of the following information:

A) A description of what occurred;

B) A description of what was done to identify the source;

C) The name and address of the nondomestic user responsible;

D) The reason(s) why the incident occurred;

E) A description of the corrective actions taken; and

F) An examination of the local and federal discharge limits and requirements for the purposes of determining whether any additional limits or changes to existing requirements may be necessary to prevent other Upset, Interference or Pass Through incidents.

5) **Influent, Effluent and Biosolids Monitoring Results**

The Discharger shall evaluate the influent, effluent and biosolids monitoring results as specified in Appendix H-4 in preparation of this report. The Discharger shall retain the analytical laboratory reports with the Quality Assurance and Quality Control (QA/QC) data validation and make these reports available upon request.

This section shall include:

A) Description of the sampling procedures and an analysis of the results (see Appendix H-4 for specific requirements);
B) Tabular summary of the compounds detected (compounds measured above the detection limit for the analytical method used) for the monitoring data generated during the reporting year as specified in Appendix H-4;
C) Discussion of the investigation findings into any contributing sources of the compounds that exceed NPDES limits; and
D) Graphical representation of the influent and effluent metal monitoring data for the past five years with a discussion of any trends.

6) **Inspection, Sampling and Enforcement Programs**

This section shall include at a minimum the following information:

A) Inspections: Summary of the inspection program (e.g., criteria for determining the frequency of inspections and inspection procedures);

B) Sampling Events: Summary of the sampling program (e.g., criteria for determining the frequency of sampling and chain of custody procedures); and

C) Enforcement: Summary of Enforcement Response Plan (ERP) implementation including dates for adoption, last revision and submission to the Regional Water Board.

7) **Updated List of Regulated SIUs**

This section shall contain a list of all of the federal categories that apply to SIUs regulated by the Discharger. The specific categories shall be listed including the applicable 40 CFR subpart and section, and pretreatment standards (both maximum and average limits). Local limits developed by the Discharger shall be presented in a table including the applicability of the local limits to SIUs. If local limits do not apply uniformly to SIUs, specify the applicability in the tables listing the categorical industrial users (CIUs) and non-categorical SIUs. Tables developed in Sections 7A and 7B can be used to present or reference this information.

A) CIUs - Include a table that alphabetically lists the CIUs regulated by the Discharger as of the end of the reporting period. This list shall include:

i. Name;

ii. Address;

iii. Applicable federal category(ies);

iv. Reference to the location where the applicable Federal Categorical Standards are presented in the report;

v. Identify all deletions and additions keyed to the list submitted in the previous annual report. All deletions shall be briefly explained (e.g., closure, name change, ownership change, reclassification, declassification); and

vi. Information, calculations and data used to determine the limits for those CIUs for which a combined waste stream formula is applied.

B) Non-categorical SIUs - Include a table that alphabetically lists the SIUs not subject to any federal categorical standards that were regulated by the Discharger as of the end of the reporting period. This list shall include:

i. Name;

ii. Address;

iii. A brief description of the type of business;

iv. Identify all deletions and additions keyed to the list submitted in the previous annual report. All deletions shall be briefly explained (e.g., closure, name change, ownership change, reclassification, declassification); and

v. Indicate the applicable discharge limits (e.g., different from local limits) to which the SIUs are subject and reference to the location where the applicable limits (e.g., local discharge limits) are presented in the report.

8) **SIU (categorical and non-categorical) Compliance Activities**

The information required in this section may be combined in the table developed in Section 7 above.

A) **Inspection and Sampling Summary:** This section shall contain a summary of all the SIU inspections and sampling activities conducted by the Discharger and sampling activities conducted by the SIU over the reporting year to gather information and data regarding SIU compliance. The summary shall include:

i. The number of inspections and sampling events conducted for each SIU by the Discharger;

ii. The number of sampling events conducted by the SIU. Identify SIUs that are operating under an approved Total Toxic Organic Management Plan;

iii. The quarters in which the above activities were conducted; and

iv. The compliance status of each SIU, delineated by quarter, and characterized using all applicable descriptions as given below:

a. Consistent compliance;

b. Inconsistent compliance;

c. Significant noncompliance;

d. On a compliance schedule to achieve compliance (include the date final compliance is required);

e. Not in compliance and not on a compliance schedule; and

f. Compliance status unknown, and why not.

B) **Enforcement Summary:** This section shall contain a summary of SIU compliance and enforcement activities during the reporting year. The summary may be included in the summary table developed in section 8A and shall include the names and addresses of all SIUs affected by the actions identified below. For each notice specified in enforcement action "i" through "iv," indicate whether it was for an infraction of a federal or local standard/limit or requirement.

i. Warning letters or notices of violations regarding SIUs' apparent noncompliance with or violation of any federal pretreatment categorical standards and/or requirements, or local limits and/or requirements;

ii. Administrative Orders regarding the SIUs' apparent noncompliance with or violation of any federal pretreatment categorical standards and/or requirements, or local limits and/or requirements;

iii. Civil actions regarding the SIUs' apparent noncompliance with or violation of any federal pretreatment categorical standards and/or requirements, or local limits and/or requirements;

iv. Criminal actions regarding the SIUs' apparent noncompliance with or violation of any federal pretreatment categorical standards and/or requirements, or local limits and/or requirements;

v. Assessment of monetary penalties. Identify the amount of penalty in each case and reason for assessing the penalty;

vi. Order to restrict/suspend discharge to the Discharger; and

vii. Order to disconnect the discharge from entering the Discharger.

C) **July-December Semiannual Data:** For SIU violations/noncompliance during the semiannual reporting period from July 1 through December 31, provide the following information:

i. Name and facility address of the SIU;

ii. Indicate if the SIU is subject to Federal Categorical Standards; if so, specify the category including the subpart that applies;

iii. For SIUs subject to Federal Categorical Standards, indicate if the violation is of a categorical or local standard;

iv. Indicate the compliance status of the SIU for the two quarters of the reporting period; and

v. For violations/noncompliance identified in the reporting period, provide:

a. The date(s) of violation(s);

b. The parameters and corresponding concentrations exceeding the limits and the discharge limits for these parameters; and

c. A brief summary of the noncompliant event(s) and the steps that are being taken to achieve compliance.

9) **Baseline Monitoring Report Update**

This section shall provide a list of CIUs added to the pretreatment program since the last annual report. This list of new CIUs shall summarize the status of the respective Baseline Monitoring Reports (BMR). The BMR must contain the information specified in 40 CFR 403.12(b). For each new CIU, the summary shall indicate when the BMR was due; when the CIU was notified by the Discharger of this requirement; when the CIU submitted the report; and/or when the report is due.

10) **Pretreatment Program Changes**

This section shall contain a description of any significant changes in the Pretreatment Program during the past year including, but not limited to:

A) Legal authority;

B) Local limits;

C) Monitoring/ inspection program and frequency;

D) Enforcement protocol;

E) Program's administrative structure;

F) Staffing level;

G) Resource requirements;

H) Funding mechanism;

I) If the manager of the Discharger's pretreatment program changed, a revised organizational chart shall be included; and

J) If any element(s) of the program is in the process of being modified, this intention shall also be indicated.

11) **Pretreatment Program Budget**

This section shall present the budget spent on the Pretreatment Program. The budget, either by the calendar or fiscal year, shall show the total expenses required to implement the pretreatment program. A brief discussion of the source(s) of funding shall be provided. In addition, the Discharger shall make available upon request specific details on its pretreatment program expense amounts such as for personnel, equipment, and chemical analyses.

12) **Public Participation Summary**

This section shall include a copy of the public notice as required in 40 CFR 403.8(f)(2)(viii). If a notice was not published, the reason shall be stated.

13) **Biosolids Storage and Disposal Practice**

This section shall describe how treated biosolids are stored and ultimately disposed. If a biosolids storage area is used, it shall be described in detail including its location, containment features and biosolids handling procedures.

14) **Other Pollutant Reduction Activities**

This section shall include a brief description of any programs the Discharger implements to reduce pollutants from nondomestic users that are not classified as SIUs. If the Discharger submits any of this program information in an Annual Pollution Prevention Report, reference to this other report shall satisfy this reporting requirement.

15) **Other Subjects**

Other information related to the Pretreatment Program that does not fit into any of the above categories should be included in this section.

16) **Permit Compliance System (PCS) Data Entry Form**

The annual report shall include the PCS Data Entry Form. This form shall summarize the enforcement actions taken against SIUs in the past year. This form shall include the following information:

A) Discharger's name,

B) NPDES Permit number,

C) Period covered by the report,

D) Number of SIUs in significant noncompliance (SNC) that are on a pretreatment compliance schedule,

E) Number of notices of violation and administrative orders issued against SIUs,

F) Number of civil and criminal judicial actions against SIUs,

G) Number of SIUs that have been published as a result of being in SNC, and

H) Number of SIUs from which penalties have been collected.

APPENDIX H-2

REQUIREMENTS FOR JANUARY-JUNE PRETREATMENT SEMIANNUAL REPORT

The pretreatment semiannual report is due on July 31 for pretreatment program activities conducted from January through June unless an exception has been granted by the Regional Water Board's Executive Officer (e.g., pretreatment programs without any SIUs may qualify for an exception to the pretreatment semiannual report). Pretreatment activities conducted from July through December of each year shall be included in the Pretreatment Annual Report as specified in Appendix H-1. The pretreatment semiannual report shall contain, at a minimum the following information:

1) **Influent, Effluent and Biosolids Monitoring**

The influent, effluent and biosolids monitoring results shall be evaluated in preparation of this report. The Discharger shall retain analytical laboratory reports with the QA/QC data validation and make these reports available upon request. The Discharger shall also make available upon request a description of its influent, effluent and biosolids sampling procedures. Violations of any parameter that exceed NPDES limits shall be identified and reported. The contributing source(s) of the parameters that exceed NPDES limits shall be investigated and discussed.

2) **Significant Industrial User Compliance Status**

This section shall contain a list of all SIUs that were not in consistent compliance with all pretreatment standards/limits or requirements for the reporting period. For the reported SIUs, the compliance status for the previous semiannual reporting period shall be included. Once the SIU has determined to be out of compliance, the SIU shall be included in subsequent reports until consistent compliance has been achieved. A brief description detailing the actions that the SIU undertook to come back into compliance shall be provided.

For each SIU on the list, the following information shall be provided:

A) Name and facility address of the SIU;

B) Indicate if the SIU is subject to Federal Categorical Standards; if so, specify the category including the subpart that applies;

C) For SIUs subject to Federal Categorical Standards, indicate if the violation is of a categorical or local standard;

D) Indicate the compliance status of the SIU for the two quarters of the reporting period; and

E) For violations/noncompliance identified in the reporting period, provide:

i. The date(s) of violation(s);

ii. The parameters and corresponding concentrations exceeding the limits and the discharge limits for these parameters; and

iii. A brief summary of the noncompliant event(s) and the steps that are being taken to achieve compliance.

3) **Discharger's Compliance with Pretreatment Program Requirements**

This section shall contain a discussion of the Discharger's compliance status with the Pretreatment Program Requirements as indicated in the latest Pretreatment Compliance Audit (PCA) Report or Pretreatment Compliance Inspection (PCI) Report. It shall contain a summary of the following information:

A) Date of latest PCA or PCI report;

B) Date of the Discharger's response;

C) List of unresolved issues; and

D) Plan(s) and schedule for resolving the remaining issues.

APPENDIX H-3

SIGNATURE REQUIREMENTS FOR PRETREATMENT ANNUAL AND SEMIANNUAL REPORTS

The pretreatment annual and semiannual reports shall be signed by a principal executive officer, ranking elected official, or other duly authorized employee who is responsible for the overall operation of the Discharger [POTW - 40 CFR 403.12(m)]. Signed copies of the reports shall be submitted to the USEPA, the State Water Board, and the Regional Water Board at the following addresses unless the Discharger is instructed by any of these agencies to submit electronic copies of the required reports:

Pretreatment Program Reports
Clean Water Act Compliance Office (WTR-7)
Water Division
Pacific Southwest Region
U.S. Environmental Protection Agency
75 Hawthorne Street
San Francisco, CA 94105-3901

Submit electronic copies only to State and Regional Water Boards:
Pretreatment Program Manager
Regulatory Unit
State Water Resources Control Board
Division of Water Quality-15th Floor
1001 I Street
Sacramento, CA 95814
DMR@waterboards.ca.gov
NPDES_Wastewater@waterboards.ca.gov

Pretreatment Coordinator
NPDES Wastewater Division
SF Bay Regional Water Quality Control Board
1515 Clay Street, Suite 1400
Oakland, CA 94612
(Submit the report as a single Portable Document Format (PDF) file to the Pretreatment Coordinator's folder in the Regional Water Board's File Transfer Protocol (FTP) site. The instructions for using the FTP site can be found at the following internet address:
http://www.waterboards.ca.gov/sanfranciscobay/publications_forms/documents/FTP_Discharger_Guide-12-2010.pdf.)

APPENDIX H-4

REQUIREMENTS FOR INFLUENT, EFFLUENT AND BIOSOLIDS MONITORING

The Discharger shall conduct sampling of its treatment plant's influent, effluent and biosolids at the frequency shown in **the pretreatment requirements table** of the Monitoring and Reporting Program (MRP, Attachment E). When sampling periods coincide, one set of test results, reported separately, may be used for those parameters that are required to be monitored by both the influent and effluent monitoring requirements of the MRP and the Pretreatment Program. The Pretreatment Program monitoring reports as required in Appendices H-1 and H-2 shall be transmitted to the Pretreatment Program Coordinator.

1. Reduction of Monitoring Frequency

The minimum frequency of Pretreatment Program influent, effluent, and biosolids monitoring shall be dependant on the number of SIUs identified in the Discharger's Pretreatment Program as indicated in Table H-1.

Table H-1: Minimum Frequency of Pretreatment Program Monitoring

Number of SIUs	Minimum Frequency
< 5	Once every five years
> 5 and < 50	Once every year
> 50	Twice per year

If the Discharger's required monitoring frequency is greater than the minimum specified in Table H-1, the Discharger may request a reduced monitoring frequency for that constituent(s) as part of its application for permit reissuance if it meets the following criteria:

The monitoring data for the constituent(s) consistently show non-detect (ND) levels for the effluent monitoring and very low (i.e., near ND) levels for influent and biosolids monitoring for a minimum of eight previous years' worth of data.

The Discharger's request shall include tabular summaries of the data and a description of the trends in the industrial, commercial, and residential customers in the Discharger's service area that demonstrate control over the sources of the constituent(s). The Regional Water Board may grant a reduced monitoring frequency in the reissued permit after considering the information provided by the Discharger and any other relevant information.

2. Influent and Effluent Monitoring

The Discharger shall monitor for the parameters using the required sampling and test methods listed in **the pretreatment table** of the MRP. Any test method substitutions must have received prior written Executive Officer approval. Influent and effluent sampling locations shall be the same as those sites specified in the MRP.

The influent and effluent samples should be taken at staggered times to account for treatment plant detention time. Appropriately staggered sampling is considered consistent with the requirement for collection of effluent samples coincident with influent samples in Section III.A.3.a(2) of Attachment D. All samples must be representative of daily operations. Sampling and analysis shall be performed in accordance with the techniques prescribed in 40 CFR 136 and amendments thereto. For effluent monitoring, the reporting limits for the individual parameters shall be at or below the minimum levels (MLs) as stated in the Policy for Implementation of Toxics Standards for Inland Surface Waters, Enclosed Bays, and Estuaries of California (2000) [also known as the State Implementation Policy (SIP)]; any revisions to the MLs shall be adhered to. If a parameter does not have a stated ML, then the Discharger shall conduct the analysis using the lowest commercially available and reasonably achievable detection levels.

The following report elements should be used to submit the influent and effluent monitoring results. A similarly structured format may be used but will be subject to Regional Water Board approval. The monitoring reports shall be submitted with the Pretreatment Annual Report identified in Appendix H-1.

A) Sampling Procedures, Sample Dechlorination, Sample Compositing, and Data Validation (applicable quality assurance/quality control) shall be performed in accordance with the techniques prescribed in 40 CFR 136 and amendments thereto. The Discharger shall make available upon request its sampling procedures including methods of dechlorination, compositing, and data validation.

B) A tabulation of the test results for the detected parameters shall be provided.

C) Discussion of Results – The report shall include a complete discussion of the test results for the detected parameters. If any pollutants are detected in sufficient concentration to upset, interfere or pass through plant operations, the type of pollutant(s) and potential source(s) shall be noted, along with a plan of action to control, eliminate, and/or monitor the pollutant(s). Any apparent generation and/or destruction of pollutants attributable to chlorination/dechlorination sampling and analysis practices shall be noted.

3. Biosolids Monitoring

Biosolids should be sampled in a manner that will be representative of the biosolids generated from the influent and effluent monitoring events except as noted in (C) below. The same parameters required for influent and effluent analysis shall be included in the biosolids analysis. The biosolids analyzed shall be a composite sample of the biosolids for final disposal consisting of:

A) Biosolids lagoons – 20 grab samples collected at representative equidistant intervals (grid pattern) and composited as a single grab, or

B) Dried stockpile – 20 grab samples collected at various representative locations and depths and composited as a single grab, or

C) Dewatered biosolids - daily composite of 4 representative grab samples each day for 5 days taken at equal intervals during the daily operating shift taken from a) the dewatering units or b) each truckload, and shall be combined into a single 5- day composite.

The USEPA manual, POTW Sludge Sampling and Analysis Guidance Document, August 1989, containing detailed sampling protocols specific to biosolids is recommended as a guidance for sampling procedures. The USEPA manual Analytical Methods of the National Sewage Sludge Survey, September 1990, containing detailed analytical protocols specific to biosolids, is recommended as a guidance for analytical methods.

In determining if the biosolids are a hazardous waste, the Discharger shall adhere to Article 2, "Criteria for Identifying the Characteristics of Hazardous Waste," and Article 3, "Characteristics of Hazardous Waste," of Title 22, California Code of Regulations, sections 66261.10 to 66261.24 and all amendments thereto.

The following report elements should be used to submit the biosolids monitoring results. A similarly structured form may be used but will be subject to Regional Water Board approval. The results shall be submitted with the Pretreatment Annual Report identified in Appendix H-1.

- Sampling Procedures and Data Validation (applicable quality assurance/quality control) shall be performed in accordance with the techniques prescribed in 40 CFR 136 and amendments thereto. The Discharger shall make available upon request its biosolids sampling procedures and data validation methods.

- Test Results – Tabulate the test results for the detected parameters and include the percent solids.

- Discussion of Results – Include a complete discussion of test results for the detected parameters. If the detected pollutant(s) is reasonably deemed to have an adverse effect on biosolids disposal, a plan of action to control, eliminate, and/or monitor the pollutant(s) and the known or potential source(s) shall be included. Any apparent generation and/or destruction of pollutants attributable to chlorination/dechlorination sampling and analysis practices shall be noted.

The Discharger shall also provide a summary table presenting any influent, effluent or biosolids monitoring data for non-priority pollutants that the Discharger believes may be causing or contributing to interference, pass through or adversely impacting biosolids quality.

EXHIBIT H

Numeric Nutrient Endpoint Development for San Francisco Bay Estuary: Literature Review and Data Gaps Analysis

Lester McKee
Martha Sutula
Alicia Gilbreath
Julie Beagle
David Gluchowski
Jennifer Hunt

Southern California Coastal Water Research Project

Technical Report 644 - June 2011

EXHIBIT H

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June 30, 2011

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Executive Summary

The California State Water Resources Control Board (SWRCB) is developing nutrient water quality objectives for the State's surface waters, using an approach known as the Nutrient Numeric Endpoint (NNE) framework. The NNE establishes a suite of numeric endpoints based on the ecological response of an aquatic waterbody to nutrient over-enrichment (eutrophication, e.g. algal biomass, dissolved oxygen). In addition to numeric endpoints for response indicators, the NNE framework must include models that link the response indicators to nutrient loads and other management controls. The NNE framework is intended to serve as numeric *guidanc*e to translate *narrative* water quality objectives. The NNE framework is currently under development for estuaries. Because San Francisco Bay represents California's largest estuary (70% by area of estuarine habitat statewide), it merits development of an estuary-specific NNE framework. The purpose of this document is to review literature and data relevant to the assessment of eutrophication in San Francisco Bay, with the goal of providing information to formulate a work plan to develop NNEs for this estuary. The review had three objectives: 1) Evaluate indicators to assess eutrophication and other adverse effects of anthropogenic nutrient loading in San Francisco Bay, 2) Summarize existing literature in SF Bay using indicators and identify data gaps, and 3) Investigate what data and tools exist to evaluate the trends in nutrient loading to the Bay.

Recommended NNE Indicators for SF Bay

The NNE assessment framework is the structured set of decision rules that helps to classify the waterbody in categories from minimally to very disturbed, in order to determine if a waterbody is meeting beneficial uses, or to establish TMDL numeric targets. Development of an assessment framework begins by choosing response indicators, which were reviewed using four criteria: 1) strong linkage to beneficial uses, 2) well -vetted means of measurement, 3) can model the relationship between the indicator, nutrient loads and other management controls, and 4) has an acceptable signal: noise ratio to assess eutrophication. Indicators varied among four habitat types: 1) unvegetated subtidal, 2) seagrass and brackish SAV, 3) intertidal flats, and 4) tidally muted habitats (e.g. estuarine diked Baylands). Two types of indicators were designated. Primary indicators are those which met all evaluation criteria and would therefore be expected to be a primary line of evidence of the NNE assessment framework for SF Bay. Supporting indicators fell short of meeting evaluation criteria, but may be used as supporting lines of evidence. This terminology is used in order to provide a sense of level of confidence in how the indicators should be employed in a multiple lines of evidence context.

The review found four types of indicators met all evaluation criteria and are designated as primary: dissolved oxygen, phytoplankton biomass, productivity, and assemblage, and cyanobacterial abundance and toxin concentration (all subtidal habitats), macroalgal biomass and cover (intertidal habitat, tidally muted habitats, and seagrass habitats). Other indicators evaluated met three or fewer of the review criteria and designated as supporting indicators: HAB cell counts and toxin concentration, urea and ammonium (all subtidal), light attenuation and epiphyte load (seagrass/brackish SAV). Ultimately, the real distinction between “primary” and “supporting” and how these classes of indicators would be used as multiple lines of evidence in an NNE assessment is entirely dependent on indicator group and particular applications to specific habitat types. Some primary indicators (e.g. dissolved oxygen) could be stand-alone, while for others such as phytoplankton biomass, productivity and assemblage, the SF Bay Technical Advisory Team strongly recommends using them as multiple lines of evidence, as use of any one alone is likely to be insufficiently robust.

The use of ammonium as an indicator received review, due to its hypothesized role in limiting phytoplankton primary production via nitrate uptake inhibition in Suisun Bay and the lower Sacramento River. The SF Bay Technical Advisory Team chose to include it as a supporting indicator because the importance of ammonium inhibition of diatom blooms relative to other factors controlling primary productivity Bay wide is not well understood. Additional review and synthesis are recommended, pending currently funded studies, to identify potential ammonium thresholds.

To What Extent is SF Bay Demonstrating Symptoms of Eutrophication, Utilizing NNE Indicators?

Of the four habitat types, only unvegetated subtidal habitat had adequate data to make an assessment of eutrophication. Dissolved oxygen in SF Bay subtidal habitat is much higher and phytoplankton biomass and productivity is lower than would be expected in an estuary with such high nutrient enrichment, implying that eutrophication is controlled by processes other than a simple nutrient-limitation of primary production. However, all regions of the SF Bay have experienced significant increases in phytoplankton biomass since the late 1990's. Recent analysis of water quality data collected by USGS from 1978 to 2009 show a significant increase in water column chlorophyll a (30-50% per decade from Suisun to South Bay respectively) and a significant decline in DO concentrations (1.6 to 2.5% per decade in South Bay and Suisun Bay respectively). Thus evidence is building that the historic resilience of SF Bay to the harmful effects of nutrient enrichment is weakening. The causes for the Bay wide trends include changes in water clarity due to less suspended sediment, lower metal inhibition due to improvements in wastewater treatment, increased seeding from ocean populations, declines in consumption by bivalves due to increases in predation by juvenile English sole and speckled sanddabs, and declines in phytoplankton consumption by consumers due to recent new invasive species introductions. Data suggest that primary productivity in Suisun Bay is limited by strong grazing pressure by invasive clams, light limitation by high turbidity, and ammonium inhibition of diatom uptake of nitrate. Few harmful algal blooms (HABs) have been reported recently in SF Bay. However, there have been historical occurrences, and recently cyanobacteria and dinoflagellate blooms have been increasingly documented.

What Are the Nutrient Loads to SF Bay From Various Sources?

Nutrients loads to SF Bay from external sources are poorly understood, though data exist with which to improve published load estimates from some sources. For the most part, published load estimates are outdated by one or even two decades or based on data that were not collected for loads estimation.

Data Gaps and Recommended Next Steps

The SF Bay NNE framework consists of two principle components: 1) primary and supporting indicators used in an assessment framework to assess eutrophication of SF Bay habitats and 2) models that link these indicators back to nutrient loads and other management controls on eutrophication. There are five major recommendations: 1) develop an NNE assessment framework for SF Bay, 2) quantify external nutrients loads, 3) develop a suite of models that link NNE response indicators to nutrient loads and other co-factors, 4) implement a monitoring program to support the use of the NNE in SF Bay to manage nutrients, and 5) Coordinate development of the SF Bay NNE workplan with nutrient management activities in Sacramento and San Joaquin Delta. **The SF Bay Technical Advisory Team assumed the San Francisco Bay Water Board will prioritize these next steps, with review/feedback from its advisory groups.**

Develop an NNE assessment framework for SF Bay

Development of an NNE assessment framework for SF Bay involves specifying how primary and supporting indicators would be used as multiple lines of evidence to diagnose adverse effects of eutrophication. The table below summarizes data gaps and recommended next steps for development of an SF Bay NNE assessment framework by habitat type. Data gaps and recommendations generally fall into four categories: 1) Monitoring to assess baseline levels of indicators of interest where data are currently lacking, 2) Analysis of existing data, 3) Field studies or experiments to collect data required for endpoint development, and 4) Formation of expert workgroups to recommend approach to assessment framework development and synthesize information to be used in setting numeric endpoints.

Type	Indicator	Designation	Data Gaps	Recommended Next Steps
Subtidal Habitat	Dissolved oxygen	Primary	Wealth of data exists. Technical Advisory Team does not have expertise to review adequacy of DO objectives. Review did not address dissolved oxygen data in the tidally muted habitats of SF Bay.	Consider update of science supporting Basin Plan dissolved oxygen objectives, if warranted by additional review by fisheries experts. Review could be for entire Bay or limited to the tidally muted areas of the Bay.
	Phytoplankton biomass , productivity, and assemblage	Primary	Need a review of science supporting selection of endpoints. Improved prediction of factors controlling assemblage	Recommend development of a white paper and a series of expert workshops to develop NNE assessment framework for phytoplankton biomass, productivity, taxonomic composition/assemblages,

Type	Indicator	Designation	Data Gaps	Recommended Next Steps
	HAB species abundance and toxin conc.	Cyanobacteria = primary; Other HAB =supporting	Little data on HAB toxin concentrations in surface waters and faunal tissues.	abundance and/or harmful algal bloom toxin concentrations. Recommend augmentation of current monitoring to include measurement of HAB toxin concentrations in water and faunal tissues.
Subtidal Habitat (Continued)	Ammonium and urea	Supporting	Lack of understanding of importance of ammonia limitation of nitrate uptake in diatoms on Bay productivity vis-à-vis other factors. Lack of data on urea in SF Bay	Recommend formulation of a working group of SF Bay scientists to synthesize available data on factors known to control primary productivity in different regions in the Bay, and evaluate potential ammonium endpoints. Recommend collecting additional data on urea concentrations in SF Bay via USGS's water quality sampling over a two year period.
	Macrobenthos taxonomy, abundance and biomass	Co-factor	Lack of information on how to use combination of taxonomy, abundance, and biomass to assess eutrophication	Recommend utilization of IE-EMP dataset to explore use of macrobenthos to be used reliably to diagnose eutrophication distinctly from other stressors in oligohaline habitats. This may involve including biomass in the protocol to improve ability to diagnose eutrophication.
Seagrass Habitat	Phytoplankton biomass, epiphyte load and light attenuation	Phytoplankton biomass = primary, epiphyte load and light attenuation = secondary	Poor data availability of data on stressors to SF Bay seagrass beds. Studies needed to establish light requirements for seagrass and to assess effects of light attenuation	Recommend 1) Continued monitoring of aerial extent of seagrass every 3-5 years (currently no further system scale monitoring is planned beyond 2010), 2) studies to establish light requirements for SF Bay seagrass species, 3) development of a statewide workgroup to develop an assessment framework for seagrass based on phytoplankton biomass, macroalgae, and epiphyte load and 4) collection of baseline data to characterize prevalence of macroalgal blooms on seagrass beds. Studies characterizing thresholds of adverse effects of macroalgae on seagrass currently underway in other California estuaries should be evaluated for their applicability to SF Bay.
	Macroalgae biomass and cover	Primary	Data gaps include studies to establish thresholds of macroalgal biomass, cover and duration that adversely affect seagrass habitat	
Intertidal Flat Habitat	Macroalgal biomass and cover	Primary	Lack of baseline data on frequency, magnitude (biomass and cover) and duration of macroalgal blooms in these intertidal flats	Recommend collection of baseline data on macroalgae, microphytobenthos and sediment bulk characteristics. Recommend inclusion of SF Bay scientists and stakeholders on statewide workgroup to develop an assessment framework for macroalgae on intertidal flats.
	Sediment nutrients	Supporting		
	MPB taxonomy and biomass	Supporting		

Type	Indicator	Designation	Data Gaps	Recommended Next Steps
Muted Subtidal Habitat	Macroalgae	Primary	Lack of baseline data on biomass and cover in muted habitat types	Recommend collection of baseline data on macroalgae, dissolved oxygen, phytoplankton biomass, taxonomic composition and HAB species/toxin concentration in these habitat types. Recommendation to develop an assessment framework based on macroalgae, phytoplankton and dissolved oxygen in these habitat types. One component of this discussion should be a decision on beneficial uses that would be targeted for protection and to what extent the level of protection or expectation for this habitat type differ from adjacent subtidal habitat.
	Phytoplankton biomass, assemblage, HAB toxin conc.	Phytoplankton biomass, cyanobacteria = primary; assemblage and other HABs= supporting	Lack of baseline data on biomass and community composition, HAB toxin concentrations	
	Dissolved oxygen	Primary	Some data on dissolved oxygen exist. Unclear what levels of DO required to protect muted habitat beneficial uses	

Quantify Nutrient Loads

The table below provides a summary of data gaps and recommended next steps. Recommendations generally fall into two categories: 1) Revising and updating estimates of nutrients from the different sources, based on existing data and 2) Identification of data needed to develop a dynamic loading model.

Source	**Data Gaps Identified**	**Recommended Next Steps**
Atmospheric Deposition	No recently published data on wet & dry atmospheric deposition	Loads likely relatively small. Literature review to determine range of N and P deposition rates for West Coast coastal urban areas. Recommend baseline atmospheric deposition monitoring of wet and dry N and P deposition over 1-2 year period to better constrain estimates.
Terrestrial Loads from Delta	Dry weather concentrations available through RMP. No data available on wet weather concentrations	Loads likely large. Recommend analysis of existing RMP data to estimate dry weather nutrient loads. Initiate wet weather data collection of nutrients at the Mallard Island DWR sampling location (head of Suisun Bay) to support improved daily loads estimates for 1995-present.
Municipal Effluent	Data available through 15 of approx. 40 Publicly Owned Treatment Works	Loads likely large. Synthesize nutrient discharge and concentration data to estimate loads over period of last 10-20 years. Encourage all treatment plants that discharge to the Bay to begin analyzing effluent for total and dissolved inorganic nutrients and to submit these data to the SFRWQCB on a regular basis. Recommend that the POTWs conduct a laboratory inter-comparison on nutrient methods to assure comparability of estimates.
Industrial Effluent	Some data available from the 1990s	Loads likely small relative to municipal wastewater. Synthesize available data to provide information for prioritization of any future steps.
Stormwater	Lack of wet weather data sufficient to develop a dynamic loading model	Loads likely large. Synthesize data to provide an updated estimate of stormwater contributions to assist prioritization of next steps. Scope the data needs associated with the development of a dynamic loading model.
Groundwater	Data available from 79 USGS monitoring stations. Flow data not well understood	Loads likely small. Refine current loads estimates after review by local USGS groundwater experts in order to support prioritization of next steps if any.
Exchange with Coastal Ocean	Some data available for fluxes of water and sediments during selected tides and seasons	Initiate a workgroup of local experts to design a sampling program for nutrient flux at the Golden Gate boundary. The intent with this program would be to develop models that simulate flux at the ocean-bay interface.

Develop Load-Response Models

An important component of implementing the NNE framework in SF Bay is the development of load-response models that can simulate the ecological response of the Estuary to nutrients and other important co-factors. Several types of models need to be developed, fitting into two general categories: 1) Air, oceanic and watershed loading model(s), which estimate the amount of nutrients and sediment reaching the SF Bay estuary and where they originate, and 2) an Estuary water quality model, which simulates the ecosystem response to nutrient loads and other management controls. Sufficient data and knowledge of SF Bay must exist to support the development of system wide dynamic simulation models to predict phytoplankton biomass/community response and relationships to models of secondary productivity. This is not likely in the short term, so it is important to consider that the development of a more complex model should follow the testing out of key concepts and assumptions in smaller, simpler models.

Scoping the development of these NNE load response models should begin through use of empirical data and studies to develop coarse nutrient budgets for SF Bay. Existing data that describe the timing and magnitude of external sources, internal sources, sinks, and pathways of transformation such as benthic nutrient flux, nitrification, denitrification, etc. would be compiled in order to synthesize current understanding of sources and fate of nutrients as well as identify critical data gaps in advance of the modeling strategy development.

Second, a review of existing models and their applications should be undertaken, with the intent of understanding what existing tools may be used to leverage efforts.

During this strategy workshop, participants would describe the modeling objectives, determine whether existing tools can be used in this effort, identify key data gaps and studies, and identify additional work elements needed to begin this major work element. The product of this effort would be the identification of the appropriate models, a phased workplan, timeline and budget to develop these models, and identification of and coordination among key institutions, programs and stakeholders. This information could be synthesized into a workplan to develop the loading and estuary water quality models and a preliminary timeline and budget for Phase I of the effort.

Conduct a Monitoring Program to Develop and Implement the NNE Framework in SF Bay

The development and use of an NNE framework for San Francisco Bay is completely contingent on the continued availability of monitoring data to formulate, test and periodically assess the status of the Bay with respect to eutrophication. Over the past forty years, the USGS has conducted a research program in the subtidal habitat of SF Bay, with partial support by the SF Bay Regional Monitoring Program (RMP) since 1993. This USGS research program cannot be considered replacement for a regularly funded monitoring program. The SF Bay Technical Advisory Team strongly recommends that a nutrients/eutrophication monitoring strategy be developed and funded for successful development and implementation of the NNE in SF Bay.

Coordinate Development of the SF Bay NNE Framework with Nutrient Management in the Delta

Development and implementation of a NNE framework for SF Bay will require improve coordination with nutrient management activities in the San Joaquin and Sacramento River Delta. Preliminary discussions on this topic have just begun with the Central Valley Water Board staff. Other entities, for example, the Interagency Ecological Program should be engaged. Coordination should be improved, at minimum, with respect to any future monitoring and/or modeling of nutrient loading, transport and source identification, as SF Bay and the Delta exchange nutrients across their aquatic and terrestrial boundaries. Coordination would be further enhanced by a similar review of NNE candidate indicators, summary of existing science, and identification of data gaps and recommended next steps specifically for the Delta.

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1. Introduction

1.1 Background

Cultural eutrophication of estuaries and coastal waters is a global environmental issue, with demonstrated links between anthropogenic changes in watersheds, increased nutrient loading to coastal waters, harmful algal blooms (HABs), hypoxia, and impacts on aquatic food webs (Valiela, Foreman et al.,1992; Kamer and Stein, 2003). These ecological impacts of eutrophication of coastal areas can have far-reaching consequences, including fish-kills and lowered fishery production (Glasgow and Burkholder, 2000), loss or degradation of seagrass and kelp beds (Twilley, 1985; Burkholder, Noga et al., 1992; McGlathery, 2001), smothering of bivalves and other benthic organisms (Rabalais and Harper, 1992), nuisance odors, and impacts on human and marine mammal health from increased frequency and extent of HABs and poor water quality (Bates et al., 1989; Bates, DeFreitas et al., 1991; Trainer, Hickey et al., 2002). These modifications have significant economic and social costs, some of which can be readily identified and valued, while others are more difficult to assess (Turner, Qureshi et al., 1998). According to United States Environmental Protection Agency (USEPA), eutrophication is one of the top three leading causes of impairments of the nation's waters (USEPA, 2001).

In California, the impacts of nutrient loading on estuaries and coastal waters have not been well monitored (Bricker, Clement et al., 1999), with the notable exception of San Francisco (SF) Bay where there has been research and ongoing publication by a number of authors (Cloern, 1982; Cloern, Cole et al., 1985; Cloern, 1991; Cloern, 1996; Cloern, 1999). Without management actions to reduce anthropogenic nutrient loads and other factors controlling eutrophication, symptoms are expected to develop or worsen in the majority of systems, in part due to projected population increases along the coastal areas. Scientifically-based statewide water quality objectives and tools that relate these objectives to management controls are needed to prevent eutrophication from occurring and to provide targets for restoration or mitigation of systems where adverse effects of eutrophication have already occurred.

The USEPA initiated the National Nutrient Management Strategy in 1998 to begin addressing the pervasive impacts of excessive nutrient loading to both fresh and marine waters (Wayland, 1998). A primary goal of the strategy was to develop numeric nutrient criteria to measure the progress of the management strategy. The USEPA issued a series of technical guidance manuals for the development of nutrient criteria. Initial national guidance on nutrient criteria development advocated the use of a statistical approach to establish thresholds based on the nutrient concentrations in surface waters (USEPA, 1998). In this approach, reference conditions were based on 25th percentiles of all nutrient concentration data including a comparison of reference condition for the aggregate ecoregion versus the subecoregions. These 25th percentile concentrations were characterized as criteria recommendations that could be used to protect waters against nutrient over-enrichment. The "Nutrient Criteria Technical guidance Manual: Estuarine and Coastal Waters" was released by USEPA in 2001.

Several studies have demonstrated the shortcomings of using ambient nutrient concentration criteria alone to predict eutrophication, in streams (Welch, Horner et al., 1989; Fevold, 1998; Chetelat, Pick et al., 1999; Heiskary and Markus, 2001; Dodds, Smith et al., 2002) and estuaries (Cloern, 2001; Dettman, Kohn et al., 2001; Kennison, Kamer et al., 2003). Use of ambient, surface water nutrient concentrations is generally not effective for assessing eutrophication and the subsequent impact on beneficial use because ambient concentrations reflect the biological processing that has already occurred. In addition, biological response to nutrients (e.g., algal productivity) depends on a variety of mitigating factors such as basin morphology and substrate characteristics, tidal energy, stratification, temperature, light availability, biological community structure, and seed populations. Thus high concentrations are not an obligatory indicator of eutrophication and low concentrations do not necessarily indicate absence of eutrophication.

Given these problems, in 1999 the USEPA Region 9 and the California State Water Resources Control Board (SWRCB) chose an alternative approach to developing nutrient objectives (USEPA, 2006). This approach, known as the Nutrient Numeric Endpoint (NNE) framework, establishes a suite of numeric endpoints based on the ecological response of an aquatic waterbody to nutrient over-enrichment (eutrophication, e.g., algal biomass, dissolved oxygen). It was suggested that numeric endpoints, if successfully developed, would serve as *guidanc*e to translate *narrative* water quality objectives (State of California's term for water quality criteria) for nutrients and biostimulatory substances. A key component of the NNE framework is the availability or development of stressor- response tools that link the ecological response indicators with nutrient loads and other potential management controls for TMDL development and implementation.

The California NNE framework was first developed for streams and lakes (USEPA, 2006) and is currently under development for estuaries. A scientific framework has been presented to support the development of numeric endpoints for a suite of biological response indicators and highlight data gaps and research recommendations for their development (USEPA, 2007). A subsequent document articulated a broad work plan to address data gaps, develop numeric endpoints, and support the efficient and cost-effective development of stressor-response TMDL tools (USEPA, 2008). Within this work plan, one key step was to summarize existing literature relevant to the development of a set of NNEs and TMDL tools in relation to monitoring and assessment of eutrophication in SF Bay estuary. A key outcome of this initial step is a work plan vetted by the scientists and stakeholders that work and live around the estuary.

1.2 Objective, Geographic Scope and Organization of this Report

The purpose of this document is to present the review of literature and monitoring programs relevant to the assessment of eutrophication in SF Bay, with the goal of providing a baseline of available information to formulate a work plan to develop NNEs for this estuary. The review had four specific objectives:

- Evaluate appropriate indicators to assess eutrophication in SF Bay;
- Summarize existing literature and identify data gaps on the status of eutrophication in SF Bay with respect to these indicators;

- Describe what data and tools exist to evaluate the trends in nutrient loading to the Bay; and

- Summarize, to the extent possible (What do they reveal about trends in nutrient loads over time?)

For the purposes of this literature review, the geographic scope of this effort is limited to the areas of the Bay included within the San Francisco Bay Regional Water Quality Control Board (SF Water Board) jurisdiction (Figure 1). The upstream boundary of the SF Water Board is roughly coincident with the 2 ppt isohaline of bottom waters (a.k.a. "X2", Jassby et al., 1994). This X2 isohaline has a significant statistical relationship with measures of SF estuary resources, including: 1) supply of phytoplankton and phytoplankton-derived detritus, 2) benthic macroinvertebrate, 3) larval fish survival, and the abundance of fish.

The intention is this will be a living document, updated over time to reflect input from scientists, stakeholder groups, and the interested public. Drafts will be identified by date of released and should be cited as such. The report is organized into six sections:

Section 1 gives the introduction, purpose of the document, the organization, and definitions of key terms used throughout the report.

Section 2 gives a brief summary of the conceptual framework of the NNE, preliminary classification and consideration of habitat types, and candidate estuarine NNE (E-NNE) indicators.

Section 3 provides an overview of relevant physiographic information for the Bay Area including human population trends, climate, habitats (both in Bay and fringing), beneficial uses and water quality criteria designated by the State of California through the San Francisco Regional Water Quality Control Board (hereto referred to as the "SF Water Board").

Section 4 provides a review of the current understanding of external nutrient loads and ambient nutrient concentrations in SF Bay.

Section 5 reviews and summarizes existing information on candidate NNE indicators for the SF Bay estuary. The section focuses on seven main indicator groups: phytoplankton blooms and HAB species, dissolved oxygen (Hypoxia and anoxia), macroalgae, submerged aquatic vegetation (sea grass and brackish submerged aquatic vegetation), benthic macroinvertebrates, jellyfish, and ammonium including ammonium nitrate ratio, urea, and toxicity.

Section 6 summarizes the review, identifies important data gaps and recommends next steps.

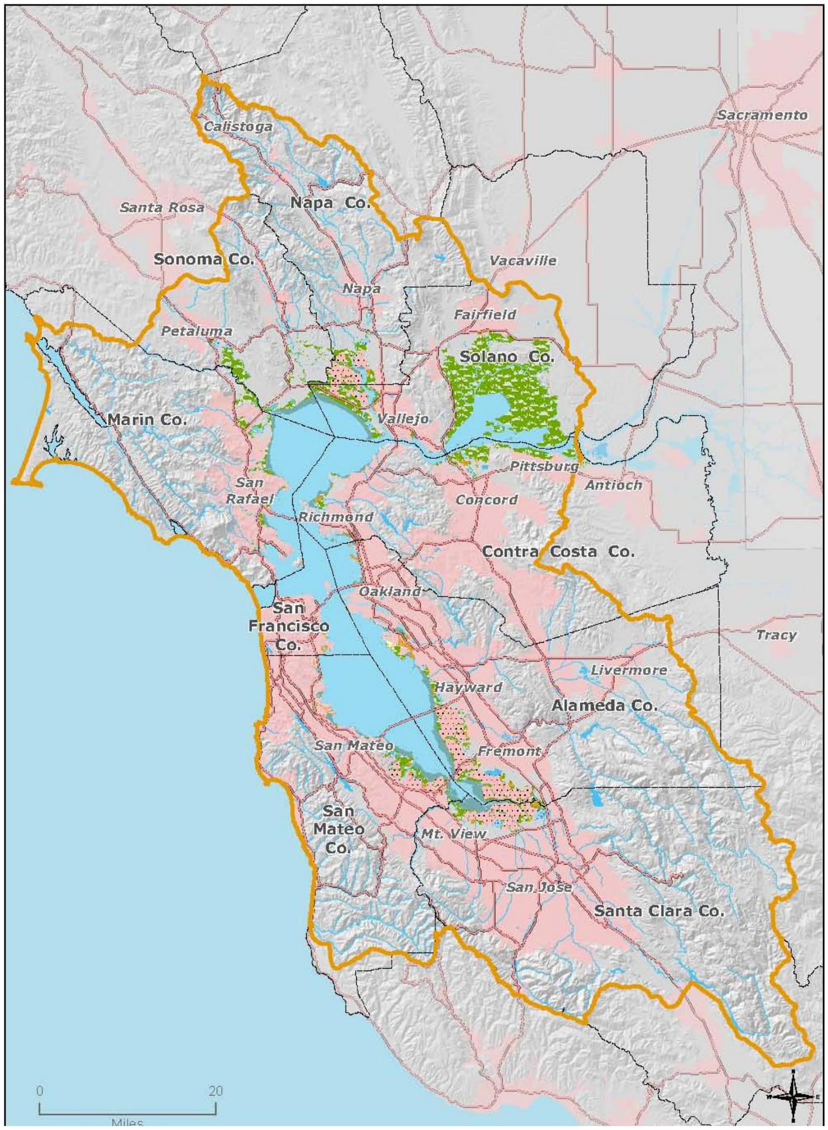


Figure 1.1. Geographic scope of the literature review, defined by SF Water Board jurisdiction.

1.3 Important Definitions

For those outside the regulatory world, distinction between terms like “criteria,” “standards”, “objectives,” and “endpoints” can be confusing. The purpose of this section is to provide definitions of the terms that are linked closely to how the NNE framework will be implemented.

Eutrophication: Eutrophication is defined as the acceleration of the delivery, in situ production of organic matter, and accumulation of organic matter (Nixon, 1995). One main cause of eutrophication in estuaries is nutrient over enrichment (nitrogen, phosphorus and silica). However, other factors influence primary producer growth and the build-up of nutrient concentrations, and hence modify (or buffer) the response of a system to increased nutrient loads (hereto referred to as **co-factors**). These co-factors include hydrologic residence times, mixing characteristics, water temperature, light climate, grazing pressure and, in some cases, coastal upwelling.

***Indicator*:** A characteristic of an ecosystem that is related to, or derived from, a measure of biotic or abiotic variable, that can provide quantitative information on ecological condition, structure and/or function. With respect to the water quality objectives, indicators are the ecological parameters for which narrative or numeric objectives are developed.

Water Quality Standards: Water quality standards are the foundation of the water quality-based control program mandated by the Clean Water Act. Water Quality Standards define the goals for a waterbody by designating its uses, setting criteria to protect those uses, and establishing provisions to protect water quality from pollutants. A water quality standard consists of three basic elements:

1. **Designated uses** of the water body (e.g., recreation, water supply, aquatic life, agriculture)
2. **Water quality criteria** to protect designated uses (numeric pollutant concentrations and narrative requirements)
3. A**ntidegradation policy** to maintain and protect existing uses and high quality waters

Water Quality Criteria: Section 303 of the Clean Water Act gives the States and authorized Tribes power to adopt water quality criteria with sufficient coverage of parameters and of adequate stringency to protect designated uses. In adopting criteria, States and Tribes may:

- Adopt the criteria that USEPA publishes under §304(a) of the Clean Water Act;
- Modify the §304(a) criteria to reflect site-specific conditions; or
- Adopt criteria based on other scientifically-defensible methods.

The State of California's water criteria are implemented as "water quality objectives," as defined in the Water Code (of the Porter Cologne Act; for further explanation, see below).

States and Tribes typically adopt both **numeric** and **narrative** criteria. **Numeric** criteria are quantitative. **Narrative** criteria lack specific numeric targets but define a targeted condition that must be achieved.

Section 303(c)(2)(B) of the Clean Water Act requires States and authorized Tribes to adopt numeric criteria for priority toxic pollutants for which the Agency has published §304(a) criteria. In addition to narrative and numeric (chemical-specific) criteria, other types of water quality criteria include:

- Biological criteria: a description of the desired biological condition of the aquatic community, for example, based on the numbers and kinds of organisms expected to be present in a water body.
- Nutrient criteria: a means to protect against nutrient over-enrichment and cultural eutrophication.

- Sediment criteria: a description of conditions that will avoid adverse effects of contaminated and uncontaminated sediments.

Water Quality Objectives: The Water Code (Porter-Cologne Act) provides that each Regional Water Quality Control Board shall establish water quality objectives for the waters of the state i.e., (ground and surface waters) which, in the Regional Board's judgment, are necessary for the reasonable protection of beneficial uses and for the prevention of nuisance. The State of California typically adopts both **numeric** and **narrative** objectives. **Numeric** objectives are quantitative. **Narrative** objectives present general descriptions of water quality that must be attained through pollutant control measures. Narrative objectives are also often a basis for the development of numerical objectives.

Numeric Endpoint: Within the context of the NNE framework, numeric endpoints are thresholds that define the magnitude of an indicator that is considered protective of ecological health. These numeric endpoints serve as guidance to Regional Boards in translating narrative nutrient or biostimulatory substance water quality objectives. They are called “numeric endpoints” rather than “numeric objectives” to distinguish the difference with respect to SWRCB policy. Objectives are promulgated through a public process and incorporated into basin plans. Numeric endpoints are guidance that can evolve over time without the need to go through a formal standards development process.

2. NNE Conceptual Approach, Classification, and Key Indicators

This section describes the NNE conceptual approach, estuarine classification and key habitat types and the rationale for selection of candidate NNE indicators identified for SF Bay estuary. The material in this section is derived from Sutula et al. (2011), which conducted an extensive review of candidate NNE indicators for California estuaries.

2.1 NNE Conceptual Approach

The Nutrient Numeric Endpoints (NNE) framework is a term coined to describe the SWRCB staff strategy for developing nutrient objectives for the State of California. This draft strategy includes developing a narrative objective, plus numeric guidance that would be incorporated by default into the Basin Plans of the Regional Water Quality Control Boards. The purpose of developing NNEs for California estuaries is to provide the State Water Resources Control Board and the Regional Water Quality Control Boards with a scientifically-defensible framework that can serve as guidance for adopting water quality objectives for nutrients.

The development of an NNE assessment framework for SF Bay is consistent with the findings of the review of candidate indicators for California estuaries (Sutula et al., 2011), but this work represents a more focused effort to develop a framework for assessment eutrophication in SF Bay, with the intent to incorporate specific indicators and thresholds into the SF Regional Water Quality Control Board's (hereto referred to as "SF Water Board") Basin plan.

2.1.1 Why Nutrient Concentrations Should Not Be Used to Set Nutrient Water Quality Objectives in Estuaries

Nutrient objectives are scientifically challenging. Nutrients are required to support life, but assessment of how much is "too much" is not straightforward. Typical paradigms used to set thresholds for toxic contaminants do not apply, in part because adverse effects of nutrient over enrichment are visible at orders of magnitude below recognized toxicity thresholds for ammonium and nitrate.

USEPA guidance on nutrient objective development generally recommends three means to set nutrient criteria (USEPA, 2001): 1) reference approach, 2) empirical stress-response approach, and 3) cause-effect approach. The reference waterbody approach involves characterization of the distributions of nutrient in "minimally disturbed" waterbodies. Nutrient concentrations are chosen at some statistical percentile of those reference waterbodies. The empirical stress-response approach involves establishing statistical relationships between the causal or stressor (in this case nutrient concentrations or loads) and the ecological response (changes in algal or aquatic plant biomass or community structure, changes in sediment or water chemistry (e.g., dissolved oxygen, pH). The cause-effect approach involves identifying the ecological responses of concern and mechanistically modeling the linkage back to nutrient loads and other co-factors controlling response (e.g., hydrology, grazers, denitrification, etc.).

SWRCB staff and USEPA Region 9 staff evaluated these three approaches for setting nutrient objectives in California waterbodies and determined that, while it may choose to ultimately incorporate some elements of all approaches into California's strategy for setting nutrient objectives, it would rely most heavily on the cause-effect approach. There were several reasons for this. First, the cause-effect approach has a more direct linkage with beneficial uses and is generally thought to lend itself to a more precise diagnosis of adverse effects. Second, the alternative approaches require a tremendous amount of data not currently available in such a large state. Third, the reference approach is particularly problematic because it automatically relegates a certain percentage of the reference sites to an "impaired" status. In addition, for many waterbody types, minimally disturbed reference sites are largely unavailable. Fourth, statistical stress-response relationships can be spurious, or have lots of unexplained variability (i.e., poor precision). This poor precision is translated to a larger margin of safety required (more conservative limits) for load allocations and permit limits. While waterbody typology, to some degree, can assist in explaining some of this variability, it cannot completely remove the concern. Thus, while simpler than the cause-effect approach, the empirical stress-response approach will result in more false negative and false positive determinations of adverse effects and in the end will be more costly to the public.

For estuaries, reliance on the cause-effect approach is strongly suggested, because in the majority of circumstances, the reference or empirical stress-response approaches are simply untenable (Cloern 2001). Estuaries within California are highly variable in how they respond to nutrient loading due to differences in physiographic setting, salinity regime, frequency and timing of freshwater flows, magnitude of tidal forcing, sediment load, stratification, residence time, denitrification, etc. This combination of "co-factors" results in differences in the dominant primary producer communities (i.e., phytoplankton, macroalgae, benthic algae, submerged aquatic vegetation, emergent macrophytes). It also creates variability in the pathways that control how nutrients cycle within the estuary. At times, these co-factors can play a larger role in mitigating estuarine response to nutrient loads or concentrations, blurring or completely obscuring a simple prediction of primary productivity limited by nutrients (e.g., Figure 2.1). For example, in many lagoonal estuaries, benthic algal blooms can act to reduce surface water concentrations of nutrients to non-detectable levels. Thus while the estuary may be in a clearly impacted state, it would appear to meet N and P ambient water quality objectives. In estuaries such as SF Bay, synthesis by Cloern and Dugdale (2010) have clearly shown that ambient nutrient concentrations do not correlate with measures of primary productivity , in part because of important co-factors that override simple nutrient limitation of primary production.

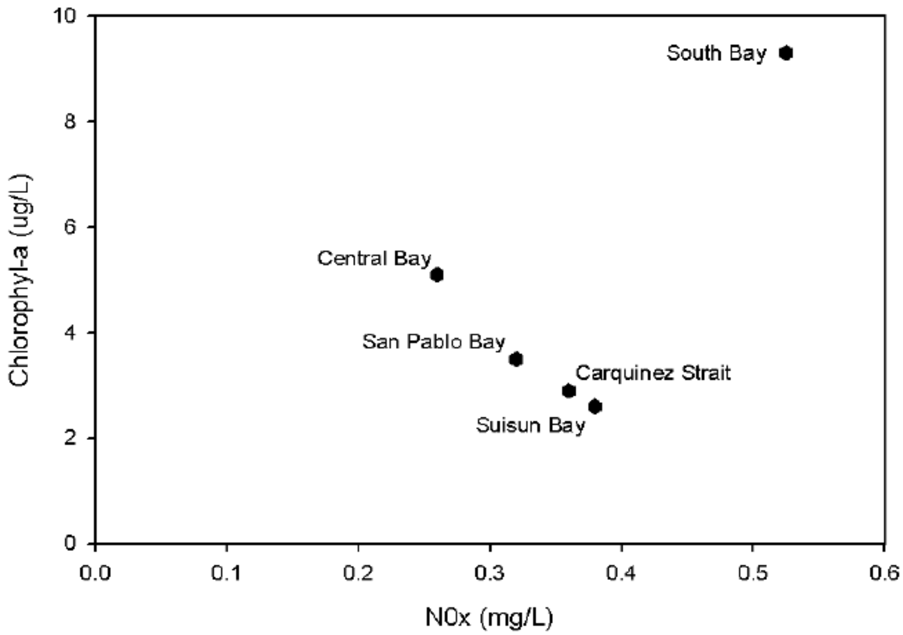


Figure 2.1. Mean chlorophyll a concentrations as a function of mean nitrate+nitrite (NOx) concentrations in SF Bay for the period January 1999 to February 2009 (Data Source: USGS: http://sfbay.wr.usgs.gov/access/wqdata).

2.1.2 Key Tenets of the NNE Approach

The NNE framework for California waterbodies is basely largely on the cause-effect approach. The framework has three organizing principals (Creager, Butcher et al., 2005):

1. *Ecological response indicators provide a more direct risk-based linkage to beneficial uses than nutrient concentrations or loads alone. Thus the NNE framework is based on the diagnosis of eutrophication or other adverse effects and its consequences rather than nutrient over enrichment.*

Except in extreme cases such as unionized ammonium causing toxicity, nutrients themselves do not impair beneficial uses. Rather, ecological response to nutrient loading causes adverse effects that impair uses. Instead of setting objectives solely in terms of nutrient concentrations, it is preferable to use an analysis that takes into account the risk of impairment of these uses. The NNE framework needs to target information on ecological response indicators such as dissolved oxygen, surface water phytoplankton and HAB biomass (e.g., chlorophyll a, water clarity), macroalgal biomass and percent cover, benthic algal biomass (sediment chlorophyll a) and submerged aquatic vegetation (SAV) density and percent cover, and aesthetics (e.g., foul odors, unsightliness). These ecological response indicators provide a more direct risk-based linkage to beneficial uses than the ambient nutrient concentrations or nutrient loads. Given this approach, it is critical that tools be developed that link the response indicators back to nutrient loads and other co-factors and management controls (hydrology, etc.).

2. *A weight of evidence approach with multiple indicators will produce a more robust assessment of eutrophication.*

When possible, the use of multiple indicators in a "weight of evidence" approach provides a more robust means to assess ecological condition and determine impairment. This approach is similar to the multimetric index approach, which defines an array of metrics or measures that individually provide limited information on biological status, but when integrated, functions as an overall indicator of biological condition (Karr and Chu, 1999).

3. *Use of models to convert response indicators to site-specific nutrient loads or concentrations.*

A key premise of the NNE framework is the use of models to convert numeric endpoints, based on ecological response indicators, to site- specific nutrient load goals appropriate for assessment, permitting, and TMDLs.

Thus the intent of the NNE framework is to control excess nutrient loads to levels such that the risk or probability of impairing the designated uses is limited to a low level. If the nutrients present – regardless of actual magnitude – have a low probability of impairing uses, then water quality standards can be considered met.

2.2 How Response Indicators Would Be Used: Development of a Diagnostic Assessment Framework

Within the regulatory context, waterbody assessments are made in order to make determination of whether the waterbody is meeting beneficial uses or impaired, as an example, for nutrients. In this context, a diagnostic assessment framework is the structured set of decision rules and guidance for interpretation that helps to classify the waterbody in categories of minimally disturbed (fully sustaining beneficial uses) to moderately disturbed (still sustaining beneficial uses, but functions reduced), to very disturbed (clearly not meeting beneficial uses). Although scientists can provide a lot of guidance and data synthesis to illustrate how the assessment framework could be formed, ultimately the decision of what levels to set thresholds that separate the categories (e.g., minimally versus moderately and very disturbed) is a policy decision. These thresholds are what are referred to as "nutrient numeric endpoints."

Development of the diagnostic assessment framework begins by choosing indicators that would be measured and used to determine waterbody status. It is important to distinguish between three types of indicators for an NNE assessment framework:

1. Primary indicators
2. Supporting indicators
3. Co-factor indicators required for data interpretation

Primary indicators will play a central role in the NNE assessment framework. Designation of these indicators as "primary" implies a higher level of confidence in these indicators to be used to make an assessment of adverse effects, based on a wealth of experience and knowledge about how this indicator captures and represents ecological response. Primary indicators are those which are considered to meet all explicit criteria (see Section 2.5) established to evaluate candidate NNE indicators.

Supporting indicators are those which could be collected to provide supporting lines of evidence. These indicators may have met many, though not all evaluation criteria, but are consider important because they are commonly used to assess eutrophication in scientific studies, albeit with a lower level of confidence to assess adverse effects of eutrophication. Use of the indicator as supporting evidence over time may increase confidence and cause it to be promoted to "primary."

Finally, co-factors are indicators that could be part of a routine monitoring program and important for data interpretation and trends analysis.

2.3 Context for Indicator Selection: Estuarine Classes and Major Habitat Types

Discussion of estuarine numeric nutrient endpoint (E-NNE) candidate ecological response indicators requires mention of estuarine classes and key habitat types. The approximately 400 estuaries found in the State of California are highly variable in terms of physiographic setting, salinity regime, frequency and timing of freshwater flows, magnitude of tidal forcing, sediment load, stratification, residence time, etc. (Engle et al., 2007). This combination of factors results in differences in the dominant primary producer communities (i.e., phytoplankton, macroalgae, microphytobenthos, submerged aquatic vegetation, emergent macrophytes). It also creates variability in the pathways for nutrient cycling within estuaries. As a result of these differences, estuaries are expected to be variable in how they respond to nutrient loading (NRC, 2000). Partitioning this apparent natural variability into classes will improve the E-NNE framework by eliminating the need to research and define indicators for each of the 400 individual estuaries. Instead, indicators will be defined and tested for each estuarine class (numbering just six).

Classification approaches can by driven by conceptual, empirical or statistical approaches. The NNE Technical Team has proposed a preliminary classification of California estuaries, based on a conceptual approach modeled after the Coastal Marine Ecological Classification Standard (CMECS; Madden et al., 2005; Sutula et al., 2011). The preliminary classes are shown in Table 2.1.

Table 2.1. Preliminary classification of California estuaries.

GEOFORM	SEASONALITY OF SURFACE WATER CONNECTION TO OCEAN
Enclosed Bay	Perennial
Lagoon	Perennial Intermittent Ephemeral
River mouth	Perennial Intermittent

According to this classification, SF Bay estuary is an enclosed bay. However, the estuary contains at least four compartments that are hydrologically distinct from each other. The extreme northern compartment of the estuary receives the largest inflow of fresh water into the estuary. The central component of the estuary receives very little freshwater input and is greatly influenced by tidal action. The lower two compartments include the "south bay" and "extreme south bay." The extreme south bay encompasses the area between San Jose and the Dumbarton Bridge and is semi-hydrologically distinct and has a slower "flushing rate" than its northern neighbor the "south bay", which extends north from the Dumbarton Bridge to just south of the Oakland – Bay Bridge. Given the size and geomorphic complexity of the estuary, a more detailed review of estuarine classification and dominant habitat types of SF Bay estuary is required in order to understand relevant ecological response indicators (USEPA, 2007).

Within these classes, several key habitat types can be distinguished that organize what indicators may be relevant to consider. For example, Table 2.2 summarizes the relevant aquatic primary producer groups that could be used to diagnose eutrophication, expressed across a range of water depth and salinity regime (Table 2.2.; Day et al., 1989). Thus within each estuarine class, the indicators appropriate to assess eutrophication can change by habitat type.

Table 2.2. Dominant primary producer groups present in California estuaries as a function of water depth and salinity range.

Depth	Dominant Primary Producers
Intertidal	Macroalgae Microphytobenthos Seagrass (intertidal Central & No.Calif.)
Shallow subtidal (<10 m)	Macroalgae Microphytobenthos Brackish water SAV and Seagrass Phytoplankton
Deep or light limited subtidal (>=10 m)	Microphytobenthos Phytoplankton Drift or Floating Macroalgae (in oligohaline habitats)

2.4 Conceptual Models and Candidate Ecological Response Indicators

Eutrophication is defined as the acceleration of the delivery, in situ production of organic matter, and accumulation of organic matter within an aquatic ecosystem (Howarth, 1988; Nixon, 1995; Cloern, 2001). One of the main causes of eutrophication in estuaries is nutrient over enrichment (nitrogen, phosphorus and silica). Other factors influence primary producer growth and nutrient availability, and hence modify (or buffer) the response of a system to increased nutrient loads (referred to as **co-factors**). These co-factors include hydrologic residence times, mixing characteristics, water temperature, light climate, grazing pressure and, in some cases, coastal upwelling (Figure 2.1). A simple conceptual model of estuarine ecological response to eutrophication can be described (Figure 2.1). The increased nutrient loads and alterations in co-factors can result in:

1. Changes to aquatic primary producers,
2. Altered water and sediment biogeochemistry, and
3. Altered community structure of secondary (invertebrates) and tertiary consumers (fish, birds, mammals).

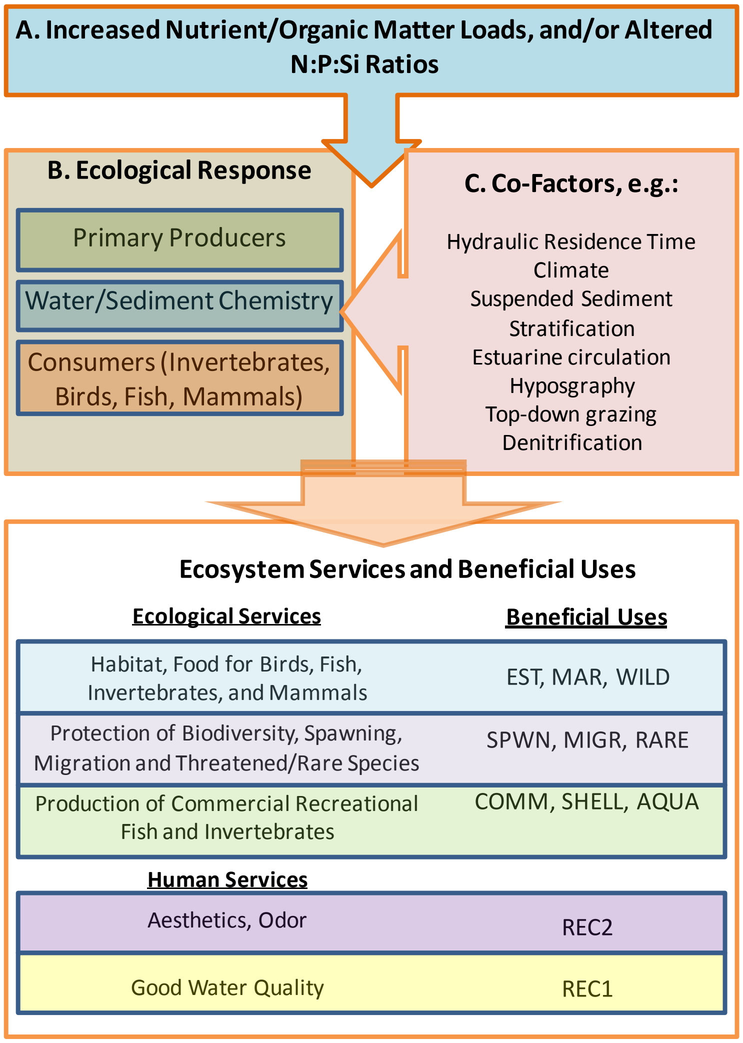


Figure 2.2. Conceptual framework of linkage of nutrient loading (A), ecological response (B), which includes altered primary producers, sediment and water biogeochemistry, and secondary & tertiary consumers), co-factors modulating response (C), and altered ecological services and beneficial uses.

This cascade of effects has a direct effect on the ecosystem services and beneficial uses an estuary provides, including reduced:

- Habitat for aquatic life (including EST, MAR, WILD)
- Protection of biodiversity including rare, threatened and endangered species and migratory and spawning habitat (RARE, SPWN, MIGR)
- Productivity of commercial and recreational fisheries (SHELL, COMM, AQUA).
- Good aesthetics and lack of odors (REC2)
- Maintenance of good water quality (REC1, COMM, AQUA, SHELL)

The three identified components of the ecological response to eutrophication (Figure 2.4 component (B), Figure 2.5) can be used as an organizing framework within which to list and review possible indicators for the E-NNE. Each component is further explained below, along with a list of corresponding indicators under consideration for the E-NNE framework.

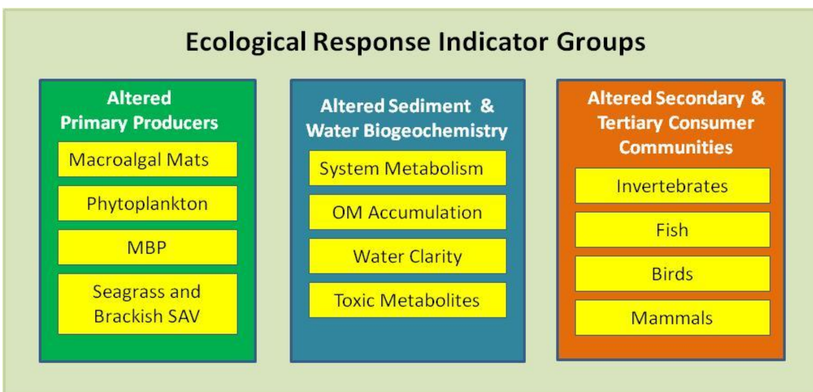


Figure 2.3. Ecological indicator groups, which include altered primary producers, sediment and water biogeochemistry, and secondary & tertiary consumers. MPB = microphytobenthos, OM = sediment organic matter accumulation.

2.4.1 Changes in Aquatic Primary Producer (APP) Community Structure

As an estuary becomes increasing eutrophic, predictable changes occur with respect the types and relative abundance of the primary producer communities, as depicted in Figure 2.6. Estuaries in a "minimally disturbed" condition are typically dominated by primary producers tolerant of low nutrient conditions, such as microphytobenthos (benthic microalgae), seagrasses, or, in deep or turbid estuaries, a high diversity of phytoplankton at relatively low biomass. As nutrient availability increases, the growth of epiphytic micro-, macroalgae as well as opportunistic ephemeral macroalgae is favored in shallow subtidal estuaries. In deep or turbid estuaries, phytoplankton biomass increases, favoring nutrient tolerant and often, HAB species that can produce toxins harmful to marine life and humans (Fong et al., 1993, Valiela et al., 1997, Viaroli et al., 2008). In the extreme end of the eutrophication gradient, macroalgae and cyanobacterial mats dominate intertidal and shallow subtidal habitat, while in deepwater or turbid habitat, cyanobacteria and/or picoplankton blooms can dominate, causing dystrophy.

These changes along a gradient of increasing nutrient availability provide the basis for selecting one or more primary producers as indicators for the E-NNE framework. The precise indicators that will be relevant are dependent on the habitat type and estuarine class. Table 2.3 lists the indicator groups and specific metrics under evaluation for the E-NNE framework. Literature used to evaluate these indicators is summarized in Sutula (2011).

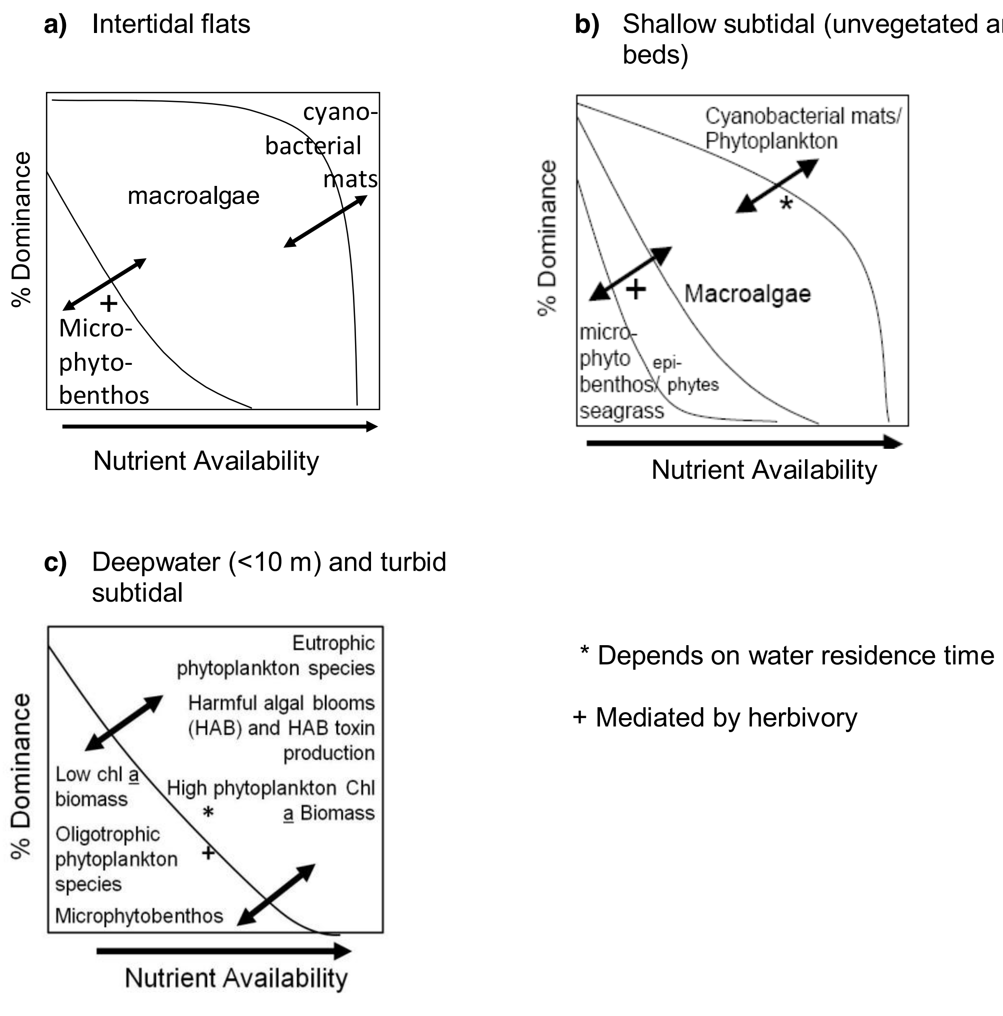


Figure 2.4. Conceptual model of relationship between nutrient availability and relative dominance of primary producers in California estuaries by major habitat type: (a) intertidal flats, (b) shallow subtidal and (c) deepwater or turbid subtidal.

Table 2.3. List of primary producer indicator groups and specific metrics reviewed as candidate indicators for the E-NNE.

Primary Producer Indicator Group	Indicator or Metric
Macroalgae	Percent Cover Biomass
Phytoplankton	Chlorophyll a Concentration (Biomass) Productivity Assemblage/Taxonomic Composition Harmful Algal Bloom Species Abundance Harmful Algal Bloom Species Toxin Concentration
Microphytobenthos	Sediment Chlorophyll a Taxonomic Composition
Seagrass and Brackish Water Submerged Aquatic Vegetation	SAV Aerial Distribution SAV Taxonomic Composition SAV Biomass SAV Density Epiphyte Load on SAV Macroalgal Biomass/Cover on SAV Water Column Chlorophyll a Water Column Light Attenuation

2.4.2 Altered Water and Sediment Chemistry (Biogeochemical Cycling)

As the process of eutrophication progresses, elevated live and dead aquatic primary producer (APP) biomass provide an elevated supply of labile organic matter, setting off a cascade of altered biogeochemical cycling in the sediments and surface waters. These effects include increased respiration in the sediments and surface waters, increased extent, frequency and duration of hypoxia, and increased concentrations of sediment pore water ammonium, sulfide, increasing the potential for toxicity to benthic organisms (D'Avanzo and Kremer, 1994; Nixon, 1995; Diaz, 2001; Howarth, Sharpley et al., 2002). The efficiency of nitrogen and carbon cycling decreases, which fuels increased organic matter accumulation in the sediments.

With respect to review of candidate E-NNE indicators, changes in biogeochemical cycling in estuarine sediments and surface waters due to eutrophication can be broken down into four general categories (Table 2.4) each having a set of discrete candidate indicators:

- Changes in water clarity, due to increased suspended live and dead biomass
- Altered rates of system metabolism, which capture the relative rates of carbon production and respiration within a system
- Increased sediment organic matter accumulation, which is the rate at which organic matter is accumulates within sediments
- Altered rates of nutrient cycling, which can be defined as the rates of in key transformation mechanisms for nitrogen, phosphorus, and associated elements involved in redox reactions such as sulfur, iron and manganese

Table 2.4. Table of candidate water column and sediment chemistry indicators reviewed for the E-NNE framework (from Sutula, 2011).

Indicator Group	Indicator or Metric
Nutrients	Ammonia Urea N:P Ratio
Water Clarity	Secchi Depth Kd (Light Extinction) Turbidity
Dissolved Oxygen	Dissolved Oxygen Concentration Biological or Chemical Oxygen Demand Sediment Oxygen Demand
Benthic Metabolism	Benthic Production: Respiration Ratio Benthic TCO2 Flux
Organic Matter Accumulation and Sediment Redox Status	Sediment %OC, %N, and %P Sediment C:N: P Ratio
	Sediment TOC:TS and Degree of Pyritization
Nitrogen Cycling	Denitrification Efficiency

2.4.3 Altered Community Composition of Secondary and Tertiary Consumers

Poor habitat quality and altered abundance of primary producers causes shifts in the secondary consumers (benthic infaunal, epifauna and pelagic invertebrates) that are directly impacted by alterations in primary producer community structure and degradation in water and sediment chemistry. Higher level consumers, such as fish, birds, mammals, and other invertebrates that prey upon these secondary consumers (referred to here as tertiary consumers), experience reduced food availability and quality, reduce reproductive success, increased stress and disease, and increased mortality.

While secondary and tertiary consumers are closely linked to ecosystem services and beneficial uses (Figure 2.1), use of these organisms as indicators for the E-NNE framework is problematic because organism and population measures of health are impacted by a variety of different stressors in a complex environment which is not easy to model. Within the group of secondary and tertiary consumers, benthic macroinvertebrates are the sole taxonomic group recommended pursuing for possible inclusion as an E-NNE indicator in some key habitat types and estuarine classes.

Because invertebrates that live in or on sediments are exposed to environmental stressors on an ongoing basis, the benthic life present at a particular location often provides a good indicator of sediment habitat quality. Benthic community composition can be impacted by contamination, eutrophication as well as natural variations in habitat and physical disturbance. The State of California has been developing a benthic response index (BRI) for bays and estuaries with salinities of 18 ppt or greater. Benthic indices apply standard mathematical formulas to data on the number and diversity of benthic organisms at a particular location to find a score that rates the disturbance of the community. This provides a simple means for communicating complex ecological data to environmental managers. The BRI is a component of the SWRCB's sediment quality objectives (www.waterboards.ca.gov/water_issues/programs/bptcp/sediment.shtml), which establishes numeric endpoints for sediment quality due to toxic contaminants.

2.5 Indicator Review Criteria and Candidate NNE Indicators for SF Bay

Sutula (2011) reviewed candidate indicators for use in assessing eutrophication in California estuaries. The following criteria were used in the reviews of existing science to evaluate the utility of each indicator for the E-NNE assessment framework.

Indicators Should:

- Have well documented links to estuarine beneficial uses and, if possible, organisms at multiple trophic levels
- Have a predictive relationship with causal factors such as nutrient concentrations/loads and other factors known to regulate response to eutrophication (hydrology, etc.). This relationship could be empirical (modeled as a statistical relationship between load/concentration and

response or modeled mechanistically through tools such as a simple spreadsheet or dynamic simulation models)

- Have a scientifically sound and practical measurement process that can be accurately and precisely measured over large areas and over multiple years (long term) to quantify the spatial and temporal variability in the forcing and response variables typical of California estuaries
- Must be able to show a trend either towards increasing or/and decreasing eutrophication with an acceptable signal: noise ratio

Based on the review by Sutula (2011) and early discussions with the SF Bay Technical Team, the following indicators were short-listed for further review and synthesis of existing data for the SF Bay estuary (Table 2.5).

Table 2.5. Short-list of candidate E-NNE indicators for SF Estuary by applicable habitat type. Shaded boxes represent applicable habitat.

Indicator	**Habitat Type**			
	Tidal Flats	Shallow Subtidal Unvegetated	Seagrass/Brackish SAV	Deepwater/Turbid Subtidal
Dissolved oxygen		√	√	√
Macroalgae biomass/% cover	√	√	√	
Epiphyte load			√	
Phytoplankton biomass and productivity		√	√	√
Phytoplankton taxonomy, abundance, and/or harmful algal bloom toxin concentrations		√	√	√
Macrobenthos taxonomy/biomass		√		√
Ammonium and urea		√	√	√
Light attenuation		√	√	√

Note that seagrass areal extent and density and macrobenthos taxonomy are known to be affected by a variety of stressors including eutrophication, but cannot be considered to be specific diagnostic indicators of eutrophication (see Sutula, 2011). These indicators would be considered if part of a multimetric assessment protocol for eutrophication, but not as stand-alone indicators.

3. Geographic Setting and Regulatory Context

3.1 Geographic Setting: San Francisco Bay Estuary

The San Francisco Bay estuary (37°27' - 38°10' N, 121°45' - 122°31' W) lies between the Sacramento-San Joaquin Delta and the Pacific Ocean and receives flow from approximately 160,000 km^2 (37% of California). The "urbanized estuary" (Comomos (ed.), 1979) is surrounded by nine counties with a total resident population of 6.78 million (2000 census) (Figure 3.1) 70.0% of whom reside within watersheds draining to the Bay south of the Richmond San Rafael Bridge (Hwy 580) (the Central and Southern portions of the Bay) within and south of the cities of Larkspur and El Cerrito.

Population increase has not been uniform and build-out has occurred mainly through conversion from agriculture to urban land use. By far the most rapid population growth occurred in the Bay Area during the decades of 1940, 1950 and 1960 largely through medium density residential urban infill adjacent to the Bay (the populations of Contra Costa, Santa Clara and San Mateo increased by 5-6 times and the population of Marin increased by 4 times from the 1940 census to the 1970 census). However more recently (1970 – 2000 census), the largest population increases have been occurring in outlying cities of Napa County (e.g., Calistoga, Napa, and American Canyon more than doubling in population), Solano County (e.g., Suisun City increasing by 9 times), Contra Costa County (e.g., Hercules, Oakley, San Ramon, Brentwood, Clayton averaged together increasing by a staggering 25 times), Alameda (e.g., Pleasanton increasing 3.5 times) and Santa Clara County (e.g., Gilroy and Morgan Hill increasing by over 3 and 6 times respectively). During the more recent decades, urban build out has been through conversion from mainly agricultural land to a mix of medium density urban and lower density suburban residential. It is likely that agricultural and urban lands are continuing to release nutrients that get to the Bay via river and urban stormwater runoff and this release might be exacerbated by disturbances during land use conversion and related construction activities.

The climate in the area is generally mild. Average temperature in the summer ranges from the low to high 60's, and in the winter between the mid-forties to mid-50's F (Figure 3.2). Daylight hours in the region range from 9.5 to 15 per day. Available data for 2008 indicates mean hourly solar radiation was 362 Ly/hr in Oakland (Oakland Hills gage), 415 Ly/hr in the Napa area (Carneros gage) and 408 Ly/hr in Santa Clara (Morgan Hill gage) (hourly data, CIMIS, 2008). Peak daily solar radiation occurred during June and July at all three stations.

According to analysis of precipitation data available between 1907 and 1956 from gauges across the Bay Area, mean annual rain directly over the Bay ranges between 14.75 inches (375 mm) in the far South Bay to 28 inches (710 mm) on the western margins of San Pablo Bay in the North (Figure 3.3). In general, rain over the land area of the nine counties adjacent to the Bay is greater than over the Bay itself ranging from about 14 inches (350 mm) near sea level to 48 inches (1,220 mm) on high western facing slopes at higher elevations.

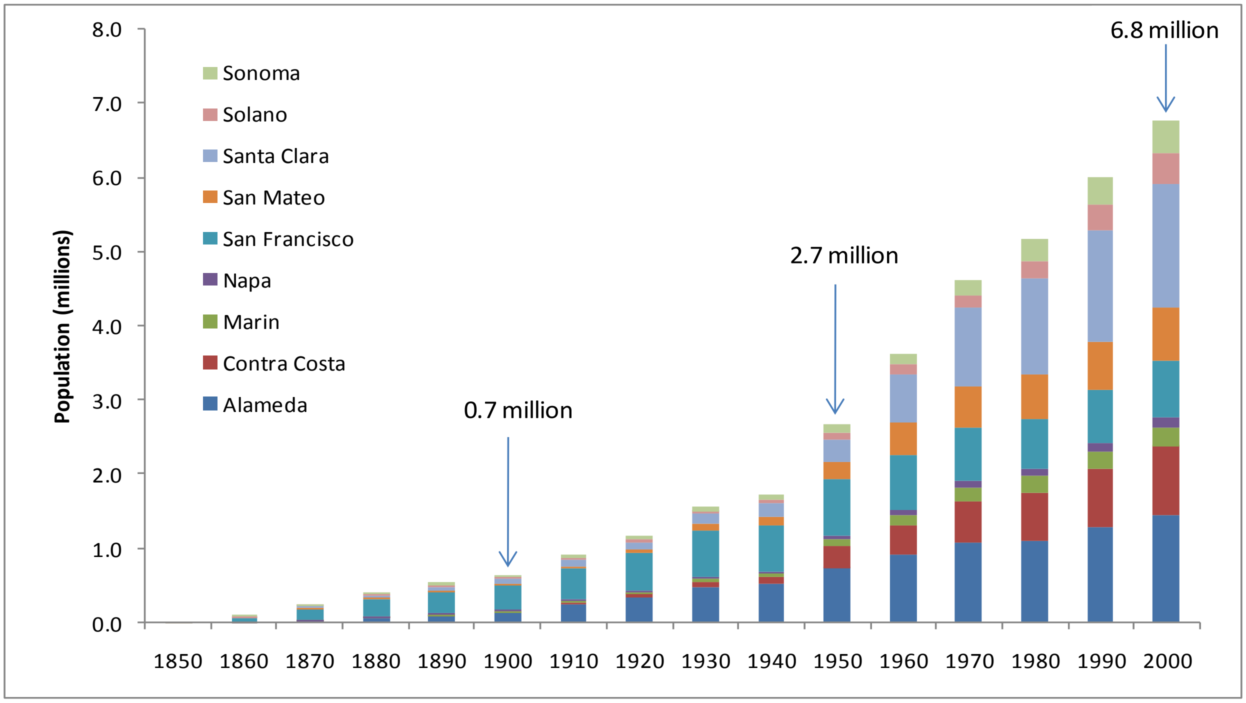


Figure 3.1. Population totals in the nine Bay Area Counties on a decadal time series since 1850. Source: Census Bureau.

