Pacific Gas and Electric Company®

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Via UPS Next Day Delivery
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AvillaLobos@waterboards.ca.gov

Ms. Amber Villalobos- Division of Water Rights
STATE WATER RESOURCES CONTROL BOARD
1001 "I" Street
Sacramento, CA 95814-2828

Re: DeSabla-Centerville Hydroelectric Project, FERC Project No. 803
PG&E's Comments on Draft Water Quality Certification

Dear Ms. Villalobos:

Pacific Gas and Electric Company ("PG&E") hereby respectfully submits comments on the April 12, 2013 Draft Water Quality Certification issued by the State Water Resources Control Board ("State Water Board") for PG&E's DeSabla-Centerville Hydroelectric Project, FERC Project No. 803 ("Project").

As discussed herein, PG&E has concerns with several of the conditions set forth in the Draft Water Quality Certification ("Draft Conditions").

A. Draft Condition 1: Minimum Instream Flows

Butte Creek: As set forth in Table 1, the magnitude and duration of minimum instream flows at Lower Centerville Diversion Dam from September 1 to March 14th in normal water years and from September 1 to April 30th in dry water years exceeds the storage resources in Philbrook Reservoir (see Attachment 1 for results of the operations model analysis). PG&E ran the operations model developed during relicensing and determined that in the 20-year period from 1986-2005, there would have been insufficient water resources to provide these proposed fall and winter minimum instream flows in Butte Creek in as many as nine years during the period, depending upon the operation of Philbrook Reservoir. Generally, every dry year produced a water shortage the following year. In addition, this time period (1986-2005) did not include a critically dry year (e.g., 1976-1977) where region-wide water resources were extremely low. When Philbrook Reservoir was modeled to operate to provide supplemental water during the spawning season, there was still insufficient water resources to meet the minimum instream flows specified in Table 1 in three of those 20 years (15% of the time). Philbrook Reservoir is relatively small (5,000 acre-ft) and there is insufficient storage to provide the proposed volume of cold water during the hot summer (June through August) while also maintaining high flows in

Butte Creek during the salmon spawning and incubation period. The proposed flows are likely to have a negative impact on spawning in such years. PG&E requests that the State Water Board adopt the flows stated in the draft License Article listed on page A-2 of Appendix A of the Final FERC Environmental Assessment ("EA"). These minimum flows are more conservative and are compatible with the limited storage at Philbrook.

In addition, PG&E's compliance with the flows stipulated in Table 1 depends upon the combined operation of Hendricks and Butte canals. If either canal goes out of service for any reason, including an emergency outage, it will not be possible to meet the minimum flows at Lower Centerville Diversion Dam from September 1 to March 14th, except possibly during very wet years. Therefore, PG&E requests that the State Water Board add the following footnote to this requirement: "In the case of an emergency outage of either Hendricks or Butte canals, the minimum instream flows will be the total canal inflow into DeSabla Forebay."

Draft Condition 1.A. states: "*The effects of the increased Butte Creek flows on temperature, anadromous fish and cold water habitat shall be monitored in accordance with Conditions 10, 16, and 17.*" With the initiation of full flows in Butte Creek below Centerville Diversion, it will not be possible to meet the monitoring requirements stated in Draft Conditions 16 and 17 (see discussion below under those draft conditions, respectively).

Lower West Branch Feather River below Hendricks Diversion Dam: Draft Condition 1.B. states that the "*Deputy Director may increase minimum mean daily flows in Table 2 if the design, testing of the fish ladder required in Condition 12 demonstrates that higher flows than listed in Table 2 are required for the attraction and passage of fish over Hendricks Diversion Dam.*" PG&E requests that this paragraph be removed from Draft Condition 1.B. Water temperature modeling conducted during relicensing studies clearly showed that increases in instream flow below Hendricks Diversion Dam increased water temperatures in Butte Creek. Therefore, this condition threatens listed spring-run Chinook salmon holding in Butte Creek and potentially conflicts with Draft Condition 43 which states: "*This WQC does not authorize any act which results in the taking of a threatened, endangered or candidate species or any act, which is now prohibited, or becomes prohibited in the future, under either the California ESA (Fish & Game Code §§ 2050-2097) or the federal ESA (16 U.S.C. §§ 1531 - 1544). If a "take" will result from any act authorized under this WQC or water rights held by the Licensee, the Licensee must obtain authorization for the take prior to any construction or operation of the portion of the Project that may result in a take. The Licensee is responsible for meeting all requirements of the applicable ESAs for the Project authorized under this WQC.*" As explained further below with respect to Draft Condition 12, natural barriers downstream of Hendricks

Diversion Dam will prevent upstream migrating fish from reaching a fish ladder at that location. Consequently, the aforementioned paragraph in Draft Condition 1.B should be deleted.

Draft Condition 1.B. also states *"Table 2 flows may be increased by the Deputy Director following a recommendation from the Licensee or a resource agency and submission of study data and analysis of the relationship of flow releases at Hendricks Diversion Dam and water temperature in Butte Creek, as required in Condition 10."* The relationship between flow releases at Hendricks Diversion Dam and water temperature in Butte Creek was well established during the relicensing study using water temperature models and is not improved by implementation of Draft Condition 10 (see comments below); thus, this paragraph is not necessary and should be removed.

Upper West Branch Feather River (Downstream of Round Valley Dam): Draft Condition 1.C. states: *"The Licensee shall release mean daily flows of 0.5 cfs in normal water year types and 0.1 cfs in dry water year types year-round to the Upper West Branch Feather River reach as measured at USGS gage 11405100:* This reservoir dries up annually and it is not possible to release the year-round mean daily flows stipulated in this condition. The West Branch Feather River from Round Valley Dam to Coon Hollow Springs is naturally ephemeral. Permanent flow does not occur in this river until it reaches Coon Hollow Springs. The State Water Board should therefore make the following change to this condition to make it consistent with the system: "The Licensee shall release mean daily flows of 0.5 cfs in normal water year types and 0.1 cfs in dry water years, **or natural inflow, whichever is less,** to the Upper West Branch Feather River reach as measured at USGS gage 11405100."

Philbrook Creek (below Philbrook Dam to confluence with West Branch Feather River): Draft Condition 1.D. states: *"In years when the snow water equivalent at the Humbug snow pillow sensor (HMB #823) is at least 40 inches on April 1, minimum instream flow releases to Philbrook Creek below Philbrook Dam shall be 10 cfs between April 1 and May 15."* PG&E evaluated this sensor in relation to the water year type as defined by the Sacramento Valley index. In the 30-year period from 1983 to 2012 there were 10 occurrences of the Humbug sensor being at least 40 inches on April 1. Two of those years were below normal, three were above normal, and only five were wet years. Reliance on only the Humbug snow pillow sensor to determine spring flows may result in conditions where Philbrook Reservoir will not fill and will thereby jeopardize the cold-water pool that protects spring-run Chinook salmon. PG&E notes that it can request lower releases if the reservoir is not filling as expected; however, by the time actual runoff is reliably predicted, it could be too late in the snowmelt season to capture enough water. To avoid such risk, PG&E recommends the State Water Board adopt the following modification: "In years when the snow water equivalent at the Humbug snow pillow sensor (HMB #823) is at least 40 inches on April 1 **and the forecast of unimpaired Feather River runoff at Oroville is indicative of a wet year**, minimum instream flow releases to Philbrook Creek below Philbrook Dam shall be 10 cfs between April 1 and May 15."

B. Draft Condition 6: Canal and Powerhouse Operation Water Quality Monitoring

Number 4 of Draft Condition 6 stipulates that "*Monitoring parameters shall include water temperature, dissolved oxygen, and turbidity, with sampling at defined intervals.*" PG&E requests that the requirement for dissolved oxygen (DO) be clarified. During the relicensing studies, all DO measurements taken over a two year monitoring program met Basin Plan objectives; thus, there has been no indication that the Project impacts DO. PG&E requests that clarifying language be inserted stipulating that "DO measurements are to be taken during water temperature and turbidity monitoring calibration checks."

Number 6 of Draft Condition 6 requires inclusion of "*monitoring protocol(s) for sampling and analyzing water for herbicides in receiving streams, during or immediately after scheduled herbicide treatments. . . .*" PG&E requests clarification as to whether the State Water Board's concern is related to run-off during a storm event or to drift during herbicide applications. If the intent is to determine if herbicides are entering the waterway during a run-off producing rain event, it would be more appropriate to sample during a rain event within a certain timeframe of the treatments (i.e. within 90 days of herbicide application). If the intention is to document whether herbicides are entering receiving waters due to drift, the samples would need to be taken immediately after or during treatments at pre-designated points downstream of treatments.

PG&E currently employs best management practices (BMPs) to prevent herbicides from entering waterways, including implementing no-spray buffers where treatments occur adjacent to streams and using backpack sprayers set to create large sized droplets. With these and other BMPs implemented, it is very unlikely that any herbicides will enter receiving streams from drift. Implementing sampling "*during or immediately after*" treatment would be logistically difficult because treatment crews would be moving through the system visiting several locations throughout a given treatment day. Therefore, it would be extremely difficult to define a sampling location immediately downstream of a treatment area that also allows for taking samples at the appropriate time to detect herbicides. Furthermore, any samples collected are very unlikely to capture any herbicide that may be entering the waterway due to drift because of the speed with which the waterway will move chemicals downstream.

Consequently, PG&E recommends the State Water Board require water quality sampling for herbicides during run-off producing storm events to verify that the streamside buffers and other BMPs are functioning properly. Specifically, PG&E proposes taking samples before treatments to determine a baseline level of chemicals in the water that are not due to PG&E's applications (e.g., from illegal marijuana growing), and again after treatments if there is a run-off producing storm event within a mutually agreed upon timeframe (i.e. 90 days after treatment). In addition, PG&E recommends that water quality sampling for herbicide runoff be performed for the first three years after herbicide treatments are initiated. If there are no herbicide detections within that timeframe, PG&E proposes to stop sampling. If any herbicides are detected within that timeframe, PG&E will re-evaluate and/or modify streamside buffers and other BMPs and continue sampling until there have been three consecutive years of no post-herbicide treatment detections.

Number 7 of Draft Condition 6 requires that the plan include "*Identification of the known locations of California red-legged frog, mountain yellow-legged frog, foothill yellow-legged frog,* and *Yosemite toad.*" PG&E notes that of the species listed in this requirement, only foothill yellow-legged frogs were detected in the Project Area during relicensing. PG&E further notes that foothill yellow-legged frog monitoring is addressed in Draft Condition 20, *Foothill Yellow-Legged Frog monitoring*. Consequently, PG&E requests the removal of this requirement. Furthermore, PG&E request that the reference to species other than foothill yellow-legged frog be removed from all other requirements associated with herpetofauna.

In addition, Number 9 of Draft Condition 6 requires that the required water quality monitoring plan include "*installation and operation of turbidity monitors upstream of Centerville Powerhouse in the Lower Centerville Canal spill channel and downstream of the Centerville Powerhouse. . . .*" PG&E requests the removal of the requirement to install turbidity monitors in the Lower Centerville Canal spill channel because this is a turbulent environment and monitoring equipment cannot reliably measure turbidity in such conditions. In addition, the turbidity sensor would not likely survive the high-energy environment present at this location. Finally, this measure should apply, if at all, only when the lower Centerville canal is in operation, and should be suspended when "full flow" conditions in Butte Creek take effect.

Number 10 of Draft Condition 6 requires that the water quality monitoring plan include "*Specific, measureable criteria to be used in combination with monitoring data and the list of drivers to objectively evaluate if the goals and objectives of the Water Quality Plan are being met or if the Project may be adversely affecting water quality, California red-legged frog, mountain yellow-legged frog, foothill yellow-legged frog, and Yosemite toad.*" PG&E notes that of the species listed in this requirement, only foothill yellow-legged frogs were detected in the Project Area during relicensing. And again, PG&E notes that foothill yellow-legged frogs are addressed in Draft Condition 20. Pursuant to that Draft Condition, foothill yellow-legged frog monitoring, details of methods, monitoring schedules, and reporting will be submitted to the resource agencies (which includes the State Water Board) for review and comment. Consequently, PG&E requests that this requirement be removed.

Within the body of Draft Condition 6 there is also the following requirement; "*surveys for California red-legged frog, mountain yellow-legged frog, foothill yellow-legged frog and Yosemite toad shall be on-going and data shall be recorded and provided to the Deputy Director annually by the end of January for the preceding year and to participants at the annual meeting.*" As noted above, no California red-legged frog, mountain yellow-legged frog, or Yosemite toad were detected during relicensing surveys and none would be expected to occur in future surveys. Thus, there is no project nexus for such surveys with respect to these species and PG&E requests that the requirement to perform such surveys be deleted. Furthermore, PG&E requests that any requirement to perform surveys for foothill yellow-legged frog be consistent with the U. S. Forest Service ("USFS") modified 4(e) Condition 20 Part *2* Foothill Yellow-Legged Monitoring Plan (as referenced in the State Water Board's Draft Condition 20, *Foothill Yellow-Legged Frog Monitoring*) which requires monitoring for this species "*for the first four consecutive years after License issuance and every three years thereafter*".

C. Draft Condition 9: DeSabla Forebay Water Temperature Improvements

Number 3 of the Construction section requires PG&E to provide "*A description of how the Project will be operated to continue to provide cold water to lower Butte creek during construction and when the Butte canal or pipeline is in or out of service.*" It is not feasible to continue to provide cold water to lower Butte Creek during construction of the DeSabla Forebay water temperature reduction structure because the distance (approximately 1 mile), elevation difference (1,530 feet), and slope (29%) between the forebay and Butte Creek would make construction of a temporary bypass more difficult to construct and more disturbing to the environment than the temperature control structure itself. Construction of this structure is a major undertaking and will require DeSabla Forebay to be drained, dredged, and the work to be completed under dry reservoir bed conditions. There is no feasible way to divert the canal water around the forebay during construction, nor is there a spillway that can accommodate the 50-100 cfs that is normally diverted from the West Branch Feather River. The construction will require 4-6 months and should occur during the late spring to fall period (i.e., during the summer holding period of spring-run Chinook salmon). It will not be possible to construct this structure during the winter, since rain and snow events will compromise the work site, increase environmental and safety risks, and require a longer construction period.

The condition also requires that "*The Temperature Improvement Plan shall also contain a provision for continued diversions to Upper Centerville Canal during construction.*" Similarly, it will not be possible to continue diversions to Upper Centerville Canal during construction as the reservoir will need to be drained and dry. In addition, this condition is not consistent with the 1942 Butte Creek Adjudication, paragraph 57, which allows PG&E to use the Upper Centerville Canal as a conduit for conveying water at its discretion.

Model CE-QUAL-W2 Validation and Validated Model Application: The Licensee does not support the described validation and application of the CE-QUAL-W2 (W2) water temperature model. Draft Condition 1.A. requires implementation of the existing minimum instream flows (see Table 1) for one year following completion of the DeSabla Forebay water reduction structure. Therefore, direct monitoring of the "realized" water temperature reduction within the forebay and the resulting water temperature changes downstream will be the best measure as to whether the device is achieving the intended results. The existing W2 model does not explicitly model the thermal reduction device within DeSabla Forebay. In the W2 modeling analysis, performed during relicensing, the percent of temperature reduction in DeSabla Forebay was simulated by reducing the difference between the outflow and inflow temperature by the target percentage (i.e., 50% or 80%). The actual temperature reduction achieved after construction of the structure can be directly determined through the monitoring program identified in Draft Condition 10. An evaluation of the resulting "reduction" in stream temperatures can be achieved by using the long water temperature monitoring record (available at the Butte Creek stations listed in Draft Condition 10 since 2004). This will give the actual temperature reduction, whereas the W2 model is just an estimate and is subject to error. Comparisons between model predictions and actual observed changes will not distinguish between model error and implementation issues.

D. Draft Condition 12: Hendricks Diversion Fish Screen and Passage

PG&E requests that the requirement for a fish ladder at Hendricks Diversion Dam be removed from Draft Condition 12. PG&E has already conducted an assessment of the migration corridor between Hendricks Diversion Dam and Big Kimshew Creek (Attachment 2). This assessment confirmed three natural barriers within this reach comprised of both large physical (vertical) barriers and high-velocity (flow) barriers, and three additional probable velocity barriers. The construction of the ladder is not justified because (1) natural barriers downstream of Hendricks Diversion Dam will prevent upstream migrating fish (resident rainbow trout and brown trout) from reaching such a fish ladder, and (2) there are potential adverse impacts to spring-run Chinook salmon from an increase in water temperature in Butte Creek if higher instream flows are required in the West Branch Feather River to operate the ladder.

Numbers 4 and 5 of Draft Condition 12 call for development of drivers and criteria that will address the success of a fish screen and ladder at Hendricks Diversion Dam; number 6 calls for a plan of corrective action '*in cases when the Hendricks Fish Plan's goals and objectives are not being achieved*". Because natural barriers to fish migration, located on private property downstream of Hendricks Diversion Dam, will prevent fish in the designated reach from utilizing such a fish ladder, defining meaningful standards for ladder "success" is not possible. PG&E requests that references to a fish ladder be removed.

Number 8 of Draft Condition 12 requires that the plan make a "*recommendation for the minimum flow required for operation of the fish ladder (to provide both attraction and passage). The fish screen shall be designed to comply with NMFS and CDFW fish screen criteria.*" This provision for increased flows in the West Branch Feather River for a fish ladder creates the potential for adverse impacts to spring-run Chinook salmon. Water temperature modeling done during relicensing clearly showed that increases in instream flow below Hendricks Diversion Dam directly increased the water temperatures in Butte Creek. Therefore, increased minimum flows to supply a fish ladder could adversely impact a threatened species. Moreover, natural migration barriers would severely undermine the efficacy of a fish ladder at Hendricks Diversion Dam. For these reasons, this requirement should be deleted.

E. Draft Condition 14: Resident Fish Population Monitoring

Language in Number 2 of Draft Condition 14 states that the Resident Fish Population Monitoring Plan shall include "*A description of the proposed monitoring and monitoring protocol(s) consistent with those prescribed by the USFS in its modified 4(e) Condition 20;*" Number 5 of Draft Condition 14 adds, "*At a minimum, the schedule for monitoring shall include monitoring during the third year after the license issuance and every five years thereafter for the term of the license and any annual extensions*". PG&E requests that the State Water Board adopt the language of the USFS modified 4(e) Condition 20 exactly, or delete the second statement (Number 5 of Draft Condition 14) referring to schedule, because it conflicts with the USFS modified 4(e) Condition 20 which reads: "*Fish surveys will be conducted beginning in year 3 after license issuance, and then every 5 years thereafter for the life of the license. <u>If sampling is scheduled in wet water years, it will be postponed to the next year to avoid the potential confounding effect of high flows on fish recruitment and populations</u>.*"(emphasis added) Number 5 of the Board's Draft Condition 14 contradicts the USFS 4(e) requirement that sampling only occur during non-wet years. PG&E notes that during high flows the reliability of sampling methods used (snorkeling and backpack electrofishing) will decrease. The additional error introduced in wet years, makes comparing data from multiple types of water years difficult and misleading.

The minimum monitoring requirement of Draft Condition 14 states: "*Sampling at the following locations (not limited to): the West Branch Feather River below Philbrook Creek; West Branch Feather River upstream of Hendricks Diversion; West Branch Feather River downstream of Hendricks Diversion; Butte Creek upstream of Butte Dam; Butte Creek downstream of Butte Dam; and Butte Creek upstream of DeSabla Powerhouse.*" PG&E requests that the "*not limited to*" statement be removed from this paragraph, as it implies additional sites may be required, thereby adding uncertainty.

F. Draft Condition 15: Fish Stocking

The purpose of planting fish is to mitigate for project canal impacts on fish and recreational opportunities. Measures such as Draft Condition 13 (two required fish rescues) and Draft Condition 12 (installation of the fish screen at Hendricks, if implemented) already reduce fish entrainment and mortality to fish within the Project area while increasing the opportunity of natural recruitment of fish within the system. Since the number of fish being diverted out of the stream will be reduced by the previously mentioned measures, the proposed requirement to increase trout stocking to 8,000 pounds annually is excessive. PG&E requests that the State Water Board revise Draft Condition 15 to read "The Licensee shall stock 4,130 pounds of trout annually in years in which CDFW stocks trout within the Project". This language is consistent with the 1985 Fish Stocking Agreement between PG&E and the California Department of Fish and Wildlife, in which CDFW states, "Licensee will reimburse Fish and Game for annually stocking 14,435 fish, with an approximate minimum catchable size of 3.5 trout per pound" (i.e., 4,124 pounds).

G. Draft Condition 16: Federally-and State-Listed Anadromous Fish Monitoring

Draft Conditions 16 and 17 are generally duplicative and create confusion as to how they should be implemented. As a general comment, PG&E requests that the State Water Board combine the two conditions into one condition, and permit two years to develop a plan. Due to the detail and complexity of the plans required, the two-year time frame for plan development proposed in Draft Condition 17 will be necessary. Comments on specific details of Draft Condition 16 are provided below.

Draft Condition 16 requires that PG&E create a Federally- and State-Listed Anadromous Fish Monitoring Plan for lower Butte Creek that ensures "*funding for CDFW to continue annual monitoring*" without specifying a nexus to the Project. CDFW activities on lower Butte Creek extend far beyond the bounds of the Project, and PG&E should not be required to fund activities not directly related to the Project. PG&E requests that this requirement read "...funding for CDFW adult Spring-run Chinook monitoring on Butte Creek between the Quartz Bowl barrier and the Covered Bridge..." This language reflects the scope of feasible monitoring with a nexus to the Project.

Draft Condition 16 specifies as one minimum requirement "*Annual snorkel surveys to monitor adult distribution and abundance, pre-spawn mortality surveys, and carcass surveys.*" As written, this condition is unacceptable to PG&E since under the flow requirements imposed by this draft water quality certification, snorkel surveys will not be feasible. The full flow requirements at Lower Centerville Diversion Dam under Draft Condition 1 will result in flows between 185 cfs and 300 cfs (Clint Garman, CDFW, personal communication 5/16/13). It would be unsafe to conduct snorkel surveys within the Butte Creek canyon from Quartz Bowl to the Centerville Powerhouse at these flow rates. This section of the canyon/river exhibits a very high gradient with large boulders. At full flow, surveyors could be subject to severe, even fatal, injuries. Obviously, PG&E will not expose its employees or contractors to such conditions. Secondly, even if snorkel surveys could be safely conducted, any pre-spawning data collected above the Centerville Powerhouse after the implementation of full flows would not be comparable to those data collected in that section during current conditions. Increased flows will result in faster traveling time of surveyors, increased bubble curtains that decrease visibility, and other changes in sampling methods related to the higher velocities (such as ability to dive into deeper areas). Because carcass surveys occur in the fall, when available flow is lower due to the natural hydrograph, these surveys won't be as affected by the full flow requirement. For all these reasons, the annual snorkel survey requirement should be deleted from the certification.