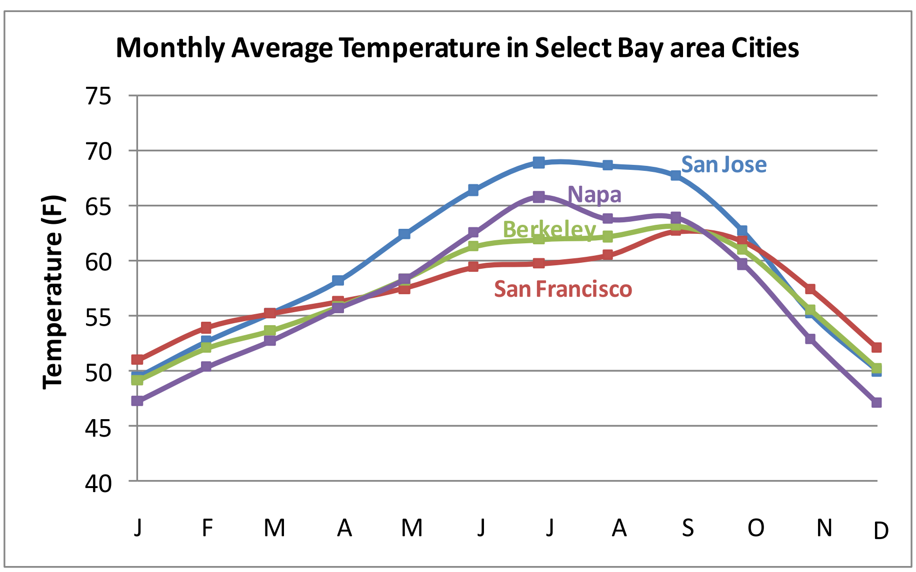


Figure 3.2. The monthly average temperature. Data downloaded from the Western Regional Climate Center.

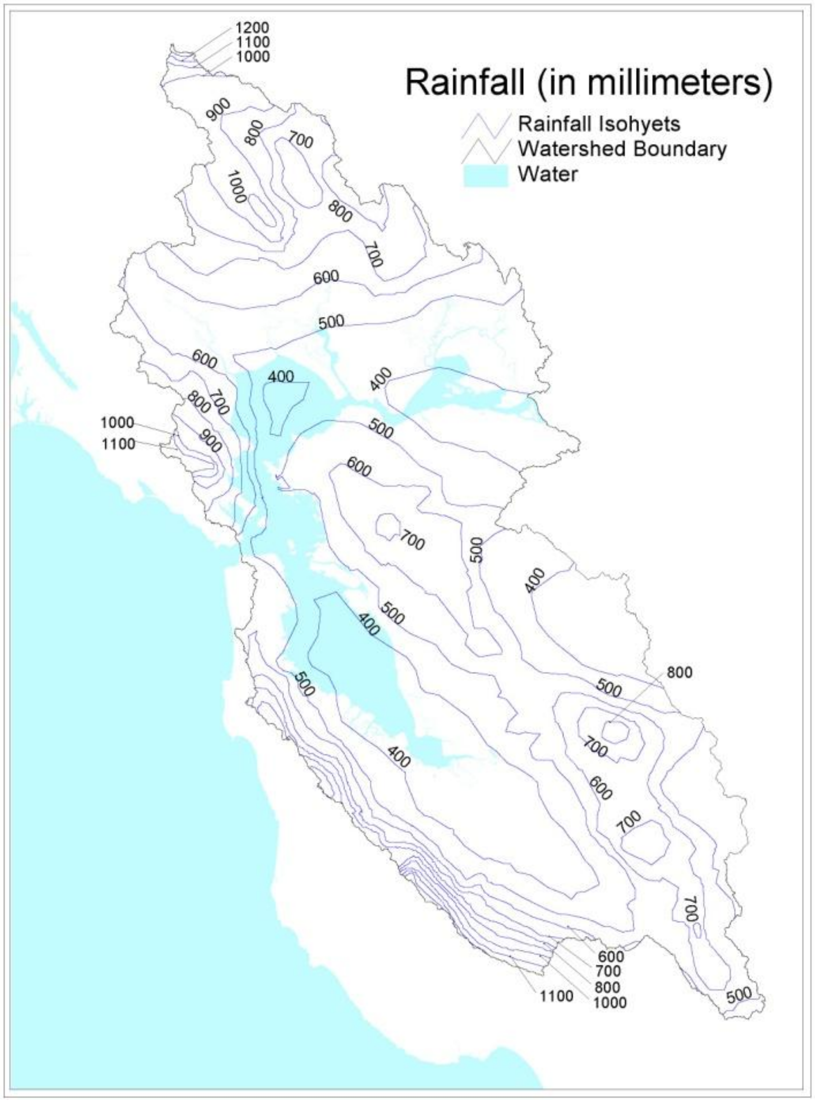


Figure 3.3. Distribution of rainfall (mm) in the Bay Area (after Rantz, 1971).

The estuary itself has an open water surface area of 460 mi^2 (~1200 km^2) at mean sea level (msl) with a maximum depth of 469 ft (143 m) below msl under the Golden Gate Bridge, and an average depth across the estuary of 23 ft (7m) combining to a total msl volume of 8.4 km^3. Tides are mixed semidiurnal with a tidal amplitude (mean high water to mean low water) at San Francisco near the Golden Gate Bridge (NOAA station 18649 established 1854) of 4.1 feet (1.25 m) http://co-ops.nos.noaa.gov/geo.shtml?location=9414290. The latest information from NOAA shows that msl presently rising at a mean rate of 2.01±0.21 mm (0.08±0.008 in) per year. Water temperatures range from about 46-50°F (8-10°C) in the winter to 68-77°F (20-25°C) in the summer. Of interest to both phytoplankton productivity and density gradient driven water fluxes, temperatures at the

Golden Gate are warmer in the winter than in the South or North Bays. In contrast the reverse is true in the summer months (Table 3.1).

Table 3.1. Water temperatures in San Francisco Bay (Source: USGS Surface-Water Data for USA, URL: http://waterdata.usgs.gov/nwis/sw).

Bay Segment	Representative Gage	Winter Mean Low (ºC)	Summer Mean High (ºC)
Lower South Bay	Marker 17	9.5	25
South Bay	San Mateo Br	10	24
Central Bay	Alcatraz	10	19.5
San Pablo Bay	Point San Pablo	9	22
Carquinez Straight	Carquinez	8.5	22.5
Suisun Bay	Benicia Br	8	23

Major components of the freshwater flux into the estuary include precipitation, evaporation, STP effluent influx, river flow and runoff. Smith and Hollibaugh (2006) computed a water budget for the northern and southern segments of the SF Bay for the period 1990-1995 (Figure 3.4). Based on this work it appears that the North SF Bay is overwhelmingly dominated by river inflows and runoff. In contrast, all inputs are important in the South Bay budget (Smith and Hollibaugh, 2006). STP effluent (assumed to be constant) is particularly important in the South Bay, as are evaporative losses in the summertime which sometimes results in net water loss from the South Bay during summer periods. Other important notes include the strong seasonality in runoff between winter and summer months, as well as high inter-annual variability (e.g., 1993 and 1995 are much wetter than the other years). The North and South Bays each exchange water with the Central Bay segment (budget not computed for this segment), which in turn exchanges water with the Pacific Ocean.

San Francisco Bay, like most estuaries, is a complex mix of a variety of habitats which can be conceptually categorized as subtidal, intertidal, and seasonal (fringing) wetlands (locally many of these are diked Baylands) (Figure 3.5). Although, in fact, there is a continuum with multiple subcategories within each, SFEI has mapped the intertidal and diked Baylands. A geographic information system (GIS) geo-referenced map of bathymetry (Figure 3.6) along with substrate character (texture) and habitat types is important for managing and modeling nutrient related water quality, especially, the linkage between nutrient loads and endpoint response. The proportions of habitat and bathymetry vary between Bay segments Table 3.2). The most common habitat is deep-Bay/channel[1] followed by shallow Bay/channel. Historically there were about one third more tidal marshes but this was converted to

[1] Definitions of habitat type.

Deep Bay/Channel:	Bottom is deeper than 18 ft (5.5 m) below MLLW.
Diked Wetland:	Areas of historical tidal marshes that have been isolated from tidal influence by dikes or levees, but which remain primarily wetland features.
Shallow Bay/Channel:	Bottom is entirely between 18 ft (5.5 m) below Mean Lower Low Water (MLLW) and MLLW.
Tidal Flat:	Occurs from below MLLW to Mean Tide Level (MTL) and supports less than 10% cover of vascular vegetation, other than eelgrass. Includes mudflats, sandflats, and shellflats.
Tidal Marsh:	Vegetated wetland that is subject to tidal action.

either dike wetland or salt pond habitat. Today there is a large effort to restore many of the salt pond areas following the South Bay Salt Pond Restoration 30-year Restoration Plan[2].

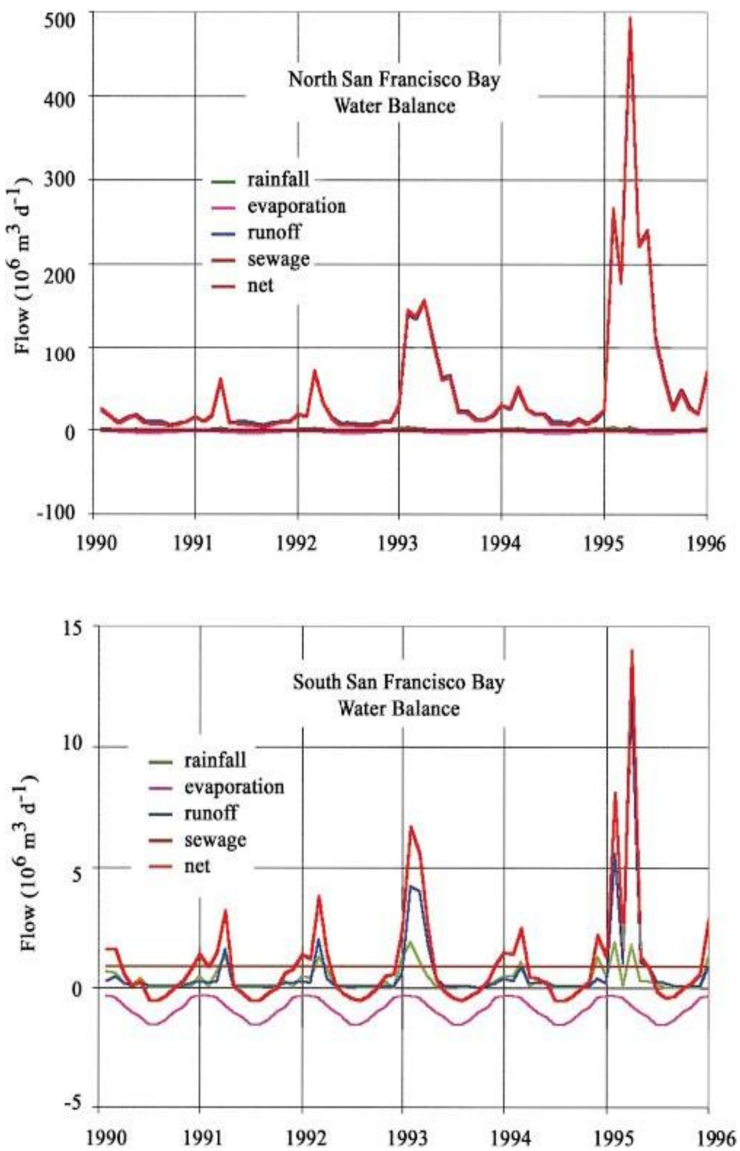


Figure 3.4. Water budget for the northern and southern segments of SF Bay (reproduced without permission from Smith and Hollibaugh, 2006).

[2] South Bay Salt Pond Restoration official website http://www.southbayrestoration.org/

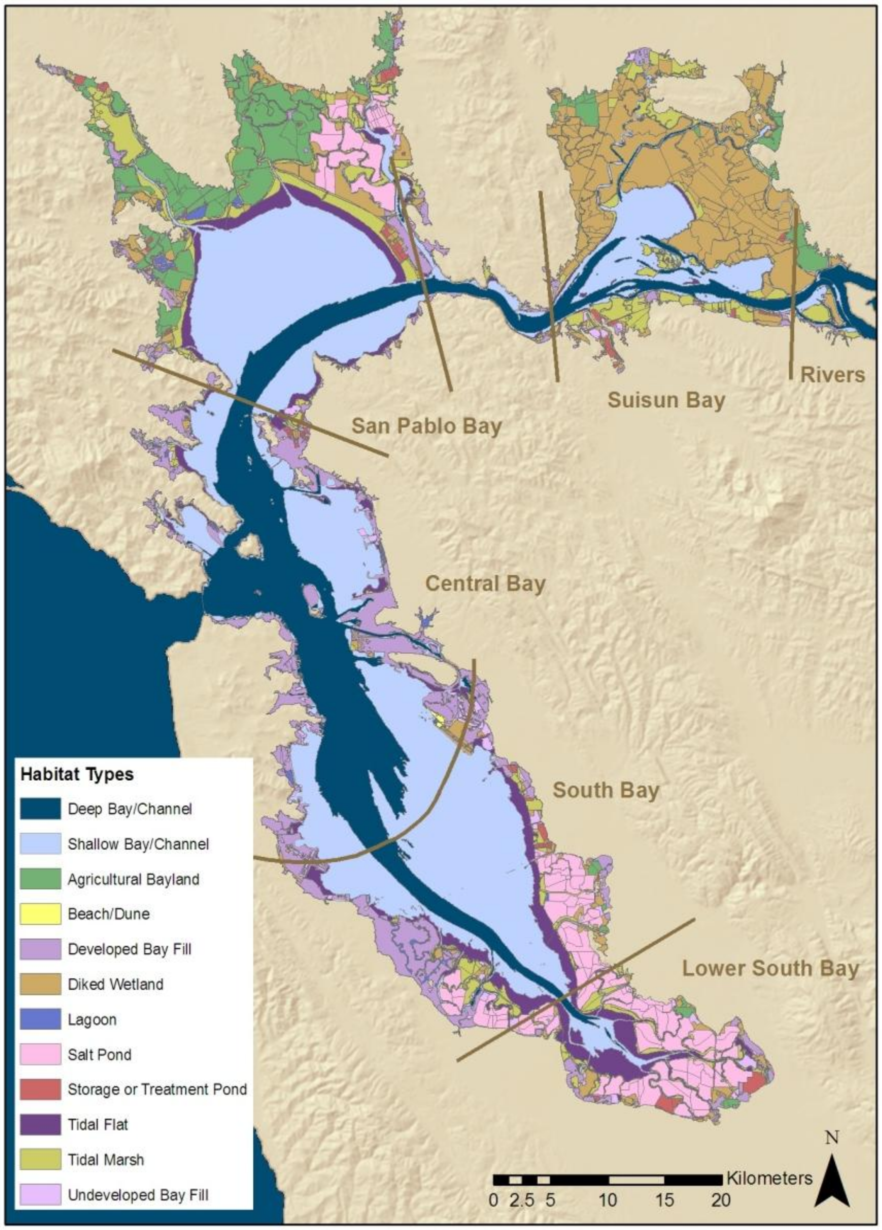
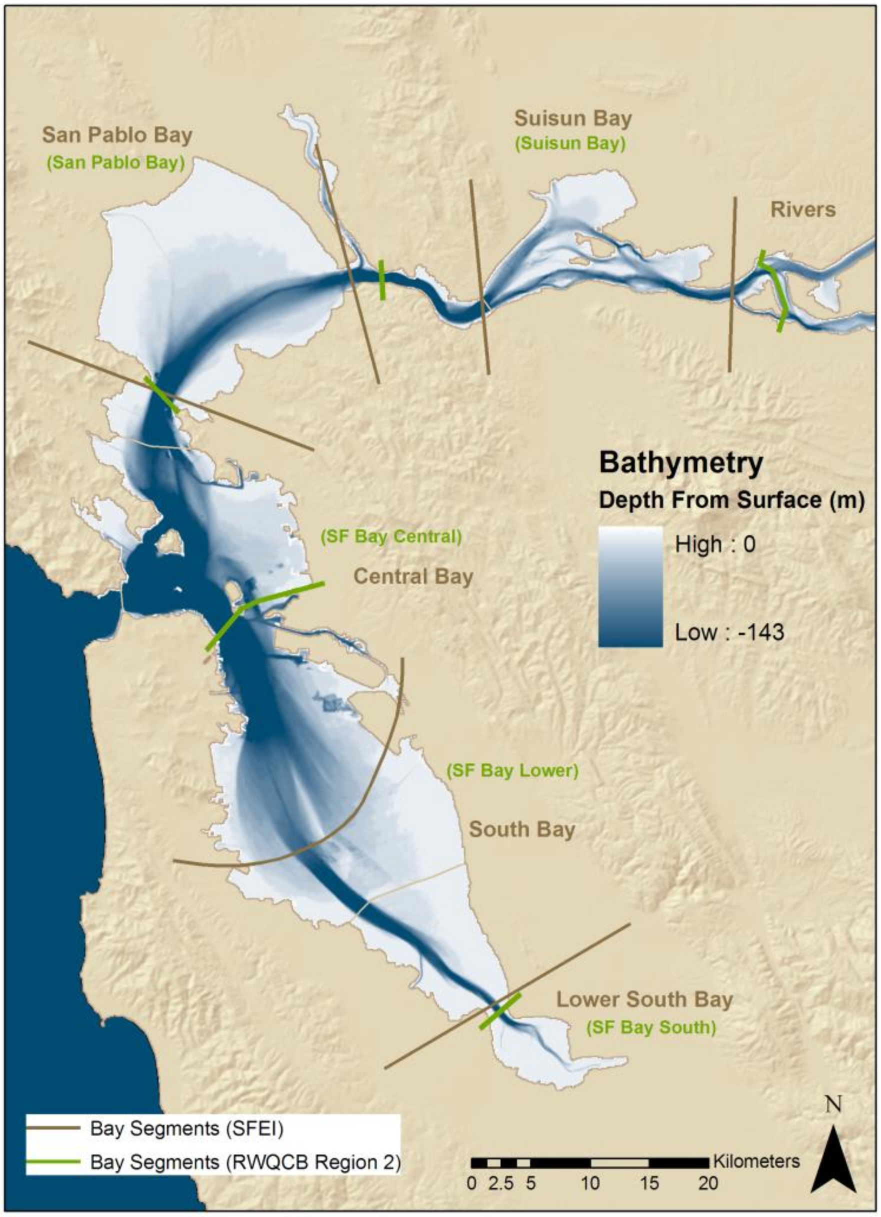


Figure 3.5. Habitat types of SF Bay and adjacent Baylands. Primary sources underlying map data include: CA State Lands Commission, US Geological Survey, US Fish and Wildlife Service, US National Aeronautical and Space Administration, and local experts. Science coordination, GIS and map design by the San Francisco Estuary Institute (1997).

Table 3.2. Relative percent of habitat types by RMP segment for select habitat categories. Diked wetlands are those isolated from tidal influence and dominated by mudflats and marsh. Source: SFEI Ecoatlas: (www.sfei.org/ecoatlas/gis)

	Rivers		Suisun Bay		Carquinez Str.		San Pablo Bay		Central Bay		South Bay		Lower South Bay	
	(km^2)	**(%)**	**(km^2)**	**(%)**	**(km^2)**	**(%)**	**(km^2)**	**(%)**	**(km^2)**	**(%)**	**(km^2)**	**(%)**	**(km^2)**	**(%)**
Tidal Marsh	4.0	19	51	14	5.9	14	61	14	4.5	1.0	20	6.2	19	13
Tidal Flat	0	0	3.7	1.0	2.2	5.5	34	8.1	14	3.1	36	11	28	19
Shallow Bay/Channel	5.6	27	80	22	10	25	212	50	201	45	174	55	13	8.8
Deep Bay/Channel	5.4	26	31	8.5	13	31	39	9.3	216	49	26	8.4	2.4	1.7
Diked Wetland	6.0	28	198	54	5.0	12	35	8.5	5.3	1.2	8.6	2.7	11	7.3
Lagoon	0	0	0	0	0	0	4.0	1.0	2.7	0.6	2.7	0.84	0	0
Salt Pond	0	0	0	0	2.7	6.5	30	7.2	0	0	48	15	68	47
Storage or Treatment Pond	0	0	3.1	0.84	1.9	4.6	4.6	1.1	1.5	0.34	2.3	0.74	5.0	3.4
Total	**21**		**367**		**41**		**420**		**445**		**317**		**146**	



*Bathymetry data does not apply to ocean area.

Figure 3.6. Segmentation and bathymetry on SF Bay (Source: NOAA bathymetric soundings). Green lines show the Regional Water Quality control Board (RWQCB) segmentation scheme and the brown lines show the Regional Monitoring Program for Water Quality (RMP) segmentation scheme developed at San Francisco Estuary Institute (SFEI).

Recently, a series of new maps of habitat types have been developed for sub-tidal areas of the Bay that define soft substrate, rocky outcrops, artificial structures, shellfish beds (Olympia oysters-*Ostrea lurida*, California mussels-*Mytilus californianus*, hybridized Bay mussels-*Mytilus trossulus/galloprovinciallis*), SAV (Widgeon grass-*Ruppia maritima*, Eelgrass-*Zostera marina*), and macro algae-insufficient data for map development (*Ulva spp., Gracilaria pacifica, Fucus gardneri*, and introduced *Sargassum muticum*)[3] (NOAA, 2010).

The Regional Water Quality Control Board divides the SF Bay into seven hydrological segments for regulatory purposes based on CalWater, USGS drainage basin maps and largely defined by major bridges. In 2002, a Regional Monitoring Program Work Group developed a new segmentation scheme based on expert opinion of natural hydrological and ecological boundaries, as well as a cluster and partition analyses using 10 years of sediment and water quality data (Grosso and Lowe, 2002; Lowe et al., 2005). The most dramatic difference in the two segmentation schemes is the division between the Central and South Bay segments. The RWQCB makes this division at the Bay Bridge, while the RMP defines this division at the San Bruno Shoal (Figure 3.6).

3.2 San Francisco Bay Beneficial Uses and Existing Water Quality Objectives Relevant to Eutrophication

The SF Water Board has existing standards for SF Bay estuary, consisting of designated beneficial uses, narrative and numeric water quality objectives, and an implementation plan to achieve these standards. The purpose of this section is to summarize the beneficial uses and water quality objectives relevant to the development of NNEs in SF Bay estuary.

3.2.1 Beneficial Uses

Aquatic ecosystems have many resources, services, and qualities that provide the basis for a variety of benefits to the people of the state. Beneficial uses are designated uniquely to aquatic systems based on resources, services, and qualities. The Water Board's definitions for each of these beneficial uses is listed in Table 3.3 below (see RWQCB Basin Plan, 2011 for more information about each beneficial use category). The Water Board is charged with establishing water quality objectives and discharge limits to protect these beneficial uses from pollution and nuisance (RWQCB, 2011). In general most Bay segments have similar designation however there are some exceptions (Table 3.4). For example, the northern reaches of the Bay (Sacramento-San Joaquin Delta) are managed for freshwater and therefore are designated for agricultural, municipal and freshwater uses. The Sacramento-San Joaquin Delta, Suisun Bay, and SF Bay Central reaches are used for industrial process supply and all reaches south of San Pablo Bay (inclusive) are designated for harvesting shellfish (e.g., clams, oysters, and mussels) for human consumption, commercial, or sport purposes.

[3] Marilyn Latta at the Coastal Conservancy is heading up the Subtidal Goals project for San Francisco Bay.

Table 3.3. Definitions of Beneficial Uses Designated within SF Bay.

OCEAN, COMMERCIAL, AND SPORT FISHING (COMM): Uses of water for commercial or recreational collection of fish, shellfish, or other organisms in oceans, bays, and estuaries, including, but not limited to, uses involving organisms intended for human consumption or bait purposes.

SHELLFISH HARVESTING (SHELL): Uses of water that support habitats suitable for the collection of crustaceans and filter-feeding shellfish (e.g., clams, oysters, and mussels) for human consumption, commercial, or sport purposes.

ESTUARINE HABITAT (EST): Uses of water that support estuarine ecosystems, including, but not limited to, preservation or enhancement of estuarine habitats, vegetation, fish, shellfish, or wildlife (e.g., estuarine mammals, waterfowl, shorebirds), and the propagation, sustenance, and migration of estuarine organisms.

FISH MIGRATION (MIGR): Uses of water that support habitats necessary for migration, acclimatization between fresh water and salt water, and protection of aquatic organisms that are temporary inhabitants of waters within the region.

PRESERVATION OF RARE AND ENDANGERED SPECIES (RARE): Uses of waters that support habitats necessary for the survival and successful maintenance of plant or animal species established under state and/or federal law as rare, threatened, or endangered.

FISH SPAWNING (SPWN): Uses of water that support high quality aquatic habitats suitable for reproduction and early development of fish.

WILDLIFE HABITAT (WILD): Uses of waters that support wildlife habitats, including, but not limited to, the preservation and enhancement of vegetation and prey species used by wildlife, such as waterfowl.

WATER CONTACT RECREATION (REC1): Uses of water for recreational activities involving body contact with water where ingestion of water is reasonably possible. These uses include, but are not limited to, swimming, wading, water-skiing, skin and scuba diving, surfing, whitewater activities, fishing, and uses of natural hot springs.

NONCONTACT WATER RECREATION (REC2): Uses of water for recreational activities involving proximity to water, but not normally involving contact with water where water ingestion is reasonably possible. These uses include, but are not limited to, picnicking, sunbathing, hiking, beachcombing, camping, boating, tide pool and marine life study, hunting, sightseeing, or aesthetic enjoyment in conjunction with the above activities.

NAVIGATION (NAV): Uses of water for shipping, travel, or other transportation by private, military, or commercial vessels.

Table 3.4. Designated beneficial uses for segments of SF Bay based on the 2011 basin plan.

	Human Consumptive Uses											
			Aquatic Life Uses						Wildlife Use	Recreational Uses		
	IND	PROC	COMM	SHELL	EST	MIGR	RARE	SPWN	WILD	REC-1	REC-2	NAV
Suisun Bay	E	E	E		E	E	E	E	E	E	E	E
Carquinez Straight	E		E		E	E	E	E	E	E	E	E
San Pablo Bay	E		E	E	E	E	E	E	E	E	E	E
San Francisco Bay Central	E	E	E	E	E	E	E	E	E	E	E	E
San Francisco Bay Lower	E		E	E	E	E	E	P	E	E	E	E
San Francisco Bay South	E		E	E	E	E	E	P	E	E	E	E

*Adapted from Table 2-1 in the Basin Plan (RWQCB, 2007). Segments listed are those defined by the RWQCB. "E" means existing beneficial use.

3.2.2 Existing Water Quality Criteria Related to Nutrients and/or Eutrophication

SF Water Board numeric and narrative objectives relevant for SF Bay are given in Table 3.5. Water quality criteria specifically for nutrients in surface waters are not defined in the Basin Plan.

Table 3.5. Numeric objectives for constituents related to nutrient over enrichment or eutrophication in SF Bay.

<table>
<tr><th>Constituent</th><th>Numeric Objectives</th></tr>
<tr><td>Ammonia</td><td>The discharge of wastes shall not cause receiving waters to contain concentrations of un-ionized ammonia in excess of the following limits:

<u>Un-ionized ammonia (mg L^{-1} as N)</u>
Annual Median 0.025
Maximum, Central Bay 0.16
Maximum, Lower Bay 0.4

The intent of this objective is to protect against the chronic toxic effects of ammonia in the receiving waters. An ammonia objective is needed for the following reasons: 1) Ammonia (specifically un-ionized ammonia) is a demonstrated toxicant. Ammonia is generally accepted as one of the principle toxicants in municipal waste discharges. Some industries also discharge significant quantities of ammonia, 2) Exceptions to the effluent toxicity limitations in Chapter 4 of the Plan allow for the discharge of ammonia in toxic amounts. In most instances, ammonia will be diluted or degraded to a nontoxic state fairly rapidly. However, this does not occur in all cases, the South Bay being a notable example. The ammonia limit is recommended in order to preclude any buildup of ammonia in the receiving water, and 3) A more stringent maximum objective is desirable for the northern reach of the Bay for the protection of the migratory corridor running through Central Bay, San Pablo Bay, and upstream reaches.</td></tr>
<tr><td>Dissolved Oxygen</td><td>For all tidal waters, the following objectives shall apply:

Location — Tidal minimum (mg L^{-1})
Downstream of Carquinez Bridge 5.0
Upstream of Carquinez Bridge 7.0

Dissolved oxygen is a general index of the state of the health of receiving waters. Although minimum concentrations of 5 mg L^{-1} and 7 mg L^{-1} are frequently used as objectives to protect fish life, higher concentrations are generally desirable to protect sensitive aquatic forms. In areas unaffected by waste discharges, a level of about 85 % of oxygen saturation exists. A three-month median objective of 80 % of oxygen saturation allows for some degradation from this level, but still requires consistently high oxygen content in the receiving water.</td></tr>
</table>

4. Summary of Trends in Nutrient Loading to San Francisco Bay

4.1 Introduction

Development of the NNE framework for the SF Bay requires an accurate understanding of the sources, magnitude and timing of nutrient loads delivered to the Bay. These data are important to properly calibrate our understanding of the biological effects of nutrients on the Bay. It is also important to understand the primary sources and predominant forms of nutrients delivered to the Bay. The purpose of this section is to assess the availability of data and summarize, to the extent possible, the trends in nutrient loading to SF Bay. In most cases it was not possible to find loading information specific to the three major Bay segments (the northern reaches north of the Richmond-San Rafael Bridge, Central Bay between the Richmond-San Rafael Bridge and San Bruno shoals (RMP Central Bay segment) and the southern portions of the Bay south of the San Bruno shoals (RMP south Bay and Lower South Bay segments). Spatial resolution of data overall remains a pervasive gap in current knowledge. In addition, there was generally a lack of understanding of inter-annual variability of nutrient loads. This is of particular concern given that the freshwater inflow to the estuary can vary considerably between dry years and wet years.

4.2 A Primer on Nutrients: Sources and Forms

Nutrients are supplied to SF Bay via a variety of pathways including:

- Atmospheric deposition (both wet and dry) directly to the Bay surface,
- Stormwater from watersheds that drain to the Bay from the nine counties adjacent to the Bay,
- Groundwater from these same tributaries,
- Terrestrial runoff from 37% of the Central Valley via the Sacramento and San Joaquin Rivers,
- Urban wastewater,
- Industrial wastewater, and
- Exchange with coastal ocean (via Golden Gate).

Although each of these pathways is not entirely mutually exclusive (for example atmospheric deposition is probably a large component of urban runoff for some nutrient forms), this section focuses on what passes into the Bay via the main pathways rather than the ultimate source. Should a call for management of nutrient supply to the Bay occur in the future, it will become important to learn more about ultimate sources and the processes that cause the release of and transport of various forms of nutrients into the Bay. In addition, it is important to note that this section focused on "new" sources of nutrients to SF Bay and makes no attempt to account for additional sources or sinks for nutrients within the Bay. As an example, within an estuary, nutrients can undergo a variety of transformations and exchanges among the “compartments” (e.g. water column, sediment, animal and plant biomass, etc.). Nutrients that are deposited to the estuary from a watershed can undergo a series of biological and chemical processes cause the buildup and net release of nutrients (and other compounds) from the sediment pore waters to surface waters in a process known as “benthic flux” (Berner 1980). Net benthic fluxes of nutrients in some estuaries can support a major percentage of primary productivity (e.g. Cowan and Boynton 1996). By the same token, processes such as denitrification can be responsible for the loss of nitrogen from an estuary.

Analytically, nutrients are divided into a number of forms (Table 4.1). Practically, in terms of estimating nutrient loads in relation to standing nutrient concentrations and impacts to beneficial uses in SF Bay, a nutrient budget should primarily focus on total nitrogen and total phosphorus and the main dissolved inorganic species of each. The organic components for nitrogen can then be derived by subtraction using the equation that follows Table 4.1.

Table 4.1. Nutrient species relevant to estimating nutrient loads in relation to standing nutrient concentrations and impacts to beneficial uses in San Francisco Bay.

	Nitrogen	**Phosphorus**
Dissolved Inorganic	Nitrate (NO_3^-) + nitrite (NO_2^{2-}) collective called NO_x almost wholly in the dissolved phase	Phosphate (PO_4^+) mostly in dissolved phase but also adsorbs readily to particles
	NH_3/NH_4^+ (in a dynamic equilibrium in natural waters influenced mainly by temperature and pH)	
Dissolved Organic	Dissolved organic nitrogen (often a large portion of total nitrogen in natural waters especially those less impacted by human activities)	Dissolved organic phosphorus (can be a large portion of total phosphorus in natural waters unless impacted by human activities or there is a natural source of phosphate from mineral or animal (guano) origin
Particulate	Particulate organic nitrogen (detritus left from pieces of undecayed or partially decayed organic matter)	Particulate organic phosphorus (detritus left from pieces of undecayed or partially decayed organic matter)
	Particulate inorganic nitrogen (insignificant in natural waters and usually not considered)	Particulate inorganic phosphorus (PO_4^+ sorbs readily to inorganic and organic particle; also associated with minerals)

Organic nitrogen = Total nitrogen – (Nitrate+nitrite (NO_x)) –ammonium (NH_4) (making the reasonable assumption that negligible inorganic nitrogen is particulate)

or from laboratory analysis of Total Kjeldahl Nitrogen (TKN) which is the sum of organic nitrogen, and ammonium (NH_4^+)

Organic nitrogen = TKN – NH_4^+

Similarly, total nitrogen can be determined by the addition of concentrations found in analyzed natural water samples:

Total nitrogen = TKN + NO_x

Organic forms are typically only a small portion of total phosphorus. As such, in most cases, literature describing studies of phosphorus in watersheds and estuaries largely ignores organic forms. That said, with effort, all forms of phosphorus can be determined and relate via the following equation:

Total phosphorus = dissolved inorganic phosphorus (DIP, phosphate, PO_4^{2-}) + dissolved organic phosphorus (DOP) + total particulate phosphorus (TPP).

Practically, quantification of these forms is made using just two methods, the molybdate blue method for phosphate and the persulfate method applied to filtered samples and whole water samples.

4.3 Freshwater budget for the Estuary

Freshwater enters the Estuary predominantly via freshwater flow from the Central Valley and from flow from smaller tributaries in the nine-county Bay area (Figure 4.1). Freshwater flow from the Central Valley via the Delta dominates (89%) and flow from the smaller tributaries in the nine-county Bay Area is about double that of wastewater input. Flows are highly variable. For example, annual flow into the Bay from the Central Valley via the Delta varied by 26 times between wetter years and drier years from 1971-2000 (McKee et al., 2006). Daily inflow from the Delta is even more variable ranging from near zero to 1,540 million m^3 on which occurred on February 20th, 1986 (Figure 4.2). In order to measure accurate loads of any contaminant of interest including nutrients, it will be important for future studies to focus on capturing data during high flow events when daily flow exceeds about 40,000 cfs (98 million m^3/day) (e.g. David et al., 2009 who discussed monitoring design in relation to mercury).

Flow from local small tributaries is much more difficult to qualify given there are more than 250 individual drainages that flow to the Bay within the nine-county Bay Area. Recently SFEI has developed a 5-station index for the South Bay south of the Bay Bridge and an 3-station index for the North Bay north of the Bay Bridge (L. McKee unpublished). These indexes were developed for the period Water Year 1971 to 2010 (40 years) and adjusted using average annual flow from a calibrated rainfall-runoff model developed for the whole watershed of the nine-county Bay Area (Lent and McKee, 2011). Based on this analysis, annual flow from the small tributaries south of the Bay Bridge has varied from 84-2,419 million m^3 and maximum daily flow was 121 million m^3 on February 19th, 1986 (Figure 4.3) and annual flow from the small tributaries north of the Bay Bridge has varied from 16-2,911 million m^3 and maximum daily flow was 348 million m^3 on February 17th, 1986 (Figure 4.3). It is interesting to note that flow from northern watersheds peaked on a different day in the northern watersheds although all were wet for a full 7 days of heavy rain during, this, the largest storm in the past 40 years. It is also interesting to note that the maximum daily discharge entering the Bay from the Central Valley via the Delta is of the same magnitude at the average annual flow from the small tributaries in the nine-county Bay Area (1,589 million m^3 for the period WY 1971-2000).

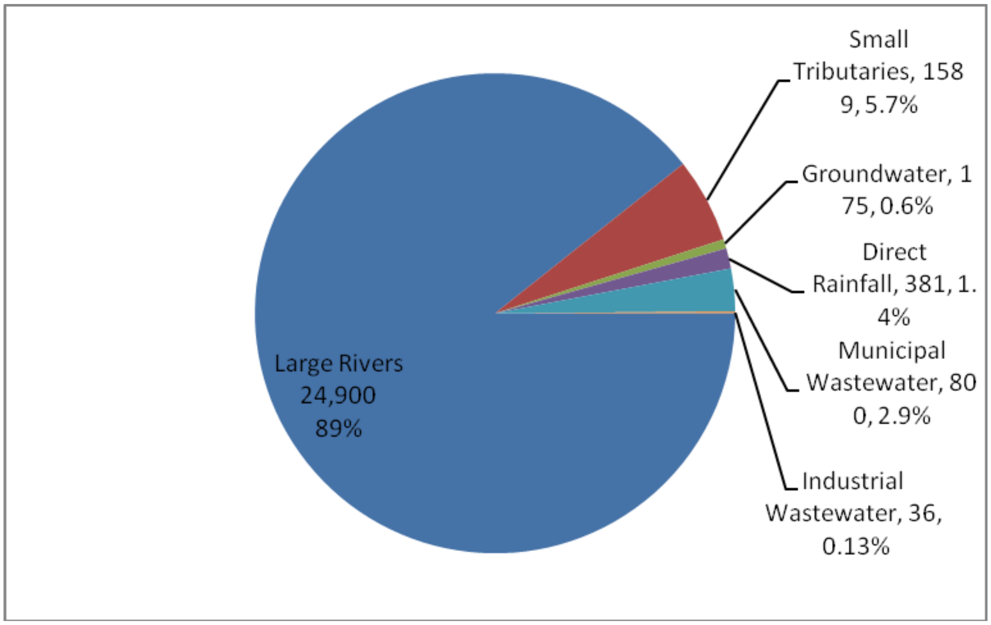


Figure 4.1. Relative flow from each of the freshwater main sources to the Bay (million m^3 per year).

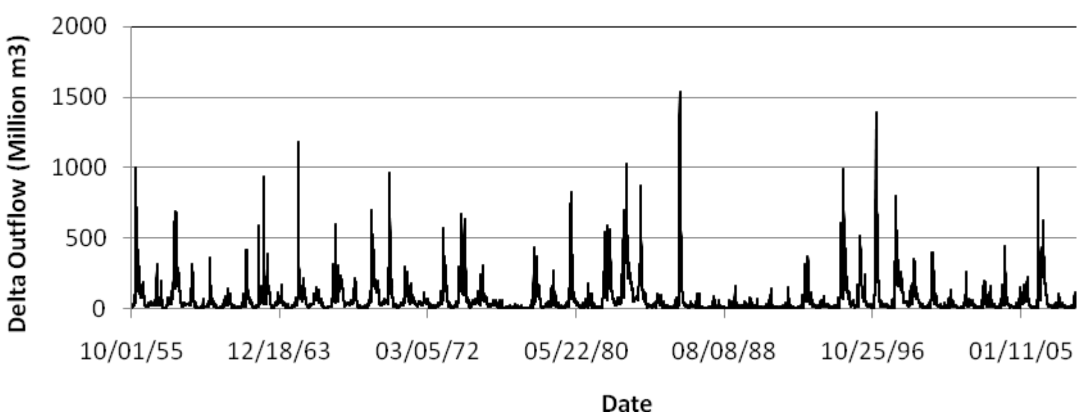


Figure 4.2. Daily Delta outflow from the Dayflow model (Source DWR website: http://www.water.ca.gov/dayflow/output/Output.cfm).

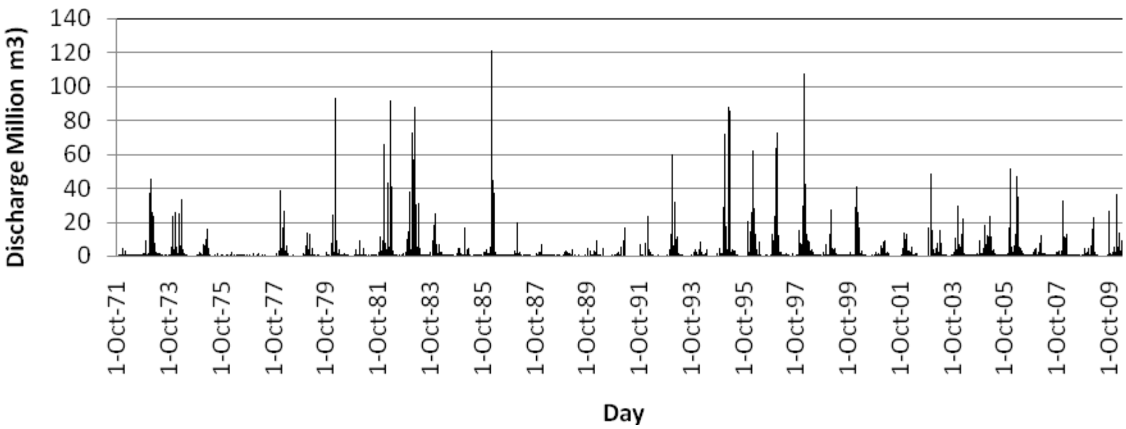


Figure 4.3. Daily flow from the local tributaries in the nine-county Bay Area to the South Bay south of the Bay Bridge based on an 5 station index (Dry Creek at Union City, Alameda Creek at Niles, Guadalupe River at Hwy 101, San Francisquito at Stanford University, and Saratoga Creek at Saratoga) adjusted to the annual average flow (586 million m^3) for water years 1971-2000 (Lent and McKee, 2011).

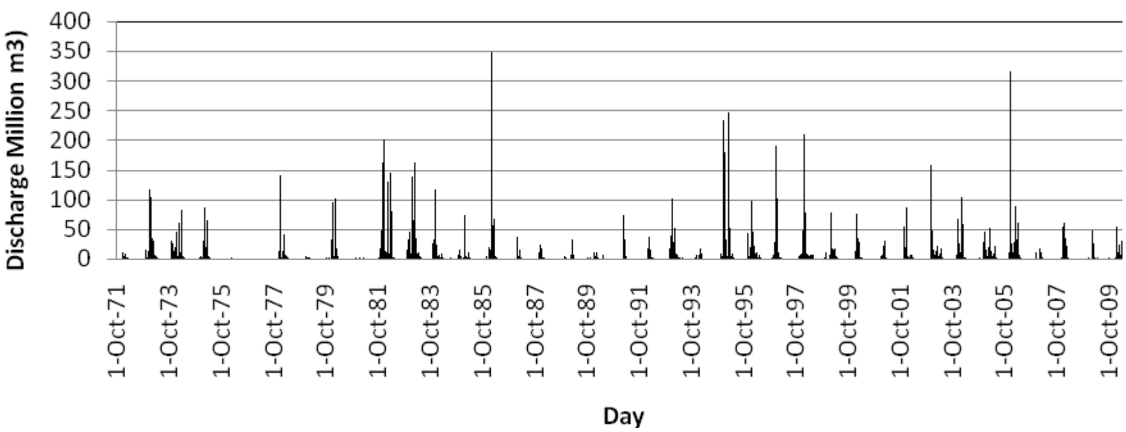


Figure 4.4. Daily flow from the local tributaries in the nine-county Bay Area to the North Bay north of the Bay Bridge based on an 3 station index (Novato Creek at Novato, Napa River near Napa, and San Ramon Creek at San Ramon) adjusted to the annual average flow (1,003 million m^3) for water years 1971-2000 (Lent and McKee, 2011).

4.4 Atmospheric Nutrient Loads Direct to the Bay Surface

Nutrients derived from atmospheric deposition to estuary surfaces has gained attention especially in oligotrophic (low nutrient) systems or systems with a small watershed area to surface area ratio where direct load to the surface may be a larger portion of the overall annual loads. In addition, there is some evidence that atmospheric derived nitrogen may be more bioavailable than terrestrially derived loads (Paerl, 1995). There is a wide variety of methods used to collect and analyze nutrient atmospheric deposition with some studies collecting wet deposition only and others focusing on dry deposition. The portion of deposition in wet and dry loads is variable. For example, in the case of nitrogen, perhaps only an additional 20% is associated with dry deposition over the ocean (Paerl, 1995). In contrast, Jassby et al. (1994) reported dry deposition on Lake Tahoe comprising 28% of nitrate input, 33% of NH_4^+ input, 70% of phosphate input, 58% of total nitrogen input (equivalent to 1.4:1 dry:wet) and 70% of total phosphorus input (equivalent to 2.3:1 dry:wet). The estimates of Kratzer et al. (2010) for the Central Valley ranged between 1.7-2.8 dry:wet for total nitrogen deposited on land surfaces.

There was only one previous estimate of nutrient deposition to SF Bay. Russell et al. (1980) estimated an annual wet and dry deposition of total nitrogen and total phosphorus of 980 and 120 metric tons (mt) respectively (Table 4.2) however they did not disclose the source of data or methods for their calculations. These estimates were made for the whole Bay only and are not spatially resolvable. Normalizing them to the whole area of the Bay (1,200 km^2), Russell et al.'s estimates are equivalent to 817 and 100 kg km^{-2} y^{-1}. Comparison of these measurements to those in Lake Tahoe provided by Jassby et al. (1994) (562 kg N km^{-2} y^{-1} and 32.6 kg P km^{-2} y^{-1}) suggest that the estimates of Russell et al. (1980) might be reasonable. Recently, estimates were made for the Central Valley of between 1.18 (1987) and 3.55 (1998) tons mi^{-1} y^{-1} (Kratzer et al., 2010) equivalent to 413-1,243 kg N km^{-2} y^{-1}. These also bracket Russell's estimates. However, population in the Bay Area has increased by 31% from 1980-2000 (2010 census data pending), vehicle miles traveled has also increased, laws regarding vehicle emissions have improved, industrial land use has decreased, and trends in fossil fuel combustion for home and office heating have undoubtedly occurred. All these changes in particular likely render previous estimates of nitrogen deposition outdated. That said, in comparison to currently available nutrient loads from wastewater and stormwater, atmospheric loads appear to be about 5% of the annual average load. It is recommended that recent data on nitrogen and phosphorus in wet and dry deposition from western US cities (Los Angeles, Portland, and Seattle) be reviewed and used to make more up-to-date estimates for the Bay Area.

4.5 Nutrient Loads from the Delta via Delta Outflow to the Bay

Nitrogen and phosphorus loads entering the Bay from the Delta have been estimated by six authors. Russell et al. (1980) estimated annual inputs of total nitrogen and total phosphorus of 13,000 and 2,400 mt respectively for 1978 but did not describe their calculation methods or data sources. Russell et al. speculated that loads would decrease due to the balance of sediment load trends, continuing changes in population and wastewater treatment and changes to agricultural drainage water practices. Jassby and

Cloern (2000) made an estimate of total organic nitrogen load from the Central Valley to the Bay of 6,205 mt. Smith and Hollibaugh (2006) made an estimate of total dissolved inorganic nitrogen (NO_x+$NH_{3/4}$) and phosphate entering the Bay from the Delta of 13,404 and 1,880 mt per year respectively. Russell's estimates are dwarfed by those of Davis et al. (2000) who combined concentration measurements from the SF Bay Regional Monitoring Program (RMP) collected during base flow conditions at the most upstream Bay locations with average annual Delta outflow. These estimates were 45,200, 5,100, and 6,400 mt for nitrate, ammonium, and phosphate respectively. Through a large data synthesis exercise to support management of drinking water supply in the Delta, Heidel et al. (2006) estimated monthly nutrient loads exported from the Delta by combining monthly Dayflow Delta outflow with total nitrogen (sum of nitrate, nitrite, and TKN) and total phosphorus concentrations. They estimated loads of 7,435 metric t for dry years and 30,885 metric t for wet years for total N and 1049 metric t and 4473 metric t for total P for dry and wet years. Most recently, Kratzer et al. (2010) reported loads based on a thorough compilation of data collected through the U.S. Geological Survey's National Water Information System database, the California Department of Water Resources, the University of California at Davis, and the U.S. Environmental Protection Agency's STOrage and RETrieval database. They estimated loads at Freeport on the Sacramento River and at Vernalis on the San Joaquin River from 1974 to 2004. Taking the average for and summing the two stations (assuming no storage or losses between these stations and the head of the Bay at Mallard Island) the loads in metric t were nitrate (6,593), ammonia (1,857), total nitrogen (16,642), phosphate (1,130), and total phosphorus (2,635). These loads are equivalent to a flow weighted average of 0.265 mg L^{-1} nitrate, 0.075 mg L^{-1} ammonia, and 0.045 mg L^{-1} phosphate assuming an annual average flow of 24,900 million m^3 (McKee et al., 2006). These concentrations are very similar to averages calculated from the RMP monitoring at the head of the estuary (sites BG20 and BG30) (nitrate: 0.286 mg L^{-1}; phosphate: 0.069 mg L^{-1}).

Comparing all these estimates (Table 4.2) it can be seen that the available estimates to do not make a lot of sense. For example, the estimates of dissolved forms of nitrogen (Davis et al., 2000; Smith and Hollibaugh, 2006) are greater than the estimate of total nitrogen load by Russell et al. (1980). Similarly, the estimate of phosphate load by Davis et al. (2000) is about 3 times greater than the estimate by Smith and Hollibaugh (2006) and greater than the total phosphorus load estimate that Russell et al. made. Some of these discrepancies are probably due to temporal trends, however, in truth, no data used by these authors were collected during high flow or for the purposes of calculating loads.

The RMP has continued to collected data in the northern segments of the Bay. While these data are not collected during wet weather flow, they can be used to make more up-to-date estimate of nutrient loads during the dry season that are relatively accurate; wet season loads estimate can be improved by careful manipulation of the data taking into account knowledge about sediment transport (McKee et al., 2006). It is recommended that some effort be put into making these improved estimates as an interim measure to help support immediate planning efforts and decisions about priority information development.

In the medium term, to support the development of a hydrodynamic model on estuarine nutrient response, it is recommended that wet weather data collection of nutrients be initiated at the DWR sampling location at Mallard Island at the head of Suisun Bay. Nutrient forms monitored should include nitrate, ammonium, total nitrogen, phosphate and total phosphorus. Given the existence of a long term turbidity data set at Mallard Island (supported by the USGS) and the likelihood that total phosphorous and total nitrogen correlate at least to some extent with turbidity, just a few years of data will likely support a reasonable estimate of daily loads during wet and dry seasons for the period Water Year 1995 – present if we make the assumption that nutrient loads are not trending. Given the size of the Sacramento River system and the fact that it can take many days to weeks for a flood wave to pass down the system, a daily time step is entirely sufficient for describing loading dynamics at the head of the estuary. Sampling and interpretation methods have been developed by McKee et al. (2006) and further refined by David et al. (2009). These methods could be augmented with automated sampling technology.

4.6 Nutrient Loads from tributaries in the Nine-County Bay Area

There have been several annual scale estimates of nutrient loads entering the Bay via urban and non-urban tributary flow emanating from the nine counties that fringe the Bay. Russell et al. (1980) estimated that approximately 2,300 and 470 mt of total nitrogen and phosphorus was entering the Bay on average in 1978 and suggested that there would likely be no change into the future. The estimate for total nitrogen appears consistent with the estimate of 1,500 mt of nitrate per year made by Davis et al. (2000) in contrast to the estimate of Smith and Hollibaugh (2006) for total dissolved inorganic nitrogen (nitrate and ammonium/ammonium) of just 245 mt. The estimates for phosphate (34 mt) are similarly not in agreement with other estimates (Table 4.2). Smith and Hollibaugh's estimates were made in the absence of any data on nutrient concentrations in local tributaries and were based on assuming concentrations in urban runoff were the same as Delta outflow. In addition, estimate of water flow from small tributaries from around the Bay were challenged by a lack of data. There have been a number of reliable spatially resolved estimates made of flow associated with small tributaries in the nine-county Bay Area (Russell et al., 1980; Davis et al., 2000; McKee et al., 2003; Lewicki and McKee, 2009; Lent and McKee, in preparation). For the most part, these authors have reported an annual average flow of approximately 1,000 million m^3 per year. However, runoff from Bay Area tributaries is very well understood based on a number of currently well maintained USGS and county operated gauges. In addition, Gilbreath and McKee (2010) collated runoff data from 44 urban stormwater pump stations; a data set that could be continually maintained. In addition, nutrient data have been collected in the Napa River and Sonoma Creek watersheds (McKee and Krottje, 2005), Pinole Creek watershed (Pearce et al., 2006), Cerrito Creek and Ettie Street pump station watershed (EBMUD, 2010), and in Zone 4 Line A watershed (Gilbreath and McKee et al., in preparation) and perhaps others. It is recommended that these data be used to make new estimates of nutrient loads for tributaries entering the Bay from the urbanized counties around the Bay to support planning efforts to prioritize new information development. Depending on the data needs of an estuarine nutrient response model, new data collection may need to be initiated to support either improved empirical loads calculations or the development of a watershed loads model with outputs at needed at potentially an hourly if not daily time step.

4.7 Nutrient Loads from Municipal Wastewater

Modern sewage treatment practices are designed to remove solids, biological oxygen demand, and pathogens during primary and secondary treatment phases. During the first two phases some phosphorus and, to a lesser extent, nitrogen is removed through sedimentation, but greater nutrient removal is achieved through tertiary treatment. On average, about 871 million m^3 of wastewater is currently discharged to the Bay annually (Oram et al., 2008), just 10% less than that of stormwater discharge (McKee et al., 2003). There have been three estimates of nutrient loads from wastewater. Smith and Hollibaugh (2006) remarked on the importance of wastewater nutrient loads; they found that wastewater accounted for 50% of the wet season nutrient loads and 80% of the dry season loads in the South Bay. They collated flow information from 12 wastewater treatment plants and nutrient concentration data for five of the larger plants for the period 1990-1995 and interpolated the data to make estimates for all of the plants. Using these data, estimates of 5,983 and 1323 mt of dissolved inorganic nitrogen ($NOx+NH_{3/4}$) and dissolved inorganic phosphorus (DIP or PO_4) were made were made for the South Bay and 1,994 and 230 mt of DIN and DIP were made for the North Bay. These estimates appear to be similar to those of Davis et al. (2000) for nitrogen and about 4 times lower for phosphate. In contrast, the load estimates for total nitrogen by Russell et al. (1980) appear to be much greater (given most nitrogen discharged after secondary treatment is likely to be nitrate and ammonium). In contrast, the total phosphorus loading estimate of Russell et al. is 10 to 100 times lower. Again these numbers are not making sense; there are large discrepancies between authors and partitioning between total and dissolved phases are not logical.

Table 4.2. Published nutrient loading estimates for San Francisco Bay (mt). Note these estimates are mostly based on very limited data assembled from monitoring programs that were not designed for estimating mass loadings.

Source or Pathway	Author	Bay Segment	Time period of estimate	Total Nitrogen (TN-N)	Nitrate + Nitrite (NOx-N)	Ammonium (NH4-N)	Total Inorganic Nitrogen (TIN)	Total organic nitrogen (TON-N)	Total phosphorus (TP-P)	Phosphate (PO4-P)
Aerial deposition	Russell et al., 1980	Whole Bay	1978	980					120	
		Whole Bay	2000 (Authors estimate)	980					120	
Delta outflow	Russell et al., 1980	North Bay	1978	13,000					2,400	
		North Bay	2000 (Authors estimate)	78,000					1,600	
	Heidel et al., 2006	North Bay	Wet year	30,885					4,473	
	Davis et al., 2000	North Bay	Average year		45,200	5,100				6,400
	Jassby et al., 1993	North Bay	Average year (1980 estimate)							
	Jassby and Cloern, 2000	North Bay	Average year (1978-91 estimate)					6,205		
	Smith & Hollibaugh, 2006*	North Bay	Average (1990-95)				13,404			1,880
	Kratzer et al., 2010	North Bay	Average (1974-04	16,642	6,593	1,857			2,635	1,130
Local small tributaries (Urban + non-urban stormwater)	Russell et al., 1980	Whole Bay	1978	2,300					470	
		Whole Bay	2000 (Authors estimate)	2,400					480	
	Davis et al., 2000	Whole Bay	Average year		1,500					510
	Smith & Hollibaugh, 2006*	South of the Richmond Bridge (Central and South Bays)	Average (1990-95)				245			34
Waste water	Russell et al., 1980	Whole Bay	1978	21,000					10	
		Whole Bay	2000 (Authors estimate)	24,000					15	
	Davis et al., 2000	Whole Bay	Average year		3,110					970
	Smith & Hollibaugh, 2006*	South of the Richmond Bridge (Central and South Bays)	Average (1990-95)				5,983			1,323
		North Bay	Average (1990-95)				1,994			230

* Converted from moles to mass using a molecular weight of 14.01 g per mol for N and 30.97 g per mol for P.

The local Water Board issues permits effluent limits to wastewater agencies. In response to these permits, a number of data sets on both flow rates and nutrient concentrations have been generated in recent times. We are currently aware that 15 of the roughly 40 treatment plants in the Bay Area have data available for ammonium concentrations on a monthly basis. Six of these 15 data sets are for systems with tertiary treatment; four of these six measure nitrate. In one case (Fairfield-Suisun WWTP) there are also data for organic nitrogen, total nitrogen, and total phosphorus.

It is recommended that available data be combined with flow data for each of the plants to make new estimates of nutrient loads taking into account treatment methods and population trends. It should be possible to make estimates for annual and wet and dry season loads for at least the last 10 years with reasonable confidence and for the last 20 years with lower confidence for ammonium, nitrate and with overall lower confidence for phosphate. In addition, all treatment plants that discharge to the Bay should be encouraged to begin analyzing effluent for total and dissolved inorganic nutrients and to submit these data to the SFRWQCB on a regular basis. Finally, it is recommended that the POTWs conduct a laboratory inter-comparison on nutrient methods to assure comparability of estimates.

4.8 Loads from Industrial Dischargers

Presently there is no estimate for nutrient loads for industrial discharges to the Bay. For the most part, industrial waste is not treated on site but rather introduced to the municipal sewer system and treated by the local wastewater treatment plant. However, in a few cases treatment is performed on site and treated wastewater is discharged to the Bay. Examples include the oil refineries and C&H sugar (Table 4.3). While we do know that these industries have characterized their effluent streams in the 1990s, we are not aware if there is more recent data available or if their reuse practices have changed in the last 15 years. It is recommended that a request be made to the industrial dischargers of the Bay Area to provide the latest data on flow and concentrations of nutrients in their waste effluent streams.

Table 4.3. Industrial dischargers in the Bay Area with data from the 1990s on flow and nutrient concentrations.

Facility	Volume (MGD)	Treatment type
C&H Sugar	1	Activated sludge
Tosco Corp. at Avon	5	Pond/RBC/carbon
Tosco Corp. at Rodeo	3	Pond/RBC/carbon
Shell Oil Company	6	Activated sludge/carbon
EXXON	3	Activated sludge/carbon
Chevron U.S.A.	8	Activated sludge/wetland

4.9 Nutrient Loads from Groundwater

Nutrient loads entering the Bay from groundwater sources are not available. A number of drinking water supply agencies in the Bay Area monitor losses from their groundwater recharge systems via seepage to the Bay (SFPUC, 1997; Hanson et al., 2004; Thomas Neisar pers. comm., 2010; Muir, 1996 cited in Water Board, 2010). Based on these four study areas, it is estimated that groundwater discharge occurs at an average rate of 0.7 Mm^3 per km shoreline length per year. The perimeter of the Bay is approximately 250 km thus ground water discharge for the whole Bay is estimated to be 175 Mm^3 or about 17.5% of the surface water discharge. Given the extensive use of ground water recharge in the Bay Area for drinking water supply, the use of extensive landscape irrigation which maintains dry-weather flow in our urban drainage systems, and the presence of large alluvial deltas at the months of our larger urban tributaries that ring almost the entire Bay margin (Alameda Creek, Coyote Creek, Guadalupe River, San Francisquito Creek, Novato Creek, Petaluma River, Sonoma Creek, Napa River, Green Valley Creek, Walnut Creek, San Pablo/Wildcat Creeks, San Leandro Creek, and San Lorenzo Creek), this portion of groundwater discharge seems believable despite our clay soils nearer the surface.

Nutrient concentrations have been measured by the USGS in 79 wells tapping the ground water systems of the Bay Area (Ray et al., 2009). Data is available for ortho-phosphorus (phosphate or DIP) for all 79 wells, whereas data are sparser for ammonium, and NOx (Table 4.4). It can be seen that nitrate concentrations are very high in our groundwater systems whereas ammonium and phosphate are at lower concentrations. The nitrate concentrations in this study are not dissimilar to those observed in the groundwater basins of Sonoma and Napa Counties where maximum concentrations of nitrate of 5.2 mg L^{-1} were observed (Kulongoski et al., 2010; USGS Scientific Investigations Report 2010–5089). Combining median concentrations with estimates of groundwater flow provides first order estimates of nutrient loads to the Bay from groundwater (Table 4.4). The load of nitrate in groundwater moderately large relative to other pathways and is greater than the nutrient loads estimate for small tributaries made of Smith and Hollibaugh (2006). However, as mentioned, the estimates by these authors were based on very limited data and assumptions. Loads of ammonium and phosphate are estimated to be small relative to other pathways. Given its overall magnitude of these groundwater estimates in comparison to other pathways, further work may not be a high priority. It is recommended that we seek expert review from the USGS groundwater section as part of the decision making and prioritization process for any next steps with regards to groundwater flows and loads of nutrients to the Bay.

Table 4.4. Nutrient concentrations and loads estimate for San Francisco Bay based on median concentrations found in groundwater of 79 wells in the Bay Area (Ray et al., 2009) and an estimate of groundwater discharge to San Francisco Bay of 175 million m^3 per year.

	Ammonium (mg L^{-1})	NOx (mg L^{-1})	PO4 (mg L^{-1})
Count (n)	22	66	79
Minimum	0.017	0.05	0.006
Maximum	3.88	12.7	1.27
Mean	0.488	3.38	0.102

Median	0.099	3.01	0.051
Load estimate (mt per year)	17	530	8.9

4.10 Exchange with Coastal Ocean

Nutrients and biogenic materials pass in and out of estuaries in response to tides and freshwater forcing. It is well known that in systems like SF Bay which have seasonal freshwater patterns, the net flux during the wet season is from estuary to ocean (e.g., McKee et al., 2000). In contrast, during the dry season, net flux for some nutrient forms (e.g., organic nitrogen) can be from the ocean into the estuary (McKee et al., 2000) and this can be enhanced during upwelling events when nutrient concentrations (particularly phosphorus) in the coastal ocean can be enhanced. Over the years there have been a number of measurements made of water and salt flux through the ocean boundary of the Bay known as the Golden Gate (e.g., Largier, 1996; Fram et al., 2007). There have been no estimates of nutrients flux in this x-section that we are aware of. However, one study (Martin et al., 2007) did quantify chlorophyll a flux during a neap and spring tide during wet season runoff (March 2002), summer upwelling (July 2003), and autumn relaxation (October 2002). The found that that net flux (advective + dispersive) was large and net seaward during the wet season observations, large and net landward during the summer observations and small and indiscernible from zero in the autumn. It is this very type of outcome that could be enhanced to build a statistical relationship between hydrological forcing and flux conditions (e.g., McKee et al., 2000). In their case, they sampled during spring and neap tides during wet season, mid and late dry season conditions (upwelling) and during three flood events of a range of sizes and use the data to build a statistical understanding between freshwater flow and season and net flux. Therefore we recommend that a data set be developed during the next deployment of ACDP instrumentation that quantifies the nutrient concentrations in any surface layer and the bottom layer in the x-section every 1-1.5 hours for 25 hours during spring and neap tides. This should be repeated for a range of seasonal and flow conditions. Alternatively, a data set that captures the annual variability in the ocean-estuary gradient could be combined with the estimates of exchange coefficients (Fram et al., 2007; Martin et al., 2007) to define a net nutrient flux (Mark Stacey, UC Berkeley, personal communication, March 2011). Stacey suggests that the nutrient data set should include samples from a few depths along a line from Central Bay out to the Gulf of the Farallones perhaps monthly, but ensuring that the samples are consistently collected on the same tidal phase (like the USGS Polaris cruises) but final design of a sampling program would need to be the subject of a workshop that would include a number of local experts. Ultimately the data set collected should be suitable for both immediate flux estimates based on either statistical or event modeling and would provide data to support the calibration and verification of the ocean boundary of a system scale hydrodynamic model.

4.11 Summary and Recommendations

SF Bay is regarded as a nutrient enriched estuary, based on the ambient concentrations and estimated loads of nutrients to the Bay (Cloern and Dugdale, 2010). As discussed in this section, estimates of nutrients loads from external sources and pathways are poorly understood. For the most part, published load estimates are outdated by one or even two decades and were either based on data collection methods that were not designed for loads estimation, were based on assumptions that provided guesses at best or were based on data sets that have now been substantially improved with ongoing collection through time. Given changes to wastewater treatment technologies, increases in population, changes to land use, home heating methods, pet husbandry, fertilizer use in agricultural and urban areas, and other factors that influence nutrient loads, it would seem likely that nutrient loads are changing through time. However, data sets are of limited use to make any suggestion of the overall effect of these factors on nutrient load trends through time.

In order to develop models that provide a linkage between indicators of SF Bay health in relation to nutrient enrichment and the nutrient management knobs that can be turned, accurate estimates of nutrient loads are needed with sufficient temporal and spatial resolution. Given the magnitude of the nutrient loads from the Central Valley, wastewater, and stormwater, it is recommended that these pathway a major focus; loads from atmospheric deposition and groundwater are much smaller and together constitute no more than 10% of the total loads to the Bay and thus should receive a smaller emphasis. Table 4.5 provides a summary of data gaps and recommended next steps. Recommendations generally fall into two categories:

1) Revising and updating estimates of nutrients from the different sources, based on existing data
2) Identification of data needed to develop a dynamic watershed model.

The exercise of revising and updating estimates of nutrients from the various sources, based on existing data would help to better inform our understanding of the dominant nutrient sources for each distinct region of the Bay. This would, in turn, assist in decision-making to prioritize new data collection to develop the watershed, airshed and oceanic exchange/loading subcomponents of the loading model.

The loading model would be used to establish load allocations of nutrients that the SF Bay estuary can sustainably assimilate. Although data could be collected to make empirical estimates, the ultimate utility of a loading model is to generate simulations of the past, present or future state of the Estuary and watershed, airshed and ocean (e.g., population growth, climate change, etc.) to explore potential effects of management actions and evaluate alternatives. Thus these models would be a key component of a strategy to adaptively manage SF Bay. The loading model, which would incorporate information about land use, industrial and wastewater plant discharges, wet and dry atmospheric deposition, oceanic exchange, weather and other sources, would include four components: 1) a hydrologic sub-model, 2) a non-point source sub-model (wet and dry weather runoff), 3) a river sub-model which routes flow and associated nutrient loads to the Estuary from the Delta and other major tributaries that drain to the Bay, and 4) a oceanic submodel that would create boundary conditions for exchange of the estuary with the coastal ocean.

Table 4.5. Summary of data gaps and recommended next steps for quantification of nutrient loads to San Francisco Bay.

Source	Data Gaps Identified	Recommended Next Steps
Atmospheric Deposition	No recently published data on wet & dry atmospheric deposition.	Loads likely relatively small. Literature review to determine range of N and P deposition rates for West Coast coastal urban areas. Recommend baseline atmospheric deposition monitoring of wet and dry N and P deposition over 1-2 year period to better constrain estimates.
Terrestrial Loads from Delta	Data available through RMP on dry season concentrations. No data available on wet weather concentrations during storm flow.	Loads likely large. Recommend analysis of existing RMP data to estimate dry season nutrient loads. Initiate wet weather sampling at the DWR gauge at Mallard Island at the head of Suisun Bay to support improved daily loads estimates for 1995-present.
Municipal Effluent	Data available for 15 of approx. 40 POTWs.	Synthesize existing nutrient discharge and concentration data to estimate loads over period of last 10-20 years. Encourage all treatment plants that discharge to the Bay to begin analyzing effluent for total and dissolved inorganic nutrients and to submit these data to the SFRWQCB on a regular basis. Recommend that the POTWs conduct a laboratory inter-comparison on nutrient methods to assure comparability of estimates.
Industrial Effluent	Some data available from the 1990s. Recent data availability unknown.	Synthesize available data to provide information for prioritization of any future steps.
Stormwater	Some data available but general lack of land use-specific wet weather data sufficient to calibrate and verify a watershed loads model.	Synthesize data to provide an updated estimate of stormwater contributions to assist prioritization of next steps. Scope the data needs for development of a dynamic watershed loading model.
Groundwater	Some data available from 79 USGS monitoring stations surrounding the Bay. Flow data currently less well understood.	Refine current loads estimates with review from local USGS groundwater experts in order to support prioritization of next steps.
Exchange with Coastal Ocean	Some data available for fluxes of water and sediments during selected tides and seasons in the past decade collected by USGS and US Berkeley using comparable methods.	Initiate a workgroup of local experts to design a sampling program for nutrient flux at the Golden Gate boundary, with the intent of developing a hydrodynamic and material flux dynamic model to describe exchange with coastal ocean

5. Evaluation of Candidate NNE Indicators for Application in San Francisco Bay Estuary and Summary of Existing Literature

5.1 Introduction

Development of an NNE framework for SF Bay estuary requires the selection of appropriate ecological response indicators to diagnose eutrophication or other adverse effects of nutrient over enrichment. The purpose of this section was to summarize existing information available on each indicator (Table 5.1.1), evaluate the appropriateness of candidate NNE candidate indicators for SF Bay, and identify data gaps in information needed to develop NNE thresholds.

Table 5.1.1. Candidate indicators reviewed for potential development within the NNE framework for San Francisco Bay.

Type	Indicator Group	Indicator or Metric	Section to refer to
Primary Producers	**Phytoplankton**	Phytoplankton Biomass (chlorophyll a concentration)	5.2
		Productivity (carbon fixed per unit volume and time)	
		Assemblage/Taxonomic Composition	
		Harmful algal bloom species -- cell count	
		Harmful algal bloom species – toxins	
	Macroalgae	Percent Cover and Biomass	5.3
	Seagrass and Brackish Water Submerged Aquatic Vegetation	Phytoplankton Biomass	5.4
		Macroalgal Biomass and Cover	
		Epiphyte Load	
		Light Attenuation	
Consumers	**Benthic Macroinvertebrate**	Benthic infauna taxonomic composition, abundance and biomass	5.5
	Jellyfish	Taxonomic composition and abundance	5.6
Water Column Physio-chemistry	**Nutrient Concentrations and Ratios**	Ammonium	5.7
		Urea	
	Dissolved Oxygen	Concentration	

The appropriateness each of the candidate indicators was evaluated relative to four criteria:

1. Ample scientific evidence demonstrating a linkage to SF Bay estuary beneficial uses
2. The existence or potential to develop a predictive relationship with causal factors such as nutrient concentrations/loads and other factors known to regulate response to eutrophication (hydrology, etc.)
3. Availability of a scientifically sound and practical method to measure the indicator
4. The ability to show a trend either towards increasing or/and decreasing eutrophication with an acceptable signal: noise ratio

5.2 Phytoplankton

Phytoplankton have a variety of characteristics that make them potentially useful as indicators of eutrophication in estuaries. Phytoplankton are highly sensitive indicators of nutrient availability in surface waters since their growth rates are relatively rapid, growth responses occur at a wide range of nutrient concentrations and photosynthetic responses can be measured using an array of sensitive techniques (Paerl et al., 2007). Phytoplankton can be described by a number of indicators that may be relevant for use in the SF Bay NNE framework. They include:

- Biomass, as measured by water column chlorophyll a;
- Productivity, as measured by the rate of carbon fixed per unit time per square meter (areal) or per cubic meter (volumetric)
- Assemblage), as measured by the species taxonomic composition, the relative abundance of species (as measured by cell counts), and/or size class of the cells.
- Abundance of HAB species and HAB toxins

In this sub-section we describe the current understanding of spatial and temporal variation in phytoplankton on seasonal, interannual and decadal scale trends, the factors affecting phytoplankton biomass and community structure in SF Bay, and discuss the suitability of phytoplankton as an indicator of eutrophication.

5.2.1 Applicable Habitat Types

Phytoplankton require light to photosynthesis and therefore are typically limited to the shallow to deepwater subtidal regions of an estuary. As depths decrease towards the shallow subtidal zone and particularly in macrotidal estuaries, benthic microalgae and macroalgae that are attached to sediment are at a competitive advantage over phytoplankton which can be easily flushed out during tidal cycles or torn apart by tidal currents or wave energy. With increasing depth, phytoplankton's advantage over benthic algae and rooted bed-forming submerged aquatic vegetation and seagrass increases, because phytoplankton are able to position themselves in the upper portion of the water column and outcompete other primary producers for light and nutrients. In shallow subtidal habitats, phytoplankton can be found in codominance with SAV, microphytobenthos, and macroalgae. In turbid or deepwater subtidal habitats, particularly in wave dominated environments, phytoplankton species tends to be the dominant primary producer, or co-dominant with microphytobenthos in deepwater habitats with high water clarity (Day et al., 1989; Wetzel, 2001).

North, Central and South Bay are dominated by subtidal habitat (71, 96, and 68%, respectively). For this reason, phytoplankton is the largest component of primary producer biomass in SF Bay (Cloern et al., 2000) with carbon production from planktonic species historically making up roughly 70% of total production (Jassby et al., 1993). Measures of phytoplankton are thus key candidate indicators for the SF Bay NNE framework in subtidal habitats.

5.2.2 Available Data on Phytoplankton Biomass, Productivity, and Assemblage

Although nutrient concentrations are relatively high in SF Bay, algal biomass has been relatively low compared to other River dominated systems (e.g., Chesapeake Bay; Cloern, 2001), though most recent estimates for the Bay as a whole show productivity in the normal range of other temperate latitude estuaries (Cloern et al., 2006). Much of the annual production occurs not from the year-round baseline persistence of phytoplankton but rather when algae blooms occur. Algal blooms have been defined by Cloern (1996) as:

> *…events of rapid production and accumulation of phytoplankton biomass that are usually responses to changing physical forcings originating in the coastal ocean (e.g., tides), the atmosphere (wind), or on the land surface (precipitation and river runoff). These physical forcings have different timescales of variability, so algal blooms can be short-term episodic events, recurrent seasonal phenomena, or rare events associated with exceptional climatic or hydrologic conditions (Cloern, 1996, p 127, 133).*

Cloern (1982) defined blooms in SF Bay to be chlorophyll a concentrations >10 µg L^{-1}. Algal blooms are natural events and are the foundation for the secondary productivity which supports the SF Bay food web. There is a concern that increases in the phytoplankton biomass or changes in species composition (in particular, shifts in the frequency and duration of blooms dominated by harmful algal species) may occur in the future in response to changing nutrient loads, turbidity and other limiting factors.

The USGS (Menlo Park Laboratory) has been collecting water quality data in SF Bay on nutrient concentrations and related ancillary data continuously for 39 years beginning 1968 and on phytoplankton since 1977. Their research program includes measurements of water quality from a monthly ship cruse of 39 fixed locations 3-6 km apart along the 145 kilometer spine of the entire Estuary. Since the USGS sample-collection was driven by research questions, it has not always been as regular or systematic as would occur in a monitoring program. For example, the USGS stopped sampling completely in 1981 after the spring bloom and didn't sample in the North Bay from about 1980-1987. That accepted, the database generated presently includes >11,000 discrete laboratory measurements of the chlorophyll a in water samples and 156,610 estimates of chlorophyll a made from a linear relationship between fluorometer voltage and discrete lab measurements. In addition to information collected during these regular monthly cruses, real-time remote observing instrumentation has greatly enhanced the surveillance in recent years (Cloern et al., 2005b), though it should be noted that remote sensing captures only surface blooms and many blooms dominated by HAB species are not easily distinguished using readily available, multi-spectral remote sensing products (e.g., MODIS). On many occasions, the USGS group and collaborating coauthors have also carried out special studies in locations off transect (e.g., Cloern and Oremland, 1983; Cloern et al., 1985; Powell et al., 1986; Lucas and Cloern, 2002; Thompson et al., 2008; discussed in detail below). Estimates of pelagic primary production were made estimated either directly using the ^{14}C radioisotope tracer method (Steeman Nielson 1952, Cole and Cloern 1984) or indirectly through an empirical model that derives productivity from biomass and light attenuation (Cole and Cloern 1987).

More recently, continuous monitoring has also conducted by scientists at the San Francisco State University's Romberg Tiburon Center for Environmental Studies. They have been collecting information on chlorophyll a and a number of ancillary parameters every 6 minutes using instruments mounted just offshore at the end of a 200-ft pier adjacent to the RTC campus on the Tiburon peninsular (R. Dugdale and F. Wilkerson, pers. comm.). This data is part of the observing networks of the Council on Ocean Affairs, Science and Technology (COAST) and the Central and Northern California Ocean Observing System (CeNCOOS). Data on chlorophyll a have been collected 0.5 m below the water surface using a flotation platform that adjusts with the tides from April 2006 to January 2009 and at a fixed datum 1 meter below lower low tide from 12/2008-present. In addition, the group has been publishing on a number of focused research projects on the ecology and controls of diatom productivity in the northern reaches of SF Bay. The research groups at the USGS and the RTC have been responsible for the majority of systematically collected measurements on phytoplankton biomass and community composition in SF Bay downstream from the Sacramento – San Joaquin confluence near the Region 2/Region 5 Water Board boundary.

5.2.3 Factors Effecting Temporal and Spatial Variation of Indicator Phytoplankton Biomass and Productivity

In SF Bay, the biomass and primary productivity associated with phytoplankton varies in space and time in response to nutrient availability from external loads (e.g., Wilkerson et al., 2006; Dugdale et al., 2007) and internal regeneration (Grenz et al., 2000), grazing (Cloern et al, 1985; Thompson et al., 2008), stratification (Cloern, 1991; Cloern, 1996), water temperature (Cloern et al., 2007; Lehman et al., 2008), tidal energy (Lucas and Cloern, 2002), transparency (May et al., 2003), wind/wave energy (May et al., 2003), the availability of seed cysts (Cloern and Dufford, 2005; Cloern et al., 2007), UV radiation effects on nitrate versus ammonium assimilation perhaps due to disruptions of enzyme pathways (Hogue et al., 2005), differential uptake of nitrate and ammonium by larger versus smaller cells (Wilkerson et al., 2006), inhibition of nitrate uptake by ammonium (Wilkerson et al., 2006; Dugdale et al., 2007), predation by benthic invertebrates (e.g., Thompson et al., 2008), and variations in the phase of the Pacific Decadal Oscillation and related changes to top down predation of benthic invertebrates (Cloern et al., 2007). These factors lead to spatial gradients across shoals to the axis, between segments of the Bay, and temporal variation at scales ranging from days to years to decades.

Spatial Variability

In the broadest sense, the Bay can be divided into two main regions, the North Bay and the South Bay. The North Bay is a river dominated estuary where spatial and temporal variability is driven by intra- and inter-annual variations in freshwater, sediment, and nutrient input from urban and agricultural sources within the Sacramento and San Joaquin River watersheds (Sigleo and Macko, 2002; Smith and Hollibaugh, 2006; Wilkerson et al., 2006). The estimated average freshwater flushing time of the North Bay is 72 days (Engle et al., 2007). The South Bay in contrast acts more like a tidal lagoon with relatively low freshwater input relative to basin volume; it is dominated in the summer months by wastewater discharge (Cloern et al., 2000; Smith and Hollibaugh, 2006). The average estimated freshwater flushing time of the South Bay is over 4,000 days (Engle et al., 2007). Within these broad classes, due mainly to physiographic controls on freshwater and tidal flow (Powell et al., 1986), the Bay can be further divided into six strata or segments (see Figure 3.6 in Section 3 of this report) that have small within strata variance relative to variability along the whole gradient between marine and freshwater conditions (Cloern et al., 2000).

Chlorophyll a varies laterally from shallow areas to the axis (Cloern et al., 1985; Thompson et al., 2008) often associated with variations in turbidity and the timing of wind relative to the tidal cycle, fetch, and tidal forces (May et al., 2003). For example, while the focus of an early study by Cloern et al. (1985) was on intra-annual temporal variability in phytoplankton biomass, the paper also illustrated biomass variability across lateral gradients in SF Bay (Figure 5.2.1). More recently Thompson et al. (2008) discussed strong lateral gradients in the South Bay (Figure 5.2.3). Their observations supported the hypothesis that bloom generation began on the east shoals in most years and spread into the channels if the bloom persisted. There was one instance, however, when a channel produced phytoplankton bloom was observed perhaps attributable to persistent stratification (Lucas et al., 1998).

However, by far the most persistent spatial gradient of phytoplankton biomass variation occurs between the ocean entrance at the Golden Gate Bridge and the fresh water extremities in the Lower South Bay and the Sacramento River Delta (Cloern et al., 2000). Algal productivity varies widely in each region of the Bay. Based on data collected from 1995 to 2009, average chlorophyll a concentrations vary from 13 $\mu g\ L^{-1}$ in the lower South Bay to 2.6 $\mu g\ L^{-1}$ in the river-dominated North Bay (Table 5.2.1). Suisun Bay, although high in nutrients, exhibits relatively low mean chlorophyll a concentrations relative to the South Bay (Wilkerson et al., 2006). Concentrations are more temporally variable both within a year and between years further from the Golden Gate.

The causes for the Bay wide trends include changes in water clarity due to less suspended sediment (Schoellhamer, 2009), lower metal inhibition due to improvements in wastewater treatment, increased seeding from ocean populations (Figure 5.2.3; Cloern et al., 2005), declines in consumption by bivalves due to increases in predation by juvenile English sole and speckled sanddabs, and declines in phytoplankton consumption by bivalves and zooplankton due to recent new invasive species introductions (Cloern et al., 2006).