The second minimum requirement under Draft Condition 16 requires "*juvenile emergence and outmigration monitoring*". PG&E requests the State Water Board remove this requirement because such monitoring will be ineffective in assessing the effect of Project conditions. CDFW has found that downstream migrant trapping on Butte Creek is too prone to error to permit accurate estimates of downstream migrants. It is very difficult to calibrate trap efficiency (a key factor in determining total numbers of downstream migrants). In addition, the accuracy of trapping data in Butte Creek is compromised by high flows during the juvenile out-migration period (January through February), when the traps have to be taken out of service for the safety of personnel and to decrease mortality of juvenile spring-run Chinook salmon (Clint

Garman, CDFW, personal communication 5/16/13). The poor data quality inevitably resulting from this monitoring requirement does not justify the significant cost of conducting the study.

The third minimum requirement under Draft Condition 16 requires *"Monitoring and mapping the changes in adult SR Chinook and steelhead habitats (e.g., undercut banks, spawning gravel locations and quantity) as a result of a change in Project operation (e.g., minimum instream flows) downstream of the Lower Centerville Diversion Dam."* PG&E requests that this monitoring condition be removed because of a lack of nexus to the Project. The availability of spawning gravel as a function of minimum instream flow was assessed during relicensing. Annual variation in the quantity of gravel observed in Butte Creek is mainly due to the pattern and frequency of high storm flows with a return frequency of 1.5 years or greater, and local canyon hydraulic and geological conditions controlling scour and deposition. It is highly unlikely that changes in minimum instream flows will have a detectable influence on this annual variation. Similarly, the location of spawning riffles is not determined by instream flows, but by the shape, gradient, and geology of the Butte Creek Canyon which control the location of gravel deposition during channel forming flows. Undercut banks in this area will not change as a function of minimum instream flow because they are mainly formed by bedrock and large boulders that are resistant to change even at very high flows.

H. Draft Condition 17: Spring-Run Chinook Salmon Monitoring

As noted above, PG&E requests that the State Water Board consolidate Draft Conditions 16 and 17 and use the two-year time frame for plan development proposed in Draft Condition 17 in the combined condition.

Because Draft Condition 17 specifies many of the same technical requirements as Draft Condition 16, the comments provided regarding Draft Condition 16 also apply here. Specifically, poor visibility and unsafe conditions with full flow below Lower Centerville Diversion Dam will make it unsafe and infeasible to monitor effectively the change in distribution, abundance, and summer mortality above and below Centerville Powerhouse and to compare the results to pre-full flow conditions.

I. Draft Condition 18: Long-term and Annual Operations and Maintenance Plans, and Annual Meeting

Draft Condition 18 requires that *"The Long-Term Operation Plan shall include the Licensee's requirement to hold an annual meeting in April of each year."* and *"During the annual meeting, <u>the Licensee shall present the results of any monitoring conducted in the</u>*

previous year (emphasis added), a summary of the past year's operation and maintenance activities, and the draft Annual Operations and Maintenance Plan for the next twelve months." Later Draft Condition 19 states "*The Licensee shall provide results of benthic macroinvertebrate monitoring to the Deputy Director in a technical report following completion of each sampling effort and at least 30 days prior to the annual meeting required in Condition 18*". PG&E requests that this condition be modified to delete the requirement to report on monitoring results of the benthic macroinvertebrate monitoring and other such studies at the Annual Operations Meetings. Presenting the results of approximately 9 monitoring studies (studies required by the USFS 4(e) Conditions, the draft EA, and the water quality certification will interfere with the purpose of the Annual Operations Meetings which is to focus on how the Operations Group[1] will meet the delivery of cold water during the critical holding period of the spring-run Chinook salmon. PG&E requests that water quality temperature and anadromous fish monitoring results from previous monitoring years continue to be the focus at the Annual Operations Meeting.

PG&E also recommends that all other monitoring data be presented at the Annual Consultation Meeting which is required by the Final USFS 4(e) Condition 1 which reads "*The date of the consultation meeting will be mutually agreed to by the Licensee and the Forest Service but in general will be held 60 days prior to the beginning of the recreation season to facilitate implementation of flow management requirements and recreational management activities. Representatives from the U.S. Fish and Wildlife Service, California Department of Fish and Game, or other interested agency representatives concerned with operation of the project may request to attend the meeting.*" This would also be consistent with the draft License Article listed on page A-9 of Appendix A of the Final FERC EA. That Draft Article states "*Consistent with Forest Service 4(e) condition 1, the licensee shall also annually consult with: the California Department of Fish and Game; the California State Water Resources Control Board; the National Marine Fisheries Service, the U.S. Fish and Wildlife Service, and the U.S. Geological Survey.*"

J. Draft Condition 19: Benthic Macroinvertebrate Monitoring

This condition requires PG&E to submit a Benthic Macroinvertebrate Monitoring Plan within 180 days of license issuance. PG&E requests that the submittal of the Plan be extended to "within one year of License issuance" to make it consistent the USFS 4(e) Condition 20, Part 3 which also requires a Benthic Macroinvertebrate Monitoring Plan.

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[1] "Operations Group" is defined in a draft License Article on page A-9 of Appendix A of the Final FERC Environmental Assessment)

In addition, Number 5 of Draft Condition 19 requires that, "*At a minimum, monitoring shall be conducted during the third year of the license and every five years thereafter for the term of the license. . . . .*" PG&E recommends that the State Water Board use the same language as USFS 4(e) Condition 20, Part 3 which states that "*Surveys shall be coincident with the fish monitoring in Part 1 (unless an alternative monitoring schedule is approved in consultation with interested governmental agencies).*" This language will keep the schedule consistent with the fish monitoring required by USFS 4(e) Condition 20, which delays sampling events to the following year if sampling is scheduled during a wet year. As is the case with fish, sampling benthic macro-invertebrates during wet water years introduces error and makes annual comparisons difficult and/or misleading. The California Stream Bioassessment Protocol is designed for wadeable streams; however high flows during wet years make much, if not all, of the project streams unwadeable.

K. Draft Condition 21: Bald Eagle Monitoring

Draft Condition 21 requires that the Bald Eagle Plan monitoring, at a minimum, shall include "*One breeding and one wintering survey every three years beginning within three years of license issuance*". This is contradictory to the language of Number 6 of Draft Condition 21 which reads, "*At a minimum, reports shall be provided to the Deputy Director each year*". If surveys occur every three years then there won't be a report to submit every year. PG&E requests that the State Water Board change the language of Number 6 of Draft Condition 21 to read: "*At a minimum, reports shall be provided to the Deputy Director in the year following any year in which surveys are conducted.*"

L. Draft Condition 24: Wet Meadow

PG&E requests that this condition be deleted as not relevant to the water quality certification. The parcel that relates to this condition, Butte Creek House Ecological Reserve, is well outside of the Project boundary, is not related to the operation of the Project, and is the subject of a comprehensive agreement between PG&E and CDFW which will expire when a new license is issued.

M. Draft Condition 25: Transportation System Management

Under Number 4 of Draft Condition 25 a sentence reads "*Develop a design for reconstruction of the North Fork Feather River road crossing, below Round Valley Reservoir to the Licensee's BW45 gage.*" PG&E believes that the reference to North Fork Feather River was meant to refer to West Branch Feather River.

N. Draft Condition 26: Long-Term Operations of Centerville Development

Draft Condition 26 requires development of a plan to "allow both upstream and downstream fish passage at Lower Centerville Diversion Dam." PG&E requests that this objective be deleted because it is not supported by the facts. Upstream fish passage is not supported because natural barriers upstream, downstream, and underneath the Lower Centerville Diversion Dam ("LCDD") will prevent upstream migration even if LCDD were removed. Measures for downstream migration of resident fish are not supported because there will not be any entrainment into the Lower Centerville canal with full flows below LCDD.

A 2005 review of measurements of natural barriers above and below Lower Centerville Diversion Dam (Attachment 3) suggests that even prior to the construction of LCDD, this site was a total barrier to anadromous fish in most years. The falls at Quartz Bowl (1 mile downstream) is 11.1 feet high, while a slightly higher pre-project natural barrier (11.4 feet) forms the foundation of LCDD (Watanabe 2000[2]). Powers and Orsborn (1985[3]) identified 11 feet as a criterion for a total barrier for all species of Pacific salmon and steelhead. Observations of spring-run Chinook salmon above the Quartz Bowl barrier have confirmed this site to be a barrier to salmon migration for all but a few fish in only the wettest years.

Further, even if passage above Quartz Bowl and LCDD were available, the spawning habitat between LCDD and the impassible natural barriers upstream is only adequate to support two pairs of spawning salmon and eight pairs of spawning steelhead. Therefore, a requirement for upstream passage for anadromous fish at LCDD does not have a reasonable nexus with the Project, nor would there be a significant biological benefit in providing such passage.

Similarly a nexus for providing formal downstream fish passage is also potentially lacking. It is possible that Centerville Powerhouse will remain out of operation; if that is the case, no turbine mortality would occur for any downstream migrant resident trout entering the canal. The draft certification calls for ceasing diversion at LCDD as soon as a temperature control structure can be completed in DeSabla Forebay. Thus, all downstream migrant resident

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[2] Powers, P.D. and J.F. Orsborn. 1985. Analysis of barriers to upstream fish migration. An investigation of the physical and biological conditions affecting fish passage success at culverts and waterfalls. Part 4 of 4 of a BPA fisheries project on the development of new concepts in fish ladder design. Contract DE-A179-82BP36523. Project No. 82-14

[3] Watanabe, C. 2000. Preliminary engineering requirements for fish passage on Upper Butte Creek: An assessment of the natural barriers-DRAFT. California Department of Fish and Game.

fish would pass directly downstream from LCDD, making construction of a fish screen and bypass unnecessary. Further discussion of the possible retirement of the powerhouse with the various stakeholders will be conducted before the final water quality certification is issued and we ask the State Water Board to be open to revision of this requirement based on the conclusions reached.

In a related comment, PG&E requests that the Butte Creek barrier discussion on page 46 of the draft Mitigated Negative Declaration be revised to reflect the data in Watanabe (2000), including the fact that even if LCDD were removed, the 11.4-foot natural barrier at its foundation would still block upstream migration.

O. Draft Condition 28: Philbrook Reservoir Instream Flow Releases

Draft Condition 28 states: "*The Licensee shall make any adjustments to the minimum instream flow release valve as quickly as possible, in response to heat-related events. In any case, these adjustments should be made in less than two hours. <u>The Licensee shall submit a Philbrook Reservoir summary of valve adjustments report that includes response times every three years to the Deputy Director, by December 31. In the event that the Licensee fails to respond within two hours for any reason including unsafe conditions, the Licensee shall submit a report to the Deputy Director within 10 days of the incident. The report will include response time, reason for the delay in response, unsafe conditions and remediation to delay and/or unsafe conditions that will prevent a delay in response time in the future.</u>*"(emphasis added)

The procedures identified in the annual operation plan provide for extensive interaction with, and report out to, several agencies (including the State Water Board). Overall guidelines for operation of Round Valley and Philbrook reservoirs contain detailed contingency procedures to be followed in the case of an extreme heat event, including:

- Biweekly (Monday and Thursday) weather forecasts to the resource agencies to anticipate heat events;
- Alerting the agencies if air temperatures are forecast to exceed 105 degrees at Cohasset, with the potential for compression heating;
- Adjusting flow in consultation with the agencies;
- After temperature forecasts have returned to normal levels, reduction of releases at Philbrook Reservoir to pre-event levels (or other levels

as determined appropriate in consultation with CDF&G and NOAA Fisheries); and

- Assessment and notification of the quantity of water available for the remainder of the season.

This type of forecast-based collaborative management has proven to be an effective means of providing timely releases of additional cool water to Butte Creek and managing the limited amount of cool water available in Philbrook Reservoir.

PG&E requests that the part of the draft condition underlined above be removed since it requires redundant reporting. Details regarding releases from Philbrook Reservoir are part of the operations plans required under Draft Condition 18. PG&E originally proposed the two-hour response time as a general guideline, which it already follows. Strict reporting of literal adherence to this guideline is unnecessary, as decisions to increase flow at Philbrook Reservoir are made days in advance. Because there is a 23-hour travel time for water released from Philbrook Reservoir to reach DeSabla Forebay, the effective use of cool water from Philbrook Reservoir to moderate water temperatures in Lower Butte Creek requires anticipation of heat events. Tracking and reporting on whether the two-hour requirement on a specific flow adjustment is met increases the difficulty of implementing the license, without significantly improving the response to extreme heat events.

P. Draft Condition 30

Draft Condition 30 states: *"Project activities shall not cause an increase in turbidity downstream of the Project area greater than those identified in the SR/SJR Basin Plan. Waters shall be free of changes in turbidity that cause nuisance or adversely affect beneficial uses, and shall comply with the turbidity requirements defined in the SR/SJR Basin Plan. If monitoring shows that turbidity has exceeded the water quality objective, construction will cease and the violation will be reported immediately to the State Water Board's Deputy Director for the Division of Water Rights (Deputy Director) and the Executive Officer for the Central Valley Water Board (Executive Officer). Construction may not re-commence without the permission of the Deputy Director."*

This condition is unnecessary and does not take into consideration the major efforts PG&E has undertaken to control turbidity releases in the project area. The requirements in Draft Condition 30 are already covered by Draft Condition 6 which requires that "the Licensee shall file a Canal and Powerhouse Operation Water Quality Monitoring Plan" that requires extensive monitoring throughout the life of the license, including the development of corrective measures and a timetable for action. In addition, Draft Condition 7 requires that "the Licensee shall file a Project Canal Maintenance, Inspection, and Hazard Prevention Plan". PG&E has worked closely with the RWQCB to develop BMP's and operational protocols to prevent turbidity releases from the project.

Review of turbidity data from previous studies show that the turbidity standards in the SR/SJR Basin Plan are not biologically based and PG&E's concern is that these standards do not allow for reasonable, less than significant alterations in turbidity associated with normal hydro project operations (e.g., bringing canals back on-line after an outage). The effects of turbidity on salmonids have been studied in detail (see Newcombe 2003; Attachment 4) and have been found to be a function of the magnitude and duration of the elevated turbidity. PG&E commissioned a white paper in 2010 (Attachment 4) which indicates that short term (one day) low magnitude (+15 NTU) deviations from background levels have little to no effect on resident salmonids in cold water streams. The Basin Plan allows for a mixing zone in which turbidities can exceed criteria and allows for the establishment of averaging periods as long as beneficial uses are protected. At a minimum, Draft Condition 30 should be modified to allow a +15 NTU above background buffer based on a daily average. But again, as demonstrated in the white paper, a short-term elevation in turbidity does not impair cold water habitat beneficial uses. Furthermore, the limitations on construction, embedded within this condition, are unnecessary, duplicative, and potentially contradictory as construction projects require multiple permits that specifically identify appropriate conditions (including project specific water quality certification conditions).

Q. Draft Condition 31

Draft Condition 31 reads "*All imported riprap, rocks, and gravels used for construction shall be pre- washed.*" PG&E requests clarification: is the Water Board proposing that these materials be pre-washed specifically when used within a water course? If so, PG&E requests the State Water Board change this condition to read "*All imported riprap, rocks, and gravels used for construction within a water course shall be pre- washed.*" If not, please clarify when pre-washing should occur.

R. Draft Condition 38

Draft Condition 38 includes a purported reservation of authority "*to modify the conditions of this water quality certification to incorporate load allocations developed in a total maximum daily load developed by the State Water Board or a Regional Water Quality Control Board.*" However, Draft Condition 38 does not reference any statute authorizing this reserved authority.

The State's effort to retain jurisdiction as stated in this Draft Condition would allow the State Water Board to unilaterally change the requirements of PG&E's FERC license, in violation of the Federal Power Act. Such reservation of authority appears to contravene the express terms of the Federal Power Act, which provides in relevant part that "Licenses . . . may be altered . . . only upon mutual agreement between the Licensee and the Commission. . . ." 16 U.S.C. § 799. Therefore, PG&E recommends that this Draft Condition be removed or substantially re-drafted to conform to the State Water Board's statutory authorities in connection with the issuance of a water quality certification under the Clean Water Act.

S. Draft Condition 39

Draft Condition 39 includes a purported reservation of authority to add to or modify this water quality certification under certain stated circumstances in the future. However, Draft Condition 39 does not reference any statute authorizing this reserved authority. The State's effort to retain jurisdiction as stated in this Draft Condition would allow the State Water Board to unilaterally change the requirements of PG&E's FERC license, in violation of the Federal Power Act. Such reservation of authority appears to contravene the express terms of the Federal Power Act, which provides in relevant part that "Licenses . . . may be altered . . . only upon mutual agreement between the Licensee and the Commission. . . ." 16 U.S.C. § 799. Therefore, PG&E recommends that this Draft Condition be removed or substantially re-drafted to conform to the State Water Board's statutory authorities in connection with the issuance of a water quality certification under the Clean Water Act.

T. Draft Condition 40

Draft Condition 40 includes a purported reservation of authority to modify this water quality certification as a result in the change in baseline assumptions caused by future climate change. However, draft Condition 40 does not reference any statute authorizing this reserved authority. The State's effort to retain jurisdiction as stated in this Draft Condition would allow the State Water Board to unilaterally change the requirements of PG&E's FERC license, in violation of the Federal Power Act. Such reservation of authority appears to contravene the express terms of the Federal Power Act, which provides in relevant part that "Licenses . . . may be altered . . . only upon mutual agreement between the Licensee and the Commission. . . ." 16 U.S.C. § 799. Furthermore, it is not appropriate potentially to require PG&E to mitigate for a harm to which the Project is not contributing. There must be some nexus between this Draft Condition and a project effect that is contributing to the identified harm. The required nexus is lacking since the Project is not a cause of climate change. Therefore, PG&E recommends that this Draft Condition be removed or substantially re-drafted to conform to the State Water Board's statutory authorities in connection with the issuance of a water quality certification under the Clean Water Act.

U. Draft Condition 41

Draft Condition 41 requires compliance with all applicable requirements of the SR/SJR Basin Plan. It is unfair and inappropriate for a future compliance determination to hinge on the opinion of future regulators as to what may or may not have been intended as an "applicable requirement" of such a lengthy document, particularly one that may be changed from time to time.

Furthermore, in *East Bay Municipal Utility District et al. v. State Water Resources Control Board et al.*, Alameda County Case No. RG 10512151, the State Water Board argued – and the court agreed – that Basin Plan provisions assigning mass-based numerical waste load allocations to named dischargers "do not by themselves prohibit any conduct or require any actions on the part of dischargers. They merely set goals. What dischargers are required to do is *specified* in the waste discharge permits (NPDES permits) that they are required to obtain from Regional Water Boards." State Water Board's December 22, 2010 Brief on the Merits, 7:11-13 (emphasis added).

Thus, the State Water Board took the position that there could be no enforcement jeopardy associated with the Basin Plan unless and until *specific* requirements were articulated in a future approval issued to the discharger. Here, the "future approval" – a 401 certification – does not have the requisite *specificity* to put PG&E on notice of "[w]hat dischargers are required to do."

It is PG&E's understanding, then, that the Basin Plan's primary purpose is to provide guidance to permit writers as to what measures to incorporate into a permit; it is not itself intended primarily as a compliance document. Consequently, PG&E questions the propriety of purporting to incorporate wholesale "all applicable requirements" of the Basin Plan.

The State Water Board agreed to delete this type of draft condition from other certifications for PG&E's hydroelectric projects. PG&E requests that Draft Condition 41 be deleted here as well.

V. Draft Condition 42

Draft Condition 42 requires PG&E to comply with all water quality standards and implementation plans applicable in the future under the Porter-Cologne Water Quality Control

Act or section 303 of the CWA, and to take all reasonable measures to protect beneficial uses of Butte Creek and the West Branch of the Feather River and their tributaries. It is unfair and inappropriate for a future compliance determination to hinge on such broad and undefined requirements. PG&E requests that Draft Condition 42 be deleted or more specifically clarified.

W. Draft Condition 45

Draft Condition 45 includes a purported reservation of authority to add to or modify this water quality certification in response to a suspected violation of any condition of the water quality certification. However, Draft Condition 45 does not reference any statute authorizing this reserved authority. The State's effort to retain jurisdiction as stated in this Draft Condition would allow the State Water Board to unilaterally change the requirements of PG&E's FERC license based on a suspected violation of the water quality certification. This kind of reserved authority is in violation of the Federal Power Act. Such reservation of authority appears to contravene the express terms of the Federal Power Act, which provides in relevant part that "Licenses . . . may be altered . . . only upon mutual agreement between the Licensee and the Commission. . . ." 16 U.S.C. § 799. Therefore, PG&E recommends that this Draft Conditions be removed or substantially re-drafted to conform to the State Water Board's statutory authorities in connection with the issuance of a water quality certificate under the Clean Water Act.

X. Draft Condition 49

Draft Condition 49 states "*The Deputy Director and the Executive Officer shall be notified one week prior to the commencement of ground disturbing activities.*" PG&E requests that the State Water Board clarify this requirement since "ground disturbing activities" can range from very minor activities to those that require permits. A sweeping notification condition can hinder the scheduling and performance of minor project activities and put PG&E at risk of non-compliance. PG&E suggests limiting the notification requirement to activities for which a permit pertaining to water quality is required.

Y. Draft Condition 50

Draft Condition 50 purports to make this water quality certification subject to modification or revocation upon judicial or administrative review. Section 401 of the federal Clean Water Act, 33 U.S.C. § 1341, does not allow a water quality certification to be withdrawn once it is issued. Therefore, this Draft Condition should be removed from the water quality certification for this Project.

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CONCLUSION

PG&E would like to thank the State Water Board for the opportunity to submit these comments and welcomes the opportunity to discuss them with the State Water Board. It is PG&E's hope that it can continue to work cooperatively with the State Water Board to achieve reasonable solutions that fulfill all necessary water quality requirements while protecting existing beneficial uses, including the continuation of a clean, reliable, and economic energy source for California.

If you have any questions regarding these comments and/or would like to schedule a meeting to discuss them, please contact me at the e-mail or phone number listed above. You may also contact Tom Jereb at (415) 973-9320.

Very truly yours,

Matthew A. Fogelson

cc: Gail Cismowski, Division Chief – State Water Board
Tom Jereb, Project Manager, Power Generation – PG&E
Kimberly D. Bose, Secretary, Federal Energy Regulatory Commission
Service List for DeSabla-Centerville Hydroelectric Project, FERC Project No. 803

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Memo

To:	Ed Cheslak		
From:	Fred Holzmer	Project:	PG&E - DeSabla-Centerville Relicensing
CC:	Rick Jones, Megan Lionberger		
Date:	6/10/2013	Job No:	00394.177880

RE: Assessment of Operational Impacts to the Proposed Project for DeSabla-Centerville based on Flow Requirements of SWRCB's Draft 401 Water Quality Certification

1. Executive Summary

A numerical modeling assessment was performed by HDR to evaluate the impacts to PG&E's DeSabla-Centerville Hydroelectric Project (FERC No. 803) as a result of the California State Water Regional Control Board's draft 401 Water Quality Certification (SWRCB 401). The primary metrics evaluated in this study were: minimum instream flow violations (i.e., inability to meet minimum flow requirements); Philbrook Reservoir storage and cold water pool impacts; and power generation. An existing operations model, developed and used during the FERC relicensing process for the Project, was utilized in the assessment.

<u>Results Summary</u>

Minimum instream flow violations occurred in at least three years, or in as many as nine years, under the relicensing period of record (Water Years 1986 – 2005) at Butte Creek below Lower Centerville Diversion Dam, depending upon whether flows were augmented by release from storage at Philbrook Reservoir to help meet the minimum instream flow requirements under Condition 1 of the draft SWRCB 401. Flow violations occurred primarily during the fall spawning period for spring-run Chinook (SRC) salmon and sometimes extend into the early winter (i.e., September to early February). Flow violations generally occurred in water years classified as Normal following a water year classified as Dry.

Philbrook Reservoir cold water pool conditions were similar under SWRCB Condition 1 in summer as under the Base Case (Existing Operations) and PG&E's Proposed Project (License Application). Reservoir storage impacts, due to implementation of SWRCB Condition 1, were greatest in late summer (September) through early spring, under dry to near-normal water years, owing to increased releases from Philbrook Reservoir to meet minimum flows below Lower Centerville Dam. Generally, this is the period of time when the cold water pool in Philbrook runs out due to releases during July and August to benefit SRC holding in Butte Creek. In Dry years, when Philbrook Reservoir does not completely fill, this cold water pool is reduced in volume. No cold water pool impacts are expected to occur in above-normal to wet water years.

Relative to PG&E's Proposed Project, the average annual generation from the Project is reduced by approximately 20 percent under the SWRCB 401 Condition 1, or approximately 30 gigawatt-hours per year (GWh/yr). The primary driver for the generation loss is SWRCB's proposed cessation of diversions from Butte Creek into Lower Centerville Canal, resulting in a significant change in water available for power generation when compared to both historic (Base Case) Project operation and PG&E's Proposed Project.

2. Introduction and Background

The DeSabla-Centerville Hydroelectric Project (Project) is divided into three developments: Toadtown, DeSabla and Centerville. The Toadtown development diverts water from the West Branch of the Feather River (WBFR); the DeSabla development diverts water from upper Butte Creek as well as utilizes the outflow of the Toadtown development; and the Centerville development diverts a portion of the flow of Butte Creek downstream of the DeSabla development.

The Project recently went through the FERC relicensing process, resulting in several operating proposals from PG&E, various resource agencies, and FERC itself. FERC issued its "Staff Alternative with Mandatory Conditions" as part of its issuance of the Final Environmental Assessment for the Project on July 24, 2009. The final step in the relicensing process for the Project is the issuance of the final 401 Water Quality Certification from the SWRCB. On April 12, 2013, SWRCB issued its draft 401 Water Quality Certification for the Project. The sections below provide pertinent details of the draft SWRCB 401 with respect to operation of the Project. For the purposes of this assessment, HDR assumed the release of full flow into Butte Creek at Lower Centerville Diversion Dam (i.e., no diversions into the Lower Centerville Canal). Other assumptions, where necessary, are described below.

<u>Minimum-Instream Flows</u>

Minimum-instream flows under SWRCB 401 Condition 1 are summarized below:

A. <u>Butte Creek</u>

Within approximately five years following issuance of the FERC License, PG&E shall cease diverting water into the Lower Centerville Canal at the Lower Centerville Diversion Dam, thereby allowing full flow below Lower Centerville Diversion Dam into Butte Creek (Condition 1(A)). For modeling purposes, release of full flows in Butte Creek below Lower Centerville Diversion Dam (i.e., no diversions to the Lower Centerville Canal) was simulated for the full 20-year period of record (Water Years 1986-2005) while attempting to honor the SWRCB 401 minimum instream flow requirements below.

Butte Creek below Lower Centerville Diversion Dam	Mean Daily Flow (cfs) by Water Year	
Month	**Normal***	**Dry***
Sep 1 – Mar 14	100	75
Mar 15 – Apr 30	80	75
May	80	65
Jun – Aug	40	40

* Water year types defined per SWRCB 401 Condition 2

Butte Creek below Butte Creek Diversion Dam	Mean Daily Flow (cfs) by Water Year	
Month	**Normal***	**Dry***
Mar 1 – May 30	30	20
Jun 1 – Feb 28/29	16	10

* Water year types defined per SWRCB 401 Condition 2

Inskip Creek below Inskip Creek Diversion Dam	Mean Daily Flow (cfs) by Water Year	
Month	**Normal***	**Dry***
Year Round	0.25	0.2

* Water year types defined per SWRCB 401 Condition 2

Kelsey Creek below Kelsey Creek Diversion Dam	Mean Daily Flow (cfs) by Water Year	
Month	**Normal***	**Dry***
Year Round	0.25	0.2

* Water year types defined per Condition 2

Clear Creek below Clear Creek Diversion Dam	Mean Daily Flow (cfs) by Water Year	
Month	**Normal***	**Dry***
Year Round	0.5	0.25

* Water year types defined per SWRCB 401 Condition 2

B. Lower West Branch Feather River below Hendricks Diversion Dam

Lower West Branch Feather River	Mean Daily Flow (cfs) by Water Year	
Month	**Normal***	**Dry***
Sep - Feb	15	7
Mar - Aug	15	15

* Water year types defined per SWRCB 401 Condition 2

C. Upper West Branch Feather River below of Round Valley Dam

Upper West Branch Feather River	Mean Daily Flow (cfs) by Water Year	
Month	**Normal***	**Dry***
Year Round	0.5	0.1

* Water year types defined per SWRCB 401 Condition 2

D. Philbrook Creek below Philbrook Dam

Philbrook Creek	WY Type		
Month	**Dry****	**Normal****	**Wet***
Jan - Mar	2	2	N/A
Apr 1 - May 15	2	2	10
May 16 - Dec 31	2	2	N/A

* If Humbug snow pillow reports a Snow Water Equivalent of 40 inches or more on April 1, a "Wet year" instream flow will be implemented from April 1 through May 15.

** When instantaneous flows into Philbrook Reservoir are less than 0.5 cfs, minimum-instream flow shall be reduced to 1 cfs.

E. Hendricks Canal Feeder Creeks

PG&E shall be required to install three 4-inch pipes, one at each diversion point, to convey minimum flows. For modeling purposes, the following minimum flows were simulated:

Long Ravine	Mean Daily Flow (cfs) by Water Year	
Month	**Normal***	**Dry***
Year Round	1.0**	1.0**

* Water year types defined per SWRCB 401 Condition 2

**Or natural flow, whichever is less

Cunningham Ravine	Mean Daily Flow (cfs) by Water Year	
Month	**Normal***	**Dry***
Year Round	1.0**	1.0**

* Water year types defined per SWRCB 401 Condition 2
** Or natural flow, whichever is less

Little West Fork Creek	Mean Daily Flow (cfs) by Water Year	
Month	**Normal***	**Dry***
Year Round	1.0**	1.0**

* Water year types defined per SWRCB 401 Condition 2
**Or natural flow, whichever is less

F. Helltown Ravine

No minimum-instream flow requirement, assuming no diversions at Lower Centerville Diversion Dam to Lower Centerville Canal (Condition 1(F)). The cessation of diversions at Lower Centerville Diversion Dam is expected to occur within the first five years of new License issuance, based on other conditions proposed by SWRCB.

Water Year Types

Water Year Types under SWRCB 401 Condition 2 are summarized below:

Dry	Fifty percent or less of the average April though July unimpaired runoff of the Feather River at Oroville.*
Normal	Greater than fifty percent of the average April though July unimpaired runoff of the Feather River at Oroville.*

* Based on DWR Bulletin 120 April-July Forecast.
Bulletin 120 is tracked monthly from February through May and the model's flow requirements are adjusted as needed if water year type changes during those months.

3. Methods

The HEC-ResSim operations model, developed in collaboration with the agencies to support the DeSabla-Centerville Hydroelectric Project relicensing, was used to simulate system-wide water and power impacts under the draft SWRCB 401 Conditions. The ability to meet minimum-instream flow requirements, impacts to Philbrook Reservoir overall storage and cold water pool, and impacts to power generation were assessed.

The operations model was previously set up to simulate two system operating scenarios: the Existing Project (existing License conditions; model run code "*Base Case*"), and PG&E's Proposed Project (PG&E's proposed future License conditions; model run code "*Run 6*"). Both runs include minimum-instream flow releases below Project dams.

Model run SWRCB 401 was developed as two scenarios. In the first Scenario, Project operations were simulated using current operations whereby Philbrook Reservoir releases are managed primarily for downstream temperature control in Butte Creek during the summer and fall. In the second scenario, Philbrook Reservoir releases were managed for temperature control through August, as under Scenario 1, then starting on September 1 managed to meet the SWRCB 401 minimum instream flow requirements in Butte Creek below Centerville Diversion Dam for spring-run Chinook (SRC) spawning. The following summarizes the assumption for these two scenarios.

Scenario 1 – Reservoir Management for Temperature Control

- Perform a Period of Record Simulation (Water Years 1986 – 2005)
- Minimum instream flow requirements below Project impoundments, as specified under SWRCB 401 Condition 1
- Cessation of diverted flow to Lower Centerville Canal (i.e. full flows to Butte Creek below Lower Centerville Diversion Dam)
- Abandonment of Centerville Powerhouse
- Periodic releases of 35 cfs below Philbrook Reservoir during heat storm events
- Typical Philbrook Reservoir (guide curve) operation, comparable to the Licensee's Proposed Project

Scenario 2 – Reservoir Management for Temperature Control with Flow Augmentation for SRC Spawning

- Perform a Period of Record Simulation (Water Years 1986 – 2005)
- Minimum instream flow requirements below Project impoundments, as specified under SWRCB 401 Condition 1
- Cessation of diverted flow to Lower Centerville Canal (i.e. full flows to Butte Creek below Lower Centerville Diversion Dam)
- Abandonment of Centerville Powerhouse
- Periodic releases of 35 cfs below Philbrook Reservoir during heat storm events
- Beginning September 1, modified Philbrook Reservoir (guide curve) operation to conserve storage for potential release later as needed to meet increased minimum instream flow requirements in Butte Creek below Centerville Diversion Dam
- A new reservoir release rule at Philbrook Reservoir to release additional water from storage, as needed starting September 1, to meet increased minimum instream flow requirements in Butte Creek below Centerville Diversion Dam

Results from the SWRCB 401 Scenarios were analyzed to identify any minimum instream flow violations, with particular focus on instream flow violations below Lower Centerville Diversion Dam. Results were also assessed for impacts to the cold water pool at Philbrook Reservoir, and compared against power generation for the Base Case and the Proposed Project.

4. Results

Results of the SWRCB 401 Scenarios are summarized below:

Minimum instream Flow Violations – Minimum instream flow violations occurred at Butte Creek below Lower Centerville Diversion Dam in nine years under Scenario 1 and in three years under Scenario 2, as a result of proposed SWRCB 401 instream flow requirements in Butte Creek below Lower Centerville Diversion Dam. Violations occurred in both Dry and Normal Water Years. Flow violations occur primarily during the fall and sometimes extend into the early winter (i.e., September to early February). The number of daily average flow violations in each "Release Year"[1] under the relicensing period of record (Water Years 1986 – 2005) is summarized in Table 1 along with the Water Year type (Dry or Normal) and May-1 Bulletin 120 (B120) percent of average runoff at Oroville that dictates the September 1 release.

Under both Scenarios 1 and 2, fall and winter minimum instream flow violations are most pronounced in Release Years 1990 and 1992, each of which represents a "Normal" water year that was preceded by a "Dry" water year. It is also worth noting that minimum instream flow violations projected in HEC-ResSim assume a fully-operational canal system for the Period of Record, i.e., no unplanned outages (the model does, however, incorporate a typical annual outage for Hendricks Canal from April 16– May 9). Any unplanned canal outages may lead to additional and more severe minimum instream flow violations in Butte Creek below Lower Centerville Diversion Dam due to the frequent need to rely upon flow augmentation from Philbrook Reservoir to meet minimum flow requirements under the SWRCB 401 proposal.

[1] Note the first column in Table 1 is characterized as "Release Year (Sept 1-Aug 31)" which coincides with the onset of increased instream flows on September 1 for SRC spawning. This convention allows the spawning-flow period violations to be grouped into the "flow release year" that the spawning flows occurred. For example, Release Year 1990 is September 1, 1989 – August 31, 1990.

Table 1. Summary of minimum instream flow violations, in days per Release Year (September 1-August 31), over the full relicensing period of record (Water Years 1986-2005) for Scenarios 1 and 2. The September-1 Water Year type and May-1 B120 value controlling the September-1 minimum instream flow requirement is also summarized.

Release Year (Sept 1-Aug 31)	Number of Daily Flow Violations		September-1 Condition 2 Water Year Type	May-1 B120 Percent of average
	Scenario 1	Scenario 2		
1986	0	0	Normal	65
1987	0	0	Normal	78
1988	0	0	Dry	36
1989	5	0	Dry	35
1990	65	33	Normal	70
1991	0	0	Dry	35
1992	138	118	Normal	61
1993	27	0	Dry	46
1994	14	0	Normal	137
1995	0	0	Dry	40
1996	0	0	Normal	203
1997	0	0	Normal	111
1998	0	0	Normal	67
1999	0	0	Normal	149
2000	0	0	Normal	115
2001	3	0	Normal	99
2002	0	0	Dry	46
2003	23	4	Normal	70
2004	28	0	Normal	97
2005	7	0	Normal	70

Philbrook Reservoir Cold Water Pool – Implementation of SWRCB 401 instream flows under Scenario 1 does not greatly impact Philbrook Reservoir levels, and only marginally so from mid-November to late spring. Summer water levels, and thus the cold water pool, would be similar to PG&E's Proposed Project when compared to the SWRCB 401 Conditions. However, under Scenario 2, with the assumption that WBFR diversions are necessary to support the SRC-spawning minimum instream flow requirements in Butte Creek below Lower Centerville Diversion Dam, the SWRCB 401 proposed conditions have a much more significant impact on Philbrook Reservoir operations. The difference in operations occurs in late summer and early fall. This is generally when the cold water pool has been fully utilized by releases during July and August. Under Scenario 2, Philbrook Reservoir storage would be reserved starting on September 1 until needed in order to provide flow augmentation into Hendricks Canal in support of minimum flow criteria in Butte Creek to support SRC spawning. This change to operation at Philbrook Reservoir would also impact late summer hydropower generation for the overall project; this is addressed in greater detail within the "Power Generation" impacts section below.

Figure 1 provides simulated Philbrook Reservoir elevations for a representative sub-set of water years in the period of record (i.e., Water Years 1991-1995).

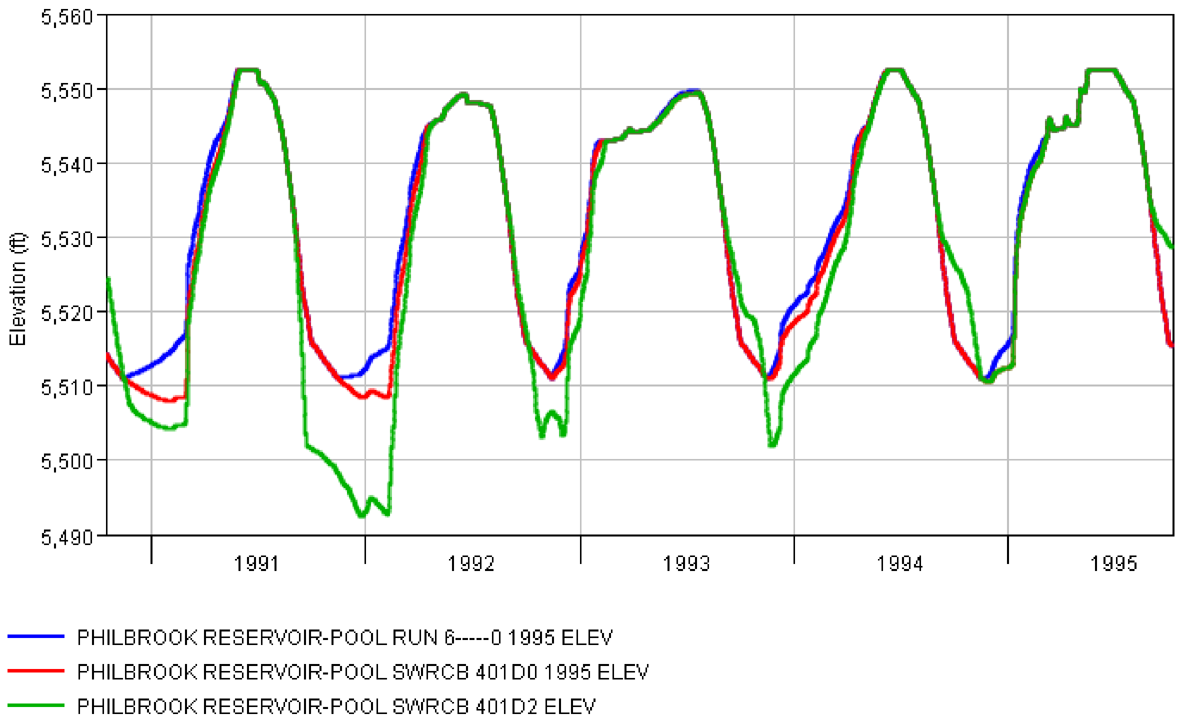

Figure 1. Time series of simulated Philbrook Reservoir water-surface elevations under PG&E's Proposed Project (Blue), Scenario 1 (Red), and Scenario 2 (Green). While not shown here, results of the Base Case simulation of Philbrook Reservoir are identical to PG&E's Proposed Project.

An analysis was performed to describe the frequency, duration and magnitude of exceedances of Philbrook Reservoir water-surface elevation over the 20-year relicensing Period of Record (Water Years 1986-2005), provided as Figures 2 through 4.

The analysis shows that in the wettest 10 percent of seasonal hydrologic conditions (i.e., 10% exceedance) over the 20-year period of record (Figure 2), there is no difference between PG&E's Proposed Project reservoir elevations and reservoir elevations under Scenario 1. Under Scenario 2 during wet hydrologic conditions, storage reserved to augment downstream minimum instream flows is generally not needed (shown in Figure 2 as unused late-year storage).

The median seasonal hydrologic condition (50% exceedance) for reservoir elevations during the period of record, shown in Figure 3, shows small variations between scenarios during mid November to early March. Under Scenario 2 (green line), the reservoir is drawn down as needed between September 1 and November 30 to augment downstream minimum instream flow requirements, but unused late-year storage remains.

In the driest 10 percent of years (Figure 4), variations between scenarios occur between mid September and early May and reservoir elevations vary by as much as 10 feet. Figures 2 through 4 all show that the cold water pool would be relatively unchanged in summer months. However, in dry year conditions, the reduced storage that occurs under all scenarios (Figure 4) would reduce the cold water pool.

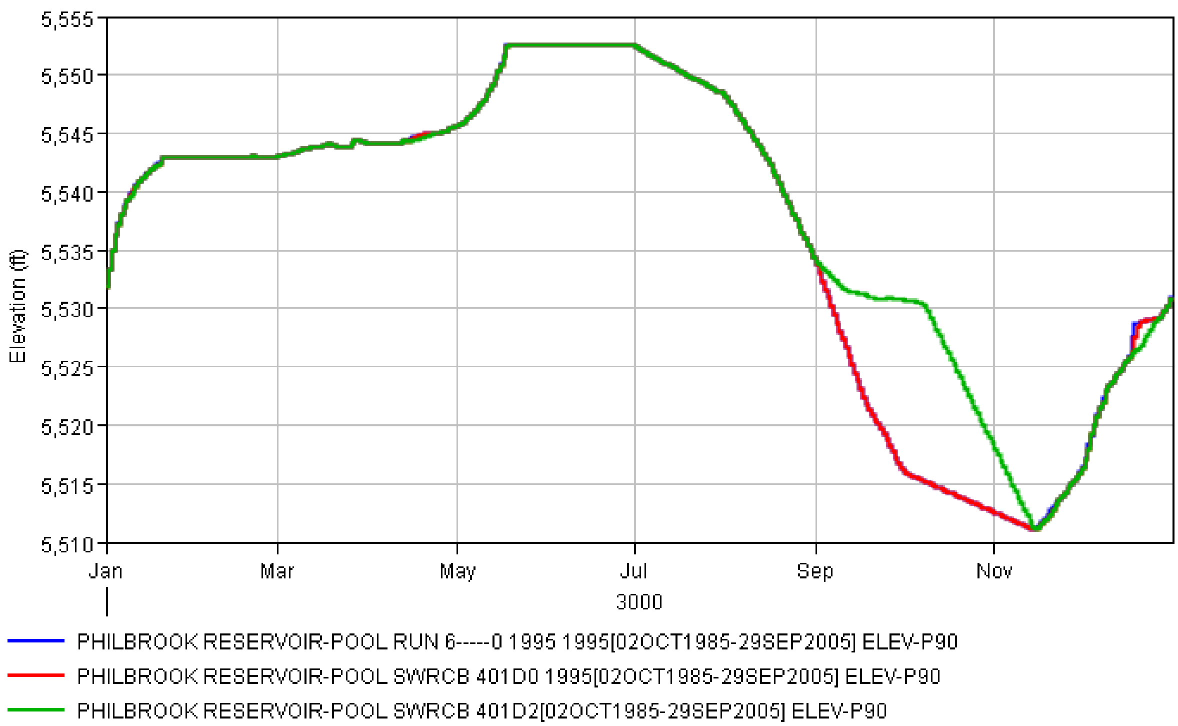

Figure 2. Wet Hydrological Conditions: 10% Exceedance water-surface elevations at Philbrook Reservoir for PG&E's Proposed Project (Blue), Scenario 1 (Red), and Scenario 2 (Green) over the Period of Record (Water Year 1986-2005). The blue line is hidden by the red where the blue line isn't visible. While not shown here, results of the Base Case simulation of Philbrook Reservoir are identical to PG&E's Proposed Project.