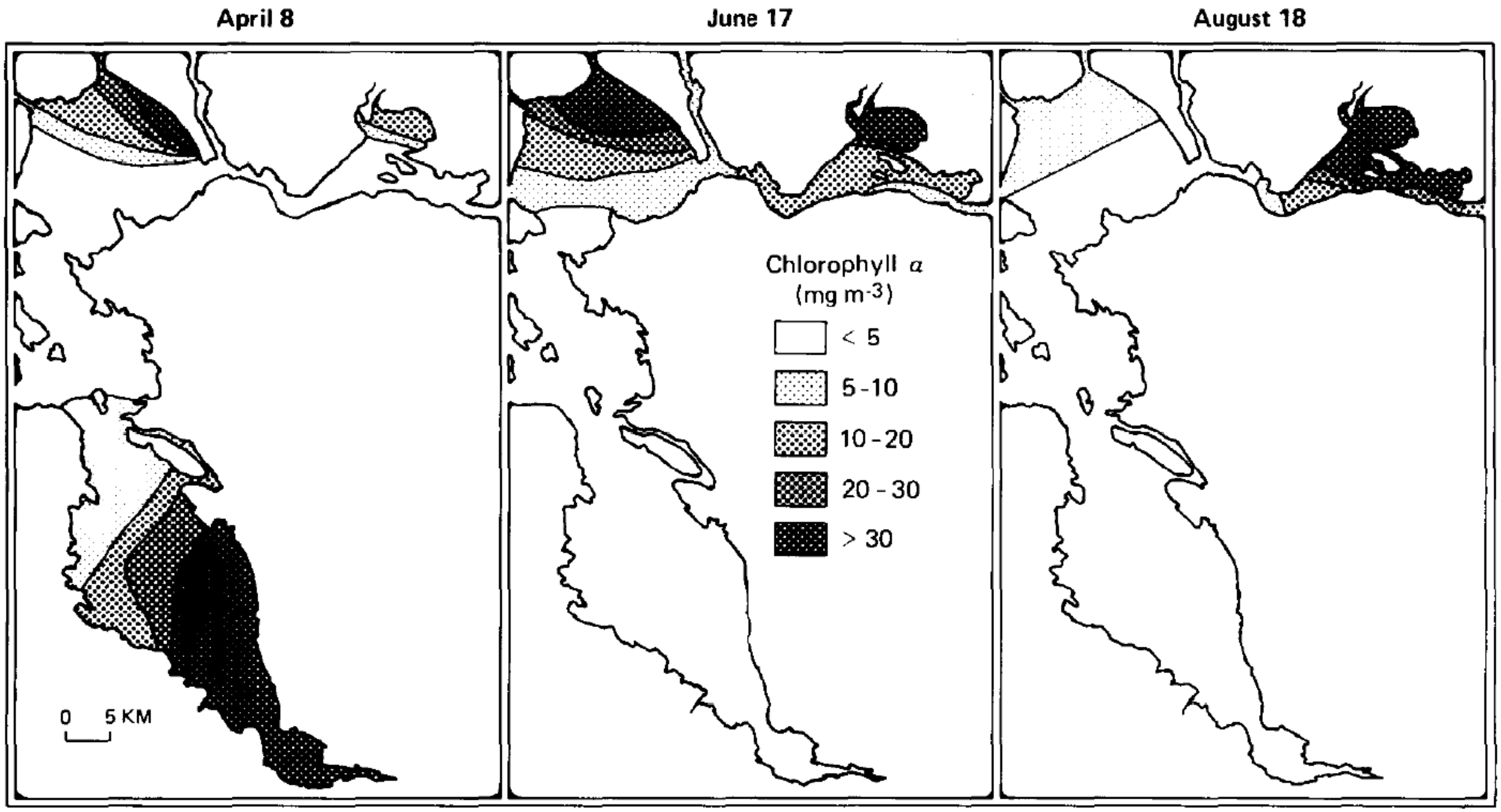


Figure 5.2.1. Lateral variability in chlorophyll a concentrations based on measurements at 106 sites during 1980. Figure extracted from Cloern et al. (1985).

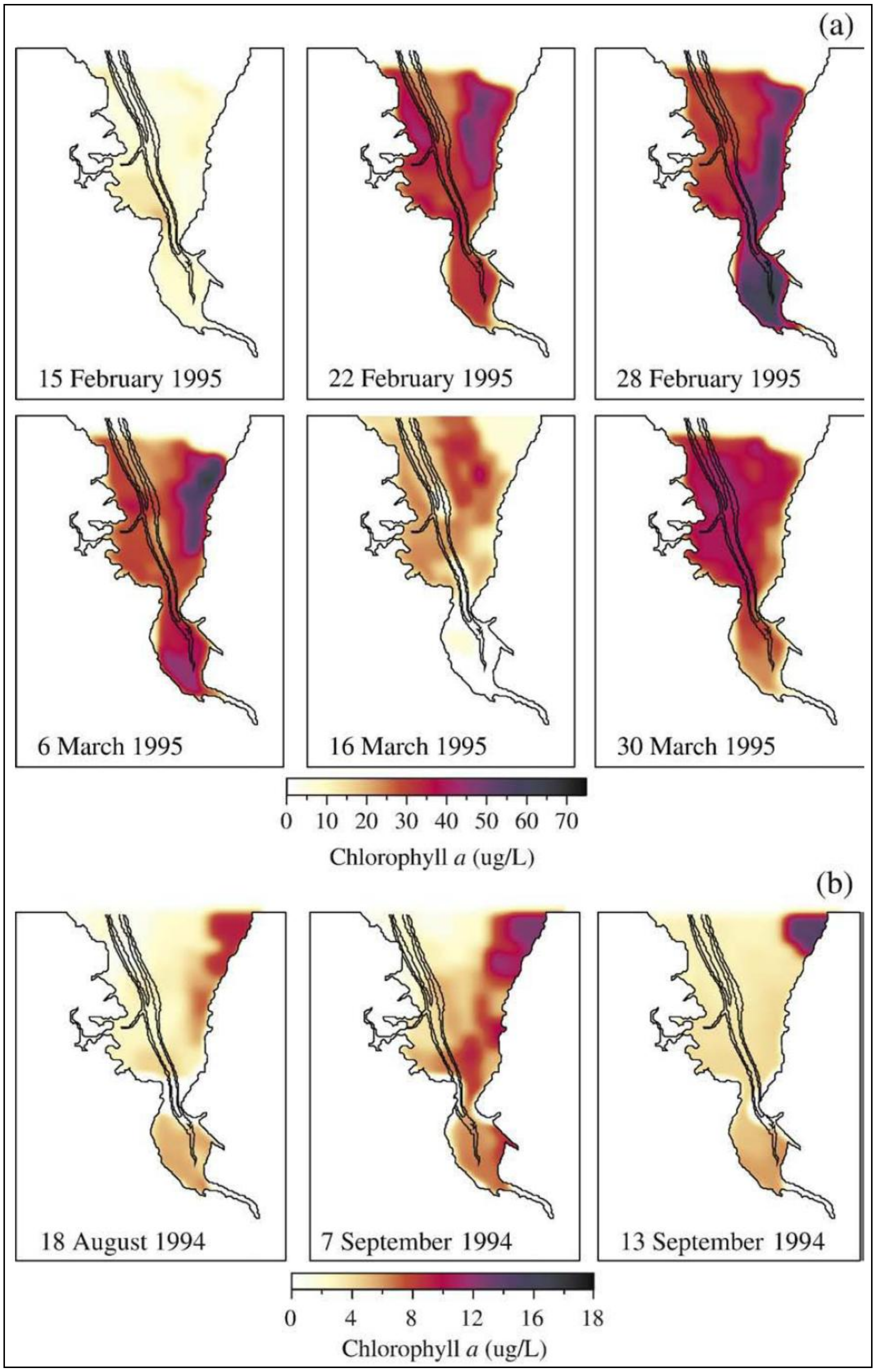


Figure 5.2.2. Lateral variability in chlorophyll a concentrations in the South Bay sites during 1995 (59 stations; a) and 1994 (49 stations; b). Figure extracted from Thompson et al. (2008).

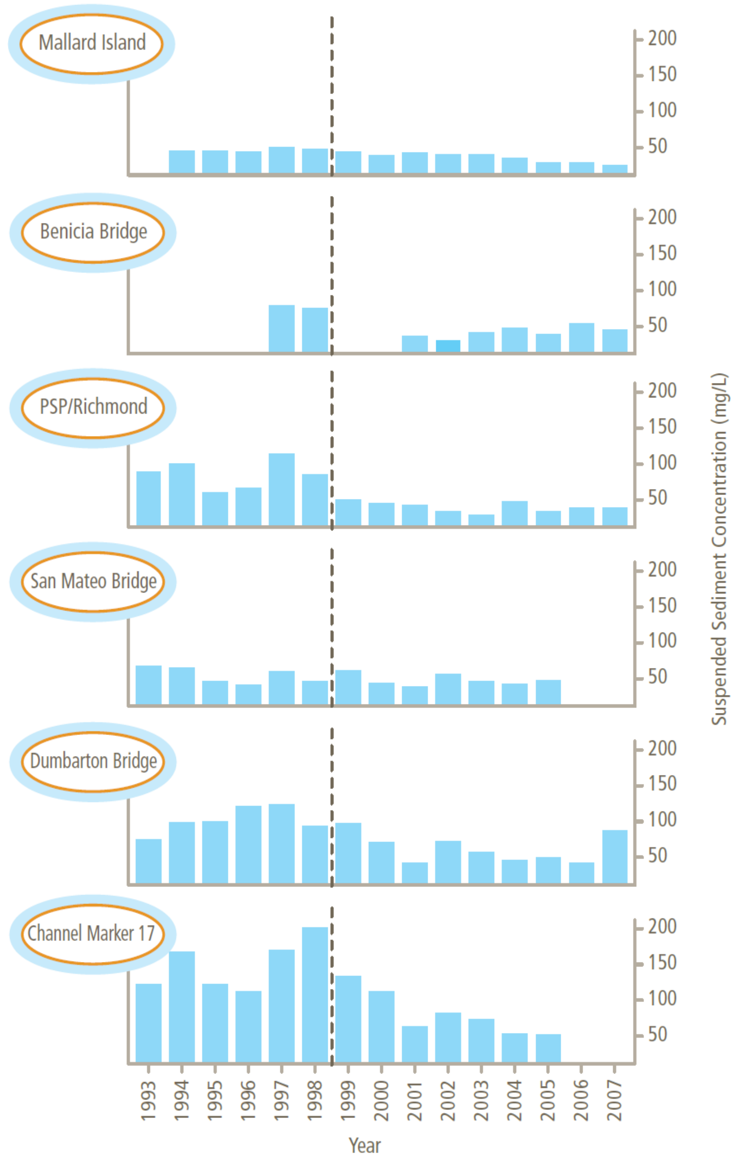


Figure 5.2.3. Trends in suspended sediment concentrations in San Francisco Bay (source Schoellhamer, 2009).

Table 5.2.1. Variation of chlorophyll a among estuary segments. Period 1999-2009. (Source: James Cloern, USGS: http://sfbay.wr.usgs.gov/access/wqdata).

Style	Segment	Chlorophyll a ($\mu g\ L^{-1}$ equivalent to mg/m^3)		
		Minimum	Maximum	Mean
River dominated	Rivers	0.4	14	2.6
	Suisun Bay	0.1	12	2.6
	Carquinez Straight	0.4	30	2.9
	San Pablo Bay	0.1	44	3.5
Oceanic	Central Bay	0.1	48	5.1
Lagoonal	South Bay (SF Bay Lower)	0.9	106	9.3

Intra- and Inter-annual Temporal Variability

Temporal variability in chlorophyll a and/or phytoplankton has been observed at scales ranging from hours to years (Cloern et al., 1985, 2000; Hogue et al., 2001; Cloern et al., 2003; Thompson et al., 2008) and to decades (Cloern et al., 2007; Jassby, 2008; Cloern et al., 2010). In general, both the North Bay and the South Bay experience low phytoplankton concentrations during the winter (December, January) and summer months (June, July) typically <5 $\mu g\ L^{-1}$ and greater concentrations during most spring periods (Figure 5.2.4). The blooms in the North Bay reach much lower peak concentrations than the blooms in the Central or South Bay and can be absent all together during years of low runoff (Cloern et al., 2000). Averaging the data since 1999, it is seen that the largest blooms occur in the South Bay during the spring (February to May inclusively; Figure 5.2.4), when river runoff sufficiently stratifies the water column and light penetrates more easily (Cloern et al., 2006). Phytoplankton biomass in the South Bay are characterized by strong intra-annual or within season variability; concentrations vary markedly between months over short time scales (Cloern and Jassby, 2010). That said, larger more prolonged blooms appear to last for 6 weeks or more during wetter years (e.g., 1993, 1995) reaching >70 $\mu g\ L^{-1}$, whereas blooms in drier years (e.g., 1991, 1992) lasted only 2 weeks and reached concentrations <20 $\mu g\ L^{-1}$ (Thompson et al., 2008).

Other than supply of nutrients and stratification, bivalve grazing appears to be a strong control on bloom magnitude, extent and longevity (Thompson et al., 2008). The seasonal absence of bivalve grazers in the winter months on the shoals sets up the potential for bloom each spring. Phytoplankton dynamics are strongly controlled by timing and recruitment process of bivalves, which in turn may be controlled by predation from fall migratory birds and fish (Thompson et al., 2008). However, ultimately bivalve biomass is also triggered and controlled by the available phytoplankton food resources; for example in 1995, bivalves that recruited on the shoals at the beginning of the bloom grew sufficiently in six weeks to control the shoal phytoplankton biomass (Thompson et al., 2008). This concept of coupled ecosystem capacity through the transfer of nutrients from phytoplankton biomass into secondary consumers, senescence and death, and recycling of regenerated nutrient back to primary producers was also discussed by Cloern (2007).

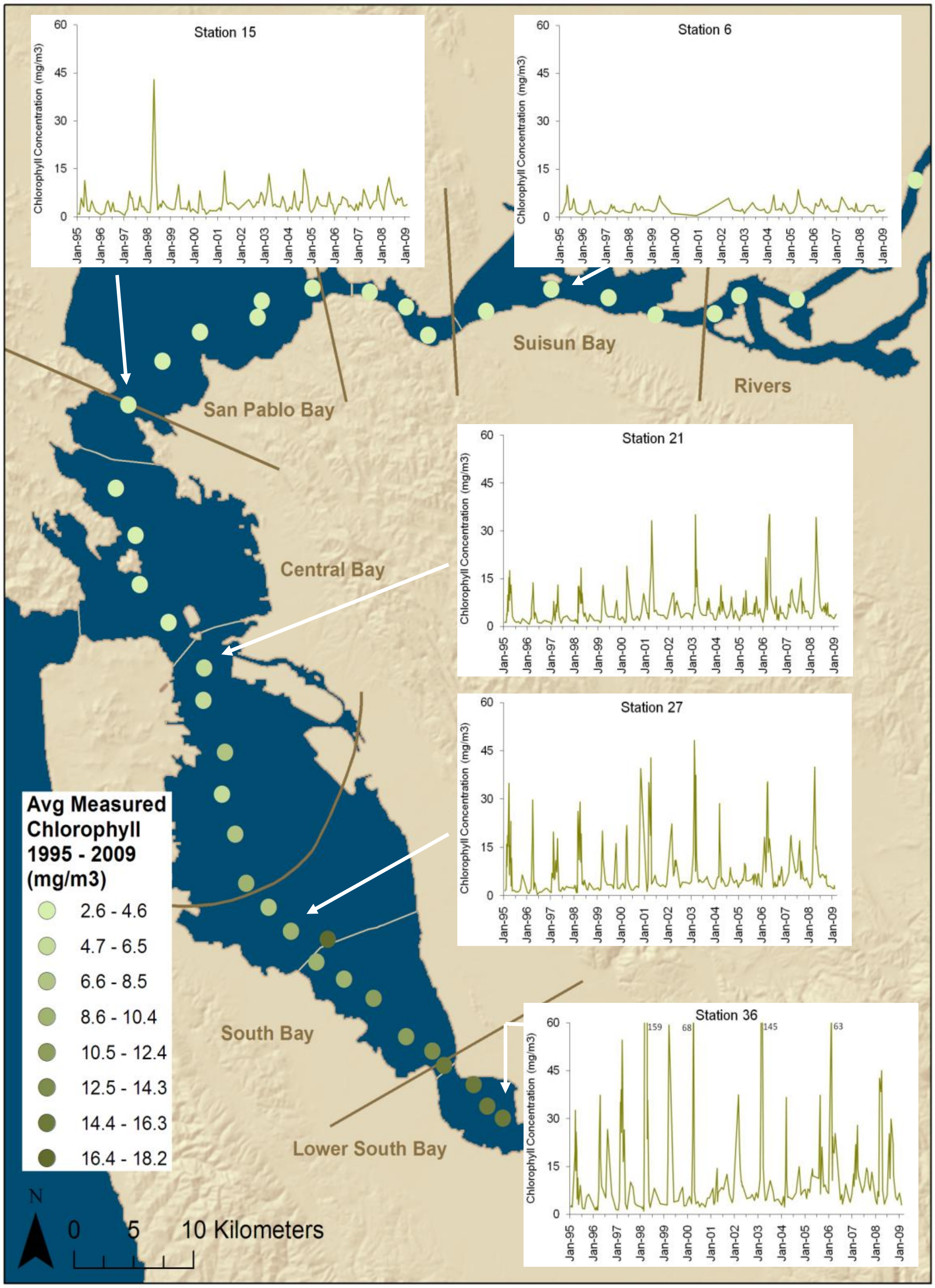


Figure 5.2.4. Seasonal chlorophyll a concentrations in San Francisco Bay based on monthly sampling by the USGS (Source: USGS: http://sfbay.wr.usgs. gov/access /wqdata).

Blooms in the North Bay occur when vertical salinity stratification occurs also in the late winter/spring (improving light penetration) and in the presence of low ammonium (Wilkerson et al., 2006). For similar reasons, the largest spring blooms occur after the wettest winters (e.g., 1998, 2003) (Figure 5.2.4, station 15 and station 6). In a similar fashion to the South Bay, averaging data collected since 1999 indicates a pattern of bloom in the spring and again in the fall (Figure 5.2.5).

Dugdale et al. (2007) summarized work to-date for the North Bay describing a conceptual model that includes a sequence of events that lead to blooms in the North Bay: 1) stabilization of the water column by stratification and or reduced tidal energy, 2) reduced NH4 concentrations (to a critical level below 4 µM) through dilution during runoff or by phytoplankton uptake, and 3) uptake (secondarily) of NO_3 autumn blooms are characteristically smaller than the spring blooms perhaps because phytoplankton does not deplete the ammonium enough to switch over to NO_3 uptake (Dugdale et al., 2007). In the spring, phytoplankton more often depletes the ammonium (especially in the North Bay) perhaps because ammonium in the Bay at this time is diluted by spring runoff or because ammonium regeneration is lesser than ammonium consumption by the growing bloom. Phytoplankton biomass in the North Bay is characterized by weaker variability between months but higher and dominant intra-annual variation in phytoplankton biomass (Cloern and Jassby, 2010), with some years exhibiting little bloom activity and other years having significant events. This strong inter-annual variability appears to be driven by variation in river runoff, the balance between ammonium and nitrate (in relation to sources that include wastewater, urban, and agricultural runoff; Wilkerson et al., 2006; Dugdale et al., 2007), and the introduction of the nonindigenous clam *Potamocorbula amurensis* (Alpine and Cloern, 1992).

Many models of phytoplankton mass in SF Bay have been developed over the past three decades. For example, Cloern and Cheng (1981) developed a pseudo-two-dimensional model to simulate the dynamics of a single dominant phytoplankton species in the North Bay. Using this model they were able to account for most of the variability of biomass as a function of light availability, temperature, salinity and copepod grazing; nutrients were deemed non-limiting. The model supported the premise that populations established over the shoals and were enhanced by reduced transport due to estuarine gravitational circulation. Later Lucas et al. (1998) presented a model for South SF Bay that included benthic grazing, zooplankton grazing, vertical phytoplankton sinking through a stratified water column, and respiration losses. They specifically did not incorporate nutrient availability since, in South SF Bay, bloom initiation was not thought controlled by nutrients; rather bloom termination can sometimes occur when nutrients are depleted although this still warrants further investigation (Thompson et al., 2008).

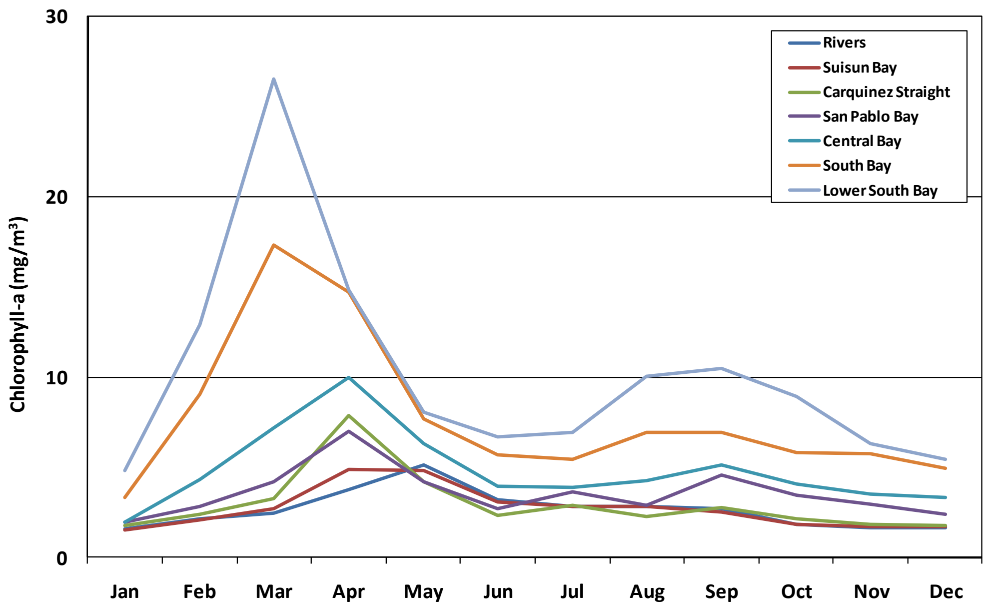


Figure 5.2.5. Average monthly chlorophyll a concentrations in San Francisco Bay based on monthly sampling by the USGS between January 1999 and February 2009 (Source: James Cloern, USGS: http://sfbay.wr.usgs.gov/access/wqdata).

In a later two-paper series, Lucas et al (1999 a, b) looked at the two main processes for governing bloom formation, 1) local mechanisms such as depth, light availability, and benthic grazing, and 2) transport related mechanisms which control the extent and distribution of the resulting bloom (Lucas et al., 1999a,b). They concluded that local conditions control the balance between phytoplankton loss and production and that initiation can occur in both shallow and deeper areas. They also pointed out that due to transport, greatest biomass may occur in areas that are not the most productive and vice-versa. Interestingly, again they did not include nutrients in the components of the models that simulated phytoplankton production, but in order to constrain peak biomass during long simulation times, an ad-hoc representation of nutrient limitation for bloom termination was included (Lucas et al., 1999a).

Lucas and Cloern (2002) explored the influence of tidal deepening and shallowing on phytoplankton production. They also assumed nutrients were not limiting and concluded that if tidal range is large relative to water depth, then tidal range may significantly influence net phytoplankton growth. Following on from this study, May et al. (2003) developed a coupled one dimensional model that simulated vertical and horizontal mixing processes to explore the impacts of turbidity on phytoplankton dynamics. Turbidity variation associated with wind strength was implicated as a control mechanism for the development of spring blooms in the South Bay. It was suggested that during years with high wind during the critical bloom period, phytoplankton productivity can be suppressed in contrast to years of lower wind (May et al., 2003).

The concept of the role of physical and biological processes in one sub-region controlling phytoplankton biomass and bloom production in an adjacent sub-region was explored using a coupled pseudo-two-dimensional model Lucas et al. (2009). They concluded that positive coupling occurs between productive shallow shoal areas and adjacent relatively unproductive deeper water channels. They also further supported the earlier premise that turbidity (May et al., 2003), benthic grazing (Thompson et al., 2008), and vertical density stratification (Lucas et al., 1999a,b; Thompson et al., 2008) control bloom occurrence, longevity, and spatial extent.

Most recently, it has been proposed that the cause of the annual autumn bloom might be attributed to sharp declines in bivalve mollusks (phytoplankton consumers) resulting from a trophic cascade caused by the onset of the East Pacific "cold phase" (Cloern et al., 2007). In the south Bay, Cloern et al. (2007) deduced that trends are not likely caused by changes in other reasonable factors alone such as nutrients (an observed decline), temperature (no change), stratification (no change), and turbidity (an observed weak increase) (note – this turbidity trend appears to contrast with data in Figure5.2.3: Schoellhamer, 2009). Cloern et al. (2007) argued instead that a 20-fold decrease in benthic water column filtering herbivores (e.g., *Corbula amurensis, Venerupis japonica, Musculista senhousia*, and *Mya arenaria*) has coincided with phytoplankton increases in southern areas of SF Bay in part caused by an collective increase in shrimp, crab, and sole biomass of about 4x (Cloern et al., 2007). An argument is now presented that the classic model of nutrient enrichment and light limitation as primary controls on phytoplankton in South SF Bay (and other estuaries) may be overshadowed by shifts in top-down control sometimes associated with connective shifts in sea surface temperatures and upwelling. In the case of SF Bay, sea surface temperatures and upwelling are a function of the Pacific Decadal Oscillation (PDO) and the broader ocean Basin (Smetacek and Cloern, 2008).

Decadal Scale Temporal Trends in Phytoplankton Biomass and Productivity

Long term monitoring data indicate decadal scale trends. Blooms are generally on the rise in the marine domains of the Bay with the exception being the River dominated Suisun Bay (Cloern et al., 2006) where *Corbula amurensus* is implicated as providing high grazing pressure (Alpine and Cloern, 1992) and ammonium may be inhibiting growth (Wilkerson et al., 2006; Dugdale et al., 2007). In an analysis of chlorophyll a concentrations since 1980, Cloern et al. (2006) showed that spring blooms since 1999 have been much larger than those prior to 1999, and that autumn-winter blooms are now occurring where they did not previously regularly occur, an observation they have called a "regime shift" (Cloern et al., 2006; 2007; 2010). In addition, baseline chlorophyll a concentrations have increased since the mid-1990s and these trends are significant year round in all locations from San Pablo Bay south. Suisun Bay and the Delta appear to be different (Jassby, 2008). Although overall since 1970 there has been a decrease on productivity in Suisun Bay and the Delta, since 1996 phytoplankton biomass appears to have stabilized in Suisun Bay and shown a positive increase throughout the Delta (Jassby, 2008). Beginning in 1999, the Bay began exhibiting autumn/winter blooms (September to December inclusively) (Figure 5.2.4), although these are generally have lower biomass than the annual spring bloom. In later years this annual autumn/winter bloom, although mainly comprised of diatoms, even included dinoflagellate red tides (Cloern et al., 2007) (see HABs discussion below). Increasing phytoplankton in the central and southern sectors of the Bay is manifested as increasing baseline concentrations of small cell plankton, increasing magnitude of spring blooms (larger cell diatoms), and occurrence of small cell autumn/winter blooms (Cloern et al., 2007).

While the causes of these majors changes are still being evaluated, over all, from San Pablo Bay to the lower South Bay, mean annual primary production has increased 75% over 1993-96 levels (Cloern et al., 2006). Carbon production by phytoplankton was estimated to be 200,000 US tons, or about 150 g C m^{-2} in 1980 (Jassby et al., 1993). At that time, the carbon budget of the south Bay was dominated by autotrophic production (92%); in contrast North Bay carbon was 68% allochthonous and supplied from Rivers (Jassby et al., 1993). Phytoplankton was responsible for 67% of the autochthonous production in the South Bay and 70% in the North Bay. Estimates of autochthonous total carbon production in 1993 - 1996 were about 120 g C m^{-2} (similar to the 1980 figure) and most recently production has increased again to an annual average of about 215 g C m^{-2} associated with both enhanced bloom and non-bloom biomass (Cloern et al., 2006). This has included a more than doubling of the autumn/winter (August-December) production from 32 g m^{-2} (pre-1998 mean) to 73 g m^{-2} (post 1998 mean). Based on an analysis of monthly trends, eight out of 12 months distributed across the whole year showed an upward trend (Cloern et al., 2007). Presently, a reanalysis of data is being completed to further evaluate summer trends. Preliminary data analysis conducted by Alan Jassby and James Cloern shows increasing chlorophyll a in South Bay, San Pablo Bay, and Suisun Bay since the mid-late 1980s at an average rate of 3-5% per year (James Cloern Personal Communication, March 2011). These new analyses provide further evidence that the Bay is changing, perhaps motivating further interest to understand the effects of nutrient loads and other co-factors.

Future trends are hard to predict. One hypothesis for the northern Bay (particularly Suisun Bay) is that any alleviation of the mechanisms currently limiting phytoplankton growth during the spring bloom, whether it is ever proven unequivocally what these mechanisms are, should lead to greater dominance of larger celled diatoms (Wilkerson et al., 2006). Because many of the HAB species common to the West coast are large celled (R. Kudela, personal communication March 2011), it is not clear whether additional factors may promote the dominance of HABs, including the toxic diatom genus Pseudo-nitzschia, versus non-harmful diatoms which better support Bay beneficial uses. In contrast, if the autumn bloom increases due to increased ammonium regeneration, phytoplankton species which have a preference for ammonium, including HAB species such as Pseudo-nitzschia and many toxic dinoflagellates, may become more prevalent (Kudela et al. 2010).

5.2.4 Phytoplankton Assemblage and Harmful Algal Blooms

The benefits of enhanced primary production during blooms are directly correlated with the species that dominate the bloom. Large cell diatom production tends to fuel the pelagic food web supporting zooplankton including jellyfish, filter feeding shell fish and crustaceans, fishes, and mammals including humans. In contrast, blooms of toxic smaller celled flagellates and some large-celled HAB species can suppress herbivores and impact beneficial uses for aquatic wildlife and humans (Cloern, 1996; Ning et al., 2000; Cloern et al., 2005b). This section covers two types of indicators: 1) assessment of health based on phytoplankton community structure and 2) abundance of HAB species and HAB toxin concentrations.

Phytoplankton Assemblage

San Francisco Bay contains over 500 phytoplankton taxa. Based on analysis using light microscopy, it appears that 10 and 20 species account for 77% and >90% of the total biomass respectively (Cloern and Dufford, 2005). Diatoms (Bacillariophyta) dominate the biomass making up 81% of the total cumulative biomass; dinoflagellates and cryptophytes (Pyrrophyta and Cryptophyta) made up 11% and 5% respectively (Cloern and Dufford, 2005). Cell sizes range between <3 and >100 um but in the nutrient enriched SF Bay system, large cells >30 µm contribute 40% of the biomass; attributed to the lack of a competitive advantage for smaller species. Like many nutrient enriched systems, SF Bay is characterized by a bloom-bust cycle of larger cell species periodically dominating a more stable community of small cell species (Hogue et al., 2001; Cloern and Dufford, 2005; Wilkerson et al., 2006); an observation attributed to the close coupling of small cell consumers in the microbial food web, the lagged response of metazoan consumers (Cloern and Dufford, 2005), and the take up of nitrate by larger cells (Wilkerson et al., 2006). Presently there is no explanation as to why diatoms dominate in SF Bay during blooms; hypotheses range from bottom up (inherently fast division rate, high N assimilation under high nitrate conditions, high growth rate in relatively low light conditions, ability to utilize bicarbonate) or a top down view (silica cell wall is better at resisting predation and/or buoyancy regulation allows avoidance of bottom dwelling filter feeders in shallow estuarine conditions).

In contrast there is a more constant crop of small cell picoplankton composed primarily of cyanobacteria and small eukaryotes *(Nannochloropsis sp., Teleaulax amphioxeia, Plagioselmis prolonga*) that occur across a wide range of salinities and seasonal conditions (Ning et al., 2000; Cloern and Dufford, 2005). Picoplankton make up <15% of the Bay biomass and <2% during blooms (Ning et al., 2000; Cloern and Dufford, 2005) and 11% of the total measured spatially and temporally averaged results for the whole North and South Bay combined. In relation to the possibility of using phytoplankton community structure as an ecological response indicator, some phytoplankton taxa (*Prorocentrum aporum*, *Coscinodiscus marginatus, Protoperidinium depressum, Eucampia zodiacus*) have not been seen since 1996 while others (*Protoperidinum bipes, Pseudo-nitzschia delicatissima, Scrippsiella trochoidea, Thalassiosira nodulolineata*) have appeared perhaps attributable to the Pacific Decadal Oscillation (PDO) (Cloern and Dufford, 2005).

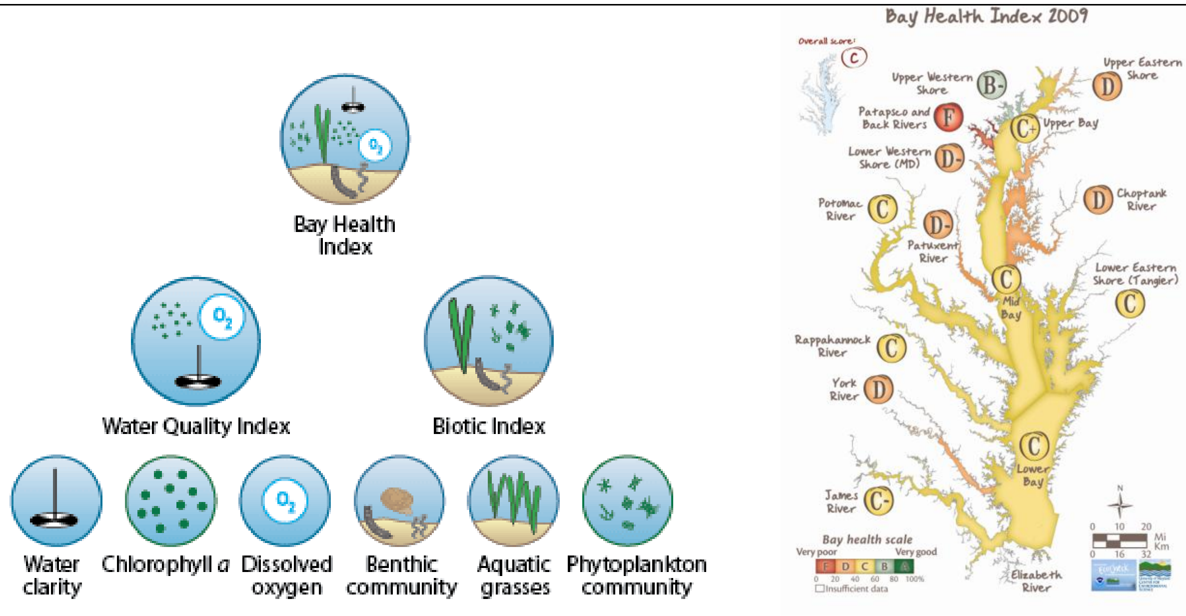
One use of data on phytoplankton community structure is to combine it into an index of biological integrity (IBI). IBIs are becoming more common for assessment of estuarine ecological condition and management focus in the face of physical and chemical transformation, habitat destruction, and changes in biodiversity (Borja et al., 2008). An IBI describes the biological condition of an assemblage of plants or animals, typically based on the diversity and relative abundance of species or the presence or absence of pollution tolerant species. A key element of developing an IBI is the ability to describe the community response of the assemblage (e.g., benthic invertebrates, phytoplankton, etc.) along gradient of physical or chemical stress from minimally disturbed or “reference state” to highly disturbed. IBIs are most commonly used in stream bioassessment, but several examples exist for estuarine environments as well including submersed aquatic vegetation (Dennison et al., 1993; Corbett et al., 2005), benthic macroinvertebrates (Weisberg et al., 1997; Graves et al., 2005), fish populations (Deegan et al., 1997;

Bortone et al., 2005), zooplankton (Carpenter et al., 2006), micro-algae (Paerl et al., 2005) and phytoplankton (Lacouture et al., 2006).

IBIs developed and used in Chesapeake Bay present an example of how phytoplankton community structure data can be synthesized to provide information about the ecological health of the Estuary and about the ability to support specific beneficial uses. A Phytoplankton Index of Biotic Integrity (P-IBI) was developed in Chesapeake Bay using an 18 year data set (Lacouture et al., 2006). Thirty-eight phytoplankton metrics were tested for their ability to discriminate between impaired and least-impaired habitat conditions. Twelve discriminatory metrics were chosen from a tested set of 38 to discriminate between impaired and least-impaired habitat conditions. Combinations of these twelve metrics were scored and used to create phytoplankton community indexes for spring and summer in the four salinity regimes. The P-IBI, thus developed, combined the scores of pollution-sensitive, biologically important metrics of the phytoplankton community into a single index. Like other multi-metric indexes, the P-IBI is more sensitive to habitat conditions than its component metrics, which include chlorophyll a, the abundances of several potentially harmful species, and various indicators of cell function and species composition (Lacouture et al., 2006).

Following on from the work of Dennison et al. (1993) on the use of submerged aquatic vegetation (SAV), Carpenter et al. (2006), who developed an IBI for zooplankton, and Lacouture et al. (2006) on the development and testing of a P-IBI for the Chesapeake, a Bay Health Index (BHI) that combined three water quality and three biological measures was developed to assess the ecological effects of nutrient and sediment loading in Chesapeake Bay (Williams et al., 2009). A Water Quality Index (WQI) was generated by averaging concentrations of chlorophyll a, dissolved oxygen, and Secchi depth. A P-IBI and B-IBI was developed from the biological measures of the phytoplankton and benthic community composition and combined with the area of SAV to create the Biotic Index (BI). The WQI and BI were then averaged to give a BHI for the growing season (March–October) (Figure 5.2.6; http://www.eco-check.org/reportcard/chesapeake/2009/). Least impaired regions of Chesapeake Bay exhibited low chlorophyll a, high dissolved oxygen, greater transparency, higher phytoplankton and benthic indices relative to ecological health-based thresholds, and greater SAV area. All three indexes were significantly correlated with nitrogen (N), phosphorus (P) and sediment loads and the sum of developed and agricultural land use. The BHI is used annually to track progress as part of the annual environmental report card.

The development of multi-metric indexes of estuarine quality are not without challenges which include the formation of multidisciplinary scientific teams and stakeholder groups that are committed to the outcome more than representation of their individual interest, long term multi-parameter data sets on a wide range of biotic and abiotic indicator species, co-factors, and stressors, and empirically demonstrated and perhaps modeled cause and effect relationships that can demonstrate trends with a high signal to noise ratio. Following from the example set in Chesapeake Bay (Carpenter et al., 2006; Lacouture et al., 2006; Williams et al., 2009; Williams et al., 2010); it would seem that SF Bay, with its rich multi-parameter long term data sets, may be a suitable living laboratory to develop such an index.



2009 Water Quality Index, Biotic Index, and Bay Health Index Scores

Bay health scale: Very poor — Very good; F D C B A; 0 20 40 60 80 100%

	Patapsco and Back Rivers	Lower Western Shore (MD)	Patuxent River	York River	Upper Eastern Shore	Choptank River	James River	Potomac River	Mid Bay	**Overall Bay**	Lower Eastern Shore (Tangier)	Rappahannock River	Lower Bay	Upper Bay	Upper Western Shore	Elizabeth River*
Water Quality Index	15	26	33	32	43	47	52	47	48	45	52	49	39	52	42	25
Biotic Index	23	21	14	27	24	21	34	44	44	47	44	48	64	66	79	Incomplete assessment
Bay Health Index	**19**	**24**	**24**	**30**	**34**	**34**	**43**	**45**	**46**	**46**	**48**	**49**	**51**	**59**	**61**	**19***

(*score based on average of 4 indicators)

Figure 5.2.6. The Chesapeake Bay Report Card. The report card aims to provide a transparent, timely, and geographically detailed annual assessment of Chesapeake Bay health. (See Chesapeake EcoCheck: Assessing and Forecasting Ecosystem Status. http://www.eco-check.org/reportcard/chesapeake/2009/**).**

Harmful Algal Blooms and Toxins

Harmful algal blooms are blooms of phytoplankton, algae, or cyanobacteria that can produce potent toxins, nuisance levels of biomass, and suppress oxygen causing harm to humans, fisheries resources, and coastal ecosystems. While increased anthropogenic nutrients increase the potential for HAB development, the conversion of nutrients into biomass is dependent on other factors including clarity, temperature, stratification, and seed populations (Cloern et al., 2005b).

Despite the persistent nutrient enriched status of San Francisco Bay, few harmful algal blooms (HABs) have been reported recently in San Francisco Bay, apparently because nutrient enriched turbid conditions in the estuary favor larger celled diatoms associated with new production as opposed to nutrient regeneration (Cloern, 1996; Ning et al., 2000). A lack of monitoring may also play a role, given the large number of potentially harmful algae present in San Francisco Bay (Cloern and Dufford, 2005). However, there have been historical occurrences (see Cloern et al., 1994 referenced in Cloern, 1996), and recently cyanobacteria and dinoflagellate blooms have been documented. For example, blooms of the cyanobacteria *Microcystis aeruginosa* have been occurring in the late summer/autumn in the northern reaches of the Bay since 1999 (Lehman et al., 2005), the raphidophyte *Heterosigma akashiwo* created a red tide in the Central Bay in summer 2002 (Herndon et al., 2003), and the dinoflagellate *Akashiwo sanguinea* caused a red tide in the Central and South Bay areas during September 2004 (Cloern et al., 2005a; Table 5.2.2; Figure 5.2.7). The conditions under which these blooms occurred are presented in greater detail below.

Microcystis aeruginosa blooms have occurred in the Delta and the North Bay during July through November of each year since 1999. The colonial form of *M. aeruginosa* is the first recorded toxic phytoplankton bloom in the northern reach of SF Bay and may have been recently introduced because it was not recorded in historic samples taken between 1975 and 1982 (Lehman and Smith, 1991 in Lehman et al., 2005). *M. aeruginosa* can form surface scums and is a nuisance to recreational users, reduce aesthetics and oxygen and can produce microcystin, a hepatoxin to humans and wildlife (Lehman and Walker, 2003; Lehman et al., 2005; Lehman et al., 2008). Concentrations found at Benicia, in Suisun Bay, and at Chips Island were low relative to upstream locations (Lehman et al., 2005) perhaps because of dilution or cell death at higher salinities (Lehman et al., 2008). Blooms occurred at salinities less than 18 ppt, although growth was probably limited to <7 ppt (Lehman and Walker, 2003; Lehman et al., 2005; Lehman et al., 2008).

Several surveys of *M. aeruginosa* blooms have documented that the blooms can be widespread, often with microcystin concentrations that exceed World Health Organization guidelines for risks to humans and wildlife (e.g., Lehman and Walker, 2003; Lehman et al. 2005; Lehman et al., 2008). For example, Lehman et al. (2005) documented that an extensive *M. aeruginosa* bloom was found to extend 180 km from Benicia to near Rio Vista on the Sacramento River to 20 km downstream from Tracy on the San Joaquin River side of the Delta, with toxicity exhibited at all stations. Concentrations of microcystin were measured in greater concentrations in zooplankton and clam tissue relative to algal tissue although concentrations were not greater than lethal limits known to cause acute death (Lehman et al., 2005; Lehman et al., 2008). This appears to support the hypothesis that microcystin are transferred or perhaps biomagnified in the food web, the exceptions being clams which appear to be able to depurate toxins from their tissue rapidly (Lehman et al., 2008). However, concentrations they found may be chronically obstructive to food quality, feeding ability, growth, and fecundity in zooplankton (Lehman et al., 2008). Given *M. aeruginosa* seems to prefer high light and warm shallow water eutrophic conditions, any change in the management of the flows from the Sacramento River that leads to increased or more persistent but steady flow rate and improved salinity stratification may expand the population in the late summer/autumn. Given the potential threats to humans and wildlife, Lehman et al. (2005) recommended annual monitoring and further assessment of the causes and controls on this species.

Table 5.2.2. Reported harmful algal blooms in San Francisco Bay since 1995 (See Figure 5.2.5 for approximate locations and extent of blooms).

Map ID	Author	Bloom Type	Bloom Location(s)	Bloom Date(s)
1	Lehman and Waller, 2003	Cyanobacteria: *Microcystis aeruginosa*	Delta	July-November, 1999-2002
2	Herndon et al., 2003	Red Tide: raphidophyte *Heterosigma akashiwo*	Richardson Bay	June, July, and Sept 2002
3	Cloern et al., 2003	Red Tide: raphidophyte *Heterosigma akashiwo*	Central Bay	September 2002
4	Lehman et al., 2005	Cyanobacteria: *Microcystis aeruginosa*	180 km of waterways in northern SF Bay (Carquinez Straight to Suisun and Rivers segments).	October 2003
5	Lehman et al., 2008	Cyanobacteria: *Microcystis aeruginosa*	Rivers	August, September 2004
6	Cloern et al., 2005	Red Tide: dinoflagellate *Akashiwo sanguinea*	Central and South Bay (Angel Island down into South Bay)	September 2004

Red tides associated with a bloom of *Heterosigma akashiwo* have occurred in Richardson Bay (Herndon et al., 2003). Three bloom events were observed in northern Richardson Bay during the summer and autumn of 2002 and all coincided with clear skies, warm air temperatures >25ºC, and calm and warm (>20°C) waters (Herndon et al., 2003). The blooms were a near monoculture with other species comprising <7% of the samples (by cell count) (Herndon et al., 2003). A fourth bloom occurred between September 1 and 12 and covered a wider geographic area including most of the coastline of Tiburon Peninsular over to the Berkeley frontage (Herndon et al., 2003). That same year it was identified by O'Halloran et al., (2006) at the Berkeley pier in April and September. This harmful alga is a new occurrence and has been associated with fish kills in other coastal ecosystems (Cloern et al., 2003). In this case it was widespread outside of the Golden Gate with similar reports at Stinson Beach and in Bodega Bay. Although there was some evidence that the bloom was seeded from the near-field ocean, it is not clear what other factors including nutrients supplied from terrestrial sources, turbulence, and temperature played in bloom sustenance and degradation.

Figure 5.2.7. Harmful algal bloom (HAB) occurrences reported in the literature since 1995. Large segments show locations of HABs as reported in the literature (usually from a diagram) and small segments indicate general location of HAB in which more descriptive location information was not provided in the literature. Locations are approximated based on location description in the referenced journal publication.

Red tides associated with a bloom of the dinoflagellate species *Akashiwo sanguinea* have been observed in the southern area of Central Bay (September 2004). This species is not normally described as a HAB but can be potentially disruptive if biomass is high and is indicative of recent increases in red tides on coastal California (Kudela et al., 2008). An advantage of this and other dinoflagellates is their ability to move lower in the water column to feed on nutrients during the night hours and reside within the photic zone during daylight. The bloom, which had chlorophyll a concentrations approaching 200 µg L^{-1}, coincided with unusually weak neap tides, calm winds, and four consecutive high air temperature days creating a shallow (<3 m) surface layer above a thermocline that persisted long enough for the motile dinoflagellate species to proliferate (Cloern et al., 2005a,b). This bloom reduced ammonium and then nitrate concentrations to some of the lowest concentrations measured (Cloern et al., 2005a,b; Wilkerson et al., 2006). While climatic conditions were intimated as the key factor in bloom development, the bloom followed a summer of weak coastal upwelling and high dinoflagellate biomass in nearfield coastal waters, apparently providing seed organisms (Cloern et al., 2005b).

A common theme emerging from observations of all recent HAB blooms have been there occurrence in the summer and autumn months, perhaps associated with the decline of the spring and summer diatom blooms and consumption of regenerated nutrients. If blooms become more common and magnitude increases, the occurrences of hypoxia/anoxia may also rise in relation of higher punctuated organic matter loading and resulting biological oxygen demand (BOD).

5.2.5 Utility of Phytoplankton Biomass, Productivity, and Community Composition as an NNE Indicator for San Francisco Bay

Clear Linkage to Beneficial Uses

Phytoplankton has a well-documented linkage to beneficial uses of SF Bay. Phytoplankton are the dominant primary producer in SF Bay, and therefore the ultimate source of carbon for the entire food web (Cole and Cloern 1982). Food supply is smaller than in many other estuaries (largely because SF Bay is turbid) and, as a result, consumers such as zooplankton, mysid shrimp, and clams are limited by carbon productivity (Cloern et al., 2003). Only during blooms is the rate of carbon production sufficient in SF Bay to keep pace with consumption (Cloern, 1996). Additionally there is compelling evidence of the linkage between phytoplankton and the pelagic foodweb. For example, there was a remarkable change in phytoplankton post 1986 in Suisun Bay when the *Potamocorbula amurensis* (now called *Corbula amurensis*; Coan, 2002, referenced in Wilkerson et al., 2006) was introduced. The summer bloom was decimated and primary production decreased around 2.5-fold from 106 g C m^{-2} to just 39 g C m^{-2} (Alpine and Cloern, 1992) arguably directly caused by increased consumption faster than phytoplankton reproduction by the invasive clam and perhaps leading to massive failures in several competing pelagic organisms, the copepod *Eurytrmora affinis* and the native mysid shrimp *Neomysis mercedis* (Orsi and Mecum, 1996). Note that Wilkerson et al. (2006) more recently argued that grazing could not be the dominant cause of low phytoplankton in Suisun Bay because surface growth rates are an order of magnitude less than clam pumping rates, the similarity of NH_4 uptake rates between Central, San Pablo, and Suisun Bays despite differences in clam populations, and the fact that the clam population is depressed during the spring bloom period. There is also evidence that phytoplankton biomass is linked to water clarity (May et al., 2003). Recently, Schoellhamer (2009) provided evidence that all regions of the Bay are showing decreasing trends in turbidity mainly associated with declines in suspended sediment loads (McKee et al., 2006). This is likely one factor that is contributing to increasing trends in primary productivity.

Although there is clearly complexity, these studies provide a broad base of evidence that phytoplankton have a direct linkage to important SF Bay beneficial uses, including food web support for marine and estuarine aquatic organisms (EST, MAR) including the commercial and sport fisheries (COMM), shellfish such as clams, oysters and mussels (SHELL and AQUA), migratory (MIGR) birds and fish, support for fish nursery habitat (SPAWN). Harmful algal blooms can adversely affect the health of humans (REC-1) by irritation and injury to recreational swimmers, sailboarders, and boaters (Lehman et al., 2005). In addition, elevated phytoplankton biomass could impact estuarine and wildlife habitat by shading and degrading eelgrass habitat and impact aesthetics (REC-2) through nuisance buildup and smell during decay.

Predictive Relationships to Causal Factors

Use of phytoplankton as an NNE indicator for SF Bay requires the ability to develop a predictive model that links phytoplankton response variables back to nutrient loads and other causal factors. Specifically, this requires, at minimum, the development of models that establish the relationship between nutrient loads and phytoplankton response (biomass, productivity, or assemblage). These models can be empirical or computer spreadsheet or dynamic simulation models.

There has been some success in relating empirical phytoplankton to both external nutrient loads and *in situ* nutrient concentrations in some estuaries, particularly when data are averaged over annual time periods. Table 5.2.3 shows relatively high correlation coefficients published by various authors for both phytoplankton biomass and production. In general, variations in N loading rates are reflected in concentrations of N in receiving water bodies, particularly when the residence time of that water body is long (on the order of weeks). Although many processes act at various rates to modify nutrient concentrations, mean total nitrogen (TN) concentrations are significantly correlated to TN loading for 5 sub-systems of Chesapeake Bay averaged over a decadal period (Boynton and Kemp 2008). Conley et al. (2000) reported that on an annual basis about 70% on the variation in TN concentration could be explained by variation in TN loads in a large sample of Danish estuaries. Madden et al. (2010) found a strong correlation between SEAWIFS remotely sensed chlorophyll a and TN loading for 108 estuaries in the United States. A survey of the fundamental nutrient forms and processes in several major estuaries was performed by Smith (2006) using data from 92 estuarine and coastal sites worldwide. It demonstrated a strong correspondence between log transformed annual mean concentrations of total P and N and standing stock of chlorophyll a.

Table 5.2.3. Modeled relationships between nutrient loading and phytoplankton response in world estuaries. (From Boynton and Kemp, 2008).

Location	Variable, X (units)	Variable, Y	r^2 / n	Reference
Multiple estuaries	TN-loading (g N m^{-2} y^{-1})	Phytoplankton Production	0.60 / 14	Boynton et al. 1982
SF Bay and other estuaries	Composite parameter X = f(B, Z_p, I_0)	↓	0.82 / 211	Cole and Cloern 1987
Narragansett Bay	Composite parameter X = f(B, Z_p, I_0)	↓	0.82 / 1010	Keller 1988
Multiple estuaries	DIN-loading (mol N m^{-2} y^{-1})	↓	0.93 / 19	Nixon et al. 1996
Multiple estuaries	TN-loading (g N m^{-2} y^{-1})	↓	0.36 / 51	Borum and Sand-Jensen 1996
Boston Harbor	Composite parameter X = f(B, Z_p, I_0)	↓	0.66 / 12	Kelly and Doering 1997
Waquoit Bay system	Annual average DIN conc (µM)	↓	0.61 / 12	Valiela et al. 2001
Chesapeake Bay	TN(x_1), TP(x_2) load (kg mo^{-1})	↓	0.67 / 11	Harding et al. 2002
Multiple estuaries	DIN (mM m^{-3}); tidal range (m)	Phytoplankton Biomass	na / 163	Monbet 1992
Multiple systems / MERL	DIN input (mmol m^{-3} y^{-1})	↓	na / 34	Nixon 1992
Ches Bay mesohaline	River flow (m^3 d^{-1}); proxy for N-load	↓	0.70 / 34	Harding et al. 1992
Maryland lagoons	TN load (g N m^{-2} y^{-1})	↓	0.96 / 9	Boynton et al. 1996
Danish coastal waters	TN concentration (ug l^{-1})	↓	0.64 / 168	Borum 1996
Canadian estuaries	TN concentration (ug l^{-1})	↓	0.72 / 15	Meeuwig 1999
Ches Bay and Tributaries	TN Load; (mg N m^{-2} yr^{-1}) (R_{time}, yrs)$^{-1}$	↓	0.82 / 17	Boynton and Kemp 2000
Danish estuaries	TN concentration (ug N l^{-1})	↓	0.30 / 1347	Nielsen et al. 2002

However, San Francisco Bay has long been recognized as an estuary in which phytoplankton biomass and pelagic primary productivity is not driven by simple nutrient-limitation, due to a variety of co-factors that modulate primary producer response to nutrients (Figure 5.2.8, Cloern and Dugdale, 2010). Substantial effort has gone into the development of empirical relationships between phytoplankton and causal indicators in SF Bay (Cloern and Cheng, 1981; Lucas et al., 1998; Lucas, et al., 1999a,b; Lucas and Cloern, 2002; May et al., 2003; Thompson et al., 2008; Lucas et al., 2009). Typically the basis of the models has been temperature, light (surface irradiance and photic depth), stratification, predation, and senescence. The premise of Cole and Cloern (1987) that 80% of the spatial and temporal variability in productivity is correlated with variations in three easily measured parameters (phytoplankton chlorophyll a, photic depth, and surface irradiance) seems to largely hold true. In addition to these main factors, modelers have explored other cofactors such as turbidity and transport. In no single case have the authors used nutrient concentrations or external loads in the bloom initiation components of the models, however, the limitation of phytoplankton biomass by nutrients deserves more study (Thompson et al., 2008).

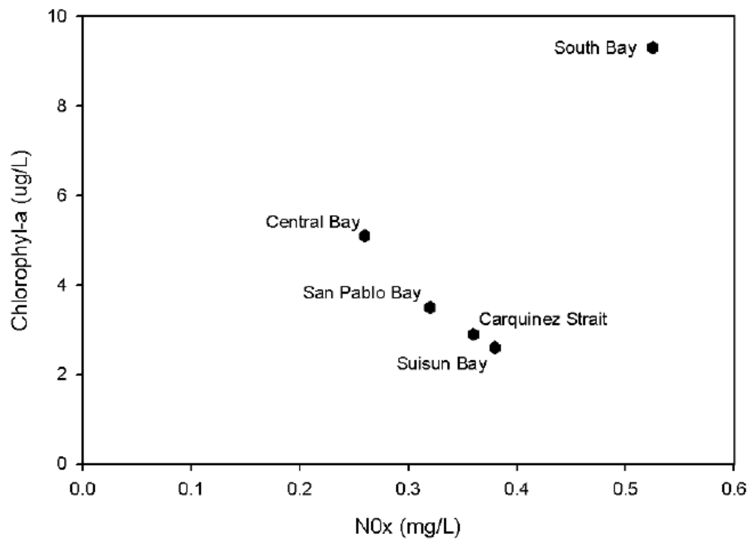


Figure 5.2.8. Mean chlorophyll a concentrations as a function of mean nitrate+nitrite (NOx) concentrations in San Francisco Bay for the period January 1999 to February 2009 (Source: James Cloern, USGS: http://sfbay.wr.usgs.gov/access/wqdata).

From nearly four decades of research, a number of common themes have emerged about the relationships between phytoplankton production and causal or limiting factors: 1) Light limitation plays a role in bloom production and magnitude in all reaches of the Bay; if turbidity continues to decrease, overall production will likely increase; 2) in all areas of the Bay, nitrate and phosphate never limit the initiation of spring bloom phytoplankton production; 3) in all areas of the Bay, predation by bottom dwelling filter feeders limit the maximum bloom magnitude as can depletion of nitrate more occasionally; 4) in the North Bay, the ratio of ammonium to nitrate along with salinity based stratification and the magnitude of freshwater outflow appear to control initiation of spring blooms whereas in the South Bay, the timing and magnitude of blooms appears to be controlled by the rate of recruitment of macro-invertebrates in the late winter and spring; and 5) greater winter runoff (and nutrient supply) can increase the size of spring blooms in both the North and South Bays, however, the occurrence is controlled by other factors; this implies that if at some future time blooms cause impairment of the Bay, nutrient load controls may only have the potential to bring the Bay back to its current “2010” condition. Another general theme that has emerged is the role of macro-invertebrates as both response variable as a secondary consumer that bloom in response to availability of planktonic food resources, and as a cofactor causal variable where the presence or absence of winter populations can strongly influence bloom initiation.

Northern SF Bay (North Bay) has undergone radical changes in ecosystem structure, from nutrients to phytoplankton and up the food chain to zooplankton and fish. Although much of these changes have been ascribed to the appearance of invasive species, e.g., of zooplankton and bivalves, a new analysis of the long term time series for Suisun Bay has provided a bottom-up view of the causes of these changes based on principles of ecological stoichiometry (Glibert, 2010). Glibert describes an initial change from a diatom-based foodweb that began in 1982 when the Sacramento Regional Sewage Treatment plant came on line and converted to secondary treatment, releasing NH4 into the Sacramento River, eventually discharging 15 tons N daily by 2002 (Jassby, 2002). Phytoplankton blooms became rare in Suisun Bay after 1987 coincident with the arrival of the Asian clam, Corbula amurensis. However, the diatom population had been declining for the previous 5 years, now believed to be the result of increasing NH4 input (Wilkerson et al., 2006; Dugdale et al., 2007; Glibert, 2010). The composition of the zooplankton also changed as the phytoplankton community became dominated by Chryptophytes and Flagellates and the growth rate of the delta smelt declined. In this scenario, the causal agent was the conversion of the Sacramento River from a nitrate-based diatom phytoplankton system, to an NH4 –based, small phytoplankton, small zooplankton (*Eurytemora, Pseudodiatomus, Limnothiona*, and the introduction of the invasive clam which came to dominant phytoplankton blooms in some times of the year (not in spring when clam populations are seasonally low). The next major perturbation in the northern estuary nutrient regime was the decline in phosphate, another anthropogenic effect as phosphate was removed from detergents on the basis of fear of eutrophication. The northern estuary then became an enriched NH4-N, low P ecosystem, which now favored the development of Cyanobacteria, in particular Microcystis blooms began to occur, as that group is able to grow well at low P concentrations and compete with the Cryptophyte and Flagellate functional groups, diatoms having been eliminated by the blocking of access to nitrate by high NH_4 concentrations. These three stages in the degradation of the Suisun Bay ecosystem are diagrammed (Figure 5.2.8).

Although the changes described above are well documented, there is no consensus among the scientific community that these changes can be attributed to a single factor, such as wastewater inputs of ammonium. For example, recent analyses of population declines of pelagic fish and their food resources in the upper estuary show strong associations with changes in water clarity, export flows, and salinity distribution measured as X2 (Mac Nally et al., 2010; Thomson et al., 2010). The conclusion of Dugdale et al., 2007, that the northern SF estuary is locked most of the time in an NH_4-based low-primary productivity condition (due to NH_4 control of access to the larger N pool, NO_3) is not incompatible with changes in productivity driven by other factors, e.g., in the NH_4 based system, an increase in transparency will result in an increase in the depth integrated primary productivity, and other modulating factors will also come into play to increase or decrease productivity. Therefore, the SF technical advisory team views that much remains to be learned about the ecological consequences of ammonium enrichment in the context of other drivers of population declines, and recommends formation of a working group to review the relevant data and identify and evaluate potential ammonium endpoints.

With respect to South Bay, most recently, Cloern et al. (2010) presented an analysis of the influence of the Pacific Decadal Oscillation (PDO) on biomass trends in SF Bay. They found that populations of demersal fish, crabs, and shrimp covary with PDO and the North Pacific Gyre Oscillation (NPGO) accounting for upwards of two thirds of the variability. They argue in this and their previous paper

(Cloern et al., 2007) that population decreases in these predators after 1999 were followed by declines in bivalve suspension feeders, and increasing abundance of phytoplankton biomass. It appears that at the scales of decades, the supply of nutrients may not be strongest driver of productivity in an estuary where nutrients are not limiting; the existence of a unique and long term phytoplankton time series for SF Bay appear to suggest responses on the decadal scale to climate variability.

In order to capture the complexity of these relationships, the consensus among the SF Bay NNE Technical Team is that computer models are required to predict watershed, airshed loadings and oceanic exchange with SF Bay and models that simulate response of the Bay to nutrient loads and other factors. Dynamic simulation models are mathematical representations of the real world that estimate environmental events and conditions. Models can be used to predict pollutant delivery as well as simulate how various changes or pollution-reduction actions could affect a waterbody's beneficial uses, especially with respect to water quality, aquatic life, and wildlife. Because estuaries and their watersheds are typically complex, scientists and managers can rely on computer models to synthesize information about the ecosystem's characteristics and the effects of various environmental actions to reduce pollution. To-date, no comprehensive predictive calibrated model exists for the Bay or the watershed, airshed or ocean that is able to couple forcing factors, co-factors, and biological response. Instead, these models have tended to support empirical observations

The conceptual approach to development of models for the SF Bay estuary could be similar to that done for the Chesapeake Bay Estuary (http://archive.chesapeakebay.net/pubs/backgrounder_CBP_Models.pdf), in which models were developed and refined through a 30 years of collaboration by federal, state, academic and private partners. Initially, two types of models would need to be developed:

1) Watershed, airshed, and oceanic exchange model, which incorporates information about loadings or exchanges from land use, fertilizer applications, wastewater plant discharges, septic systems, wet and dry air deposition, exchange with the coastal ocean, weather and other variables to estimate the amount of nutrients and sediment reaching the SF Bay estuary and where these pollutants originate. The watershed model would include three components:
    - A hydrologic sub-model that uses rainfall, evaporation and meteorological data to calculate runoff and sub-surface flow for all land uses.
    - A non-point source sub-model, which simulates soil erosion and nutrient loads from the land to rivers, driven by the hydrologic sub-model
    - A river sub-model which routes flow and associated nutrient loads from the land through lakes, rivers and reservoirs to the Estuary.
    - An ocean exchange submodel can be used to force exchange of flow, chemical and biological constituents across the estuarine-oceanic boundary.

2) Estuary water quality model, which simulates the ecosystem response to pollutant loads, which would consist of two sub-models

- A hydrodynamic sub-model that will simulates the exchange with rivers, oceans, mixing of waters in the Estuary and its tidal tributaries.
- A water quality sub-model that simulates the Estuary's biological, chemical and physical dynamics in response to nutrient loads and other factors (light, temperature, grazing, etc.).

The models would be used to establish load allocations of nutrients that the SF Bay estuary can sustainably assimilate. It would also be used to generate simulations of the past, present or future state of the Estuary, ocean, watershed, and airshed (e.g., population growth, climate change, etc.) to explore potential effects of management actions and evaluate alternatives. Thus these models would be a key component of a strategy to adaptively manage SF Bay.

Ideally sufficient data and knowledge of SF Bay should exist to support the development of system wide dynamic simulation models to predict phytoplankton biomass/community response and relationships to models of secondary productivity. At the macro scale, the relationship between phytoplankton based primary production is a given; phytoplankton need nutrients to grow; nutrient loads to each Bay compartment, the standing nutrient mass, and speciation exert a strong control on primary production and are primary causal variables. However, as shown in Figure 5.2.8, the relationship for the Bay is complex. Empirical models as well as any subsequently developed dynamic simulation models will need to capture this complexity in order to be of use in a management context.

Unfortunately, there are some data sets that are less well developed that may also hamper the development of predictive models and a fuller understanding of the causes of change. For example, we have no reliable annual estimates of nutrient loads from either the Central Valley or local tributaries and current annual average loads can only be considered 1st order estimates (See section 4 of this report). We presently have only limited understanding of sediment loads entering the Bay from local tributaries although recent evidence suggests that this source may now be larger than Central Valley loads (Lewicki and McKee, 2009). Data on HABs for SF Bay are presently weak due mainly to limited recent occurrences and few research programs aimed at such ephemeral events. There is a lack of data on the causes, timing, extent of toxin production (influences and concentrations), nutrient consumption during blooms, and processes leading to bloom termination and predictive models that link HAB production to nutrient loads are lacking. There is presently no systematic monitoring program for either zooplankton benthic macro-invertebrates or eelgrass and other submerged aquatic vegetation. Given the rate of change of these populations in recent decades (see later sections); we suggest that comprehensive surveys of these system components should be completed about every five years. In addition, Cloern et al. (2006) point out that our understanding of key processes of change is limited by the lack of systematic measurement of phytoplankton growth and transport rates, and there is no systematic measurement of phytoplankton biomass outside the Golden Gate. There is presently no systematic collection of urea concentrations and there has been no research done to-date on the role of ammonium or urea limitation on phytoplankton growth rates in the Central and South Bays.

To-date no system-wide dynamic simulation model has been developed, but given the spatial and temporal data richness of the system and the basis provided by existing sub-system models, of any system, SF Bay hold great promise for the development of such a unified model. Wilkerson et al., 2006 suggest in their closing statement that: “The effect of water management changes, i.e., changing DIN loading, can now be modeled using these nitrogen productivity data as a framework to understand the importance of different nutrient concentrations in the development of phytoplankton blooms in the northern SFB.” In addition, given that the majority, if not all the parameters that supported the Chesapeake Bay, Bay Health Index (BHI), have been collected in SF Bay, we can’t help but conclude that further effort to develop such an index for SF Bay may yield a useful tool for tracking and predicting nutrient related water quality changes in SF Bay.

Sound and Practical Measurement

Phytoplankton is among the best studied assemblages of estuarine organisms, with over six decades of research. As a consequence, there are a variety of sound and scientifically well-vetted means of measuring phytoplankton biomass, productivity, community composition, and growth efficiency (Table 5.2.4). See Sutula (2011) review of phytoplankton indicators for the estuarine NNE for additional detail. In addition, San Francisco Bay has the advantage of an established long term USGS research program on phytoplankton that began in 1977 and currently spans 29 years and that includes >11,000 discrete laboratory measurements of the chlorophyll a in water samples and 156,610 estimates of chlorophyll a made from a linear relationship between fluorometer voltage and discrete lab measurements. Thus, a long-term data set exists to support decisions on regulatory endpoints as well as for use in developing a load-response model. It is important to note that the USGS research program is not mandated. There is the critical need for a commitment to support regular sampling to measure and understand future changes in water quality, including those related to nutrient enrichment.

Table 5.2.4. Summary of methods for measuring phytoplankton biomass and community structure (from Sutula, 2011).

Group	Indicator	Methods	Information
Water Clarity	Light Attenuation	Grab samples or Continuously deployed Instrumentation (e.g., data sondes)	Attenuation of light reaching bottom
	Turbidity or TSS		Turbidity or TSS
	Secchi Depth	Field survey	Coarse measure of water clarity
Phytoplankton Biomass	Discrete water column chlorophyll a	Grab samples with laboratory analysis	Precise measure of water column chlorophyll-a
	Chlorophyll a fluorescence	*In situ* probes and flow through instrumentation	Chlorophyll a fluorescence, which must be calibrated to grab samples
	Remote sensing of color	Satellite (SeaWiFS, MERIS, MODIS) or wide variety of multispectral and hyperspectral airborne sensors	Water color as a proxy for chlorophyll a
Phytoplankton Productivity	Photosynthesis versus irradiance curves	Modeled production	Rate of carbon fixation per unit time per square meter (areal) or cubic meter (volumetric)
	Isotope	Direct measure of gross productivity, respiration, and net productivity	
Phytoplankton Community Structure	Number of species and relative abundance	Taxonomy and cell counts	Dominant species and presence/absence of rare or pollutant tolerant taxa
	Chemotaxonomic phaeopigments	HPLC	Relative composition of broad taxonomic group composition by determining chlorophyll and carotenoid presence phaeopigments (e.g., Chlorophytes, Cryptomonads, diatoms, dinoflagellates and zeaxanthin)
HAB species and toxin concentrations	HAB species abundance	Taxonomy/cell counts or Q-PCR	Abundance of HAB species
	Toxin concentrations	HPLC or Elisa Assay	Concentration of toxins associated with water column or sediment

Acceptable Signal to Noise Ratio

Phytoplankton are subject to a high degree of spatial and temporal variability (see previous sections summarizing this variability), due to a number of physical, chemical and biological co-factors. However, given the long-term data set available for SF Bay, it has been possible to determine statistically-significant trends with respect to phytoplankton biomass at the decadal time scale (e.g., Cloern et al., 2006). Thus our ability to use phytoplankton as an NNE indicator in SF Bay is possible because of this 39-yr data set. However, this may be too long of a time scale to be useful for determining the cause of more ephemeral system responses such as HABs. Smetacek and Cloern (2008) comment that because phytoplankton species populations appear and disappear within weeks, assessing change on shorter timescales may require higher resolution monitoring of annual cycles over many years. It should be noted that while high spatial and temporal variability is characteristic of all biological indicators, these indicators tend to integrate better over time and space than stressors, such as nutrient concentrations. Ultimately, our understanding and the various hypotheses about controls on spatial and temporal variability in phytoplankton biomass, productivity and community structure and linkages to consumers can be tested and refined through predictive models.

The use of any particular indicator of phytoplankton (biomass, productivity, assemblage) alone to assess eutrophication is not recommended. Each of these indicators has strengths and limitations which, when measured and used as multiple lines of evidence provide a more holistic assessment of adverse effects with an acceptable signal:noise ratio. By contrast, use of any single indicator may produce a false negative or positive assessment of adverse effects without supporting information.

For example, phytoplankton productivity is the measure of the rate of biomass production and is in fact a more immediate measure of the influence of nutrients on autotrophic production and potential eutrophication than biomass alone. Cole and Cloern (1984) showed that regions of the Bay which may have higher chlorophyll a do not necessarily provide a net transfer of carbon to herbivores because, in some cases, respiration exceeds gross production in deeper or more turbid areas. Therefore, high chlorophyll a alone is not necessarily indicative of trophic status if that high chlorophyll a is the depth of the photic zone is limited to a fraction of total water depth. In addition, productivity and assemblage information (e.g. % diatoms) have a much stronger linkage to beneficial use than biomass per se.

However, though the rate of productivity may be a good indicator of nutrient concentration, the ultimate disposition of the production may vary across estuaries or even within an estuary based on several factors. High productivity in deep and well-mixed waters may not result in problematic levels of phytoplankton biomass as the biomass produced can be mixed throughout the water column, and the balance of productivity to respiration (P:R) within the entire water column constrains the production within acceptable limits. Moreover, even in shallow estuaries where biomass may accumulate in the euphotic zone, if grazer or filter feeding communities are present, the biomass may be efficiently removed, contributing to a healthy and productive estuary, without causing negative impacts. Second, direct measures of productivity are relatively difficult and time consuming, so gathering data over a large and representative spatial area is not typically widely conducted in monitoring programs for coastal waters (Anderson et al. 2006).

Phytoplankton indicators can be used in tandem to provide information not only about the accumulation of organic matter in the system, but also information about the health of the phytoplankton community and factors that may lead to trophic level changes that underpin key estuarine beneficial uses. For example, the ratio of productivity: biomass is an index of growth potential and is a meaningful indicator of the physiological state of phytoplankton from ammonium or other toxic contaminant (Yoshiyama and Sharp, 2006).Thus a combination of measures of phytoplankton biomass, productivity, and assemblage are needed in order to make a more robust assessment of adverse effects of nutrient over-enrichment or eutrophication.

Approaches to Setting Numeric Endpoints Based on Phytoplankton

Paradigms for establishment of estuarine numeric endpoints based on phytoplankton typically separate seagrass from subtidal unvegetated habitats. For seagrass, precedents for establishment of numeric endpoints exist based on biomass, based on light limitation for photosynthesis of seagrass beds (e.g., Janicki et al., 2000; Kemp et al., 2004; Brown et al., 2004; Sutula, 2011). Turbidity, total suspended solids (TSS), chlorophyll a, and dissolved organic matter are measured to determine light available in the water column that reaches the seagrass bed (Biber et al., 2008). For example, In the mid-Atlantic, environmental conditions that allow adequate light penetration for SAV survival are total suspended solids (TSS) less than 15 mg L^{-1} and chlorophyll a less than 15µg L^{-1} (Kemp et al., 2004). Bio-optical models predicting light attenuation under various environmental conditions have been calibrated for the Chesapeake Bay (Gallegos, 2001), Indian River Lagoon in Florida (Gallegos and Kenworthy, 1996), and North River in North Carolina (Biber et al., 2008), Yakina Estuary in Oregon (Brown et al., 2007), and Tampa Bay in Florida (Janicki et al., 2000). Explicit studies are needed to understand the precise light requirements of seagrass in SF Bay. This information can be used to develop a bio-optical model that could be used to establish a combination of chlorophyll a thresholds and turbidity to establish levels of light attenuation that will be protective of SF Bay seagrass beds (see Section 5.4 for further discussion).

For unvegetated subtidal habitats, some precedent for setting chlorophyll a endpoints for biomass (e.g., Bricker et al., 2003; Soucho et al., 2000; Ferreira et al., 2006; Zalidvar et al., 2008) and phytoplankton productivity (Devlin et al., 2007) to assess eutrophication exists, though use of phytoplankton for regulatory purposes is not widespread. Ultimately, confidence in setting NNE endpoints based on biomass, productivity and/or community structure is more easily accomplished with long-term data sets that describe the range in variability in these indicators and relationship to consumer communities linked to beneficial uses. In SF Bay, this would be done by convening a workshop of experts to synthesize data that could be used to establish thresholds based on biomass, productivity and community structure.

With respect to HAB species abundance and toxin concentrations, experience with establishing numeric thresholds is more evident for freshwater cyanobacteria species such as *Microsystis spp*. A summary underway of suggested action levels for adverse health effects of anatoxin-a, cylindrospermospin, and four microcystins (LA, LR, RR, and YR) by California EPA Office of Environmental Health Hazard Assessment will be an excellent starting point for consideration of numeric endpoints for cyanobacteria. That report is currently in peer review. These thresholds are most applicable for oligohaline environments, where cyanobacteria are most prevalent, but should also be considered for downstream impacts to polyhaline or euhaline habitats, as cyanobacteria toxins such as microcystin can accumulate in marine invertebrates and thus adversely affect marine mammals. As an example of this, Miller et al. (2010) found that microcystin poisoning was the likely cause of death in sea otters in Monterey Bay Marine Sanctuary.

For estuarine or marine HAB species typically found in California, there is a lack of understanding on the controls of relative abundance and toxins production that limit our ability to use these as NNE indicators at this time. Additional research is needed to understand controls on marine HAB frequency and occurrence and controls on toxin production. Additional work is required to understand chemical controls on community structure (ammonium, trace elements, and micronutrients).

5.2.6 Summary: Use of Phytoplankton as an NNE Indicator

Overall, phytoplankton appears to satisfy the four evaluation criteria to be considered as an NNE indicator for SF Bay. However, several key data gaps and recommended next steps are required in order to further pursue its use for this purpose. The steps are:

1. Select the precise indicator and numeric endpoints

   The following indicators are recommended for use in the SF Bay NNE: 1) phytoplankton biomass, 2) productivity, 3) phytoplankton assemblage and in particular, HAB species abundance and toxin concentrations. The SF Bay TAT recommends a series of expert workshops to synthesize data, identify data gaps and create a phytoplankton assessment framework that would be used by policy makers to set numeric endpoints based on these indicators.

2. Scope the development of a series of dynamic watershed, atmospheric, and oceanic loading and SF Bay hydrodynamic and water quality models to simulate the ecological response of the Bay to nutrient loads and other factors. This would be done through a series of workshops to develop a modeling strategy for SF Bay. The product of this effort would be the identification of the appropriate models, a phased workplan, timeline and budget to develop these models, and identification of and coordination among key institutions, programs and respectively roles. This scoping must include three elements:
   1) *Conceptual Model Development*. There is a need to develop conceptual models that explicitly show linkage between watershed, airshed, ocean and estuarine hydrology, nutrient loads, ecological response indicators, and "co-factors" that control ecological response to eutrophication or oligotrophication. The conceptual model would identify key sources, sinks and

processes of transformation that would need to be incorporated into the models. Areas of disagreement on causal mechanisms should be synthesized as alternative hypotheses than can be tested through experiments, field studies and model sensitivity analyses.

2) *Model Selection*. The next step in the scoping of model development is to select the appropriate models. This should be done by reviewing available loading and receiving waterbody models and present an analysis of the advantages and disadvantages of their use for modeling eutrophication and other adverse responses to nutrients, based on the explicit conceptual models.
3) *Data Needs Assessment*. Based on explicit conceptual models and the modeling platform selected, the next step would be to identify data required to support model development, calibration and validation.

This information could be synthesized into a workplan to develop the loading and estuary water quality models and a preliminary timeline and budget for Phase I of the effort.

5.3 Macroalgae

Macroalgae are an ancient group of single to multicellular primary producers found in all aquatic ecosystems. They provide the same ecological functions as vascular plants in terrestrial ecosystems, but lack the structural tissues characteristic of plants. Marine macroalgae form an important component of productive and highly diverse ecosystems in estuaries worldwide and in moderate abundances provide vital ecosystem services. They are important primary producers in intertidal and shallow subtidal estuaries, providing food and refuge for invertebrates, juvenile fish, crabs and other species. However, some species of macroalgae thrive in nutrient-enriched waters, out-competing other primary producers (Sutula, 2011). For this reason, macroalgae have been proven to be useful indicators of eutrophication in estuaries. Estuarine ecosystems have been subjected to increased frequencies and magnitudes of harmful macroalgal blooms, outcompeting seagrasses and other primary producers and resulting in hypoxia, reduced biodiversity, fish and invertebrate mortality, altered food webs and energy flow, and disruption of biogeochemical cycling (Sfriso et al., 1987; Valiela et al., 1992, 1997; Coon, 1998; Young et al., 1998; Raffaelli et al., 1989; Bolam et al., 2000). Fong, Green and Kennison provide a detailed review of the utility of macroalgae as an NNE indicator in estuaries (Chapter 5, Sutula, 2011). This section provides a brief synopsis of that work and presents literature directly relevant to known abundance, distribution and variability of macroalgae in SF Bay estuary.

5.3.1 Applicable Habitat Types

As an NNE indicator, macroalgae are most applicable to intertidal or shallow subtidal habitat, including seagrass beds. Macroalgae are also applicable to diked Baylands and salt ponds.

5.3.2 Available Data on Macroalgae in San Francisco Bay

No regular program of monitoring of macroalgal abundance exists in SF Bay. A survey of macroalgal abundance in seagrass beds is slated for completion in Spring 2011.

5.3.3 Macroalgal Relationship to Nutrients and Water Quality

Macroalgae are important members of the primary producer community in rocky and shallow soft-sediment systems worldwide where light penetrates to large areas of the benthos. They are present in all estuarine geoforms, but their relative abundance is, at least in part, proportional to the amount of suitable habitat and nutrient supply. In oligotrophic systems, macroalgae are a component of the primary producer community, but are generally not dominant (Figure 5.3.1). Rather, in shallow subtidal and intertidal portions of these estuaries, benthic communities may be dominated by the microphytobenthos (MBP), an assemblage of diatoms, dinoflagellates, cyanobacteria, and sporling green macroalgae living on the sediment surface that can contribute up to 50% of the primary production in an estuary (Underwood and Kromkamp 1999). In larger, well-flushed California estuaries, shallow subtidal portions are often dominated by seagrasses.

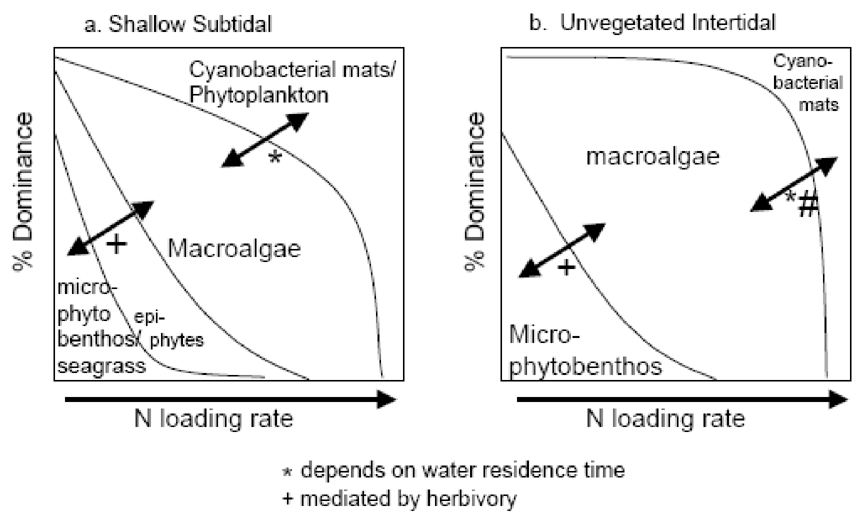


Figure 5.3.1. Conceptual model of the relationships between N loading rate and the community composition of primary producers in a) shallow subtidal and b) unvegetated intertidal habitat in California estuaries.

As nutrient availability increases, it has been well-documented in many parts of the world that blooms of green or red macroalgae become dominant in shallow subtidal and intertidal estuaries and lagoons, replacing seagrass or MPB (Figure 3.2; e.g., Sfriso et al., 1987, 1992; Raffaelli et al., 1989,; Valiela et al., 1992, 1997; Peckol et al., 1994; Marcomini et al., 1995; Hernández et al., 1997; Hauxwell et al., 1998; Kamer et al., 2001). Macroalgal carbon moves more easily through microbial and consumer food webs than carbon derived from seagrasses due to the more labile nature of its carbon (Valiela et al., 1997). In shallow estuaries, macroalgae can dominantly influence the O_2 profile of the water column, further affecting the biogeochemistry of the sediments and exchange of nutrient to and from sediments. Macroalgal blooms affect the abundance of macrofauna through more frequent hypoxia/anoxia events (caused by metabolism of organic matter that depletes the benthos of dissolved oxygen) and habitat changes (Valiela et al., 1997; Cloern, 2001). Thus blooms of opportunistic macroalgae can cause in loss of critical habitat, hypoxia, reduced biodiversity, fish and invertebrate mortality, altered food webs and energy flow, and disruption of biogeochemical cycling (Sfriso et al., 1987; Valiela et al., 1992, 1997; Coon, 1998; Young et al., 1998; Raffaelli et al., 1989; Bolam et al., 2000).

Macroalgae have a well-documented linkage to nutrients and water quality. Common bloom-forming species of Ulva have been used as biological indicators of nutrient supplies in estuaries. They are good indicators because of their ability to rapidly take up large pulses of inorganic nitrogen (Fujita, 1985; Pedersen and Borum, 1997; Lotze and Schramm, 2000; Runcie et al., 2003) and store it for future growth (Fujita, 1985; Bjornsater and Wheeler, 1990; Fong et al., 1994; Pedersen and Borum, 1997; Lotze and Schramm, 2000; Naldi and Viaroli, 2002). Thus, tissue nutrients in macroalgae integrate nutrient supplies over time (Wilson, 1994). This is especially important in Mediterranean systems, where nutrient supply and availability can be variable due to pulses of nutrients that are delivered by runoff from seasonal storms in the wet season as well as during periodic discharges of sewage and agricultural waste in both the wet and dry seasons (Zedler, 1996). In addition, in eutrophic estuaries with organically –enriched sediments, macroalgal biomass accumulation can be partially or wholly supported by benthic regeneration of nutrients; macroalgae have in fact been shown to increases the magnitude of benthic fluxes by increasing the concentration gradient between surface waters and sediment pore waters. Therefore, traditional water column nutrient sampling methods may miss pulsed nutrient signals, and not provide an accurate estimate of nutrient enrichment. With the combination of a high affinity for nitrogen and ability to store nutrients, macroalgal tissue nutrient status can be used as a biological indicator (Harrocks et al., 1995; Fong et al., 1998; Costanzo et al., 2000; Huntington and Boyer, 2008b) to determine nutrient availability.

The effects of nutrient loading rate on macroalgal distribution and biomass accumulation are heavily influenced by the hydrological connection to the ocean of each estuary. Due to the ability of most bloom species to shift habitat usage from benthic to floating stages, macroalgae are able to occupy all estuarine habitats by rafting in surface waters or depositing on subtidal or intertidal sediments. Biomass accumulation, however, is linked to nutrient supply. Thus, low abundances of macroalgae may co-occur in low nutrient systems with subtidal and intertidal seagrasses and the microphytobenthos (benthic microalgal community). It is only as nutrient loads increase that proliferation of macroalgae has negative impacts on other producer groups across all estuarine classes (see Sutula, 2011 for a full review).

5.3.4 Species Composition of Macroalgae in San Francisco Bay

In the SF Bay 162 species of macroalgae have been identified, the most common species are *Ulva clathrata, U. intestinalis, U. linza, U. angusta, U. lactuca* (commonly known as sea lettuce), *Cladophora sericea, Antithamnion kylinii* and *Polysiphonia denudate* (Figure 5.3.2). Besides the last two which are red and brown algae respectively, all of those species are green algae. Other common macroalgal species with smaller spatial distributions include Fucus distichus spp. edentatus, *Gracilaria verrucosa, Bryopsis hynoides, Grateloupia doryphora, Gigartina exasperata, Cryptopleura violacea,* and *Gelidium couheri* (Josselyn and West, 1985). The kelp (*Laminaria*) has been identified in Raccoon Straight between Tiburon and Angel Island (Josselyn and West, 1985; BCDC, 2010).