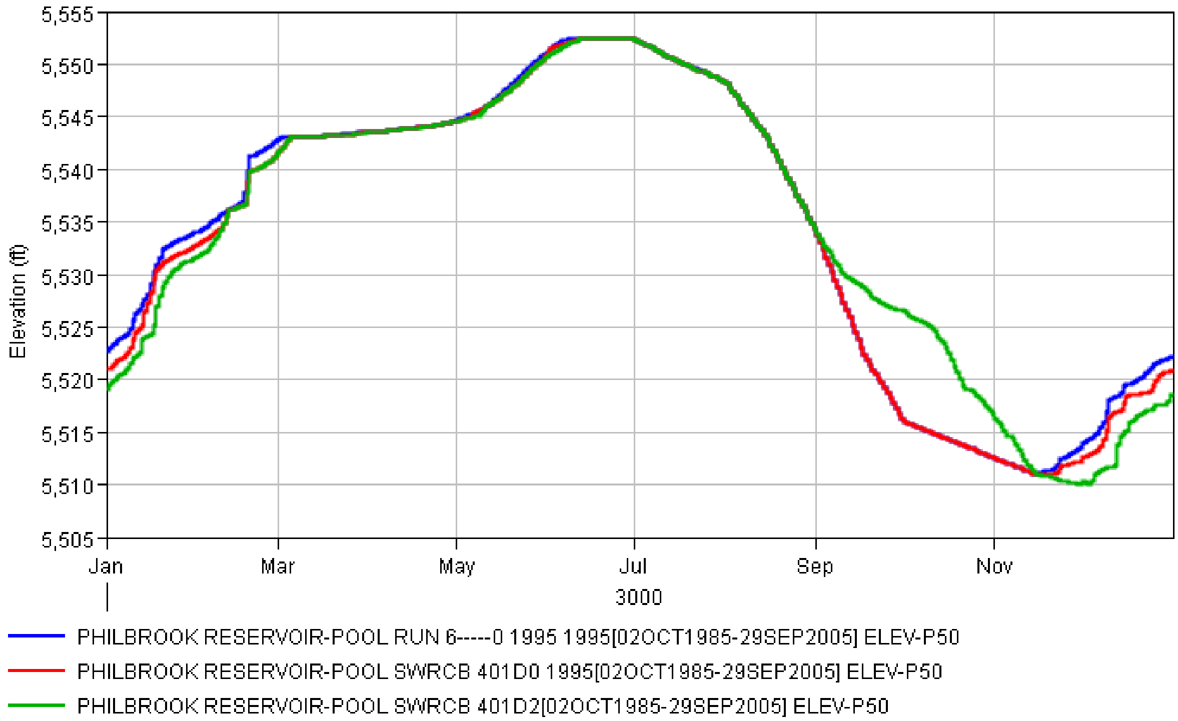

Figure 3. Median Hydrological Conditions: 50% Exceedance (median) water-surface elevations at Philbrook Reservoir for PG&E's Proposed Project (Blue), Scenario 1 (Red), and Scenario 2 (Green) over the Period of Record (Water Year 1986-2005). While not shown here, results of the Base Case simulation of Philbrook Reservoir are identical to PG&E's Proposed Project.

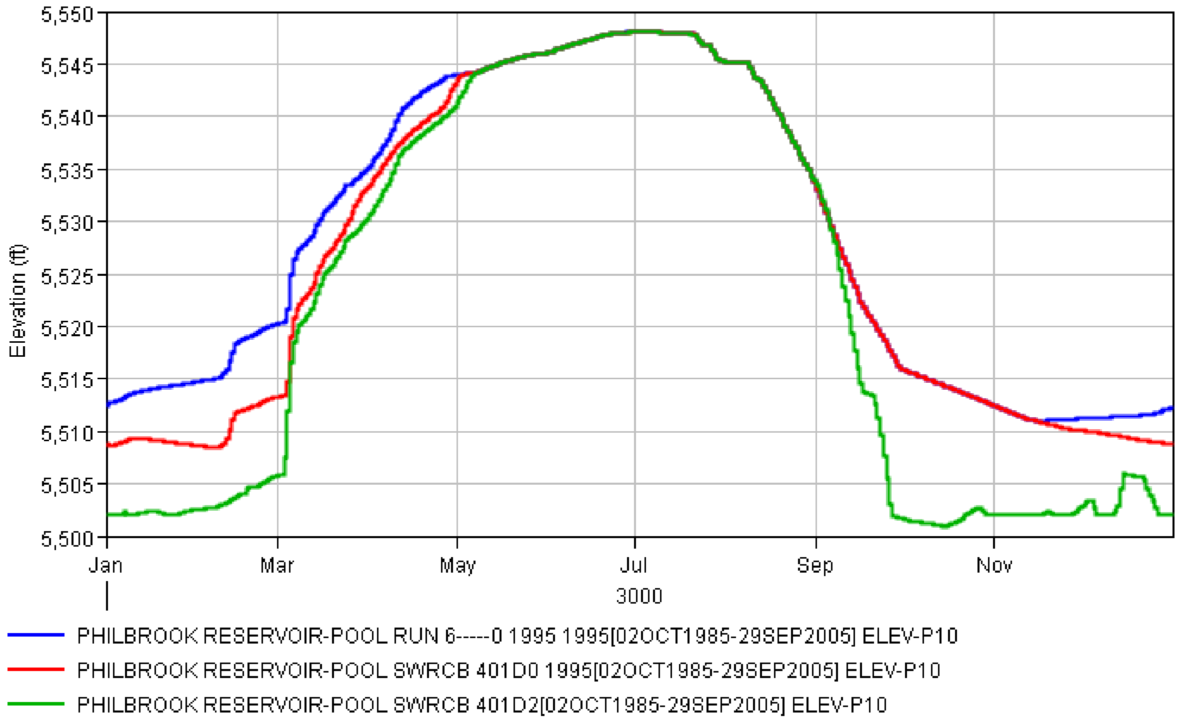

Figure 4. Dry Hydrological Conditions: 90% Exceedance water-surface elevations at Philbrook Reservoir for PG&E's Proposed Project (Blue), Scenario 1 (Red), and Scenario 2 (Green) over the Period of Record (Water Year 1986-2005). While not shown here, results of the Base Case simulation of Philbrook Reservoir are identical to PG&E's Proposed Project.

Power Generation – The following tables summarize generation results for the Base Case, PG&E's Proposed Project, and SWRCB 401 Scenarios 1 and 2. Table 2 reports results for Scenario 1 and Table 3 reports results for Scenario 2. Average generation per year for the period of record is given in the left hand side. Relative percent difference between scenarios is given in the right hand side.

Table 2. Comparison of average annual Project generation under the Base Case, PG&E's Proposed Project and Scenario 1 over the full relicensing period of record (Water Years 1986-2005).

Powerhouse	PG&E DeSabla-Centerville Project GWh/yr			Percent Difference	
	Base Case	PG&E's Proposed Project	Scenario 1	Scenario 1 vs. Base Case	Scenario 1 vs. PG&E's Proposed Project
Toadtown	7.6	7.3	7.5	-1.9%	2.0%
DeSabla	109.3	107.2	108.1	-1.1%	0.8%
Centerville	34.6	31.6	0	-100%	-100%
PROJECT TOTAL	151.5	146.1	115.6	-23.7%	-20.9%

Table 3. Comparison of average annual Project generation under the Base Case, PG&E's Proposed Project and Scenario 2 over the full relicensing period of record (Water Years 1986-2005).

Powerhouse	PG&E DeSabla-Centerville Project GWh/yr			Percent Difference	
	Base Case	PG&E's Proposed Project	Scenario 2	Scenario 2 vs. Base Case	Scenario 2 vs. PG&E's Proposed Project
Toadtown	7.6	7.3	7.5	-1.9%	2.0%
DeSabla	109.3	107.2	108.3	-0.9%	1.0%
Centerville	34.6	31.6	0	-100%	-100%
PROJECT TOTAL	151.5	146.1	115.7	-23.6%	-20.8%

Generation loss for Scenarios 1 and 2, as compared to the Base Case or PG&E's Proposed Project, is similar. Figure 5 shows simulated daily average DeSabla Powerhouse flow for a representative period, calendar years 1994 and 1995.

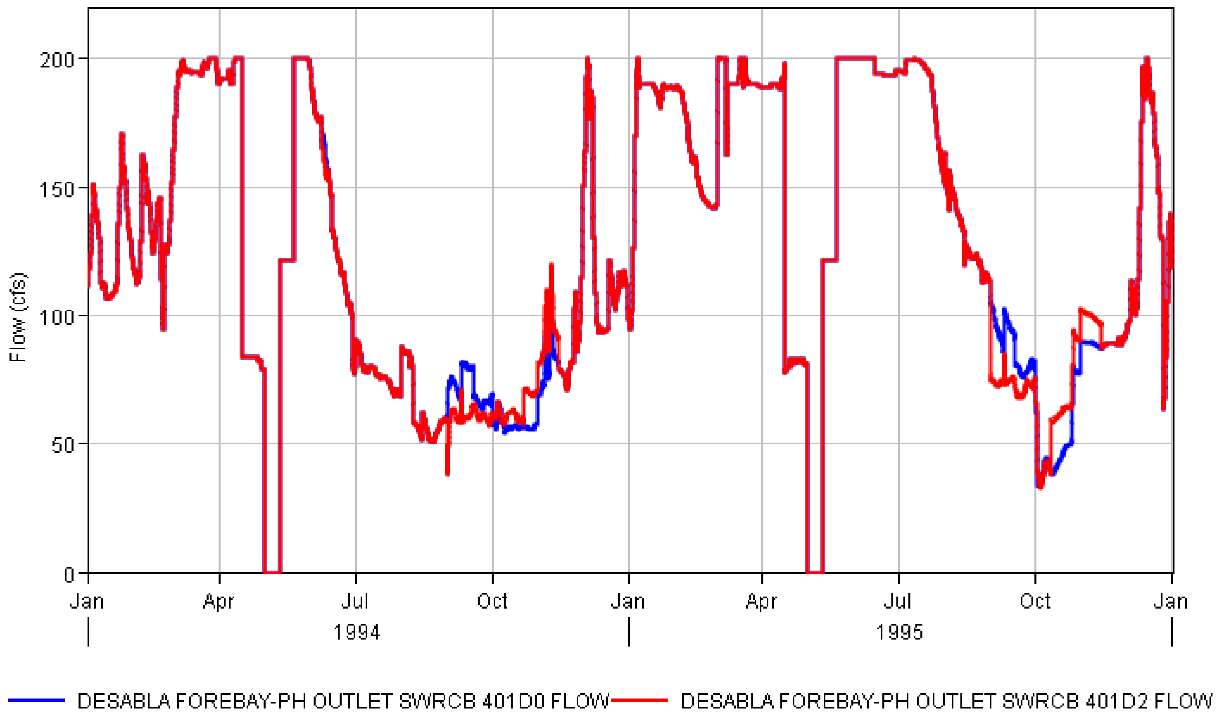

Figure 5. Time series of simulated daily average DeSabla Powerhouse flow under Scenario 1 (Blue), and Scenario 2 (Red).

5. Discussion and Conclusions

SWRCB 401 was modeled two ways: under normal operating conditions managed for temperature control (Scenario 1); and with modified operation of Philbrook Reservoir to manage for temperature control and SRC spawning flows (Scenario 2). Minimum flow violations during the fall and winter occur under both scenarios, but to a lesser degree under Scenario 2 with on-demand releases from storage starting in September to meet minimum instream flows in Butte Creek below Lower Centerville Diversion Dam (Table 1). Violations were most likely to occur in fall and early winter under slightly above-Normal Water Years that follow a Dry Water Year, such as in Release Years 1990, 1992 and 2003 (Table 1).

It is also important to consider that this modeling assessment was based on the use of the relicensing Period of Record, Water Years 1986-2005, which does not include a critically dry period, such as Water Years 1976-1977, which would further limit the availability of water resources for flow and temperature control.

Increased minimum-instream flow below Philbrook Dam in April and May, when the Humbug snow pillow sensor is at least 40 inches on April 1, had very little impact on Philbrook Reservoir water levels or the cold water pool, as reservoir releases usually exceed the 10 cfs minimum in wetter years during this period. Increased minimum-instream flow in dry months (fall and early winter) had a much greater impact on reservoir levels, especially in moderate to dry years. But, impacts were limited to times during the year when cold water is less critical downstream. The Reservoir was always able to recover to Base Case reservoir levels by mid-May, if not sooner, under both SWRCB 401 Scenarios.

Under the draft SWRCB 401, diversions from Butte Creek into Lower Centerville Canal are assumed to be eliminated for the life of the new License (Condition 1(A)). This is a significant deviation from both historical and PG&E's Proposed Project operations. As such, the impacts to Project generation are on the order of 20 percent relative to both the Base Case and PG&E's Proposed Project. Median flows over the period of record in Butte Creek below Lower Centerville Diversion Dam increased from 46 cfs under the Base Case and PG&E's Proposed Project to 184 or 186 cfs under Scenario 1 and Scenario 2, respectively. Minimum and maximum flows over the period of record were relatively unchanged under SWRCB 401.

PACIFIC GAS AND ELECTRIC COMPANY

DeSabla-Centerville Hydroelectric Project

FERC Project No. 803

ASSESSMENT OF FISH MIGRATION BARRIERS ON THE WEST BRANCH FEATHER RIVER: 2011 FIELD SURVEY AND DATA COMPILATION — DRAFT REPORT

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DESABLA-CENTERVILLE PROJECT
FERC PROJECT NO. 803

ASSESSMENT OF FISH MIGRATION BARRIERS ON THE WEST BRANCH FEATHER RIVER: 2011 FIELD SURVEY AND DATA COMPILATION

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SECTION 1.0

Introduction and Background

As part of Pacific Gas & Electric's (PG&E's) DeSabla-Centerville Hydroelectric Project (Project), PG&E operates Hendricks Diversion on the West Branch Feather River (WBFR) in Butte County, California to route water to Hendricks and Toadtown canals and then to DeSabla Powerhouse on Butte Creek. The diversion currently contains no passage facilities for resident fishes. As directed in FERC Draft License Article 415 and Forest Service 4(e) condition 19, PG&E is required to retrofit the diversion dam to include fishway structures for upstream and downstream passage of brown trout (*Salmo trutta*) and rainbow trout (*Oncorhynchus mykiss*).[1] The primary purpose of adding a fish ladder at the diversion dam would be to provide resident fish access to thermal refuge in the upper watershed during dry years when water temperatures may be elevated in downstream areas.[2] However, resource agencies had additional concerns about passage within the river, downstream of the diversion dam. Specifically, the United States Forest Service (USFS) noted that the PHABSIM-calibration[3] flows at the Retson Camp site (approximately 1.5 miles downstream of Hendricks Diversion) indicated that 7 cubic feet per second (cfs) may not support passage through shallow sections of the stream reach between Hendricks Diversion and the first major tributary, Big Kimshew Creek (FERC 2009). To ensure passage connectivity within the river, the prescribed minimum instream flow releases below Hendricks Diversion under the new License, pursuant to USFS 4(e) Condition No. 18,[4] *Streamflow*, is 15 cfs year-round, with the exception of dry water-type years, where releases may be lowered to 7 cfs[5] between September and February. Flows downstream of the reach increase with perennial input from Big Kimshew Creek.

As an alternative to increasing minimum instream-flow releases above 7 cfs at Hendricks Diversion during dry water-type years, FERC recommended that a fish passage and screen plan be developed that specifies how migration connectivity through the stream reach would be provided using fish habitat structures or other such means to increase connectivity in dry years.

This technical memorandum summarizes an assessment of fish passage barriers on the WBFR between Hendricks Diversion and Big Kimshew Creek, near the town of Stirling City (Figure 1).

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[1] The Draft License Articles are preliminary and the number was assigned based on the order listed in the FERC EA (2009). License conditions may change when the Final FERC License is accepted.

[2] FERC EA Section 5.4 referencing the June 29, 2009, section 10(j) meeting discussion

[3] PHABSIM: Physical Habitat Simulation software developed by the U.S. Geological Survey

[4] As modified April 16, 2010

[5] Flows may increase above 7 cfs if higher streamflows are needed for proper functioning of the Hendricks Dam fish passage facility.

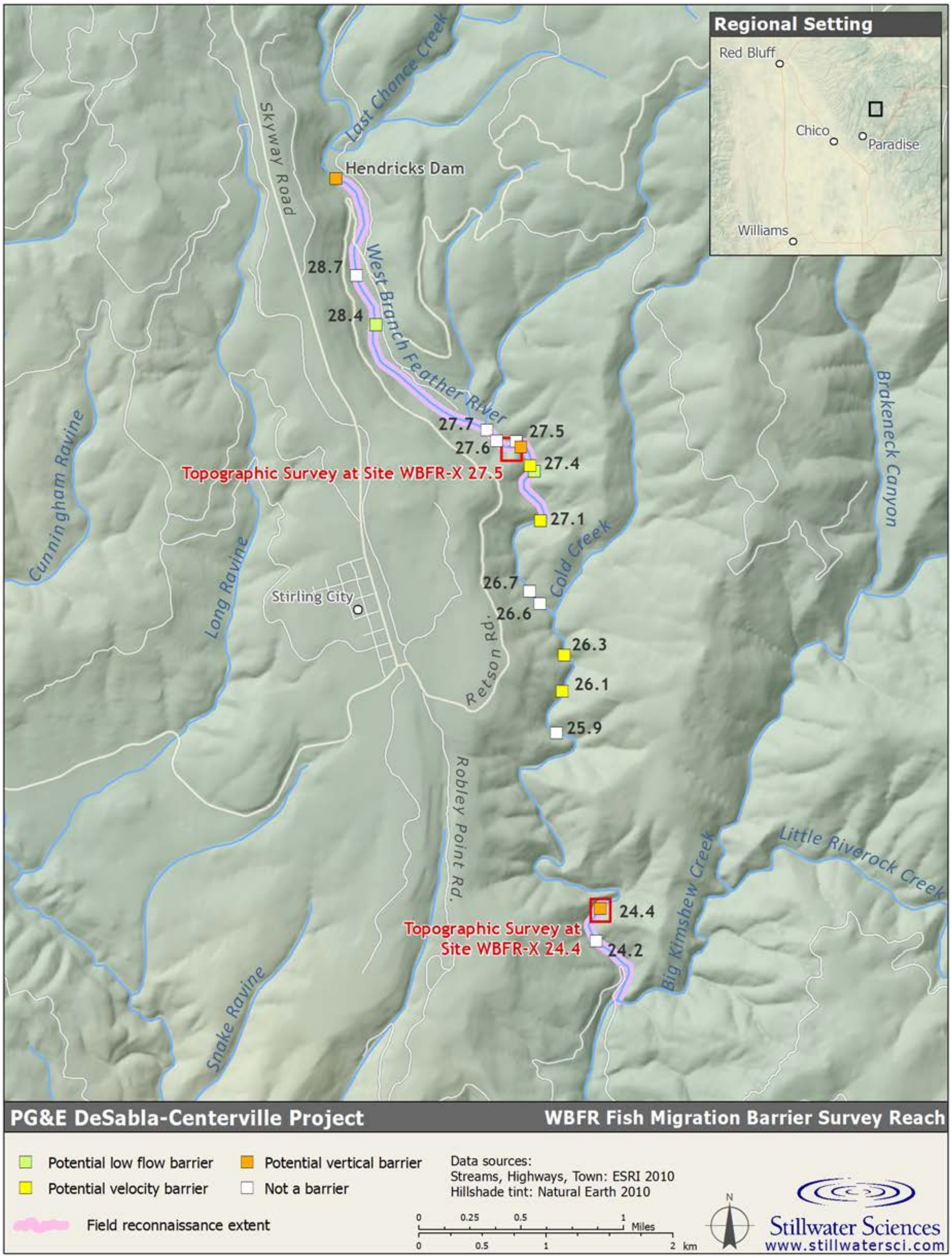

Figure 1. Overview map showing the assessment reach extent and potential fish migration barriers identified during this assessment

SECTION 2.0

Field Survey and GIS Processing Methods

This section describes the activities conducted and methods followed in the identification and characterization of potential fish-migration barriers in the assessment reach. All initial screening and field surveys were conducted in October 2011 and digital data compilation and map production using a geographical information system (GIS) was conducted during November–December 2011.

2.1 Initial Screening of Potential Fish-Migration Barriers

To initially screen the assessment reach for potential fish-migration barriers, available spatial-data sources were reviewed. These data sources included:

- California Fish Passage Assessment Database (CFPAD);[6]
- High-resolution aerial imagery collected in September 2005 by PG&E; and
- Low-elevation flyover video collected in June 2005 by PG&E.

2.1.1 California Fish Passage Assessment Database

The CFPAD is a spatial data layer that contains locations and attributes of known and potential barriers to salmonid migration in California streams. Queries of the CFPAD made within and near the assessment reach revealed three potential barriers, which are summarized in Table 1.

While the CFPAD is spatially comprehensive, its authors (CalFish, a California cooperative anadromous fish and habitat data program) stress that it is not error-proof and should only be used as an initial screening tool. Only CFPAD ID No. 715749 (Hendricks Dam) was verified during our field surveys, which did not include the middle section of the assessment reach where CFPAD ID Nos. 737357 and 736834 were reported to occur; these two barriers were not observed during initial screening of the barriers using the high-resolution aerial imagery or low-elevation flyover video.

[6] Available at: http://www.calfish.org

Table 1. Potential fish passage barriers in the assessment reach from the California Fish Passage Assessment Database (CFPAD)

CFPAD barrier identification No.	Approximate river mile	Barrier description	Barrier status	Assessed by
737357	23.2	Falls below Big Kimshew Creek (16–19 ft in height with a 16–23 ft deep base pool)	Total	California Department of Water Resources
736834	26.2	Historical upstream limit to Chinook salmon runs at Stirling City	Total	California Department of Fish and Game
715749	29.2	Hendricks Diversion Dam	Unknown	California Department of Water Resources

2.1.2 High-resolution aerial imagery

High-resolution aerial imagery was reviewed to help identify potential barriers in the assessment reach. This imagery was collected in 2005 to support various studies conducted during the Project's license application process, and was produced in a digital, georectified format with a resolution of 1 ft per pixel. Using a GIS application to view the imagery, no potential barriers were directly identified because the relief of various geomorphic features, such as bedrock outcrops and boulder riffles, could not be ascertained in this perspective. However, the aerial photos were revisited later following field survey efforts to help delineate areal dimensions of the potential barriers and associated features identified in the field.

2.1.3 Low-elevation flyover video

A preliminary assessment of the reach was made using low-elevation, oblique-perspective videography, flown in support of Project relicensing studies on June 29, 2005. The video was reviewed in support of this assessment, to locate potential migration barriers. Both potential physical (vertical drop) barriers and shallow-water locations potentially resulting from low-flow releases were noted throughout the assessment reach, and are summarized in Table 2 and shown in Figure 1. Many of the noted potential vertical barriers were field-verified during the subsequent survey effort. The shallow-water locations were assessed for passability by adult trout at 15 cfs (the flow at the time of the flyover survey), and were noted whether or not the locations would likely be passable at 7 cfs. The preliminary low-flow barrier assessment was revisited after field verification of habitat conditions and many sites were eliminated from further consideration.

2.2 Field Surveys in the Assessment Reach

Building on the results generated during the initial screening process, a field survey of most of the assessment reach was conducted from October18–20, 2011. The field team consisted of two fisheries biologists and one fluvial geomorphologist, all having experience in fish passage evaluations in similar mountain stream systems. Only the upper and lower sections of the reach were visited in the field (see Figure 1), while the remote, middle section of the reach could not be visited due to access and time constraints. Field methods entailed walking along the river bed in search of features that appeared to be potential barriers, including potential physical and hydraulic barriers.

Physical barriers were formed by the channel morphology and included vertical barriers. Features considered to be potential physical barriers were features formed by large boulders or bedrock outcrops in the river channel that created a distinct vertical step in the river's longitudinal profile (e.g., falls). Vertical barriers included a channel morphology that lacked a jump pool 1.25 times deeper than the jump height at the base of the vertical step (Flosi et al. 1998), or a resting pool at the top, thus having the potential to impede passage.

Hydraulic barriers consisted of two categories, either low-flow barriers (at flows between 7 and 15 cfs), or high-velocity barriers (such as chutes or high gradient cascades). Low-flow barriers occurred where seasonal low flows might cause a discontinuity in surface flows across a particular channel feature such as a coarse riffle, where flows may go subsurface through substrate interstices. Low-flow barriers were estimated to have water depths less than 0.4 ft and/or disconnected surface flow (Thomson 1972) at flows less than 15 cfs. High-velocity barriers generally occurred when a combination of steep slope and confined channel width created velocities in excess of 5 feet per second (fps)—the minimum adult trout burst swim speed from Alexander (1967) and Clay (1961), in combination with the range of fish lengths previously observed within the reach in 2006–2007 (PG&E 2007). General swim speeds, given in mean fish lengths per second, can be multiplied by the length of fish observed to obtain speeds in feet per second (fps). A general rule of thumb is that a fish can sustain a speed equal to about four fish-lengths per second for long periods, and a speed of about ten fish-lengths per second for short bursts (Alexander 1967 and Clay 1961). For example, a fish 3-in. long (total length) would be capable of a sustained speed of about 1 fps and a burst speed of about 2.5 fps, while a 6-in. fish could sustain a speed of 2 fps and a burst speed of 5 fps. Water velocities were considered a potential barrier if greater than 5 fps over the entire channel width with no resting locations (*e.g.*, a cascade over a bedrock sheet).[7]

Locations within the assessment reach where the PHABSIM results (PG&E 2007 [Vol. II Sec. E6.3.2.8]) indicated potential shallow-water conditions at 7 cfs were carefully assessed to

[7] Rainbow trout observed in the WBFR downstream of Hendricks Diversion ranged from 55–250 mm (2–10 in.), which equates to a sustained swim speed of 0.7 to 3.3 fps and a burst speed of 1.8 to 8.2 fps. There was one brown trout observed at 450 mm (18 in.), which equates to a sustained swim speed of 5.9 fps and a burst speed of 14.7 feet per second; however, the remaining brown trout were within the size range of the rainbow trout.

determine if shallow areas would present a barrier during flow releases less than 15 cfs. When a potential barrier was encountered, several data types were recorded, including a GPS waypoint, photos, and detailed notes. The barrier coordinates were collected using a mapping-grade, handheld GPS unit (Garmin® eTrex Venture HC) that recorded horizontal position with approximately ±20-ft accuracy. Digital photographs were taken using photographic equipment with at least 5-megapixel resolution.

Topographic surveys were made at two features that were considered to have the highest potential to impede upstream migration. These potential barriers were classified as high-velocity (WBFR-X 24.4) and vertical (WBFR-X 27.5) barriers during the field surveys (Table 2). The topographic surveys utilized total station survey equipment to capture detailed profiles of the river-channel form at and adjacent to the barriers. A Trimble® S8 robotic total station with angular accuracies up to 0.5-in. was used in combination with a Trimble Ranger controller to collect profile data. The survey methodology entailed taking a 'shot' at select locations along the river's longitudinal profile and cross-sections. A survey crew member waded in the river holding a stadia rod with attached survey prism at each survey point. Typically, these points were taken at profile inflections (*i.e.*, 'break in slope') and at the water's edge (to estimate water surface elevation). Every data point logged by the total station controller contained a unique ID and a basic descriptor.

Geo-positioning of the survey equipment was accomplished by establishing benchmarks at each of the two survey sites. Accurate GPS measurements were taken at the benchmarks and survey instrument locations using a handheld Trimble GeoExplorer XT unit (differential GPS).

Stream releases at Hendricks Diversion were approximately 17 cfs during the surveys; however, tributary accretion raised the discharge measurements at the surveyed sites to 24 cfs. The field measurements were made by establishing a transect perpendicularly across a section of the river that appeared to have uniform, steady flow (*i.e.*, non-accelerating). The flow measurement entailed gauging velocity and depth incrementally across the transect using a Marsh-McBirney® Flo-Mate 2000 flow meter and top-setting wading rod, respectively. The field-measured discharge is generally consistent with typical autumn flow conditions in this section of the river. At the inactive USGS river gage[8] just downstream of Hendricks Dam at the Retson Road Bridge, mean monthly flows for September, October, and November through the period of water years 1987–1998 were calculated to be 20, 16, and 14 cfs, respectively. The slightly higher-than 'normal' flow encountered in October 2011 is likely an artifact of the wet spring in 2011 that contributed to the persistent snow pack and late runoff occurring well into the summer and fall.

2.3 Data Reduction and GIS Processing

Upon completion of the field survey, all digitally recorded and hand-written data were promptly transferred to a common electronic project folder. All photographs were initially inventoried in a

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[8] USGS 11405200 WB FEATHER R BL HENDRICKS DIV DAM CA; data available at: http://waterdata.usgs.gov/ca/nwis/inventory/?site_no=11405200&agency_cd=USGS&amp.

Microsoft® Excel spreadsheet along with narrative descriptions, then georeferenced and imported into a GIS (ESRI® ArcGIS 10) geodatabase. Quality control and quality assurance measures were applied to check for positional and attribute errors in each photo point within shapefiles.

The topographic survey data was differentially corrected to absolute global positioning using Trimble Pathfinder Office version 5.0. All data were output in English units; the horizontal system is State Plane CA Zone II, North American Datum 1983 (NAD83); the vertical data are reported as height above ellipsoid (HAE) based on the GPS positions.

The total-station survey data were geometrically transformed (coordinate axis rotation and translation) based on the survey control points (*i.e.*, the benchmarks and instrument locations). Plot diagrams of the surveyed data in planform, longitudinal profile, and cross-section views were generated using Microsoft Excel (see below). These plots along with the survey data points were then brought into the geodatabase using ArcGIS 10. Once completed in the GIS, the geodatabase was subsequently used to create a user-friendly spatial dataset for use in Google Earth. These spatial data are included on the DVD that accompanies this technical memorandum.

2.4 Passage Assessment

In order to quantitatively assess adult rainbow trout and adult brown trout passage conditions at the two potential barriers that were believed to be complete impediments to upstream migration based on the field surveys (WBFR-X 24.4 and WBFR-X 27.5), the topographic survey data was evaluated based on methodology developed by Powers and Orsborn (1985), which uses burst swimming speed to estimate fish jumping capabilities. Fish leaping profiles were developed assuming ideal leaping conditions in the jump pool at angles of 80, 60, and 40 degrees. Burst swimming speeds were determined using Alexander (1967) and Clay (1961) based on the upper size limits of rainbow trout and brown trout observed in the WBFR downstream of Hendricks Diversion during relicensing studies (PG&E 2007).

Additionally, passage conditions at the surveyed velocity barriers (WBFR-X 24.4) were assessed using FishXing software (USFS 2012) to determine if fish could swim up the barrier.

SECTION 3.0

Results

3.1 Potential Fish-Migration Barrier Inventory

The potential fish-migration barriers identified during this assessment are summarized in Table 2 and shown in Figure 1. The table and figure do not, however, include the CFPAD-identified barriers because they include one that is downstream of the assessment reach (#737357), one that is not identified as a physical barrier (#736834), and Hendricks Diversion Dam at river mile (RM) 29.2, which is an acknowledged, but non-natural barrier structure. From the 14 potential barriers identified during the initial assessment, 7 barriers were confirmed following field surveys, including: 3 velocity barriers, 1 vertical barrier, 1 combination vertical and velocity barrier, and 2 potential low-flow barriers at flows between 7 and 15 cfs.

All of the potential barriers initially identified are summarized in Table 2 and described below in Section 3.2. Two of the barriers having the greatest potential for preventing migration were topographically surveyed for further analysis: WBFR-X 24.4 (a combination vertical and velocity barrier, located approximately 4.8 miles downstream from Hendricks Diversion Dam) and 27.5 (a vertical barrier located 1.7 miles downstream from Hendricks Diversion). The topographic survey results and detailed passage assessments are included in Section 3.3 and Appendix A. All spatial data compiled for this assessment are included on the attached DVD, included in Appendix B.

Table 2. Migration barriers in the assessment reach based on the initial review of the low-elevation flyover video and field surveys

Potential migration barrier (listed downstream to upstream)	Barrier coordinates (lat, long) [WGS 84]	Initial barrier type (based on video)	Description of feature	Method of barrier observation	Final barrier classification (based on field survey)
WBFR-X 24.2	39.879279, -121.511537	Velocity/Vertical	Bedrock cascades	Video and field	Not a barrier
WBFR-X 24.4	39.885588, -121.509125	Velocity	Bedrock cascades	Video and field	Confirmed combination vertical and velocity barrier
WBFR-X 25.9	39.898553, -121.512352	Low-flow	Boulder-cobble riffle	Video	Not a barrier
WBFR-X 26.1	39.900784, -121.512176	Velocity	Bedrock cascades	Video	Probable velocity barrier [a]

Potential migration barrier (listed downstream to upstream)	Barrier coordinates (lat, long) [WGS 84]	Initial barrier type (based on video)	Description of feature	Method of barrier observation	Final barrier classification (based on field survey)
WBFR-X 26.3	39.905058, -121.511493	Velocity	Bedrock cascade	Video	Probable velocity barrier [a]
WBFR-X 26.6	39.907747, -121.51365	Low-flow	Boulder-cobble riffle	Video	Not a barrier
WBFR-X 26.7	39.908627, -121.514593	Low-flow	Boulder-cobble riffle	Video	Not a barrier
WBFR-X 27.1	39.913529, -121.514111	Velocity	Bedrock cascade	Video and field	Confirmed velocity barrier
WBFR-X 27.4	39.917146, -121.513961	Vertical	Bedrock cascade	Field	Potential velocity barrier between 7 and 15 cfs [b]
WBFR-X 27.5	39.918696, - 121.516288	Vertical	Bedrock fall	Video and field	Confirmed vertical barrier
WBFR-X 27.6	39.918996, -121.516484	Low-flow	Boulder-cobble riffle	Video and field	Not a barrier
WBFR-X 27.7	39.919732, -121.517644	Vertical	Boulder-cobble riffle	Video and field	Not a barrier
WBFR-X 28.4	39.927876, -121.528280	Low-flow	Boulder-cobble riffle	Video and field	Potential low-flow barrier between 7 and 15 cfs [b]
WBFR-X 28.7	39.931241, -121.530214	Low-flow	Boulder-cobble riffle	Video and field	Not a barrier

WBFR-X = West Branch Feather River potential barrier; number that follows is the closest river-mile station.

[a] Barriers identified as “probable” were classified based on review of the low-elevation flyover video and follow-up field validation surveys of other potential barrier sites with comparable features within the assessment reach.

[b] Hydraulic barriers (i.e., low-flow or velocity) were assessed at surveyed flows of 24 cfs; at flows less than 24 cfs, passable portions of the channel may be dewatered, or flow could go subsurface through coarse-grained substrates, creating a discontinuity in surface flow.

3.2 Descriptions of Observed Potential Barriers

This section presents narrative and photographic descriptions of the potential barriers visited during surveys of the assessment reach. Potential passage barriers WBFR-X 24.4 and 27.5 had topographic surveys performed at each and are described in more detail in Section 3.3 below. See Figure 1 for the locations of all potential barriers discussed below.

3.2.1 Potential barrier WBFR-X 24.2

Potential barrier WBFR-X 24.2 consists of a high-velocity, bedrock cascade situated within a bedrock-confined section of the river (Figure 2). The height of the cascade as measured between the water surface below and above is approximately 6 feet. The entire feature spans the width of the channel and consists of four sub-parallel, narrow cascades. Within each cascade there are intermittent steps that could support small, temporary holding places for fish as they ascend upstream. Large calm, holding pools are present below and above the cascades. Through qualitative assessment of the feature's physical and hydraulic characteristics, it was determined that this site is not considered a barrier to fish passage.

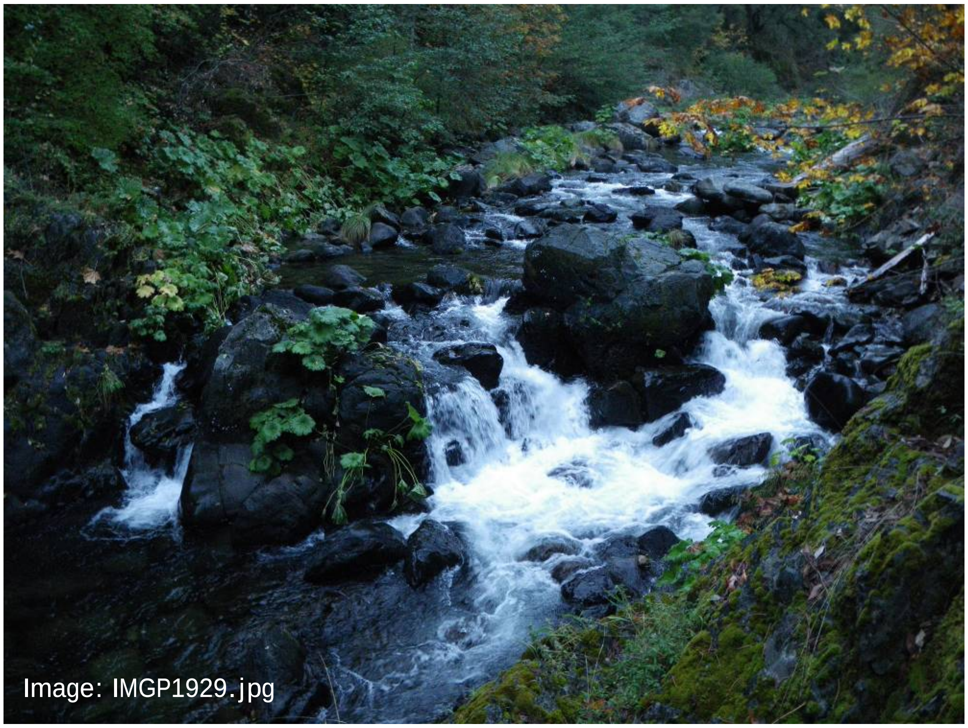

Figure 2. Photograph of WBFR-X 24.2, taken from below the barrier and looking upstream

3.2.2 Potential barrier WBFR-X 24.4

Potential barrier WBFR-X 24.4 consists of a series of three cascades formed by bedrock constrictions protruding from the valley walls and channel bed (Figure 3). Deep holding pools are present above and below each cascade. However, the steep gradient and constricted morphology of the cascades creates potential high-velocity barriers to upstream passage. Because this barrier was considered to have a higher potential to prevent passage to trout, it was surveyed in more detail, as described in Section 3.3, *Detailed Survey Data from Barriers WBFR-X 24.4 and 27.5*.

Figure 3. Oblique aerial photograph of WBFR-X 24.4, taken from below the site and looking upstream, as viewed from helicopter

3.2.3 Potential barrier WBFR-X 25.9

Potential barrier WBFR-X 25.9 is a coarse-grained riffle that spans the river channel and is composed of cobbles and boulders. This site was identified during review of the low-elevation flyover video and was described as a potential low-flow barrier (Figure 4). Although this site was not visited in the field, after ground truthing at similar potential low-flow barrier sites, like WBFR-X 28.7, it was determined that there are likely no low-flow impediments that would be anticipated to restrict passage at 7 cfs. Therefore, this site is not considered a barrier to fish passage.

Figure 4. Oblique aerial photograph of WBFR-X 25.9, taken from below the site and looking upstream, as viewed from helicopter

3.2.4 Potential barrier WBFG-X 26.1

Potential barrier WBFR-X 26.1 consists of two in-line, high-velocity, bedrock cascades situated within a bedrock-confined section of the river. This site was identified during review of the low-elevation flyover video and was described as a potential velocity or vertical barrier (Figure 5). Although this site was not visited in the field, its appearance in the video exhibits physical and hydraulic characteristics similar to those observed at potential velocity barrier sites visited in the field, such as WBFR-X 27.1. The similar features include narrow and presumably steep cascades with high velocity flow, as evidenced from whitewater constrained by bedrock and boulders in the wetted channel (see Figure 5). Therefore, this site is classified as a probable velocity barrier to fish passage.

Figure 5. Oblique aerial photograph of WBFR-X 26.1, taken from below the barrier and looking upstream, as viewed from helicopter

3.2.5 Potential barrier WBFR-X 26.3

Potential barrier WBFR-X 26.3 consists of a high-velocity, bedrock cascade situated within a highly confined section of the river with prominent bedrock and boulder constrictions. This site was identified during review of the low-elevation flyover video and was described as a potential velocity or vertical barrier (Figure 6). Although this site was not visited in the field, its appearance in the video exhibits physical and hydraulic characteristics similar to those observed at potential velocity barrier sites visited in the field, such as WBFR-X 27.1. The similar features include narrow and presumably steep cascades with high velocity flow, as evidenced from whitewater very narrowly constrained by bedrock and boulders in the wetted channel. Therefore, this site is considered a probable velocity barrier to fish passage.

Figure 6. Oblique aerial photograph of WBFR-X 26.3, taken from below the barrier and looking upstream, as viewed from helicopter

3.2.6 Potential barrier WBFR-X 26.6

Potential barrier WBFR-X 26.6 is a coarse-grained riffle that spans the river channel and is composed of cobbles and boulders. This site was identified during review of the low-elevation flyover video and was described as a potential low-flow barrier (Figure 7). Although this site was not visited in the field, ground truthing at similar potential low-flow barrier sites, like WBFR-X 28.7, it was determined that there are likely no low-flow impediments that would be anticipated to restrict passage at 7 cfs. Therefore, this site is not considered a barrier to fish passage.

Figure 7. Oblique aerial photograph of WBFR-X 26.6, taken from below the barrier and looking upstream, as viewed from helicopter

3.2.7 Potential barrier WBFR-X 26.7

Potential barrier WBFR-X 26.7 is a coarse-grained riffle that spans the river channel and is composed of cobbles and boulders. This site was identified during review of the low-elevation flyover video and was described as a potential low-flow barrier with a long, deep pool situated immediately downstream (Figure 8). Although this site was not visited in the field, ground truthing at similar potential low-flow barrier sites, like WBFR-X 28.7, it was determined that there are likely no low-flow impediments that would be anticipated to restrict passage at 7 cfs. Therefore, this site is not considered a barrier to fish passage.

Figure 8. Oblique aerial photograph of WBFR-X 26.7, taken from below the barrier and looking upstream, as viewed from helicopter

3.2.8 Potential barrier WBFR-X 27.1

Potential barrier WBFR-X 27.1 consists of a high-velocity, bedrock cascade situated within a bedrock-confined section of the river (Figure 9). The height of the cascade, as measured between the water surface below and above, is approximately 7.5 feet. Calm, holding pools are present below and above the cascade. There are at least two distinct 'steps' along the cascade's profile that could aid fish passage; however, the upper section of the cascade has a total height of 6.5 feet over a length of 5.5 feet (as measured at the base of the stadia rod in Figure 9), resulting in a high slope, and high velocity. There is a side channel associated with the location; however, at a total flow of 24 cfs, the side channel contained approximately 0.25 cfs and would be impassible by adult trout. Therefore, this site is considered a velocity barrier to fish passage.

Figure 9. Photograph of WBFR-X 27.1, taken from below the barrier and looking upstream, with a 10-ft stadia rod shown for scale

3.2.9 Potential barrier WBFR-X 27.4

Potential barrier WBFR-X 27.4 is a suite of high-velocity, sub-parallel cascades at a relatively broad bedrock 'step' along the river bed (Figure 10). The bed morphology and hydraulics here are accordingly complex, exhibiting a nearly random pattern of bedrock and boulders and, thus, turbulent and quiescent flow. At the distinguishable cascade features, flows are fast as they spill down the steep, 6 to 10-ft high cascades with scattered high velocity chutes and 2–3 foot vertical drops. Holding pools with slow water are present above and below the cascades. Within the cascades, short bedrock steps are interspersed that could aid in fish passage. At flows less than 24 cfs, portions of the channel containing the smaller vertical steps may be dewatered. Therefore, this site is considered a potential velocity barrier at flows less than 24 cfs.

Figure 10. Photograph of WBFR-X 27.4, taken from below the barrier and looking upstream, with a 6-ft person standing on top for scale

3.2.10 Potential barrier WBFR-X 27.5

Potential barrier WBFR-X 27.5 consists of a 5-ft high vertical drop, or waterfall, along the river bed composed of resistant bedrock (Figure 11). Deep holding pools are present above and below the barrier, which lies in a constrained gorge-like canyon with high bedrock outcrops on either side. Because this was the first significant migration barrier encountered downstream of Hendricks Diversion, it was surveyed in more detail, which is described in Section 3.3, *Detailed Survey Data from Barriers WBFR-X 24.4 and 27.5*.

Downstream of the jump pool was boulder riffle that, based on the aerial video, warranted a field visit. During the site visit, it was determined that the boulder riffle did not present any passage concerns and was not inventoried as a potential barrier; however, photos were taken and are included as part of the WBFR-X 27.5 site within the GIS files.