5.3.5 Trends and Factors Effecting Temporal and Spatial Variation of Macroalgae

There is very little previous literature regarding the historical extent of macroalgae in the SF Bay. Josselyn and West (1985) describe the spatial extent of macroalgae, but macroalgal distribution is not described quantitatively. According to Josselyn and West (1985), the SF Bay has experienced some long term changes in macroalgal species. Several species have been accidentally introduced since the 1970s including *Codium fragile*, *Ascophyllum nodosum ecad scorpioides, Sargassum muticum, Polysiphonia denudata,* and *Callithamnion byssoides*.

As with most estuarine organisms, there are number of complex and interrelated factors that influence the spatial and temporal variation of macroalgae. Spatial and temporal variations in estuarine nutrients and relationships to macroalgae have been studied extensively at various locations, mostly because of their relative importance to primary production (Valiela et al., 1992; Peckol and Rivers, 1995; Pihl et al., 1999; Boyle et al., 2004; Krause-Jensen et al., 2007). The supply of nutrients to an estuary is a primary control on macroalgal abundance (Josselyn and West, 1985; Mackas and Harrison, 1997; Boyle et al., 2004). According to Cloern (1985), continual periods of low discharge allow some marine-estuarine macroalgal species to migrate upstream, and that maximum biomass and diversity of macroalgae is reached in the summer. Variation in salinity, temperature, and available light (Josselyn and West, 1985), as well as the abundance of grazers and differences in water residence time (Valiela et al., 1997), are factors that lead to spatial gradients between different areas of the SF Bay and temporal variation on a scale from days to years.

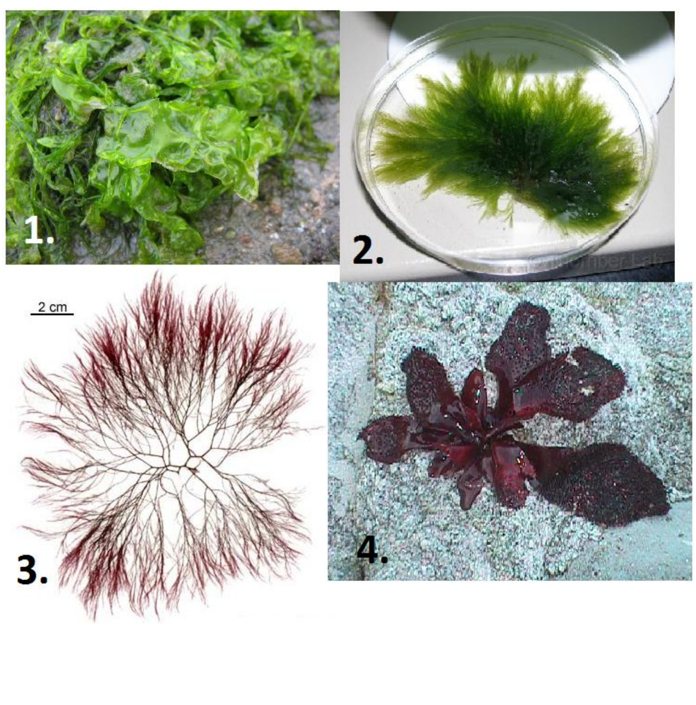


Figure 5.3.2. Examples of macroalgae found in the San Francisco Bay: 1. *Ulva lactuca* (commonly known as sea lettuce), 2. *Cladophora sericea*, 3. *Polysiphonia denudata,* and 4. *Gigartina exasperata.*

Spatial Variability

Macroalgae are most commonly found growing on hard substrate such as rock outcroppings, coarse sediment, and man-made structures but they are also found in lesser diversity on mud and salt flats. Hard substrate suitable for macroalgal growth in the intertidal zone is more common on the shores closest to the ocean and less common in San Pablo Bay, Suisun Bay, and the southern branch of SF Bay (Josselyn and West, 1985). Calm, protected areas often provide ideal locations for certain macroalgal species to grow and accumulate (Josselyn and West, 1985) however at present, macroalgae beds are less dominant by area in SF Bay than submerged aquatic vegetation (BCDC, 2010) Drifting macroalgae also can accumulate in thick mats, detached from substrate, and the current and tides can carry them away from the intertidal zone to deeper areas of the estuary (Nichols and Patamat, 1988). However, at present, there is no recent data on the distribution of macroalgae in the SF Bay.

Temporal Variability

Seasonal variability of macroalgae in SF Bay appears to be strongly influenced by temperature, salinity and light availability (Josselyn and West, 1985). Water temperature varies over the year with the highest temperatures in the summer and the lowest in the winter. There is little difference in water temperature between different areas of the Bay. Salinity drops significantly in the winter to below 10ppt and there is a reduction in macroalgal species number (Cloern and Nichols, 1985; Josselyn and West, 1985). Light attenuation, measured as the light-extinction coefficient (m^{-1}), fluctuates seasonally with the highest levels in the late-winter and spring and the lowest levels in the summer and early-fall. There are significant differences in levels of light attenuation between seaward sites (lower levels) and landward sites (higher levels) in the SF Bay. Figure 5.3.3 compares these previously listed physical factors over an annual cycle. The greatest abundance of macroalgae, measured as percent cover, occurs during May-September. Green algae contribute the most percent cover, red algae are present all year, and brown algae are present all year, but are only abundant during the summer.

Relationships between more frequent daytime exposure of mudflats and an increase in macroalgae, particularly *Ulva clathrata*, have been observed in the SF Bay (Shellem and Josselyn and West, 1982). This greater daytime exposure occurs from late spring to early summer due to the increased frequency of daytime low tides (Josselyn and West, 1985). An increase in macroalgal abundance during the summer has been found to coincide with peak periods of benthic efflux of ammonium and phosphate (McLaughlin et al., 2011). Previous studies have suggested that macroalgae can drive an increased efflux of dissolved inorganic nutrients from sediments by drawing down surface water concentration, thereby increasing the concentration gradient (Tyler et al., 2003). As these nutrients are trapped as biomass, macroalgae become an effective mechanism to retain and recycle nutrients within an estuary, diverting losses such as denitrification or tidal outflow.

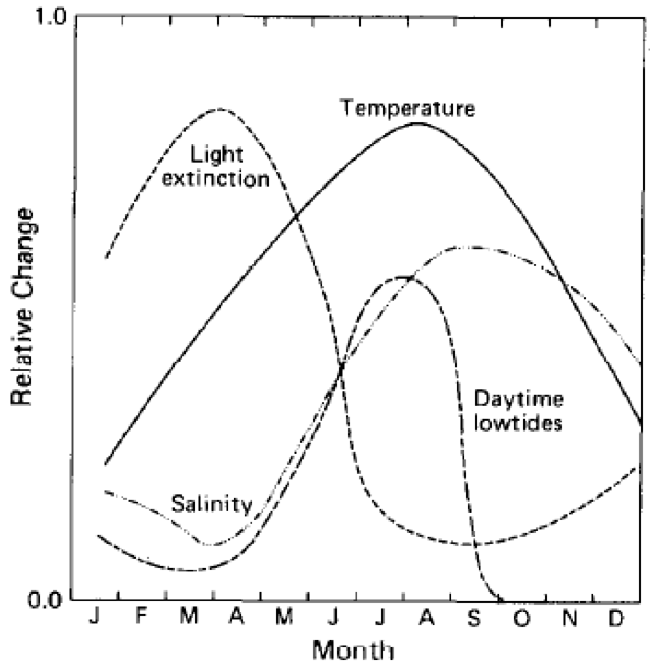


Figure 5.3.3. Relative relationships between physical factors that affect macroalgal growth in San Pablo Bay over the course of a year (adopted from Josselyn and West, 1985).

5.3.6 Utility of Macroalgae as an Eutrophication Indicator for San Francisco Bay

Clear Linkage to Beneficial Uses

Excessive macroalgal blooms have a variety of negative effects on estuarine beneficial uses including: 1) increasing frequency of water column and sediment hypoxia and heightening heterotrophic bacterial activity, resulting in poor water quality and increased frequency of diseases, 2) alteration of biogeochemical cycling, more rapid nutrient regeneration (Tubbs and Tubbs, 1980; Raffaelli et al., 1991; Wennhage and Pihl, 1994; Bolam et al., 2000), 3) shading or smothering of seagrass, shellfish beds and other important habitats (Nelson, 2009; Young 2009), 4) decreased recruitment and survival of benthic invertebrates and reduced carrying capacities for fishes and shorebirds (e.g.Raffaelli, 1999; Thomsen and McGlathery, 2006; Nezlin et al., 2009), 5) poor aesthetics and an increase in odors relating to the decomposition of organic matter and increased sulfide production, and 6) subsequent changes in both trophic and community structure of invertebrates, birds and fishes (Raffaelli et al., 1989, 1991; Bolam et al., 2000). Cumulatively, these adverse effects result in a reduction in recreational use of estuarine waters (REC1 and REC2), poor water column and benthic habitat quality for estuarine (EST) and marine (MAR) aquatic species, direct impacts to populations of threatened and endangered (RARE), migratory (MIGR) and spawning (SPAWN) birds, fish and mammals, and reduction in the economic value of commercial and sports fisheries, aquaculture, and shellfish harvesting (COMM, AQUA).

Fish and invertebrate kills as a result of lowered dissolved oxygen may occur when estuaries are stressed by mats of macroalgae, especially in conjunction with high temperatures and high cloud cover (D'Avanzo and Kremer, 1994). In addition to oxygen stress by respiring macroalgae, fish may be physically affected by drift mats. For example, cod foraging efficiency was drastically reduced with *Ulva intestinalis* cover of 70-80% (Pihl et al., 1995). Macroalgae may also affect piscine recruitment. Wennhage & Pihl (1994) found that juvenile flat fish preferred to recruit to bare sand compared to plots with dense macroalgal cover. They speculated that macroalgae invokes stress in juvenile fish through declines in dissolved oxygen and also restricts prey availability. However, drifting macroalgae is not the only form of algae that may result in the loss of fish and invertebrates. Increases in epiphytic algae on seagrass can result in dramatic reductions in the abundance and biomass of epibenthic invertebrates and fish (Isaksson and Pihl, 1992). Thus ephemeral macroalgae may cause deleterious declines in both recreational and commercial fish stocks (Raffaelli, 1999).

Field studies show that inverse correlations exist between the density of macroalgae and numbers of shorebirds. Cabral et al. (1999) made the claim, "In a long-term perspective, an increase of dense and contiguous macroalgal mats, covering large areas of the intertidal flats, may affect directly or indirectly all wader species in the Mondego estuary." Green (2010) showed that the presence of macroalgae changed foraging behavior from pecking to probing in Sandpipers and Marbled Godwits, suggesting that macroalgae hindered foraging by obscuring visual cues or physically interfering with foraging effort. Further, if macroalgal blooms reduce prey intake rates by shorebirds, then shorebirds with less flexible diets may be more negatively affected than generalist foragers that feed on a wide range of prey species. Green (2010) found avoidance of mats based on foraging ecology. For example, shorebirds that are generalist foragers, such as Least terns and Western Sandpipers and Willets, foraged on macroalgal mats and bare sediment equally. In contrast, shorebirds with more specific dietary requirements such as Marbled Godwits often avoided mats while foraging. In another study, Cabral et al. (1999) observed that Dunlin, a bird species with more restricted diets, tended to avoid dense mats. These studies suggest that as macroalgal blooms become more prevalent in estuaries, specialist species of birds may suffer losses in population numbers.

Predictive Relationships to Causal Factors

There is overwhelming evidence that blooms of macroalgae are stimulated by high nutrient loading, particularly of nitrogen (N) and phosphorus (P) (e.g., Raffaelli et al., 1989; Valiela et al., 1992; Peckol and River 1995; Pihl et al 1999; Krause-Jensen et al., 2007). Recent studies have shown that estimates of nutrient loading that include all possible sources as well as physical removal (flushing) and biological processes are accurate and generalizable predictors of macroalgal biomass. In one of the best examples of this approach, Fox et al. (2008) compared three sub-estuaries of Waquoit Bay, Massachusetts, with different nitrogen loads and found the magnitude of macroalgal standing stock was predicted by total nitrogen load over a six-year period. Notably, this level of detail of the relationship between nutrient loading and producer biomass has been quantified in only a few systems (Valiela et al., 1992, 1997; Hauxwell et al., 1998; Conley et al., 2000; Fox et al., 2008), as it is an expensive and time-consuming process. Measures of nutrient removal to the ocean via tidal flushing are also a key factor that may help to ensure accurate predictions. For example, in Mugu Lagoon (Calleguas Creek arm), southern California, water column nutrient concentrations were always high but algal biomass always low due to low hydraulic residence time and scouring of sediments (Kennison, 2008). Finally, biological processing such as nutrient uptake and algal growth, internal nutrient cycling, and grazing (see Schramm, 1999 for review) must be taken into account to improve the predictive capability of any model. For example, longer residence times that allow more biological uptake and resultant macroalgal growth may result in lower water column nutrient concentrations and more proliferation of macroalgae as found in Mugu (West arm), Tijuana Estuary, and Upper Newport Bay in southern California (Kennison, 2008). Additional work is needed to understand conditions in which phase shifts from microphytobenthos to macroalgae occur, including quantifying rates of uptake and release of nutrients from macroalgae and seasonal storage and release of inorganic nutrients in sediments, in order to parameterize dynamic simulation models of estuarine water quality and biological response to nutrient loads.

Sound and Practical Measurement

A suite of methods to assess the extent and impact of macroalgae in estuaries is well vetted and considered to be sound and practical. These methods are centered on measures of macroalgal taxonomic composition and abundance (biomass and cover). Of these, taxonomic composition is not a particularly useful indicator of eutrophication because the taxonomic composition of macroalgae in California estuaries is limited and the presence of these species alone does not indicate eutrophication.

Overall, macroalgal abundance, as measured by biomass and percent cover is a widely used and accepted method of measurement. Measures of biomass typically require field sampling, collection of samples on mudflats or shallow subtidal habitat in randomly placed transects, and processing of biomass in the laboratory and thus are labor-intensive. Percent cover is usually collected along the same transect as biomass and provide an ability to better capture spatial heterogeneity of macroalgal mats. Measures of both biomass and percent cover are essential to characterize macroalgal response to nutrient loadings. For example, 100% cover of a visible yet thin and still attached mat of algae that may weigh only grams per square meter while 100% cover of dense macroalgal bloom may weigh 12 kg per square meter. Thus the combination of both percent cover and biomass are used to integrate the magnitude and spatial heterogeneity of a macroalgal bloom.

Acceptable Signal to Noise Ratio

Macroalgal blooms are known to be spatial patchy and temporally variable. Mechanisms that control net production of macroalgae are the same as for other primary producers: geographic limits for growth are set by temperature and light and for removal by grazing and physical disturbance. Within these geographical limits, biomass accumulation is controlled by many interacting biotic and abiotic factors including light quantity and quality, water motion, intra- and inter-specific competition, herbivory, and physical disturbance. However, in most places where macroalgae proliferate and dominate estuarine communities in temperate zones, this dominance is a function of nutrient, usually nitrogen (N), availability (for reviews see Howarth and Marino, 2006; Valiela et al., 1997, Vitousek et al., 1997; Downing et al., 1999). For this reason, macroalgae have been successfully used to detect a trend towards decreasing or increasing eutrophication (Scanlan et al., 2007).

Approaches to Setting Numeric Endpoints Based on Macroalgae

Paradigms for establishment of estuarine numeric endpoints based on macroalgal biomass and cover must separate effects for three types of habitats: 1) seagrass 2) polyhaline and euhaline intertidal and shallow subtidal unvegetated habitats, and 3) oligohaline and mesohaline intertidal and subtidal habitat (vegetated and unvegetated; see Figure 5.3.4). A wealth of literature exists documenting the adverse effects of macroalgae on benthic infauna in intertidal and shallow subtidal habitat (Sutula, 2011). Two studies have been conducted that provide data useful for "range-finding" of endpoints. However, more definitive studies need to be conducted to develop better characterization of the relationship between macroalgal biomass, duration of bloom, and effects benthic infauna in intertidal flats across the diversity of intertidal habitats encountered in California (See Sutula, 2011). Some preliminary studies are now underway under the Estuarine NNE project.

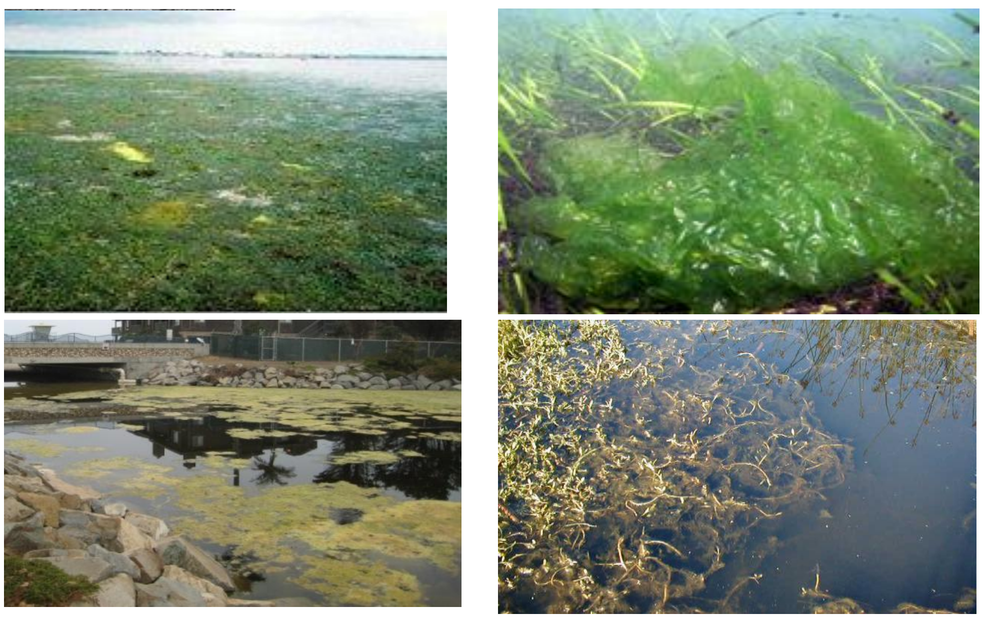
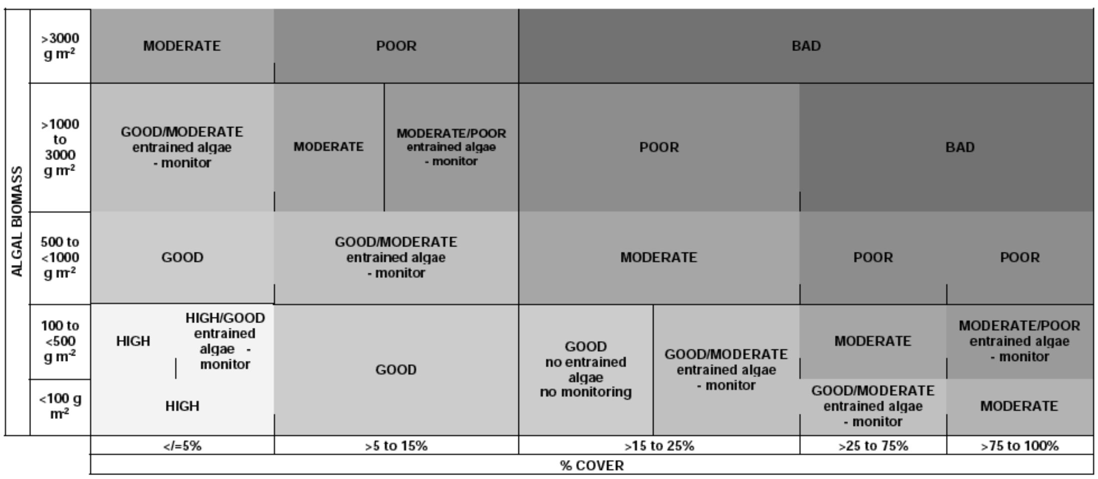


Figure 5.3.4. Examples of types of habitats in which macroalgae can occupy a dominant role among other primary producers: mats on intertidal shallow subtidal flats in polyhaline to euhaline environments (upper left), rafting mats on seagrass (upper right), and floating/rafting mats in oligohaline unvegetated (lower left) and intercalated with *Ruppia* in oligohaline environments (lower right).

Studies documenting effects of macroalgae on seagrass bed density have likewise been published (see Sutula et al., 2011 for full review), but few of these studies are useful specifically for establishing endpoints. As with macroalgae on intertidal flats, studies are likewise needed to document the effects of rafting mat biomass and duration on seagrass bed density in California estuaries. Recently, there is anecdotal evidence of macroalgal blooms found in seagrass habitat in SF Bay (Katharyn. Boyer , SFSU, personal communication, March 2011), though a comprehensive survey is needed to better document the cover and biomass found in SF Bay seagrass beds. Little documentation is available on the effects of rafting algae on microphytobenthos and brackish SAV oligohaline environments. Conceptual models from applicable habitats could be applied, but few field studies have been conducted to illustrate the effects of rafting mats on dissolved oxygen, other pelagic or benthic food webs. These conceptual models may be applicable to the Northern reaches of SF Bay.

Some precedent exists for the use of macroalgae to assess eutrophication. With the adoption of the Water Framework Directive (WDF, 2000), the European Union has been working to assess the ecological condition of its waterbodies. Scanlan et al. (2007) has proposed an assessment framework to diagnose eutrophication. This framework is moving towards adoption within the WFD (Zalidvar et al., 2008). The Scanlan et al. (2007) assessment framework utilizes both macroalgal cover and biomass in a multiple lines of evidence approach (Figure 5.3.5). Both biomass and cover are required to make a diagnosis, because the measurement of just one indicator in isolation could be misleading. For example, an estuary may have low biomass (a positive indicator for estuarine health) but high macroalgal cover (a negative indicator for estuarine health) resulting in a moderate impact to the ecosystem. On the other hand, high macroalgal biomass may be recorded locally, but be mediated by low percent cover over the whole estuary.



Quality Status	Algal Biomass	Algal Cover
High	<100 g m-2	<5%
Good	100-500 g m-2	5-15%
Moderate	500-1000 g m-2	15-25%
Poor	1000-3000 g m-2	25-75%
Bad	>3000 g m-2	>75%

Figure 5.3.5. Proposed assessment framework to diagnose eutrophication using macroalgae for macroalgae in intertidal and shallow subtidal habitat for the European Water Directive Framework (from Scanlan et al., 2007). Biomass is in wet weight.

The framework uses biomass and percent cover to classify an area within an estuary into one of five categories: High, Good, Moderate, Poor, and Bad. Each of these categories was defined as a deviation from a reference or pristine condition. They used a combination of data and expert opinion to generate their categories and assign threshold values between categories, emphasizing that more work was needed, especially to differentiate between moderate, poor, and bad conditions. Scanlan et al. (2007) emphasized that the proposed threshold values must be validated by examining multiple ecological indicators across the eutrophication gradient.

The Scanlan et al. (2007) assessment framework provides a good conceptual model for how to incorporate both biomass and cover into a diagnostic tool and as such is a good starting point for California, in general, and SF Bay, in particular. However, several caveats should be considered. First, the assessment framework does not explicitly incorporate duration of mat presence into the framework, a factor that has been determined to be important through in situ experiments and published literature (Hull, 1987; Balducci et al., 2001; Osterling and Pihl, 2001; Bolam and Fernandes, 2002) and is likely important for SF Bay. Second, Scanlan et al. (2007) did not clearly specify the geographic scope of these specific thresholds for macroalgal biomass and percent cover. Countries within the European Union span the range from Artic to Mediterranean climates and it is unreasonable to think that, given differences in water temperatures across large area, that some differences in the thresholds for biomass and cover are not warranted. Third, while reasonable, the thresholds are based on best professional judgment with little citation of the actual data used to derive the thresholds. Additional work would need to be conducted to develop an appropriate macroalgal assessment framework for SF Bay.

5.3.7 Summary: Use of Macroalgae as an NNE Indicator

Overall, macroalgae appear to satisfy the four evaluation criteria to be considered as an NNE indicator for SF Bay. However, limited data exist on the distribution and variability in macroalgae in SF Bay in seagrass and intertidal flat habitat as well in the tidally muted portions of the Bay. There are a number of data gaps that would need to be filled in order to develop macroalgae as an NNE indicator:

1. Conduct a comprehensive assessment of macroalgal biomass and cover in SF Bay habitats.

2. Develop an assessment framework for macroalgae, including data to support the development of numeric endpoints. This assessment framework ultimately needs to address effects of macroalgae on seagrass habitat, intertidal flats, and oligohaline subtidal environments. Some of this work has already begun for “other” California estuaries (see Sutula, 2011). The SF Bay TAT recommends evaluating the findings of these planned studies in tandem with better information on macroalgal biomass and cover and revisiting how macroalgae could be incorporated into the SF Bay NNE framework at that time.

3. Scope the development of a macroalgal component within SF Bay water quality models (see Section 5.2.6 for additional details on loading and SF Bay water quality models). If macroalgae is to be included, then scoping should include conceptual model development, understanding of how models under consideration can simulate macroalgae, and a data needs assessment to conduct this work. As with the development of an assessment framework for macroalgae, the SF Bay recommends that this work be considered pending the findings of planned studies for macroalgae being conducted for other California estuaries.

5.4 Submerged Aquatic Vegetation (SAV)

Rooted submerged aquatic vegetation (SAV) encompass a large diversity of species that range from obligate halophytes (e.g., seagrasses, *Zostera marina L., Z. japonica*) to mesohaline and oligohaline species (e.g., *Ruppia maritima L., Vallisneria spp., Stukenia pectinatus*) to freshwater obligates (e.g., *Elodea canadensis, Nuphar spp*.). The primary features distinguishing between groups of SAV are salinity tolerance and pollination vectors. Throughout the course of this review, the term “seagrass” will be applied exclusively to genera that are obligate halophytes, exhibit hydrophylious (underwater) pollination and form meadows; this includes but is not limited to, *Zostera, Phyllospadix, Halodule, Thalassia, Halophil*a, etc. “Brackish SAV” or “aquatic beds” will be applied to genera that are euryhaline species, exhibit aerial or surface pollination and tend to form canopies; this includes but is not limited to *Ruppia, Stukenia, Zannichellia, Myriophyllum,* etc. Seagrass and SAV can form extensive beds (Figure 5.4.1), and can also be found as solitary patches much smaller in size (Merkel & Assoc., 2004a).

Seagrass and SAV have a variety of characteristics that make them good candidates to be “end-points of concern” for eutrophication or “bio-indicators”. First, many of these species, especially the seagrasses, are perennial and form persistent rhizomes; consequently they act as “long term integrators” responding to environmental change (Burkholder et al., 2007). Second, as rooted organisms, they are not mobile and cannot move in response to changing environmental drivers. Third, for a number of key seagrass and SAV species (including *Zostera marina L*. and *R. maritima*) the biological and physiological requirements are known well enough to develop models of how the plants respond to stressors. Finally a number of very well designed monitoring programs currently use seagrasses as bio-indicators (Fonseca et al., 2001; Foden and Brazier, 2007; Madden et al., 2009) including government organizations such as Washington State Dept. of Natural Resources[4] and non-governmental organizations (http://seagrassnet.org). Although many estuarine systems do support SAV it is important to recognize that not all systems would be expected to support these plant communities based on the morphology and hydrology of the system.

Kaldy and Sutula provide a detailed review of the utility of seagrass and brackish SAV as an NNE indicator in estuaries (see Chapter 6 in Sutula, 2011). They conclude that there are three types of primary indicators relevant to the assessment of eutrophication in seagrass habitats: 1) macroalgal biomass and cover, 2) phytoplankton biomass and light attenuation, and 3) epiphyte load. Seagrass areal distribution and density can be considered supporting indicators, but are known to respond to a wider number of stressors than just eutrophication per se (e.g., excessive sedimentation, temperature stress, etc.). This section provides a brief synopsis of that work and presents literature directly relevant to known abundance, distribution and variability of seagrass and brackish SAV in San Francisco Bay estuary.

[4] www.dnr.wa.gov/ResearchScience/Topics/AquaticHabitats/Pages/aqr_nrsh_eelgrass_monitoring.aspx

Figure 5.4.1. Southern End of Eelgrass Bed, between Point San Pablo and Point Pinole (from Merkel & Assoc., 2004a).

5.4.1 Data Available on Seagrass and Brackish SAV

In general there is still little is known about abundance and distribution of SAV in SF Bay. One study exists on four types of SAV communities *Phyllospadix scouleri* (surfgrass), *Zostera marina* (eelgrass), *Ruppia maritima* (widgeongrass), and *Potamogeton pectinatus* (sago pondweed) (Schaeffer et al., 2007). The data set on eelgrass is more comprehensive made possible by the Bay Bridge construction and the associated mitigation related surveys (Merkel & Assoc., 2003, 2010). Surveys of areal distribution are available for the years 1987, 2003, and 2009 (Merkel & Assoc., 2004, 2010) and will continue for at least another 5 years. Aspects of physiology, recruitment, and growth have been studied for eelgrass (Zimmerman et al., 1995; Boyer et al., 2008).

5.4.2 SAV Relationship to Nutrients and Water Quality

Submerged aquatic vegetation (SAV) is an important component of estuarine and coastal nearshore ecosystems and form structural habitats for a diversity of plant and animal species, affect rates of sedimentation and erosion, and influence the structure of inshore benthic communities (see reviews in Zimmerman et al., 1991, 1995). SAV usually grow in unconsolidated anoxic sediments in shallow, calm nearshore areas and are therefore extremely vulnerable to human encroachment from stormwater outfalls, marinas, piers, wharfs, swimming beaches, dredging, trash and other forms of urban pollution. The depth distribution and plant density of SAV is strongly correlated to light availability, temperature, salinity, and tidal forces. Light, however, appears to be the factor that most often controls the depth distribution, density, and productivity of SAV (see review in Zimmerman et al., 1991, 1995). Unfortunately, in SF Bay, high turbidity renders much of the Bay light limited; the euphotic zone (depth where irradiance falls to 1% of surface irradiance) is <1 m in many locations (Alpine and Cloern, 1988; Zimmerman et al., 1991). Observations made by Zimmerman et al. (1991) provided evidence for the hypotheses that 1. *Zostera* populations in the Bay are adapted to low light availability, and 2. *Zostera* in SF Bay may be more controlled by short lived low light pulses of turbid water during high runoff or high wind periods rather than by average annual light conditions. Therefore, it should not be surprising that the majority of SAV beds, dominated by the seagrass *Zostera* spp., found in SF Bay occur in the higher salinity lower turbidity areas in the Central Bay at depths <2 m mean lower low water (MLLW) and at some sites at depths only shallower than 1 m MLLW (Zimmerman et al., 1991).

The response of eelgrass (*Zostera marina*) to increased nutrient loading and eutrophication has been a major research focus over the past few decades; however, the majority of this work has occurred on the east coast (Nixon et al., 2001; Burkholder et al., 2007). It has been shown that increases in nutrient loading causes degradation in *Z. marina* through the stimulation of algal production (micro- and macroalgae) and shading out seagrass (Havens et al., 2001). Algae are stronger competitors than seagrass, and when nutrients are increased, a phase shift from seagrass dominance to either phytoplankton or macroalgae dominance can occur (Burkholder et al., 2007). However, Nixon et al. (2001) did not find a predictable relationship between algal type (phytoplankton vs. microphytobenthos vs. macroalgae) and nutrient levels; a number of biotic and abiotic factors contribute to this complex relationship. More often, the impact of nutrient enrichment on seagrass is usually observed through indirect effects, although there have been direct effects reported. Burkholder et al. (2002) performed mesocosm experiments in North Carolina and found that increased nitrate levels in the water column led to declines of eelgrass, independent of macroalgal shading. They suggested that the impact was due to direct physiological effects associated with internal imbalances in nutrient ratios from sustained nitrate intake through the leaves. Nutrient issues for eelgrass in San Francisco Bay have been reviewed by Boyer and Wyllie-Echeverria (2010). They suggested there is evidence for a lack of N-limitation except in a few locations but more work needs to be done. Recently Carr et al. (2011), presented a new analysis which included some data epiphyte loads (as chl a) on eelgrass in SF Bay. Overall, there is consensus that turbidity is the most limiting factor for eelgrass in San Francisco Bay. However, recent and sustained downward trends in suspended sediment concentrations might mean that factors like phytoplankton might become the leading factors in the future (Katharyn Boyer, personal communication, March 2011). Overall, the effects of nutrients of eelgrass are not well established for the Bay. In a recent study in Tomales Bay, macroalgae was found in abundances that were determined to have an adverse effect on eelgrass density and growth rates (Huntington and Boyer 2009). In SF Bay, Boyer and Wyllie-Echeverria (2010) found biomass of macroalgae generally low in eelgrass beds, but occasionally levels approached biomass found to be detrimental to eelgrass in Tomales Bay; further work is recommended. Presently there are a number of ongoing studies being overseen by Katharyn Boyer. These include chlorophyll a data from the water column, surface sediments, and ammonium and nitrate data from four eelgrass beds and about a year of continuous dissolved oxygen data from two seagrass beds (Katharyn Boyer, personal communication, March 2011).

Given the habitat benefits associated with eelgrass beds, there are efforts to restore areas formally thought to have supported eelgrass. However, efforts, thus far, to restore or increase the area of existing meadows have been met with mixed success. In a study of eelgrass (*Zostera marina L.*) transplants in SF Bay, Zimmerman et al. (1995) found that despite the period of favorable light levels due to low Sacramento River runoff in 1989 and 1990, 40% of the transplants were lost within the depth range of the native populations (-0.5 to -1.0 m depth). Despite these losses, self-sustaining beds were established and observed over a four year period and the authors concluded that transplanting should be viable given sufficient plant C reserves and light availability (Zimmerman et al., 1995). Using new seeding techniques, Buoy-Deployed Seeding, Boyer et al. (2008) were able to recruit and establish *Zostera marina* at two restoration sites in the SF Bay and they believe that seedling recruitment will continue to contribute to eelgrass cover in the next several years. They observed fish and amphipods in the restored beds which signify that restored patches were beginning to serve as habitat for native and eelgrass dependent species (Boyer et al., 2008).

5.4.3 Species Composition

Although there is developing information on SAV in SF Bay, in general there is still little is known about abundance and distribution. There are four types of SAV communities found in the SF Bay (Figure 5.4.5: *Phyllospadix scouleri* (surfgrass), *Zostera marina* (eelgrass), *Ruppia maritima* (widgeongrass), and *Stuckenia pectinata* (sago pondweed)) (Schaeffer et al., 2007). Based on the recent surveys of SAV in the Bay, eelgrass is the most widely distributed and most abundant (Merkel & Assoc., 2003, 2010; Schaeffer et al., 2007). Eelgrass is the most commonly studied form of SAV in the SF Bay. Sago pondweed has long rhizomes and runners which allow it to better tolerate strong currents and wave action. Eelgrass has all of its life cycle stages occur underwater, including seed germination, flowering, and pollination. Widgeongrass can grow in both freshwater and high salinity environments (Schaeffer et al., 2007).

5.4.4 Trends and Factors Effecting Temporal and Spatial Variation of SAV

Zostera marina beds vary greatly during a single season in shoot density and between years in aerial extent. Several factors have been found to influence this variation including temperature and light conditions in relation to variations in water clarity (turbidity) (Merkel & Assoc., 2010). Given turbidity is gradually decreasing in the Bay (Schoellhamer, 2009) associated with decreasing sediment loads entering the Bay from the Central Valley (McKee et al., 2006); it is possible that expansion will continue as an overall trend. However, annual variations in light conditions will continue to cause inter-annual fluctuations.

Spatial Variability

Eelgrass beds are currently found in euhaline to polyhaline environments in the SF Bay which is mainly along the eastern shores of San Pablo and Central Bays. The majority of eelgrass in the Bay and the largest eelgrass bed is found between Point Pinole and Bayfarm Island (Figure 5.4.2) (Merkel & Assoc., 2010). In another example, the eelgrass patch on the southern side of the Richmond shipping channel is also large but has decreased in size by 11.9 ac (4.8 ha) over a two year period. During the same time frame, the patch at Keller Beach increased in size by 10.4 ac (4.2 ha) (Merkel, 1999 cited in Wyllie-Echeverria and Rutten, 1989).

Using data from the 2003 and 2009 surveys performed by Merkel & Assoc., the depth distribution of eelgrass was mapped (Figure 5.4.3). The depth distribution of eelgrass is narrow and it also shows the turbid nature of the SF Bay (Merkel & Assoc., 2010). The likely factor controlling the spatial distribution of eelgrass in the Bay is water clarity and turbidity. In the North and South Bay there are low light levels due to the large input of sediment per year (on average 1 million mt per year: McKee et al., 2006) from freshwater rivers. Additional suspended sediment occurs from tidal currents and wind driven waves (Merkel & Assoc., 2010).

Temporal Variability

Eelgrass distribution varies greatly within and between years in association with light and temperature conditions. For example, at the control site of the Richmond training wall transplanting experiment study, shoot density was almost 10 times greater in April 1985 than that observed in July 1985. Shoot densities for September 1985 and May 1986 were 30% and 48 % of the April densities respectively (Fredette et al. 1987 cited in Wyllie-Echeverria and Rutten, 1989). According to Keith Merkel (personal communication, 2011) monitoring performed in 2010 suggests a slight increase in eelgrass from 1,500 ha to approximately 1,522 ha in the SF Bay. He also anticipates that there will be a substantial drop in eelgrass abundance in the next year due to his anticipation of relatively high sediment loading from the 2010/11 wet weather period. In general, for a few years after a large sediment loading event, sediment resuspension will cause a decline in eelgrass, especially where wind driven resuspension and limited flushing occurs. This appears to be supported by a preliminary relationship between suspended sediment loads entering the Bay from the Central Valley via the Delta (McKee et al., 2006), and eelgrass extent data collected on three occasions beginning 1987 (Figure 5.4.4). There are of course flaws in this very simple model; it suggests a linear relationship, whereas more likely maximum possible eelgrass extent is greater than 4920 acres (possibly as high as 20,000 acres (8,100 ha) (Merkel & Assoc. 2004), and it is unlikely that total extirpation would occur if a future 3-year averaged sediment load exceeds 0.7 million mt, since there has been many times in the past 30 years when that has likely happened and 7 times since 1997 when we know it has happened (McKee et al., 2006).

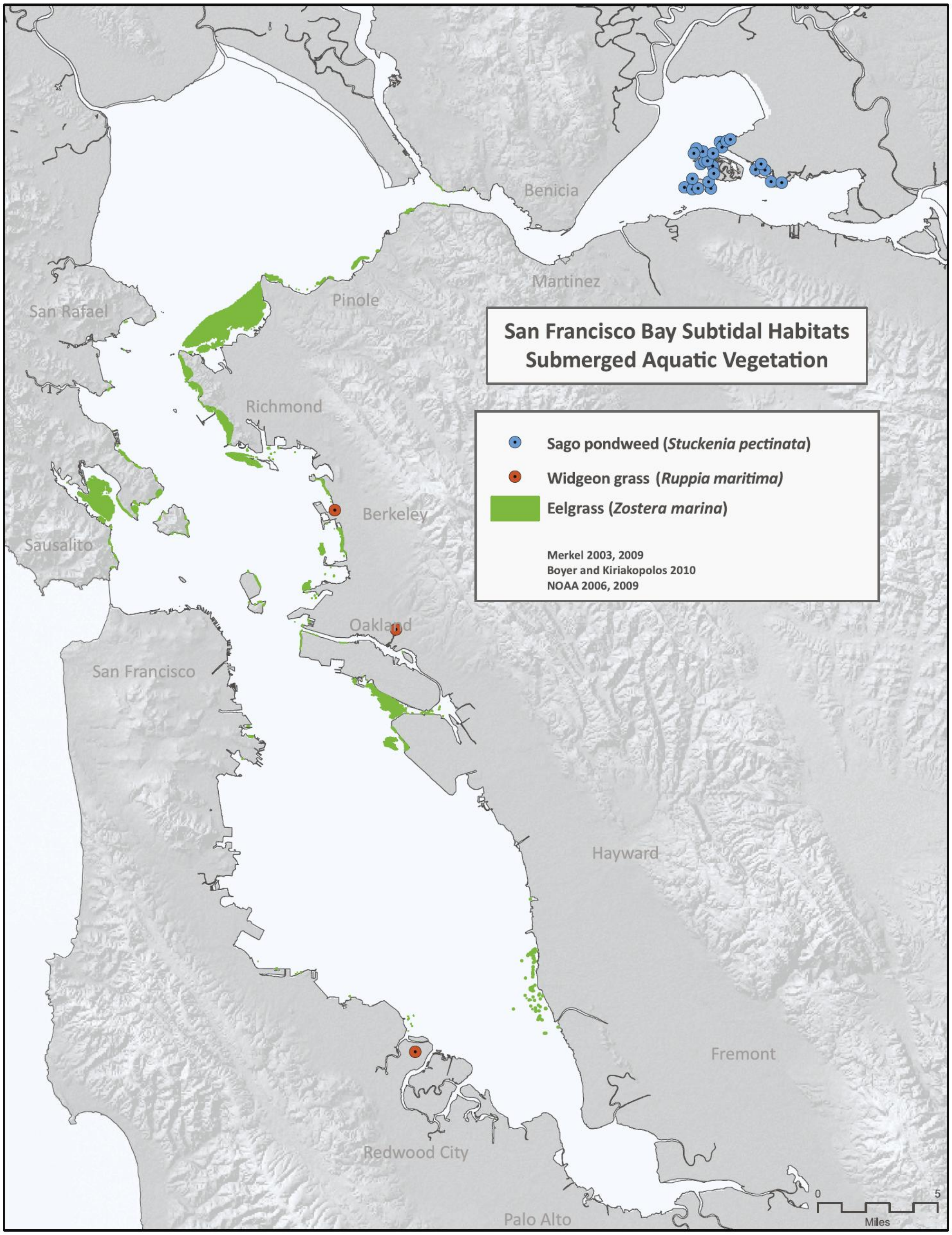


Figure 5.4.2. The distribution of three different SAV communities in San Francisco Bay (SSC, 2010).

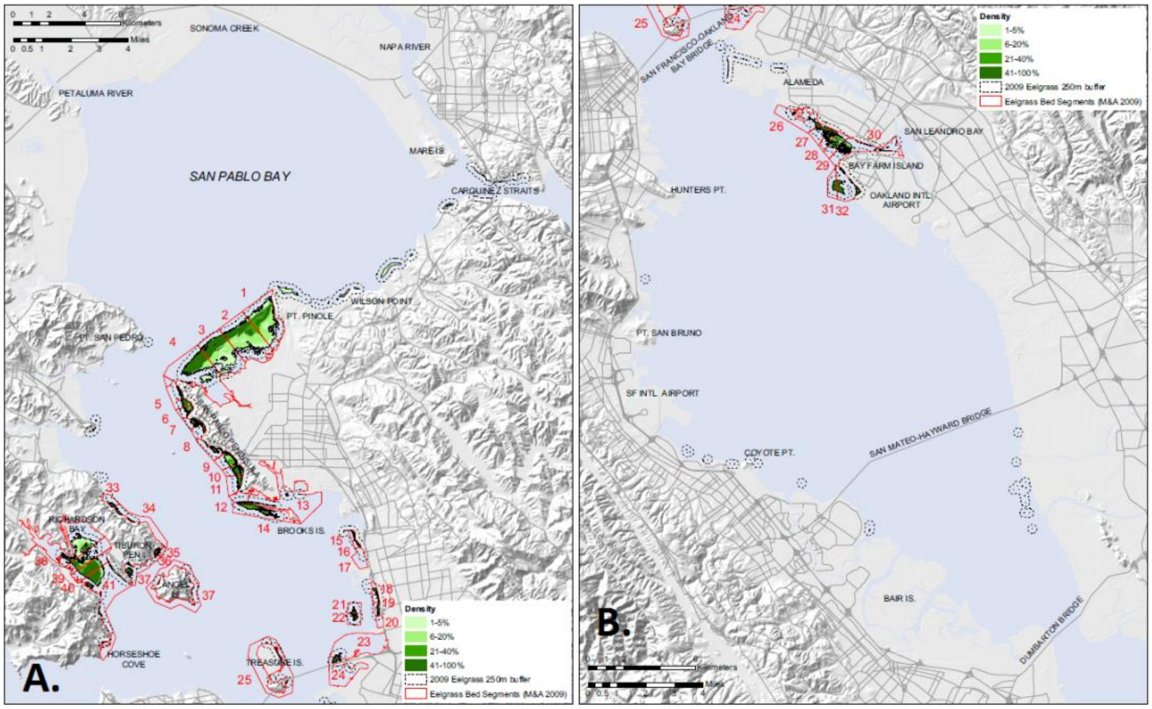


Figure 5.4.3. 2009 distribution of eelgrass meadows (*Zostera marina L.*) in A. North and Central San Francisco Bay and B. South San Francisco Bay (Source Merkel & Assoc. 2010). Note – the numbers on the Figure refer to the numbers and text in Table 5.4.2.

SAV Trends

The first formal recorded study of eelgrass in SF Bay was that of Setchell (1922) who studied *Zostera marina* patches in Keil and Paradise coves on the eastern coastline of Tiburon. Unfortunately, that we can determine, no Bay wide estimates of total bed area were published at the time. However, given the massive fluxes of sediment that were coming into the Bay in the early 1900s in response to landscape disturbances during the gold rush and as the Central Valley was opening up for agriculture (McKee et al., 2006; Jaffe et al., 2007; Ganju et al., 2008), that any eelgrass was observed at all is perhaps miraculous. A long hiatus for some 50 years occurred before research interests picked up in relation to managing and monitoring diminishing beds, spawning habitat value for Pacific herring, and ongoing deepening and widening projects in relation to shipping (Wyllie-Echeverria and Fonseca, 2003). In 1989, a study revealed that eelgrass populations were discontinuous and found on the southern shorelines of San Pablo Bay and northern reaches of central Bay and in the northern reach of South SF Bay in 23 locations ranging in size from 0.5 ac (0.2 ha) to 124 ac (50 ha) with a total estimate of 316 ac (128 ha) (Wyllie-Echeverria and Rutten, 1989). In 1993, these aerial distributions were combined with estimates of carbon production from literature (300 g $m^{-2}y^{-1}$) to estimate a total baywide carbon production from eelgrass of 384 mt (Jassby et al., 1993) or <0.2% of total autochthonous productivity at that time.

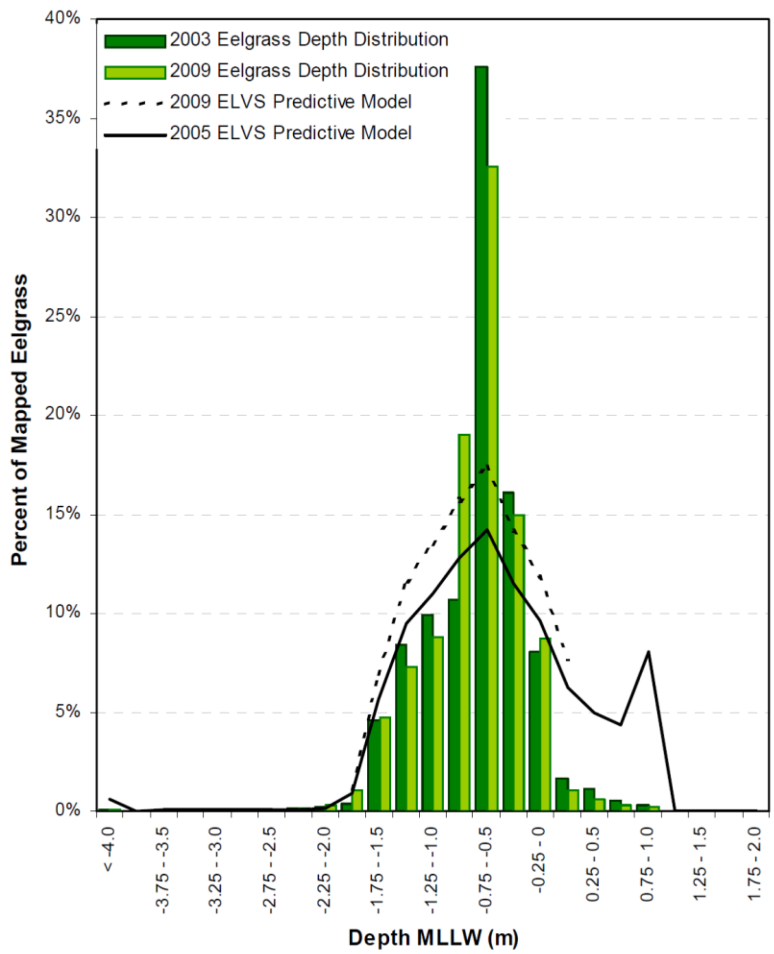


Figure 5.4.4. Eelgrass depth distribution in San Francisco Bay (from Merkel & Assoc., 2010).

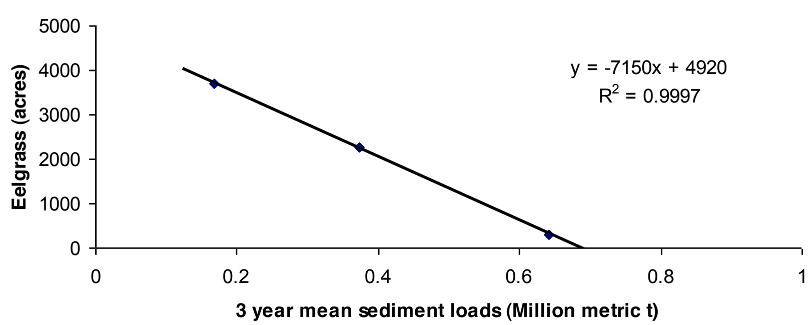


Figure 5.4.5. The quasi-relationship between eelgrass area and three- year mean sediment loads in the San Francisco Bay using estimated and measured sediment loads from the Central Valley to the Bay and the equation from McKee et al., 2006.

More recently eelgrass has been surveyed extensively in relation to the construction of the new eastern span of the Bay Bridge. To reduce time and resources needed to accurately map eelgrass in SF Bay, a prescreening model and aerial survey was used by Merkel & Associates (2003) to reduce the area of detailed survey from ~100,000 ha of sub- and inter-tidal habitat to 20,000 acres (8,100 ha) of possible habitat which was subsequently surveyed using vessel-deployed acoustic techniques. Based on this program, new estimates of eelgrass habitat are now a little less than 1,200 ha (Merkel & Assoc., 2004) (Tables 5.4.1 and 5.4.2). While their methods are not absolute, diver surveys are done to confirm there is eelgrass at a given site (as opposed to macroalgae or *Ruppia*). It is possible that small patches are missed and some deeper patches are over estimated but over all there is good confidence in the data (Katharyn Boyer, Romberg Tiburon Center for Environmental Studies, San Francisco State University personal communication, January 15th 2011).

The survey completed in 2009 by Merkel & Associates provides estimates of the eelgrass habitat area of 1,500 ha, a 28.7% increase in overall eelgrass coverage since the 2003 survey (Merkel & Assoc., 2004) and nearly a 1,200% increase since the 1987 survey (Tables 5.4.1 and 5.4.2). An expansion of eelgrass northward into the Carquinez straights and an increase along both the shoreward (shallower) and bayward (deeper) edges of existing eelgrass beds are major differences between the 2003 and 2009 surveys. These increases were not observed in the southern reaches of the Bay, where eelgrass remained comparably or even less abundant than the 2003 surveys had shown.

5.4.5 Utility of Seagrass and Brackish SAV as an Eutrophication Indicator for San Francisco Bay

Clear Linkage to Beneficial Uses

Seagrasses and other SAV are considered to be community structuring plant that forms expansive meadows or smaller beds. As a result, they are considered to be "habitat forming" species that creates unique biological, physical, and chemical environments when it occurs in the forms of submerged or intertidal aquatic beds or larger meadows. Eelgrass beds are important ecological communities of shallow bays and estuaries because of the multiple ecological services they sustain (Orth et al., 2006). Seagrass and SAV are designated marine and/or estuarine habitat that have an obligate requirement for seawater (MAR and EST beneficial uses). They are also wildlife habitat particularly waterfowl and shorebirds (WILD beneficial use). Seagrass and SAV beds function as habitat and nursery areas for commercially and recreationally important open ocean marine and estuarine fish and invertebrates, and provide critical structural environments for resident bay and estuarine species. Many commercially and recreationally (COMM beneficial use) important fisheries species have a life-history stage that is estuarine dependent and many of them utilize seagrass beds; examples include, salmonids, herring, Dungeness crab, shellfish (Blackmon et al., 2006). Seagrass also support shellfisheries (SHELL beneficial use), as a variety of bivalves used for human consumption and bait occur in seagrass beds. Presence of seagrass can influence the population structure and growth rates of clams (Peterson et al., 1984); additionally seagrass patch size and structural characteristics affect bivalve survivorship (Irlandi, 1997). Peterson and Heck (2001) suggest that bivalves and seagrass have positive interactions resulting in a facultative mutualism.

Table 5.4.1. Trends in the aerial extent of eelgrass meadows (*Zostera marina L.*) in San Francisco Bay between 1987 and 2003 (Merkel & Assoc., 2004).

	1987	
	(ha.)	(ac.)
San Pablo Bay	50.2	124
Point Orient	1.2	3
Naval Supply Depot	4.9	12
Point Molate Beach	10.5	26
Toll Plaza West	0.2	0.5
Toll Plaza East	0.2	0.5
Point Richmond, North	2.8	7
Point Richmond, South	1.6	4
Richmond Breakwater, North	7.3	18
Richmond Breakwater, South	2.8	7
Brickyard Cove	-	-
Emeryville (breakwater)	5.3	13
Emeryville Flats	-	-
Yerba Buena Island	-	-
Treasure Island	-	-
Alameda	22.3	55
Bayfarm, North	0.8	2
Bayfarm, South	1.6	4
Coyote Point	0.4	1
Richardson Bay	5.3	13
Angel Island West	1.2	3
Angel Island South	-	-
Angel Island East	-	-
Belvedere Cove	2.0	5
Point Tiburon	0.4	1
Keil Cove	4.0	10
Paradise Cove, North	1.6	4
Paradise Cove, South	1.2	3
Pt. San Quentin	-	-
Pt. San Pedro	-	-
Minor Beds and Patches	-	-
TOTAL	127.9	316

Table 5.4.2. Trends in the extent of eelgrass (*Zostera marina*) in the San Francisco Bay during the 2003 (Merkel & Assoc., 2004) and the 2009 surveys (Merkel & Assoc., 2010).

Eelgrass Segment Number	Bay Region	2003 (ha.)	2009 (ha.)
1	Pt. San Pablo/Pt. Pinole (East)	139.83	208.06
2	Pt. San Pablo/Pt. Pinole (Central-East)	169.83	248.68
3	Pt. San Pablo/Pt. Pinole (Central-West)	128.26	225.91
4	Pt. San Pablo/Pt. Pinole (West)	124.28	133.53
5	Navy Supply Depot Pt. San Pablo	14.69	21.26
6	Navy Supply Depot Pt. San Pablo	13.81	13.00
7	Point Molate Beach	7.25	9.43
8	Point Molate Beach	5.68	11.90
9	Kellers Beach North	11.74	28.17
10	Kellers Beach South	15.71	21.83
11	Kellers Beach South	9.82	13.07
12	Inside Richmond Tr. Jetty	3.85	4.17
13	Inside Richmond Tr. Jetty	4.14	4.20
14	Outside Richmond Tr. Jetty	27.59	35.13
15	Albany Beach	0.28	1.48
16	Golden Gate Fields	0.15	0.49
17	Golden Gate Fields	0.09	0.01
18	Brickyard Cove	2.42	4.05
19	Berkeley Shoreline	1.67	1.78
20	Berkeley Shoreline	3.08	4.72
21	Berkeley Shoal	4.73	4.24
22	Berkeley Shoal	6.89	7.51
23	Emeryville Shoal	3.78	3.92
24	Emeryville Shoal	6.80	8.02
25	Clipper Cove/Treasure Island	2.51	2.19
26	Crown Beach	16.36	5.14
27	Crown Beach	13.04	11.36
28	Crown Beach	10.78	18.31
29	Crown Beach	31.60	35.63
30	Crown Beach	29.95	18.22
31	Bayfarm Island	13.05	11.37
32	Bayfarm Island	28.33	24.44
33	Paradise Cove, North	4.12	2.99
34	Paradise Cove, South	2.25	4.53
35	Keil Cove	2.14	1.94
36	Keil Cove	3.06	3.16
37	Keil Cove	14.13	14.07
38	Richardson Bay	21.58	88.70
39	Richardson Bay	45.50	48.25
40	Richardson Bay	66.54	72.36
41	Richardson Bay Entrance	48.19	63.74
NA	Pt Pinole/Carquinez Bridge	<0.01	54.96
NA	Pt San Quentin/Marin Rod & Gun Club	0.88	0.36
NA	Oakland Middle Harbor	0.53	0.01
NA	Oakland Inner Harbor	0.03	0.15
NA	Alameda Western Shoreline	<0.01	1.45
NA	San Mateo to South San Francisco	1.16	0.01
NA	San Mateo Bridge to Dumbarton Bridge	0.13	0.02
NA	Other Eelgrass Beds	1.33	2.10
BAYWIDE EELGRASS AREA TOTAL		**1,061.34**	**1,500.00**

Besides providing important habitat for fish, seagrass and SAV are considered to be an important resource supporting migratory birds and spawning fish during critical life stages. Bortolus et al. (1998) found that *Ruppia maritima* was an important food sources for a variety of waterfowl species in Argentina; including swans and ducks. Along the Pacific flyway, both *Ruppia maritima* and *Z. marina* are food resources for Black Brant geese (Ward, 1983; Derksen and Ward, 1993; Moore et al., 2004). Seagrass and SAV meet the spawning beneficial uses as they provide a refuge for anadromous fish (salmonids) particularly during the transition from freshwater to seawater (see reviews in Kennedy, 1982 and Blackmon et al., 2006). Seagrass and SAV habitat provide a direct food source for migrating waterfowl (Moore et al., 2004) as well as an acclimation refuge for anadromous fish species (Blackmon et al., 2006), thus linking to MIGR, SPWN, and RARE beneficial uses. Healthy Seagrass and SAV support REC-2 beneficial uses in a number of ways. These habitats are prime areas for recreational crabbing and fishing as well as kayaking and waterfowl hunting and a focus of marine studies programs in the SF Bay as well as elsewhere in California.

Predictive Relationships to Causal Factors

Under oligotrophic conditions, increased nutrient loads may initially be beneficial to seagrass communities by stimulating primary production, leading to greater secondary production by consumers. However, under continued high nutrient loads, algae are superior competitors and their increased abundance can be deleterious to seagrass. Initial indications of eutrophication issues include decreased bed density and increased abundance of the algal flora. Under very high nutrient loading, the system can become dominated by algal competitors (phytoplankton, epiphytes or macroalgae) resulting in the degradation or loss of the seagrass community (Figure 5.4.6). The primary mechanism of seagrass loss is through light reduction caused by shading or smothering from algal competitors. Reduced light coupled with increased delivery of labile organic detritus (senescent algae and seagrass) to the sediments can lead to additional biogeochemical stressors (hypoxia/anoxia, sulfide toxicity, etc.) that further exacerbate the problem. Consequently, there tends to be a positive feedback loop between nutrient enrichment and expression of eutrophic or dystrophic conditions. Degraded seagrass beds tend to be sparse or patchy, heavily epiphytized with macroalgae and experience large diurnal swings in dissolved oxygen concentrations.

Response of seagrass to nutrient loading and eutrophication has been a major research focus over the last couple decades (Nixon et al., 2001; Nielsen et al., 2004a; Burkholder et al., 2007; and many others). Most seagrass eutrophication studies have examined the community level response in experimental systems ranging from aquaria to mesocosms to the natural environment (Table 6-1). For *Z. marina* much of this work has been conducted along the East Coast of North America and has resulted in a general theory of seagrass response. Specifically, that enhanced nutrient loading leads to a degradation of *Z. marina* habitat (Figure 5.4.6) by stimulating algal production (micro- and macroalgae) and shading out seagrass (Short et al., 1991, 1995; McGlathery, 2001; Havens et al., 2001).

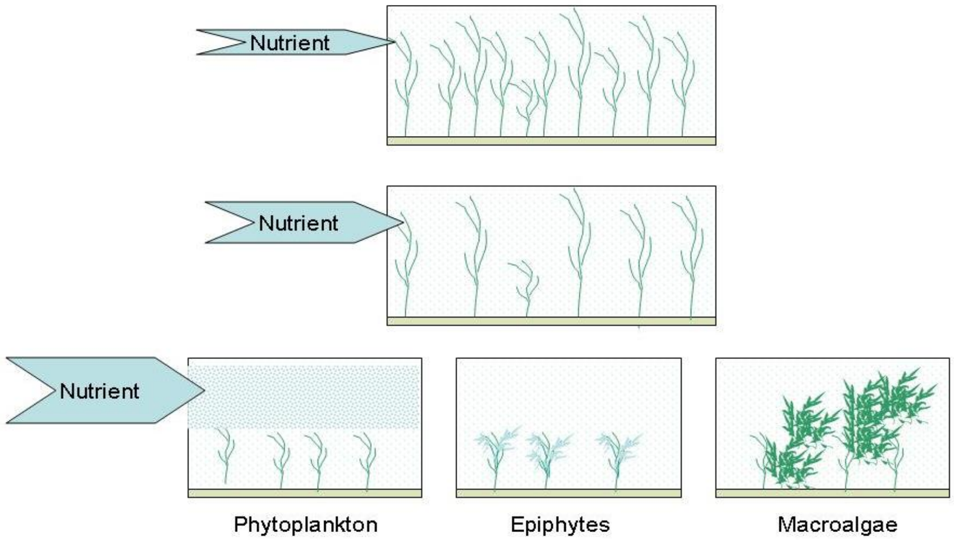


Figure 5.4.6. Conceptual model of how seagrass and some SAV communities respond to increased nutrient loading and resulting eutrophication. From Kaldy and Sutula (Chapter 5, Sutula, 2011).

A number of dose response experiments have been undertaken with *Z. marina*; however, most studies have been monocultures in experimental mesocosm experiments and this work has been primarily conducted with plants from the North Atlantic populations (Burkholder et al., 2007 and references therein). The experimental mesocosm and load response experiments clearly show that the taxonomy of the community shifts from seagrass dominance to either macroalgae or phytoplankton dominance (Burkholder et al., 2007). Field based sampling along eutrophication gradients (e.g., Waquoit Bay, MA) and field manipulations of water column nutrients exhibit similar patterns (Burkholder et al., 2007). Thus the seagrass declines through indirect effects on the seagrass (e.g., shading, increased hypoxia, increased respiration, etc.). However, there does not appear to be a predictable trajectory of development between nutrient input and the algal type (epiphyte vs. macroalgae vs. phytoplankton) that can adversely affect seagrass (Nixon et al., 2001). That is for any given load of nutrients there is no capacity to predict if the system will become dominated by macroalgae, epiphytes (e.g., microphytobenthos) or phytoplankton.

The ultimate primary producer dominance is determined by initial starting condition of the system (spore or propagules availability) interacting with various biotic and abiotic cofactors. Additionally, since seagrass occur in shallow subtidal habitats, characterization of external loads versus internal recycling of nutrients through the sediments is important for achieving a predictable load-response. These data gaps would effects the accuracy and precision of dynamic simulation models that attempt to capture the relationship between seagrass community structure and nutrient loads and other co-factors. In general then, an understanding of the relationship between nutrient loading and algal primary producers within seagrass habitats is a data gap that requires additional research.

Dynamic simulation models (as ecosystem or stress-response) have been used in many instances to develop and evaluate nutrient and other water quality criteria and restoration potential. The Chesapeake Bay approach has been to define the water quality parameters (e.g., light, temperature, salinity, nutrients) associated with SAV declines and to develop a suite of environmental characteristics that are protective of the resource and allow restoration of SAV habitat (Batuik et al., 1992, 2000). As part of this approach, light attenuation by epiphyte load, in addition to light attenuation by the water column (k_d) from water column suspended sediment and phytoplankton, is considered in efforts to evaluate SAV light requirements. These relationships have been based on extensive published and unpublished data sets developed over more than 30 years of research on a single, albeit large estuary. A dynamic simulation model is then used to model the relationship between the seagrass habitat and factors that control light availability, and other stressors that require management (nutrient loads, sediment inputs, etc.). Similarly, the USEPA Western Ecology Division has developed and used a seagrass stress response model to evaluate the impact of proposed nutrient criteria on eelgrass in a Pacific Northwest estuary (Brown et al., 2007). Potential nutrient criteria were developed using the *in situ* observations as a basis for the "Estuarine Reference Condition" using cumulative distribution functions. Proposed criteria were then incorporated into the *Zostera* stress response model to determine whether particular percentile values would adequately protect *Zostera* resources within the estuary (Brown et al., 2007). Based on this data analysis and modeling exercise, median values of most water quality parameters were protective of seagrass habitat.

Some work has been done on the direct effects of nutrients on seagrasses. Tenant (2006) conducted an *in situ* fertilization experiment in Humboldt Bay and concluded that phosphate toxicity explained field observations. The study suffers from a poor design and lack of data to evaluate the sediment nutrient pools as well as the tissue nutrient pools. Claims of phosphate toxicity are premature and not supported by the primary literature. Work from North Carolina and Europe suggests that some populations of *Z. marina* may exhibit declines in response to low level concentrations of NO_3 or NH_4 (Burkholder et al., 1992, 1994; van Katwijk et al., 1997). However, these conclusions may be confounded by other factors (e.g., high temperatures and reduced light). Oregon populations of *Z. marina* are regularly exposed to >30 µM NO_3 from coastal upwelling (Kaldy and Lee, 2007; Brown and Ozretich, 2009) and have shown no declines associated with nitrate toxicity. Furthermore, exposure to 1000 µM NO_3 for two weeks in a laboratory experiment (temp 8°C, 12:12 L:D with saturating irradiance) did not produce mortality or evidence of stress (J. Kaldy, USEPA, unpublished data). Additionally, this seagrass-nutrient loading paradigm from east coast systems may not be directly transferable to California or the West Coast in general.

There are a variety of local and regional processes that effect nutrient dynamics on the west coast that are not as prevalent on the east coast. For example, seasonally closed estuaries and large scale upwelling are not primary features in East coast systems and as a result are not considered in many of the conceptual models that develop from research in that region. Other factors include land use patterns and the fact that in some portions of *Zostera's* range "natural" nutrient loads far exceed those calculated for even the most eutrophic systems on the East Coast (e.g., Waquoit Bay, MA).

There have been relatively few studies of *Zostera marina* nutrient interactions on the West Coast or in California. In a field study from Padilla Bay, WA, Williams and Ruckelshaus (1993) demonstrated that eelgrass growth was influenced by both sediment nutrient availability and higher order effects of epiphytes and isopod grazer interactions. They conclude that "Consideration of sediment nitrogen, epiphytes or herbivores alone is unlikely to yield a predictable understanding of the control of eelgrass productivity in nature, particularly given the complexity of the eelgrass habitat with respect to its dual nutrient sources" (Williams and Ruckelshaus, 1993). This conclusion suggests that external nutrient loads and internal recycling alone do not control eelgrass growth and production, but that integrated water quality, biological and environmental factors play a considerable role (Koch, 2001). Therefore, the use of seagrass health as a primary indicator of eutrophication is problematic in that other stressors (temperature, excessive sedimentation, climate change) may be adversely impacting health.

Estuarine macrophyte community response to nutrient loading occurs primarily as a shift in dominant primary producers. Macroalgae interact with *Zostera* spp. in a variety of ways. An overabundance of macroalgae can cause degradation of *Zostera* habitat through two related mechanisms. First, mats or rafts of algae can develop over-topping or among seagrass shoots that effectively limit the amount of light available to seagrass. However, some systems can exhibit large accumulations of macroalgae growing among *Z. marina* shoots with no apparent decline in seagrass condition driven by seasonal upwelling of nutrients (J. Kaldy, USEPA, pers. observation). Secondly, mats or rafts of algae that settle on top of the seagrass and effectively smother the plants, cutting off light and oxygen leading to anaerobic conditions with a build of toxic metabolites (e.g., sulfides).

Work conducted in European estuaries indicates that the brackish SAV species *Ruppia* spp. acts as a seagrass analog and is susceptible to degradation based on the same types of interactions (shading, smothering, biogeochemical stressors, etc.). There are several examples from Europe that examine how the systems respond to losses of *Ruppia* associated with eutrophication or other anthropogenic activities (Bachelet et al., 2000; Lenzi et al., 2003; Pergent et al., 2006; Shili et al., 2007). Bachelet et al. (2000) investigated an eutrophication gradient along the coast of France; the intermediate site was characterized by *Ruppia* with a constant biomass with sporadic spring blooms of macroalgae. In contrast, the eutrophic site was dominated by macroalgae and had low biomass and abundance of macrozoobenthos (Bachelet et al., 2000). In the Orbetello lagoon (Italy), eutrophication abatement measures (macroalgal harvesting, increased circulation and waste water phytotreatment) resulted in reductions of algal biomass and increased seagrass (Giusti and Marsili-Libelli, 2005). More recent macroalgal blooms appear to be a "legacy effect" of sediment nutrient release (Lenzi et al., 2003). At Biguglia lagoon in Corsica, a healthy bed of *R. cirrhosa* was replaced by Ulvoid algae between 1997 and 1998 with re-appearance of *R. cirrhosa* in 1999. Pergent et al. (2006) attribute these shifts to nutrient availability related to agricultural runoff and wastewater discharge. Several studies have investigated the response of *Ruppia* spp. nutrient loading; however, these studies use a gradient approach where there is little control over or quantification of the loading to the system. Thus, for European *Ruppia*, well defined load -response experiments do not appear to exist. The USEPA Chesapeake Bay Program explicitly assumes that all SAV species follow the same conceptual model where nutrients increase light attenuation by phytoplankton and epiphytes leading to declines of SAV (Batuik et al., 2000).

Manipulative experiments in Maryland concluded that epiphytes, stimulated by nutrient additions, caused declines in *Potamogeton perfoliatus* (Staver, 1984).

In the Delta and in southern and central California lagoons, very dense and apparently healthy brackish SAV populations appear to persist under very eutrophic conditions (high nutrient loading, high organic loading to the sediments, fish kills, large diurnal dissolved oxygen swings, etc.). The presence of dense brackish SAV beds has been observed primarily in Southern California ICOLL (Sutula & McLaughlin, SCCWRP, unpublished data), but also occurs in the Klamath River in Northern CA (Lee and Brown, USEPA, unpublished data). It is not clear if these beds are adapted to and thrive under high nutrient conditions or if these populations are an expression of eutrophication symptoms. The beds tend to be seasonal and it is unknown what triggers the reduction of biomass and subsequent decline of these apparently annual populations. Alternatively, the presence of these dense, ephemeral California populations may be an expression of the natural life-cycle of this species. In Chesapeake Bay, the growth form of seagrass and SAV are classified as “canopy forming” and “meadow forming,” respectively (Batuik et al., 2000). Brackish SAV species tend to be “canopy formers” with biomass concentrated in the top half of the water column and exhibit rapid growth toward the surface early in the growing season. Canopy formation results in shading of older portions and the sloughing of lower leaves. Epiphytes accumulate on the older portions of the leaves and continued growth results in epiphyte free apical leaves near the surface of the water that actively photosynthesize. In contrast, “meadow forming” species concentrate biomass in the lower portion of the water column and new leaf production occurs near the base of the plant. Older leaf tissue near the surface may be heavily epiphytized but rapid leaf turn-over rates allow the plants to maintain positive carbon balance. Additionally, it should be noted, that changes in the distribution of *Ruppia*, and probably other brackish SAV as well, can be related to factors other than nutrients and eutrophication. In the Ichkeul lagoon (Tunisia) rapid changes (1993-1998) in the species composition and distribution of SAV, including *Ruppia cirrhosa* were linked primarily to water management activities (e.g., dams) coupled with drought and not eutrophication (Shili et al., 2007). In San Diego, California, a shift in community dominance from *Z. marina* to *R. maritima* in San Diego Bay were likely related to increased water temperature associated with the 1997-1998 El Niño event (Johnson et al., 2003).

In general, a better understanding of the response of brackish SAV to alterations in nutrient loading requires substantial research before it could be used as an indicator. Key research questions that need to be addressed before brackish SAV will be useful indicators of eutrophication include: First, the basic physiological requirements (salinity tolerances, temperature tolerances, nutrient requirements, minimum light requirements, etc.) of brackish SAV species need to be defined for California. Second, the environmental triggers to seasonal cycles of biomass (temp, salinity, day length, etc.) of both meadow and canopy forming SAV need to be elucidated. Third, nutrient dose-response relationships need to be determined with emphasis on how the response is manifested (e.g., epiphyte loads, light reduction, self-shading from canopy development, etc.).

Sound and Practical Measurement

A suite of indicators are generally used to assess seagrass health and effects from stressors. For the purposes of this review, these indicators can be grouped into three categories:

1) Indicators of seagrass and SAV community structure (taxonomy, biomass, aerial distribution, density)
2) Factors that affect seagrass health through reduced light availability to the plant (e.g., water column light attenuation, total suspended solids, phytoplankton biomass, epiphyte load, macroalgal biomass or cover)
3) Other indicators (environmental or water quality)

Of these, three groups, Sutula (2011) conclude that those indicators that affect seagrass through reduced light availability are candidates for the California Estuarine NNE framework. Indicators of seagrass community structure (taxonomy, biomass and aerial distribution) are important collateral data to track overall trends in the condition of this key habitat type, but do not uniquely respond to stressors.

The primary mechanism of seagrass loss from eutrophication is through the reduction in available light to plant leaves caused by shading or smothering from algal competitors. Reduced light coupled with increased delivery of labile organic detritus (senescent algae and seagrass) to the sediments can lead to additional biogeochemical stressors (hypoxia/anoxia, sulfide toxicity, etc.) that further exacerbate the problem. Seagrass and SAV beds adversely affected by eutrophication tend to be sparse or patchy, heavily epiphytized with microalgae, and/or shaded with phytoplankton or macroalgal blooms. Thus epiphyte load, water column light attenuation (from attendant phytoplankton biomass and turbidity), and macroalgal biomass are indicators of eutrophication that directly affect light availability to seagrass. Canopy forming SAV are even more complicated because in addition to all of the other factors that attenuates irradiance the canopy formers also self-shade. That is, by having most of their biomass at the surface of the water, the plant absorbs and attenuates light before it can reach the deeper leaves.

Water column light penetration is a dominant factor controlling the growth and distribution of seagrass and SAV. Although, water column light attenuation cannot be directly related to nutrient loading, monitoring of underwater light is likely to be a critical component of evaluating eutrophication because all of the algal groups that respond to nutrients influence the underwater light field. Water column turbidity is generally not related to nutrient loading except in circumstances such as river dominated portions of estuaries and is likely to not be a useful indicator of eutrophication, although it does contribute to water column light attenuation. Water column chlorophyll a (chl a), which is a surrogate measure for phytoplankton, responds to nutrient loading and influences underwater light availability for seagrasses and SAV. Consequently, monitoring of chl a may be a strong indicator of eutrophication under some conditions. Table 5.4.3 provides an overview of available methods to measure reduced light availability to seagrass. Overall, sound and practical methods exist to characterize light attenuation to seagrass beds (see Sutula (2011) for detailed review).

Table 5.4.3. Summary of literature reviews for candidate SAV and seagrass related indicators for E-NNE. Excerpted from Sutula (2011).

Group	Indicator	Methods	Information	Summary of Review
Light Attenuation	Epiphyte Cover or Load	Visual rapid assessment, empirical sampling	Relative abundance of competing primary producers	There is a relationship between nutrient loading and epiphyte biomass but epiphyte load is confounded with a variety of other parameters and is unlikely to have a good "signal to noise ratio". As a result it is not likely to be a good stand-alone tool for detecting eutrophication. A field assessment may work in conjunction with additional metrics. **Recommend to pursue in conjunction with other metrics associated with light attenuation.**
	Light Attenuation	Grab samples or Continuously deployed Instrumentation (e.g., data sondes)	Attenuation of light reaching seagrass or SAV bed	Science exists, but assessment framework needs to be refined for California use. Due to species specific requirements and location specific characteristics application of this metric will require additional research and validation. **Recommend to pursue in conjunction with other metrics associated with light attenuation (epiphyte load, chlorophyll a, turbidity), possibly as a rapid assessment to determine whether additional intensive diagnosis is warranted.**
	Chlorophyll a, Turbidity or TSS		Surface water Chl a biomass and turbidity or TSS	Science exists, but assessment framework needs to be refined for use in California. **Recommend to pursue in conjunction with other metrics associated with light attenuation (epiphyte load, chlorophyll a, turbidity, macroalgal cover/biomass), possibly as a rapid assessment to determine whether additional intensive diagnosis is warranted.**
	Macroalgae	Field survey	Biomass or cover	Science exists but data is required to develop an assessment framework for California estuaries (see Sutula (2011). **Recommend to pursue in conjunction with other metrics associated with light attenuation.**

*parameters may include enzyme assays, photosynthetic characteristics, carbohydrate content, etc.

Acceptable Signal to Noise Ratio

Sutula (2011) reviewed indicators of eutrophication that affect seagrass habitat. The primary indicators, epiphyte load, macroalgal cover/biomass, and light attenuation (monitored through a combination of light attenuation, water column chlorophyll a, and turbidity) are known to have considerable spatial and temporal variability. As noted for phytoplankton, high spatial and temporal variability is characteristic of all biological indicators, these indicators tend to integrate better over time and space than stressors, such as nutrient concentrations. Ultimately, our understanding and the various hypotheses about controls on spatial and temporal variability in seagrass aerial distribution vis-à-vis nutrient loads and other stressors can be refined through predictive models.

Approaches to Setting Numeric Endpoints Based on Rafting Macroalgae, Epiphytes, and Water Column Light Attenuation

In order to develop NNEs that are protective of seagrass and SAV habitat, an assessment framework is needed to integrate the effects of rafting macroalgae, epiphyte load, and water column light attenuation on seagrass health. The fundamental step in developing such a framework is to assess the availability of studies which document the effects of these stressors, either as a single or multiple effects, on seagrass.

Precedent for establishment of numeric endpoints for light attenuation to seagrass beds from phytoplankton biomass and turbidity or water clarity, based on light limitation for photosynthesis of seagrass (e.g., Janicki et al., 2000, Kemp et al., 2009, Brown et al., 2004, Sutula, 2011). Turbidity, total suspended solids (TSS), chlorophyll a, and dissolved organic matter are measured to determine light available in the water column that reaches the seagrass bed (Biber et al., 2008). For example, in the mid-Atlantic, environmental conditions that allow adequate light penetration for SAV survival are TSS less than 15 mg L^{-1} and chlorophyll a less than 15µg L^{-1} (Kemp et al., 2004). Bio-optical models predicting light attenuation under various environmental conditions have been calibrated for the Chesapeake Bay (Gallegos, 2001), Indian River Lagoon in Florida (Gallegos and Kenworthy, 1996), and North River in North Carolina (Biber et al., 2008), Yakina Estuary in Oregon (Brown et al., 2007), and Tampa Bay in Florida (Janicki et al., 2000). Explicit studies are needed to understand the precise light requirements of seagrass in SF Bay. This information can be used to develop a biooptical model that could be used to establish a combination of chlorophyll a thresholds and turbidity to establish levels of light availability required to support a healthy seagrass bed.

A strong relationship exists between epiphyte load and light reduction (Boese et al., 2009). However, epiphyte load is generally not quantified in most seagrass or SAV monitoring programs or is quantified using relative abundance. Epiphyte load and subsequent light reduction are highly variable both spatially and temporally, even at the scale of individual plants. There are differences in epiphyte load between wet and dry seasons, location in the estuary and between younger inner leaves and older outer leaves (Boese et al., 2009). One approach to develop an assessment framework to diagnose eutrophication in seagrass beds may involve quantifying effects of rafting macroalgae or water column light attenuation with categories (high, medium and low) epiphyte loading. Another approach, used by the Chesapeake Bay Program, utilizes light attenuation by epiphyte load, in addition to light attenuation by the water column (k_d) from water column suspended sediment and phytoplankton, in efforts to evaluate SAV light requirements. An epiphyte attenuation coefficient is also calculated (k_e) and used with epiphyte biomass (B_e) to predict the percent light reaching the leaf surface (PLL) as described by Batuik et al. (2000).

$$PLL = \left(e^{-(k_d)(z)}\right)\left(e^{-(k_e)(b_e)}\right) * 100$$

This is based on site-specific empirical measurements made over 30 years throughout Chesapeake Bay. Exporting this concept to other estuaries is problematic and would need to be developed for specific estuaries in California.

Studies documenting effects of rafting macroalgae on seagrass bed density have likewise been published (see Sutula (2011) for full review), but few of these studies are useful specifically for establishing endpoints. As with macroalgae on intertidal flats, studies are likewise needed to document the effects of rafting mat biomass and duration on seagrass bed density in California estuaries. These studies are ongoing in other California estuaries as part of the larger estuarine NNE research program. The outcome of these studies should be evaluated for applicability to SF Bay seagrass habitats.

Little documentation is available on the effects of rafting algae on brackish SAV environments. Conceptual models from applicable habitats could be applied, but few field studies have been conducted to illustrate the effects of rafting mats on dissolved oxygen, other pelagic or benthic food webs. Thus, foundational studies are needed to better understand the effects of macroalgae on brackish SAV.

5.4.6 Summary: NNE Indicators Protective of Seagrass and Brackish SAV Habitat

Seagrass

Overall, rafting macroalgae, epiphyte loads and the portion of water column light attenuation influenced by phytoplankton biomass appear to satisfy the four evaluation criteria to be considered as an NNE indicator for seagrass habitats in SF Bay. In order to pursue the use of these indicators for diagnosing and managing eutrophication, several key data gaps need to be addressed. These include:

- Studies to establish thresholds of macroalgal biomass, cover and duration that adversely affect seagrass habitat
- Studies that establish light requirements for seagrass beds in different regions of SF Bay and assessment of duration of reduced light/photosynthesis that results in adverse effects to the seagrass bed.
- Determination of thresholds of the frequency, duration and magnitude of phytoplankton biomass which would result in adverse effects of phytoplankton.
- Development and validation of site-specific dynamic simulation models that simulate reduced light availability to seagrass beds from nutrient loads and other co-factors. This modeling could be done in concert or separately from SF Bay water quality models discussed.

Brackish SAV

Though brackish water SAV are an important component of the Delta, little documentation exists on the extent and ecology of these primary producer communities in the North Bay. Studies are funded to characterize the structure and stressors associated with *Stuckenia pectinata* (Sago Pondweed) beds in the North Bay (K. Boyer, personal communication. Literature from intermittently tidal Mediterranean estuaries and the Chesapeake Bay suggests that brackish SAV species decline in response to eutrophication. The mechanism of decline is presumably mediated through light limitation caused by epiphytes, phytoplankton or macroalgal blooms, though there is relatively poor documentation of response of SAV with nutrient loads and other co-factors (temperature, salinity, etc.). This literature contradicts anecdotal observations of brackish water SAV in the Bay Delta and in intermittently tidal estuaries, where very dense and apparently health *Ruppia* populations exist under very eutrophic conditions (high nutrient loading, high organic loading to the sediments, fish kills, large diurnal dissolved oxygen swings, etc. No clear documentation exists of dose-response relationship between elevated biomass of SAV and secondary consumers such as neither water column macroinvertebrates, nor documentation of changes in bed extent, biomass or density as a function of nutrient loading. The following key data gaps should be addressed in order to pursue the use of brackish water SAV for diagnosis of eutrophication in the North Bay.

- Document nutrient load- SAV community response through long-term monitoring at established sites
- Document the relationship between SAV community structure, indicators of light availability (epiphyte load, chlorophyll a biomass, macroalgal cover/biomass, etc.), dissolved oxygen, pH, and indicators of aquatic life use (macroinvertebrates, fish, etc.)
- Document the growth habits of these plants and elucidate mechanisms of water column versus sediment response to nutrient loads. Detailed physiological and autecological studies of brackish SAV species need to be undertaken in order to better understand the habitat requirements of these communities.

It should be noted that this indicator will likely be of interest in the Delta. Therefore, opportunities exist for synergy on research to address this data gap with what could be proposed for the Delta.

5.5 Macroinvertebrates

Macrobenthic fauna or macrobenthos are invertebrates living on and within the sediments of aquatic waterbodies. Macrobenthos are one of the primary tools used to assess the ecological condition of estuaries and coastal nearshore habitat because 1) they live in bottom sediments, where many stressors accumulate; 2) most macrobenthos are sedentary and therefore reflect the quality of their immediate environment (Pearson and Rosenberg, 1978; Dauer,1993; Weisberg et al., 1997); 3) most communities are comprised of a diverse array of species with a variety of tolerances to stress, so the presence or absence of different taxa can provide information about the types of stressors present (Christman and Dauer, 2003; Lenihan et al., 2003); and 4) they serve as food sources for many ecologically and economically important estuarine fish and birds (Virnstein, 1979; Phil et al., 1992; Gillett, 2010). Macrobenthic community-based assessment tools have traditionally been designed to assess overall habitat quality, successfully integrating a variety of anthropogenic stressors (e.g., contaminants, eutrophication, or physical disturbance) while accounting for gradients in natural stressors/environmental conditions (e.g., salinity, sediment type, or depth). Within the macrobenthos, there are a variety of aspects that can be used in environmental assessment, including individual responses (e.g., condition indices, cellular bioindicators, or contaminant loads) (Ringwood and Keppler, 1998; Brylawski, 20089), as well as community-level responses (e.g., abundance of sensitive/tolerant taxa, community composition changes) (Weisberg et al., 1997; Borja et al., 2000; Smith et al., 2001; Llansó et al., 2002). The community structure of macrobenthic infauna has been used as an indicator of ecosystem health and environmental stress for a number of years in a variety of estuarine habitats around the United States, including the USEPA Environmental Monitoring and Assessment Program (EMAP), National Coastal Assessment (NCA), Chesapeake Bay Benthic Monitoring Program, Southern California Bight Regional Monitoring Program, California Sediment Quality Objective, and internationally the European Union Water Framework Directive (WFD).

Three types of macrobenthic indicators have been considered for assessing eutrophication: 1) taxonomic composition, 2) abundance, and 3) biomass. Gillett provides a full review of these indicators for the California estuarine NNE framework (see Chapter 7 in Sutula (2011), and that work is incorporated in this review.

5.5.1 Applicable Habitat Types

Macrobenthic infauna are valuable tools for environmental assessment in estuaries because, in the absence of long-term hypoxic conditions (e.g., main-stem Chesapeake Bay), they can be found throughout all soft sediment habitats found in estuaries; from euhaline sandy sediments through tidal freshwater muds or from deep subtidal waters through the littoral zone. These macrobenthic communities, however, are not uniform across these gradients in physical habitat, with relatively unique communities in each salinity and sediment regime (e.g., Sanders, 1958; Holland et al., 1987; Attrill and Rundle, 2002; Ranasinghe et al., 2010). Consequently, assessment tools developed to work across the entire spectrum of estuarine habitats use a categorical approach to ecological condition assessment. Different aspects of community structure and/or different thresholds of community characters for the different salinity zones – typically following the Venice classification scheme (International Association of Limnology, 1958) – and sediment types – typically either sands or muds – found in an estuary (e.g., Weisberg et al., 1997; Van Dolah et al., 1999; Llansó et al., 2002). The lower salinity (<5 psu) portions of estuaries are notoriously difficult systems for the application of macrobenthic community changes in assessing habitat quality due to the salinity fluctuations and high turbidity, which act as stressors to the community, as well as the pervasive human perturbations typically found there throughout the United States and Europe (Draheim, 1998; Alden et al., 2002; Attrill, 2002; Diaz et al., 2004). Estuarine habitats that encompass a large amount of intertidal area additionally problematic, because at low tide the air exposure can create desiccation and large fluctuations in temperature that can impact community diversity, abundance, and biomass in comparison to adjacent subtidal habitats (Van Dolah et al., 2000; Holland et al., 2004).

5.5.2 Availability of Data on Macrobenthos in San Francisco Bay

Surprisingly, there are no recent summaries of the soft-bottom benthic community of the entire Bay and Schaeffer et al (2007) bemoan the challenge of keeping such a summary up to date given the high rate of non-native species introductions into the system (Schaeffer et al., 2007). The only comprehensive summary of any compartment of the soft bottom benthos was written in 1986

(Nichols and Pamatmat, 1988), just prior to the invasion of *Corbula amurensis*, a bivalve that greatly altered the soft-bottom benthos community structure (Alpine and Cloern, 1992; Schaeffer et al., 2007).

The data used to describe macrofauna in this system are limited to the post-Corbula invasion period due to the dominance of the bivalve *Corbula* in the communities where it resides. The major data sources and their abbreviations include: 1) long-term monitoring data collected monthly from the freshwater Delta to the Richmond San Rafael Bridge by the California Department of Water Resources (DWR); 2) 2 to 3 year bimonthly data collected by the Regional Effects Monitoring Program in 1986 to 1989 (REM); 3) long-term near-monthly data collected by the USGS south of Dumbarton Bridge in Palo Alto (USGSPA); 4) the summary of semi-annual data collected by various agencies as listed in Thompson et al. 2000 (Regional Monitoring Program [RMP], Long-term monitoring program [LMP], and Bay Protection and Toxic Clean-Up Program [BPTCP]); 5) samples taken as part of NOAA's National Coastal Assessment and the USEPA Environmental Monitoring and Assessment Program (EMAP) West Coast pilot (2000-2001); 6) a monthly study of the bivalves south of San Mateo Bridge collected by the USGS in 1990 to 1996 (Thompson, 2005, 1999); and 7) unpublished rapid assessment survey data from the California Academy of Sciences (C. Brown, Smithsonian Institute, pers. comm.; Schaeffer et al., 2007).

5.5.3 Indicator Relationship to Nutrients and Water Quality

Excessive amounts of nutrients that lead to excessive amounts of primary production and eutrophication typically do not have direct impacts on macrobenthic fauna, with the exception of HABs (e.g., Anderson et al., 2002). Eutrophication primarily affects the macrobenthos via two basic microbially-mediated, indirect paths: 1) water column hypoxia/anoxia or 2) the accumulation of toxic reduced sulfides and ammonium in the sediment.

As heterotrophic microbes consume the organic matter from the primary producers, oxygen is removed by aerobic microbes and reduced compounds are created as metabolic byproducts. Even in natural, non-eutrophic conditions, these processes occur in both muddy and sandy sediment environments and the fauna that live there are adapted to deal with low-oxygen, reducing environments. As the amounts of organic matter produced and accumulated in the system, the low oxygen and reduced conditions begin to either smother or poison the benthic fauna. These processes lead to progressive changes in the abundance, biomass, and composition of the macrobenthic community and eventually lead to azoic conditions. By looking at trajectories and magnitudes of these changes in community, one should be able to distinguish between the effects of eutrophication, as well as changes brought about by other common estuarine stressors (e.g., contaminants, physical disturbance, or salinity fluctuation).

Effects of Hypoxia

Most of the information detailing the response of macrobenthic fauna to eutrophication is related to the effects of low-oxygen (i.e., hypoxia or anoxia) on macrobenthic communities. Benthic sediments in estuaries are naturally low-oxygen environments because of the large amounts of organic matter and large number of heterotrophic microbes there. As a consequence, most benthic fauna have evolved to deal with those conditions (Pearson and Rosenberg, 1978; Hargrave et al., 2008), but hypoxic or anoxic conditions in the overlying water can be an important factor structuring the composition of an ecosystem (e.g., Rosenberg et al., 1991; Diaz and Rosenberg ,1995; Baustin and Rabalais, 2009; Seitz et al., 2009).

The response of the macrobenthic community to hypoxic conditions is primarily negative. Increases in frequency and duration of hypoxic (<2.0 mg O_2 L^{-1}) or anoxic (<0.5mg O_2 L^{-1}) conditions lead to reduced community diversity, biomass, and productivity and eventually complete absence of macrofauna (Gray et al. 2002; Rakocinski 2009; Seitz et al. 2009). The degree of the response in these broad, community attributes and the trajectory of community changes will vary, depending upon the severity and duration of hypoxic conditions. Tolerance to low oxygen conditions varies widely among the taxonomically diverse macrobenthic community, though persistent anoxic conditions will eventually kill all metazoans (e.g., main stem Chesapeake Bay, Gulf of Mexico, coast of Oregon) (Holland et al., 1977; Diaz and Rosenberg, 2008; Rabalais et al., 2010). Among the most common types of estuarine macrofauna, crustaceans and gastropods are typically the most sensitive (LT_{50} anoxia <1 d), annelids the most tolerant (LT_{50} >5 d), and bivalve mollusks in-between, as different species have differing capabilities of sealing themselves off to the environment and waiting for better conditions (Llansó, 1992; Sagasti et al., 2001; Gray et al., 2002; Calle-Delgado, 2007).

Water column hypoxia can also have indirect effects on macrobenthic survival and community structure by altering behavior that increases the risk of being preyed upon. As oxygen concentrations near the bottom decline, many species of infauna will start to move closer the sediment surface in an effort to extend appendages or siphons further up into the water column in search of oxygenated water (Rosenberg et al., 1991; Llansó, 1992; Long et al., 2008). Eventually, continued exposure to low oxygen forces many infaunal species from the sediment entirely and they remain moribund on the sediment surface, which greatly increases their exposure to predation by benthivoric nekton (Nestlerode and Diaz 1992; Pihl et al. 1992; Seitz et al. 2003; Powers et al. 2005).

Effects of Increased Sediment Organic Matter Accumulation

Eutrophic conditions do not always lead to hypoxia and anoxia, but can still have effects on the macrobenthic community of estuaries. Hypoxic conditions are, in part, a function of water column stratification and water residence time (Diaz and Rosenberg, 1995; Hagy et al., 2004; Kemp et al., 2009) and many of California's estuaries that are always connected to the open ocean are not always prone to the formation of chronic hypoxic bottom waters. As such, understanding the effects of non-hypoxic eutrophication on the macrobenthos will be particularly relevant to California's estuaries.

Almost every modern work on the effects of eutrophication and the accumulation of organic matter on benthic fauna is based upon the conceptual model of Pearson and Rosenberg (1978). This paper summarizes one of the central tenets of benthic ecology: that there are relatively consistent and predictable changes in macrobenthic community structure with increasing accumulation of organic matter in marine sediments (Figure 5.5.1). In short, the model proposes that: 1) under normal, non-eutrophic conditions, a benthic community should be composed of a trophically and functionally diverse array of species that span different body sizes and lifespans, as well as live at various depths through the sediment, often extending 10's of cm below the sediment-water interface[5]; 2) as organic matter begins to accumulate in the sediment and there will be changes in the community, shifting towards a less diverse community composed of smaller fauna with relatively short lifespans living near the sediment surface; and 3) eventually the sediments are devoid of macrofauna and are covered in mats of sulfur-oxidizing bacteria (i.e., *Beggiatoa*). The presence of benthic infauna will typically enhance the depth of oxygen penetration due to tube building/ventilating and bioturbation. As a system becomes more eutrophic and organic matter begins to accumulate at greater rates in the sediment, bacterial production is stimulated and the demand for oxygen outstrips the rates of diffusion. This leads to anoxic, reducing processes dominating formally oxygenated sediment, and a variety of bacterial metabolic pathways that produce byproducts (primarily sulfide and ammonium in saline sediments) that are toxic to most metazoans (Pearson and Rosenberg, 1978; Jørgensen, 1996; Gray et al., 2002; Hargrave et al., 2008). These compounds and the reducing environment of the sediments are thought to be the mechanism behind the mortality leading to changes in community structure. Many of the species that are community dominants in disturbed habitats are always present at low densities and presumably at a competitive disadvantage to non-disturbed community dominants. Only when the non-disturbed dominants die off are there available resources that allow tolerant fauna to flourish (e.g., Gillett et al., 2007).

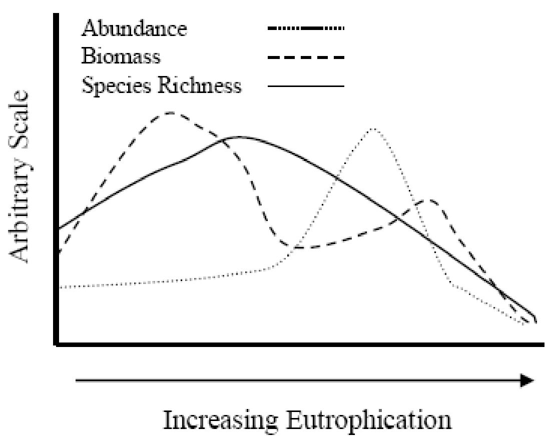


Figure 5.5.1. Conceptual patterns of abundance, biomass and species richness with increasing eutrophication. After Pearson and Rosenberg, 1978.

[5] In practice, this kind of community should only be expected in relatively high salinity environments (>10-15 psu) with relatively little salinity fluctuation. The premise of community change is still appropriate in lower, more variable salinity environments, but the baseline community will likely be less trophically diverse and more tolerant of environmental stressors than higher salinity communities, as detailed in our subsequent discussion of the so called "estuarine quality paradox".

Most of the historical literature (summarized in Nixon, 1995; Grall and Chauvaud, 2002; Gray et al., 2002; Hyland et al., 2005) focuses on the negative impacts of eutrophication in general and macrobenthic fauna in specific. There has been recent interest, however, in how eutrophication can have both positive and negative effects on the functioning of macrobenthic communities in estuarine ecosystems (e.g., Beukema and Cadée, 1997; Nixon and Buckley, 2002; Rakocinski and Zapf, 2005; Gillett, 2010). By definition, eutrophication typically leads to an increase in the primary production of a system and this represents an increase in food availability for primary consumers, which has been linked to increases in benthic production, as well as fisheries yields (Nixon and Buckley, 2002; Breitburg et al., 2009; Nixon, 2009). Rakocinski and Zapf (2005) put forth a conceptual model of changes in macrobenthic function with increasing eutrophication that incorporates both the positive and negative aspects of eutrophication on benthic communities (Figure5.5.2). In this model, as a system begins to become eutrophic, there is an increase in the rate of macrobenthic function. This increase is related to increases in primary production, which provide a release from food limitation for existing fauna (e.g., Marsh and Tenore, 1990; Sterner et al., 2002; Brylawski, 2008), as well as beginning to alter the sediment biogeochemistry, allowing for the eutrophication-tolerant taxa to increase their proportion within the community before the sensitive taxa are severely impacted.

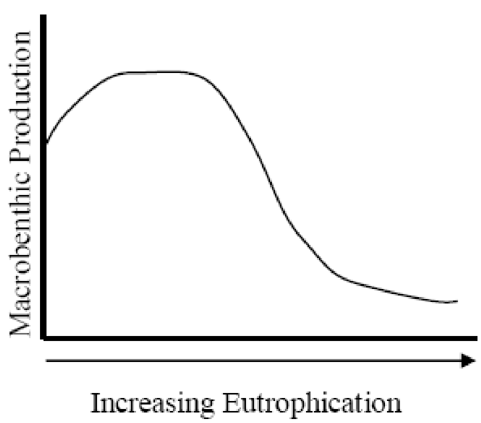


Figure 5.5.2. Conceptual relationship between macrobenthic production and eutrophication. After Gillett, 2010.

As the degree of eutrophication progresses, the model predicts that there will be decline in community function due to the negative aspects of organic matter accumulation (reduced O_2 penetration and the build-up of toxic reduced compounds) outweigh the benefits of additional food and the composition of the macrobenthic community changes, following the classic pattern of habitat degradation described in the preceding paragraphs. Gillett (2010) showed an initial increase in secondary production with increasing eutrophication, followed by a decline in production as eutrophic conditions continued to intensify. There are two aspects of this eutrophication pattern that bear further detail: 1) macrobenthic community composition is important – the positive aspects of eutrophication (i.e., the fertilization effect) most strongly affected filter- and interface-feeding fauna that could directly utilize the increases in phytoplankton production in the water column (Gillett, 2010), whereas other types of benthic fauna remained unchanged until the negative aspects became dominant; and 2) the benthic-pelagic setting is important – the macrobenthos from sandy, non-depositional habitats appear to have a greater buffer to eutrophication and the macrobenthic community experience negative impacts slower than their counterparts from depositional habitats, where the sediments are naturally rich with organic matter. These depositional habitats can be quickly oversaturated with organic matter and therefore habitat quality will start to degrade with only a small increase in eutrophication (Molinaroli et al., 2009).

The concepts of eutrophication having positive benefits to the macrobenthic community, while still representing a change in ecosystem condition from reference, has been incorporated into a small number of environmental monitoring programs that utilize the macrobenthos as their assessment tool (Chesapeake Bay Program – Weisberg et al., 1997; Mid-Atlantic US – Llansó et al., 2002; European Water Framework Directive – Lavesque et al., 2009). In these indices, which are largely built upon the Pearson and Rosenberg (1978) paradigm, macrobenthic abundance and biomass do not have a simple, positive linear relationship with habitat quality. Instead, they have a concave, unimodal relationship to habitat quality, where a sample can be assessed as degraded for having too much or too little biomass/abundance (Weisberg et al., 1997). It should be noted, however, that these indices were developed to assess overall habitat quality or integrity, not individual stressors on the macrobenthic community. However, there has been some work in recent years to use specific aspects of the macrobenthic community to assess multiple stressors impacting ecosystem quality. Christman and Dauer (2003) and Dauer et al. (2000) were able to detect the differential response of the macrobenthic community in Chesapeake Bay to low oxygen stress and chemical contaminant stressors by looking at variation benthic multi-metric index (Chesapeake Bay B-IBI [Weisberg et al., 1997]) scores in relation to environmental conditions. Furthermore, Dauer et al. (2000) were able to relate index score to different types of watershed development (urban, agricultural, and forested) and local water/sediment quality. Lenihan et al. (2003) were able to differentiate macrobenthic community responses to either organic matter enrichment or heavy metal contamination. In this study, they showed positive responses among annelids (i.e., increases in abundance and biomass) with organic matter enrichment, even when combined with increasing concentrations of heavy metals. Conversely, echinoderms had slightly positive responses to organic enrichment, but declined when exposed to heavy metals and arthropods declined with increased exposure to both types of stressor (Lenihan et al., 2003). This type of differential response by separate components of the macrobenthic community to different stressors could be used to delineate eutrophic impacts from the mix of co-occurring stressors typically found in estuarine ecosystems.

5.5.4 Species composition in the San Francisco Bay

Populations of aquatic organisms in the upper portions of SF Bay have undergone significant declines over the past several decades (Jassby et al., 1995; Carlton, 1979). The benthic macroinvertebrate community of the SF Bay is composed of less than 40 species, most of which were introduced in the 19th century when oysters were imported from the eastern coast of the United States and grown in the Bay, and from a lack of regulation of ballast water (Carlton, 1979; Nichols et al., 1986; Nichols and Pamatmat, 1988; Thompson et al., 1999; Cohen and Carlton, 1998). The results of a 10-year study of the invertebrates living on a mudflat at the south end of the Bay (Nichols and Thompson, 1984) show that species composition and relative abundance have remained fairly constant, at least in the second half of the 20th century (Nichols and Pamatmat, 1988). Studies are often separated geographically along a salinity gradient (Figure 5.5.3), as well as by composition of the substrate of the Bay floor (Figure 5.5.4).

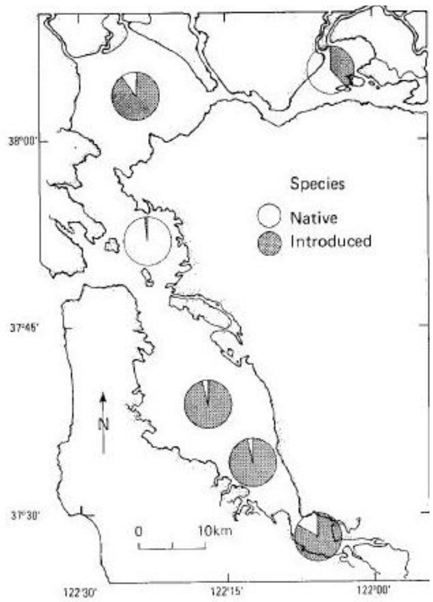


Figure 5.5.3. Proportions of introduced and native species relative to biomass of mollusks in San Francisco Bay (Nichols and Pamatmat, 1988).

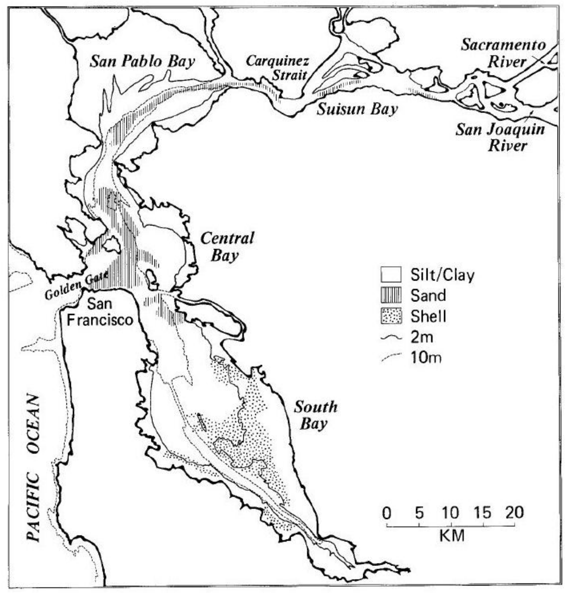


Figure 5.5.4. Generalized distribution of surface sediment composition in San Francisco Bay (Nichols and Pamatmat, 1988).

5.5.5 Factors Effecting Temporal and Spatial Variation of Indicator

The SF Bay experiences natural fluctuations in tide, salinity, nutrient and sediment loads, sediment deposition, water clarity and temperature due to variations in freshwater flows, semidiurnal tides, and seasonal winds as well as historic and recurring anthropogenic influences including nutrient and organic enrichment, and contamination. It follows that the benthic community composition in the SF Estuary responds to these many types of physical, chemical, and biological fluctuations, both spatially and temporally (Nichols, 1985; Nichols and Pamatmat, 1988; Thompson et al., 1999).

Much of the spatial distribution of benthic organisms can be tracked along a salinity gradient between the marine environments of the Central and South Bay, to the northern areas of the San Pablo and Suisun Bays which are characterized by temporally shifting brackish and freshwater landward and seaward boundaries of the estuary (Nichols and Pamatmat, 1988; Jassby et al., 1995). In terms of temporal variability, in the temperate climate of SF Bay, food availability (associated with phytoplankton and microphytobenthos blooms) may be the most important factor in the reproductive cycle and timing of macroinvertebrates (Nichols and Pamatmat, 1988), but as mentioned earlier, predation by shallow feeding birds can also be important some years. Observed seasonal changes in abundance of benthic macrofauna have also been linked to seasonally predictable variations in freshwater inflow, winds and tides leading to water column mixing, microalgal biomass, and sediment erosion and deposition patterns (Nichols and Thompson, 1985; Nichols and Pamatmat, 1988).

Spatial Variability

The SF Bay, except the central portion nearest the Golden Gate, is very shallow with characteristically wide intertidal and shallow subtidal mudflats incised by narrow mid-Bay channels (Thompson and Nichols, 1985; Lucas et al., 2009). However, some conceptualize the Bay as having two estuarine systems, each with a different hydrodynamic and freshwater inflow regime. Historically, North and South Bays have had very different macrobenthic invertebrate communities, whose distribution is most strongly influenced by spatial variations in salinity and composition and stability of sediments (Nichols, 1979, Nichols, 1985; Nichols and Pamatmat, 1988). Suspension-feeding bivalves were found in the 1970s to be less abundant in the northern Bay, potentially because of high-suspended sediment loads and large seasonal variations in salinity (Cloern, 1982). Later, beginning about 1986, an invasive clam Corbula amurensis recruited and is now thought to be one of the dominant controls on algal productivity in the northern reaches of the Bay (Jassby, 2008). The South Bay is a “lagoonal system” (salinity 26-30 PSU) whose major inputs of fresh water and nutrients are more dominated by treated wastewater releases (Nichols et al., 1986; Caffrey, 1995). In the North Bay, salinity ranges from 0- 30 PSU. Although there are some nutrient inputs in treated wastewater (for example the Sacramento Regional Wastewater Treatment Facility) that can be important during lower flow summer and autumn months, the majority of nutrients loads and fresh water are delivered to the North Bay during the winter from the Sacramento and San Joaquin rivers, which converge in the Delta (Caffrey, 1995).

Suisun Bay is made up of brackish water embayments and is inhabited by less than 10 permanent macrobenthic species, and because the region is inundated each winter by freshwater it is considered a mesohaline community (Nichols and Pamatmat, 1988; Schaeffer et al, 2007). Species that survive here include mollusks *Corbicula amurensis*, which dominates, with *Macoma balthica, Mya arenaria*, and *Corbicula fluminea* - though only when the river inflow is particularly high, lowering salinity levels (Nichols and Patatmat, 1988; Schaeffer et al., 2007); the amphipods *Corophium stimsoni*, and *C. spinicorne*; and the annelids *Nereis succinea* and *Limnodrilus hoffmeisteri*. During periods of low river flow, which leads to increased salinity, the populations of some fauna like the polychaete *Streblospio benedicti* and the amphipod *Ampelisca abdita* are shown to expand upstream towards Suisun Bay. Normally however, these two species can only be found west of the Carquinez strait because of their intolerance to freshwater (Nichols and Patatmat, 1988). Suspension-feeding bivalves are less abundant in the northern San Francisco Bay estuary. Cloern (1982) suggested that this is due to high suspended sediment loads and large seasonal variations in salinity in this area.

West of the Carquinez straight, where salinity rarely dips below 5 psu, the macrobenthic community diversity increases. The macrobenthic community of the broad shallow subtidal expanses of the San Pablo Bay (Figure 5.5.5) includes, in addition to the mollusks found further upstream: *Gemma gemma*, *Musculista senhousia*, *Tapes philippinarum* and *Ilyanassa obsolete*; amphipods *Ampelisca abdita*, *Grandideirella japonica,* and *Corophium spp*. The polychaetes *Steblospio bendicti, Hetermastus filiformis, Glycinde* spp., *Polydora* spp., and several other species of oligochaetes (Thompson and Nichols 1985; Nichols, 1988).

The Central Bay is characterized by stronger currents, deeper waters and a more marine environment. The strong tides create a highly dynamic bottom of large sand waves that reverse directions with each tide. The benthic community is dominated by species that are found in sand sediments along the outer coast, demonstrating a more marine influence. Islands and other rock outcrops in the Bay are inhabited by hard-substrate marine organisms, as well as the "cosmopolitan Bay mussel *Mytilus edulis*" (Thompson and Nichols, 1985; Nichols and Pamatmat, 1988).

In the South SF Bay, several of the same species found in San Pablo Bay occur, but in the subtidal mud areas, the large tube-dwelling polychaetes *Asychis elongata* is common in shallow and deep water (Figure 5.5.5). These maidanid polychaetes have burrows that can reach up to a meter deep in sediments, and can occur in very dense patches in South Bay and can influence structure of the entire community (Nichols, 1988; Thompson and Nichols, 1985; Nichols and Pamatmat, 1988; Schaeffer et al., 2007). In the intertidal and shallow subtidal reaches of South Bay, *Gemma gemma*, *Ampelisca abdita* (Figure 5.5.5), and *Streblospio benedicti* tend to dominate (Thompson and Nichols 1985; Nichols and Pamatmat, 1988; Schaeffer et al., 2007). Many of the species in this assemblage are patchy with some, like *A. abdita*, showing very high abundance one year and low abundance the next year (Schaeffer et al., 2007). This is also where introduced macrofauna tend to be most abundant (Nichols et al., 1990).

Thompson and Lowe (2004) conducted assessments of "benthic condition" between 1994 and 1997 using a multimetric Index of Biotic Integrity (IBI) on two major benthic assemblages in the Estuary: the polyhaline assemblage in the Central Bay and the mesohaline assemblage from the moderate salinity portions of the Estuary (Thompson and Lowe, 2004). The same assessment methods were subsequently applied to samples from San Pablo Bay, Napa and Petaluma rivers, and three sites in the Napa-Sonoma Marsh in 2000–2001. They found that elevated TOC and sediment contamination in those areas had more influence on benthic species composition and abundances than did changes in the hydrodynamic regime (e.g., river or marsh channel), or seasonal and tidal differences in salinity, flow, turbidity, or temperature which supports the idea that anthropogenic nutrient enrichment impacts outweigh environmental co-factors (Thompson and Lowe, 2004; Thompson, et al., 2007).

While Nichols and Pamatmat (1988) largely focused on the ecology of the soft bottom benthos, Schaeffer et al. (2007) also discuss the differences along salinity gradients in areas of the Bay with hard bottom surfaces. In the mesohaline regions hard bottoms surfaces include the large filter-feeding mussel *Mytilus trossulus/gallopgrovincialis* and filter-feeding barnacle *Amphibalanus improvisus* and other attached, non-mobile species such as anemone, sponges, and several suspension feeder species. In polyhaline regions, invertebrates are much more diverse than that seen in mesohaline regions (Schaeffer et al., 2007). In euhaline regions, with higher species richness, amphipods are still a major component of the community, as are omnivorous and carnivorous polychaetes, including two species of scale worms. Pacific rock crab (*Cancer antennarius*) and the red rock crab (*C. productus*) inhabit rocky, intertidal and subtidal areas in the Pacific Ocean, and likely use SF Bay as an extension of their coastal habitats (Schaeffer et al., 2007).

Temporal Variability

Estuarine invertebrates are relatively short lived, however the highest densities are normally observed between spring and autumn, peaking in the summer months, reflecting the high reproductive capability and productivity rates of many benthic species. Abundance decline in winter usually indicates species die off after reproducing and lack of endurance for winter conditions, including lack of food availability in shallow waters. However, because of the temperate climate in the SF Bay some species, such as *Ampelisca abdita, Gemma gemm*a, and *Streblospio benedicti*, can reproduce year round (Nichols and Thompson, 1985).

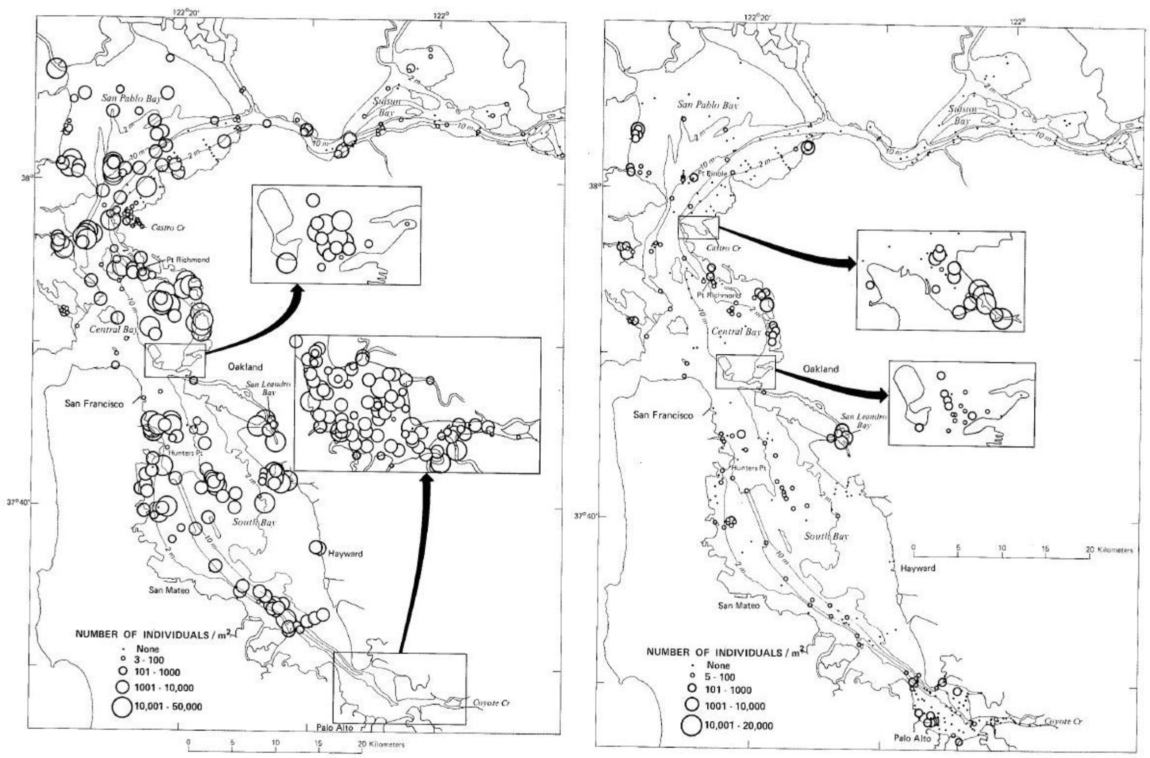


Figure 5.5.5. Distribution and abundance of *Ampelisca abdita* (left) and a *Capitella* polychaete (worm) (right) - an opportunistic species (Nichols, 1988).

Additionally, there is evidence that species interactions can contribute greatly to interannual variations. For example, Nichols and Thompson (1985) observed that Macoma balthica establishes large populations only when the amphipod *A. abdita* is not abundant. Similarly, abundance patterns for the tube-dwelling, surface-deposit feeding polychaete *M. viridis* (peak abundance late spring-early summer, followed by a several month minimum) varies alternately with *Monocorophium alienense*, an amphipod with similar habits and food but which peaks in abundance in late fall/early winter. While Nichols and

Thompsons long term studies have shown that species composition does not change considerably over long periods of times, year-to-year predictability of species abundances is low (Nichols and Thompson, 1985).

Several separate studies of growth of SF Bay benthic invertebrates have shown a strong coincidence in the timing of most rapid growth. Maximum growth in Macoma balthica on a South SF Bay mudflat occurred at the same time as the spring phytoplankton bloom and coincided with the peak in benthic microalgal biomass (Nichols and Thompson, 1985). There is also a strong positive relationship between California bay shrimp annual abundance and freshwater outflow in spring (Schaeffer et al., 2007).

The benthic community in the northern part of the Bay is dominated by *C. amurensis*, which peaks in abundance in fall during the wet and normal water years, and peaks in summer and fall in dry and below normal years. Lowest abundance for this bivalve is in spring or early summer in most years (Schaeffer et al., 2007). The species in the South Bay such as the bivalves *C. amurensis, Mya arenaria, Venerupis japonica, Macoma petalum*, and *Musculista senhousia* show very strong seasonal patterns with decline in abundances to near zero each winter/early spring. The bivalves are therefore mostly annual species in this habitat with peaks in abundance occurring in late spring/early summer. The amphipods (*Corophium heteroceratum* and *Ampelisca abdita*) can show similar annual patterns except during dry years when A. abdita in particular seems to persist through the winter (Schaeffer et al., 2007).

5.5.6 Indicator Trends

Despite the long term scientific and growing management interests in benthic responses to nutrient enhancement, there is considerable debate about the actual impacts of these factors or their potential mechanisms of effect (Posey, 2006). Results from multi-year investigations in the SF Bay estuary show that large inter and intra-annual abundance fluctuations within benthic macroinvertebrate populations, depending on variable predation on adults and on planktonic larvae, variation in the number of larvae in the water column, river inflow from the Sacramento San Joaquin system increasing or decreasing salinity, and changes in the phytoplankton community (Nichols and Pamatmat, 1988; Nichols and Thompson, 1985). After a 10-year study, Thompson et al. (2002) found that the Bay benthic community was composed of non-indigenous, opportunistic species that dominated the community due to their ability to survive the many physical disturbances on the mudflat (Thompson et al., 2002).

Analyses of the benthic community structure over a 28-year period show that changes in the community have occurred concurrent with reduced concentrations of metals in the sediment and in the tissues of the clam *Macoma balthica* (Thompson et al., 2002; Shouse et al., 2003). In addition, two of the opportunistic species (*Ampelisca abdita* and *Streblospio benedicti*) that brood their young and live on the surface of the sediment in tubes have declined in dominance coincident with the decline in metals. *Heteromastus filiformis*, a subsurface polychaete worm that lives in the sediment, consumes sediment and organic particles residing in the sediment has shown an increase in dominance. These changes in species dominance reflect a change in the community from one dominated by surface dwelling, brooding species to one with species with varying life history characteristics, though these trends are directly related to decreases in trace metals, not nutrients (Shouse, 2002).

Multi-year studies reveal that year-to-year variations in species abundances are much larger than would be expected from repetitive cycling of environmental conditions, and have more to do with larger scale shifts in weather patterns, disturbances due to storms and droughts, and anthropogenic influences (Nichols and Thompson, 1985; Cloern et al., 2010). For example, as the PDO has changed signs since 1999, we have seen an uptick in populations of shrimp and crab in the SF Bay. The mean annual catch per unit effort (CPUE) between 1980-2008 for Cancer crabs and caridean shrimp was calculated and reflected 3–6 fold increases. Population increases of these predators after 1999 were followed by population declines of bivalve suspension feeders and increasing phytoplankton biomass that persisted from 2000 through 2008. A strong argument has been put forward that these observations may indicate a dramatic restructuring of biological communities in SF Bay in relation to larger climate driven phenomenon at the decadal scale (Cloern et al., 2010).

5.5.7 Utility of Indicator as an Eutrophication Indicator for San Francisco Bay

Relationships to Beneficial Uses

Macrobenthos play a critical role in the biotic and abiotic functioning of the estuary; thus a diverse, fully functional macrobenthic community is an essential part of maintaining ecosystem services and related estuarine beneficial uses. The State of California has designated six "Estuarine Beneficial Uses" upon which to evaluate the estuarine natural resources (structure) and ecosystem services (function) (Chapter 2). These beneficial uses broadly address biodiversity and threatened/endangered species (rare [RARE], spawning [SPWN], and migratory [MIGR] uses), commercially valuable resources (commercial [COMM], shellfish [SHELL], and aquaculture [AQUA] uses), and the inherent value estuarine habitat for aquatic life (estuarine [EST] and wildlife [WILD] uses). The structure and function of the macrobenthic community encompass: 1) their contribution to estuarine and marine biodiversity; 2.) direct recreational and fisheries harvest; 3) a food resource for a variety of estuarine aquatic life forms, including fish, birds, marine mammals; 4) a critical role in the maintenance of water column and sediment biogeochemical cycling; and 5) the consumption of a variety of organic matter sources and subsequent regeneration of nutrients to the water column.

From the estuarine beneficial use perspective, macrobenthos are part of diversity of aquatic life and as such a direct measure of EST beneficial uses. The State of California has recognized the intrinsic value of macrobenthos and as such, is currently developing a biocriteria program that includes macrobenthos as a primary indicator of aquatic life in streams (J. Bishop, SWRCB, Pers. Comm.). Development of macrobenthic-based assessment tools for California's estuaries will provide the State the same opportunity to establish biocriteria in estuaries.

In terms of commercial value, many species of macrobenthos are directly harvested (e.g., oysters, mussels, clams, shrimp, and lobsters) by humans, which would be classified as COMM and SHELL uses. Within California, commercial shellfish harvest represented approximately $100 million in fisheries landings in 2008 (NMFS pers. comm.), in addition to the creation of jobs and revenue related to harbor infrastructure, seafood processing and distribution, and tourism. The harvest of macrobenthos also provides recreational value. Beyond their direct commercial value, the macrobenthos provide an important source of food for estuarine and marine fish, birds and marine mammals (EST), including migratory fish and marine mammals (MIGR), spawning fish (SPAWN), and threatened/endangered species of fish and birds (RARE). Numerous commercially important nekton (e.g., *Embiotaca jacksoni*, *Umbrina rancador,* or *Hypsopetta guttulata*) from California's estuaries are dependent upon the macrobenthos as a food source and thus provide indirect support for COMM beneficial uses (Allen et al., 2006).

The macrobenthos play a key role in sediment nutrient and contaminant cycling through bioturbation and bioirrigation (the mixing of sediment and advective exchange of sediment pore waters with surface waters) and thus are a key component of maintenance of good estuarine and marine habitat and water quality (EST and MAR). Active burrowing and the building of tubes or galleries in the sediment increases the penetration of oxygen into the sediment and the surface area of oxic/anoxic sediment horizons, which can enhance coupled nitrification/denitrification and ultimately remove nitrogen from the estuary (Aller, 1982; Mayer et al., 1995; Aller and Aller, 1998). As infauna ventilate their burrows and tubes, there is an increase in the flux of pore water through the sediment and the exchange of porewater with overlying waters, which will carry dissolved nutrients and organic matter with it (Michaud et al., 2005, 2006). In estuarine systems where the mixed layer extends to the bottom, filter-feeding benthos will enhance benthic-pelagic coupling by collecting water column production and depositing waste products at or below the sediment surface (Graf, 1992; Gerritsen et al., 1994; Thompson and Schaffner, 2001). Macrobenthos have been shown to be a major control in both the North and South Bay (accepting light limitation and temperature) on phytoplankton populations. Analogously, head-down deposit-feeders (e.g. *Asychis elongata*) feed on bacteria and organic matter centimeters below the sediment surface and depositing waste at the surface, which exposes and recycles organic matter back to the water column (Lopez and Levinton, 1987; Clough and Lopez, 1993; Levin et al., 1997). Autumn (pico) phytoplankton blooms including HABs occur once the peak biomass of water column filter feeders passes in the summer and regenerated nutrients from decaying detritus are able to feed the planktonic foodweb in the later summer and autumn months. Therefore, macrobenthos play an important role in processing organic matter, recycling nutrients, and sequestering contaminants, all of which support healthy estuarine and marine habitat.

Finally, from the biotic, food web perspective, a healthy, well-developed macrobenthic community consists of a diverse array of trophic levels and feeding guilds that utilize the variety of organic matter produced or deposited in the shallow waters of estuaries (e.g., Diaz and Schaffner, 1990; Fauchald and Jumars, 1979; Gaudênci and Cabral, 2007). Much of this production though (e.g., microphytobenthos, bacteria/detritus, phytoplankton) is not directly available to these transient fauna. Macrofauna however, can directly consume most types of bacterial or primary production and via their own somatic growth, accumulate the energy and material in a form that can be consumed by fish or birds (Levin, 1984; Iwamatsu et al., 2007; Neuman et al., 2008). In this respect, the macrobenthos serve as a conduit for the transfer of carbon from bacterial and primary production to higher trophic levels in estuaries, most of which cannot directly consume all of these types of organic matter (Gillett, 2010). Thus macrobenthos play a key role in transfer of energy and carbon to higher trophic levels, a key ecosystem function.

Predictive Relationships with Causal Factors

Water residence time and flushing in estuaries, shallowness and salinity of the estuary, and food availability are key drivers of the relationship between benthic communiy composition and nutrient enrichment. Though retention of nutrients in estuaries is positively correlated with residence time of the water mass, the underlying mechanisms are not well understood and it is conceivable that differential effects of limiting factors other than food may obscure a relationship between nutrient load and benthic biomass production (Martinetto et al., 2006; Josefson and Rasmussen, 2000; Heip et al., 1996). In the North Bay, benthic macroinvertebrate biomass is dominated by C. amurensis and peaks in abundance in fall during the wet and normal water years, and peaks in summer and fall in dry and below normal years. Lowest abundance for this bivalve is in spring or early summer in most years. In the North Bay, where greater than 50% of the carbon budget is allochthonous, there is a weaker relationship between phytoplankton biomass and production rates of secondary macroinvertebrate biomass. In contrast, in the South Bay were the majority of carbon production in autochthonous, benthic biomass is tightly coupled with phytoplankton blooms in the spring; the bivalves C*. amurensis, Mya arenaria, Venerupis japonica, Macoma petalum,* and *Musculista senhousia* show very strong seasonal patterns with decline in abundances to near zero each winter/early spring and peaks in abundance in late spring/early summer. The amphipods (*Corophium heteroceratum* and *Ampelisca abdita*) can show similar annual patterns except during dry years when *A. abdita* can persist through the winter.

This synthesis suggests that accounting for benthic biomass is a critical co-factor in modeling the relationship between nutrient loads and phytoplankton productivity. Development of a model that predicts benthic taxonomic composition as a function of nutrient loads and other co-factors is complicated. It remains difficult to identify a benthic response to eutrophication when contamination commonly covaries with many of these other environmental factors (Nichols, 1979).

Scientifically Sound and Practical Measurement Process

Macrobenthos are relatively easy to quantitatively sample, especially in soft sediments. Samples of sediment and macrobenthos can be collected with a variety of grabs (e.g., Smith-MacIntyre grab, Van Veen grab, Young grab) or cores (e.g., box cores, push cores, or vibrating cores) that can be deployed from various sized vessels, by divers, or by wading in shallow water (see review in Holme and McIntyre ,1984). Once collected, organisms can be separated from the sediment using sieves with a variety of mesh sizes. Macrobenthic fauna are typically collected with a 500-µm sieve, though larger sized meshes can be used to simplify sample processing or to establish size-spectra within the community, while smaller sized meshes are used to sample juvenile macrobenthic fauna (e.g., Edgar, 1990). The selection of sampling gear and sieve size is an important consideration, as they will both influence the characterization of the macrobenthic community. Different gear types sample to different depths in the sediment and larger sample areas will have a greater likelihood of collecting rarer taxa. Different sieve sizes will retain or exclude different size classes of organisms, which will influence abundance and biomass measurements – especially for small fauna like oligochaetes and polychaetes (Gillett et al., 2005). Most macrobenthic monitoring programs in California have refined their protocols to using a Van Veen grab for sample collection and a 1-mm mesh sieve for sample processing to balance community characterization and ease of sample processing (Smith et al., 2001; Bay et al., 2009; Ranasinghe et al. 2009).Standardized protocols for sampling of taxonomic composition and abundance are currently part of the SWRCB 's sediment quality objective protocol (www.swrcb.ca.gov/water_issues/programs/bptcp/sediment.shtml).

It should be noted that these are applicable in shoal areas, and channel edges, however, the difficulty lies in the ability to sample the benthos in open water. In the SF Bay, that there is no recent comprehensive study of species composition and distribution is a major limitation and data gap. Long term studies in specific areas by Thompson and others are helpful, but frequent repetitions of Nichols 1986 survey is imperative in order to develop and maintain an accurate and useful dataset.

Acceptable Signal to Noise Ratio

There is a large amount of information available on the effects of eutrophication on the macrobenthic community. The major impediment to the development of community-based indicators specifically in estuaries is partially due to the variable nature of the estuarine environment and the physiological stress this places upon endemic estuarine fauna (Dauvin, 2007; Dauvin and Ruellet, 2009). The estuary represents an ecotone between the marine and freshwater systems and the fauna that inhabit this area are a mix of organisms invading (at geologic time scales) landward from the coastal ocean and seaward from riverine systems (Attrill and Rundle, 2002). The osmotic stresses of fluctuating salinity, the physical stress of tidal erosion/deposition of surface sediments, and other natural stressors act in concert to select for fauna that are relatively predisposed to be tolerant of environmental stressors, which may make them better adapted to deal with eutrophic stressors than fauna from more stable marine or freshwater systems. This problem has been referred to as “the estuarine quality paradox” (Elliot and Quinto, 2007); where the paradox is how to define or detect anthropogenic reductions in habitat quality on a community that is adapted to deal with changing physical conditions and high rates of primary production naturally occurring in estuaries (Dauvin, 2007; Dauvin and Ruellet, 2009). This problem is even further complicated when looking at eutrophic impacts, particularly at the beginning of the eutrophication process where impacts maybe more subtle. Though this paradox makes it a challenge to use macrobenthos as an indicator of eutrophication in estuaries, it is not impossible given the use of the macrobenthos as a monitoring tool in estuaries around the world (Diaz et al., 2004). If the community characteristics that are chosen to be used as indicators are sensitive/or unique only to eutrophication and if the choice of reference condition(s) incorporates the environmental variation of the estuarine ecosystem by stratifying sampling and assessment tools by environmental gradients (e.g., Weisberg et al., 1997; Llansó et al., 2002) then the problems associated with the estuarine quality paradox can be reduced.

There are a number of reasonable conceptual models and experimental data to describe how eutrophication alters the composition and functioning of macrobenthic communities. Using the macrobenthic community for detection and quantification of eutrophic conditions in estuarine systems is complicated however, because most water bodies that experience eutrophication are also subject to a variety of other stressors (e.g., chemical contamination or physical disturbances) that have been shown to effect macrobenthic community structure (USEPA, 2008). There is a wide array of different chemicals that that accumulate in estuarine sediments, including organic compounds, heavy metals, pesticide, pharmaceuticals (Sanger et al., 1999a,b; Kennish, 2002). Many of the these chemicals can have toxic effects on the macrobenthos at the community level, reducing the number of sensitive taxa and overall community species richness, but without the potentially positive effects that the extra organic matter from eutrophication can create (Peterson et al., 1996; Gaston et al., 1998; Dauvin, 2008). Additionally, the effects of many contaminants are taxonomically specific (organotins and gastropods, pesticides and crustaceans, or metals and annelids) (Rand et al., 2000; Valiela, 1995). Because of the diversity of chemical contaminants in estuarine sediments, there are not any generalized models of contaminant-driven changes in community structure like the Pearson-Rosenberg (1978) model. An overall loss in community diversity and disproportionate mortality among sensitive taxa with increasing chemical contamination should be expected (Peterson et al., 1996; Rakocinski et al., 1997; Gaston et al., 1998), but the impact on community abundance and biomass is unknown. That said, abundance and biomass should not increase, as they can with non-hypoxic eutrophication.

As alluded to earlier, physical disturbance of the benthic habitats, either natural (large storms or ice scour) or anthropogenic (dredging or benthic trawling), can have important influences on the structure of the macrobenthic community. These types of disturbance can defaunate a habitat and the recovery of the community will occur in a relatively predictable fashion through time (Rhoads and Boyer, 1982; Rhoads and Germano, 1986). At the beginning of the successional process, many of the same species that are pioneering, opportunistic organisms are also resistant to the stressors of eutrophication and the accumulation of organic matter in sediments (Pearson and Rosenberg, 1978; Rhoads and Boyer, 1982; Gray et al., 2002). The model of macrobenthic community succession of Rhoads and Boyer (1982) is conceptually almost the mirror image of Pearson and Rosenberg's (1978) organic enrichment model. Physical disturbances severe enough to "restart" the successional process in estuarine systems are, however, much more stochastic than eutrophic stressors, which are persistent, systemic problems that even when corrected in the water column, have a legacy of organic matter in the sediments that will continue to negatively affect benthic fauna for a 5-10 years (e.g., Rosenberg, 1976; Borja et al., 2006; Tett et al., 2007; Diaz et al., 2008). In contrast, recovery time from large-scale physical habitat disturbance like dredge-material disposal occurs over 2-3 years, with detectable changes in community structure in the short-term (e.g., Zajac and Whitlach, 1982b; Wilbur et al., 2008; Schaffner 2010). Given these temporal differences, year-to-year comparisons of community data should allow for the separation of physical stress (significant year-to-year change in structure) and eutrophic stress (less year-to-year change) on the macrobenthic community. Additionally, like the chemical stressors, physical stressors should not have the positive biomass/production benefits to the macrobenthic community that accompany eutrophication of an ecosystem.

5.5.8 Summary: Use of Macrobenthos for San Francisco Bay NNE

Overall, macrobenthos appear to satisfy three of four review criteria. Macrobenthic taxonomic composition, abundance and biomass have the potential to be used as a supporting indicator in a NNE assessment framework in enclosed bays and estuaries with salinities of > 10 psu. Using only singular aspects of macrobenthic community structure (i.e., taxonomy, abundance, or biomass) will likely not be a robust method to assess eutrophication or, more generally, the trophic-state of an estuary. Though simpler metrics should be tested as well, it is most likely that a combination of all three aspects of macrobenthic community structure will prove to yield the best assessment tool. Measures of mean per capita biomass (community biomass ÷ community abundance), relative biomass distribution among different taxonomic or ecological groups, or the species-specific abundance in different size classes of organisms are slightly more complex measures than total abundance or a species list, but they also have the potential to capture more subtle changes in community structure brought upon by eutrophic stressors.

The review criterion not well satisfied is the ability to establish a predictive model between macrobenthic taxonomic composition, abundance, biomass and nutrient loads. Science is evolving in this area, and thus predictive capacity may be possible in the future. However, available data and our understanding of factors driving macrobenthos in SF Bay are lacking. Thus while macrobenthos may be a useful supporting indicator or co-factor in shallow muddy, subtidal habitats of SF Bay, it is not likely to be a primary NNE indicator in the near term. Because macrobenthic taxonomic composition and abundance are already standardized components of the State's ambient monitoring program, it merits considering what key data gaps and next steps would be necessary to use macrobenthos as an additional supporting line of evidence to diagnose eutrophication.

Several key data gaps exist and a number of steps will be required in order to determine the ultimate utility of macrobenthos in this capacity. First, we recommend assembling a workgroup to identify potential species or metrics based on taxonomic composition and abundance, then use existing data collected through EPA Environmental Monitoring and Assessment Program (EMAP) and regional monitoring programs to test out the utility of these metrics as a tool for eutrophication. Second, though it is not currently collected in California's existing, state-wide monitoring programs, biomass data (or a reasonable approximation thereof) are probably going to have to be collected to successfully distinguish eutrophication from other stressors. We recommend a small pilot project, which would include the collection of macrobenthic biomass in a new ambient monitoring framework, in order to test out the applicability of these kinds of data to detect eutrophication. Finally, looking at spatial variation will also likely be necessary to separate different stressors: where measures of poor/impacted community structure with less spatial variation would be indicative of eutrophic stress on the community, compared to those with larger spatial variation, which would be indicative of other types of physical disturbance or successional changes in the community. Different combinations of metrics or differential thresholds will likely have to be implemented to tailor any assessment tool to the different sediment, salinity, or flow regimes within and among California's varied estuarine systems.

5.6 Jellyfish

The term 'jellyfish' refers to free-floating gelatinous animals (Figure 5.6.1) belonging to the phylum *Cnidaria* (hydromedusae, siphonophores and scyphomedusae) and to planktonic members of the phylum *Ctenophora* (Mills, 2001; Richardson, 2010). Although many *Cnidaria* are able to actively swim by contracting the muscles of their bells and *Ctenophora* are able to propel themselves by the sequential beating of cilia, neither can swim against currents and are therefore defined as zooplankton (Bushek, 2005). These organisms share many characteristics including their watery or 'gelatinous' nature, and a role as higher-order carnivores in plankton communities. They are often referred to as gelatinous zooplankton (Mills, 2001; Lucas, 2001; Pitt, 2007).

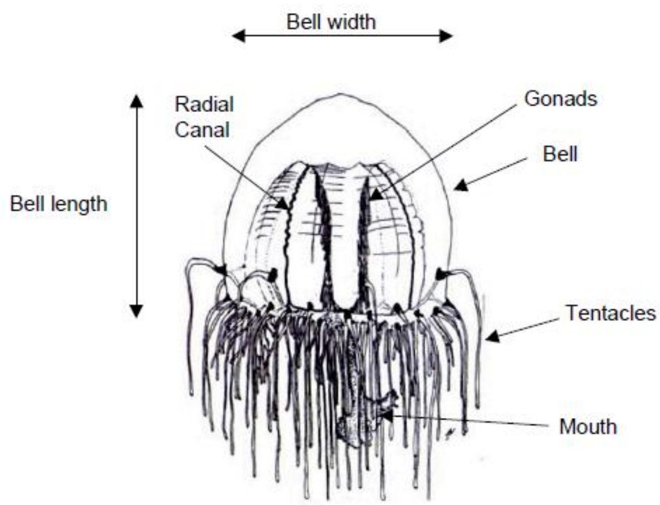


Figure 5.6.1. Main body parts of a jellyfish (Herdon et al., 2003).

The life cycle of the jellyfish is biphasic; an alternation between a small, often unseen benthic polyp and a free-floating/swimming, open-water medusa. The polyp produces medusae by asexual budding, whereas the medusa reproduces sexually (Bushek, 2005; Lucas, 2001; Pitt, 2009; Purcell, 2005). The asexual reproduction process is seasonal in temperate climates and varies between species, with the period of medusa budding varying from days to months long, but nearly always resulting in seasonal appearance and disappearance of medusa populations (Mills, 2001). This alternation of life stages means jellyfish can be present even when they are not obvious to the casual observer and more information is known about the medusa stage than the polyp stage (Wintzer, 2010; Bushek, 2005). The ability of these gelatinous species of cnidarians (*Scyphozoa, Cubozoa, Hydrozoa*) to occur in large numbers, or, to bloom, is due to having both asexual and sexual reproduction (Purcell, 2007). The life history of ctenophores does not include an asexual benthic stage. Larval ctenophores grow and bloom directly into adults without passing through the polyp stage. Perhaps because of their fragility, little is known about factors that affect their sexual reproduction and population abundances (Purcell, 2005).

Most populations are found in small shallow coastal embayments, fjords, and estuaries where there are suitable substrates for the benthic polyp to attach, and limited tidal exchange, with variable productivity, and food availability (Lucas, 2001). Most gelatinous zooplankton are suspension-feeders and respond quickly to changes in food availability by exploiting their high feeding and growth rate potentials; during starvation periods of up to 25 days, they survive by shrinking, instead of dying (Moller, 2007).

The life cycle of a jellyfish is normally less than one year, yet gelatinous zooplankton come and go seasonally (seasons of reproduction vary greatly by species), appearing and disappearing with annual regularity (Mills, 2001). However, medusa and ctenophore populations also undergo interannual variation, some years blooming with much greater intensity and much larger populations of each species than others (Mills, 2001).

5.6.1 Applicable Habitat Types

As an indicator, jellyfish would be applicable to the subtidal habitats of the North, Central and South Bays.

5.6.2 Indicator Relationship to Nutrients and Water Quality

The input of excessive nutrients from fertilizer runoff, sewage and other anthropogenic sources into estuaries has been shown to greatly alter pelagic communities (Richardson, 2010; Pitt, 2007; Mills, 2001). Nutrient enrichment stimulates primary production, increasing the biomass of phytoplankton. Feeding rates of grazers then increase, which stimulates secondary and, potentially, higher-order production (Pitt, 2007). As more food becomes available, polyps and jellyfish increase asexual production and sexual reproduction, allowing populations to "bloom" (Pitt, 2007; Purcell, 2001; Bushek, 2005).

Jellyfish populations are characterized by large and rapid fluctuations in abundances and "boom and bust population dynamics" and thus often represent a substantial proportion of the pelagic consumer biomass (Condon, 2010; Pitt, 2009). During bloom formation, when both individuals and populations are increasing in size, jellyfish and ctenophores act as a net sink for C, N and P, rapidly assimilating carbon and nutrients from their planktonic prey (Pitt, 2009). Because of their high biomass during blooms, gelatinous zooplankton can influence nutrient cycling as they both excrete and take up dissolved organic matter, inorganic nitrogen and phosphorus (Condon, 2010; Pitt, 2007; Welsh, 2009). When in high abundances, gelatinous species may contribute significantly to nitrogen and phosphate budgets (Pitt 2007; West, 2009).

Eutrophication and Hypoxia

Large phytoplankton blooms, as discussed above, and dead jellyfish resulting from nutrient enrichment can sometimes sink to the seafloor, where their bacterial degradation can cause localized hypoxia (Richardson, 2010; West, 2009). Polyps and medusae are more tolerant to lower oxygen conditions than fish, which often ensure jellyfish survival over fish during hypoxic events (Richardson, 2010). Fish avoid, or die in, waters of ≤2–3 mg O_2 L^{-1} but many jellyfish species are tolerant of ≤1 mg O_2 L^{-1} (Pitt, 2007). However, though tolerant of low dissolved oxygen at adult stages, several species can, in fact, be intolerant at polyp stage (Purcell, 2001; Wintzer, 2010).

Eutrophication has been suggested to be an important environmental factor for increasing mass occurrence of jellyfish (Mills, 2001). As jellyfish are tolerant to low dissolved oxygen concentrations (Pitt, 2007; Purcell, 2001), they can take over oxygen depleted waters previously inhabited by zooplanktivorous fish. For example, "Skive Fjord (Denmark) suffers every summer from oxygen depletion in the near-bottom water causing large amounts of nutrients (phosphate and ammonium) to be released from the anoxic sediment. This subsequently stimulates a phytoplankton bloom, followed later on by an increase in the zooplankton. The surface chlorophyll a concentrations may become very high during periods with exceptionally severe oxygen depletion. In certain years when biomass occurrence of *Aurelia aurita* in high, peak chlorophyll a concentrations as high as 60 to 80 µg L^{-1} have been measured in Skive Fjord because the jellyfish effectively eliminate the zooplankton-grazing impact on the phytoplankton bloom (Moller, 2007). It seems that jellyfish may benefit from eutrophication, which can increase small-zooplankton abundance, turbidity and hypoxia, all conditions that favor jellyfish over fish. Overfishing can also remove predators of jellyfish and zooplanktivorous fish competitors as well as cause large-scale ecosystem changes that improve conditions for jellyfish (Purcell, 2007).

On a micro-scale, the presence of individual jellyfish was shown to have an influence on benthic oxygen and nutrient dynamics by researchers at the Australian Rivers Institute (Welsh, 2009). Sediment patches occupied by individual jellyfish showed dramatically different dynamics than adjacent unoccupied or "bare" sediments, and also varied temporally. For example, during the night the presence of a *Cassiopea spp*. individual enhanced benthic respiration by 3.6-fold and benthic ammonium regeneration rates by 4.5-fold. "However, during the high light period, photosynthetic oxygen production by the jellyfish increased benthic oxygen production by almost 100-fold and although the sediment alone was net source of ammonium to the water column, ammonium assimilation by the jellyfish reversed this flux creating a benthic sink for water column ammonium (Welsh, 2009)."

Another relationship change which favors jellyfish is the decline, in the Chesapeake Bay for example, of oyster populations. Phytoplankton that would have been consumed by oysters now is available to zooplanktivores such as medusae, increasing their populations, leading to the hypothesis that if oysters were restored to their former abundance in Chesapeake Bay there would be a reduction in gelatinous zooplankton (Purcell, 2001).

Impact on Pelagic and Benthic Communities

In addition to eutrophic conditions, increased nutrients from sewage effluents or fertilizers in estuarine environments may change plankton food webs towards small phytoplankton (or microplankton) and zooplankton species causing a trophic cascade. This size reduction to lower trophic levels is considered to favor gelatinous zooplankton which are non-visual and consume small and large size of prey, over fish (Purcell, 2001, 2007; Pitt, 2007; Mills, 2001). Predators such as *A. aurita* may also affect fish standing stocks, either directly by predation on fish larvae, or indirectly by competing with fish larvae for available food resources (Lucas, 2001). Studies in the Baltic Sea of populations of A. aurita show considerable variability, but in peak densities, can consume more than 60% of the daily production of copepods and other zooplankton (Bushek, 2005). Similarly, a tendency of algal blooms related to high abundances of jellyfish was noticed by researchers in a Canadian fjord due to reduced herbivore grazing caused by a high predation impacts by medusa on zooplankton (Moller, 2007).

5.6.3 Species Composition in San Francisco Bay

San Francisco Bay has 20 native jellyfish species and four introduced, based on year round studies at 52 sites in the estuary, conducted by the California Department of Fish and Game's San Francisco Bay Study in 2000 (Herndon et al., 2003). Native species tended to be found in high salinity and cooler waters. Introduced species have been primarily found in Suisun Bay and the Sacramento San Joaquin delta where there is lower salinity, and higher water temperature in the summer (Herndon et al., 2003). Of the most common species found in the Bay, the most abundant species, both native and non-native are discussed in this section (Figure 5.6.2 and 5.6.3). Jellyfishes of all sizes capture food from suspension as they drift and swim in the water column (Bushek, 2005). However, Rees and Kitting (2002) noted that *M. marginata* were able to kill juvenile fishes in laboratory experiments, and Schroeter (unpublished data) found goby larvae in 6 out of 39 medusae collected in July 2004 from Suisun Marsh (May, 2006).

Invasive jellyfish have been documented to have severe effects on the ecosystems they invade because many species are voracious predators, consuming large amounts of prey and disrupting planktivorous food webs more than native hydrozoan species (Mills and Sommer, 1995). Preliminary information on diet of invasive hydromedusae in the SF Bay shows that they feed on a wide variety of planktonic species, including larval fishes (Mills and Sommer, 1995, R. E. Schroeter, unpublished data).

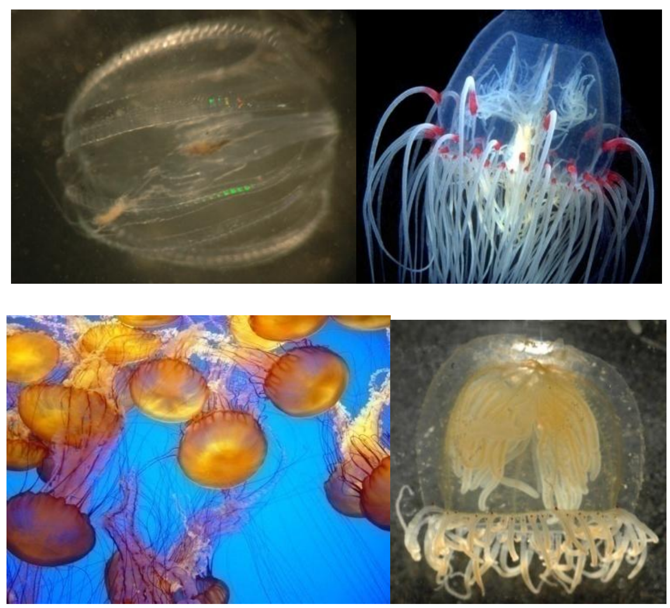


Figure 5.6.2. Native species: Starting from top left clockwise: *Pleurobrachia bachei*, "Sea Gooseberry" or "Comb Jelly" (photo: Dave Cowles 2007), *Scrippsia pacifica* (photo: Garry McCarthy, 2000), *Polyorchis penicillatus* (photo: Dave Cowles 2006), *Chrysaora fuscescens* or "Sea Nettle" (photo: Monterey Bay Aquarium).

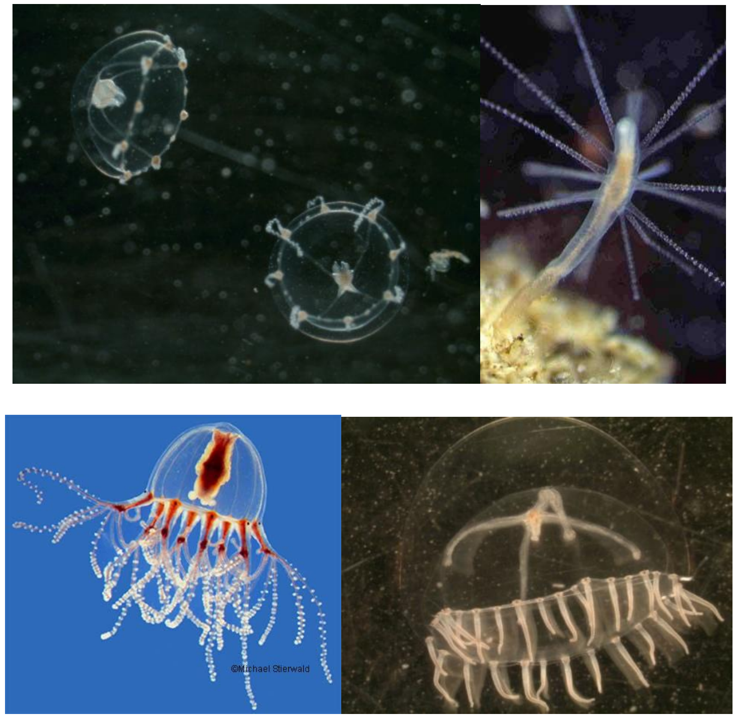


Figure 5.6.3. Introduced species: Starting top left clockwise: *Blackfordia virginica* (photo: Meek, UC Davis, 2010), *Cordylophora caspia* (photo: USGS invasive aquatic species), *Maeotias margina* (photo: USGS invasive aquatic species), *Moerisia sp.* (photo: Meek, UC Davis, 2010).

5.6.4 Indicator Trends and Factors Effecting Temporal and Spatial Variation

Climate and micro-climates, seasonal variation, water quality changes from land uses, the Pacific decadal oscillations, and other oceanic shifts such as upwelling are factors which effect the temporal and spatial variation of jellyfish populations. Furthermore, substrate composition, temperature and salinity levels impact communities on a shorter time scale. A study on spatial and temporal variation and abundance looking specifically at the polyp stage of the four non-natives species demonstrated that water quality factors are strongly correlated with the majority of variation of seasonal observations (Wintzer, 2010).

Spatial Variability

Polyorchis penicillatus has been found as far upstream as Suisun Bay and as far south as the Dumbarton Bridge, with the highest concentration in Central Bay (Figure 5.6.4) (Herndon et al., 2003). Pleurobrachia bachei has been found as far upstream as Suisun Bay and as far south as the Dumbarton Bridge, with the highest concentration in South Bay (Figure 5.6.4) and in a study of macroplankton species composition in the SF Bay between 1997 and 2000 was found to be the most abundance macroplanktonic species living in the estuary (Herndon et al., 2003; Gewant, 2005). *Maeotias marginata* is an introduced species, most commonly found in Suisun Bay and West Delta areas (Figure 5.6.4). *Scrippsia pacifica* is a native jellyfish with a distribution ranging from the southern half of San Pablo Bay to most of South Bay (Figure 5.6.4). *Chrysaora fuscescens* or "Sea Nettle" is a native species of jellyfish found in the southern part of San Pablo Bay and Central Bay (Figure 5.6.4) (Herndon et al., 2003).

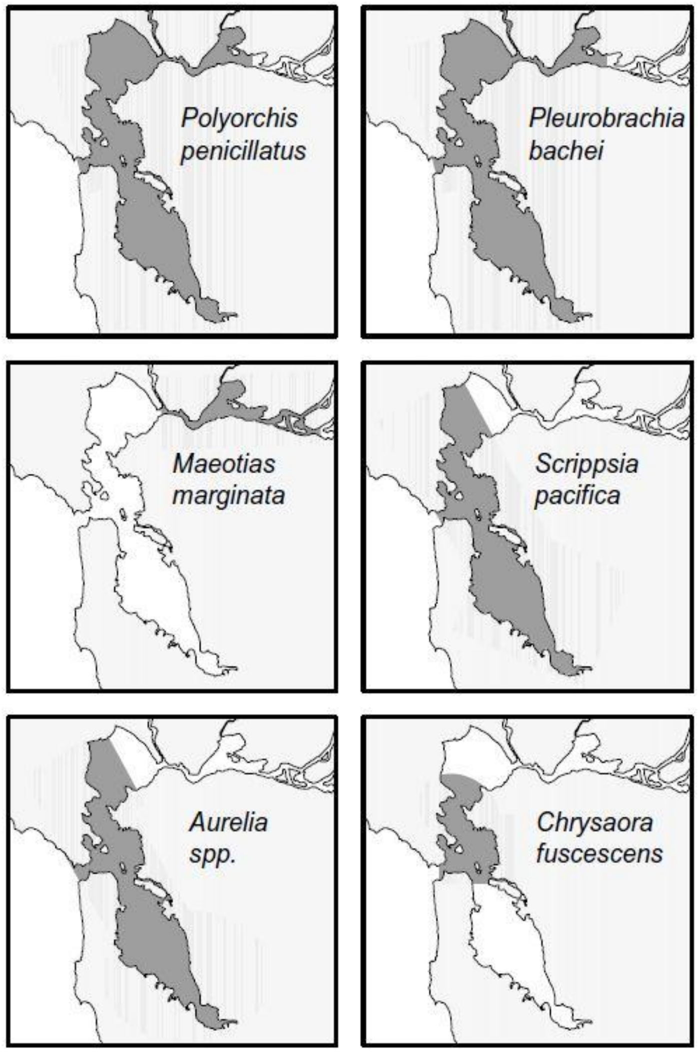


Figure 5.6.4. Maps of common jellyfish species found in the San Francisco Bay (Herndon et al., 2003).

Researchers at the Romberg Tiburon Center trolled the Bay monthly between September 1997 and December 2000 at six stations spanning the North, South and Central Bays. They found that macrozooplankton and micronekton communities were dominated by four fishes and seven invertebrates which comprised 98% of the total catch (Gewant, 2005). Polyorchis penicillatus, was the most commonly occurring, specifically in Central Bay (station 17), but reached maximum densities (70 individuals/ 1,000 m^3) in the North Bay in the winter of 1998-1999 (station 13 and 15) (Figure 5.6.5) (Gewant, 2005).

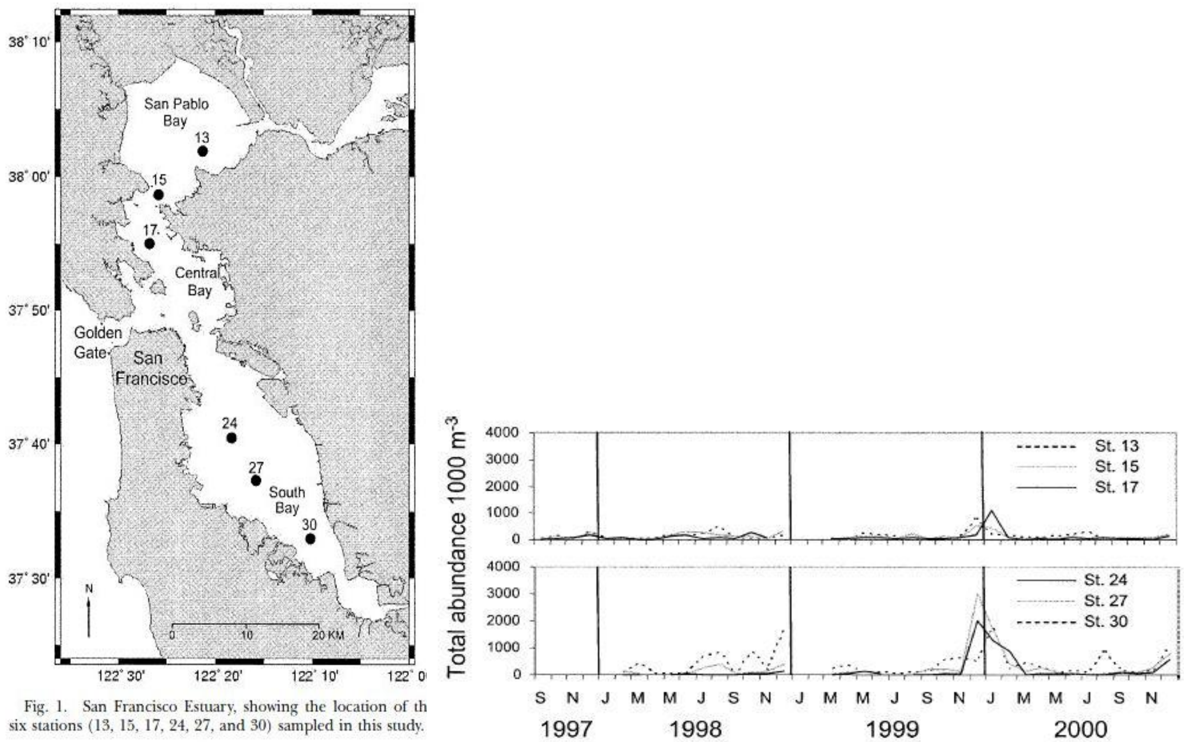


Figure 5.6.5. San Francisco Bay showing locations of the six sample stations, and abundance of macrozooplankton and micronekton in the San Francisco Bay (1997-2000) (Gewant, 2005).

At a more localized scale, individual jellyfish may represent local sources/sinks for oxygen and nutrients their distributions may enhance spatial heterogeneity of the benthos at the small (adjacent patches of sediment with and without a jellyfish) and medium (nearby areas of sediment colonized or not colonized by jellyfish) scales (Welsh, 2009).

Temporal Variability

Polyorchis penicillatus is a native species of jellyfish, found year-round, but with increased abundance during the winter months, perhaps due to their preference for lower salinity levels exhibited by native species, specifically November and December although largely absent during the winter of 1998– 1999 according to Gewant et al. (Herndon et al, 2003, Gewant, 2005). Pleurobrachia bachei, “Sea Gooseberry” or “Comb Jelly” is a native ctenophore that only occurs only in winter and spring. Maeotias marginata, an introduced species, is highly seasonal, found only during summer and fall (Herndon et al., 2003).

In Gewant’s study, *P. bachei* and *Polyorchis spp.,* showed very strong seasonal occurrence patterns. They were captured exclusively in the late fall and winter months, during periods of high salinity, and then disappearing by early spring (Gewant, 2005). However, questions remain as to whether these large numbers were advected into the Bay by wind or currents or if they bloomed within the Bay from a resident seed population. The appearance of peak abundances in North and South Bays, but not Central Bay, which is most proximate to the coastal ocean, is particularly noteworthy (Gewant, 2005).

In another study, researchers at UC Davis found that the polyp phases of *Moerisia sp*. and *B. virginica* followed the common recruitment pattern in benthic hydroids in temperate waters, showing a peak in productivity during spring and summer, which slows down in fall and winter (Wintzer, 2010).

Invasive Species

The four invasive hydrozoans, *Maeotias marginata, Blackfordia virginica, Moerisia* sp., and *Cordylophora caspia*, have become established in the brackish waters of the SF Bay (Figure 5.6.6), where they reach seasonally high abundances during medusae blooms (June-November) (Mills and Sommer 1995; Mills and Rees, 2000). In Suisun Marsh, R. E. Schroeter (UC Davis) recorded Moerisia densities of more than 500 individuals per m^3 (R. E. Schroeter, unpublished data) and "tens of thousands" of *M. marginata* have been collected in the Napa River during July 2003 surveys alone (May, 2006).

Indicator Trends

Many jellyfish populations appear to be increasing around the world, most likely in response to human-induced alterations of the oceanic environment, such as global warming, eutrophication, and over-harvesting of fish stocks (Mills, 2001; Wintzer, 2010). Analyses of several long-term (8- to 100-year) trends in jellyfish populations demonstrate that their abundances vary with climate, often at decadal scales (reviewed in Purcell 2005). Some evidence suggests continued upward trends; however, recent time series are still too short to exclude decadal climate cycles (Purcell, 2007). Bushek (2005) hypothesizes that climate fluctuations serve as the major source for interannual variation in jellyfish populations, due to the changes in primary production, zooplankton, fishes and seabird abundance in the 20th century (Bushek, 2005). During the summer of 2007 in the Gulf of Mexico, nutrient-rich outflows from the Mississippi River resulted in large phytoplankton blooms and 25 000 km^2 of oxygen-depleted waters, favoring jellyfish because of their tolerance for low dissolved oxygen as compared to commercially valuable fish and shellfish (Richardson, 2010). Finally, warming of the oceans may increase many populations of gelatinous species and also shift the population distributions poleward, as seems to be occurring for the ctenophore *Mnemiopsis leidyi* (Purcell, 2007). Warming of the sea surface can enhance water column stratification, leading to nutrient-poor surface waters where flagellates, because of their ability to migrate vertically into nutrient-rich deeper waters, can out-compete diatoms. Such flagellate-dominated food webs might be more favorable for jellyfish than for fish (Richardson, 2010). Warmer temperatures have also been shown to accelerate medusae growth and reproduction (Purcell, 2007; Richardson, 2010).

Locally, limited data are available on jellyfish populations, though Peter Moyle has tracked the invasion of *Maeotius marginata* in the Suisun Marsh since 1981, showing a growing abundance of the invasive gelatinous zooplankton (Moyle, ppt, 2009). The Pacific Decadal Oscillation (PDO) reversed signals in 1999 and has remained negative (Cloern et al., 2010; Peterson and Schwing, 2003) which has been supported by cooler temperatures, increased salinity, weakened winds, a doubling of zooplankton biomass and a favoring of cool water species over warm (Peterson and Schwing, 2003). These changes also imply a shift to stronger coastal upwelling which would continue to support jellyfish populations, and though Anderson and Piatt (1999) note that jellyfish populations increased with the last positive PDO cycle, there is perhaps, conflicting evidence regarding what impact the negative PDO cycle will have on gelatinous zooplankton in the SF Bay (Cloern et al., 2010; Anderson and Piatt, 1999).

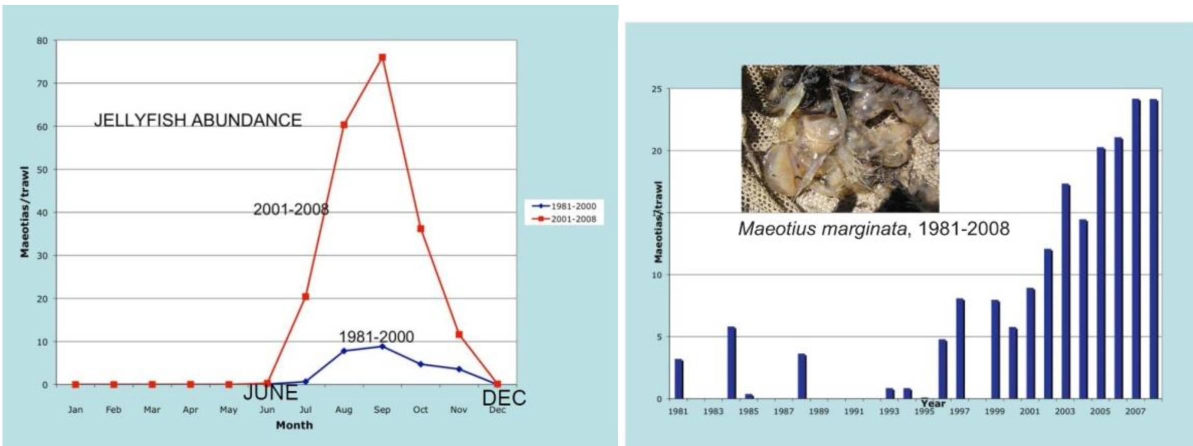


Figure 5.6.6. Invasive jellyfish abundance over time in Suisun Marsh (Moyle ppt. 2009).

5.6.5 Utility of Indicator as an Eutrophication Indicator for San Francisco Bay

Clear Linkage to Beneficial Uses

Jellyfish often have direct negative impacts on beneficial uses of estuaries and open waters. They impact primary and secondary contact recreation by stinging and deterring swimmers sail boarders, and boaters, they can increases costs to marine businesses by clogging fishing nets and can kill fish in aquaculture net-pens. They can also interfere with power plants by clogging cooling-water intake screens (Purcell, 2007; Richardson, 2010). They have indirect effects on fisheries by feeding on zooplankton and ichthyoplankton, and, therefore, becoming predators and direct competitors of planktivorous fish for available food resources (Purcell, 2007; Purcell, 2001).

Predictive Relationships to Causal Factors

While jellyfish populations appear to increase as a consequence of nutrient enrichment and increases in hypoxic conditions, they also increase in relation to mortality rates of competitors, harvesting of zooplanktivorous fish and overfishing, and other predators. There are also changes in jellyfish populations due to increases in water temperatures and other global processes, such as the reverse in the Pacific decadal oscillation (Cloern et al., 2010; Purcell, 2001). These many reasons which explain changes in jellyfish populations create a challenge in how to isolate nutrient enrichment from other causes of change. Therefore, presently there is not a proven predictive relationship with causal factors which indicate eutrophication and the many cofactors which influence this relationship would need to be explored. Anderson and Piatt (1999) suggest the need to observe changes over a complete climate cycle, if not several, before confidently drawing conclusions between climate change and the biological impacts of jellyfish (Anderson and Piatt, 1999).

Sound and Practical Measurement

Although Peter Moyle's long term study on Suisun Marsh has shown increasing invasive jellyfish populations, there are very few other long term studies tracking invasive species colonization of the SF Bay, as well as long term monitoring of blooms of native jellyfish populations. The use of jellyfish as an indicator for nutrient numeric endpoints in the SF Bay is challenged by this lack of long term distribution and abundance datasets from species specific sampling for both invasive and native species, as well as a lack of biological understanding of life histories of jellyfish (May, 2006; Purcell, 2007; Mills, 2001, Mills and Sommer, 1995). Environmental changes affect both the benthic and pelagic stages of the jellyfish though, even less is known about the benthic polyp stages than the pelagic jellyfish (Purcell, 2007). Knowledge of both life stages is essential to understanding the causes of blooms well as being able to predict and manage any resulting impacts (Mills, 2001; Wintzer, 2010).

Acceptable Signal to Noise Ratio

The existing data on the relationship between jellyfish and eutrophication, although anecdotally compelling, do not support an acceptable signal to noise ratio as understood thus far. The changes in nutrient enrichments over time do not correlate to the studies of population dynamics and spatial and temporal variation of jellyfish and thus more models or experiments would need to be developed to confidently associate jellyfish dynamics with a nutrient numeric endpoint.

5.6.6 Summary: Use of Jellyfish as an NNE Indicator

Based on this review, jellyfish do not meet the evaluation criteria as an acceptable indicator for the SF Bay NNE. We do not recommend pursuing jellyfish as an indicator further for this purpose.

5.7 Dissolved Oxygen

Dissolved oxygen (DO) refers to the concentration of oxygen in the water column (reported in units of concentration (mg L^{-1}) or percentage of estimated saturation). Adequate DO is required for the health of aquatic systems and organisms. Hypoxia is the term used by scientists studying aquatic systems to describe stress of organisms (usually fish but also invertebrates) due to low oxygen. Hypoxia as a stressor differs from chemical toxicants in that it can occur naturally; hypoxia is a consequence of the balance of atmospheric oxygen diffusion to surface waters, the *in situ* production of oxygen by primary producers during daylight hours, their night time respiration, in combination with the respiration of decaying organic matter and other biogeochemical processes that consume oxygen within surface waters and sediments. In cases where hypoxia has anthropogenic origins, the assumption is that hypoxia may be reduced by controlling nutrient availability and reducing the supply oxygen-demanding material to a waterbody.

Hypoxia exhibits temporal variability, on diurnal, tidal, lunar, and seasonal timescales. Seasonal hypoxia often develops in association with stratification. Hypoxic water can occur as stratified water prevents the oxygenated surface water from mixing downward or when upwelled hypoxic water is advected into an estuary from offshore. Hypoxia can appear in water near the sediment interface when respiration in the water and sediment depletes oxygen faster than it can be replenished. Breakdown of the stratification allows the surface and bottom waters to mix. Stratification can occur in both deepwater habitat of perennially tidal enclosed bays, such as SF Bay, or in lagoonal or river mouth estuaries that are intermittently closed to tidal exchange and that are known to "trap salt" (Largier et al., 1991). Diel cycles of hypoxia often appear in stratified or unstratified shallow habitats where nighttime respiration, in combination with water column and sediment dissolved oxygen demand, can deplete DO. Tidal and lunar frequencies can become apparent, particularly in poorly flushed areas where greater exchange occurs on flood or ebb tides or during a spring tide.