Figure 11. Oblique aerial photograph of WBFR-X 27.5, taken from below the barrier and looking upstream, as viewed from helicopter

3.2.11 Potential barrier WBFR-X 27.6

Potential barrier WBFR-X 27.6 is a coarse-grained, low gradient riffle at the lower end of the relicensing PHABSIM site located near Retson Camp. The riffle spans the river channel and is composed of cobbles and boulders (Figure 12). The PHABSIM study conducted during relicensing indicated that the coarse nature of this site had the potential for limited surface flow at 7 cfs; however, the survey crew determined that passage would be provided at 7 cfs given the low channel gradient and sufficient depths between large substrate particles. Therefore, this site is not considered a barrier to fish passage.

Figure 12. Photograph of WBFR-X 27.6, view looking upstream toward coarse riffle with standing person in center for scale

3.2.12 Potential barrier WBFR-X 27.7

Potential barrier WBFR-X 27.7 was identified during review of the low-elevation flyover video and was initially described as a vertical, bedrock-controlled feature along the river bed and left bank (Figure 13). It was later determined in the field, however, that this feature does not present a continuous barrier across the river's wetted width. While bedrock impinges on the left and right banks of the channel, it does not form a continuous outcrop across the channel bed. Instead, the bedrock forms a deep, long pool just below a short, cobble-boulder riffle. It was determined that the riffle does not present a potential impediment to migrating fish and, therefore, this site is not considered a barrier to fish passage.

Figure 13. Photograph of WBFR-X 27.7, view looking upstream from the deep pool adjacent to the bedrock outcrop and towards the short, coarse riffle in the distance

3.2.13 Potential barrier WBFR-X 28.4

Potential barrier WBFR-X 28.4 is an abrupt, coarse-grained riffle composed of cobbles and boulders that spans the majority of the river channel, with a small side channel along the right bank (Figure 14). The riffle and the side channel are perched above the long, wide pool immediately downstream. The height of the riffle, as measured between the water surface below and above, it is approximately 6 feet. This feature was determined to be a potential low-flow barrier, due to its coarseness, that may contain limited surface flow at a dry-year flow release of 7 cfs. At 7 cfs, flow through this feature could go subsurface through the coarse-grained substrates and create a discontinuity in surface flow. The side channel also appears to be a potential low-flow barrier at 7 cfs due to low water volume and depth. Therefore, this site is a potential low-flow barrier between flows of 7 and 15 cfs.

Figure 14. Photograph of WBFR-X 28.4, view looking upstream toward tall, coarse riffle with surveyor on right side (left bank) for scale

3.2.14 Potential barrier WBFR-X 28.7

Review of the low-elevation flyover video identified potential barrier WBFR-X 28.7; however, similar to potential barrier WBFR-X 27.7, it was determined during the field survey that no barrier spanning the width of the river is present here (Figure 15). The river morphology is plane-bedded with very little topographic expression that could interrupt fish migration. There were no low-flow impediments identified (e.g., cobbles or boulders) that would be anticipated to restrict passage at 7 cfs and, therefore, this site is not considered a barrier to fish passage.

Figure 15. Oblique aerial photograph of WBFR-X 28.7, taken from below the site and looking upstream, as viewed from helicopter

3.3 Detailed Survey Data and Quantitative Passage Assessment for Barriers WBFR-X 24.4 and 27.5

This section presents images, profile plots, and narrative descriptions of the two migration barriers considered to have the greatest potential to prevent passage to trout. These sites are located at vertical bedrock drops and/or swift-water cascades along the river bed. Their locations are shown in Figure 1 relative to the other potential barriers identified in the assessment reach. The leaping abilities of rainbow trout and brown trout used in the assessment, based on upper size limits observed in the WBFR downstream of Hendricks Diversion during relicensing studies, are summarized in Table 3.

Table 3. Leap height and distance ability of rainbow trout and brown trout in the WBFR

Species (upper size-limit length)	Leaping angle (degrees)	Burst speed (fps)[1]	Height of leap (ft)[2]	Distance at high point (ft)[2]
Rainbow trout (10 in)	40°	8.2	1.01	0.36
	60°	8.2	0.78	0.90
	80°	8.2	0.43	1.03
Brown trout (18 in)	40°	14.7	3.25	1.15
	60°	14.7	2.52	2.91
	80°	14.7	1.39	3.30

[1] Based on Alexander (1967) and Clay (1961): Rainbow trout observed in the WBFR downstream of Hendricks Diversion ranged from 55–250 mm (2–10 in.), which equates to a sustained swim speed of 0.7–3.3 fps and a burst speed of 1.8–8.2 fps. One brown trout was observed at 450 mm (18 in.), which equates to a sustained swim speed of 5.9 fps and a burst speed of 14.7 fps; the remaining brown trout were within the size range of the rainbow trout.

[2] Based on Powers and Osborn (1985)

3.3.1 Potential barrier WBFR-X 24.4

The lowermost classified barrier in the assessment reach is site WBFR-X 24.4, approximately 4.8 miles downstream of Hendricks Diversion and 0.5 miles upstream of Big Kimshew Creek. The barrier consists of a series of three cascades formed by bedrock constrictions protruding from the valley walls and channel bed. Flow was fast at these cascades due to the steep gradient (near-vertical) and constricted wetted widths (~6 feet). Deep holding pools are present above and below each cascade. Due to the constricted morphology of the cascades, it is assumed that they convey flow even during much lower flows; however, this has not been verified. Within the pools, a mixture of sand, gravel, and cobble mantled the underlying bedrock.

An aerial perspective of the site is shown in Figure 16, along with the locations of the surveyed longitudinal profile and cross-sections. A representative field photo of the potential barrier and longitudinal profile is presented in Figures 17 and 18. The planform map of the surveyed data points and 11 cross-sectional profiles are included in Appendix A.

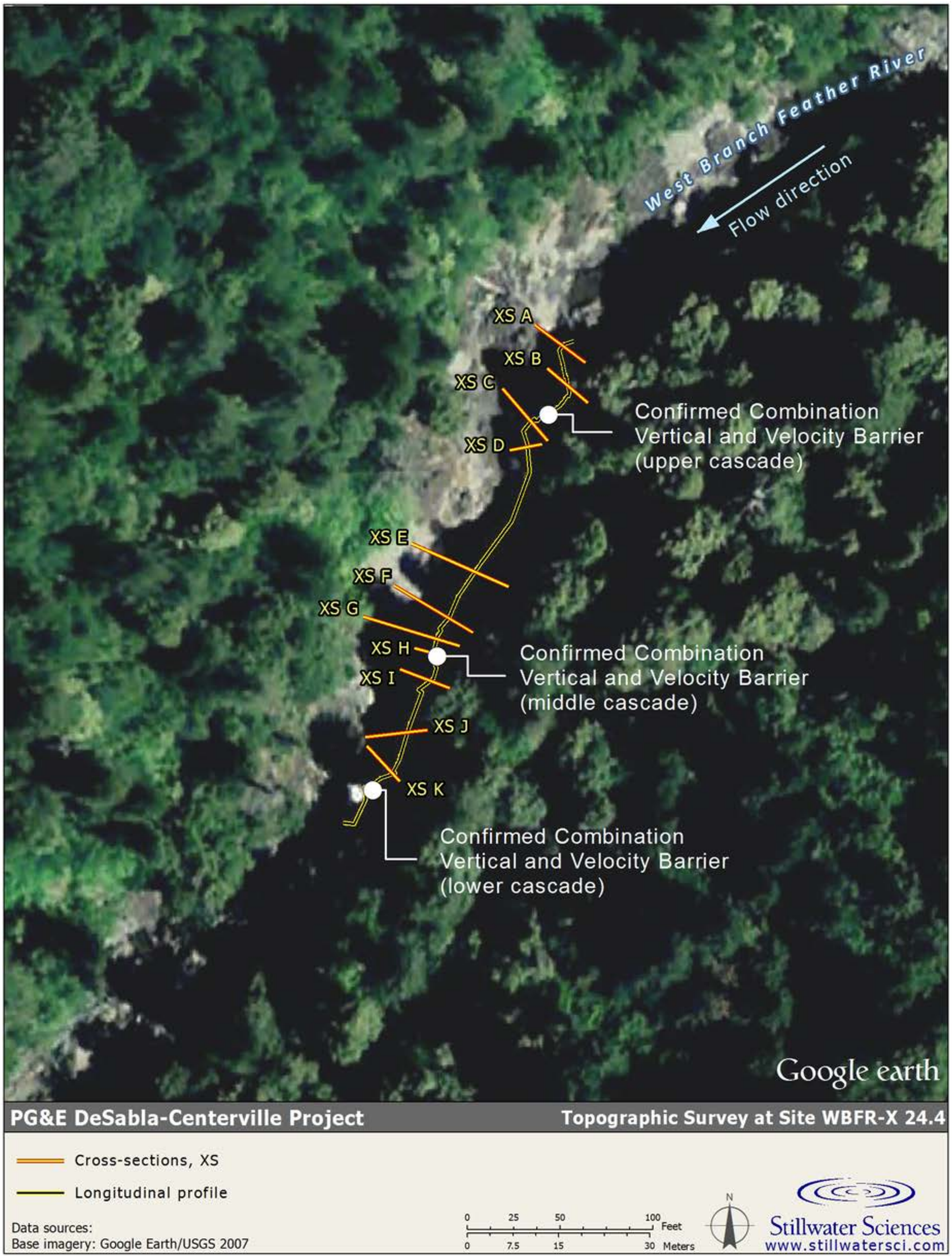

Figure 16. Overview map of WBFR-X 24.4 showing the surveyed longitudinal profile and cross-sections

Figure 17. Photograph of WBFR-X 24.4, view from the right bank looking upstream toward the upper cascade with a 6-ft tall person standing in the cascade for scale

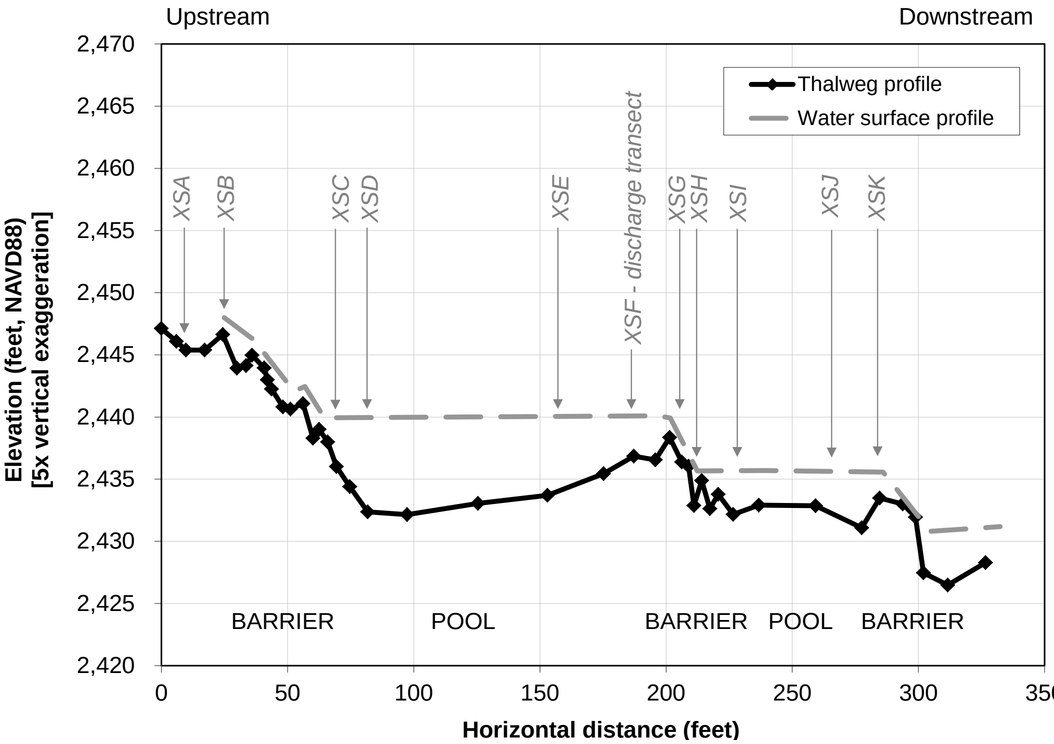

Figure 18. Longitudinal profile and surveyed cross-sections (XS) at locations A-K of WBFR-X 24.4, showing three cascades (upper, mid, and lower)

The three cascades surveyed at WBFR-X 24.4 were all classified as vertical migration barriers to species and size ranges of fish expected to occur within the study reach, based on fish observed during relicensing studies. Leaping abilities of both rainbow trout and brown trout limit passage at three locations at the flows measured (24 cfs) (Table 3 and Figures 19–21). It is expected that passage ability would not improve with lower flows; however, as flows increase, the vertical drop (i.e., fish jumping height) has the potential for reduction, which may allow passage.

Additional analysis using FishXing software (USFS 2012) was conducted at each of the cascades located at WBFR-X 24.4 in order to assess the ability of rainbow trout and brown trout to swim up the cascades between flows of 7 cfs and 15 cfs. Both depth and velocity were found to limit upstream passage at each of the individual cascades. Therefore, each of the cascades surveyed at WBFR-X 24.4 are both vertical and velocity barriers between 7 cfs and 24 cfs.

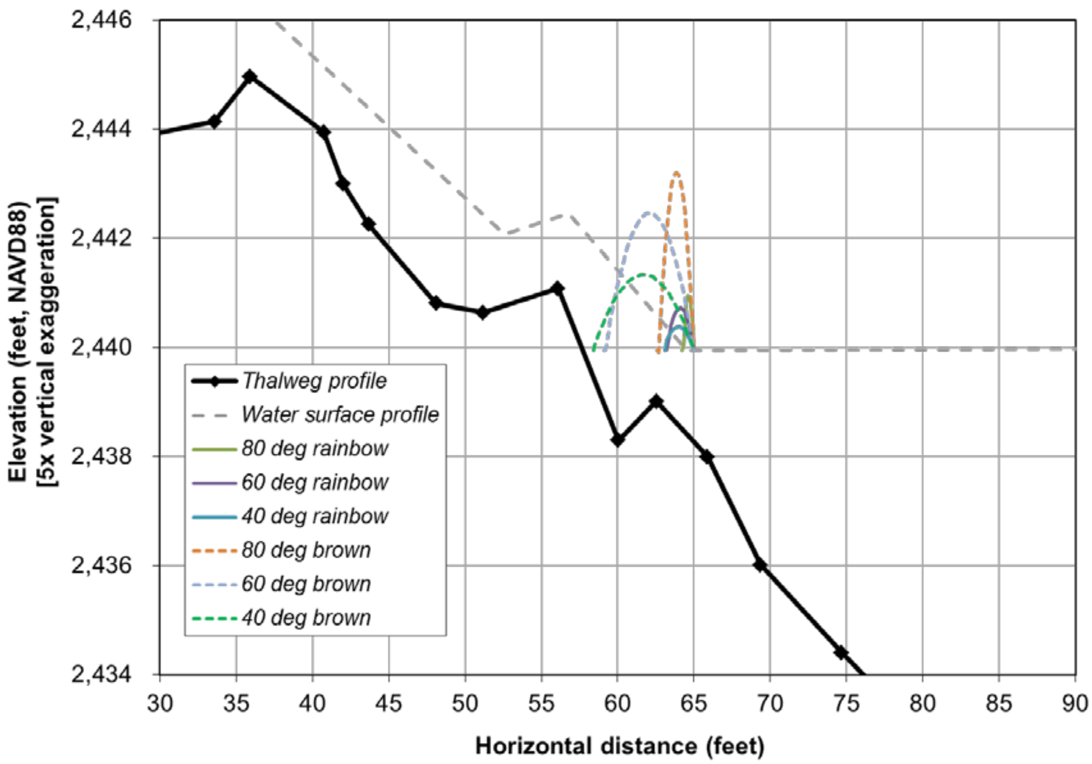

Figure 19. WBFR-X 24.4 upper cascade showing fish leaping capabilities at 80, 60, and 40 degree angles for rainbow trout and brown trout

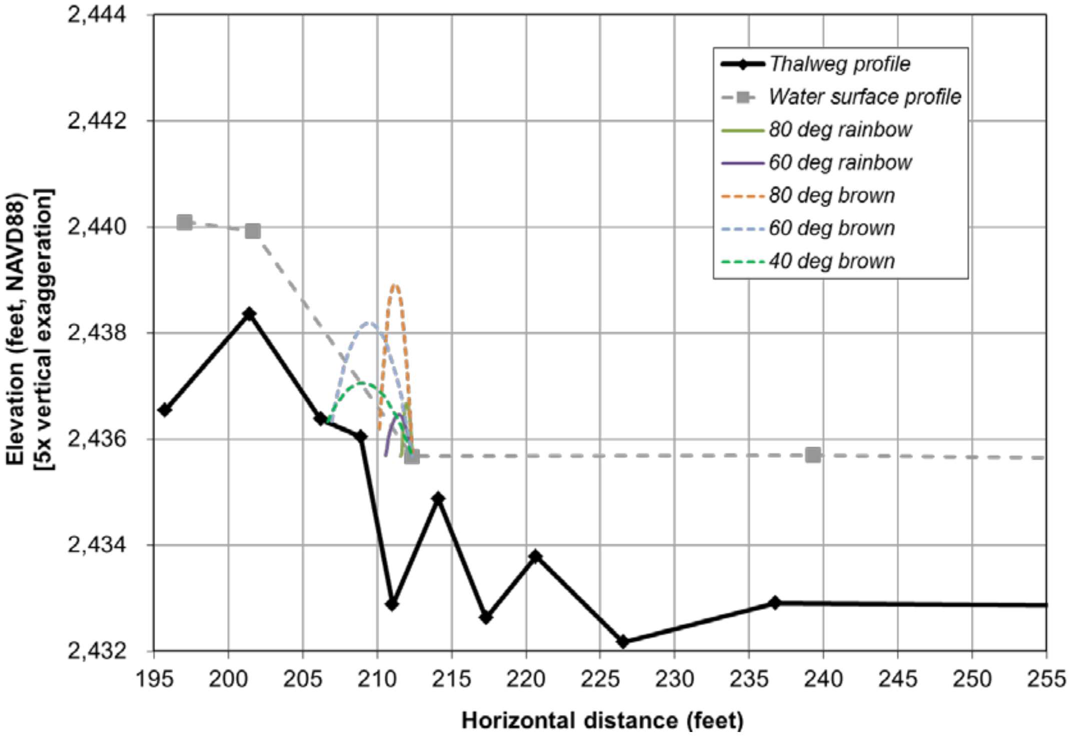

Figure 20. WBFR-X 24.4 mid cascade showing fish leaping capabilities at 80, 60, and 40 degree angles for rainbow trout and brown trout

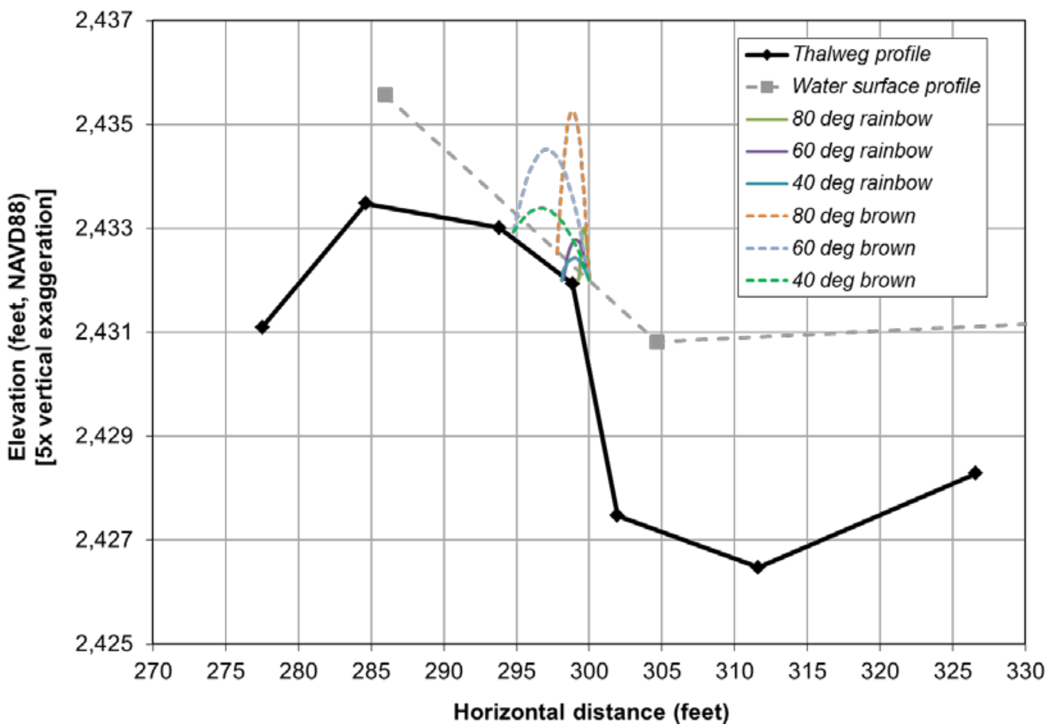

Figure 21. WBFR-X 24.4 lower cascade showing fish leaping capabilities at 80, 60, and 40 degree angles for rainbow trout and brown trout

3.3.2 Potential barrier WBFR-X 27.5

The uppermost potential vertical barrier in the assessment reach is site WBFR-X 27.5, consisting of a 5-ft high vertical drop, or waterfall, along the river bed composed of resistant bedrock. The entire river segment here runs through a constrained gorge-like canyon with high bedrock outcrops on either side, and underlying the river channel itself. Deep holding pools are present above and below the barrier, but the downstream pool is substantially longer, measuring about 300 feet in length from the base of the barrier and on through the pool to its downstream grade-control (coarse-grained riffle). This barrier likely always conveys flow even during low-flow conditions, although this condition has not been verified.

An aerial perspective of the site is shown in Figure 22, along with the locations of the surveyed longitudinal profile and cross-sections. A representative photo of the potential barrier and longitudinal profile is shown in Figures 23 and 24.