Oxygen demand and resulting reductions in dissolved oxygen vary spatially and temporally and may be more or less persistent. The response of aquatic organisms to low DO will depend on the intensity of hypoxia, duration of exposure, and the periodicity and frequency of exposure (Rabalais et al., 2002). Organisms have developed several physiological and behavioral adaptations to deal with temporary periods of low oxygen availability. Organisms can: 1) temporarily utilize anaerobic pathways to produce energy (ATP); 2) scavenge oxygen from hypoxic waters and increase the efficiency of oxygen transport to cells; 3) emigrate from hypoxic zones; 3) utilize the abundant oxygen from the surface or breathing aerial sources; or 4) reduce demand for oxygen by reducing activity. However, these are all short-term strategies and will not enable the animal to survive during long hypoxic periods. Adaptations are well developed in animals such as intertidal and burrowing animals that commonly experience hypoxia but poorly developed in animals that inhabit well-oxygenated environments such as the upper water column. If oxygen deficiency persists, death will ensue. Sublethal effects also occur. For example, reduced motor activity from mild hypoxia may make the animal more vulnerable to predators or decrease its growth or reproduction. Several components of SF Bay can have an associated oxygen demand, e.g.:

- Organic rich waste loads from agricultural, municipal, or industrial sources
- Allochthonous or autochthonous organic matter produced as live or dead plants, algae, and animal tissue, found in sediments or surface waters
- Chemical oxygen demand resulting from redox reactions in sediments or surface waters

5.7.1 Applicable Habitat Types

Dissolved oxygen is applicable principally to the subtidal habitats of the North, Central and South Bays. Utility and applicability of DO to diked Baylands requires additional discussion, particularly because muted habitats are to some degree subject naturally to hypoxia. In addition, these habitat types are known to influence subtidal DO. Low DO water can exist in salt ponds that, if breached, can supply high organic discharge or low DO discharge to the Bay (Shellenbarger et al., 2008; Thebault et al., 2008). Similarly, managed duck ponds can also have local influences on DO if waters from duck clubs are allowed to mix with Bay waters too rapidly.

5.7.2 Available Data on Dissolved Oxygen

The USGS has been collecting water quality data in San Francisco Bay on water quality and nutrients continuously for 39 years beginning in 1968 (see Section 5.2). It is important to note that this was a research program drive by hypothesis related questions. There are gaps in the data set. For example, the USGS didn't collect DO data from the mid 1970s to the mid 1980s. That accepted, the database generated presently includes over 6900 discrete laboratory measurements of dissolved oxygen in water samples and 119,685 estimates of dissolved oxygen made from a linear relationship between the oxygen electrode voltage output and discrete lab measurements. Without support data sets like this may not always continue into the future since the USGS research program is not mandated. There is the critical need for a commitment to support regular sampling to measure and understand future changes in DO related to nutrient enrichment.

5.7.3 Indicator Relationship to Nutrients and Water Quality

San Francisco Bay has generally been considered a nutrient enriched but low primary production environment (Cloern, 1987; Cloern et al., 2005b; Wankel et al., 2007; Cloern et al., 2007). Factors that limit primary production in the Bay include light limitation due to high turbidity in the water column, strong physical tidal/wind forcing that prevents thermal/saline stratification, and high predation from filter feeding bivalves (Cloern, 2001; Cloern et al. 2007). Due to these controls, SF Bay has not experienced the water quality issues e.g., hypoxia and anoxia associated with high nutrients and high primary production that have impacted other coastal estuaries (Cloern et al., 2001; Diaz, 2001; Wankel et al., 2006; Breitburg et al., 2009).

Therefore, SF Bay is not considered impaired by low dissolved oxygen conditions. Based on data collected by the USGS, minimum concentrations have only rarely dipped below 5 mg L^{-1} (the water quality objective outlined in the San Francisco Bay Basin Plan (SFBRWQCB, 2007)) on a few occasions in the Central, South and Lower South Bay segments (Table 5.7.1). Central Bay had the highest incidence of dissolved oxygen levels less than 5 mg L^{-1}. However, in the past decade there has been an increase in primary production, including a new South Bay autumn phytoplankton bloom. This period of increased primary production has coincided with decreases in nutrient inputs (due to improved wastewater treatment) over the same time period (Cloern et al., 2007). There has also been a co-occurring shift in Eastern Pacific oceanic conditions and the California Current that has resulted in favorable Bay conditions for bivalve predators, a subsequent reduction in bivalve biomass, and increased primary production (Cloern et al., 2007).

A major threat, in high nutrient estuarine systems, is the potential for increased primary production, including HABs, and the subsequent increase in heterotrophic activity that can lead to hypoxic and anoxic conditions (Diaz, 2001). This scenario has played out in many estuaries around the world (Diaz, 2001; Breitburg et al., 2009). San Francisco Bay has had incidents of HABs in the past 15 years (See phytoplankton section Table 5.2.2). However, there were no reports of reduced dissolved oxygen during or following a large dinoflagellate bloom in South SF Bay (maximum chlorophyll a concentrations were 195 mg/m^3 which was 65 times higher than the 27 year average for August – October time period)

(Cloern et al., 2005b). Adequate dissolved oxygen levels were most likely maintained due to a quick dissipation of the bloom from physical forcing of the tides and winds mixing the water column. There is little information for SF Bay on dissolved oxygen concentrations during other HAB episodes. That said, it seems feasible that if the increased production trend continues or if the incidence of harmful algae blooms increases, there might be a concomitant increase in the frequency of lower DO events especially if they happen to coincide with neap tides or lower wind conditions which can lead to stratification.

Table 5.7.1. Minimum, maximum, and mean dissolved oxygen concentrations in each segment of San Francisco Bay based on data from 1999-present. The number of samples in each measurement and the percent of dissolved oxygen measurements less than 5.0 mg L^{-1} are also shown. (Source: J. Cloern, USGS): http://sfbay.wr.usgs.gov/access/wqdata).

	Calculated Oxygen* (mg/L)				
Segment	**Minimum**	**Maximum**	**Mean**	**n**	**% < 5 mg/l**
Rivers	7.2	10.9	8.9	5,071	0.00%
Suisun Bay	6.9	11.0	8.8	5,435	0.00%
Carquinez Straight	6.6	11.0	8.3	7,563	0.00%
San Pablo Bay	5.6	10.7	7.9	5,802	0.00%
Central Bay	4.2	12.3	7.7	30,616	0.20%
South Bay	4.7	14.4	7.8	18,517	0.04%
Lower South Bay	4.8	14.6	7.7	5,657	0.05%

*Note: Calculated Oxygen is the estimated concentration of dissolved oxygen, calculated from the oxygen electrode voltage calibrated with the discrete measures of the dissolved oxygen using linear regression. (USGS website: http://sfbay.wr.usgs.gov/access/wqdata).

5.7.4 Factors Affecting Temporal and Spatial Variation of Indicator

Dissolved oxygen concentration and percent saturation varies both spatially and temporally in SF Bay. Dissolved oxygen levels can range from anoxic (oxygen depletion) to super saturation (>100% saturation). Factors that control dissolved oxygen variability include tidal forcing (advection of oxygen from and to coastal waters), wind stress (oxygen exchange between atmosphere and water surface), biological activity (photosynthesis and respiration), and freshwater input (Conomos et al., 1979). Gaseous atmospheric exchange with the Bay is the greatest oxygen source.

Spatial Variability

Freshwater outflow from the Sacramento/San Joaquin Delta is the largest freshwater supply to the Bay (Conomos et al., 1979). Freshwater input from South Bay tributaries is much lower. Dissolved oxygen concentrations tend to be higher in the northern reaches of the Bay (decreasing the oxygen concentration gradient from north to south) (Table 5.7.1 and Figure 5.7.1) and in the upper portions of the water column (Conomos et al., 1979; Peterson, 1979). Water column oxygen variability is controlled by density/thermal stratification and results in higher oxygen concentrations in the upper portions of the water column. Isohaline conditions prevail during summer due to strong wind and tidal forcing which results in a mixed water column with less vertical variability in oxygen levels (Conomos et al., 1979). Decreased dissolved oxygen generally occurs in the lower portions of the water column during periods of stratification when there is high availability of organic matter (Diaz, 2001).

Temporal Variability

There are two major scales of dissolved oxygen temporal variation: seasonal and hourly (tidal and variation in photosynthesis/respiration). The Bay has a Mediterranean climate with mild wet winters and dry temperate summers. The majority of precipitation occurs from October through April with rain runoff occurring during the winter months and snowmelt runoff during the early summer (Conomos, 1979). Freshwater runoff is a source of oxygen to the Bay with maximum runoff occurring during winter storm events (Peterson, 1979; Cloern, 1996; McKee et al., 2006). Dissolved oxygen concentrations are highest during winter due to increased solubility of oxygen in colder water, decreased oxygen dependent biological activity, and increased oxygen supply from freshwater inflow (Figure 5.7.1 and 5.7.2) (Conomos et al., 1979; Peterson, 1979). Dissolved oxygen levels can also increase during the spring phytoplankton bloom (Peterson, 1979; Cloern, 1996) and can become supersaturated during these blooms (Cloern, 1996; Cloern et al., 2005). Dissolved oxygen levels are lowest during the summertime in all Bay segments (Figure 5.7.1 and 5.7.2).

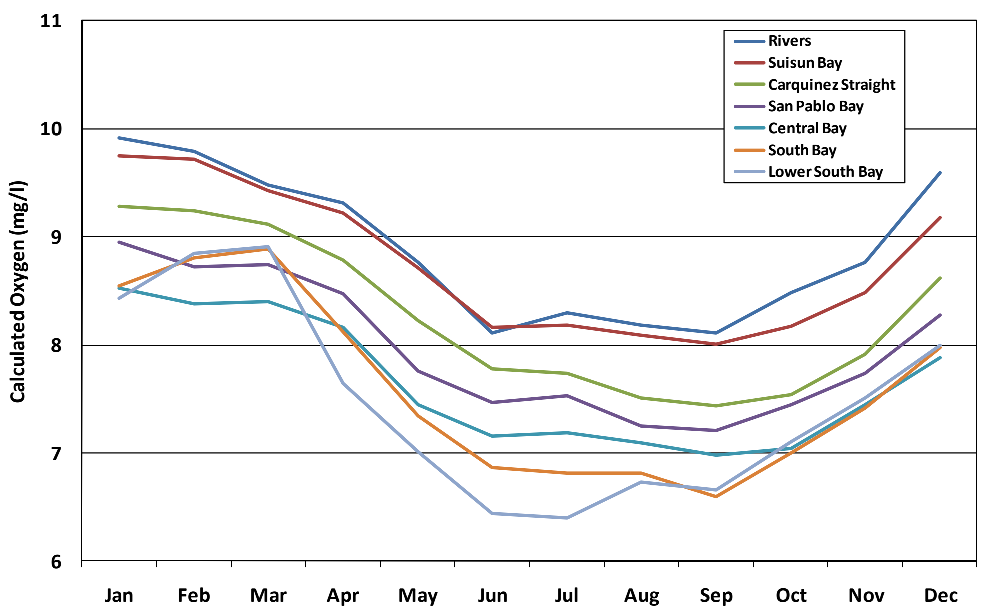


Figure 5.7.1. Monthly averaged dissolved oxygen concentrations in the defined segments of the San Francisco Bay for 1999 – 2010. Data obtained from the USGS: http://sfbay.wr.usgs.gov/access/wqdata).

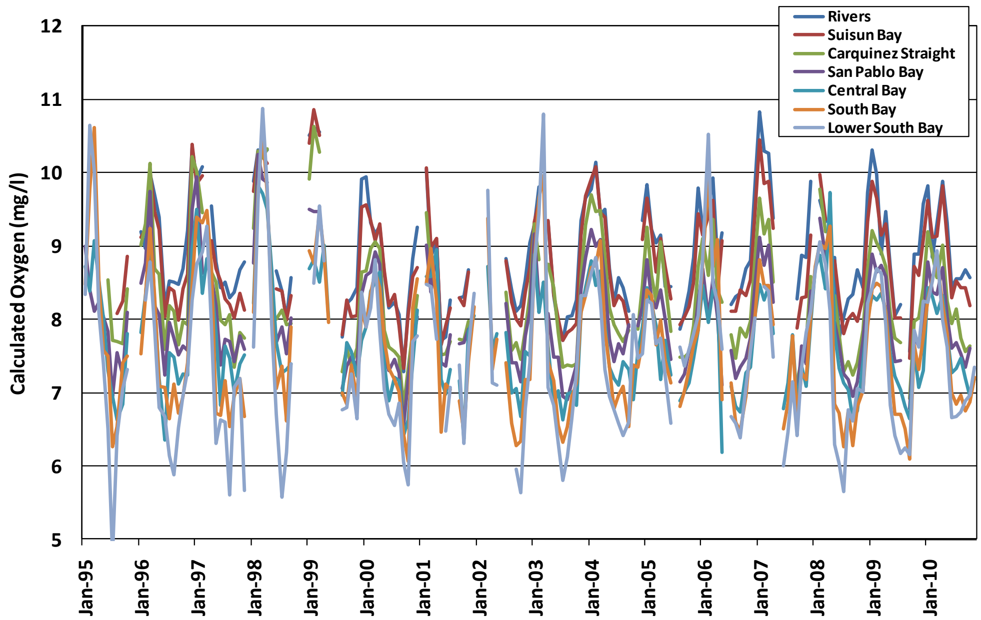


Figure 5.7.2. Average dissolved oxygen concentrations in the defined segments of the San Francisco Bay for measurements between 1995 – 2010. Data obtained from the USGS: http://sfbay.wr.usgs.gov/access/wqdata).

Hourly variability in dissolved oxygen is controlled by tidal forces (semi-diurnal) and the balance of photosynthesis and respiration. There are daily (2 high and low tidal cycles per day) tidal cycles that produce varying tidal currents with spring tides producing maximum tidal currents. Tidal currents generally supply lower oxygenated waters to the bottom waters of the Bay (Peterson, 1979). Hourly variation of dissolved oxygen is due to increased primary productivity during daylight hours and subsequent decreases in primary productivity during nighttime hours (Thebeault et al., 2008).

Indicator Trends

Generally speaking, dissolved oxygen concentrations and percent saturation in the Bay have been fairly consistent, over the long term, since implementation of secondary wastewater treatment (Cloern, 2003). Figure 5.7.3 below illustrates no marked shift over the 15 year period (1995 – 2010) in either the lowest or highest concentrations. This was not always the case, however. Low oxygen events were a common occurrence in the 1960s, but since the introduction of secondary wastewater treatment in the 70s, events of low oxygen (<5 mg L^{-1}) have been rare (Table 5.7.1 and Figure 5.7.3) (Cloern et al., 2003). The most notable historic example studied was a sewage spill that occurred near the mouth of Coyote Creek in the South Bay in 1979. For two weeks fishermen noted a lack of fish and pelagic invertebrates in Coyote Creek. Within several weeks of resuming treatment, at the San Jose/Santa Clara plant, both fish and invertebrates returned to the Creek (Cloern and Oremland, 1983). Presently, a reanalysis of DO data is being completed to further evaluate summer trends. Preliminary data analysis conducted by Alan Jassby and James Cloern suggests decreasing DO in bottom waters (>5 m) in South Bay, San Pablo Bay, and Suisun Bay since the early 1990s at an average rate of 1.5-2.5% per decade (note a much lower rate of change compared to chlorophyll a) (James Cloern Personal Communication, March 2011). These new analyses provide further evidence that the Bay is changing, perhaps motivating further interest to understand the effects of nutrient loads and other co-factors. While for DO the rate of change is low, the ramifications could be large and the rate may not be linear.

5.7.5 Utility of Dissolved Oxygen as an NNE Indicator for San Francisco Bay

Clear Linkage to Beneficial Uses

Dissolved oxygen has a clear and well established linkage to beneficial uses in the SF Bay. While the response of aquatic organisms to low DO will depend on the intensity of hypoxia, duration of exposure, and the periodicity and frequency of exposure (Rabalais et al., 2002), USEPA has an extensive database documenting adverse effects of low DO on a variety of fish and invertebrates with respect to juvenile and adult survival, reproduction and recruitment (EPA, 2003). Impacts of hypoxia on SF Bay pelagic and benthic organisms would have a direct impact on important SF Bay beneficial uses, including food web support for marine and estuarine aquatic organisms (EST, MAR) including the commercial and sport fisheries (COMM), shellfish such as clams, oysters and mussels (SHELL and AQUA), migratory (MIGR) birds and fish, support for fish nursery habitat (SPAWN). Poor water quality and increase heterotrophic bacterial production would adversely affect the health of recreational swimmers, sailboarders, and boaters (REF-1) and decrease aesthetic enjoyment of the Bay (REC-2) through nuisance buildup and smell during decay.

San Francisco Bay currently has a dissolved oxygen water quality objective (See section 3, Table 3.5) for maintaining organism/ecosystem health. However, the science supporting the selection of these regulatory thresholds have not been reviewed recently and it is likely that additional science is available that may shed light on whether revised DO objectives are needed for SF Bay. Currently, as part of the NNE framework for estuaries, a review of the science supporting dissolved oxygen objectives is currently being undertaken for California estuaries, exclusive of SF Bay. A review of this nature could be beneficial for SF Bay, including applicability of DO to diked Bayland and salt pond habitat.

Predictive Relationships to Causal Factors

Reduced dissolved oxygen is a measurable indirect impact of high nutrients and high primary productivity under certain conditions. Reduced dissolved oxygen can occur under conditions of water column stratification, high nutrients, ample sunlight, and high primary production and subsequent decomposition by heterotrophs. Modeling these conditions using empirical SF Bay data could help predict when and if reduced/low dissolved oxygen responses could occur. But to-date, that we are aware of, there has been no models developed to predict the spatial and short or long term temporal characteristics of DO in relation to external and internal causative factors. Open-source dynamic simulation models exist to predict dissolved oxygen concentrations from nutrient and organic matter loading and other co-factors.

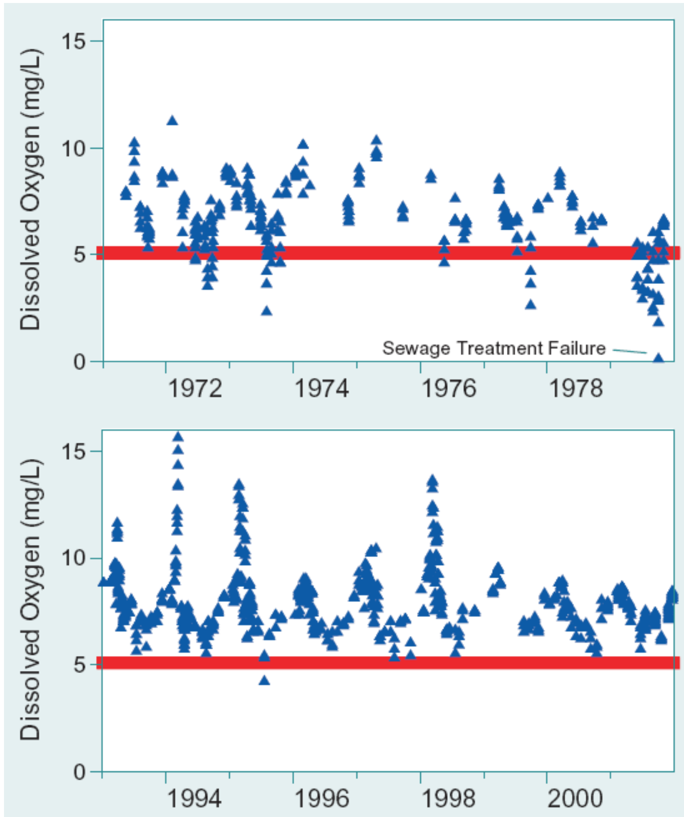


Figure 5.7.3. Dissolved oxygen concentrations in bottom waters of the South Bay. Red line shows the San Francisco Bay water quality objective for dissolved oxygen. Figure directly excerpted from Cloern et al., 2003.

Sound and Practical Measurement

Dissolved oxygen has a well-established and practical means of measurement. In addition, there is a long term dissolved oxygen data set available for SF Bay. USGS collects monthly measurement of dissolved oxygen along the spine of the Bay. RMP collects measurements of dissolved oxygen once a year in the summer according to a randomized study design. Both data sets provide a measure of the spatial variability in dissolved oxygen. USGS data can be used to indicate short-term temporal dissolved oxygen variability within a year as well as long term trends while RMP data can be used to indicate long-term trends and spatial variation of the Bay axis. There is ongoing monthly monitoring of the Bay for multiple water quality indicators including dissolved oxygen and chlorophyll a which, in combination, could be useful indicators.

Acceptable Signal to Noise Ratio

As with phytoplankton, dissolved oxygen concentrations in SF Bay have a high degree of spatial and temporal variability. However, DO is a well-vetted indicator of eutrophication and there is considerable experience with its use in a regulatory context to manage eutrophication. Due to its ease of measurement and the existence of a long term data set spanning 39 years, and an advance understanding of and measurement of phytoplankton blooms in the Spring and autumn, there is very high potential for development of DO ad an indicator of Bay health. The existence of a water quality guideline and clear relationships to beneficial use impairment provides a strong opportunity to measure trends either towards increasing or/and decreasing eutrophication with a very high signal: noise ratio.

5.7.6 Summary: Use of Dissolved Oxygen as an NNE Indicator for San Francisco Bay

Dissolved oxygen satisfies all four evaluation criteria and is recommended as an NNE indicator for the subtidal habitats of SF Bay. Existing basin plan objectives for dissolved oxygen exist for SF Bay. This review focused exclusively on summarizing status and trends of dissolved oxygen along the longitudinal axis of the Bay. Existing DO data available for tidally muted habitats (managed ponds) or in shallow areas of the Bay were not summarized in the review.

The SF Bay TAT did not have the expertise or budget to review the adequacy of dissolved oxygen objectives for SF Bay. A helpful framework to consider the scientific basis for dissolved oxygen objectives is the the USEPA Virginia Province Salt Water Dissolved Oxygen Criteria approach (USEPA, 2003). The fundamental goal of this approach is to maintain and support aquatic life communities and their designated uses. The approach was developed for the region of the east coast of the US from Cape Cod, MA, to Cape Hatteras, NC. This approach has been adapted for use in setting DO criteria for Chesapeake Bay (Batiuk et al., 2009), and has been applied, with appropriate modification, the other coastal regions of the US including Maine and Alabama. A review of the science supporting dissolved oxygen objectives in"other" California estuaries is onging, utilizing the EPA Virgina Province approach. If desired a similar review could be undertaken for SF Bay in its entirety or for selected habitat types such as tidally muted areas.

5.8 Ammonium and Urea Concentrations

San Francisco Bay is a nutrient-enriched estuary (Peterson, 1985a; Alpine and Cloern, 1988; Cloern et al., 2007). While primary productivity has been trending upwards for the past decade and the incidence of short lived harmful algae blooms appear to be on the increase, the Bay has not seen impacts from high nutrients like other eutrophic systems and for the last 30 years has been classified as a high nutrient, low primary productivity environment (e.g., Chesapeake Bay, Gulf of Mexico) (Cloern et al., 2007; Dugdale et al., 2007).

The primary pathways of nitrate and ammonium loads to the Bay are freshwater inflow (Sacramento/San Joaquin River and smaller tributaries), atmospheric deposition (Anderson et al., 2002), oceanic tidal exchange, and wastewater effluent (Conomos et al., 1979; Hanger and Shemel, 1992; Wankel et al., 2006). Nitrogen sources to the North Bay are primarily anthropogenic and include agricultural runoff as a main source in addition to nutrient loads from the Sacramento Regional Wastewater Treatment Facility (Sac Regional) (Hanger and Shemel, 1992; Wankel et al., 2006; Dugdale et al., 2007). Urea dominantly enters coastal ecosystems through the anthropogenic sources of stormwater and wastewater discharge (Glibert et al., 2006). The main sources of urea in these two types of discharge is uric acid from animal and human sources, direct application of urea as a fertilizer, production during industrial process (see review by Glibert et al., 2006, p447). Urea can also be found in atmospheric deposition and concentrations that can be similar to nitrate. While external loads typically exceed *in situ* production, urea is also known to be produced through excretion from zooplankton, bacterial regeneration, and release from bottom sediments (see review in Glibert et al., 2006, p446). Based on measurements in few coastal systems, rates of in situ urea production appear to be lower than ammonium production and too low to explain concentrations found in the water column suggesting that external sources are dominant (see review in Glibert et al., 2006, p446).

Inclusion of ammonium and urea concentration in the list of candidate NNE indicators represents a slight departure from the NNE framework, in which diagnosis of adverse effects is made on the ecological response to nutrients. The reason for this exception is based on recent evidence that ammonium concentrations may be responsible for limiting the spring diatom blooms in the North Bay and the lower Sacramento River (Wilkerson et al., 2007, Dugdale et al., 2006). In addition, urea has been proposed as one important factor that favors the dominance of certain species of HABs. This work is reviewed here in order to determine whether it is appropriate to include either ammonium and/or urea as NNE indicators in SF Bay.

5.8.1 Available Data on Nitrate+Nitrite, Ammonium, and Urea in the San Francisco Bay

The USGS has been collecting water quality data in SF Bay on nitrite, nitrate, ammonium and other nutrients for 39 years beginning 1968. Their research program has included many measurements of water quality from a monthly ship cruse at 39 fixed locations along the spine of the Bay. Since the USGS sample-collection was driven by research questions, it has not always been as regular or systematic as would occur in a monitoring program. For example, the USGS stopped sampling completely in 1981 after the spring bloom and didn't sample in the North Bay from about 1980-1987. That said, the database of these measurements presently includes over 9,000 discrete laboratory measurements of nitrate + nitrite (NOx) and over 8500 discrete laboratory measurements of ammonium (NH_4^-). There are isolated measurements of urea in Bay waters collected during specific studies (Cochlan and Herndon, unpublished data (cited in Kudela et al., 2008, p108); Herndon et al., 2003; Kudela et al., 2008). In addition, monthly concentrations of urea, ammonium and nitrate data are being collected at a series of sites in SF Bay as part of the National Estuarine Research Reserve (see comment by Kudela et al., 2008, p108). We have not reviewed this data.

5.8.2 Indicator Trends and Factors Affecting Temporal and Spatial Variation

Nutrients vary over multiple temporal and spatial scales. Temporal scales of variability include daily (flux of nutrients with daily photosynthetic activity and tidal exchange), seasonal (influx in winter freshwater flows and flux from phytoplankton blooms/decay), and interannual (long-term variability due to changes in watershed management/use). Spatial variability includes vertical (vertical flux in the water column) and horizontal (freshwater/wastewater effluent inflow, tidal exchange) scales.

Nitrogen and phosphorus are required nutrients for growth of photosynthetic organisms such as unicellular phytoplankton. Nitrate (NO_3^-) and ammonium (NH_4^-) are the two primary dissolved inorganic nitrogen forms in aquatic systems that are assimilated by primary producers (Peterson, 1985). Ammonium is the preferred nitrogen source for assimilation by phytoplankton and can become limiting in the environment (Peterson, 1985b; Hogue et al., 2001; Hogue et al., 2005; Wankel et al., 2006). However, in SF Bay nitrate is generally in high supply and phytoplankton can switch to nitrate assimilation when ammonium is depleted (Hogue et al., 2001). There is also evidence in North Bay (Suisun Bay and lower Delta) studies that ammonium has an inhibitory effect on phytoplankton nitrate uptake when ammonium concentrations exceed 4 µmol L^{-1} (Wilkerson et al., 2006; Dugdale et al., 2007). During springtime blooms in the North Bay there is an initial ammonium uptake by phytoplankton, subsequent ammonium depletion below 4 µM, followed by high rates of nitrate uptake resulting in a bloom period (Wilkerson et al., 2006). This inhibitory effect may prevent phytoplankton blooms from occurring since phytoplankton uptake of ammonium occurs at slower rates than nitrate uptake thus limiting rates of primary productivity (Dugdale et al., 2007). Dugdale et al. suggest that ammonium inhibition could be one of the limiting factors that control primary productivity in the Bay and contributes to the Bay's low primary productivity. Since, the greatest rates of production can occur when ammonium is low and nitrate is high, the concept of an ammonium:nitrate ratio as a key indicator have been advanced (Wilkerson et al., 2006; Dugdale et al., 2007).

In addition, urea has also been identified as a nutrient source taken up by phytoplankton (Gilbert et al., 2001; Anderson et al., 2002; Kudela et al., 2008). Loadings of urea have increased in certain areas, primarily due to the increased use of urea-based fertilizers (as reviewed in Anderson et al., 2002; Gilbert et al., 2006). There is some evidence that certain phytoplankton species, mostly flagellates, prefer uptake of urea over other nitrogen forms and that urea can increase the toxicity of a bloom (as reviewed in Anderson et al., 2002 and Kudela et al., 2008). Some of these flagellates have been identified as harmful or nuisance species (Gilbert et al., 2001; Anderson et al., 2002; Gilbert et al., 2006; Kudela et al., 2008). Experiments using local harmful algal species (coastal California and Bay species) showed some preferential uptake of urea when ambient nutrient concentrations were low (Kudela et al., 2008). It has also been suggested that urea may maintain harmful blooms (Kudela et al., 2008; Gilbert et al., 2001). In Chesapeake Bay, high urea concentrations were measured prior to a spring HAB bloom (Gilbert et al., 2001). The unusually high urea levels were correlated with high springtime precipitation that may have increased urea loading prior to the bloom. Urea has been measured in Bay waters but it is unclear how urea may be stimulating and/or maintaining phytoplankton blooms, particularly HABs, in the Bay (Herndon et al., 2003; as reviewed in Wilkerson et al., 2006). However, there is some evidence that urea may have an inhibitory effect on nitrate uptake by phytoplankton (as reviewed in Kudela et al., 2008). Given urea use as fertilizer continues to rise, urea will likely to continue to form a greater portion of the dissolved organic nitrogen (DON) pool available for primary production in coastal ecosystems (Glibert et al., 2006). Quantification of the standing stock of DON and the processes of assimilation remain an under studied area of research.

Spatial Variability

Nutrient spatial variability is explained by the balance of nutrient inputs (source), transport, assimilation by primary producers (sink), and re-mineralization/vertical flux (regeneration) (Grenz at al., 2000). In the North Bay, there is a decreasing gradient in nitrate and ammonium concentrations from the fresher Suisun Bay to the more saline Central Bay (Hogue et al., 2001; Wilkerson et al., 2006). The Lower South Bay has the highest concentrations of NO_x due to the influence of wastewater effluent (Table 5.Y) (USGS data; Wankel et al., 2006). High ammonium levels are also associated with wastewater effluent and are more equally distributed around the Bay (Table 5.8.1). Ammonium and NO_x can be drawn down during spring phytoplankton blooms in all Bay segments (Hogue et al., 2001; Wilkerson et al., 2006). There are limited measurements of urea in the Bay. Data have been primarily collected from North Bay locations so spatial variation is not currently known.

Table 5.8.1. Average nitrate + nitrite and ammonium concentrations in each Bay segment 1999 – present (Source: J. Cloern, USGS: http://sfbay.wr.usgs.gov/access/wqdata). Data for urea-N are from short term studies by various authors as indicated.

Segment	(NO_x) Nitrate + Nitrite-N (mg L^{-1}) Average	Ammonium-N (mg L^{-1}) Average	Urea-N (mg L^{-1}) Location, date	Urea-N (mg L^{-1}) Average (Range)
Suisun	0.38	0.11	[1]"North Bay", unknown date	>0.024
Carquinez Straight	0.36	0.11		
San Pablo Bay	0.32	0.09		
Central Bay	0.26	0.09	[2]Richardson Bay, Jun, Jul, Sep 2002	0.0090 (0.0063-0.0137)
			[3]Western Richardson Bay and Paradise Cay, May-Oct, 2005	(0.00056 – 0.0051)
South Bay	0.35	0.08		
Lower South Bay	0.70	0.09		

[1] Cochlan and Herndon, unpublished data (cited in Kudela et al., 2008).

[2] Herndon et al., 2003.

[3] Kudela et al., 2008.

Temporal Variability

Temporal variability of dissolved nitrate and ammonium are controlled by river inflow, inflow of wastewater treatment plant effluent, photosynthetic drawdown, and tidal exchange with nutrient rich coastal waters (Conomos et al., 1979; Cloern and Nichols, 1985; Peterson, 1985a; Cloern et al., 2001). In the North Bay, there is a temporal dynamic of nutrient concentrations (interplay between source and sink) that is dependent on winter freshwater flow. Winter periods are generally characterized by low primary productivity and high nutrient concentrations while during spring blooms, primary productivity dominates nutrient dynamics (Figure 5.8.1 and 5.8.2) (Peterson, 1985a; Wilkerson et al., 2006; Dugdale at al., 2007). There is much interannual variation in this dynamic depending on the magnitude of winter river flow, the duration of high river flow into the spring/summer, and the magnitude of the spring phytoplankton bloom. Nitrate and ammonium concentrations can be drawn down during phytoplankton blooms (mostly occurring during spring) due to removal by primary producers (Peterson, 1985a; Cloern, 1996; Hogue et al., 2001; Wilkerson et al., 2006; Cloern et al., 2007). In the South Bay there is interannual variation of nitrate and ammonium during springtime dependent on timing of the spring phytoplankton bloom (Cloern, 1996) however, there is presently no peer-reviewed articles that describe any influence ammonium inhibiting nitrate uptake in the South Bay.

Ocean dynamics also contributes to the temporal variation in nutrients, especially for nitrate (Peterson, 1985a). The coastal upwelling maximum occurs in June which can result in high nitrate concentrations in coastal waters outside the Golden Gate which then enter the Bay via incoming tides. Upwelling variation occurs on a longer-term timescale known as the Pacific Decadal Oscillation (PDO). The California coast is currently in a cold phase of the PDO which results in stronger upwelling bringing higher nitrate concentrations to surface coastal waters (Cloern et al., 2007).

There is also temporal variation in the ratio of NH_4^- and NO_x (NO_3^- + NO_2^-). This ratio may be used as an indicator of an ammonium inhibited environment. The ratio generally remains below 1.0 most months except for two peak periods in the South Bay (April and September) (Figure 5.8.3). This peak in ammonium concentrations, relative to NO_x, coincides with the spring and fall phytoplankton blooms. This may indicate that ammonium is not limiting primary productivity in the southern portions of the Bay. There are limited measurements of urea in the Bay. Data that have been collected show variability in urea concentrations with urea representing between 3 and 42% of total nitrogen (Kudela et al., 2008). However, measurements were only taken between May and October so overall seasonal variation is not known.

Indicator Trends

There are long-term data sets for nitrate and ammonium in SF Bay (USGS and RMP data). Long-term trend analysis of nutrients is sparse in the literature (Figure 5.8.4, 5.8.5, and 5.8.6). In South SF Bay, daily nitrogen loads from the San Jose – Santa Clara wastewater treatment plant have declined since the late 1970s (Cloern et al., 2007). Longer-term trends in the South Bay (1990 – 2005) suggest that dissolved inorganic nitrogen levels are decreasing up to a maximum of 10% per year (Cloern et al., 2007). There are no long-term data on urea concentrations in the Bay. However, there have been documented increases of urea loadings to other estuaries due to increased usage of urea-based fertilizers.

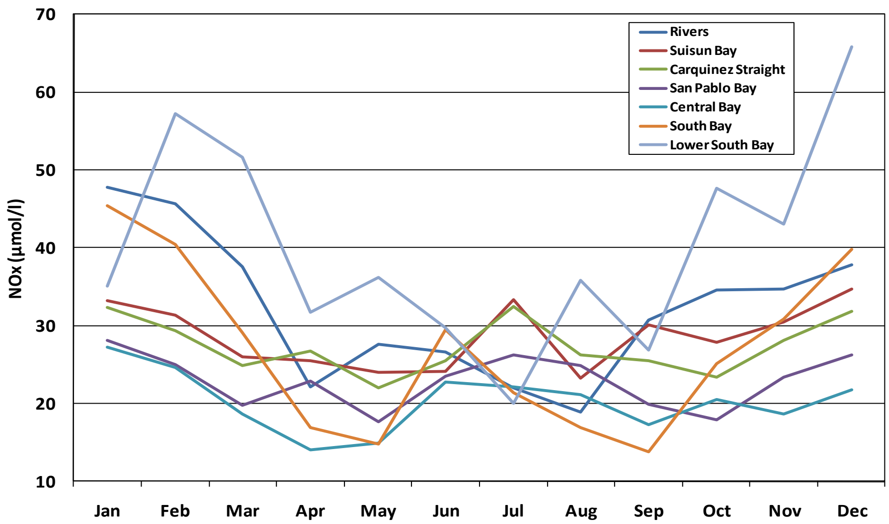


Figure 5.8.1. Monthly averaged nitrate + nitrite (NOx) concentrations in the defined segments of the San Francisco Bay for 1999 – 2010. Data obtained from the USGS: http://sfbay.wr.usgs.gov/access/wqdata).

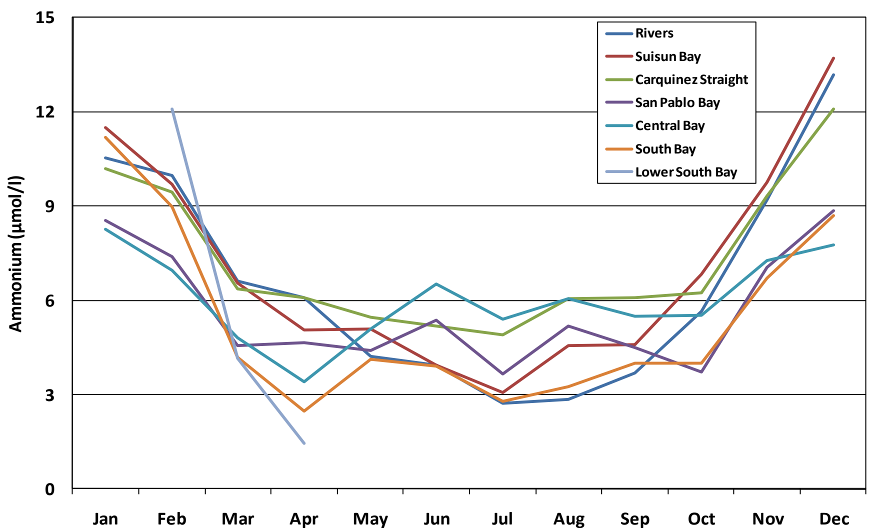


Figure 5.8.2. Monthly averaged ammonium concentrations in the defined segments of the San Francisco Bay for 1999 – 2010. Data obtained from the USGS: http://sfbay.wr.usgs.gov/access/wqdata).

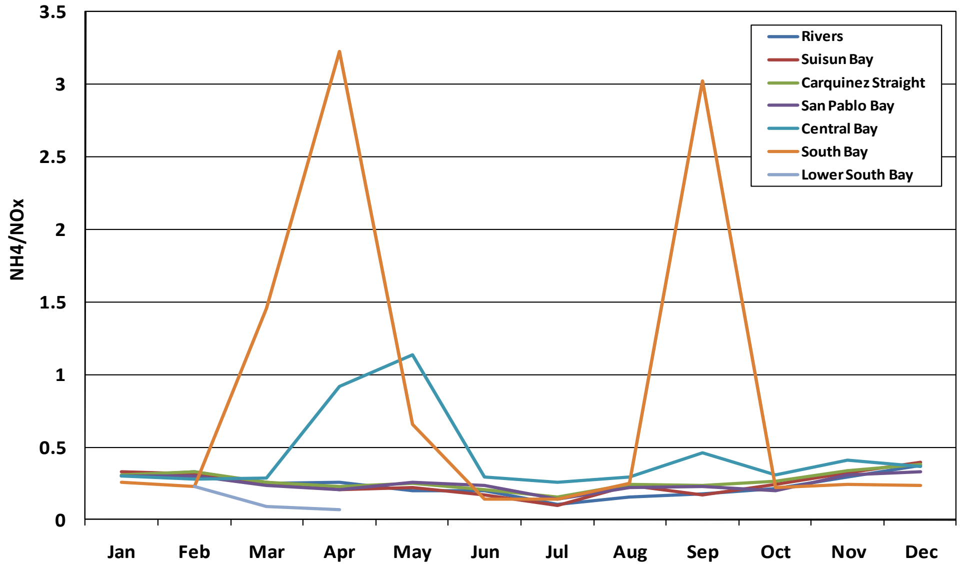


Figure 5.8.3. Monthly averaged ratios of ammonium to nitrate + nitrite in the defined segments of San Francisco Bay 1999 - 2010. Data from the USGS: http://sfbay.wr.usgs.gov/access/wqdata).

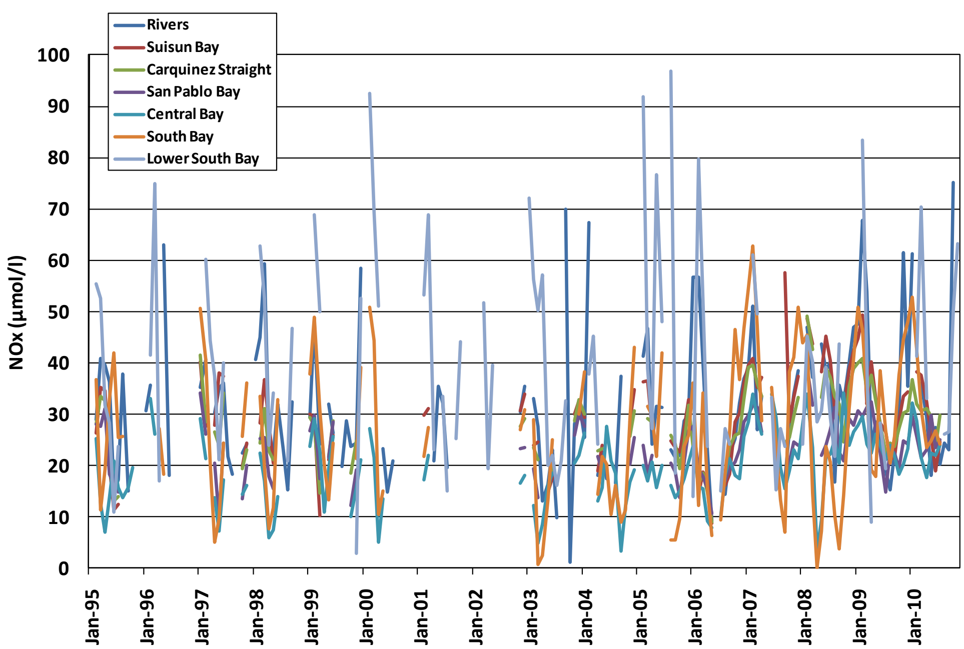


Figure 5.8.4. Monthly averaged nitrate + nitrite (NOx) in the defined segments of San Francisco Bay 1995 – 2010. Data from the USGS: http://sfbay.wr.usgs.gov/access/wqdata).

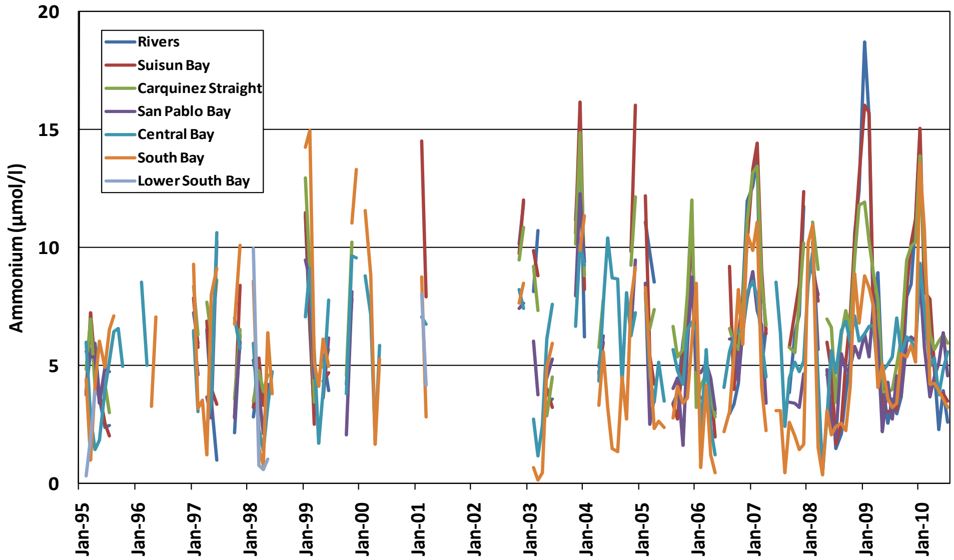


Figure 5.8.5. Monthly averaged ammonium in the defined segments of San Francisco Bay 1995 – 2010. Data from the USGS: http://sfbay.wr.usgs.gov/access/wqdata).

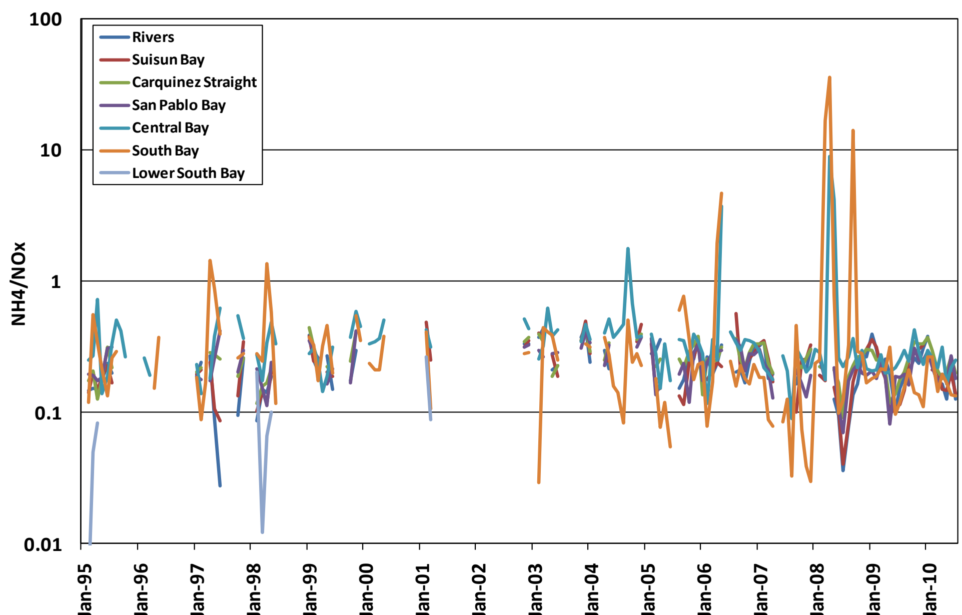


Figure 5.8.6. Monthly averaged ratios of ammonium to nitrate + nitrite in the defined segments of San Francisco Bay 1995 – 2010. Data from the USGS: http://sfbay.wr.usgs.gov/access/wqdata).

5.8.3 Utility of Indicator as an Eutrophication Indicator for San Francisco Bay

Clear Linkage to Beneficial Uses

Several studies (cited in Section 5.2) provide a broad base of evidence that phytoplankton have a direct linkage to important SF Bay beneficial uses, including food web support for marine and estuarine aquatic organisms (EST, MAR) including the commercial and sport fisheries (COMM), shellfish such as clams, oysters and mussels (SHELL and AQUA), migratory (MIGR) birds and fish, support for fish nursery habitat (SPAWN). Increased incidence of HABs can adversely affect the health of humans (REC-1) by irritation and injury to recreational swimmers, sailboarders, and boaters (Lehman et al., 2005). Thus adverse effects to phytoplankton primary production and the occurrence of dominant assemblages known to support Bay ecosystem services (e.g., fisheries) would be considered to adverse effecting Bay beneficial uses.

Ammonium inhibition of nitrate uptake by diatoms has been documented in several regions of the Bay and ammonium concentrations above 4 μM have been suggested as a major mechanism by which spring diatom blooms appear to be suppressed in the North Bay and lower Sacramento River (Wilkerson et al., 2006; Dugdale et al., 2007). Despite this evidence, the ecological importance of ammonium inhibition of spring diatoms blooms is not well understood relative to factors known to control primary productivity, particularly in other regions of the Bay where water column chlorophyll a appears to be increasing.

It has been suggested that HABs in the North Bay may be supported by regenerated ammonium in the summer and autumn (North Bay: Dugdale et al., 2007; South Bay: Thompson et al., 2008) and that HAB species that utilize ammonium as a nitrogen source can produce greater toxicity (see review by Glibert et al., 2006 and in Kudela et al., 2008). Likewise, some HAB organisms that can preferentially uptake urea may have a competitive advantage (Glibert et al., 2006). However, as with ammonium inhibition of diatom nitrate assimilation, the influence of ammonium on HABs in SF Bay has not been sufficiently investigated. Thus the linkage between ammonium concentrations and Bay beneficial uses is not at this time universally accepted. SF Bay TAT members agree that additional data synthesis is required to better understand the role of ammonium in SF Bay.

Kudela et al. (2008) noted that some HAB species can show a preference for urea versus other inorganic nitrogen constituents (e.g., *A. sanguinea*) or similar affinities for urea versus other nitrogen substrates (*L. polyedrum*), which under conditions of high urea concentrations could favor these HAB species. However, the importance of urea in promoting HABs in SF Bay is not understood, in part because measurement of urea is not part of a routine research programs conducted in the Bay and the focus of HAB research has been largely restricted to cyanobacteria. Thus the linkage between urea and adverse effects on the phytoplankton community and Bay beneficial uses are not well documented.

Predictive Relationships to Causal Factors

Conceptual models have shown a complex relationship between nutrients and primary productivity in SF Bay. There are multiple factors that have limited primary productivity in the Bay including light limitation, strong tidal/wind forcing that prevents long periods of stratification, and recent evidence that ammonium concentrations may be limiting the magnitude of spring blooms in the North Bay and that nitrate limitation might be an important factor on some occasions to causing bloom termination in the South Bay (Thompson et al., 2008). An added challenge in developing linkages between nutrient loads, standing concentrations, and productivity response is that these are often out of phase in our dry summer Mediterranean climate (Kudela et al., 2008). Empirical modeling could be beneficial to better understand relationships and predictive power between nutrients and impacts to beneficial uses. Existing models for the North Bay have not yet utilized recent empirical observations of the role of ammonium in bloom formation, and ammonium thresholds in relation to nitrate uptake rates (Dugdale et al., 2007). The impact of ammonium on nitrate uptake rates may also extend into San Pablo and Central Bays as advected ammonium that is not taken up in Suisun Bay due to unfavorable conditions and clam grazing appears to help maintain ammonium concentrations that do not allow nitrate uptake (Dugdale et al., 2007). However, as stated previously, the effect of ammonium inhibition on phytoplankton productivity throughout the Bay has not been modeled vis-à-vis other controlling factors.

There is growing evidence that DON, and in particular urea, is able to support greater population of HAB species. However at this time a causative linkage has not been established (Glibert et al., 2006); "while there is clear evidence that HABs can utilize multiple sources of nitrogen, this is not a unique characteristic and does not imply that urea selects for HAB species" (Kudela et al., 2008).

Given these data gaps, the logical next step is to the develop models that synthesize understanding of the relative importance of ammonium and urea versus other factors in controlling phytoplankton assemblages.

Sound and Practical Measurement

The laboratory methods for analysis of nitrate, nitrite, ammonium, and urea in natural waters are well established. However care should be taken to ensure reporting limits are sufficient to provide enumeration throughout all areas of the Bay during all periods of the annual cycle. There are long-term data sets on ammonium and nitrate levels in the Bay that are ongoing through the USGS and the RMP. During the peak of algal blooms, concentrations can be low and bloom termination can sometimes result from very low nitrate concentrations. Therefore, in order to study these phenomenon, using laboratories and methods that are able to report concentrations when they occur at $<10\mu g\ L^{-1}$ should be maintained. Data on urea concentrations are very limited but Kudela et al. (2008) proposed that this is an important data gap that if filled will provide an additional basis for improving our understanding of anthropogenic nutrient loadings and relationship to HAB species. In order to get useful information, monitoring programs should aim for reporting limits of $<1\mu g\ L^{-1}$ for urea.

Acceptable Signal to Noise Ratio

Since much of the nitrate and ammonium present in SF Bay has been cycled many times through the processes of assimilation and regeneration, there will be challenges with using either of these two nutrients or a ratio between the two as an indicator for changing Bay conditions. It seems that a more complex indicator model would give us a better understanding of the dynamics that lead to WQ impacts (low DO, HABs) and give us better predictive power on when/where these events might occur. The ratio of NH_4:NO_3 may be a good indicator if it can be developed to predict ammonium limitation, bloom initiation, or bloom termination. Urea, other hand perhaps holds great promise as the presence of urea in the Bay likely directly correlated with external anthropogenic loads since proportionally much less urea is likely derived from within the Bay through *in situ* generation (Kudela et al., 2008). However at this time the science of urea and HABs is in its infancy and needs further development to increase the signal to noise ratio especially for SF Bay.

5.8.4 Summary: Use of Ammonium and/or Urea as an NNE Indicator for San Francisco Bay

Elevated ammonium concentrations have been suggested as a major mechanism by which spring diatom blooms appear to be suppressed in the North Bay and lower Sacramento River (Wilkerson et al., 2006; Dugdale et al., 2007). Despite this evidence, the ecological importance of ammonium inhibition of spring diatoms blooms is not well understood relative to factors known to control primary productivity, particularly in other regions of the Bay where water column chlorophyll a appears to be increasing. Thus the linkage between ammonium concentrations and Bay beneficial uses is not at this time universally accepted. San Francisco Bay TAT members agree that additional data synthesis is required to better understand the role of ammonium in SF Bay. The SF Bay TAT recommends that a broader review of the subject, with the intent of synthesizing expert opinion on factors that influence the importance of this phenomenon in different regions of SF Bay and identifying critical data gaps.

While there is growing evidence that urea has a role in support an increased frequency of certain HAB species, a causative linkage has not been established (Glibert et al., 2006) and it is not possible to state that urea selects for HAB species (Kudela et al., 2008). Moreover little data are available on urea concentrations in SF. Therefore, recommended next steps are to begin collecting urea data in USGS SF Bay research cruises events and to undertake a broader review of the effects of urea on phytoplankton community composition, with the intent of identifying critical data gaps for its use in the NNE.

6.0 Synthesis, Data Gaps, and Recommendations

The ultimate goal of NNE program in SF Bay is to develop a diagnostic assessment framework to determine when SF Bay is experiencing adverse effects of nutrient and models to link those effects with nutrient loading and other controls on the estuary.

As a step towards this goal, this review had several objectives:

- Evaluate appropriate indicators to assess eutrophication and other adverse effects of anthropogenic nutrient loading in SF Bay
- Summarize existing literature and identify data gaps on the status of eutrophication in SF Bay with respect to these indicators
- Investigate what data and tools exist to evaluate the trends in nutrient loading to the Bay and, summarize, to the extent possible, what do they reveal about trends in nutrient loads over time?

This section summarizes recommends specific indicators, based on explicit indicator evaluation criteria, summarizes current data available in the Bay for those indicators, identifies data gaps, and recommended next steps.

6.1 Which Indicators Met Review Criteria?

The review found a suite of indicators met all evaluation criteria, while others met three of four criteria (Table 6.1). Selected indicators for the SF Bay NNE framework vary by habitat type. For the purposes of developing an NNE, SF Bay can be separated into four main habitat types, the first of which is tidally muted, while the remaining three habitat types are found in areas of the Bay which are not subject to anthropogenic tidal muting.

- Muted tidal areas (i.e., possessing tidal hydrology that is restricted by impoundments, dikes, or weirs such as estuarine diked Baylands, managed ponds (e.g., former salt ponds)

- Unvegetated intertidal habitat (intertidal flats)

- Unvegetated subtidal habitat

- Seagrass habitat

Unvegetated intertidal habitat represents 9.8% of SF Bay (~1200 km^2). Unvegetated intertidal habitat is currently believed to be dominated by benthic microalgae (microphytobenthos) and, to a lesser extent, macroalgae. For such habitat types, Sutula (2011) propose the use of macroalgae to assess eutrophication for the NNE. However, there is a general lack of data on the spatial and temporal variation in biomass, cover, and taxonomic composition of microphytobenthos and macroalgae in SF

Bay. The SF Bay TAT recommends additional data collection establish a baseline of information that can be used to further consider the establishment of a NNE for intertidal flats.

Table 6.1. Summary of review of candidate NNE indicators for San Francisco Bay.

Indicator	Applicable Habitat Type	Met Review Criteria	Comments
Dissolved oxygen	All subtidal	Yes	Wealth of data, existing objectives in SF Water Board basin plan
Phytoplankton biomass, and productivity, and assemblage	All subtidal	Yes	Wealth of data and reliable if used together in multiple lines of evidence. lack of experience predicting phytoplankton assemblages
Harmful algal bloom cell counts and toxin concentration	All subtidal	Three of four	Limited data and understanding of controls on HAB occurrence and toxin production, ***with exception of cyanobacteria***
Ammonium	All subtidal	Three of Four	Evidence of ammonium inhibition of diatom nitrate assimilation, but ecological importance of process not well understood for entire Bay.
Urea	All subtidal	Three of Four	Causative link between urea and HABs has not been established and little data are available on urea concentrations in SF Bay.
Light attenuation	Seagrass, All subtidal	Two of Four	Phytoplankton biomass a component of light attenuation, but other factors unassociated with eutrophication can contribute (e.g., suspended sediment), so poor linkage to BUs and signal:noise
Macroalgae biomass and cover	Seagrass, intertidal flats, muted subtidal	Yes	Insufficient data on macroalgal blooms on intertidal flats and seagrass habitat.
Epiphyte load	Seagrass habitat	Three of four	Epiphyte load difficult to quantify. Use in conjunction with macroalgal biomass/cover and phytoplankton biomass.
Macrobenthos taxonomy, abundance and biomass	Subtidal	Three of four	Macrobenthos not uniquely affected by eutrophication; lack of experience predicting community measures as a function of nutrient loads. Additional research required. May be used as a supporting indicator or co-factor in so much as it could part of a routine sediment quality ambient monitoring

6.2 Recommended Primary, Supporting Indicators to Develop a NNE Diagnostic Assessment Framework for San Francisco Bay

Within the regulatory context, waterbody assessments are made in order to make a determination of whether the waterbody is meeting beneficial uses or impaired, as an example, for nutrients or to set TMDL numeric targets. In this context, a diagnostic assessment framework is the structured set of decision rules and guidance for interpretation that helps to classify the waterbody in categories of minimally disturbed, to moderately disturbed, to very disturbed. Although scientists can provide guidance and data synthesis to illustrate how the assessment framework could be formed, ultimately the decision of what levels to set thresholds (i.e., NNEs) that separate the categories (e.g., minimally versus moderately and very disturbed) is a policy decision.

Development of the diagnostic assessment framework begins by choosing indicators that would be measured and used to determine waterbody status. It is important to distinguish between three types of indicators for an NNE assessment framework: 1) Primary indicators, 2) Supporting indicators, and 3) Co-factor indicators required for data interpretation.

Primary indicators are those for which regulatory endpoints could be developed. Designation of these indicators as “primary” implies a higher level of confidence in these indicators to be used to make an assessment of adverse effects, based on a wealth of experience and knowledge about how this indicator captures and represents ecological response. Primary indicators are those which met all explicit criteria (see Section 2.5) established to evaluate candidate NNE indicators. Supporting indicators are those which could be collected to provide supporting lines of evidence, but would not be used alone to make determination of whether the waterbody was meeting beneficial uses. These indicators may have met many, though not all evaluation criteria, but are considered important because they are commonly used to assess eutrophication. Use of the indicator as supporting evidence over time may increase confidence and cause it to be promoted to “primary.” Finally, co-factors are indicators that could be part of a routine monitoring program and important for data interpretation and trends analysis.

The SF Bay TAT has recommended a suite of “primary” (those for which regulatory endpoints would be developed), “supporting” (those for which no regulatory endpoints would be developed in the near term, but could be used as supporting lines of evidence) and “co-factors” (data that would be used for interpretation of primary and supporting indicators) for four major habitat types in SF Bay. The real distinction between “primary” and “supporting” and how these classes of indicators would be used as multiple lines of evidence in an NNE assessment is entirely dependent on indicator group and particular applications to specific habitat types.

San Francisco Bay is dominated by unvegetated subtidal habitat. In this habitat type, phytoplankton dominate primary production. Therefore, the primary NNE indicators to assess eutrophication in SF Bay are measures of phytoplankton (biomass, productivity, taxonomic composition and abundance), algal toxin concentration and dissolved oxygen. In the designation of phytoplankton indicators (biomass, productivity, and assemblage) as primary, the SF Bay TAT stresses the importance of using these indicators together in as multiple lines of evidence, as use of any one alone is likely to be insufficiently robust. We use the term algal here to remain broad and acknowledge that cyanobacteria do not constitute a significant percentage of the phytoplankton community within the geographic scope of the review. However, there is an expressed intent to capture potential adverse effects of cyanobacteria that may be transported downstream from the Delta as whole cells or as toxin.

Seagrass habitat found in subtidal and intertidal areas of the Bay are an important habitat, though currently a minor percentage of Bay habitats (approximately 1.3% of 1200 km^2). The primary NNE indicators for seagrass habitat represent a combination of factors that result in reduced light attenuation to the bed, resulting in reduced photosynthetic activity by the plants. Thus phytoplankton biomass, macroalgal biomass and percent cover and epiphyte load are the primary NNE indicators for this habitat type.

Muted intertidal and subtidal habitat also represent an important but minor habitat type, representing an additional 269 km^2 in addition to the 1200 km^2 of habitat subjected to full tidal action these areas include salt ponds, duck clubs, and diked Baylands, many of which are actively managed and undergoing active change through restoration. The primary NNE indicators include macroalgal biomass and cover, which are found in both intertidal and subtidal habitat, and measures of dissolved oxygen and phytoplankton (biomass, productivity, HAB species abundance and toxin concentration), which applies to subtidal habitat only.

6.3 Is San Francisco Bay Currently Experiencing Eutrophication Based on These Indicators?

Ultimately, the conclusion of whether SF Bay is adversely affected by the consequences of nutrient over enrichment is made by the SFRQCB and the SWRCB. This review does not attempt to assume that responsibility, but rather summarizes relevant information on the topic based on available data and peer-reviewed articles. The review utilized available data to summarize spatial and temporal trends in the candidate NNE indicators for the Bay (Table 6.3). Brief synopsis of the review is provided below for each of the three habitat types for NNE development is recommended (subtidal, seagrass and muted subtidal).

Table 6.3. Summary of available data sets with which to assess eutrophication. Unvegetated and vegetated (seagrass) subtidal habitats and intertidal habitats are not tidally muted.

Habitat Type	Phytoplankton	Dissolved Oxygen	Nutrients (including ammonium & urea)	Macroalgae & epiphytes
Un-vegetated subtidal habitat	40 years of water quality data from USGS research cruises with some gaps depending on research questions in any one year. Data include phytoplankton biomass, productivity, and taxonomic composition as well as DO and nutrients[6]. The IEP Environmental Monitoring Program (EMP) for the Sacramento-San Joaquin Delta, Suisun Bay, and San Pablo Bay has 40 years of data phytoplankton, zooplankton, and benthos, and water quality. Little data available on water column or tissue phytotoxin concentrations, with exception of cyanobacteria from IEP-EMP in North Bay. Little data available on urea concentrations.			N/A
Seagrass habitat	Limited data available through CALTRANS mitigation monitoring (ended in 2010)	No data available	No data available	Limited data from via recent SFSU monitoring
Intertidal habitat	Not applicable	Not applicable	No data available	No known data
Muted Tidal Habitat (managed ponds)	Limited data available through recent monitoring by USGS			No known data

[6] Note that USGS started measuring chlorophyll in 1977 and DO has not been measured consistently (e.g., not in early 1980's). Nutrients are not measured on every cruise and there are big gaps.

Table 6.2. Table of proposed primary and supporting NNE indicators by habitat type. Primary indicators are those for which regulatory endpoints could be developed. Supporting indicators are those which could be collected to provide supporting lines of evidence. Co-factors are indicators that could be part of a routine monitoring program and important for data interpretation and trends analysis. The list of co-factor indicators is provided as an example and not exhaustive. Note that primary and supporting indicators recommended for unvegetated subtidal habitat are also applicable for seagrass habitat.

Habitat	Primary Indicators	Supporting Indicators	Co-Factors
All Subtidal Habitat	Phytoplankton biomass, productivity, and assemblage Cyanobacteria cell counts and toxin concentration Dissolved oxygen	Water column nutrient concentrations and forms[7] (C, N, P, and Si) HAB species cell count and toxin concentrations	Water column turbidity, pH, conductivity, temperature, light attenuation Macrobenthos taxonomic composition, abundance and biomass Sediment oxygen demand Zooplankton
Seagrass Habitat	Phytoplankton biomass Macroalgal biomass & cover Dissolved oxygen	Light attenuation, suspended sediment conc. Seagrass areal distribution and cover Epiphyte load	Water column pH, temperature, conductivity Water column nutrients
Intertidal Flats	Macroalgal biomass and cover	Sediment % OC, N, P and particle size Microphytobenthos biomass (benthic chl a)	Microphytobenthos taxonomic composition
Muted Intertidal and Subtidal	Macroalgal biomass & cover Phytoplankton biomass Cyanobacteria toxin concentration	Sediment % OC, N, P and particle size Phytoplankton assemblage Harmful algal bloom toxin concentration	Water column pH, turbidity, temperature, conductivity Water column nutrients

[7] Forms referred to relative distribution of dissolved inorganic, dissolved organic, and particulate forms of nutrients, including urea

6.3.1 Phytoplankton, Dissolved Oxygen, and Nutrients in Subtidal Habitats

Cloern and Dugdale (2010) provide an excellent synthesis of the status of eutrophication in the subtidal habitats of SF Bay, based on available phytoplankton, dissolved oxygen and nutrient data in SF Bay. Our review largely restates their observations and conclusions.

San Francisco Bay is a nutrient-enriched estuary, receiving external loads of N and P comparable to Chesapeake Bay (Cloern and Dugdale, 2010). However, dissolved oxygen is much higher and phytoplankton biomass and productivity is lower than would be expected, implying that phytoplankton dynamics and ultimately eutrophication are driven by processes other than nutrient-limitation of primary production. However, all regions of the SF Bay system, from Suisun to South Bay, have experienced significant increases in phytoplankton biomass since the late 1990's (e.g., Cloern et al., 2007, 2010). Recent analysis of water quality data collected by USGS from 1978 to 2009 show of a significant increase in water column chlorophyll a per decade (30-50% per decade from Suisun to South Bay respectively) and a significant decline in DO concentrations (1.6 to 2.5% in South Bay and Suisun Bay respectively ; J. Cloern, personal communication March 2011). Thus evidence is building that the historic resilience of SF Bay to the harmful effects of nutrient enrichment is weakening.

A synthesis of research (Cloern and Dugdale, 2010) has pointed to a number of factors which can be controlling phytoplankton production in the Bay: 1) high turbidity that constrains phytoplankton productivity, 2) top-down control by benthic suspension feeders, 3) salinity stratification that promotes blooms by stationing cells in a high-light and high-nutrient surface layer and isolates them from benthic consumers, 4) connectivity to the Ocean which provides a source of phytoplankton cells that seed blooms, 5) the necessity of approx. 5 days of favorable irradiance for bloom initiation and interactions between diel light and semi-diurnal tidal cycles which control available light to sustain blooms, 6) currents that transport phytoplankton between habitats that function as a net source or sink of algal biomass, and 7) importance of high ammonium inputs in reducing the frequency and intensity of spring blooms through inhibition of nitrate uptake in the lower Sacramento River and Suisun Bay. Temporally, data show that phytoplankton productivity is higher and is increasing in Central Bay, San Pablo Bay and Suisun Bay since the mid 1990s. The causes for the Bay wide trends include changes in water clarity due to less suspended sediment (Schoellhamer, 2009), lower metal inhibition due to improvements in wastewater treatment, increased seeding from ocean populations (Cloern et al., 2005), declines in consumption by bivalves due to increases in predation by juvenile English sole and speckled sanddabs, and declines in phytoplankton consumption by bivalves and zooplankton due to recent new invasive species introductions (Cloern et al., 2006). Data suggest that primary productivity in Suisun Bay is limited by strong grazing pressure by the invasive clam Corbula amurensis (Alpine and Cloern 1992), light limitation by high turbidity (Cloern, 1999), and undetermined chemical inhibition, (Cloern and Dugdale 2010).

San Francisco Bay contains over 500 phytoplankton taxa. Diatoms (Bacillariophyta) dominate the biomass making up 81% of the total cumulative biomass; dinoflagellates and cryptophytes (Pyrrophyta and Cryptophyta) made up 11 and 5%, respectively (Cloern and Dufford, 2005). Despite the persistent nutrient enriched status of SF Bay, few HABs have been reported recently in SF Bay, apparently because nutrient enriched turbid conditions in the estuary favor diatoms associated with new inputs of nutrients as opposed to nutrient regeneration (Cloern, 1996; Ning et al., 2000). However, there have been occasional historical occurrences (see Cloern et al., 1994 referenced in Cloern, 1996), and recently cyanobacteria and dinoflagellate blooms have been documented.

Low oxygen events were a common occurrence in the 1960s, but since the introduction of secondary wastewater treatment in the 70s, events of low oxygen (<5 mg L^{-1}) have been rare. Currently, SF Bay is not experiencing periods of hypoxia and is currently meeting Basin Plan objectives for dissolved oxygen. Water column oxygen variability is controlled by density/thermal stratification and results in higher oxygen concentrations in the upper portions of the water column. Isohaline conditions prevail during summer due to strong wind and tidal forcing which results in a mixed water column with less vertical variability in oxygen levels (Conomos et al., 1979). Decreased dissolved oxygen generally occurs in the lower portions of the water column during periods of stratification when there is high availability of organic matter (Diaz, 2001). Adequate dissolved oxygen levels are most likely maintained due to a quick dissipation of blooms from physical forcing of the tides and winds mixing the water column. There is little information for SF Bay on dissolved oxygen concentrations during other HAB episodes. That said, it seems feasible that if the increased production trend continues or if the incidence of harmful algae blooms increases, there might be a concomitant increase in the frequency of lower DO events especially if they happen to coincide with neap tides or lower wind conditions which can lead to stratification. Recent analysis of DO data collected by USGS from 1978 to 2009 indicates that DO concentrations have significantly declined on the order of 1.6 to 2.5% per decade in South Bay and Suisun Bay respectively (J. Cloern, personal communication March 2011).

6.3.2 Phytoplankton, Macroalgae and Epiphytes in Seagrass Habitat

Very limited data exist on symptoms of eutrophication in seagrass habitats in SF Bay. Some beds have been documented to have persistent macroalgal biomass and cover, but data are inadequate to make an assessment of effects and it is unclear as the sources of nutrients responsible for maintaining macroalgal blooms in this area (K. Boyer, personal communication). There was consensus by SF Bay TAT members that seagrass is currently limited by turbidity associated with suspended sediments rather than a combination of phytoplankton biomass, macroalgae and/or epiphytes.

6.3.3 Macroalgae and Microphytobenthos in Intertidal Flats

No assessment of current status of eutrophication on intertidal flats can be made because of limited data.

6.3.4 Dissolved Oxygen, Macroalgae and Phytoplankton in Muted Intertidal and Subtidal Habitats

Data that exist on symptoms of eutrophication in muted subtidal habitats (such as estuarine diked Baylands and restored salt ponds) are a result of monitoring associated with restoration or special studies conducted by USGS. Utility and applicability of DO to diked Baylands requires additional discussion, particularly because muted habitats are to some degree subject naturally to hypoxia. In addition, these habitat types are known to influence subtidal DO. Low DO water can exist in salt ponds that, if breached, can supply high organic discharge or low DO discharge to the Bay (Shellenbarger et al., 2008; Thebault et al., 2008). Limited monitoring data of phytoplankton biomass and community composition of these habitats are insufficient to make an assessment of current status of eutrophication in these systems, although it is established that some managed ponds in lower South Bay harbor species of toxin-producing phytoplankton (Thebault et al., 2008). Data on macroalgal biomass and cover in these habitat types are not known to exist.

6.4 What Are the Nutrient Loads to San Francisco Bay From Various Sources and How Are These Loads Changing over Time?

San Francisco Bay is regarded as a nutrient enriched estuary, based on the ambient concentrations and estimated loads of nutrients to the Bay (Cloern and Dugdale, 2010). Nutrients loads from external sources and pathways are poorly understood, though data exist with which to improve published load estimates from some sources. For the most part, published load estimates are outdated by one or even two decades and were either based on data collection methods that were not designed for loads estimation, were based on assumptions that provided guesses about loads at best or were based on data sets that have now been substantially improved with ongoing collection through time. Given changes to wastewater treatment technologies, increases in population, changes to land use, etc., nutrient loads have likely changed over the past four decades.

6.5 Data Gaps and Recommended Next Steps

The SF Bay NNE framework consists of two principle components: 1) primary and supporting indicators used in an assessment framework to assess eutrophication of SF Bay habitats and 2) models that link these indicators back to nutrient loads and other controlling factors that mitigate the ecological response to eutrophication. A set of data gaps and recommended next steps are recognized for both of these components of the SF Bay NNE framework.

The development and use of an NNE framework for SF Bay is completely contingent on the continued availability of monitoring data to formulate, test and periodically assess the status of the Bay with respect to eutrophication. Over the past forty years, the USGS has conducted a research program in the subtidal habitat of SF Bay, with partial support by the SF Bay Regional Monitoring Program (RMP). This USGS research program cannot be considered replacement for a regularly funded monitoring program. **The SF Bay TAT strongly recommends that a nutrients/eutrophication monitoring strategy be developed and funded for successful development and implementation of the NNE in SF Bay. This program should be coordinated and complementary to the IEP Environmental Monitoring Program that terminates in San Pablo Bay. The Regional Monitoring Program for Water Quality in San Francisco Bay is currently in the early stages of developing a "nutrient strategy." Efforts to develop the NNE assessment framework should be coordinated with the SF Bay RMP.**

6.5.1 Data Gaps and Recommended Next Steps for Development of a San Francisco Bay NNE Assessment Framework

Development of an NNE assessment framework for SF Bay involves specifying how primary and supporting indicators would be used as multiple lines of evidence to diagnose adverse effects of eutrophication in SF Bay. While development of the assessment framework begins as a scientific or technical work element, ultimately the selection of thresholds that would be used to determine whether the Bay is meeting beneficial uses is a policy decision. That policy decision is made by the SWRCB and the SF Water Board, with advice from its advisory groups (the SF Bay SAG and the STRTAG).

Assessment frameworks would need to be created for the habitat types identified in this review: 1) subtidal habitat, 2) seagrass habitat, 3) intertidal flats, and 3) muted intertidal and subtidal habitat. The Table 6.4 summarizes data gaps and recommended next steps for development of a SF Bay NNE assessment framework by habitat type. **Note that no attempt is made to prioritize or reduce/eliminate "next steps" any habitat types, despite acknowledged limitation in available resources. The SF Bay TAT assumes this prioritization and focusing of resources would be done by the SWRCB, the SF Water Board, with advice from its advisory groups.**

Unvegetated Subtidal Habitats. For unvegetated subtidal habitats in SF Bay, a long term data set on primary indicators such as dissolved oxygen, phytoplankton biomass, productivity, and taxonomic composition exist, albeit with some data gaps and poor information on the prevalence of HAB toxins in SF Bay. Adequate understanding exists of factors controlling long-term temporal and spatial trends in these indicators. The SF Bay TAT recommends that development of an NNE assessment framework for this habitat type proceed by:

- Sponsoring a series of expert workshops to develop a draft assessment framework based on phytoplankton biomass, productivity, cyanobacteria cell counts and toxin concentrations, and dissolved oxygen as the primary indicators

- Augmenting baseline monitoring data on HAB toxin concentrations

Use of supporting indicators such as ammonium, urea, and phytoplankton taxonomic composition or assemblages, and HAB species cell abundance and toxins in the NNE assessment framework would be greatly benefited by additional work and, in some cases, baseline data collection. This work includes:

- Formation of a workgroup of SF Bay scientists and outside expertise to develop indices of Bay health based on measures of phytoplankton taxonomic composition or assemblages.
- Formulating of a working group of SF Bay scientists to synthesize available data on factors known to control primary productivity in different regions in the Bay, developing consensus on relative importance of ammonium inhibition of phytoplankton blooms to Baywide primary productivity, and determining next steps with respect to incorporating ammonium into the NNE assessment framework for SF Bay.
- Augment USGS water quality data collection with sampling urea, HAB cell counts and toxin concentrations (water and faunal tissues, for a minimum of two years.

Table 6.4. Indicator status and recommended next steps for development of an NNE Assessment framework for San Francisco Bay.

Habitat	Indicator	Indicator Designation	Data Gaps	Recommended Next Steps
Subtidal Habitat	Dissolved oxygen	Primary	Wealth of data exist. TAT does not have expertise to review adequacy of DO objectives. Review did not address dissolved oxygen data in the tidally muted habitats of SF Bay. Additional analysis of existing data and consideration of scientific basis for DO objectives in these habitats is warranted.	Consider update of science supporting Basin Plan dissolved oxygen objectives, if warranted by additional review by fisheries experts. Review could be for entire Bay or limited to the tidally muted areas of the Bay.
	Phytoplankton biomass, productivity, and taxonomy	Primary.	Wealth of data exist. Need a review of science supporting selection of endpoints. Additional work required to improve ability to predict phytoplankton assemblage.	Recommend development of a white paper and a series of expert workshops to develop NNE assessment framework for phytoplankton biomass, productivity, taxonomic composition/assemblages, abundance and/or harmful algal bloom toxin concentrations. Recommend augmentation of current monitoring to include measurement of HAB toxin concentrations in water and faunal tissues.
	Harmful algal bloom abundance and toxin conc.	Cyanobacteria cell counts and toxin = primary; HAB cell counts and toxin = supporting	Little data on HAB toxin concentrations in surface waters and faunal tissues.	
	Ammonium and urea	Supporting	Lack of understanding of relative importance of ammonia limitation of nitrate uptake in diatoms on Bay productivity vis-à-vis other factors Lack of data on urea concentrations in SF Bay	Recommend formulation of a working group of SF Bay scientists to synthesize available data on factors known to control primary productivity in different regions in the Bay and evaluate potential ammonium endpoints. Recommend collecting data on urea concentrations in SF Bay over a two year period.
	Macrobenthos taxonomy, abundance and biomass	Co-factor	IEP-EMP has data on macrobenthos, but data lack in regions south of San Pablo Bay; lack of information on how to use combination of taxonomy, abundance, and biomass to assess eutrophication.	Recommend utilization of IEP-EMP data to explore use of macrobenthos to assess eutrophication in oligohaline habitats. Consider including biomass in the protocol to improve diagnosis of eutrophication. Determine whether combination of indicators can be used reliably to diagnose eutrophication distinctly from other stressors.

Seagrass Habitat	Phytoplankton biomass, epiphyte load and light attenuation	Phytoplankton biomass = primary, epiphyte load and light attenuation = secondary	Poor data availability of data on stressors to SF Bay seagrass beds. Studies needed to establish light requirements for species of seagrass found in SF Bay and studies to assess duration of reduced light/photosynthesis that results in adverse effects to the seagrass bed.	Recommend 1) Continued monitoring of aerial extent every 3-5 years (currently no further system scale monitoring is planned beyond 2010), 2) studies to establish light requirements for SF Bay seagrass species, 3) development of a statewide workgroup to develop an assessment framework for seagrass based on phytoplankton biomass, macroalgae, and epiphyte load and 4) collection of baseline data to characterize prevalence of macroalgal blooms on seagrass beds. Studies characterizing thresholds of adverse effects of macroalgae on seagrass currently underway in other California estuaries should be evaluated for their applicability to SF Bay.
	Macroalgae biomass and cover	Primary	Data gaps include studies to establish thresholds of macroalgal biomass, cover and duration that adversely affect seagrass habitat	
Intertidal Flat Habitat	Macroalgal biomass and cover	Primary	Lack of baseline data on frequency, magnitude (biomass and cover) and duration of macroalgal blooms in these intertidal flats	Recommend collection of baseline data on macroalgae , microphytobenthos and sediment bulk characteristics. Recommend inclusion of SF Bay scientists and stakeholders on statewide workgroup to develop an assessment framework for macroalgae on intertidal flats
	Sediment % OC, N, P and particle size	Supporting		
	MPB taxonomic composition and benthic chl a biomass	Supporting		

Habitat	Indicator	Indicator Designation	Data Gaps	Recommended Next Steps
Muted Habitat	Macroalgae	Primary indicator	Lack of baseline data on frequency, magnitude (biomass and cover) and duration of macroalgal blooms in muted habitat types	Recommend collection of baseline data on macroalgae, dissolved oxygen, phytoplankton biomass, taxonomic composition and HAB species/toxin concentration in these habitat types Recommendation to develop an assessment framework based on macroalgae, phytoplankton and dissolved oxygen in these habitat types. One component of this discussion should be a decision on beneficial uses that would be targeted for protection and to what extent the level of protection or expectation for this habitat type differ from adjacent subtidal habitat.
	Phytoplankton biomass and community composition	Primary indicator	Lack of baseline data on phytoplankton biomass and community composition in these habitat types	
	Dissolved oxygen	Primary indicators	Some data on dissolved oxygen exist. Unclear about what levels of DO are required to protect beneficial uses of muted habitats.	
	Phytoplankton taxonomy, abundance, and/or harmful algal bloom toxin conc.	Cyanobacteria cell counts and toxin = primary; 'taxonomic composition/assemblage and HAB cell counts and toxin = supporting	Little data on taxonomic composition, HAB toxin concentrations in surface waters and faunal tissues.	

Over time, work on macrobenthos may show that this component of the Bay ecosystem could provide useful information on eutrophication. However, science in this area is evolving and sufficient evidence does not exist to use it as a primary or supporting indicator. The SF Bay recommends including it as a co-factor.

Seagrass Habitat. For seagrass habitat, specific recommendations to develop a NNE assessment for seagrass include: 1) studies to establish light requirements for SF Bay seagrass species; 2) collection of baseline data to characterize prevalence of macroalgal blooms and other stressors on seagrass beds; and 3) inclusion of SF Bay scientists and stakeholders in statewide group to develop an assessment framework for eutrophication in seagrass, based on phytoplankton biomass, macroalgae, and epiphyte load.

It should be noted that studies characterizing thresholds of adverse effects of macroalgae on seagrass currently underway in other California estuaries. The findings of these studies should be evaluated for their applicability to SF Bay.

***Intertidal Flat Habitat**.* For intertidal flat habitat, specific recommendations to develop a NNE assessment include: 1) development of a NNE assessment framework based on macroalgae and 2) collection of baseline data to characterize frequency, magnitude and duration of macroalgal blooms on SF Bay intertidal flats.

It should be noted that studies characterizing thresholds of adverse effects of macroalgae on intertidal flats currently underway in other California estuaries. The findings of these studies should be evaluated for their applicability to SF Bay. San Francisco Bay scientists and stakeholders should be included in statewide group to develop an assessment framework for macroalgae on intertidal flats.

***Muted Intertidal and Subtidal Habitat**.* While some data on dissolved oxygen and HAB species cell counts exist for muted intertidal and subtidal habitat, there is a general lack of extended baseline data on these and other indicators (macroalgal biomass and cover and phytoplankton biomass, taxonomic composition, and HAB toxin concentrations) in these habitats needed to make a full assessment of eutrophication. Finally, it is recommended that SF Bay scientists and stakeholders be included in the statewide effort to develop an assessment framework for muted habitats, based on macroalgae, dissolved oxygen, phytoplankton measures, and HAB toxin concentrations.

6.5.2 Data Gaps and Recommended Next Steps to Quantify External Nutrient Loads

Table 6.5 provides a summary of data gaps and recommended next steps. Recommendations generally fall into two categories: 1) Revising and updating estimates of nutrients from the different sources, based on existing data; and 2) Identification of data needed to develop a dynamic loading model.

The exercise of revising and updating estimates of nutrients from the various sources, based on existing data would help to better inform our understanding of the dominant nutrient sources for each distinct region of the Bay. This would, in turn, assist in decision-making to prioritize new data collection to develop the loading model(s) (discussed further below).

Table 6.5. Summary of data gaps and recommended next steps for quantification of external nutrient loads to San Francisco Bay.

Source	Data Gaps Identified	Recommended Next Steps
Atmospheric Deposition	No recently published data on wet & dry atmospheric deposition	Loads likely relatively small. Literature review to determine range of N and P deposition rates for West Coast coastal urban areas. Recommend baseline atmospheric deposition monitoring of wet and dry N and P deposition over 1-2 yr period to better constrain estimates.
Terrestrial Loads from Delta	Data available through RMP on dry season concentrations. No data available on wet weather concentrations during storm flow	Loads likely large. Recommend analysis of existing RMP data to estimate dry season nutrient loads. It is recommended that wet weather data collection of nutrients be initiated at the DWR sampling location at Mallard Island at the head of Suisun Bay to support improved daily loads estimates for 1995-present.
Municipal Effluent	Data available through 15 of approx. 40 POTWs	Loads likely large. Synthesize nutrient discharge and concentration data to estimate loads over period of last 10-20 years Encourage all treatment plants that discharge to the Bay to begin analyzing effluent for total and dissolved inorganic nutrients and to submit these data to the SFRWQCB on a regular basis. Recommend that the POTWs conduct a laboratory inter-comparison on nutrient methods to assure comparability of estimates.
Industrial Effluent	Some data available from the 1990s. Recent data availability unknown	Loads likely small relative to municipal wastewater. Synthesize available data to provide information for prioritization of any future steps.
Stormwater	Some data available but general lack of wet weather data sufficient to calibrate and verify a dynamic loading model	Loads likely large. Synthesize data to provide an updated estimate of stormwater contributions to assist prioritization of next steps. Scope the data needs associated with the development of a dynamic loading model.
Groundwater	Some data available from 79 USGS monitoring stations surrounding the Bay. Flow data currently less well understood.	Loads likely small. Refine current loads estimates after review of local USGS groundwater experts in order to support prioritization of next steps if any.
Exchange with Coastal Ocean	Some data available for fluxes of water and sediments during selected tides and seasons in the past decade collected by USGS and UC Berkeley using comparable methods.	Initiate a workgroup of local experts to design a sampling program for nutrient flux at the Golden Gate boundary, with the intent of developing a hydrodynamic and material flux dynamic model to describe exchange with coastal ocean

6.5.3 Data Gaps and Recommended Next Steps for Development of Load-Response Models

An important component of implementing the NNE framework in SF Bay is the development of load-response models that can simulate the ecological response of the Estuary to nutrients and other important co-factors.

Models developed to manage nutrient loads and eutrophication in the Chesapeake Bay Estuary (http://archive.chesapeakebay.net/pubs/backgrounder_CBP_Models.pdf) are one example of linked watershed loading and receiving water models. These models were developed and refined through a 30 years of collaboration by federal, state, academic and private partners. Several types of models would need to be developed that fit into two general categories:

- Air, Oceanic and Watershed Loading Model, which could incorporate information about land use, fertilizer applications, wastewater plant discharges, septic systems, air deposition, farm animal populations, weather and other variables to estimate the amount of nutrients and sediment reaching the SF Bay estuary and where these pollutants originate. The loading model would include three components: 1) a hydrologic sub-model, 2) a non-point source sub-model, 3) a river sub-model which routes flow and associated nutrient loads to the Estuary, and 4) an ocean exchange model.
- Estuary water quality model, which simulates the ecosystem response to pollutant loads, and consists of two sub-models: 1) a hydrodynamic sub-model that will simulate the mixing of waters in the Estuary and its tidal tributaries and 2) a water quality sub-model that simulates the Estuary's biological, chemical and physical dynamics in response to nutrient loads from the watershed, air and ocean and other factors (light, temperature, grazing, etc.).

The models would be used to establish load allocations of nutrients that the SF Bay estuary can sustainably assimilate. It would also be used to generate simulations of the past, present or future state of the Estuary, watershed, airshed, and ocean (e.g., population growth, climate change, etc.) to explore potential effects of management actions and evaluate alternatives. Thus these models would be a key component of a strategy to adaptively manage SF Bay.

Ideally sufficient data and knowledge of SF Bay should exist to support the development of system wide dynamic simulation models to predict phytoplankton biomass/community response and relationships to models of secondary productivity. This goal is not likely in the short term, so it is important to consider that the development of a more complex model should follow the testing out of key concepts and assumptions in smaller, simpler models.

Scoping the development of these NNE load response models should begin through the iterative development of a modeling strategy and workplan. During this strategy workshop, participants would describe the types of models that would be needed. Sufficient detail would need to be given to accomplish three elements:

- *Nutrient Budget for San Francisco Bay*. This step would utilize existing data to synthesize a nitrogen and phosphorus nutrient budget for SF Bay. Existing data that describe the timing and magnitude of external sources, internal sources, sinks, and pathways of transformation such as benthic nutrient flux, nitrification, denitrification, etc. would be compiled in order to synthesize current understanding of sources and fate of nutrients as well as identify critical data gaps in advance of the modeling strategy development.
- *Conceptual Model Development*. There is a need to develop conceptual models that explicitly show linkage between watershed, airshed, ocean and estuarine hydrology, nutrient loads, ecological response indicators, and "co-factors" that control ecological response to eutrophication or oligotrophication. The conceptual model would identify key sources, sinks and processes of transformation that would need to be incorporated into the numeric simulation models. Areas of disagreement on causal mechanisms should be synthesized as alternative hypotheses than can be tested through experiments, field studies and model sensitivity analyses.
- *Review of Existing Models and Model Selection*. The next step in the scoping of model development is to select the appropriate models. This should be done by reviewing available loading and receiving waterbody models and present an analysis of the advantages and disadvantages of their use for modeling eutrophication and other adverse responses to nutrients, based on the explicit conceptual models. A review of existing hydrodynamic, sediment transport, and water quality models developed for SF Bay and their applications should be undertaken, with the intent of understanding what existing tools may be used to leverage development of load-response models.
- *Data Needs Assessment*. Based on explicit conceptual models and the modeling platform selected, the next step would be to identify data required to support model development, calibration and validation.

The product of this effort would be the a coarse nutrient budget for SF Bay, identification of the appropriate models, data gaps and recommended studies, a phased workplan, timeline and budget to develop these models, and identification of and coordination among key institutions, programs and respectively roles. This information could be synthesized into a workplan to develop the loading and estuary water quality models and a preliminary timeline and budget for Phase I of the effort.

6.5.4 Coordination of Development of the SF Bay NNE Framework with Nutrient Management in the San Joaquin and Sacramento River Delta

Development and implementation of a NNE framework for SF Bay will require improve coordination with nutrient management activities in the San Joaquin and Sacramento River Delta. At the time of writing this report, preliminary discussions on this topic are beginning with the Central Valley Water Board staff. Coordination should be improved, at minimum, with respect to any future monitoring and/or modeling of nutrient loading, transport and source identification, as SF Bay and the Delta exchange nutrients across their aquatic and terrestrial boundaries. Coordination would be further enhanced by a similar review of NNE candidate indicators, summary of existing science, and identification of data gaps and recommended next steps specifically for the Delta.

7. References

Alden, R.W. III, D.M. Dauer, J.A. Ranasinghe, L.C. Scott and R.J. Llansó. 2002. Statistical verification of the Chesapeake Bay benthic index of biotic integrity. *Environmetrics* 13:473-498.

Aller, R.C. and J.Y. Aller. 1998. The effect of biogenic irrigation intensity and solute exchange on diagenetic reaction rates in marine sediments. *Journal of Marine Research* 56:905-936.

Aller, R.C. 1982. The effects of macrobenthos on chemical properties of marine sediment and overlying water. pp. 53-102. In: P. L. McCall and M. J. S. Tevesz (eds.), Animal-Sediment Relations: The Biogenic Alteration of Sediments. Plenum Press, New York.

Alpine, A.E. and J.E. Cloern. 1992. Trophic interactions and direct physical effects control phytoplankton biomass and production in an estuary. *Limnology and Oceanography* 37:946-955.

Alpine, A.E. and J.E. Cloern. 1988. Phytoplankton growth rates in a light-limited environment, San Francisco Bay. *Marine Ecology Progress Series* 44:167-173.

Andersen, F.Ø. and E. Kristensen. 1988. The influence of macrofauna on estuarine benthic community metabolism: a microcosm study. *Marine Biology* 99:591-603.

Anderson, D.M., P.M. Glibert and J.M. Burkholder. 2002. Harmful algal blooms and eutrophication: nutrient sources, composition, and consequences. *Estuaries* 25:704-726.

Attrill, M.J. and S.D. Rundle. 2002. Ecotone or ecocline: Ecological boundaries in estuaries. *Estuarine, Coastal and Shelf Science* 55:929-936.

Attrill, M.J. 2002. A testable linear model for diversity trends in estuaries. *Journal of Animal Ecology* 71:262-269.

Bachelet, G., X. de Montaudouin, I. Auby and P.-J. Labourg. 2000. Seasonal changes in macrophyte and macrozoobenthos assemblages in three coastal lagoons under varying degrees of eutrophication. *ICES Journal Marine Science* 57:1495-1506.

Balducci, C., A. Sfriso and B. Pavoni. 2001. Macrofauna impact on Ulva rigida C. Ag. production and relationship with environmental variables in the lagoon of Venice. *Marine Environmental Research* 52 27-49.

Bates, S.S., A.S.W. DeFreitas, et al. 1991. Controls on domoic acid production by the diatom Nitzschia pungens f. multiseries in culture: Nutrients and irradiance. *Canadian Journal of Fisheries and Aquatic Sciences* 48:1136-1144.

Bates, S.S., C.J. Bird, A.S.W. DeFreitas, R. Foxall, M. Gilgan, L.A. Hanic, G.R. Johnson, A.W. McCulloch, P. Odense, R. Pocklington, M.A. Quilliam, P.G. Sim, J.C. Smith, D.V. Subba Rao, E.C.D. Todd, J.A. Walter and J.L.C. Wright. 1989. Pennate diatom Nitzschia pungems as the primary source of domoic acid, a toxin in shellfish from Eastern Prince Edward Island. *Canadian Journal of Fisheries and Aquatic Science*s 46:1203-1215.

Bates, S.S., A.S.W. DeFreitas, J.E. Milley, R. Pocklington, M.A. Quilliam, J.C. Smith and J. Worms) 1991. Controls on domoic acid production by the diatom Nitzschia pungens f. multiseries in culture: nutrients and irradiance. *Canadian Journal of Fisheries and Aquatic Science* 48:1136-1144.

Batuik, R.A., P. Bergstrom, M. Kemp, E. Koch, L. Murray, J.C. Stevenson, R. Bartleson, V. Carter, N. Rybicki, J.M. Landwehr, C. Gallegos, L. Karrh, M. Naylor, D. Wilcox, K.A. Moore, S. Alistock and M. Teichberg. 2000. Chesapeake Bay Submerged Aquatic Vegetation Water Quality and Habitat-based Requirements and restoration targets: A second technical synthesis. US Environmental Protection Agency, Annapolis, MD. http://www.chesapeakebay.net/publication.aspx?publicationid=13051.

Batuik, R.A., R.J. Orth, K.A. Moore, W.C. Dennison, J.C. Stevenson, L.W. Staver, V. Carter, N. Rybicki, R.E. Hickman, S. Kollar, S. Bieber and P. Heasly. 1992. Chesapeake Bay Submerged Aquatic Vegetation Habitat Requirements and restoration targets: A technical synthesis. U.S. Environmental Protection.

Baustian, M.M. and N.N. Rabalais. 2009. Seasonal composition of benthic macroinfauna exposed to hypoxia in the Northern Gulf of Mexico. *Estuaries and Coasts* 32:975-983.

Bay, S.M., D.J. Greenstein, J.A. Ranasinghe, D.W. Diehl and A.E. Fetscher. 2009. Sediment Quality Assessment Draft Technical Support Manual. Southern California Coastal Water Research Project. Costa Mesa, CA.

BCDC. 2010. San Francisco Bay Subtidal Habitat Goals Project Public Draft 6/16/2010, San Francisco Bay Conservation and Development Commission (BCDC).

Berner, R.A. 1980. Early Diagenesis: A Theoretical Approach: Princeton, NJ (Princeton Univ. Press).

Berry, H.D., A.T. Sewll, S. Wyllie-Echeverria, B.R. Reeves, T.F. Mumford, J.R. Skalski, R.C. Zimmerman and J. Archer. 2003. Puget Sound Submerged Vegetation Monitoring Project: 2000-2002 monitoring report. Washington State Dept. of Natural Resources, Olympia WA. 60 pp. plus appendicies. http://www.dnr.wa.gov/Publications/aqr_nrsh_00_02svmp_rpt.pdf.

Beukema, J.J. and G.C. Cadeé. 1997. Local differences in macrozoobenthic response to enhanced food supply caused by mild eutrophication in a Wadden Sea area: Food is only locally a limiting factor. *Limnology and Oceanography* 42:1424-1435.

Biber, P.D., C.L. Gallegos and W.J. Kenworthy. 2008. Calibration of a bio-optical model in the North River, NC: A tool to evaluate water-quality impacts on seagrasses. *Estuaries and Coasts* 31:177-191.

Björnsäter, B.R. and P.A. Wheeler. 1990. Effect of nitrogen and phosphorus supply on growth and tissue composition of Ulva fenestrata on Enteromorpha intestinalis (Ulvales: Chlorophyta). *Journal of Phycology* 26:603-11.

Blackmon, D., T. Wyllie-Echeverria and D.J. Shafer. 2006. The role of seagrasses and kelps in marine fish support. WRAP Technical Notes Collection (ERDC TN-WRARP-06-1). US Army Engineering Research and Development Center, Vicksburg, MS. http://el.erdc.usace.army.mil/elpubs/pdf/tnwrap06-1.pdf.

Boese, B.L., W.G. Nelson, C.A. Brown, R.J. Ozretich, H. Lee II, P.J. Clinton, C.L. Folger, T.C. Mochon-Collura and T.H. DeWitt. 2009. Chapter 8: Lower depth limit of Zostera marina in seven target estuaries. IN: Lee II, H. and C.A. Brown (eds). Classification of Regional Patterns of Environmental Drivers and Benthic Habitats in Pacific Northwest Estuaries. USEPA, Office of Research and Development, National Health and Environmental Effects Research Laboratory, Western Ecology Division. EPA 600/R-09/140. http://www.epa.gov/wed/pages/publications/authored.htm.

Bolam, S.G. and T.F. Fernandes. 2002. The effects of macroalgal cover on the spatial distribution of macrobenthic invertebrates: the effect of macroalgal morphology. Hydrobiologia 475(1):437-448.

Bolam, S., T. Fernades, et al. 2000. Effects of macroalgal mats on intertidal sandflats: an experimental study. *Journal of Experimental Marine Biology & Ecology* 249:123-137.

Borja, A., I. Muxika and J. Franco. 2006. Long-term recovery of soft-bottom benthos following urban and industrial sewage treatment in the Nervi estuary (southern Bay of Biscay). *Marine Ecology Progress Series* 313:43-55.

Borja, A., J. Franco and V. Pe'rez. 2000. A marine biotic index to establish the ecological quality of soft-bottom benthos within European estuarine and coastal environments. *Marine Pollution Bulletin* 40:1100-1114.

Borja, A., S.B. Bricker, D.M. Dauer, N.T. Demetriades, J.G. Ferreira, A.T. Forbes, P. Hutchings, X.P. Jia, R. Kenchington, J.C. Marques and C.B. Zhu. 2008. Overview of integrative tools and methods in assessing ecological integrity in estuarine and coastal systems worldwide. *Marine Pollution Bulletin* 56:1519–1537.

Bortolus, A., O.O. Iribarne and M.M. Martínez. 1998. Relationship between waterfowl and the seagrass Ruppia maritima in a Southwestern Atlantic coastal lagoon. *Estuaries* 21:710-717.

Bortone, S.A., W.A. Dunson and J.M. Greenawalt. 2005. Fishes as Estuarine Indicators. In S. Bertone (ed.) Estuarine Indicators. Boca Raton Florida: CRC Press, Inc. p381-391.

Boyer, K.E. and S. Wyllie-Echeverria. 2010. Eelgrass Conservation and Restoration in San Francisco Bay: Opportunities and Constraints. Report for the San Francisco Bay Subtidal Habitat Goals Project, Appendix 8-1. http://www.sfbaysubtidal.org/report.html

Boyer, K.E., S. Wyllie-Echeverria, S. Cohen and B. Ort. 2008. Evaluating buoy-deployed seeding for restoration of eelgrass (Zostera marina) in San Francisco Bay: A Final Report Submitted to The NOAA/UNH Cooperative Institute for Coastal and Estuarine Environmental Technology (CICEET). Romberg Tiburon Center for Environmental Studies, San Francisco State University, Tiburon, CA. December 18, 2008. 32pp.

Boyle, K.A., K. Kamer and P. Fong. 2004. Spatial and temporal patterns in sediment and water column nutrients in a eutrophic southern California estuary. *Estuaries* 27:378-388.

Breitburg, D.L., J.K. Craig, R.S. Fulford, K.A. Rose, W.R. Boynton, D.C. Brady, B.J. Ciotti, R.J. Diaz, K.D. Friedland, J.D. Hagy, D.R. Hart, A.H. Hines, E.D. Houde, S.E. Kolesar, S.W. Nixon, J.A. Rice, D.H. Secor and T.E. Targett. 2009. Nutrient enrichment and fisheries exploitation: interactive effects on estuarine living resources and their management. *Hydrobiologia* 629:31-47.

Bricker, S.B., C.G. Clement, et al. 1999. National Estuarine Eutrophication Assessment: Effects of Nutrient Enrichment in the Nation's Estuaries. Silver Springs, MD, NOAA, National Ocean Service, Speical Projects Office and National Centers for Coastal Ocean Science.

Bricker, S.B., J.G. Ferreira and T. Simas. 2003. An integrated methodology for assessment of estuarine trophic status. *Ecological Modelling* 169:39-60.

Bricker, S.B., B. Longstaff, W. Dennison, A. Jones, K. Boicourt, C. Wicks and J. Woerner. 2008. Effects of nutrient enrichment in the nation's estuaries: A decade of change. *Harmful Algae* 8:21-32.

Brown, C.A. and R.J. Ozretich. 2009. Coupling between the coastal ocean and Yaquina Bay, Oregon: Importance of oceanic inputs relative to other nitrogen sources. *Estuaries and Coasts* 32:219-237.

Brown, C.A., W.G. Nelson, B.L. Boese, T.H. DeWitt, P.M. Eldridge, J.E. Kaldy, H. Lee II, J.H. Power and D.R. Young. 2007. An approach to developing Nutrient Criteria for Pacific Northwest Estuaries: A Case Study of Yaquina Estuary, Oregon. USEPA Office of Research and Development, National Health and Environmental Effects Laboratory, Western Ecology Division. EPA/600/R-07/046. http://www.epa.gov/wed/pages/publications/authored.htm

Brylawski, B.J. 2008. Cultural Eutrophication and the Clam *Macoma balthica*: evidence for trophic disruption and effects on blue crabs. Gloucester Point, VA, The College of William and Mary. Ph.D Dissertation.

Burkholder J., E. Noga, C. Hobbs and H. Glasgow. 1992. New 'phantom' dinoflagellate is the causative agent of major estuarine fish kills. *Nature* 358:407-410.

Burkholder, J.M., D.A. Tomasko and B.W. Touchette. 2007. Seagrasses and eutrophication. *Journal Experimental Marine Biology Ecology* 350:46-72.

Cabral, J., M. Pardal, et al. 1999. The impact of macroalgal blooms on the use of the intertidal area and feeding behavior of waders (Charadrii) in the Mondego estuary (west Portugal)." Acta Oecologica 20(4):417-427.

Cabral, J., M. Pardal, R. Lopes, T. Murias and J. Marques. 1999. The impact of macroalgal blooms on the use of the intertidal area and feeding behavior of waders (Charadrii) in the Mondego estuary (west Portugal). *Acta Oecologica* 20:417-427.

Caffrey, J. 1995. Spatial and seasonal patterns in sediment nitrogen remineralization and ammonium concentrations in San Francisco Bay, California. *Estuaries and Coasts* 18:219-233.

Calle-Delgado, K.P. 2007. Tolerance of Tidal Creek Macrobenthic Organisms to Multiple Stressors: Implications on Distributional Patterns. Columbia, SC, University of South Carolina. Ph.D. Dissertation.

California Irrigation Management Information System (CIMIS), Department of Water Resources, downloaded data on 4/23/09, http://www.cimis.water.ca.gov/

Cardoso, P.G., M.A. Pardal, et al. 2004. Macroinvertebrate response to different species of macroalgal mats and the role of disturbance history. *Journal of Experimental Marine Biology and Ecology* 308:207-220.

Carlton, J.T. 1979. Introduced Invertebrates of San Francisco Bay. San Francisco Bay: The Urbanized Estuary.

Carpenter, K.E., J.M. Johnson and C. Buchanan. 2006. An index of biotic integrity based on the summer polyhaline zooplankton community of the Chesapeake Bay. *Marine Environmental Research* 62:165-180.

Carr, L.A., K.E. Boyer and A. Brooks. 2011. Spatial patterns in epifaunal community structure in San Francisco Bay eelgrass (*Zostera marina*) beds. *Marine Ecology* 32:88-103.

Chetelat, J., F. Pick, et al. 1999. Periphyton biomass and community composition in rivers of different nutrient status. *Canadian Journal of Fisheries and Aquatic Sciences* 56:560-569.

Christman, C.S. and D. Dauer. 2003. An approach for identifying the causes of benthic degradation in Chesapeake Bay. *Environmental Monitoring and Assessment* 81:187-197.

Cloern, J.E.. 1979. Phytoplankton ecology of San Francisco Bay: The status of our current understanding, in Conomos, T.J., ed., San Francisco Bay: The Urbanized Estuary: Pacific Division, AAAS, San Francisco, page 247-264.

Cloern, J.E. 1982. Does the benthos control phytoplankton biomass in South San Francisco Bay (USA)? *Marine Ecology Progress Series* 9:191–202.

Cloern, J.E. 1991. Tidal stirring and phytoplankton bloom dynamics in an estuary. *Journal of Marine Research* 49:203-221.

Cloern, J.E. 1996. Phytoplankton bloom dynamics in coastal ecosystems: a review with some general lessons from sustained investigation of San Francisco Bay, California. *Reviews of Geophysics* 34 (2):127-168.

Cloern, J.E. 1999. The Relative Importance of Light and Nutrient Limitation of Phytoplankton Growth: A Simple Index of Coastal Ecosystem Sensitivity to Nutrient Enrichment. *Aquatic Ecology* 33:3-15.

Cloern, J.E. 2001. Our evolving conceptual model of the coastal eutrophication problem. *Marine Ecology Progress Series* 210:223–253.

Cloern, J.E. 2007. Habitat connectivity and ecosystem productivity: Implications from a simple model. *The American Naturalist* 169, E21-E33.

Cloern, J.E. and F.H. Nichols. 1985. Time scales and mechanisms of estuarine variability, A synthesis of studies from San Francisco Bay. *Hydrobiologia* 129:229-237.

Cloern, J.E. and A.D. Jassby. 2010. Patterns and scales of phytoplankton variability in estuarine-coastal ecosystems. *Estuaries and Coasts* 33:230-241.

Cloern, J.E. and R. Dufford. 2005. Phytoplankton community ecology: principles applied in San Francisco Bay. *Marine Ecology Progress Series* 285:11-28.

Cloern, J.E. and R.S. Oremland. 1983. Chemistry and microbiology of a sewage spill in South San Francisco Bay. *Estuaries* 6:399-406.

Cloern, J.E., et al. 2010. Biological communities in San Francisco Bay track large-scale climate forcing over the North Pacific. *Geophysical Research Letters* 37, L21602. 6pp.

Cloern, J.E., A.D. Jassby, J.K. Thompson and K.A. Hieb. 2007. A cold phase of the East Pacific triggers new phytoplankton blooms in San Francisco Bay. Proceedings of the National Academy of Sciences 104 (47):18561-18565.

Cloern, J.E., and R.T. Cheng. 1981. Simulation model of Skeletonema costatum population dynamics in northern San Francisco Bay. *Estuarine, Coastal and Shelf Science* 12:83-100.

Cloern, J.E. and A.D. Jassby. 2008. Complex seasonal patterns of primary producers at the land-sea interface. *Ecology Letters* 11:1294-1303.

Cloern, J.E., B.E. Cole, J.L. Edmunds, T.S. Schraga and A. Arnsberg. 2000. Patterns of Water-Quality Variability in San Francisco Bay During the First Six Years of the RMP. Regional Monitoring Program for Trace Substances, San Francisco Estuary Institute. Richmond, CA. 20 pp.

Cloern, J.E., B.E. Cole, R.L.J. Wong and A.E. Alpine. 1985. Temporal dynamics of estuarine phytoplankton: a case study of San Francisco Bay. *Hydrobiologia* 129:153-176.

Cloern, J.E., T.S. Schraga and C.B. Lopez. 2005a. Heat wave brings an unprecedented red tide to San Francisco Bay. Eos,Vol. 86, No. 7, p66.

Cloern, J.E., T.S. Schraga, C.B. Lopez and R. Lablosa. 2003. Lessons from monitoring water quality in San Francisco Bay. In The Pulse of the Estuary: Monitoring and Managing the Water Quality in the San Francisco Estuary. San Francisco Estuary Institute. Oakland, CA. 62-70.

Cloern, J.E., T.S. Schraga, C.B. Lopez, N. Knowles, R.G. Labiosa and R. Dugdale. 2005b. Climate anomalies generate and exceptional dinoflagellate bloom in San Francisco Bay. *Geophysical Research Letters* 32:L14608.

Cloern, J.E., A. Jassby, T.S. Schraga and K.L. Dallas. 2006. What is causing the phytoplankton increase in San Francisco Bay? In The Pulse of the Estuary: Monitoring and Managing the Water Quality in the San Francisco Estuary. San Francisco Estuary Institute. Oakland, CA. 15-20.

Cloern, J.E. and R.C. Dugdale. 2010. 5:6. San Francisco Bay case study. In Nutrients in Estuaries: A summary report of the national estuarine experts workshop 2005-2007. Glibert, P.M., Madden, C.J., Boynton, W., Flemer, D., Heil, C., and Sharp, J. (Principal Editors). Report to the EPA produced under EPA contract numbers 68-C-02-091 and EP-C-07-025. November 2010. 188pp. http://water.epa.gov/scitech/swguidance/standards/criteria/nutrients/upload/Nutrients-in-Estuaries-November-2010.pdf

Cloern, J.E., et al., Perils of correlating CUSUM-transformed variables to infer ecological relationships (Breton et al. 2006, Glibert 2010). *Limnology and Oceangraophy* in press.

Clough, L.M. and G.R. Lopez. 1993. Potential carbon sources for the head-down deposit-feeding polychaete Heteromastus filiformus. *Journal of Marine Research* 51:595-616.

Coan, E.V. 2002. The eastern Pacific recent species of the Corbulidae (Bivalvia). *Malacologia* 44:47-105.

Cohen, A.N. and J.T. Carlton. 1998. Accelerating Invasion Rate in a Highly Invaded Estuary. *Science* 279:555-558.

Cole, B.E. and J.E. Coern. 1987. An empirical model for estimating phytoplankton productivity in estuaries. *Marine Ecology Progress Series* 36:299-305.

Condon, R. 2010. Production of dissolved organic matter and inorganic nutrients by gelatinous zooplankton in the York River estuary, Chesapeake Bay. *Journal of Plankton Research* 32:153-170.

Conley, D.,.H.Kaas, F.Mohlenberg, B.Rasmussen and J.Windolf. 2000. Characteristics of Danish estuaries. *Estuaries* 23:820-837.

Conomos, T.J., (ed.). 1979. San Francisco Bay: the urbanized estuary. Investigation into the natural history of San Francisco Bay and Delta with reference to the influence of man. San Francisco, California: Pacific Division of the American Association for the Advancement of Science. 493 p.

Corbett, C.A., P.H. Doering, K.A. Madley, J.A. Ott and D.A. Tomasko. 2005. Using Seagrass Coverage as an Indicator of Ecosystem Condition. In S. Bertone (ed.) Estuarine Indicators. Boca Raton Florida: CRC Press, Inc. p229-245.

Costanzo, S.D., M.J. O'Donohue and W.C. Dennison. 2000. Gracilaria edulis (Rhodophyta) as a biological indicator of pulsed nutrients in oligotrophic waters. *Journal of Phycology* 36:680-685.

Cowan, J.W. and W.R. Boynton. 1996. Sediment-water oxygen and nutrient exchanges along the longitudinal axis of Chesapeake Bay: Seasonal patterns, controlling factors and ecological significance. *Estuaries* 19:562-580.

Creager, C., J. Butcher, et al. 2006. Technical Approach to Develop Nutrient Numeric Endpoints for California. Lafayette, CA, Tetra Tech, INC.: 115.

Dauer, D.M., J.A. Ranasinghe and S.B. Weisberg. 2000. Relationships between benthic community condition, water quality, nutrient loads, and land use patterns in Chesapeake Bay. *Estuaries* 23:80-96.

Dauer, D.M. 1993. Biological criteria, environmental health, and estuarine macrobenthic community structure. *Marine Pollution Bulletin* 26:249-257.

Dauvin, J. and T. Ruellet. 2009. The estuarine quality paradox: is it possible to define and ecological quality status for specific modified and naturally stressed estuarine systems. *Marine Pollution Bulletin* 59:38-47.

Dauvin, J. 2007. Paradox of estuarine quality: benthic indicators and indices, consensus or debate for the future. *Marine Pollution Bulletin* 55:271-281.

D'Avanzo, C. and J. Kremer. 1994. Diel oxygen dynamics and anoxic events in an eutrophic estuary. *Estuaries* 17:131-139.

Davis, J.A., L. McKee, J. Leatherbarrow and T. Daum. 2000. Contaminant Loads from Stormwater to Coastal Waters in the San Francisco Bay Region: Comparison to Other Pathways and Recommended Approach for Future Evaluation. San Francisco Estuary Institute, Richmond, CA.

Day, J., et al. 1989. Estuarine Ecology. John Wiley & Sons Inc. 543 pp.

Deegan, L.A., J.T. Finn, S.G. Awazian, C.A. Ryder-Kieffer and J. Buonaccorsi. 1997. Development and validation of an estuarine biotic integrity index. *Estuaries* 20:601–617.

Dennison, W.C., R.J. Orth, K.A. Moore, J.C. Stevenson, V. Carter, S. Kollar, P. Bergstrom and R.A. Batuik. 1993. Assessing water quality with submerged aquatic vegetation. *Bioscience* 43:86-94.

Derksen, D.V. and D.H. Ward. 1993. Life History and Habitat needs of the Black Brant. In: Waterfowl Management Handbook. US. Fish and Wildlife Service. Fish and Wildlife Leaflet 13. University of Nebraska- Lincoln. http://digitalcommons.unl.edu/icwdmwfm/41.

Dettman, D.L., M.J. Kohn, et al. 2001. Seasonal stable isotope evidence for a strong Asian monsoon throughout the past 10.7 m.y. *Geology* 29:31-34.

Diaz, R.J.. 2001. Overview of Hypoxia around the World. Journal of Environmental Quality 30:275-281.

Diaz, R.J. and L.C. Schaffner. 1990. The Functional Role of Estuarine Benthos. pp. 25-56. In: M. Haire and E.C. Krome (eds), Perspectives on the Chesapeake Bay, 1990. Advances in Estuarine Sciences. United States Environmental Protection Agency, Gloucester Point, VA.

Diaz, R.J. and R. Rosenberg. 1995. Marine benthic hypoxia: a review of its ecological effects and the behavioral responses of benthic macrofauna. *Oceanography and Marine Biology: An Annual Review* 33:245-303.

Diaz, R.J. and R. Rosenberg. 2008. Spreading dead zones and consequences for marine ecosystems. *Science* 321:926-929.

Diaz, R.J., D.C. Rhoads, J.A. Blake, R.K. Kropp and K.E. Keay. 2008. Long-term trends of benthic habitats related to reduction in wastewater discharge to Boston Harbor. *Estuaries and Coasts* 31:1184-1197.

Diaz, R.J., M. Solan and R.M. Valente. 2004. A review of approaches for classifying benthic habitats and evaluating habitat quality. *Journal of Environmental Management* 73:165-181.

Dodds, W.K., V.H. Smith, et al. 2002. Nitrogen and phosphorus relationships to benthic algal biomass in temperate streams. *Canadian Journal of Fisheries and Aquatic Sciences* 59: 865-874.

Downing, J.A., C.W. Osenberg and O. Sarnelle. 1999. Meta-analysis of marine nutrient-enrichment experiments: variation in the magnitude of nutrient limitation. *Ecology* 80:1157-1167.

Draheim, R.C. 1998. Tidal Freshwater and Oligohaline Benthos: Evaluating the Development of a Benthic Index of Biological Integrity for Chesapeake Bay, Masters Thesis. Gloucester Point, VA, The College of William and Mary.

Duarte, C.M. 1995. Submerged aquatic vegetation in relation to different nutrient regimes. Ophelia 41:87-112.

Dugdale, R.C., F.P. Wilkerson, V.E. Hogue and A. Marchi. 2007. The role of ammonium and nitrate in spring bloom development in San Francisco Bay. 2007. *Estuarine, Coastal and Shelf Science* 73:17-29.

East Bay MUD (EBMUD). 2010. Environmental Enhancement Project and Supplemental Environmental Porject: Characterization of Stormwater Flows, Diversion of Dry Weather and First Flush Flows to a Publically-Owned Treatment Works. East Bay Municiple Untility District, Oakland.

Edgar, G.J. 1990. The use of the size structure of benthic macrofaunal communities to estimate faunal biomass and secondary production. *Journal of Experimental Marine Biology and Ecology* 137:195-214

Elliott, M. and V. Quintino. 2007. The estuarine quality paradox, environmental homeostasis and the difficulty of detecting anthropogenic stress in naturally stressed areas. *Marine Pollution Bulletin* 54:640-645.

Engle, V.D., J.C. Kurtz, L.M.Smith, C. Chancy and P. Bourgeois. 2007. A Classification of U.S. Estuaries Based on Physical and Hydrologic Attributes. *Environmental Monitoring and Assessment* 129:397-412.

Fauchald, K. and P.A. Jumars. 1979. The diet of worms: a study of polychaete feeding guilds. *Oceanography and Marine Biology: An Annual Review* 17:193-284.

Fevold, K. 1998. Subsurface controls on the distribution of benthic algae in floodplain back channel habitats of the Queets River, University of Washington, Seattle. Master's Thesis.

Foden, J. and D.P. Brazier. 2007. Angiosperms (seagrass) within the EU water framework directive: a UK perspective. *Marine Pollution Bulletin* 55:181-195.

Fong, P., K.E. Boyer and J.B. Zedler. 1998. Developing an indicator of nutrient enrichment in coastal estuaries and lagoons using tissue nitrogen content of the opportunistic alga, Enteromorpha intestinalis (L. Link). *Journal of Experimental Marine Biology and Ecology* 231:63-79.

Fong, P., R.M. Donohoe and J.B. Zedler. 1994. Nutrient concentration in tissue of the macroalga Enteromorpha as a function of nutrient history: An experimental evaluation using field microcosms. *Marine Ecology Progress Series* 106:273-81.

Fong, P., J.B. Zedler and R.M. Donohoe. 1993. Nitrogen vs. phosphorus limitation of algal biomass in shallow coastal lagoons. *Limnology Oceanography* 38:906-923.

Fonseca, M., P.E. Whitfield, N.M. Kelly and S.S. Bell. 2002. Modeling seagrass landscape pattern and associated ecological attributes. *Ecological Applications* 12:218-237.

Fox, S.E., E. Stieve, I. Valiela, J. Hauxwell and J. McClelland. 2008. Macrophyte abundance in Waquoit Bay: Effects of land-derived nitrogen loads on seasonal and multi-year biomass patterns. *Estuaries and Coasts* 31:532-541.

Fram, J.P., M.A. Martin and M.T. Stacey. 2007. Dispersive fluxes between the coastal ocean and a semi-enclosed estuarine basin. *Journal of Physical Oceanography* 37:1645-1660.

Fredette, T.J., M.S. Fonseca, W.J. Kenworthy and S. Wyllie-Echeverria. 1987. An investigation of eelgrass (*Zostera marina*) transplanting feasibility in San Francisco Bay, California. Prepared for US Army Corps of Engineers. San Francisco, CA.

Fujita, R.M. 1985. The role of nitrogen status in regulating transient ammonium uptake and nitrogen storage by macroalgae. *Journal of Experimental Marine Biology and Ecology* 92:283-301.

Gallegos, C.L. 2001. Calculating optical water quality targets to restore submerged aquatic vegetation: Overcoming problems partitioning diffuse attenuation coefficient for photosynthetically active radiation. *Estuaries* 24:381-397.

Gallegos, C.L. and J.W. Kenworthy. 1996. Seagrass depth limits in the Indian River Lagoon (Florida) Application of an optical water quality model. *Estuarine, Coastal and Shelf Science* 42:267-288.

Ganju, N.K., N. Knowles and D.H. Schoellhamer. 2008. Temporal downscaling of decadal sediment load estimates to a daily interval for use in hindcast simulations. *Journal of Hydrology* 349:512-523.

Gaston, G.R., C.F. Rakocinski, S.S. Brown and C.M. Cleveland. 1998. Trophic function in estuaries: Response of macrobenthos to natural and contaminant gradient. *Marine and Freshwater Research* 49:833-846.

Gaudeñcio, M.J. and H.N. Cabral. 2007. Trophic structure of macrobenthos in the Tagus Estuary and adjacent coastal shelf. *Hydrobiologia* 587:241-251.

Gerritsen, J., A.F. Holland and D.E. Irvine. 1994. Suspension-feeding bivalves and the fate of primary production: An estuarine model applied to Chesapeake Bay. *Estuaries* 17:403-416.

Gewant, D. and S. Bollens. 2005. Macrozooplankton and micronekton of the lower San Francisco Estuary: Seasonal, interannual, and regional variation in relation to environmental conditions. *Estuaries and Coasts* 28:473-485.

Gillett, D.J. 2010. Effects of habitat quality on secondary production in shallow estuarine waters and the consequences for the benthic-pelagic food web. Ph.D. Dissertation, The College of William and Mary. Gloucester Point, VA.

Gillett, D.J., A.F. Holland and D.M. Sanger. 2007. On the ecology of oligochaetes: Monthly variation of community composition and environmental characteristics in two South Carolina tidal creeks. *Estuaries and Coasts* 30:238-252.

Giusti, E. and S. Marsili-Libelli. 2005. Modelling the interactions between nutrients and the submersed vegetation in the Orbetello Lagoon. *Ecological Modelling* 184:141-161.

Glasgow, Jr., H.B. and J.M. Burkholder. 2000. Water quality trends and management implications from a five-year study of a eutrophic estuary. *Ecological Applications* 10:1024-1046.

Glibert, P.M., J. Harrison, et al. 2006. Escalating worldwide use of urea: A global change contributing to coastal eutrophication. *Biogeochemistry* 77:441-463.

Glibert, P.M., R. Magnien, et al. 2001. Harmful algal blooms in the Chesapeake and coastal bays of Maryland, USA: Comparison of 1997, 1998, and 1999 events. *Estuaries* 24:875-883.

Glibert, P.M. 2010. Long-term changes in nutrient loading and stoichiometry and their relationships with changes in the food web and dominant pelagic fish species in San Francisco Estuary, California. *Reviews in Fisheries Science* 18:211-232.

Goals Project. 2000. Baylands Ecosystem Species and Community Profiles: Life histories and environmental requirements of key plants, fish and wildlife. Prepared by the San Francisco Bay Area Wetlands Ecosystem Goals Project. P.R. Olofson, editor. San Francisco Bay Regional Water Quality Control Board. Oakland, CA.

Graf, G. 1992. Benthic-pelagic coupling: A benthic view. *Oceanography and Marine Biology: An Annual Review* 30:149-190.

Grall, J. and L. Chauvaud. 2002. Marine eutrophication and benthos: The need for new approaches and concepts. *Global Change Biology* 8:813-830.

Graves, G., M. Thompson, G. Schmitt, D. Fike, C. Kelly and J. Tyrrell. 2005. Using macroinvertebrates to document the effects of a storm water–induced nutrient gradient on a subtropical estuary. pp. 333-349 *in*: S. Bertone (ed.), Estuarine Indicators. CRC Press, Inc. Boca Raton, FL.

Gray, J.S., R.S. Wu and Y.Y. Or. 2002. Effects of hypoxia and organic enrichment on the coastal marine environment. *Marine Ecology Progress Series* 238:249-279.

Green, L. 2010. Effects of macroalgae on the benthic infauna on estuarine intertidal flats and implications for the foraging behavior of estuarine migratory birds. Ph.D. Dissertation, University of California, Los Angeles. Los Angeles, CA

Grenz, C., J.E. Cloern, S.W. Hager and B.E. Cole. 2000. Dynamics of nutrient cycling and related benthic nutrient and oxygen fluxes during a spring phytoplankton bloom in south San Francisco Bay (USA). *Marine Ecology Progress Series* 197:67-80.

Grosso, C. and S. Lowe. 2002. Deterministic to Probabilistic: Changing the RMP's Sampling Design. Regional Monitoring News Newsletter. Winter, 2002. Volume 6, Issue 2. http://www.sfei.org/rmp/rmp_news/rmpnews_vol6_issue2.pdf

Hager, S.W. and L.E. Schemel. 1992. Sources of nitrogen and phosphorus to northern San Francisco Bay. *Estuaries* 15:40-52.

Hagy, J.D., W.R. Boynton, C.W. Keefe and K.V. Wood. 2004. Hypoxia in Chesapeake Bay, 1950-2001: Long-term change in relation to nutrient loading and river flow. *Estuaries* 27:634-658.

Hammond, D.E., C. Fuller, et al. 1985. Benthic fluxes in San Francisco Bay. *Hydrobiologia* 129:69-90.

Hanson, R.T., Li, Zhen and C.C. Faunt. 2004. Documentation of the Santa Clara Valley regional ground-water/surface-water flow model, Santa Clara County, California: U.S. Geological Survey Scientific Investigations. Report 2004-5231.

Hargrave, B.T., M. Holmer and C.P. Newcombe. 2008. Towards classification of organic enrichment in marine sediments based on biogechemical indicators. *Marine Pollution Bulletin* 56:810-824.

Hauxwell, J., J. McClelland, P. Behr and I. Valiela. 1998. Relative importance of grazing and nutrient controls of macroalgal biomass in three temperate shallow estuaries. *Estuaries* 21:347-360.

Havens, K.E., J. Hauxwell, A.C. Tyler, S. Thomas, K.J. McGlathery, J. Cebrian, I. Valiela, A.D. Steinman and S.-J. Hwang. 2001. Complex interactions between autotrophs in shallow marine and freshwater ecosystems: Implications for community responses to nutrient stress. *Environmental Pollution* 113:95-107.

Heidel, K., S. Roy, C. Creager, C.-F. Chung and T. Grieb. 2006. Conceptual Model for Nutrients in the Central Valley and Sacramento-San Joaquin Delta. Prepared by Tetra Tech Inc. for USEPA (Region IX) and Central Valley Drinking Water Policy Workgroup.

Heip, C.H.R., N.K. Goosen, P.M.J. Herman, J. Kromkamp, J.J. Middleburg and K. Soetaert. 1995. Production and consumption of biological particles in temperate tidal estuaries. *Oceanography and Marine Biology an Annual Review* 33:1-149.

Heip, C. and J. Craeymeersch. 1995. Benthic community structures in the North Sea. *Helgoland Marine Research* 49:313-328.

Heiskary, S. and H. Markus. 2001. Establishing relationships among nutrient concentrations, phytoplankton abundance, and biochemical oxygen demand in Minnesota, USA rivers. *Journal of Lake and Resevoir Management* 17:251-262.

Hernandez, I., G. Peralta, J.L. Perez-Llorens, J.J. Vergara and F.X. Niell. 1997. Biomass and dynamics of growth Ulva species in Palmones River Estuary. *Journal of Phycology* 33:764-772.

Herndon, J., W.P. Cochlan and R.A. Horner. 2003. Heterosigma akashiwo blooms in San Francisco Bay. IEP Newsletter 16:46-48.

Hogue, V.E., F.P. Wilkerson and R.C. Dugdale. 2005. Ultraviolet-B radiation effects on natural assemblages of central San Francisco Bay. *Estuaries* 28:190-203.

Hogue, V.E., F.P. Wilkerson, R.C. Dugdale and A. Marchi. 2001. Phytoplankton and nutrient dynamics in Suisun, San Pablo, and Central Bays. IEP Newsletter 14:35-41.

Holland, A.F., A.T. Shaughnessy and M.H.Heigel. 1987. Long-term variation in mesohaline Chesapeake Bay macrobenthos: Spatial and temporal patterns. *Estuaries* 10:227-245.

Holland, A.F., D.M. Sanger, C.P. Gawle, S.B. Lerberg, M.S. Santiago, G.H.M. Riekerk, L.E. Zimmerman and G.I. Scott. 2004. Linkages between tidal creek ecosystems and the landscape and demographic attributes of their watersheds. *Journal of Experimental Marine Biology and Ecology* 298:151-178.

Holland, A.F., N.K. Mountford and J.A. Mihursky. 1977. Temporal variation in Upper Bay mesohaline benthic communities: I. The 9-m mud habitat. *Chesapeake Science* 18:370-378.

Holme, N.A. and A.D. McIntyre. 1984. Methods for the Study of Marine Benthos 2nd Ed, 2nd. Blackwell Scientific, Oxford

Horrocks, J.L., G.R. Stewart and W.C. Dennison. 1995. Tissue nutrient content of *Gracilariopsis* spp. (Rhodophyta) and water quality along an estuarine gradient. *Marine Freshwater Research* 46:975-983.

Howarth, R.W. 1988. Nutrient limitation of primary production in marine ecosystems. *Annual Review of Ecology and Systematics* 19:89-110.

Howarth, R.W., A. Sharpley and D. Walker. 2002. Sources of nutrient pollution to coastal waters in the United States: Implications for achieving coastal water quality goals. *Estuaries* 25:656-676.

Howarth, R.W. and R. Marino. 2006. Nitrogen as the limiting nutrient for eutrophication in coastal marine ecosystems: Evolving views over three decades. *Limnology and Oceanography* 51:364-376.

Hull, S.C. 1987. Macroalgal mats and species abundance: A field experiment. *Estuarine, Coastal and Shelf Science* 25:519-532.

Huntington, B.E. and K. Boyer. 2008. Effects of red macroalgal (*Gracilariopsis sp.*) abundance on eelgrass *Zostera marina* in Tomales Bay, California, USA. *Marine Ecology Progress Series* 367:133-142.

Huntington, B.E. and K. Boyer. 2008. Evaluating patterns of nitrogen supply using macroalgal tissue content and stable isotopic signatures in Tomales Bay, CA. *Environmental Bioindicators* 3:180-192.

Hurley, L. 1990. Field guide to the submerged aquatic vegetation of Chesapeake Bay. Annapolis. MD. US Fish and Wildlife Service, Chesapeake Bay Estuary Program

Hyland, J., L. Balthis, I. Karakassis, P. Magni, A. Petrov, J. Shine, O. Vestergaard and R. Warwick. 2005. Organic carbon content of sediments as an indicator of stress in the marine benthos. *Marine Ecology Progress Series* 295:91-103.

International Association of Limnology. 1958. The Venice system for the classification of marine waters according to salinity. *Limnology and Oceanography* 3:346-347.

Irlandi, E.A. 1997. Seagrass patch size and survivorship of an infaunal bivalve. *Oikos* 78:511-518.

Isaksson, I. and L. Pihl. 1992. Structural changes in benthic macrovegetation and associated epibenthic faunal communities. *Netherlands Journal of Sea Research* 30:131-140.

Iwamatsu, S., A. Suzuki and M. Sato. 2007. Nereidid polychaetes as the major diet of migratory shorebirds on the estuarine tidal flats at Fujimae-Higata in Japan. *Zoological Science* 24:676-685.

Jaffe, B.E., R.E. Smith and A.C. Foxgrover. 2007. Anthropogenic influence on sedimentation and intertidal mudflat change in San Pablo Bay, California: 1856-1983. *Estuarine, Coastal and Shelf Science* 73:175-187.

Janicki, A.J., D. Wade and J.R. Pribble. 2000. Establishing a process for tracking chlorophyll-a concentrations and light attenuation in Tampa Bay. Prepared for: Tampa Bay Estuary Program. Prepared By: Janicki Environmental, Inc.

Jassby, A.D., W.J. Kimmerer, et al. 1995. Isohaline position as a habitat indicator for estuarine populations. *Ecological Applications* 5: 272-289.

Jassby, A.D. and J.E. Cloern. 2000. Organic matter sources and rehabilitation of the Sacramento-San Joaquin Delta (California, USA). *Aquatic Conservation: Marine and Freshwater Ecosystems* 10:323-352.

Jassby, A.D. 2008. Phytoplankton in the upper San Francisco Estuary: Recent biomass trends, their causes, and their trophic significance. San Francisco Estuary and Watershed Science, volume 6, Issue 1, Article 2.

Jassby, A.D., J.E. Cloern and T.M. Powell. 1993. Organic carbon sources and sinks in San Francisco Bay: Variability induced by river flow. *Marine Ecology Progress Series* 95:39-54.

Jassby, A.D., J.E. Cloern and B.E. Cole. 2002. Annual primary production: Patterns and mechanisms of change in a nutrient-rich tidal ecosystem. *Limnology and Oceanography* 47:698-712.

Johnson, M.R., S.L. Williams, C.H. Lieberman and A. Solbak. 2003. Changes in the abundance of the seagrasses *Zostera marina L.* (eelgrass) and *Ruppia maritima L.* (widgeongrass) in San Diego, California, following an El Niño event. *Estuaries* 26:106-115.

Jørgensen, B.B. 1996. Material flux in sediments. pp. 115-136 *in*: B.B. Jørgensen and K. Richardson (eds.), Eutrophication in Coastal Marine Ecosystems, Coastal and Estuarine Studies, Vol 2. American Geophysical Union. Washington, DC.

Josefson, A.B. and B. Rasmussen. 2000. Nutrient retention by benthic macrofaunal biomass of danish estuaries: Importance of nutrient load and residence time. *Estuarine, Coastal and Shelf Science* 50:205-216.

Josselyn, M. and J. West. 1985. The distribution and temporal dynamics of the estuarine macroalgal community of San Francisco Bay. *Hydrobiologia* 129:139-152.

Kaldy, J.E. and K.S. Lee. 2007. Factors controlling *Zostera marina L.* growth in the eastern and western Pacific Ocean: Comparisons between Korea and Oregon, USA. *Aquatic Botany* 87:116-126.

Kamer, K. and E. Stein. 2003. Dissolved oxygen concentration as a potential indicator of water quality in Newport Bay: A review of scientific research, historical data, and criteria development. Southern California Coastal Water Research Project. Westminster, CA.

Kamer, K., K. Boyle and P. Fong. 2001. Macroalgal bloom dynamics in a highly eutrophic southern California estuary. *Estuaries* 24:623-635.

Karr, J.R., and E.W. Chu. 1999. Restoring Life in Running Waters: Better Biological Monitoring. ISBN 1–55963–674–2. Island Press. Washington D.C.

Kemp, W.M., J.M. Testa, D.J. Conley, D. Gilbert and J.D. Hagy. 2009. Temporal responses of coastal hypoxia to nutrient loading and physical controls. *Biogeosciences* 6:2985-3008.

Kennedy, V.S. (ed). 1982. Estuarine Comparisons. Academic Press. New York, NY.

Kennish, M.J. 2002. Environmental threats and environmental future of estuaries. *Environmental Conservation* 29:78-107.

Kennison, R., K. Kamer, et al. 2003. Nutrient dynamics and macroalgal blooms: A comparison of five southern California estuaries. Southern California Coastal Water Research Project. Westminster, CA.

Kennison, R.L. 2008. Evaluating ecosystem function of nutrient retention and recycling in excessively eutrophic estuaries. Ph.D. dissertation, University of California, Los Angeles. Los Angeles, CA.

Kimmerer and J. W. 2006. Response of anchovies dampens effects of the invasive bivalve Corbula amurensis on the San Francisco Estuary foodweb. Marine Ecology Progress Series 324: 207-218.

Koch, E.M. 2001. Beyond light: Physical, geological and geochemical parameters as possible submersed aquatic vegetation habitat requirements. *Estuaries* 24:1-17.

Krause-Jensen, D., A.L. Middelboe, et al. 2007. Spatial patterns of macroalgal abundance in relation to eutrophication. *Marine Biology* 152:25-36.

Krause-Jensen, D., J. Carstensen and K. Dahl. 2007. Total and opportunistic algal cover in relation to environmental variables. *Marine Pollution Bulletin* 55:114-125.

Krause-Jensen, D., P.B. Christensen, et al. 1999. Oxygen and nutrient dynamics within mats of the filamentous macroalga *Chaetomorpha linum*. *Estuaries* 22:31-38.

Kratzer, C.R., R.H. Kent, D.K. Saleh, D.L. Knifong, P.D. Dileanis and J.L. Orlando. 2010. Trends in Nutrient Concentrations, Loads, and Yields in Streams in the Sacramento, San Joaquin, and Santa Ana Basins, California, 1975–2004. USGS Scientific Investigations Report 2010–5228.

Kristiansen, K.D., E. Kristensen, et al. 2002. The influence of water column hypoxia on the behavior of manganese and iron in sandy coastal marine sediment. *Estuarine, Coastal and Shelf Science* 55:645-654.

Kudela, R.M., J.Q. Lane, et al. 2008. The potential role of anthropogenically derived nitrogen in the growth of harmful algae in California, USA. *Harmful Algae* 8:103-110.

Kulongoski, J.T., K. Belitz, M.K. Landon and C. Farrar. 2010. Status and Understanding of Groundwater Quality in the North San Francisco Bay Groundwater Basins, 2004: California GAMA Priority Basin Project. USGS Scientific Investigations Report 2010–5089.

Lacouture, R.V., J.M. Johnson, C. Buchanan and H.G. Marshall. 2006. Phytoplankton index of biotic integrity for Chesapeake Bay and its tidal tributaries. *Estuaries and Coasts* 29:598-616.

Largier, J.L., J.H. Slinger, et al. 1991. The stratified hydrodynamics of the Palmiet-- A prototypical bar-built estuary. Dynamics and Exchanges in Estuaries and the Coastal Zone. D. Prandle (ed). Washington D.C., American Geophysical Union: 135-153.

Laursen, A.E., S.P. Seitzinger, et al. 2002. Multiple stressors in an estuarine system: Effects of nutrients, trace elements, and trophic complexity on benthic photosynthesis and respiration. *Estuaries* 25:57-69.

Lavesque, N., H. Blanchet and X. de Montaudouin. 2009. Development of a multimetric approach to assess perturbation of benthic macrofauna in *Zoster noltii* beds. *Journal of Experimental Marine Biology and Ecology* 368:101-112.

Lee, K.S., F.T. Short, et al. 2004. Development of a nutrient pollution indicator using the seagrass, *Zostera marina*, along nutrient gradients in three New England estuaries. *Aquatic Botany* 78:197-216.

Lehman, P.W. and S. Waller. 2003. Microcystis blooms in the delta. Interagency Ecological Program for the San Francisco Estuary Newsletter. 16, 18-19. www.water.ca.gov/iep/products/newsletter.cfm

Lehman, P.W., G. Boyer, C. Hall, S. Waller and K. Gehrts. 2005. Distribution and toxicity of a new colonial Microcystis aeruginosa bloom in San Francisco Bay Estuary, California. *Hydrobiologia* 541:87-99.

Lehman, P.W., G. Boyer, M. Satchwell and S. Waller. 2008. The influence of environmental conditions on the seasonal variation of Microsystis cell density and microcystins concentration in San Francisco Estuary. *Hydrobiologia* 600:187-204.

Lenihan, H.S., C.H. Peterson, S.L. Kim, K.E. Conlan, R. Fairey, C. McDonald, J.H. Grabowski and J.S. Oliver. 2003. Variation in marine benthic community composition allows discrimination of multiple stressors. *Marine Ecology Progress Series* 261:63-73.

Lent, M.A. and L.J. McKee. 2011. Development of regional contaminant load estimates for San Francisco Bay Area tributaries based on annual scale Rainfall-Runoff and Volume-Concentration models: Year 1 Results. A technical report for the Regional Monitoring Program for Water Quality. San Francisco Estuary Institute, Oakland, CA. www.sfei.org

Lenzi, M., R. Palmieri and S. Porrello. 2003. Restoration of the eutrophic Orbetello Lagoon (Tyrrhenian Sea, Italy): Water quality management. *Marine Pollution Bulletin* 46:1540-1548.

Levin, L., N. Blair, D. DeMaster, G. Plaia, W. Fornes, C. Martin and C. Thomas. 1997. Rapid subduction of organic matter by maldanid polychaetes on the North Carolina slope. *Journal of Marine Research* 55:595-611.

Levin, L.A. 1984. Multiple patterns of development in *Streblospio benedicti Webster* (Spionidae) from three coasts of North America. *Biological Bulletin* 166:434-508.

Lewicki, M. and L.J. McKee. 2009. Watershed specific and regional scale suspended sediment loads for bay area small tributaries. A technical report for the Sources Pathways and Loading Workgroup of the Regional Monitoring Program for Water Quality: SFEI Contribution #566. San Francisco Estuary Institute, Oakland, CA. 28 pp + Appendices. www.sfei.org

Llansó, R.J. 1992. Effects of hypoxia on estuarine benthos: The lower Rappahannock River (Chesapeake Bay), a case study. *Estuarine, Coastal and Shelf Science* 35:491-515.

Llansó, R.J., L.C. Scott, J.L. Hyland, D.M. Dauer, D.E. Russell and F.W. Kutz. 2002. An estuarine benthic index of biotic integrity for the Mid-Atlantic region of the United States. II Index development. *Estuaries* 25:1231-1242.

Long, W.C., B.J. Brylawski and R.D. Seitz. 2008. Behavioral effects of low dissolved oxygen on the bivalve *Macoma balthica*. *Journal of Experimental Marine Biology and Ecology* 359:34-39.

Lopes, J.R, et al. 2000. Impact of macroalgal blooms and wader predation on intertidal macroinvertebrates: Experimental evidence from the Mondego Estuary (Portugal). Journal of Experimental Marine Biology and Ecology. Volume 249, Issue 2, 28 June 2000, Pages 165-179

Lopez, G.R. and J.S. Levinton. 1987. Ecology of deposit-feeding animals in marine sediments. *The Quarterly Review of Biology* 62:235-260.

Lotze, H.K. and W. Schramm. 2000. Ecophysiological traits explain species dominance patterns in macroalgal blooms. *Journal of Phycology* 36:287-95.

Lowe, S., B. Thompson, R. Hoenicke, J. Leatherbarrow, K. Taberski, R. Smith and D. Stevens, Jr. 2005. Re-design Process of the San Francisco Estuary Regional Monitoring Program for Trace Substances (RMP) Status and Trends Monitoring Component for Water and Sediment. San Francisco Estuary Institute, July 2005, Contribution number 109. www.sfei.org

Lucas L.V., J.R. Koseff, S.G. Monismith and J.K. Thompson. 2009. Shallow water processes govern system-wide phytoplankton bloom dynamics: A modeling study. *Journal of Marine Systems* 75:70-86.

Lucas, C.H. 2001. Reproduction and life history strategies of the common jellyfish, *Aurelia aurita*, in relation to its ambient environment. *Hydrobiologia* 451:229-246.

Lucas, L.V. and J.E. Cloern. 2002. Effects of tidal shallowing and deepening on phytoplankton production dynamics: A modeling study. *Estuaries* 25:497-507.

Lucas, L.V., J.E. Cloern, J.R. Koseff, S.G. Monismith and J.K. Thompson. 1998. Does the Sverdrup critical depth model explain bloom dynamics in estuaries? *Journal of Marine Research* 56:375-415.

Lucas, L.V., J.R. Koseff, S.G. Monismith, J.E. Cloern and J.K. Thompson. 1999b. Processes governing phytoplankton blooms in estuaries. II: the role of horizontal transport. *Marine Ecology Progress Series* 187:17-30.

Lucas, L.V., J.R. Koseff, J.E. Cloern, S.G. Monismith and J.K. Thompson. 1999a. Processes governing phytoplankton blooms in estuaries. I: the local production-loss balance. *Marine Ecology Progress Series* 187:1-15.

Luengen, A.C. and A.R. Flegal. 2009. Role of phytoplankton in mercury cycling in the San Francisco Bay Estuary. *Limnology and Oceanography* 54:23-40.

Luoma, S.N., A. Van Geen, B.-G. Lee and J.E. Cloern. 1998. Metal uptake by photoplankton during a bloom in South San Francisco Bay: Implications for metal cycling in estuaries. *Limnology and Oceanography* 43:1007-1016.

Mackas, D.L. and P.J. Harrison. 1997. Nitrogenous nutrient sources and sinks in the Juan de Fuca Strait Strait of Georgia Puget Sound estuarine system: Assessing the potential for eutrophication. *Estuarine, Coastal and Shelf Science* 44:1-21.

Madden, C.J., D.T. Rudnick, A.A. McDonald, K.M. Cunniff and J.W. Fourqurean. 2009. Ecological indicators for assessing and communication seagrass status and trends in Florida Bay. *Ecological Indicators* 9S:S68-S82.

Marcomini, A., A. Sfriso, B. Pavioni and H.H. Orio. 1995. Eutrophication of the Lagoon of Venice: Nutrient loads and exchanges. *In*: A.J. McComb (ed.),Eutrophic Shallow Estuaries and Lagoons. CRC Press. Boca Raton, FL.

Marsh, A.G. and K.N. Tenore. 1990. The role of nutrition in regulating the population dynamics of opportunistic, surface deposit feeders in a mesohaline community. *Limnology and Oceanography* 35:710-724.

Martin, M., J.P. Fram and M.T. Stacey. 2007. Seasonal Chlorophyll-a fluxes between the coastal Pacific Ocean and San Francisco Bay. *Marine Ecology Progress Series* 337:51-61.

Martinetto, P., M. Teichberg and I. Valiela. 2006. Coupling of estuarine benthic and pelagic food webst to land-derived nitrogen sources in Waquoit Bay, Massachusetts, USA. *Marine Ecology Progress Series* 307:37-48.

Matisoff, G., J.B. Fisher, et al. 1985. Effects of benthic macroinvertebrates on the exchange of solutes between sediments and freshwater. *Hydrobiologia* 122:19-33.

May, C.L., J.R. Koseff, L.V. Lucas, J.E. Cloern and D.H. Schoellhamer. 2003. Effects of spatial and temporal variability of turbidity on phytoplankton blooms. *Marine Ecology Progress Series* 254:111-128.

Mayer, L.M. 1994. Surface area control of organic carbon accumulation in continental shelf sediments. *Geochimica et Cosmochimica Acta* 58:1271-1284.

Mazik, K., N. Curtis, et al. 2008. Accurate quantification of the influence of benthic macro- and meio-fauna on the geometric properties of estuarine muds by micro computer tomography. *Journal of Experimental Marine Biology and Ecology* 354:192-201.

McGlathery, K. J. 2001. Macroalgal blooms contribute to the decline of seagrass in nutrient enriched coastal waters. *Journal of Phycology* 37:453-456.

McGlathery, K.J., D. Krause-Jense, S. Rysgaard and P.B. Christensen. 1997. Patterns of ammonium uptake within dense mats of the filamentous macroalga *Chaetomorpha linum*. *Aquatic Botany* 59:99-115.

McKee, L., B. Eyre and S. Hossain. 2000. Transport and retention of nitrogen and phosphorus in the sub-tropical Richmond River Estuary, Australia - a budget approach. *Biogeochemistry* 50:241-278.

McKee, L.J., N.K. Ganju and D.H. Schoellhamer. 2006. Estimates of suspended sediment entering San Francisco Bay from the Sacramento and San Joaquin Delta, San Francisco Bay, California. *Journal of Hydrology* 323:335-352.

Merkel & Associates, Inc. 1999. Richmond Harbor Navigation Improvement Project. Post-dredging eelgrass survey. Prepared for Tetra Tech, Inc. San Francisco, CA.

Merkel & Associates, Inc. 2003. Baywide eelgrass (*Zostera marina L.)* inventory and resource management program in San Francisco Bay. Report for: San Francisco-Oakland Bay Bridge East Span Seismic Safety Project.

Merkel & Associates, Inc. 2004a. Baywide eelgrass inventory of San Francisco Bay: Pre-survey Screening Model and Eelgrass Survey Report. Report for: San Francisco-Oakland Bay Bridge East Span Seismic Safety Project.

Merkel & Associates, Inc. 2005. Eelgrass community pilot study for San Francisco Bay: Techniques for examining invertebrate and fish assemblages within multiple eelgrass beds. Report for: San Francisco-Oakland Bay Bridge East Span Seismic Safety Project.

Merkel & Associates, Inc. 2010. San Francisco Bay Eelgrass Inventory October – November 2009. Report for: San Francisco-Oakland Bay Bridge East Span Seismic Safety Project.

Merkel & Associates, Inc. 2004. Baywide eelgrass (*Zostera marina L.*) inventory of San Francisco Bay: Pre-survey screening model and eelgrass survey report. Prepared in Cooperation with NOAA Fisheries for support for the California Department of Transportation (Caltrans) as one element of the San Francisco-Oakland Bay Bridge East Span Seismic Safety Project environmental mitigation.

Michaud, E., G. Desrosiers, F. Mermillod-Blondin, B. Sundby and G. Stora. 2005. The functional group approach to bioturbation: The effects of biodifusers and gallery-diffusers of the *Macoma balthica* community on sediment oxygen uptake. *Journal of Experimental Marine Biology and Ecology* 326:77-88.

Michaud, E., G. Desrosiers, F. Mermillod-Blondin, B. Sundby and G. Stora. 2006. The functional group approach to bioturbation: II. The effects of the Macoma balthica community on fluxes of nutrients and dissolved organic carbon across the sediment-water interface. *Journal of Experimental Marine Biology and Ecology* 337:178-189.

Mills, C.E. 2001. Jellyfish blooms: Are populations increasing globally in response to changing ocean conditions? *Hydrobiologia* 451: 55-68.

Molinaroli, E., S. Guerzoni, G. De Falco, A. Sarretta, A. Cucco, S. Como, S. Simeone, A. Perilli and P. Magni. 2009. Relationships between hydrodynamic parameters and grain size in two contrasting transitional environments: The Lagoons of Venice and Cabras, Italy. *Sedimentary Geology* 219:196-207.

Møller, L.F. and H.U. Riisgård. 2007. Impact of jellyfish and mussels on algal blooms caused by seasonal oxygen depletion and nutrient release from the sediment in a Danish fjord. *Journal of Experimental Marine Biology and Ecology* 351:92-105.

Moore, J.E., M.A. Colwell, R.L. Mathis and J.M. Black. 2004. Staging of Pacific flyway brant in relation to eelgrass abundance and site isolation, with special consideration of Humboldt Bay, California. *Biological Conservation* 115:475-486.

Moreno, A. 2003. Jellyfish of the San Francisco Estuary. Interagency Ecological Program for the San Francisco Estuary Newsletter. Volume 16(No.2). www.water.ca.gov/iep/products/newsletter.cfm

Mucha, A.P. and M.H. Costa. 1999. Macrozoobenthic community structure in two Portuguese estuaries: Relationship with organic enrichment and nutrient gradients. *Acta Oecologica* 20: 363-376.

Muir, K.S. 1996. Groundwater Discharge in the East Bay Plain Area, Alameda County, California: Alameda County Flood Control and Water Conservation District.

Naldi, M. and P. Viaroli. 2002. Nitrate uptake and storage in the seaweed *Ulva rigida* in relation to nitrate availability and thallus nitrate content in a eutrophic coastal lagoon (Saccadi Goro, Po River Delta, Italy). *Journal of Experimental Marine Biology and Ecology* 269:65-83.

Nedwell, D.B., L.F. Dong, et al. 2002. Variations of the nutrients loads to the mainland UK estuaries: Correlation with catchment areas, urbanization and coastal eutrophication. *Estuarine Coastal and Shelf Science* 54:951-970.

Neisar, T. Personal communication December 2010. Alameda County Water District. 43885 So. Grimmer Blvd. Fremont, CA 94538.

Nelson, W.G. 2009. The interaction of epiphytes with seagrasses under nutrient enrichment. IN: Seagrasses and Protective Criteria: A review and assessment of research Status. EPA/600/R-09/050. Office of Research and Development, National Health and Environmental Effects Research Laboratory.

Nestlerode, J.A. and R.J. Diaz. 1998. Effects of periodic environmental hypoxia on predation of a tethered polychaete, *Glycera americana*: Implications for trophic dynamics. *Marine Ecology Progress Series* 172:185-195.

Neuman, K.K., L.A. Henkel and G.W. Page. 2008. Shorebird use of sandy beaches in central California. *Waterbirds* 31:115-121.

Nezlin, N., K. Kamer, J. Hyde and E. Stein. 2009. Dissolved oxygen dynamics in a eutrophic estuary, Upper Newport Bay, California. *Estuarine, Coastal and Shelf Science* 82:139-151.

Nichols, F.H. 1979. Naturak and anthropogenic influences on benthic community structure in San Francisco Bay. pp. 409-426 *in*: T.J. Conomos (ed.), San Francisco Bay: The Urbanized Estuary. Pacific Division of the American Association for the Advancement of Science. San Francisco, CA.

Nichols, F. 1985. Abundance fluctuations among benthic invertebrates in two Pacific estuaries. *Estuaries and Coasts* 8:136-144.

Nichols, F.H. 1985. Increased benthic grazing: An alternative explanation for low phytoplankton biomass in northern San Francisco Bay during the 1976-1977 drought. *Estuarine, Coastal and Shelf Science* 21:379-388.

Nichols, F.H. and J.K. Thompson. 1985. Time scales of change in the San Francisco Bay benthos. *Hydrobiologia* 129:121-138.

Nichols, F.H., J.E. Cloern, et al. 1986. The modification of an estuary. *Science* 231:525-648.

Nichols, F.H. and J.M. Patamat. 1988. The ecology of soft-bottom benthos in San Francisco Bay: A community profile. US Fish and Wildlife Service Biological Report 85 (7.19).

Nichols, F.H, J.K. Thompson and L.E. Schemel. 1990. Remarkable invasion of San Francisco Bay (California, USA) by the Asian clam *Potamocorbula amurensis*. *Marine Ecology Progress Series* 66:95.

Nielsen, S.L., G.T. Banta, M.F. Pedersen (eds). 2004a. Estuarine Nutrient Cycling: The Influence of Primary Producers. Kluwer Academic Publishers. Dordrecht, Netherlands.

Ning, X., J.E. Cloern and B.E. Cole. 2000. Spatial and temporal variability of picocyanobacteria *Synechococcus sp.* in San Francisco Bay. *Limnology and Oceanography* 45:695-702.

Nixon, S.W. 1995. Coastal marine eutrophication: A definition, social causes, and future concerns. *Ophelia* 41:199-219.

Nixon, SW. 2009. Eutrophication and the macroscope. *Hydrobiologia* 629:5-19.

Nixon, S., B. Buckley, S. Granger and J. Bintz. 2001. Responses of very shallow marine ecosystems to nutrient enrichment. *Human and Ecological Risk Assessment* 7:1457-1481.

Nixon, S.W. and B.A. Buckley. 2002. A strikingly rich zone-nutrient enrichment and secondary production in coastal marine ecosystems. *Estuaries* 25:782-796.

NOAA, 2010. San Francisco Bay subtidal habitat goals project. Jointly developed through an interagency partnership between the San Francisco Bay Conservation and Development Commission (BCDC), the California Coastal Conservancy, National Oceanic and Atmospheric Administration (NOAA), and the San Francisco Estuary Partnership. http://www.bcdc.ca.gov/planning/shg/subtidal_habitat.shtml.

NRC 2000. Clean Coastal Waters: Understanding and Reducing the Effects of Nutrient Pollution. Committee on the Causes and Management of Coastal Eutrophication. Ocean Studies Board and Water Science and Technology Board, Commission on Geosciences, Environment, and Resources. National Research Council, National Academy Press, Washington DC

O'Halloran, C., M.W. Silver, T.R. Holman and C.A. Scholin. 2006. *Heterosigma akashiwo* in central California waters. *Harmful Algae* 5:124-132.

Orsi, J.J., and W.L. Mecum. 1996. Food limitation and probably cause of the long-term decline in the abundance of *Neomysis mercedis,* the Opossum Shrimp, in the Sacramento-San Joaquin Estuary. pp. 375-401 *in*: J.T. Hollibaugh (ed.), San Francisco Bay the Ecosystem. Pacific Division of the American Association for the Advancement of Science. San Francisco, CA.

Orth, R.J., T.J.B. Carruthers, W.C. Dennison, C.M. Duarte, J.W. Fourqurean, K.L. Heck, Jr., A.R. Hughes, G.A. Kendrick, W.J. Kenworthy, S. Olyarnik, F.T. Short, M. Waycott and S.L. Williams. 2006. A global crisis for seagrass ecosystems. *BioScience* 56:987-996.

Osterling, M. and L. Pihl. 2001. Effects of filamentous green algal mats on benthic macrofaunal functional feeding groups. *Journal of Experimental Marine Biology and Ecology* 263:159-183.

Paerl, H.W., J. Dyble, J.L. Pinckney, L.M. Valdes, D.F. Millie, P.H. Moisander, J.T. Morris, B. Bendis and M.F. Piehler. 2005. Using microalgal indicators to assess human- and climate-induced ecological change in estuaries. pp. 145-172 *in*: S. Bertone (ed.), Estuarine Indicators. CRC Press, Inc. Boca Raton, FL.

Patamat, F. N. a. M. 1988. The ecology of the soft-bottom benthos of San Francisco Bay: Community profile. US Fish Wildl. Serv. Rep. 85(7.19): 73.

Pearce, S., L. McKee and S. Shonkoff. 2006. Pinole Creek Sediment Source Assessment: Pavon Creeks Sub-basin. A technical report of the Regional Watershed Program, San Francisco Estuary Institute (SFEI), Oakland, California. SFEI Contribution no. 515. www.sfei.org

Pearson, T.H. and R. Rosenberg. 1978. Macrobenthic succession in relation to organic enrichment and pollution of the environment. Oceanography and Marine Biology: Annual Review 16:229-311.

Peckol, P. and J.S. Rivers. 1995. Physiological responses of the opportunistic macroalgae *Cladophora vagabunda* (L.) van den Hoek and Gracelaria tikvahiae (McLachlan) to environmental disturbances associated with eutrophication. *Journal of Experimental Marine Biology and Ecology* 190:1-16.

Peckol, P., B. DeMeo-Anderson, J. Rivers, I. Valiela, M. Maldonado and J. Yates. 1994. Growth, nutrient uptake capacities and tissue constituents of the macroalgae Cladophora vagabunda and Gracilaria tikvahiae related to site-specific nitrogen loading rates. *Marine Biology* 121:175-185.

Pedersen, M.F. and J. Borum. 1996. Nutrient limitation and the importance of nitrogen requirements and nitrogen storage among phytoplankton and species of macroalgae. *Marine Ecology-Progress Series* 142:261-272.

Pergent, G., V. Pasqualini, C. Pergent-Martini, L. Ferrat and C. Fernandez. 2006. Variability of *Ruppia cirrhosa* in two coastal lagoons with differing anthropogenic stresses. *Botanica Marina* 49:10-3-110.

Peterson, B.J. and K.L. Heck, Jr. 2001. Positive interactions between suspension-feeding bivalves and seagrass – a facultative mutualism. *Marine Ecology Progress Series* 213:143-155.

Peterson, C.H., H.C. Summerson and P.B. Duncan. 1984. The influence of seagrass cover on population structure and individual growth rate of a suspension-feeding bivalve, *Mercenaria mercenaria*. *Journal of Marine Research* 42:123-138.

Peterson, C.H., M.C. Kennicutt, II, R.H. Green, P. Montagna, D.E. Harper, Jr., E.N. Powell and P.F. Roscigno. 1996. Ecological consequences of environmental perturbations associated with offshore hydrocarbon production: A perspective on long-term exposures in the Gulf of Mexico. *Canadian Journal of Fisheries and Aquatic Science* 53:2637-2654.

Peterson, D.H., R.E. Smith, et al. 1985. Interannual variability in dissolved inorganic nutrients in northern San Francisco Bay Estuary. *Hydrobiologia* 129:37-58.

Peterson, T. William, et al. 2003. A new climate regime in northeast pacific ecosystems. Washington, DC, ETATS-UNIS, American Geophysical Union.

Pihl, L., S.P. Baden, R.J. Diaz and L.C. Schaffner. 1992. Hypoxia-induced structural changes in the diet of bottom-feeding fish and crustacea. *Marine Biology* 112:349-361.

Pihl, L., A. Svenson, P.-O. Moksnes and H. Wennhage. 1999. Distribution of green algal mats throughout shallow soft bottoms of the Swedish Skagerrak archipelago in relation to nutrient sources and wave exposure. *Journal of Sea Research* 41:281-294.

Pihl, L., I. Isaksson, H. Wennhage and P. Moksnes. 1995. Recent increase of filamentous algae in shallow Swedish bays: Effects on the community structure of epibenthic fauna and fish. *Netherlands Journal of Aquatic Ecology* 29:349-358.

Pitt, K.A., J.E. Purcell, et al. 2009. Influence of decomposing jellyfish on the sediment oxygen demand and nutrient dynamics. Jellyfish Blooms: Causes, Consequences, and Recent Advances. K. Martens (ed), Springer Netherlands. 206: 151-160.

Pitt, K.A., M.J. Kingsford, et al. 2007. Jellyfish modify the response of planktonic assemblages to nutrient pulses. *Marine Ecology Progress Series* 351:1-13.

Posey, M.H., T.D. Alphin and L. Cahoon. 2006. Benthic community responses to nutrient enrichment and predator exclusion: Influence of background nutrient concentrations and interactive effects. *Journal of Experimental Marine Biology and Ecology* 330:105-118.

Powell, T.M., J.E. Cloern and R.A. Walters. 1986. Phytoplankton spatial distribution in south San Francisco Bay: Mesoscale and small-scale variability. pp. 369-383 *in*: D.A. Wolfe (ed.), Estuarine Variability. Academic Press. New York, NY.

Powers, S.P., C.H. Peterson, R.R. Christian, E. Sullivan, M.J. Powers, M.J. Bishop and C. Buzzelli. 2005. Effects of eutrophication on bottom habitat and prey resources of demersal fishes. *Marine Ecology Progress Series* 302:233-243.

Purcell, J.E. 2005. Climate effects on formation of jellyfish and ctenophore blooms: A review. *Journal of the Marine Biological Association of the United Kingdom* 85:461-476.

Purcell, J.U and S. Lo, WT. 2007. Anthropogenic causes of jellyfish blooms and their direct consequences for humans: A review. *Marine Ecology Progress Series* 350:153-174.

Rabalais, N.N. and D. Harper. 1992. Studies of benthic biota in areas affected by moderate and severe hypoxia. In: Proceedings, Workshop on Nutrient Enhanced Coastal Ocean Productivity, NOAA Coastal Ocean Program, 150-153. Texas A&M Sea Grant, TAMU-SG-92-109, College Station, Texas

Rabalais, N.N., R.J. Diaz, L.A. Levin, R.E. Turner, D. Gilbert and J. Zhang. 2010. Dynamics and distribution of natural and human-caused hypoxia. *Biogeosciences* 7:585-619.

Raffaelli, D. 1999. Nutrient enrichment and trophic organisation in an estuarine food web. *Acta Oecologica* 20:449-461.

Raffaelli, D., J. Limia, S. Hull, and S. Pont. 1991. Interactions between the amphipod *Corophium volutator* and macroalgal mats on estuarine mudflats. *Journal of the Marine Biological Association of the United Kingdom* 71:899-908.

Raffaelli, D., S. Hull and H. Milne. 1989. Long-term changes in nutrients, weed mats and shorebirds in an estuarine system. *Cahiers de Biologie Marine* 30:259-270.

Rakoncinski, C.F. 2009. Linking allometric macrobenthic processes to hypoxia using the Peters mass balance model. *Journal of Experimental Marine Biology and Ecology* 381:S13-S20.

Rakocinski, C.F. and G.A. Zapf. 2005. Macrobenthic process-indicators of estuarine condition. pp. 315-331 *in*: S.A. Bortone (ed.), Estuarine Indicators. CRC Press. New York, NY.

Rakocinski, C.F., S.S. Brown, G.R. Gaston, R.W. Heard, W.W. Walker and J.K. Summers. 1997. Macrobenthic responses to natural and contaminant-related gradients in northern Gulf of Mexico estuaries. *Ecological Applications* 7:1278-1298.

Ranasinghe, J.A., K.I. Welch, P.N. Slattery, D.E. Montagne, D.E. Huff, H. Lee, II, J.L. Hyland, B. Thompson, S.B. Weisberg, J.M. Oakden, D.B. Cadien and R.G. Velarde. 2011. Habitat-related benthic macrofaunal assemblages of bays and estuaries of the western United States. Integrated Environmental Assessment and Management

Ranasinghe, J.A., S.B. Weisberg, R.W. Smith, D.E. Montagne, B. Thompson, J.M. Oakden, D.D. Huff, D.B. Cadien, R.G. Velarde and K.J. Ritter. 2009. Calibration and evaluation of five indicators of benthic community condition in two California bay and estuary habitats. *Marine Pollution Bulletin* Volume, 59. Issue, 1-3 (5-13)

Rand, G.M., P.G. Wells and L.S. McCarty. 1995. Introduction to aquatic toxicology. pp. 3-70. *in*: G.M. Rand (eds.), Fundamentals of Aquatic Toxicology: Effects, Environmental Fate, and Risk Assessment, 2nd Edition. Taylor and Francis. Washington, DC.

Rantz, S.E. 1971, Mean annual precipitation and depth-duration frequency data for the San Francisco Bay region, California: U.S. Geological Survey Open-File Report, 23 p.

Ray, M.T., J.T. Kulongoski and K. Belitz. 2009. Ground-Water Quality Data in the San Francisco Bay Study Unit, 2007: Results from the California GAMA Program. USGS Data Series Report 396. 94pp.

Rees, J.T. et al. 2002. Survey of Gelatinous Zooplankton (“Jellyfish”) in the San Francisco Estuary: Initial Field Survey, Annotated Species Checklist, and Field Key." Interagency Ecological Program for the San Francisco Estuary Technical Report 70 August 2002.

Rees, J.T. and L.-A. Gershwin. 2000. Non-indigenous hydromedusae in California's upper San Francisco Estuary: Life cycles, distribution, and potential environmental impacts, Consejo Superior de Investigaciones CientÃ-ficas, CSIC: Instituto de Ciencias del Mar.

Resh, V.H., R.H. Norris, et al. 1995. Design and implementation of rapid assessment approaches for water resource monitoring using benthic macroinvertebrates. *Australian Journal of Ecology* 20:108-121.

Rhoads, D.C. and J.D. Germano. 1986. Interpreting long-term changes in benthic community structure: A new protocol. *Hydrobiologia* 142:291-308.

Rhoads, D.C. and L.F. Boyer. 1982. The effects of marine benthos on physical properties of sediments a successional perspective. pp. 3-52 *in*: P.L. McCall and M.J.S. Tevesz (eds.), Animal-Sediment Relations the Biogenic Alteration of Sediments. Plenum Press. New York, NY.

Richardson, J. Anthony, et al. 2010. The jellyfish joyride: Causes, consequences and management responses to a more gelatinous future. Trends in Ecology & Evolution, Volume 24, Issue 6, 312-322

Ringwood, A.H. and C.J. Keppler. 1998. Seed clam growth: An alternative sediment bioassay developed during EMAP in the Carolinian Province. *Environmental Monitoring and Assessment* 51:247-257.

Rivers, J.S. and P. Peckol. 1995. Summer decline of *Ulva Lactuca* (Chlorophyta) in a Eutrophic embayment—interactive effects of temperature and nitrogen availability. *Journal of Phycology* 31: 223-228.

Rosenberg, R., B. Hellman and B. Johansson. 1991. Hypoxic tolerance of marine benthic fauna. *Marine Ecology Progress Series* 79:127-131.

Rosenberg, R. 1976. Benthic faunal dynamics during succession following pollution abatement in a Swedish estuary. *Oikos* 27:414-427.

Russell, P.P., T.A. Bursztynsky, L.A. Jackson and E.Y. Leong. 1980. Water and waste inputs to San Francisco Estuary – A historical perspective. Sixty First Annual Meeting Pacific Division / American Association for the Advancement of Science, Davis, California. June 1980. 20pp.

RWQCB, 2007. San Francisco Bay Basin (Region 2) Water Quality Control Plan (Basin Plan). California Regional Water Quality Control Board San Francisco Bay Region. http://www.swrcb.ca.gov/sanfranciscobay/water_issues/programs/basin_plan/docs/basin_plan07.pdf

Sagasti, A., L.C. Schaffner and J.E. Duffy. 2001. Effects of periodic hypoxia on mortality, feeding, and predation in an estuarine epifaunal community. *Journal of Experimental Marine Biology and Ecology* 258:257-283.

San Francisco Estuary Institute (SFEI). 2009. The Pulse of the Estuary: Monitoring and Managing Water Quality in the San Francisco Estuary. SFEI Contribution 583. San Francisco Estuary Institute. Oakland, CA.

Sanders, H.L. 1958. Benthic studies in Buzzards Bay. I Animal-sediment relationships. *Limnology and Oceanography* 3:245-258.

Sanger, D.M., A.F. Holland and G.I. Scott. 1999a. Tidal creek and salt marsh sediments in South Carolina coastal estuaries: I. distribution of trace metals. *Archives of Environmental Contamination and Toxicology* 37:445-457.

Sanger, D.M., A.F. Holland and G.I. Scott. 1999b. Tidal creek and salt marsh sediments in South Carolina coastal estuaries: II. distribution of organic contaminants. *Archives of Environmental Contamination and Toxicology* 37:458-471.

Scanlan, C.M., J. Foden, E. Wells and M.A. Best. 2007. The monitoring of opportunistic macroalgal blooms for the water framework directive. *Marine Pollution Bulletin* 55:162-171.