Figure 22. Overview map of WBFR-X 27.5 showing the surveyed longitudinal profile and cross-sections

Figure 23. Photograph of WBFR-X 27.5, view from the left bank looking upstream toward the waterfall barrier with a 6-ft tall person for scale

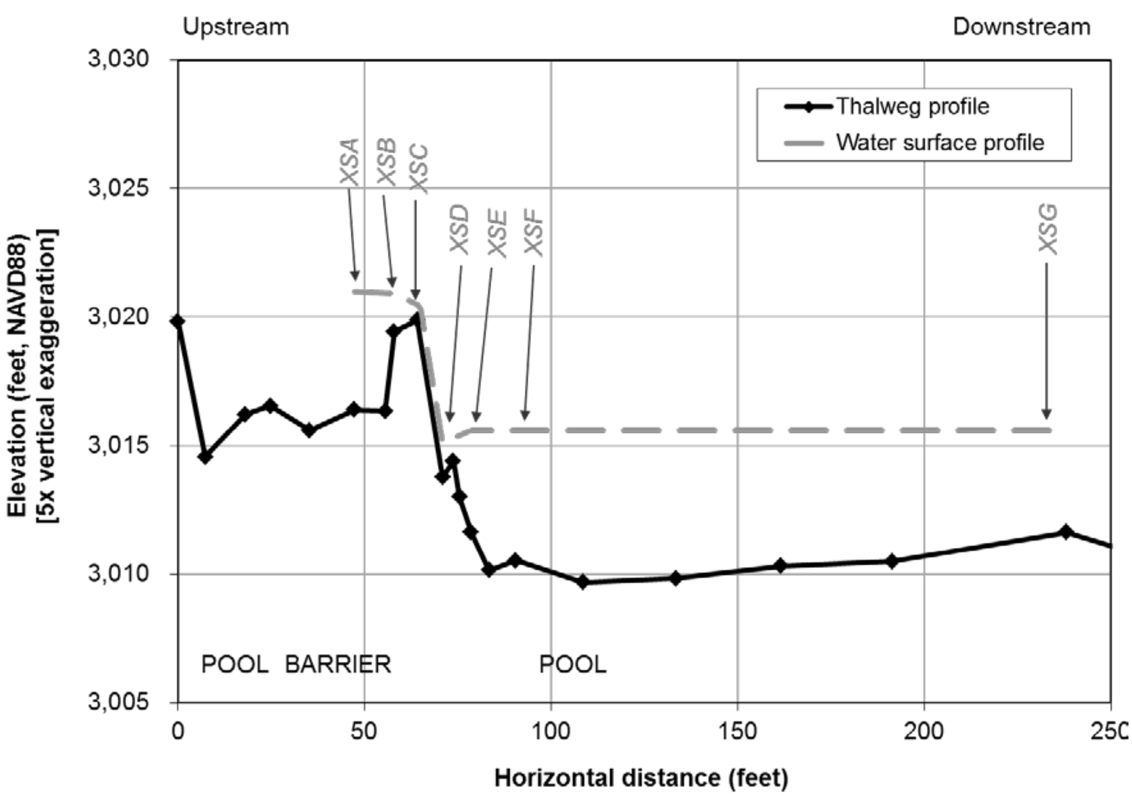

Figure 24. Longitudinal profile and cross-sections (XS) at locations A–G of WBFR-X 27.5, showing the waterfall barrier and holding pools

The waterfall surveyed at WBFR-X 27.5 was classified as migration barrier to both rainbow and brown trout, based on sizes, and therefore leaping ability, of fish observed during relicensing studies. Leaping abilities of both rainbow trout and brown trout limit passage at this location under the flows at which the survey was conducted (Table 3 and Figure 25). It is expected that passage ability would not improve with lower flows; however, as flows increase, the vertical drop (i.e., fish jumping height) has the potential for reduction, which may allow passage.

No analysis using FishXing software (USFS 2012) was conducted for this location due to the channel characteristics, which limit fish passage here strictly to leaping abilities.

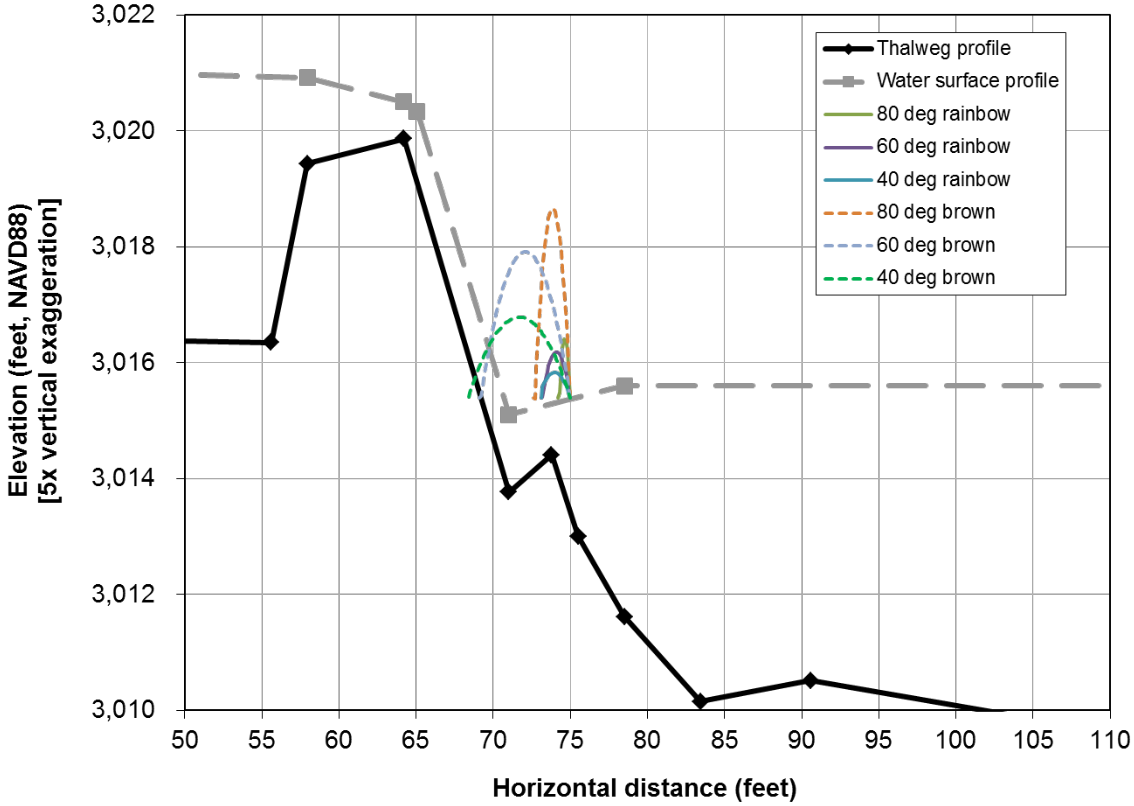

Figure 25. WBFR-X 27.5 migration barrier showing fish leaping capabilities at 80, 60, and 40 degree angles for rainbow trout and brown trout.

SECTION 4.0

Discussion

To help ensure fish passage connectivity within the river, the prescribed minimum instream flow releases below Hendricks Diversion (under the new License pursuant to USFS 4(e) Condition No. 18, Part 1, *Streamflow*) is 15 cfs year-round, with the exception of dry water-type years, where the minimum instream flow is 7 cfs between September and February. The primary purpose of adding a fish ladder at Hendricks Diversion Dam would be to provide resident fish access to thermal refuge in the upper watershed during dry years when water temperatures may be elevated in downstream areas as a result of decreased flows.[9] Additionally, FERC Draft License Article 415 and Forest Service 4(e) condition 19 (*Hendricks Diversion Fish Screen and Passage Plan*) includes measures for the successful year-round migration of trout between Hendricks Diversion Dam and Big Kimshew Creek in all water years, including dry years.[10] These measures may include the requirement for increased stream flows above those specified by Condition 18 Part 1 for the Lower West Branch Feather River below Hendricks Diversion Dam, and/or the installation of stream habitat enhancement structures.[11]

This barrier assessment was initially conducted to address USFS concerns that 7 cfs may not support passage through shallow sections of the stream reach between Hendricks Diversion and Big Kimshew Creek (FERC 2009).[12] Potential low-flow fish passage impediments downstream of the diversion were identified at two locations during this survey, which could limit access to Hendricks Diversion at flows between 7 cfs and 15 cfs. Field surveyors did not identify any low-flow passage impediments near, or along, PHABSIM transects in the assessment reach. The PHABSIM results for the site located near Retson Camp showed shallow-water conditions (depths of 4 inches or less) at 7 cfs; however, because no low-flow passage impediments were identified at these transects, it is presumed that the PHABSIM cross-section verticals did not capture the deepest pathways in spaces between large substrate particles (e.g., large cobble or boulders).

Of the two potential low-flow barriers identified, the first (WBFR-X 28.4) is located 0.8 miles downstream of Hendricks Diversion; the second (WBFR-X 27.4) is located 1.8 miles downstream of Hendricks Diversion and 0.1 miles downstream of the vertical barrier (WBFR-X 27.5). Both locations could potentially impede passage of trout within this corridor during releases of 7 cfs at Hendricks Diversion Dam; however, field verification during low flows (~7

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[9] FERC EA Section 5.4 referencing the June 29, 2009, section 10(j) meeting discussion

[10] The Draft License Articles are preliminary and the number was assigned based on the order listed in the FERC EA (2009). License conditions may change when the Final FERC License is accepted

[11] FERC EA Section 5.4 referencing the June 29, 2009, section 10(j) meeting discussion

[12] Flows downstream of the reach increase with perennial input from Big Kimshew Creek, which established the lower extent of the assessment reach

cfs) would be needed to confirm whether dry conditions would make these two potential barriers impassable to trout.

Minor modification of the stream channel to improve passage during dry water years is possible at potential low-flow barrier site WBFR-X 28.4; this abrupt high-gradient riffle is composed of large cobbles and small boulders, which may be manipulated by hand (or large pry bars) to provide continuous surface flow between 7 cfs and 15 cfs. If confirmed to be a passage barrier at 7 cfs, channel modifications at this site could increase the migration corridor by 0.9 miles during dry water years. In order to improve passage at the potential low-flow barrier site WBFR-X 27.4, mechanical work, such as the use of heavy machinery, explosives, or the construction of a fish ladder would be necessary; the channel is formed by a broad bedrock step, containing several large boulders, which cannot easily be manipulated. Also, because WBFR-X 27.4 is located downstream of vertical barrier WBFR-X 27.5, channel modifications to allow passage would not improve passage for fish to Hendricks Diversion Dam without additional significant channel modifications at site WBFR-X 27.5. The locations of both sites are inaccessible by heavy machinery and are located on privately owned lands.

Again, FERC Draft License Article 415 and USFS 4(e) Condition 19 require that the plan recommend measures to increase connectivity and year-round migration of trout, and include potentially increasing minimum instream stream flows below Hendricks Dam above those specified by Condition 18, Part 1.[13] Although the entire study reach was not assessed during the field visit, this assessment identified five barriers located between 1.7 and 4.8 miles downstream of Hendricks Diversion that would impede migration within the reach downstream of Hendricks Diversion at a wide range of velocities (including flows greater than 15 cfs). The first complete migration barrier (located 1.7 miles downstream of the diversion) consists of a 5-ft vertical drop that is not passable by rainbow or brown trout in the WBFR at flows less than 24 cfs; this barrier is expected to remain impassible at higher flows as well. The lowermost migration barrier (located 4.8 miles downstream of Hendricks diversion and 0.5 miles upstream of Big Kimshew Creek) contained three separate cascades, all of which were documented as passage barriers to rainbow and brown trout in the WBFR at flows equal to, or less than, 24 cfs (this barrier is also expected to remain impassible at higher flows).

With the exceptions of potential low-flow barriers WBFR-X 27.4 and WBFR-X 28.4, all of the barriers identified were formed from the natural morphology of the WBFR, and would impede passage at or above normal base flow. In order to increase fish passage throughout the entire assessment reach, mechanical work, such as the use of heavy machinery or explosives, would be necessary. In addition, this remote section of stream is located entirely within privately held lands, and access to the vertical or velocity barriers is very limited.

[13] The Draft License Articles are preliminary and the number was assigned based on the order listed in the FERC EA (2009). License conditions may change when the Final FERC License is accepted.

SECTION 5.0

References

Alexander, R.M. 1967. Functional design in fishes. Hutchinson and Company, London.

Clay, C.H. 1961. Design of fishways and other fish facilities. Department of Fisheries of Canada, Ottawa. Cat. No. Fs 31-1961/1.

FERC (Federal Energy Regulatory Commission). 2009. Final Environmental Assessment for new major hydropower license: DeSabla-Centerville Hydroelectric Project. FERC Project No. 803-087. Prepared by the Federal Energy Regulatory Commission, Office of Energy Projects, Division of Hydropower Licensing. Washington, DC. July.

Flosi, G., S. Downie, J. Hopelain, M. Bird, R. Coey, and B. Collins. 1998. California Salmonid Stream Habitat Restoration Manual. Prepared for the State of California Department of Fish and Game Inland Fisheries Division, Third Edition.

PG&E (Pacific Gas and Electric Company). 2007. DeSabla-Centerville Hydroelectric Project FERC Project No. 803, Updated Study Results and License Application Sections. Filed on December 31, 2007.

Powers, P. D., and J. F. Orsborn. 1985. Analysis of barriers to upstream fish migration, an investigation of the physical and biological conditions affecting fish passage success at culverts and waterfalls. Washington State University, Department of Civil Engineering, Albroook Hydraulics Lab, Pullman, WA.

Thomson, K. 1972. Determining Stream Flows for Fish Life. Presentation to Pacific Northwest Basin Commission Instream Flow Workshop, Vancouver, Washington. March 15–16.

USFS (United States Department of Agriculture, Forest Service). 2012. FishXing: U.S. Forest Services' Fish Crossing model, available at: http://stream.fs.fed.us/fishxing/.

APPENDIX A

Planform maps and cross sectional profiles of WBFR-X 24.4 and WBFR-X 27.5

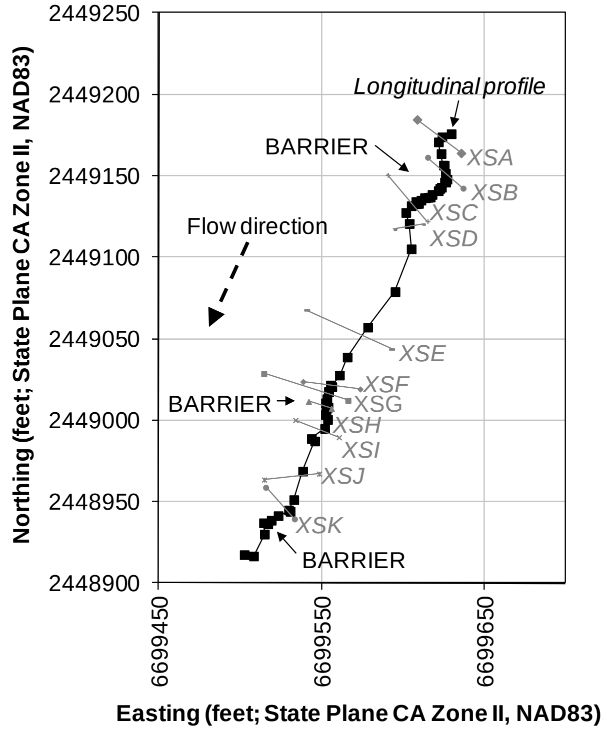


Figure A-1. Planform view of the longitudinal profile and cross-sections (XS) at locations A-K at WBFR-X 24.4, showing three cascade barriers

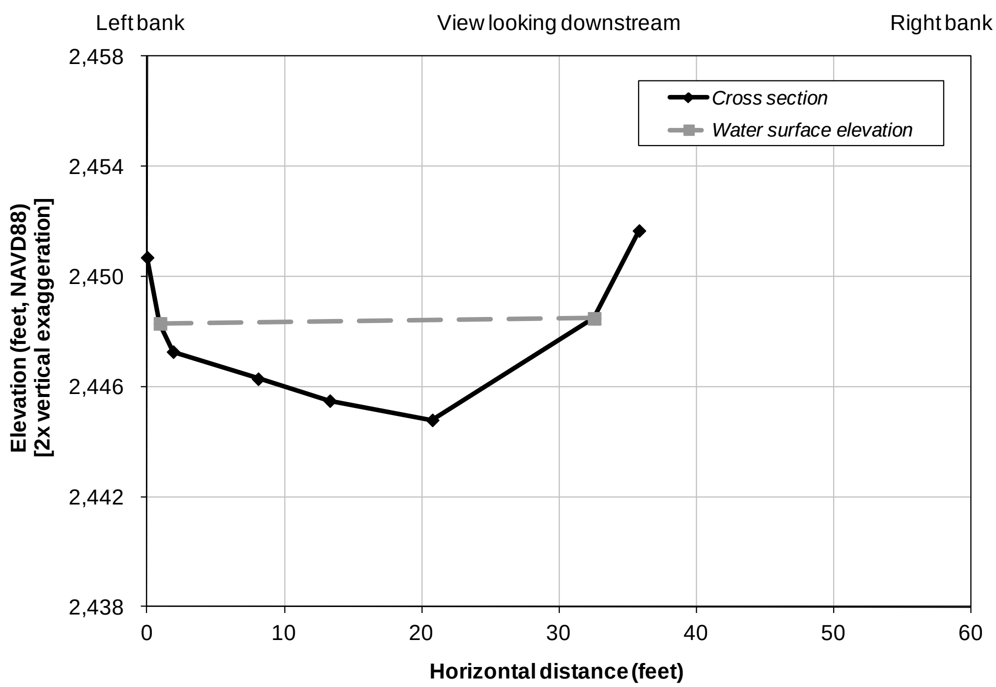


Figure A-2. Cross-section A of WBFR-X 24.4, located upstream of the upper cascade barrier

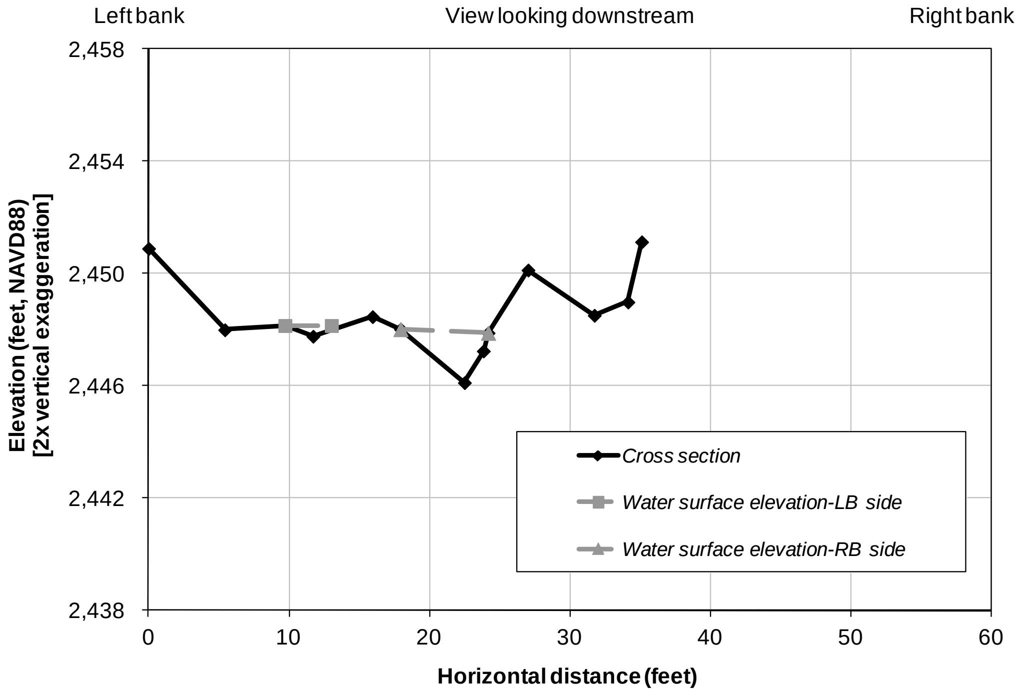


Figure A-3. Cross-section B of WBFR-X 24.4, located at the top of the upper cascade barrier

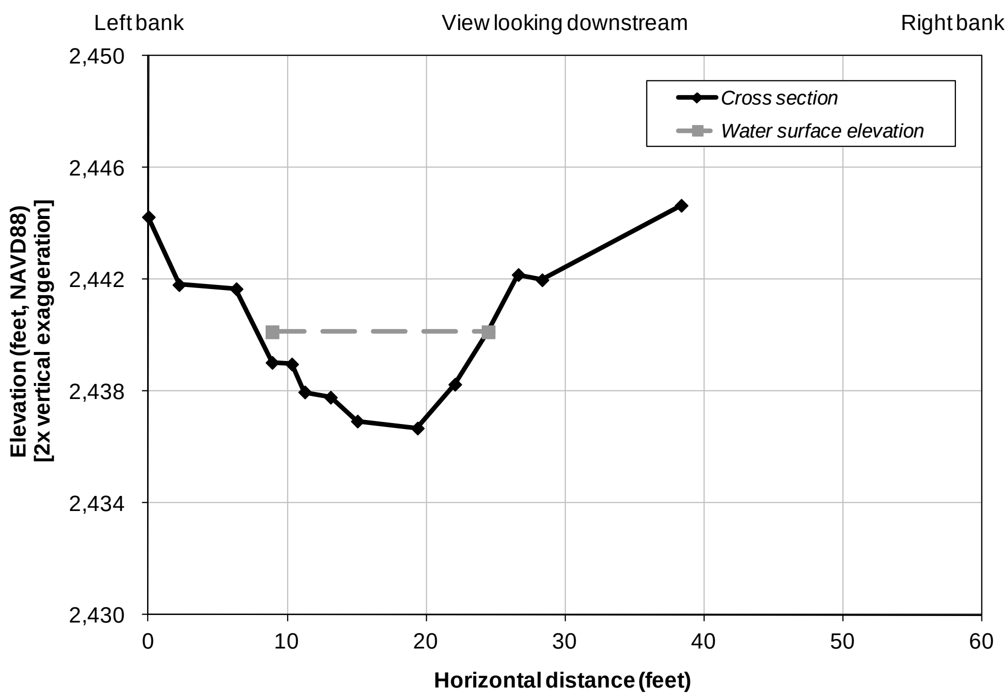


Figure A-4. Cross-section C of WBFR-X 24.4, located at the base of the upper cascade barrier