State Coastal Conservancy SCC. 2010. San Francisco Bay Subtidal Habitat Goals Project Oakland, CA, State Coastal Conservancy (SCC): 208.

Schaeffer, K.M., K; and Consentino-Manning, N. 2007. Report on subtidal habitats and associated biological taxa in San Francisco Bay. Santa Rosa, CA, National Oceanic and Atmospheric Association-National Marine Fisheries Service.

Schaffner, L.C. 2010. Patterns and rates of recovery of macrobenthic communities in a polyhaline temperate estuary following sediment disturbance: Effects of disturbance severity and potential importance of non-local processes. *Estuaries and Coasts* 33:1300-1313.

Schemel, L.E., S.W. Hager and D. Childers, Jr. 1996. The supply and carbon content of suspended sediment from the Sacramento River to San Francisco Bay. pp. 237-260 *in*: J.T. Hollibaugh (ed.), San Francisco Bay the Ecosystem. Pacific Division of the American Association for the Advancement of Science. San Francisco, CA.

Schoellhamer, D. 2009. Suspended sediment in the Bay: Past a tipping point. In The Pulse of the Estuary: Monitoring and Managing Water Quality in San Francisco Estuary. San Francisco Estuary Institute (SFEI).

Schramm, W. 1999. Factors influencing seaweed responses to eutrophication: Some results from EU-project EUMAC. *Journal of Applied Phycology* 11:69-78.

Seitz, R.D., D.M. Dauer, R.J. Llanso' and W.C. Long. 2009. Broad-scale effects of hypoxia on benthic community structure in Chesapeake Bay, USA. *Journal of Experimental Marine Biology and Ecology* 381:S1-S12

Setchell, W.A. 1922. *Zostera marina* in its relation to temperature. *Science* 56:575-577.

San Francisco Public Utilities Commission. 1997. Technical memorandum NO. 18 prepared for: DATE: May 28, 1997 (Review Draft) SUBJECT: Westside Basin Groundwater Model

Sfriso, A., B. Pavoni, A. Marcomini and A.A. Orio. 1992. Macroalgae, nutrient cycles, and pollutants in the Lagoon of Venice. *Estuaries* 15:517-528.

Sfriso, A., A. Marcomini and B. Pavoni. 1987. Relationships between macroalgal biomass and nutrient concentrations in a hypertrophic area of the Venice Lagoon Italy. *Marine Environmental Research* 22:297-312.

Sheath, R.G. and K.M. Cole. 1992. Biogeography of stream algae in North America. *Journal of Phycology* 28:448-460.

Shellem, B.H. and M.N. Josselyn. 1982. Physiological ecology of enteromorpha-clathrata (roth) grev on a salt-marsh mudflat. *Botanica Marina* 25:541-549.

Shili, A., N.B. Maïz, C.F. Boudouresque and E.B. Trabelsi. 2007. Abrupt changes in Potamogeton and Ruppia beds in a Mediterranean Lagoon. *Aquatic Botany* 87:181-188.

Short, F.T., D.M. Burdick and J.E. Kaldy. 1995. Mesocosm experiments quantify the effects of eutrophication on eelgrass, *Zostera marina*. *Limnology and Oceanography* 40:740-749.

Short, F.T., G.E. Jones and D.M. Burdick. 1991. Seagrass decline: Problems and solutions. Coastal Zone '91 Conference - ASCE. Long Beach, CA/July 1991. Pp 439-453.

Shouse, M. K. P., Francis ; Thompson, Janet K. 2003. Near-Field Receiving Water Monitoring of A Benthic Community Near the Palo Alto Water Quality Control Plant in South San Francisco Bay: February 1974 Through December 2002." U.S. Geological survey Open-File Report 02-394.

Sigleo, A.C. and S.A. Macko. 2002. Carbon and nitrogen isotopes in suspended particles and colloids, Chesapeake and San Francisco Estuaries, U.S.A. *Estuarine, Coastal and Shelf Science* 54:701-711.

Smetacek, V. and J.E. Cloern. 2008. On phytoplankton trends: How are phytoplankton at coastal sites around the world responding to ongoing global change? *Science* 319:1346-1348.

Smith, R.W., M. Bergen, S.B. Weisberg, D. Cadien, A. Dalkey, D. Montagne, J.K. Stull and R.G. Velarde. 2001. Benthic response index for assessing infuanal communities on the southern California mainland shelf. *Ecological Applications* 11:1073-1087.

Smith, S.V. and J.T. Hollibaugh. 2006. Water, salt, and nutrient exchanges in San Francisco Bay. *Limnology Oceanography* 51:504-517.

Sommer, C. E. 1995. Invertebrate introductions in marine habitats: Two species of *hydromedusae (Cnidaria)* native to the Black Sea, *Maeotias inexspectata* and *Blackfordia virginica*, invade San Francisco Bay. *Marine Biology* 122:279-288.

Staver, K.W. 1984. Responses of epiphytic algae to nitrogen and phosphorus enrichment and effects on productivity of the host plant, *Potamogeton perfoliatus* (L.), in estuarine waters. Masters Thesis, University of Maryland. College Park, MD.

Steneck, R.S., M.H. Graham, et al. 2002. Kelp forest ecosystems: Biodiversity, stability, resilience and future. *Environmental Conservation* 29: 436-459.

Sterner, R.W. and J.J. Elsner. 2002. Ecological Stoichiometry: The biology of Elements from Molecules to the Biosphere. Princeton University Press. Princeton, NJ.

Sutula, M. 2011. Review of candidate indicators for development of nutrient numeric endpoints in California estuaries. Technical Report 646. Southern California Coastal Water Research Project. Costa Mesa, CA.

Tennant, G.A. 2006. Experimental effects of ammonium on eelgrass (*Zostera marina L.*) shoot density in Humboldt Bay, California. Masters Thesis, Humboldt State University. Humboldt, CA.

Tett, P., R. Gowen, D. Mills, T. Fernandes, L. Gilpin, M. Huxham, K. Kennington, P. Read, M. Service, M. Wilkinson and S. Malcom. 2007. Defining and detecting undesirable distrubance in the context of marine eutrophication. *Marine Pollution Bulletin* 55:282-297.

Thompson, B., S. Lowe and H. Lee II. 1999. Impacts of Nonindigenous Species on Subtidal Benthic Assemblages in the San Francisco Estuary. Report to US EPA Region IX. SFEI contribution #329. www.sfei.org

Thompson, B. and S. Lowe. 2004. Assessment of macrobenthos response to sediment contamination in the San Francisco Estuary, California, USA. *Environmental Toxicology and Chemistry* 23:2178-2187.

Thompson, B., T. Adelsbach, C. Brown, J. Hunt, J. Kuwabara, J. Neale, H. Ohlendorf, S. Schwarzbach, R. Spies and K. Taberski. 2007. Biological effects of antropogenic contaminants in the San Francisco Estuary. *Environmental Research* 105:146-174.

Thompson, J.K. 1999. The effect of infaunal bivalve grazing on phytoplankton bloom development in south San Francisco Bay. Ph.D. Dissertation, Stanford University. Stanford, CA.

Thompson, J.K., F. Parchaso and M.K. Shouse. 2002. Near-Field Receiving Water Monitoring of A Benthic Community Near the Palo Alto Water Quality Control Plant in South San Francisco Bay: February 1974 Through December 2000. U.S. Geological Survey Open-File Report 02-394.

Thompson, J.K. 2005. One estuary, one invasion, two responses: Phytoplankton and benthic community dynamics determine the effect of an estuarine invasive suspension-feeder. pp. 291-316 *in*: R.F. Dame and S. Olenin (eds.), The Comparative Roles of Suspension-Feeders in Ecosystems. Springer.

Thompson, J.K., J.R. Koseff, S.G. Monismith and L.V. Lucas. 2008. Shallow water processes govern system-wide phytoplankton bloom dynamics: A field study. *Journal of Marine Systems* 74:153-166.

Thompson, M.L. and L.C. Schaffner. 2001. Population biology and secondary production of the suspension feeding polychaete Chaetopterus cf. variopedatus: Implications for benthic-pelagic coupling in lower Chesapeake Bay. *Limnology and Oceanography* 46:1899-1907.

Thomsen, M.S. and K. McGlathery. 2006. Effects of accumulations of sediments and drift algae on recruitment of sessile organsims associated with oyster reefs. *Journal of Experimental Marine Biology and Ecology* 328:22-34.

Trainer, V., B. Hickey and R. Horner. 2002. Biological and physical dynamics of domoic acid production off the Washington coast. *Limnology and Oceanography* 47:1438-1446.

Tubbs, C. and J. Tubbs. 1980. Wader and shelduck feeding distribution in Langstone Harbour, Hampshire. *Bird Study* 27:239-248.

Turner, R.E., N. Qureshi, N.N Rabalais, Q. Dortch, D. Justic, R.F. Shaw and J. Cope. 1998. Fluctuating silicate: nitrate ratios and coastal plankton food webs. Proceeding of the National Academy of Sciences 95:13048-13.

Twilley, R.R. 1985. The exchange of organic carbon in basin mangrove forests in a southwest Florida estuary. *Estuarine, Coastal and Shelf Science* 20:543-557.

Underwood, G.J.C. and J. Kromkamp. 1999. Primary production by phytoplankton and microphytobenthos in estuaries. pp. 306 *in*: D.B. Nedwell and D.G. Raffaell (eds.), Advances in Ecological Research Estuaries. Academic Press. London, UK

United States Environmental Protection Agency (USEPA). 1998. National Strategy for the Development of Regional Nutrient Criteria. Office of Water, U.S. Environmental Protection Agency. EPA 822-R-98-002.

USEPA. 2001. Nutrient criteria technical guidance manual: Estuarine and coastal marine waters. EPA-822-B-01-003. Office of Water, Office of Science and Technology, U.S. Environmental Protection Agency (USEPA). Washington, DC.

USEPA. 2006. Technical Approach to Develop Nutrient Numeric Endpoints for California. Tetra Tech, INC. Lafayette, CA.

USEPA. 2007. Technical approach to develop nutrient numeric endpoints for california estuaries. Technical Report 516. TetraTech and Southern California Coastal Water Research Project. Costa Mesa. CA.

USEPA. 2008. (540) Developing nutrient numeric endpoints and TMDL tools for California estuaries: An implementation plan. Technical Report 540. Southern California Coastal Water Research Project and TetraTech, Inc. Costa Mesa, CA.

Valiela, I. 1995. Marine Ecological Processes. 2nd edition. Springer-Verlag. New York, NY.

Valiela, I., K. Foreman, M. LaMontagne, D. Hersh, J. Costa, P. Peckol, B. DeMeo-Anderson, C. D'Avanzo, M. Babione, C.-H. Sham, J. Brawley and K. Lajtha. 1992. Couplings of watersheds and coastal waters: Sources and consequences of nutrient enrichment in Waquoit Bay, Massachusetts. *Estuaries* 15:443-457.

Valiela, I., J. McClelland, J. Hauxwell, P.J. Behr, D. Hirsch and K. Foreman. 1997. Macroalgal blooms in shallow estuaries: Controls and ecophysiological and ecosystem consequences. *Limnology and Oceanography* 42:1105-1118.

Van Dolah, R.F., D.E. Chestnut and G.I. Scott. 2000. A Baseline Assessment of Environmental and Biological Conditions in Broad Creek and the Okatee River, Beaufort County, South Carolina. Final Report. South Carolina Department of Health and Environmental Control. Columbia, SC.

Van Dolah, R.F., J.L. Hyland, A.F. Holland, J.S. Rosen and T.R. Snoots. 1999. A benthic index of biological integrity for assessing habitat quality in estuaries of the southeastern USA. *Marine Environmental Research* 48:269-283.

Van Katwijk, M.M., L.H.T. Vergeer, G.H.W. Schmitz and J.G.M. Roelofs. 1997. Ammonium toxicity in eelgrass *Zostera marina*. *Marine Ecology Progress Series* 157:159-173.

Viaroli, P., M. Bartoli, G. Giordani, M. Naldi, S. Orfanidis and J.M. Zaldívar. 2008. Community shifts, alternative stable states, biogeochemical controls and feedbacks in eutrophic coastal lagoons: A brief overview. *Aquatic Conservation Freshwater and Marine Ecosystems* 18:S105-25 S117.

Virnstein, R.W. 1979. Predation on estuarine infauna: Response patterns of component species. *Estuaries* 2:69-86.

Vitousek, P., H. Mooney, J. Lubechenko and J. Melillo. 1997. Human domination of earth's ecosystem. *Science* 277: 494-499.

Wankel, S.D., C. Kendall, et al. 2006. Nitrogen sources and cycling in the San Francisco Bay Estuary: A nitrate dual isotopic composition approach. *Limnology and Oceanography* 51:1654-1664.

Ward, D.H. 1983. The relationship of two seagrasses: *Zostera marina* and *Ruppia maritima* to the Black Brant, *Branta bernicla nigricans*, San Ignacio lagoon, Baja California, Mexico. Masters Thesis, University of Oregon. Eugene OR.

Water Board. 2010. Groundwater Committee. East Bay plain groundwater basin beneficial use evaluation report. San Francisco Bay Regional Water Quality Control Board. Final Report.

Weisberg, S.B., J.A. Ranasinghe, D.M. Dauer, L.C. Schaffner, R.J. Diaz and J.B. Frithsen. 1997. An estuarine benthic index of biotic integrity (B-IBI) for Chesapeake Bay. *Estuaries* 20:149-158.

Welch, E.B., R.R. Horner, et al. 1989. Prediction of nuisance periphytic biomass: A management approach. *Water Research* 23:401-405.

Wells, E., M. Wilkinson, P. Wood and C. Scanlan. 2007. The use of macroalgal species richness and composition on intertidal rocky shores in the assessment of the ecological quality under the European Water Framework Directive. *Marine Pollution Bulletin* 55:151-161.

Welsh, D., R. Dunn, et al. 2009. Oxygen and nutrient dynamics of the upside down jellyfish and its influence on benthic nutrient exchanges and primary production. *Hydrobiologia* 635:351-362.

Wennhage, H. and L. Pihl. 1994. Substratum selection by juvenile plaice (*Pleuronectes platessa L.*): Impact of benthic microalgae and filamentous macroalgae. *Netherlands Journal of Sea Research* 32:343-351.

West, E.J., D.T. Welsh, et al. 2009. Influence of decomposing jellyfish on the sediment oxygen demand and nutrient dynamics. *Hydrobiologia* 616:151-160.

Wilbur, D.H., G.I. Ray, D.G. Clarke and R.J. Diaz. 2008. Responses of benthic infauna to large-scale sediment disturbance in Corpus Christi Bay, Texas. *Journal of Experimental Marine Biology and Ecology* 365:13-22.

Wilkerson, F.P., R.C. Dugdale, V.E. Hogue and A. Marchi. 2006. Phytoplankton blooms and nitrogen productivity in San Francisco Bay. *Estuaries and Coasts* 29:401-416.

Williams, S.L. and M.H. Ruckelshaus. 1993. Effects of nitrogen availability and herbvory on eelgrass (*Zostera marina*) and epiphytes. *Ecology* 74:904-918.

Williams, M., B. Longstaff, C. Buchanan, D.R. Llanso and W. Dennison. 2009. Development and evaluation of a spatially-explicit index of Chesapeake Bay health. *Marine Pollution Bulletin* 59:14-25.

Williams, M.R., S. Filoso, B.J. Longstadd and W.C. Dennison. 2010. Long-term trends of water quality and biotic metrics in Chesapeake Bay: 1986 to 2008. *Estuaries and Coasts* 33:1279-1299.

Wilson, J.G. 1994. The role of bioindicators in estuarine management. *Estuaries* 17:94-101.

Wintzer, A., M.H. Meek, P.B. Moyle and B. May. 2010. Ecological insights into the polyp stage of non-native hydrozoans in the San Francisco Estuary. *Aquatic Ecology* 45:151–161 Wyllie- Echeverria, S. and P. J Rutten. 1989. Inventory of eelgrass (*Zostera marina L.*) in San Francisco/San Pablo Bay. Report SWR-89-05. Southwest Region, NOAA Administrative.

Wyllie-Echeverria, S. and M. Fonseca. 2003. Eelgrass (*Zostera marina L.*) research in San Francisco Bay, California from 1920 to present. National Centers for Coastal Ocean Science. February.

Young, D. 2009. Macroalgal interaction with the seagrasses Zostera marina and Thalassia testudinum. Seagrasses and Protective Criteria: A review and assessment of research Status, Office of Research and Development, National Health and Environmental Effects Research Laboratory.

Zajac, R.N. and R.B. Whitlatch. 1982. Responses of estuarine infauna to disturbance. II Spatial and temporal variation of succession. *Marine Ecology Progress Series* 10:15-27.

Zaldivar, J.-M., A.C. Cardoso, et al. 2008. Eutrophication in transitional waters: An overview. *Transitional Waters Monographs* 1:1-78.

Zedler, J.B. 1996. Coastal mitigation in southern California: The need for regional restoration strategy. *Ecological Applications* 6:84-93.

Zimmerman, R.C., J.L. Regguzoni and R.S. Alberte. 1995. Eelgrass (*Zostera marina L.*) transplants in San Francisco Bay: Role of light availability on metabolism, growth and survival. *Aquatic Botany* 51:67-86

Zimmerman, R.C., J.L. Reguzzoni, S. Wyllie-Echeverria, M. Josselyn and R.S. Alberte. 1991. Assessment of environmental suitability for growth of *Zostera marina L.* (eelgrass) in San Francisco Bay. *Aquatic Botany* 39:353-366.

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SRCSD

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April 27, 2012

Mr. Mike Chotkowski
Department of the Interior
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San Francisco Bay-Delta Fish and Wildlife Office
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Via email to mike.chotkowski@fws.gov

Subject: Endangered and Threatened Wildlife and Plants; 12-month Finding on a Petition to List the San Francisco Bay-Delta Population of the Longfin Smelt as Endangered or Threatened, Docket No. FWS-R8-ES-2008-0045

Dear Mr. Chotkowski:

The Sacramento Regional County Sanitation District (SRCSD) has reviewed the "Endangered and Threatened Wildlife and Plants; 12-month Finding on a Petition to List the San Francisco Bay-Delta Population of the Longfin Smelt as Endangered or Threatened, Docket No. FWS-R8-ES-2008-0045" (Finding) and has significant concerns with the document. SRCSD provides wastewater collection and treatment services to over 1.3 million residents in the greater Sacramento area. Our mission is to protect human health and keep the Sacramento River clean and safe. We take our mission seriously and work on a daily basis to meet our obligations to protect water quality and beneficial uses in the Delta. Our excellent compliance record with our discharge permit speaks to this commitment and performance.

SRCSD is providing these comments to make sure that the United States Fish and Wildlife Service (USFWS) has the best available science on water quality in the Bay-Delta. With this new information, we request USFWS to make the appropriate changes to the Finding. In summary, SRCSD has identified the following concerns with the Finding:

- Failure to consider or document best available scientific information
- Misleading presentation of information
- Lack of scientific analysis to support statements made in the Finding
- Internal inconsistencies in the Finding

Overall, in the Finding, there are statements of certainty regarding the potential effect of ammonia/ammonium on the Delta food web and longfin smelt that are incongruent with the conclusions of other scientific studies published on the role ammonia plays in the Bay-Delta. For example, the March 29, 2012, National Research Council's (NRC) report on *Sustainable Water and Environmental Management in the California Bay-Delta,* discusses alternative hypotheses to the "ammonia inhibition" hypothesis to explain food web productivity. Analyses of ammonia/ammonium in other Delta planning documents, such as the Bay Delta Conservation Plan Effects Analysis and the 5th Draft of the Delta Plan also take a more measured approach in portraying

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the state of the science regarding ammonia's role in the Bay-Delta. Additionally, the Finding also includes problematic statements related to other water quality constituents. In reaction to these and other issues with the Finding, SRCSD provides the detailed comments below to encourage USFWS to make changes to its Finding to more accurately represent the current scientific understanding regarding the role of ammonia/ammonium and other water quality constituents in the Bay Delta.

Best Available Scientific Information

Numerous statements are made in the document regarding the effects of ammonium on the Bay-Delta food web that do not reflect the best available scientific information. First, several citations are made to Dugdale et al. 2007 as a definitive source regarding the effects of ammonium on phytoplankton blooms in Suisun Bay. For instance, on Page 80, the document states:

> "In addition to direct effects on fish, ammonia in the form of ammonium has been shown to alter the food web by adversely impacting phytoplankton and zooplankton dynamics in the estuary ecosystem… ammonium impairs primary productivity by reducing nitrate uptake in phytoplankton. Ammonium's negative effect on the food web has been documented in the longfin smelt rearing areas of San Francisco Bay and Suisun Bay (Dugdale et al. 2007, pg. 26-28).

This assertion is repeated on page 124. The Dugdale study in 2007 did not consider or establish the importance of the cited "ammonium inhibition" effect on the Delta food web. In fact, the numerous subsequent studies by Dugdale, Parker and Wilkerson performed to date have failed to establish the importance of this hypothesized effect. The "ammonium inhibition" effect in Suisun Bay was carefully reviewed in a July 2011 report prepared by the San Francisco Estuary Institute and Southern California Coastal Water Research Project for the San Francisco Bay Numeric Nutrient Endpoint (NNE) effort, which is being led by the San Francisco Bay Regional Water Quality Control Board[1]. The following statements were made in the final San Francisco Bay NNE report regarding the "ammonium inhibition" hypothesis (bold emphasis added):

> "…**the ecological importance of ammonium inhibition of spring diatom blooms is not well understood** relative to factors known to control primary productivity…"
>
> "In SF Bay, the biomass associated with phytoplankton, measured as surface water chlorophyll *a* concentration, varies in space and time in response to nutrient availability from external loads and internal regeneration, grazing, stratification, water temperature, tidal energy, transparency, wind/wave energy, the availability of seed cysts, UV radiation effects on nitrate versus ammonium assimilation perhaps due to disruptions of enzyme pathways, differential uptake of nitrate and ammonium by larger versus smaller cells, inhibition of nitrate uptake by ammonium, predation by benthic invertebrates, and variations in the phase of the Pacific Decadal Oscillation and related changes to top down predation of benthic invertebrates."
>
> "…the effect of ammonium inhibition on phytoplankton productivity throughout the Bay has not been modeled vis-a-vis other contributing factors…the next logical step is to develop models that synthesize understanding of the relative importance of ammonium and urea versus other factors controlling phytoplankton assemblages."
>
> "Elevated ammonium concentrations have been suggested as a major mechanism by which spring diatom blooms appear to be suppressed in the North Bay and Lower Sacramento River…Despite this evidence, **the ecological importance of ammonium inhibition of spring diatom blooms is not well**

[1] McKee, Lester; et. al. Numeric Nutrient Endpoint Development for the San Francisco Bay Estuary: Literature Review and Data Gaps, June 2011. Pgs 46, 147. 153-154
http://www.waterboards.ca.gov/sanfranciscobay/water_issues/programs/planningtmdls/amendments/estuarineNNE/SFBay%20NNE%20Literature%20Review%20Draft%20Final%204-27-%202011.pdf

understood relative to factors known to control primary productivity, particularly in other regions of the Bay where water column chlorophyll *a* appears to be increasing. Thus, **the linkage between ammonium concentrations and Bay beneficial uses is not at this time universally accepted**. San Francisco Bay Technical Advisory Team (TAT) members agree that additional data synthesis is required to better understand the role of ammonium in SF Bay."

It is important to note that members of the TAT responsible for scientific review of and input on the NNE report include Dr. James Cloern, a highly recognized expert in San Francisco Bay ecology, and two members from the Romburg Tiburon Center, including Dr. Dugdale. The cited statements and recommendations of the NNE report should therefore be interpreted as current prevailing scientific opinion regarding the role of ammonium in Suisun Bay phytoplankton dynamics.

The recent NRC report performed a review of the available scientific information and concluded there are alternative hypotheses to the "ammonia inhibition" hypothesis, such as invasive species, residence time, optical properties, in addition to nutrients. They state:

> "Most likely, there are interacting environmental drivers at play in controlling the qualitative and quantitative makeup of food supplies at the base of the Delta and SFO Bay food webs."[2]

Their overall conclusion regarding nutrients is that they are of increasing concern from a water quality and food web perspective, which also interact with hydrologic changes in the Delta and its tributaries.

A second category of concern related to the information relied upon in the Finding is on page 80. Work by Teh et al 2011 is cited as a basis for concern that ammonium is adversely affecting prey organisms that are important to larval and juvenile fishes. In fact, the Teh et al 2011 study addresses ammonia toxicity to one of the copepods (*Pseudodiaptomus forbesi)* which is present in the Lower Sacramento River and Suisun Bay. The reliance on results from the 2011 Teh et al study does not recognize best available science regarding this issue. The subject report by Teh et al has not been formally peer reviewed, and an independent critique of the report has been performed by Pacific Ecorisk (attached) which raises serious questions as to the validity of key results in the study. Examples of the issues raised by Pacific Ecorisk regarding the 2011 Teh et al study include the following:

> Validity of toxicity threshold values. Analysis of the number of nauplii and number of juveniles produced during the chronic (31-day) exposure is flawed at a very fundamental level. The reported results for 'total number' and 'mean number per female' for the nauplii and juveniles are incorrect, and analyses of that data (including the No Observed Effect Concentration and Lowest Observed Effect Concentration resulting from those analyses) are similarly incorrect.
>
> In the testing of gravid females exposed to varying concentrations of total ammonia nitrogen, the number of nauplii produced after 3, 5 and 7 days were counted (recall that counts made on progressive count days are not all new organisms). The report focused on differences observed after 3 days. However, by extending the observation period beyond 3 days, it becomes evident that not only is there no reduction in nauplii production at 0.36 mg/L Total Ammonia Nitrogen (TAN) over the entire test period, but nauplii production actually *increased* relative to the control treatment.
>
> Finally, independent statistical analysis of the "Total # of Adults Counted on Day 31" data indicate *there are no significant differences between any of the test treatments*. This is due to the EXTREMELY high variability of the data set.

[2] Pre-publication Copy, March 29, 2012 National Research Council report on Sustainable Water and Environmental Management in the California Bay-Delta, pg 68.

The above information is important since it brings into question the validity of the key threshold value of 0.36 mg/L TAN that is relied upon to support a finding of copepod toxicity in the Lower Sacramento River and Suisun Bay.

Deficiencies in Methodology. The description of experimental methodology in the Teh et al. (2011) report is inadequate. Much of the experimental approach is not described, leaving it to the reader's imagination to assume what was actually done. Given the novel testing approach (i.e., these tests are acknowledged by the author as being "non-standard", and that the "test methods are considered developmental"), and the potential use of the information generated by this study by regulatory decision-makers, this absence of experimental description should be considered unacceptable.

The testing program did not include the use of reference toxicant tests. It should be recognized *a priori* that different tests with the same species can exhibit variable responses between tests. This results from several factors, with differences in organism sensitivity being an obvious possibility. This is precisely why the performance of concurrent reference toxicant testing is so essential in any type of testing for which the test results might be used for regulatory decision-making.

The description of "moderately hard water" from the EPA test manual[3] cited by Teh et al. does not match the alkalinity and hardness concentrations reported in Table 4 of the report. Of particular note is the reported hardness of 206.7 mg/L, which is over twice the EPA's 80-100 mg/L range. This suggests that the moderately hard water used in this test was prepared improperly.

One of the major problems in trying to conduct a toxicity test at a constant pH is the tendency of test solutions (including moderately hard water) to drift upwards (if the pH has been adjusted to <pH8) or downwards (if the pH had been adjusted to >pH8) during each 24-hr exposure period. It is exceedingly difficult to maintain constant pH at the adjusted level. Depending upon the magnitude of the pH drift that likely occurred in the Teh et al. pH7.4 experiment (and in the subsequent pH7.8 experiment), the final conclusions regarding toxicity of ammonia at a specific pH or calculations of un-ionized ammonia based on the reported data could be different than reported.

Deficiencies in Reporting. Information in various tables and figures is incorrect and/or misleading. For many of the test result descriptive figures, the data points (and error bars) represented in the figures do not appear to match the data in the tables or the raw data provided in the appendices. In some cases, hypothetical modeled data (e.g., LC point estimates) are shown in the figures as if they were actual experimental data. Data reported in various tables are apparently mislabeled and/or are misleading.

Given these unresolved issues with the Teh et al. 2011 report, the USFWS should probably not base its determinations regarding the effect of ammonia on longfin smelt or its habitat on this report.

Misleading Presentation of Information

The Finding is very uneven in its assessment of the impact of ammonium on longfin smelt in comparison to its assessment of the impacts of other factors, such as entrainment. On Page 120 of the Finding, it is stated:

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[3] U.S. Environmental Protection Agency; *Short-Term Methods for Estimating the Chronic Toxicity of Effluents and Receiving Waters to Freshwater Organisms*, Fourth Edition, EPA-821-R-02-013 (2002).

> "Entrainment losses at the SWP and CVP water export facilities are a known source of mortality of longfin smelt and other pelagic fish species in the Bay Delta, although the full magnitude of entrainment losses and population-level implications of these losses is still not fully understood…Baxter et al. (2010, p.62) hypothesize that entrainment is having an important effect on the longfin smelt population during winter…**However, Baxter et al. (2010 p.63) conclude that these losses have yet to be placed in a population context, and no conclusions can be drawn regarding their effects on recent longfin smelt abundance.**" (bold emphasis added)

The statement shown in bold above is equally or more applicable to ammonia; ammonia is not a known source of mortality of pelagic fish species and other effects of ammonia are much less certain than are the effects of entrainment. Also regarding entrainment effects, page 120 states:

> "Fujimura (2009) estimated cumulative longfin smelt entrainment at the SWRP facility between 1993 and 2008 at 1,376,432 juveniles and 11,054 adults, and estimated a 97.6 percent of juveniles and 95 percent of adults entrained were lost. Fujimura (2009) estimated cumulative longfin entrainment at the CVP facility between 1993 and 2008 at 224,606 juveniles and 1,325 adults, and estimated that 85.2 percent of the juveniles and 82.1 percent of the adults entrained were lost."

> "…efforts to reduce past delta smelt entrainment loss through the implementation of the 2008 delta smelt biological opinion for SWP and CVP operations **may have** reduced longfin smelt entrainment losses, incidentally providing a benefit to longfin smelt. These efforts to manage entrainment losses in drier years, when entrainment risk is greater, substantially reduce the threat of entrainment for longfin smelt….Estimates of entrainment have shown that it **may have been a threat** to the Bay-Delta longfin smelt DPS in the past." (bold emphasis added)

In the face of observed effects due to entrainment, the document seems to downplay the future threat of this historic stressor. In contrast, the document treats the effects and threat associated with ammonia as a certainty, despite the actual scientific uncertainty and absence of observed impacts of ammonia on fish.

> On Page 121, the document states "Entrainment is no longer considered a threat to longfin in the Bay-Delta because of current regulations. Efforts to reduce delta smelt entrainment loss…have **likely reduced longfin smelt entrainment losses**….Although larval and adult longfin smelt are lost as a result of entrainment in the water export facilities in the delta, we conclude that the risk of entrainment is generally greatest when X2 is upstream and export volumes from the CVP and SWP pumps are high. Therefore, we have determined that longfin smelt are not currently threatened by entrainment, nor do we anticipate longfin smelt will be threatened by entrainment in the future." (bold emphasis added)

This analysis raises obvious questions about the validity of this conclusion and the contrast to the treatment of ammonia effects in the document. It is clear that the avoidance of significant entrainment losses of juvenile and adult longfin smelt are entirely dependent on a new regulatory approach, which has only recently been implemented and has been strongly challenged on both legal and political levels. This regulatory shield appears to be tenuous, considering the legal and political challenges, which could provide an avenue to return to the historic significant mortalities of longfin smelt associated with entrainment noted by Fujimura in 2009.

Inexplicably, this regulatory approach is deemed to be entirely adequate and secure, while actions under the Clean Water Act, responding to concerns regarding the impacts of ammonia that have not been directly observed and which are based on incomplete knowledge regarding food web effects, have been deemed to be insufficient. This raises obvious questions about "a level playing field" and the inconsistent treatment of ammonia effects in this document, as exemplified below.

While no definitive studies have been performed to establish the "population level" effects of ammonium, and with serious issues regarding the validity of the science relied upon to make a finding of ammonia effects on longfin smelt, the report contains the following unqualified statements:

> Page 64 – "Ammonia has been shown to have negative effects on prey items that longfin smelt rely upon…"
>
> Page 87 –" Information indicates that introduced species are a threat to the Bay-Delta longfin smelt population and that ammonium may constitute a threat to the Bay-Delta longfin smelt population."
>
> Page 118 and 119 – "The release of ammonia into the estuary is having detrimental effects on the Delta ecosystem and food chain…it is likely that new ammonia limits will take effect in 2020. Until that time, the CWA protections for longfin smelt are limited, and do not reduce the current threat to longfin smelt."
>
> Page 124 - "…although no direct link has been made between contaminants and longfin smelt…ammonium has been shown to have a direct effect on the food supply that the Bay-Delta longfin smelt relies upon. Therefore, we conclude that high ammonium concentrations may be a significant current and future threat to the Bay-Delta DPS of longfin smelt."
>
> Page 127 – "high ammonium concentrations act to significantly reduce habitat suitability for longfin smelt."

Given the above cited problems regarding the proper recognition of the best available scientific information, and the lack of proper qualification of such statements based on recognized uncertainties regarding the effects of ammonia in the Delta ecosystem, it is fair to say that the above statements regarding ammonia/ammonium are misleading.

In other areas of the Finding, similarly misleading statements are made regarding the impacts of various contaminants on longfin smelt. In these cases, the report fails to link the noted effects to actual ambient concentrations in the Bay or Delta. In other words, no analysis was performed or documented linking trace metals, mercury, or selenium to effects on longfin smelt. Instead, such effects are implied in the report text. For example on page 80, statements are made regarding metals toxicity to fish in the upper Sacramento River near Redding. The document fails to note the differences in concentrations between the areas in question, which are more than a hundred and fifty miles from the Delta, and metals concentrations in the Delta. The implication that metals toxicity problems in the Upper Sacramento River would translate to problems in the Delta is misleading. The document does not cite to observed episodes of metals toxicity to fish in the Delta, mainly because such information does not exist.

On page 81, the document refers to "neurological effects in some fish species," giving the impression that such effects may exist in the Delta. In fact, the concentrations of mercury in fish tissue that would produce such effects are not observed in fish that inhabit the Delta. The notion that longfin smelt would be potentially impacted by body burdens of mercury is not likely and entirely unsupported by actual data or comparisons to effects thresholds.

On page 81, statements are made regarding the potential effects of blooms of *Microcystis aeruginosa* (blue-green algae) in the Delta. Suggestions are made that toxins produced by such blooms may impact copepods or may decrease dissolved oxygen to lethal levels for fish. These open-ended suggestions are unsupported by data on the concentration of toxins or dissolved oxygen data that would allow the assessment of these conditions. Such suggestions are insufficient to formulate a meaningful assessment of risk and are misleading.

Statements made with regard to the pyrethroid toxicity based on the work by Weston and Lydy (2010) are misleading. The Finding states on page 78:

> "Weston and Lydy (2010, p. 1835) found the largest source of pyrethroids flowing into the Delta to be coming from the Sacramento Regional Water Treatment Plant (SRWTP), where only secondary treatment occurs. Their data not only indicate the presence of these contaminants, but the concentrations found exceeded acute toxicity thresholds for the amphipod *Hyalella azteca*. This is of substantial concern because the use of insecticides in the urban environment had not before been considered the primary source of insecticides flowing into the Delta. Furthermore, this was not the case for the Stockton Waste Water Treatment facility, where tertiary treatment occurs, suggesting that the tertiary treatment that occurs at the Stockton facility could minimize or eliminate toxic effluent being dispersed from wastewater facilities (Baxter *et.al.* 2010, p. 33).

The Finding creates the impression that the SRCSD discharge was creating acute toxicity in the Delta. In fact, Weston has been clear that no pyrethroid toxicity was seen in the Sacramento River downstream from the SRCSD discharge. The statements made with regard to the comparative toxic effects of the SRCSD and Stockton discharges is also misleading. The SRCSD discharge is immediately diluted by a factor of more than 20 to 1, and typically by a factor of 50 to1 near the point of discharge. As a result, even though effluent concentrations may be higher, the ambient concentrations measured in the Sacramento River below the SRCSD secondary effluent discharge are similar to or lower than the ambient concentrations observed in the San Joaquin River below the Stockton tertiary effluent discharge, which receives much lower dilution. In both cases, importantly, effluent-associated toxicity in the receiving waters has not been observed.

Furthermore, the suggestion that tertiary treatment could minimize or eliminate toxicity in the Delta is a misrepresentation of the source statement from Baxter el. al. 2010, which states:

> "This was not the case with effluent from the Stockton Wastewater Treatment Plant, which utilizes tertiary treatment, suggesting that different treatment methods may remove or retain pyrethroids differently."[4]

The above statement from Baxter et. al. pertains to the removal efficiency of pyrethroids by different wastewater treatment plant processes, rather than a suggestion for a need for tertiary treatment to address ambient toxicity in the Delta, as stated in the Finding.

Lack of scientific analysis to support statements made in the document

In addition to examples cited above, in the Contaminants section of the document, vague statements regarding research results are made which leave impressions that effects are occurring in the Bay-Delta, but which lack the scientific analysis needed to actually assess the validity of these impressions. For instance:

> Page 79 – "Werner et al (2010b, p.3) found that larval delta smelt were between 1.8 and 11 times more sensitive than fathead minnows (*Pimephales promelas*) to copper, ammonia and all insecticides except permethrin."
>
> Page 79 – "Aquatic insects in which the longfin smelt relies upon for food have been shown to be sensitive to ammonia."
>
> Page 79 and 80 – "Delta smelt have been shown to be directly sensitive to ammonia at the larval and juvenile stages (Werner et al. 2008, p. 85-88). Longfin smelt could be similarly affected by ammonia as they utilize similar habitat and prey resources and have a physiology similar to delta smelt.
>
> Page 80 – "Ammonia also can be toxic to several species of copepods important to larval and juvenile fishes (Werner et al. 2010, p. 78-79; Teh et al, 2011, p. 25-27)"

[4] Baxter et. al., Interagency Ecological Program 2010 Pelagic Organism Decline Work Plan and Synthesis of Results, December 6, 2010. Pg 32.

> Page 123 – "Contaminants may have direct toxic effects to longfin smelt and other pelagic fish and indirect effects as a result of impact to prey abundance and composition."

In and of themselves, the above statements offer no proof regarding the existence of effects of ammonia or other contaminants to longfin smelt or other fish species in the Delta. Such proof would require the examination of ambient data in the Delta in comparison to effects thresholds. Such toxicological analysis is not performed, documented or cited in the subject document. This is in stark contrast to the South Delta pumping entrainment effects cited it the document, where direct mortalities in the millions of longfin smelt are documented. Yet, inexplicably, the document places greater emphasis on the threat to longfin smelt by ammonia as opposed to the threat that exists due to entrainment.

While the document fails to identify toxicological studies in the Delta linking fish kills to ammonia concentrations, on the other hand, it ignores an extensive assessment of paired ammonia, temperature and pH data throughout the Delta (10,000 data points) in comparison to USEPA 1999 ammonia criteria that was performed by SRCSD in 2009 and 2010. This information is well documented, having been provided in various public forums, including the August 2009 ammonia summit, the 2010 State Water Resources Control Board Delta flow Informational Proceedings, and as part of the 2010 SRCSD discharge permit hearing record. The February 2012 draft Bay Delta Conservation Plan Environmental Impact Report Water Quality chapter bases its analysis of ammonia effects on the same USEPA 1999 ammonia criteria, and reaches a similar conclusion to that documented by SRCSD, which is that ammonia toxicity to fish is not an observed or projected problem in the Delta.

Internal Inconsistencies

Early in the document, on page 37, the statement is made:

> "Despite numerous studies of longfin smelt abundance and flow in the Bay-Delta, the underlying causal mechanisms are still not fully understood (Baxter et al. 2010, p. 69; Rosenfeld 2010, p.9)".

Yet, the Finding makes definitive statements and findings regarding the impact of ammonium, an apparent contradiction. In fact, recent information strongly supports the general statement on Page 37, in that the effect of ammonium on the Bay-Delta food web is not well understood.

This inconsistency is repeated and magnified on page 124, where the document states that

> "...although no direct link has been made between contaminants and longfin smelt...ammonium has been shown to have a direct effect on the food supply that the Bay-Delta longfin smelt relies upon. Therefore, we conclude that high ammonium concentrations may be a significant current and future threat to the Bay-Delta DPS of longfin smelt."

Neither Baxter, nor Dugdale, nor any other researcher has made this definitive finding regarding the effect of ammonium on the food supply for longfin smelt.

On pages 60-61 of the document, the following statement is made:

> "The Clean Water Act has not effectively limited ammonia input into the system, and ammonia has been shown to negatively affect the longfin smelt's food supply."

This statement pre-supposes that the negative effect of ammonia is supported by best available science, and that the actions of the USEPA and State of California in considering and reacting to the studies alleging those effects were untimely and irresponsible.

In a similar vein, on page 64, the report, in referring to the December 2010 SRCSD discharge permit adopted by the Central Valley Regional Water Board, concludes:

"This regulation does not adequately mitigate potential negative effects to longfin smelt from ammonia in the Bay-Delta."

This statement ignores that the 2010 SRCSD discharge permit requires a severe reduction in ammonia by SRCSD by 2020. It also fails to qualify the uncertainty regarding the effect of ammonia on longfin smelt, given the uncertainty of effects on specific prey items and the uncertainty (and absence of research or analysis) regarding any associated food web effect on longfin smelt.

The Clean Water Act (and the California Water Code) rely on an organized system of regulation that includes, but is not limited to:

- Designating beneficial uses to be protected,
- Adopting enforceable water quality objectives to provide protection for the most sensitive of those uses in a given water body, and
- Regulating contaminant sources through National Pollution Discharge Elimination System permits, Total Maximum Daily Load allocations, and other mechanisms.

In the case of ammonia, USEPA in 1999 established the latest national criteria for ambient ammonia concentrations to protect sensitive aquatic life from the toxic effects of ammonia. USEPA is in the process of updating those criteria. USEPA considers new scientific information in the process of setting new criteria and undertakes a careful review of information before accepting it for use in the establishment of criteria.

The information developed by Dugdale and others since 2007 has not been formally considered by USEPA or the State of California as the basis for establishing water quality criteria. As noted elsewhere in this letter, if such consideration were to be made, it is likely, based on the information described in the McKee 2011 NNE document which presents expert analysis of the issue, that the work by Dugdale and others (which is heavily relied upon in this report) would not be considered adequate for the establishment of revised ammonia criteria or new ammonia objectives. This statement was made at the public hearing of the San Francisco Bay Regional Water Quality Control Board in February, 2011, where Board Chair Terry Young stated that the best available scientific information on ammonia (which included the latest work by Dugdale, Teh, and others) was not adequate to establish water quality objectives or to be used as the basis for NPDES permit decisions.

Conclusion

Significant changes should be made in the document, to modify or eliminate statements regarding the role of ammonia in the longfin smelt population decline and future risks to the species in the Bay-Delta. These changes are needed to reflect the best available scientific information, to rectify misleading impressions and to correct internal inconsistencies.

SRCSD is willing to devote resources to work with your staff to identify necessary changes in the document to address the issues raised in this letter. Additionally, we will be contacting you soon to arrange a meeting with for sharing our knowledge of water quality, learning more about public input in the Endangered Species Act process, and to discuss next steps in this matter. Please contact Linda Dorn, Environmental Program Manager, at dornl@sacsewer.com or 916-876-6030 if you have any questions.

Sincerely,

Stan Dean
District Engineer

cc: Ren Lohoefener, Regional Director, Pacific Southwest USFWS
Prabhakar Somavarapu Director of Policy and Planning
Terrie Mitchell, Legislative and Regulatory Affairs Manager

Attachment: December 26, 2011 Pacific Ecorisk's Critical Review of "Full Life-Cycle Bioassay Approach to Assess Chronic Exposure of *Pseudodiaptomus forbesi* to Ammonia/Ammonium Submitted to: Chris Foe and Mark Gowdy State Water Board / UC Davis Agreement No. 06-447-300 SUBTASK No. 14 By Swee Teh, Ida Flores, Michelle Kawaguchi, Sarah Lesmeister, and Ching Teh Aquatic Toxicology Program Department of Anatomy, Physiology, and Cell Biology School of Veterinary Medicine University of California-Davis, Davis CA 95616 Date: March 4, 2011"

FINDINGS REPORT

From A Critical Review of:

Full Life-Cycle Bioassay Approach to Assess Chronic Exposure of *Pseudodiaptomus forbesi* to Ammonia/Ammonium - Final Report

Dated August 31, 2011

Prepared by: Teh S, Flores I, Kawaguchi M, Lesmeister S, Teh C
Aquatic Toxicology Program, Department of Anatomy, Physiology, and Cell Biology, School of Veterinary Medicine, University of California Davis

This Critical Review Was Prepared By:

Pacific EcoRisk, Inc.
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This Critical Review Was Prepared For:

Larry Walker Associates 707 Fourth St. Davis, CA 95616	Central Contra Costa Sanitary District 5019 Imhoff Place Martinez, CA 94553

Submittal Date:
December 26, 2011

1. INTRODUCTION

On behalf of the Central Contra Costa Sanitary District (CCCSD), Larry Walker Associates has contracted Pacific EcoRisk, Inc. (PER) to perform a critical review of the "Final Report: *Full Life-Cycle Bioassay Approach to Assess Chronic Exposure of Pseudodiaptomus forbesi to Ammonia/Ammonium*" authored by Teh S, Flores I, Kawaguchi M, Lesmeister S, and Teh C (dated August 31, 2011).). As requested by CCCSD, the primary focus of this review were the experiments described as Subtasks 3-3 and 3-4-1 in the Teh *et al.* report. Additional comments on study methodology and data analysis were developed and can be provided to interested parties on request as evidence that additional study is needed.

2. COMMENTS ON SUB-TASK 3-3 (CHRONIC [31-DAY] LIFE CYCLE TOXICITY TESTING)

Comment #1. Teh *et al.*'s analysis of the number of nauplii and number of juveniles produced during the chronic (31-day) exposure is believed to be flawed at a very fundamental level. It is apparent in Teh et al.'s derivation of 'mean number of nauplii, juveniles, and adult *P. forbesi* produced per female' (in Teh *et al.*'s Table 11) and in the 'sum total number of nauplii, juvenile, and adult *P. forbesi* produced' (in Teh *et al.*'s Appendix III table) that they summed the counts of nauplii and juveniles that were counted on the progressive 2-3 day intervals (the raw data for these counts were provided in Teh *et al.*'s Appendix I) as if each new progressive count was of new individuals that had not been counted on the previous count day. So when 17 nauplii were counted in Control replicate A on Day 5 of the test, and 20 nauplii were counted on Day 7, and 17 were counted on Day 10, and so on, Teh *et al.* summed these up as if they were different nauplii that had been produced during the progressive 'count days'.

This would be correct had the nauplii and juveniles that were counted on each 'count day' been removed from the original replicate container and transferred to a new replicate container such that any nauplii or juveniles observed and counted in the original replicate containers on subsequent days would have been new organisms separate and distinct from the organisms that had been counted during the previous count day(s). Note that this approach would have created a logistical challenge, with a doubling of the number of experimental replicate beakers on Day 3 of the test (going from the original n=20 to n=40), a tripling of the beakers on Day 5 (n=60), a quadrupling of beakers on Day 7 (n=80), and so on and so on. This would then be compounded as nauplii that had transformed into juveniles would again need to be transferred to new replicates so as to allow observation of new juveniles produced by the remaining nauplii. The number of necessary beakers rapidly becomes logistically improbable.

However, it is not believed that this is what happened. Unfortunately, their report's inadequate description of test methodology is not explicit on this. However, it can be deduced from the nature of the study that the neonates were left in place in each replicate, as these were the source of the subsequent juveniles, which were similarly left in place to serve as the source for the subsequent adults. This was confirmed by inquiry made with one of the other authors of the report (M Kawaguchi, pers. comm.). As a result, when 20 nauplii were counted in Control replicate A on Day 7, some (if not most) of these organism were the very same organisms that had been counted on the earlier Day 5 count, and the nauplii that were counted on Day 10 were some of the same as had been counted on Days 7 and Day 5.

This conclusion is also supported by the following observations made for closely-related congener *Pseudodiaptomus annandalei* (Golez et al. 2004):

1. hatching of the first brood of nauplii occurs within 24-hrs of spawning;
2. females produced new ovisacs at ~ 1 day intervals, again with hatching occurring within that 24-hrs;
3. "females that were isolated from males produced only two clutches of viable eggs". Additional ovisacs were produced (making it appear that the female is reproductive), but the "succeeding clutches of eggs were aborted or shed off within 48 hrs and never hatched out".

Of course, the reproductive biology of *P. forbesi* may differ from that of the congener *P. annandalei*; however, in the absence of contradictory empirical evidence, Occam's razor would dictate otherwise.

We are left to conclude that **Teh *et al.*'s reported results for 'total number' and 'mean number per female' for the nauplii and juveniles are incorrect, and that their analyses of that data are similarly incorrect**.

Interestingly, in Teh *et al.*'s analyses of the 'total number' and 'mean number per female' of adults produced during the study, the number of adults counted on each progressive 'count day' were **NOT** summed in similar fashion, with Teh *et al.* instead evaluating on the count data from a single 'count day' (Day 31).

Comment #2. While it is believed that Teh *et al.*'s count data are incorrect, let us assume for a moment that they are in fact correct. The organism counts using Teh *et al.*'s summation method are summarized in Table 1 below. When their juvenile count data are analyzed using CETIS (a statistical software specifically designed to analyze aquatic toxicity data), the NOEC and LOEC are shown to be 0.79 mg/L TAN and 1.62 mg/L TAN (Table 2 below), NOT the lower concentrations reported by Teh *et al.*

It should noted that CETIS is the statistical software most commonly used by toxicity testing labs to analyze toxicity test data, and is believed to be the statistical software used at the UC Davis Aquatic Toxicology Lab; indeed, Teh *et al.* used CETIS to analyze their Subtask 3-4-1 and Subtask 3-4-2 experimental data as evidenced in Appendices IV and V of their report.

It should also be noted that our assessment of problems with Teh *et al.*'s statistical analyses should not be interpreted as indicating that there was no effect resulting from the ammonia, but simply that the experimental data do not support any differences that were observed as being statistically significant.

Table 1. Production on *Pseudodiaptomus forbesi* nauplii, juveniles, and adults (from Appendix I in Teh *et al.* report)

Test Treatment (mg/L TAN)	Test Replicate	Total # of *Pseudodiaptomus forbesi* Life Stage Counted			
		Nauplii[A]	Juveniles[A]	Adults[A] (counts made only on Day 31)	Adults[B] (counts made as for nauplii & juveniles)
Control	A	86	38	11	93
	B	100	73	26	178
	C	68	45	7	122
	D	75	52	3	52
0.36	A	60	27	0	1
	B	62	57	3	36
	C	83	79	18	167
	D	71	43	7	77
0.79	A	24	48	10	77
	B	64	31	4	45
	C	41	17	1	17
	D	52	22	8	77
1.62	A	47	1	0	0
	B	32	0	0	0
	C	46	14	5	28
	D	54	23	19	108
3.23	A	15	1	1	4
	B	39	1	1	6
	C	42	18	13	83
	D	30	13	5	34

A - For the nauplii and juveniles, Teh et al. summed the progressive counts on successive days as separate individuals; as explained in our review, this is believed to be erroneous, and is inconsistent with the counts of the "produced" adults which consist of the number of adults that were alive on Day 31 of the test.

B - Counts of "produced" adults using the summation of the progressive counts on successive days as separate individuals (as used by Teh et al. for the nauplii and juveniles); as explained in our review, this is believed to be erroneous.

Table 2. Comparative analyses of juvenile and adult production in the 31-day test (from CETIS analysis of juvenile data using Teh et al. summation method)				
Statistical Endpoint	Juveniles		Adults	
	Teh et al. Analyses	CETIS Analyses	Teh et al. Analyses	CETIS Analyses
NOEC =	0.36 mg/L TAN	0.79 mg/L TAN	**<0.36 mg/L TAN**	**3.23 mg/L TAN**
LOEC =	0.79 mg/L TAN	1.62 mg/L TAN	**0.36 mg/L TAN**	**>3.23 mg/L TAN**
Chronic Value =	1.13 mg/L TAN	1.13 mg/L TAN	<0.36 mg/L TAN	>3.23 mg/L TAN

Chronic Value = geometric mean of NOEC and LOEC.

Comment #3. Teh *et al.*'s apparently erroneous statistical analysis of the adult data is even more significant (Table 2). Teh *et al.* reported that the NOEC and LOEC for adults were <0.36 mg/L TAN and 0.36 mg/L TAN, respectively. However, their inter-replicate variability for that endpoint is so high (CVs ranged from 70% to 150%) that even qualitative evaluation suggests otherwise. CETIS analysis indicates that the NOEC and LOEC are 3.23 mg/L TAN and >3.23 mg/L TAN.

Again, it should be noted that our assessment of problems with Teh *et al.*'s statistical analyses should not be interpreted as indicating that there was no effect resulting from the ammonia, but simply that the experimental data do not support any differences that were observed as being statistically significant. Certainly, the NOECs and LOECs resulting from this experiment should not be considered suitable for use in a regulatory framework.

3. COMMENTS ON SUBTASK 3-4-1 (EFFECTS OF AMMONIA ON NAUPLII PRODUCTION OVER 3 DAYS)

Comment #4. In this test, Teh *et al.* exposed individual gravid female copepods to TAN concentrations of 0 (control treatment), 0.38, and 0.79 mg/L for 3 days after which the number of nauplii produced were counted. The results of this test have been summarized in the Table 3 below.

From data reported in Teh et al.'s Table 12 and Appendix V:

Table 3. Effects of ammonia on *P. forbesi* production of nauplii over 3 days (Teh *et al.* Subtask 3-4-1).

TAN Concentration (mg/L)	Mean # of Nauplii per Female
Control	7.6
0.38	5.5
0.79	5.4

The results from this test are somewhat troubling in that, while technically monotonically increasing as the ammonia concentration increases, no apparent concentration-response relationship is observed between the 0.38 mg/L treatment and the 0.79 mg/L treatment. One would expect that as the TAN concentration increases from 0.38 mg/L (a presumably toxic concentration) to 0.79 mg/L (a two-fold greater concentration), there should be an increase in the toxic response – this is a fundamental paradigm of toxicology.
We have already seen in the data evaluations presented above that there is variability in toxic responses made by these organisms. Indeed, in some cases, the variability has been so extreme as to preclude a meaningful statistical analysis (as in the case of the adult data from the 31-day test). The absence of the expected concentration-response in the current test (Table 3) suggests that variability in organism response is occurring (the CV was 48% in the 0.38 mg/L treatment) such that the treatment means may be deviating from the true population mean (in statistical terms, this is referred to as a "false positive" or a "false negative").

In the present case, it is impossible to determine which of the two test responses is deviating most from the true population mean response. However, it is worth noting that:

1. there were two replicates at the 0.38 mg/L treatment that had 10 nauplii (the highest number observed in ANY replicate) whereas there was only one replicate at the control treatment that had 10 nauplii, and
2. the CV at the 0.38 mg/L treatment was 48%, which was markedly higher than at the Control or 0.78 mg/L treatment.

This is suggestive that the variability at the 0.38 mg/L treatment was elevated and may have resulted in a false positive, such that the observed mean response of 5.5 nauplii per female was lower than the true population mean. If correct, then the conclusion(s) drawn from the test data may not reflect true conditions, and the true LOEC could be 0.79 mg/L, and not 0.38 mg/L. At a minimum, the absence of the expected concentration-response should cast enough uncertainty on the test results as to make them inappropriate for regulatory decision-making.

Comment #5. It is fortunate that multiple sets of test data from the study allow comparison of results between tests; for instance, the results of Subtask 3-4-1 can be compared to those generated in the earlier Subtask 3-3 (31-day) test in which gravid females were exposed to varying concentrations of TAN and counts of nauplii produced after 3 days were counted, but were also counted after 5 days and 7 days (recall that counts made on progressive count days are not believed to be all new organisms). The Subtask 3-3 data are summarized in Table 4 below, along with the data from Task 3-4-1.

If one were to "cherry-pick" the Day 3 data and exclude the additional data, then Teh *et al.*'s conclusion for the Subtask 3-4-1 might stand. However, by extending the observation period beyond 3 days, it becomes evident that not only is there no reduction in nauplii production at 0.36 mg/L TAN, but nauplii production actually appears to be *increased* relative to the control treatment (the maximum mean # of nauplii on Day 5 at the 0.36 mg/L TAN treatment is **31% greater** than the highest mean # of nauplii produced in the Control treatment on any of the count days). Furthermore, CETIS analysis indicates that there were no statistically significant reductions in nauplii production at the 0.36 mg/L (Table 5). Even if we use the count summation used by Teh et al., by extending the counts beyond 3 days, it becomes apparent that there is no statistically significant difference between the response at 0.36 mg/L TAN and the Control treatment. This certainly creates a very significant uncertainty over the results of the Subtask 3-4-1 test of the effects of ammonia on nauplii production over 3 days.

It could be argued that this phenomenon is the result of ammonia having caused a delay in egg hatching, and the 31-day data are certainly suggestive of that. However, the only way to address that would have been to have some information from the scientific literature on the egg gestation period for this species, coupled with testing being performed under the current test conditions using females with egg sacs of the same age.

Table 4. Effects of ammonia on *P. forbesi* nauplii produced over 3 and 5 days.				
Teh *et al.* Study Task	TAN Treatment (mg/L)	Mean Number of Nauplii per Female		
		Day 3	Day 5	Sum through Day 5 (Day 3 + Day 5)[A]
Subtask 3-4-1	Control	**7.6**	not counted	not counted
	0.38	**5.5**	not counted	not counted
	0.79	**5.4**	not counted	not counted
Subtask 3-3	Control-A	5.67	6.67	12.33
	Control-B	6.67	6.67	13.33
	Control-C	5	5	10
	Control-D	5	5	10
	treatment mean	**5.6**	**5.8**	**11.4**
	0.36-A	3	5	8
	0.36-B	2.33	8.33	10.67
	0.36-C	3.33	8.33	11.67
	0.36-D	3.33	3.33	6.67
	treatment mean	**3.0**	**6.3**	**9.3**
	0.79-A	0.33	1.67	2
	0.79-B	6.67	3.33	10
	0.79-C	2.67	2.67	5.33
	0.79-D	6.67	4	10.67
	treatment mean	**4.1**	**2.9**	**7.0**

A – These counts are made using method of Teh *et al.*, which assumes that the progressive counts on successive days are separate individuals; as explained in our review, this is believed to be erroneous.

Table 5. Comparison of nauplii production test results (all results expressed as mg/L TAN) (from CETIS analysis of data)						
Statistical Endpoint	**Subtask 3-4-1**	**Subtask 3-3**				
	Day 3	Day 3	Day 5	Day 3 + Day 5[A]	Total (31 days)[A]	Total (31 days)[B]
NOEC =	<0.38	3.23	0.36	0.36	0.36	0.79
LOEC =	0.38	>3.23	0.79	0.79	0.79	1.62
Chronic Value =	<0.38	>3.23	0.53	0.53	0.53	1.13

Chronic Value = geometric mean of NOEC and LOEC.

A – These counts are made using method of Teh *et al.*, which assumes that the progressive counts on successive days are separate individuals; as explained in PER's review, this is believed to be erroneous.

B – These counts are made using what is believed to be the best remaining method: identifying the maximum number of nauplii observed on any given day for each replicate (this assumes that the individuals were left in the replicate beakers and were counted again and again on progressive days [i.e. repeated measures]).

4. FINAL COMMENT

The reviewer is troubled by the absence of any discussion by Teh et al. regarding the variability in their test response data, either between tests or within tests (i.e., inter-replicate variability). Without such acknowledgement, it is left for the non-scientist to assume that the data as presented are definitive. Moreover, it raises the question of whether the data from this study are adequate (or 'ready') for use in regulatory decision-making. However, it is important to note that this critical review is not intended to negate Teh *et al.*'s general observations that ammonia is toxic to naupliar, juvenile, and/or adult *P. forbesi* at elevated concentrations and that this toxicity is strongly influenced by pH. Indeed, the primary question of 'what are the effects of ammonia on *P. forbesi*' is relevant and Teh *et al.*'s study results certainly compel a more thorough examination of this. However, the problems associated with Teh et al.'s experimental methodology for Subtasks 3-3 and 3-4-1 and significant questions regarding the analysis of the resulting data do indicate that the quality of the work should preclude the resulting 'critical threshold' data (i.e., NOECs, LOECs, and point estimates [e.g., ECx, LCx, and ICx values]) from being used for regulatory purposes.

References Cited:

Golez MSN, Takahashi T, Ishimaru T, Ohnoa A (2004) Post-embryonic development and reproduction of *Diaptomus annandalei* (Copepoda: Calanoida). Plankton Biology & Ecology 51(1):15-25.

EXHIBIT J

March 2, 2012

TO: Municipal Wastewater Dischargers (attached list)

SUBJECT: Water Code Section 13267 Technical Report Order Requiring Submittal of Information on Nutrients in Wastewater Discharges

This order requires municipal wastewater dischargers in the San Francisco Bay Region to monitor and report nutrient (nitrogen and phosphorus) concentrations and mass loadings in their wastewater discharges. The information we require includes a report of historical nutrient data, a sampling plan, quarterly reports, an interim report, and a final report. Details of these requirements, their due dates, and the basis for the requirements are described below.

Please direct your questions to Tong Yin at 510-622-2418, or by e-mail TYin@waterboards.ca.gov.

Applicability

This order is intended for, and applicable to, all dischargers under an NPDES[1] permit with the following exceptions: discharges to ocean waters, discharges of once through cooling water, discharges consisting solely of industrial process and associated wastewaters, discharges consisting solely of stormwater runoff, and discharges covered under general permits, such as for aggregate mining and sand washing, and solvent and fuels groundwater cleanup. Dischargers subject to one or more of these exceptions that discharge wastewater under an individual NPDES may be subject to a similar California Water Code section 13267 order in the future.

Purpose and Basis of Requirements

Nitrogen and phosphorus are essential nutrients for the growth of all living organisms in ecosystems. However, excessive nutrients may cause algae blooms in surface waters (eutrophication). Harmful algae blooms reduce or deplete oxygen in the water, produce toxins, stress or kill fish, and block sunlight reaching aquatic plants. There is also some evidence that certain forms of nutrients, e.g., ammonium, may inhibit phytoplankton productivity or have other effects on biota.

The San Francisco Bay estuary has long been recognized as a nutrient-enriched estuary. Despite this, the abundance of phytoplankton in the estuary is lower than would be expected, due to a number of factors, including strong tidal mixing, light limitation due to high turbidity, and grazing by clams. Bay monitoring data are indicating a significant increase in phytoplankton biomass and a small decline in dissolved oxygen concentrations in many areas of the San

[1] National Pollutant Discharge Elimination System

Francisco Bay estuary, suggesting that the historic resilience of the estuary to the effects of nutrient enrichment may be weakening.

Currently, the Regional and State Water Boards are in the process of developing nutrient water quality objectives for the San Francisco Bay estuary, using an approach known as the Nutrient Numeric Endpoint (NNE) framework. The NNE approach will likely require models that link ecological response indicators to nutrient loads and other management controls. This effort must be supported by accurate nutrient loading estimates from a variety of sources, including wastewater.

Wastewater discharges contribute a large portion of the nutrient loadings to the estuary; for example, it may be as high as 80% in such areas as the South Bay during the dry season. Wastewater discharges into tributaries of the Bay may also contribute to nutrient loadings to the Bay. There have been published studies that have developed some loading estimates; however, these studies are outdated, inadequate, or limited geographically. Thus, nutrients loads to the San Francisco Bay estuary from external sources are still poorly understood, and it is important to get an accurate estimate of the loadings.

The information collected under this order will be used by the Regional Water Board to evaluate nutrient loadings from wastewater discharges in comparison to loads from other sources, to support modeling and evaluation of loading reduction scenarios, and to inform the need for additional wastewater treatment to address nutrients. The data may also be used in the future to support development of TMDLs or other regulatory strategies.

The Regional Water Board is working collaboratively with the Bay Area Clean Water Agencies as well as other entities on studies that are being identified as part of a regional nutrient strategy. Loads analysis and modeling are included in this strategy. The San Francisco Estuary Institute (SFEI) is supporting this effort. Therefore, this order includes providing all compiled data to SFEI.

This order also requires influent nutrient monitoring. The influent data will be used to establish existing nutrient levels in raw wastewater, to examine plant performance in removing nutrients from waste streams, and to evaluate the necessity of future plant upgrades to reduce nutrient loadings from wastewater discharges to maintain or restore beneficial uses of the estuary.

<u>Nutrient Parameters to be Monitored</u>

Analytically, nitrogen and phosphorus are divided into a number of chemical forms. This order requires monitoring of those forms potentially found in influent or effluent. This order requires influent and effluent monitoring for the following nitrogen and phosphorus forms as well as some ancillary parameters:

- Total Dissolved Nitrogen (TDN)
- Total Kjeldahl Nitrogen (TKN)
- Soluble Kjeldahl Nitrogen (SKN)
- Nitrate (NO_3^-)
- Nitrite (NO_2^-)
- Total Ammonia (NH_3 and NH_4^+)

- Urea
- Total Phosphorus
- Total Phosphorus (soluble)
- Orthophosphate (dissolved/total)
- pH
- Temperature
- Total Suspended Solids (TSS)

This list includes pH and temperature, which are required for calculating ammonium (NH_4^+) from measured total ammonia concentrations. TSS results may be used to evaluate the correlation between TSS and some nutrient parameters, such as with total phosphorus.

Urea or carbamide ($CO(NH_2)_2$) is the main nitrogen-containing substance in the urine of mammals. Urea breaks down to carbon dioxide (CO_2) and ammonium in the aquatic environment. Urea may inhibit nitrogen uptake by algae. The Regional Water Board is currently investigating ambient ammonium inhibition effects on diatom blooms in the Suisun Bay; therefore, data on ammonium discharges will improve understanding of their impacts on primary productivity.

Questions have been raised about the potential quality of the urea data collected from wastewater discharges; however, there is no urea data for wastewater discharges in this region. This order only requires the region's five largest NPDES permittees, East Bay Municipal Utilities District (EBMUD), East Bay Dischargers Authority (EBDA), San Jose/Santa Clara Water Pollution Control Plant (SJSC), San Francisco Southeast Plant (SFSE), and Central Contra Costa Sanitary District (CCCSD), to collect urea data over a one year period. Harmful algal blooms show a preference for urea in ambient waters, and urea could thus be important to measure.

Table 1 lists the required parameters and suggested analytical methods:

Table 1. Parameters to be Monitored

Parameters	Units	Influent(1)	Effluent(1)	Sample type(2)	Suggested Analytical Methods(3)(4)
Total Dissolved Nitrogen(5)	mg/L and kg/day as Nitrogen (N)	Yes	Yes	24-hour composite	Standard method 4500-N
Total Kjeldahl Nitrogen	mg/L and kg/day as N	Yes	Yes	24-hour composite	Standard method 4500-N (organic)
Soluble Kjeldahl Nitrogen	mg/L and kg/day as N	Yes	Yes	24-hour composite	Standard method 4500-N (organic)
Nitrate	mg/L and kg/day as N	Yes	Yes	24-hour composite	Standard method 4500-N
Nitrite	mg/L and kg/day as N	Yes	Yes	24-hour composite	Standard method 4500-N
Total Ammonia	mg/L and kg/day as N	Yes	Yes	24-hour composite	Standard method 4500-NH_3
Urea	mg/L and kg/day as N	Yes	Yes	24-hour composite	(6)
Total Phosphorus	mg/L and	Yes	Yes	24-hour composite	Standard method 4500-P

Parameters	Units	Influent(1)	Effluent(1)	Sample type(2)	Suggested Analytical Methods(3)(4)
	kg/day as Phosphorus (P)				
Total Phosphorus (soluble)(5)	mg/L and kg/day as P	Yes	Yes	24-hour composite	Standard method 4500-P
Orthophosphate (dissolved/total) (5)	mg/L and kg/day as P	Yes	Yes	24-hour composite	Standard method 4500-P
Flow(7)	mgd	Yes	Yes	Continuous	---
pH(8)	Standard unit	Yes	Yes	Continuous/Grab	---
Temperature(8)	Degree C	Yes	Yes	Continuous/Grab	---
TSS	mg/L	Yes	Yes	24-hour composite	Standard Method 2540D
Total nitrogen and phosphorus removal, by concentration	Percent removal	---	---	Calculate from influent and effluent monitoring data	---

Footnotes for Table 1:

(1) Influent and effluent sampling shall be at the compliance monitoring locations currently specified in a discharger's NPDES permit. Sampling for all influent and effluent parameters shall fall on the same dates.
(2) 24-hour composites may be made up of a minimum of four discrete grabs, collected over the course of 24 hours or during a 24-hour period the plant is staffed, and volumetrically or mathematically flow-weighted. Grab samples may be combined prior to analysis. If only one grab sample will be collected, it should be collected during periods of maximum peak flows.
(3) Dischargers may propose other U.S. EPA-approved analytical methods, if available, with detection limits low enough to quantify concentrations in wastewater.
(4) Standard methods for the examination of water and wastewater, American Public Health Association.
(5) Soluble or dissolved is defined as filtering the sample through a 0.45 μm filter.
(6) The five dischargers identified above shall propose an appropriate analytical method for urea in their study plan.
(7) Report daily average flow, which shall correspond to the same time period when the composite samples are collected; also report daily peak flow during which grab samples are collected.
(8) Report daily maximum, minimum and average values.

Units abbreviations:

mg/L = milligrams per liter
kg/day = kilograms per day
mgd = million gallons per day

Equations for calculating mass loadings

Mass loading (kg/day) = mg/L × mgd × 3.78

Sampling Frequency and Study Duration

As indicated in Table 2 below, this order requires two years (for major dischargers) or one year (for minor dischargers) of sampling from the date each discharger starts its first sampling event, except the sampling for urea, which is one year for the five dischargers listed below. The Regional Water Board may extend the study period if more data are needed.

This order specifies different sampling requirements (e.g., sampling frequency, urea monitoring, peak wet weather monitoring) for different groups of dischargers as shown in Table 2 based on their average dry weather design flow. The difference in requirements is based on consideration of flow and nutrient mass load contributions. In addition to the once or twice-per-month effluent monitoring, major dischargers shall also conduct two additional effluent peak wet weather samplings during each wet season; the data will be used to evaluate peak wet weather flow influence on plant performance and nutrient loads during peak wet weather flow conditions. Dry season influent and effluent monitoring data will be used to establish baseline conditions and to examine possible seasonal variability. Therefore, dry season monitoring data are also necessary for seasonal dischargers.

Table 2 lists the minimum sampling frequency and duration for different groups of dischargers:

Table 2. Minimum Sampling Frequency and Study Duration (1,2,3)

Dischargers	Year round or Seasonal	Influent	Effluent	Duration
Major municipal dischargers (Flow ≥ 5 mgd)	Year round	Once during wet season, once during dry season	Twice per month and two additional samples each wet season during peak wet weather flow conditions(4)	Two years
	Seasonal	Once during discharge (wet) season, once during non-discharge (dry) season	Twice per month during discharge (wet) season; once during non-discharge (dry) season	Two years
Major municipal dischargers (Flow < 5 mgd)	Year round	Once during wet season, once during dry season	Once per month and two additional samples each wet season during peak wet weather flow conditions(4)	Two years
	Seasonal	Once during discharge (wet) season, once during non-discharge (dry) season	Once per month during discharge (wet) season; once during non-discharge (dry) season	Two years
Minor municipal dischargers (Flow < 1 mgd)	Year round	Once during wet season, once during dry season	Once per month	One year
	Seasonal	Once during discharge (wet) season, once during non-discharge (dry) season	Once per month during discharge (wet) season; once during non-discharge (dry) season	One year
Urea Only				
CCCSD, EBMUD, EBDA, SFSE, and SJSC	Year round	Once during wet season, once during dry season	Once per month	One year

Footnotes for Table 2:

(1) Influent monitoring shall fall on the same dates as effluent monitoring events.
(2) Wet season is normally from November through April, dry season is from May through October. It is preferable to conduct dry season influent sampling during July, August, and September, when the weather is the driest of the year.
(3) Sampling dates shall be as random as feasible, i.e., sampling is not to occur on the same day or weekday of a month except the two wet season events that shall coincide with the peak wet weather flows.
(4) The Dischargers shall estimate the best dates of sampling for peak wet weather flow scenarios; this decision may be based on historical peak wet weather flows, storm forecast, etc.

The Regional Water Board may also require additional sampling, if available data indicate significant variability that cannot be characterized by the current sampling frequency.

You are hereby required to provide technical information in accordance with the following:

1. **Technical reports containing available historical nutrient, flow, and other water quality data.**

   a. Many municipal wastewater dischargers were or are required to sample for some nutrient parameters by their NPDES permits. Some or all dischargers also analyze for nutrient parameters not required by their NPDES permits. This order requires each discharger to submit a report that identifies what types and quantity (nutrient parameters from Table 1, number of samples, frequency of data collection, i.e., which calendar years and detection limits) of data that are available for the period of January 1, 1975, through February 29, 2012. This report is due to the Regional Water Board June 1, 2012; submit the report to [Tong Yin, tyin@waterboards.ca.gov or via FTP].

   b. Within 90 days of the date of the report submittal in 1(a) above, each discharger shall compile and submit electronically all nutrient data available for the time period of March 1, 2004, through February 28, 2009, other than data already submitted to the Regional Water Board via the Electronic Reporting System (ERS) for compliance purposes. This submittal shall also include all available effluent flow, pH, temperature, total suspended solids, and salinity data for that time period, and be submitted to the Regional Water Board [Tong Yin, tyin@waterboards.ca.gov or via FTP] and SFEI [David Senn, sfbayeffluent@sfei.org].

   c. Within 90 days of the date of the report submittal in 1(a) above, and only if the data are available electronically as of the date of this order, each discharger shall submit all nutrient data for the time period from January 1, 1975, through February 29, 2012.

2. **A Sampling and Analysis Plan for Collecting Required Information due April 30, 2012.**

   Dischargers shall submit a sampling and analysis plan to the Regional Water Board, [Tong Yin, tyin@waterboards.ca.gov or via FTP]. The sampling plan shall include, but not be limited to, a sampling schedule, contract labs to be used, analytical methods to be used, and detection limits of the methods. The sampling plan shall also clearly identify any proposed deviations from the requirements of this order, such as proposing to monitor for fewer or different parameters, and include the bases for any proposed deviations. Dischargers are encouraged to collectively submit one sampling plan.

   If the Regional Water Board does not provide comments on the sampling plan within 45 days, the discharger shall start monitoring by July 1, 2012.

3. **Quarterly reports due 30 days after the end of each calendar quarter.**

   Monitoring results for the parameters listed in Table 1 shall be tabulated in Excel spreadsheets and reported to the Regional Water Board [Tong Yin, tyin@waterboards.ca.gov or via FTP] and SFEI, [David Senn, sfbayeffluent@sfei.org. The spreadsheets shall include the name of parameters, units, sampling location, date and times of data collection, analytical method, method detection limit, reporting level, and sampling results. A spreadsheet template will be developed for dischargers use to ensure consistency in data reporting. The Bay Area Clean Water Agencies may develop a spreadsheet template for this purpose and make it available to all dischargers. If not, Regional Water Board staff will provide the template. Dischargers are encouraged to compile their data as a group prior to submittal to the Regional Water Board.

4. **An interim report due July 31, 2013.**

   The interim report shall include all data collected through June 30, 2013, for the parameters listed in Table 1, with a cover letter summarizing significant findings, changes or upsets in treatment operations or changes in influent sources that may affect interpretation of the data, and an analysis of any issues identified during data collection effort.

5. **A final report due July 31, 2014.**

   The report shall include the information collected under this study, with the same information as required under the "interim report" above.

These requirements are made pursuant to California Water Code section 13267, which allows the Regional Water Board to require technical or monitoring program reports from any person who has discharged, discharges, proposes to discharge, or is suspected of discharging waste that could affect water quality. Failure to respond or late response may subject you to civil liability imposed by the Regional Water Board up to a maximum amount of $1,000 per day. The attached fact sheet provides additional information about these requirements. Any extension in the above deadlines must be confirmed in writing by Regional Water Board staff.

Sincerely,

Digitally signed by Bruce Wolfe
Date: 2012.03.02 10:05:05 -08'00'

Bruce H. Wolfe
Executive Officer

Attachments: Fact Sheet for Section 13267 Orders
Municipal Dischargers Mailing List

Fact Sheet – Requirements for Submitting Technical Reports

Under Section 13267 of the California Water Code

What does it mean when the Regional Water Board requires a technical report?
Section 13267[1] of the California Water Code provides that "…the regional board may require that any person who has discharged, discharges, or who is suspected of having discharged or discharging, or who proposes to discharge waste...that could affect the quality of waters...shall furnish, under penalty of perjury, technical or monitoring program reports which the regional board requires."

This requirement for a technical report seems to mean that I am guilty of something or at least responsible for cleaning something up. What if that is not so?
The requirement for a technical report is a tool the Regional Water Board uses to investigate water quality issues or problems. The information provided can be used by the Regional Water Board to clarify whether a given party has responsibility.

Are there limits to what the Regional Water Board can ask for?
Yes. The information required must relate to an actual or suspected or proposed discharge of waste (including discharges of waste where the initial discharge occurred many years ago), and the burden of compliance must bear a reasonable relationship to the need for the report and the benefits obtained. The Regional Water Board is required to explain the reasons for its request.

What if I can provide the information but not by the date specified?
A time extension may be given for good cause. Your request should be promptly submitted in writing, giving reasons.

Are there penalties if I don't comply?
Depending on the situation, the Regional Water Board can impose a fine of up to $5,000 per day, and a court can impose fines of up to $25,000 per day as well as criminal penalties. A person who submits false information or fails to comply with a requirement to submit a technical report may be found guilty of a misdemeanor. For some reports, submission of false information may be a felony.

Do I have to use a consultant or attorney to comply?
There is no legal requirement for this, but as a practical matter, in most cases the specialized nature of the information required makes the use of a consultant and/or attorney advisable.

What if I disagree with the 13267 requirements, and the Regional Water Board staff will not change the requirement and/or date to comply?
You may ask that the Regional Water Board reconsider the requirement and/or submit a petition to the State Water Resources Control Board. See California Water Code sections 13320 and 13321 for details. A request for reconsideration to the Regional Water Board does not affect the 30-day deadline within which to file a petition to the State Water Resources Control Board.

If I have more questions, whom do I ask?
Requirements for technical reports include the name, telephone number, and email address of the Regional Water Board staff contact.

[1] All code sections referenced herein can be found by going to www.leginfo.ca.gov.

Municipal Dischargers Mailing List

City of American Canyon
300 Crawford Way
American Canyon, CA 94503
Attn: Peter Lee
(plee@cityofamericancanyon.org)
Wastewater System Manager

City of Benicia
614 East Fifth Street
Benicia, CA 94510
Attn: Jeff Gregory (jgregory@ci.benicia.ca.us)
Superintendent

City of Burlingame
501 Primrose
Burlingame, CA 94010
Attn: Syed Murtuza (smurtuza@burlingame.org)
Director of Public Works

City of Calistoga
414 Washington Street
Calistoga, CA 94515
Attn: Warren Schenstrom
(wschenstrom@ci.calistoga.ca.us)
Water Systems Superintendent

Central Contra Costa Sanitary District
5019 Imhoff Place
Martinez, CA 94553
Attn: Margaret Orr (morr@centralsan.org)
Director of Operations

Central Marin Sanitation Agency
1301 Andersen Drive
San Rafael, CA 94901
Attn: Robert Cole (rcole@centramarinsa.org)
Environmental Services Manager

Port Costa Sanitation Department
Crockett Community Services District
Crockett, CA 94525
Attn: Michael Kirker
(mkirker@town.crockett.ca.us)
Department Manager

Delta Diablo Sanitation District
2500 Pittsburg-Antioch Highway
Antioch, CA 94509
Attn: Gary W. Darling (GaryD@ddsd.org)
General Manager

East Bay Dischargers Authority
2651 Grant Avenue
San Lorenzo, CA 94580
Attn: Mike Connor (mconnor@ebda.org)
General Manager

East Bay Municipal Utilities District
P.O. Box 24055
Oakland, CA 94623-1055
Attn: Ben Horenstein (bhorenst@ebmud.com)
Manager of Environmental Services

Fairfield-Suisun Sewer District
1010 Chadbourne Road
Fairfield, CA 94534
Attn: Meg Herston (mherston@fssd.com)
Senior Environmental Compliance Engineer

Las Gallinas Valley Sanitation District
300 Smith Ranch Rd
San Rafael, CA 94903-1929
Attn: Mark Williams (mwilliams@lgvsd.org)
District Manager

Sanitary District No. 5 of Marin County
P.O. Box 227
Tiburon, CA 94920
Attn: Robert L. Lynch (rlynch@sani5.org)
District Manager

City of Millbrae
621 Magnolia Avenue
Millbrae, CA 94030
Attn: Joe Magner (jmagner@ci.millbrae.ca.us)
Superintendent

Mt. View Sanitary District
P. O. Box 2757
Martinez, CA 94553
Attn: Michael Roe (mroe@mvsd.org)
District Manager

Napa Sanitation District
P.O. Box 2480
935 Hartle Court
Napa, CA 94559
Attn: Tim Healy (thealy@napasan.com)
General Manager/District Engineer

Novato Sanitary District
500 Davidson Street
Novato, CA 94945
Attn: Beverly James (BevJ@novatosan.com)
General Manager

City of Pacifica
700 Coast Highway
Pacifica, CA 94044
Attn: David Gromm, Director of Wastewater
grommd@ci.pacifica.ca.us

City of Palo Alto
2501 Embarcadero Way
Palo Alto, CA 94303
Attn: James Allen, Plant Manager
(James.Allen@CityofPaloAlto.org)

City of Petaluma
202 N. McDowell Blvd.
Petaluma, CA 94954
Attn: Lena Cox (lcox@ci.petaluma.ca.us)
Environmental Services Supervisor

City of Pinole
1 Tennant Avenue
Pinole, CA, 94564
Attn: Ken Coppo (kcoppo@ci.pinole.ca.us)
Plant Manager

Rodeo Sanitary District
800 San Pablo Avenue
Rodeo, CA 94572
Attn: Steven S. Beall (bealls@rodeosan.org)
Engineer-Manager

City of St. Helena
1480 Main Street
St. Helena, CA 94574
Attn: John Ferons (JohnF@ci.st-helena.ca.us)
Director of Public Works

San Francisco International Airport
P. O. Box 8097
676 McDonnell Road
San Francisco, CA 94128
Attn: Brian Ciappara
(brian.ciappara@flysfo.com)
Superintendent

City and County of San Francisco
1155 Market Street, 11th Floor
San Francisco, CA 94103
Attn: Tommy Moala (tmoala@sfwater.org)
Assistant General Manager

City of San Jose
Water Pollution Control
700 Los Esteros Road
San Jose, CA 95134
Attn: Jim Ervin (james.ervin@sanjoseca.gov)
Supervising Environmental Services Specialist

City of San Mateo
2050 Detroit Drive
San Mateo, CA 94404
Attn: Larry Patterson
(patterson@cityofsanmateo.org)
Director of Public Works

Sausalito-Marin City Sanitary District
#1 East Road
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Attn: Robert Simmons (bob@smcsd.net)
General Manager

Sewer Agency of Southern Marin
26 Corte Madera Ave.
Mill Valley, CA 94941
Attn: Steve Danehy
(sdanehy@cityofmillvalley.org)
Manager

Sonoma County Water Agency
P.O. Box 11628
Santa Rosa, CA 95406
Attn: Pam Jeane (pam@scwa.ca.gov)
Deputy Chief Engineer - Operations

South Bayside System Authority
1400 Radio Road
Redwood City, CA 94065
Attn: Daniel Child (dchild@sbsa.org)
Manager

South San Francisco-San Bruno Water
Pollution Control Plant
195 Belle Air Road
South San Francisco, CA 94080
Attn: David Castagnola
(Dave.Castagnola@ssf.net)
Superintendent

City of Sunnyvale
Sunnyvale Water Pollution Control Plant
P.O. Box 3707
Sunnyvale, CA 94088-3707
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Division Manager

San Francisco Bay Area
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410 Palm Avenue, Bldg 1, Suite 161
Treasure Island
San Francisco, CA 94130-1807
Attn: Michael Mentink
(michael.mentink@navy.mil)
Environmental Coordinator

Vallejo Sanitation and Flood Control District
450 Ryder Street
Vallejo, CA 94590
Attn: Humberto Molina (hmolina@vsfcd.com)
Director of Operations and Maintenance

West County Agency
2910 Hilltop Drive
Richmond, CA 94806
Attn: E.J. Shalaby, District Manager
(District.Manager@wcwd.org)

Town of Yountville
6550 Yount Street
Yountville, CA 94599
Attn: Donald Moore (dmoore@yville.com)
Wastewater Systems Supervisor

C&H Sugar
830 Loring Avenue
Crockett, CA 94525
Attn: Tanya R. Akkerman
(tanya.akkerman@chsugar.com)
Environmental Compliance Manager