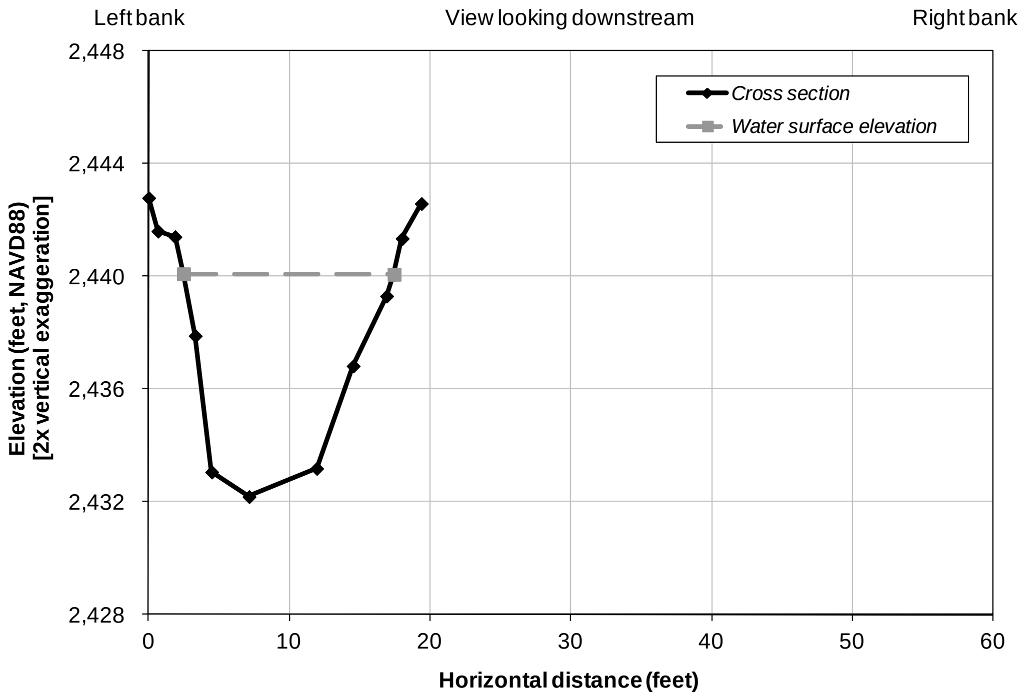


Figure A-5. Cross-section D of WBFR-X 24.4, located in the plunge pool below the upper cascade barrier

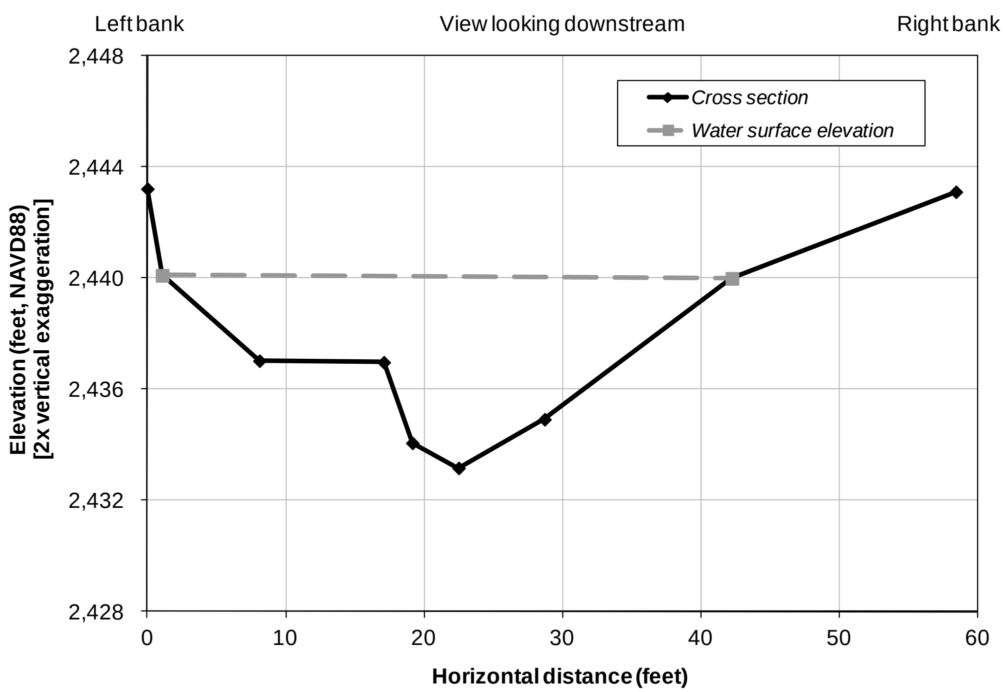


Figure A-6. Cross-section E of WBFR-X 24.4, located upstream of the middle cascade barrier

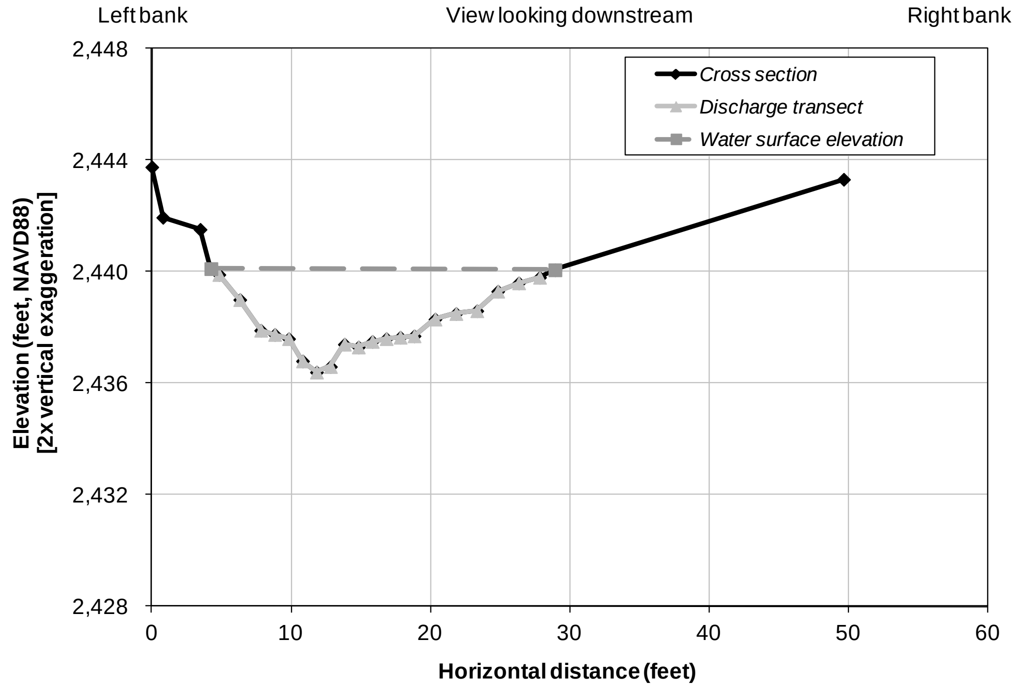


Figure A-7. Cross-section F of WBFR-X 24.4, located upstream of the middle cascade barrier and along the discharge-measurement transect

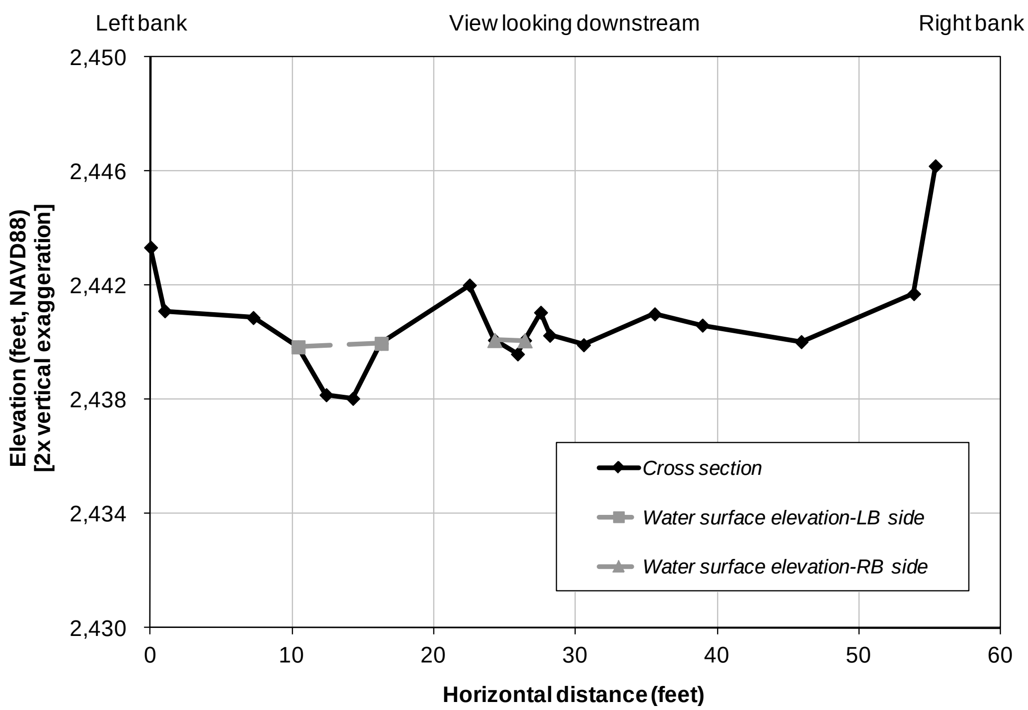


Figure A-8. Cross-section G of WBFR-X 24.4, located at the top of the middle cascade barrier

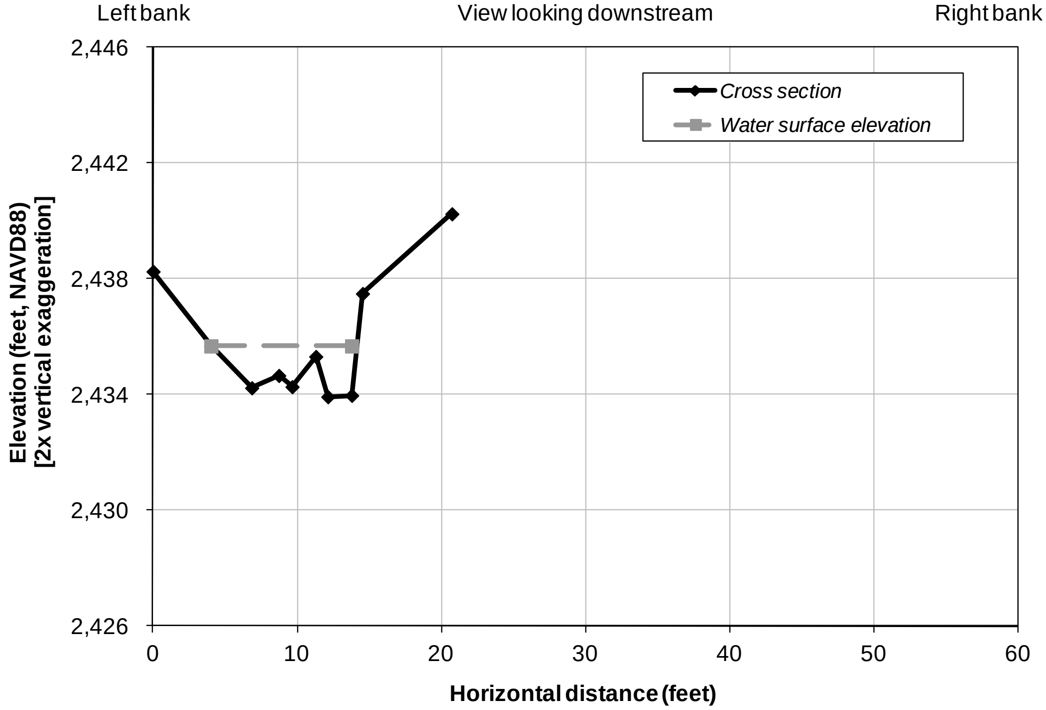


Figure A-9. Cross-section H of WBFR-X 24.4, located at the base of the middle cascade barrier

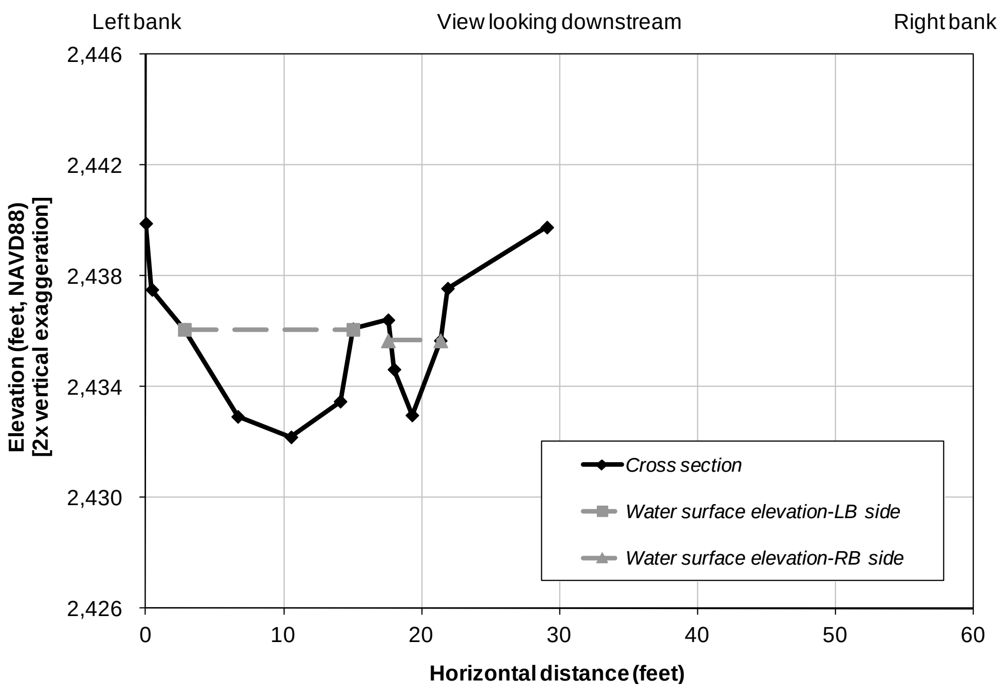


Figure A-10. Cross-section I of WBFR-X 24.4, located in the plunge pool below the middle cascade barrier

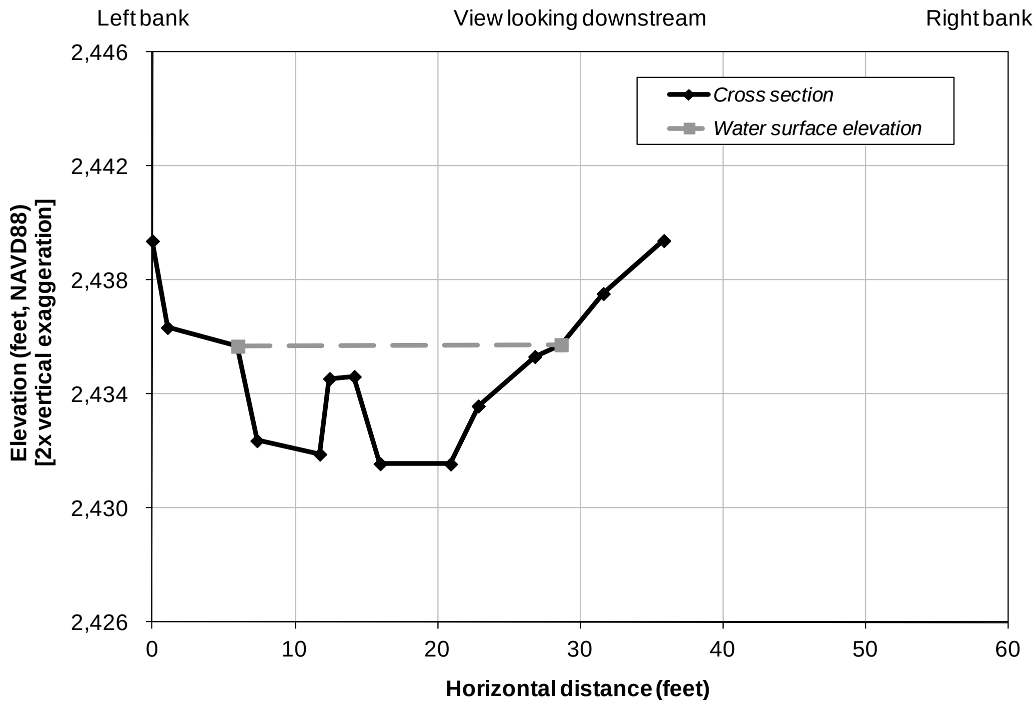


Figure A-11. Cross-section J of WBFR-X 24.4, located upstream of the lower cascade barrier

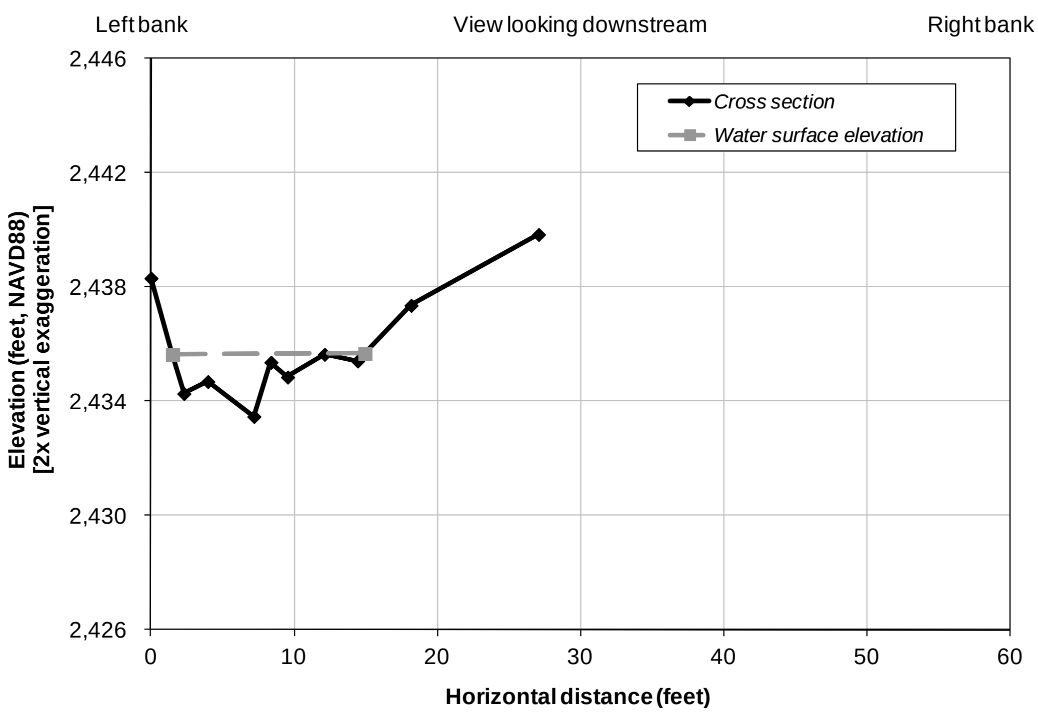


Figure A-12. Cross-section K of WBFR-X 24.4, located at the top of the lower cascade barrier

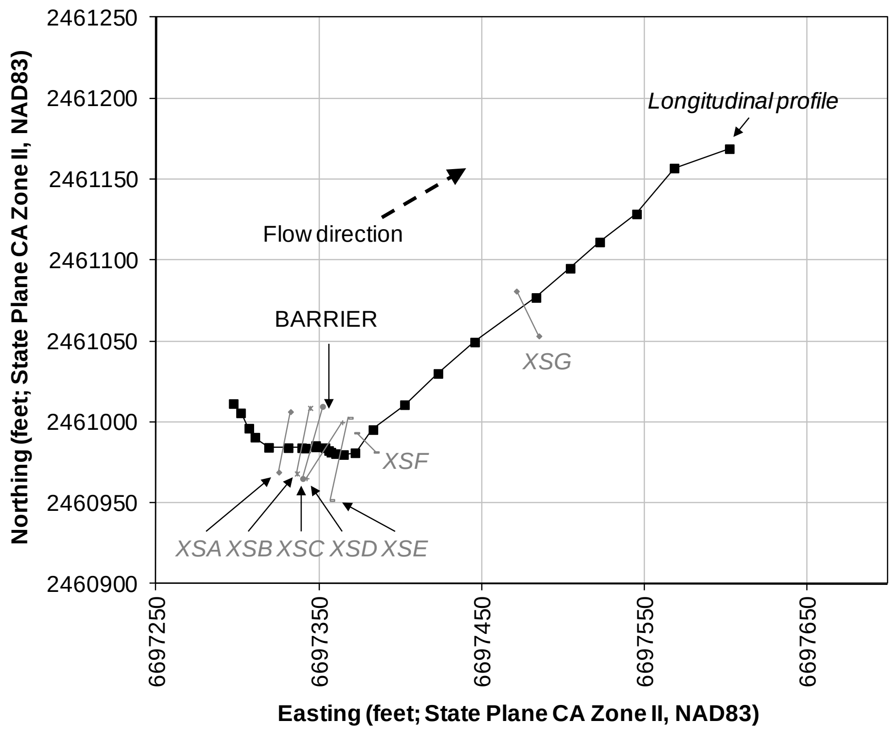


Figure A-13. Planform view of the longitudinal profile and cross-sections (XS) at locations A–G at WBFR-X 27.5, showing the barrier location

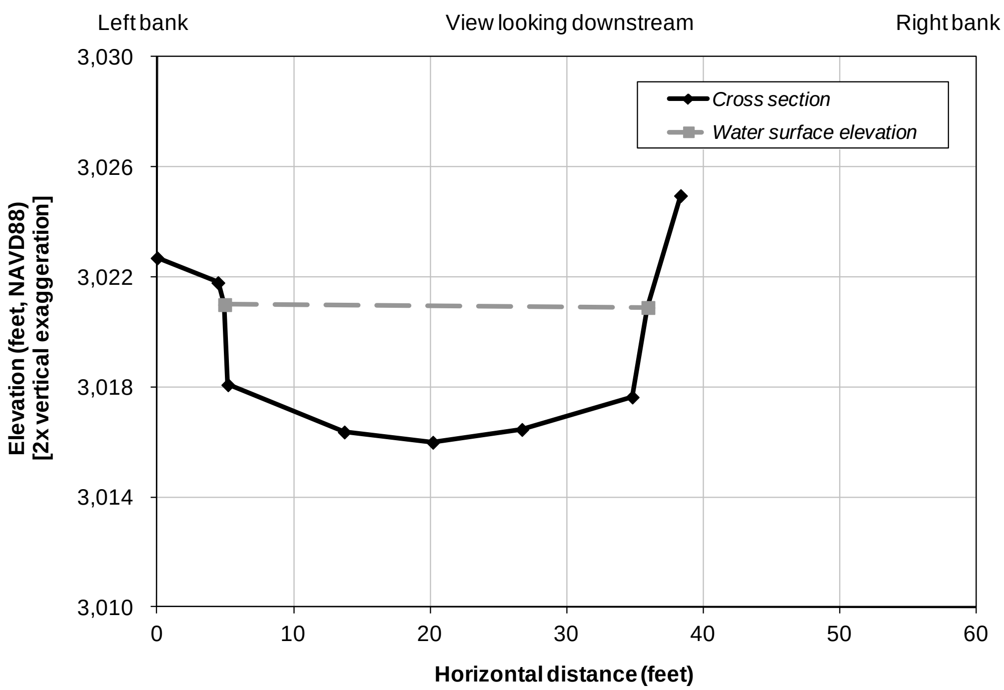


Figure A-14. Cross-section A of WBFR-X 27.5, located in the holding pool upstream of the waterfall barrier

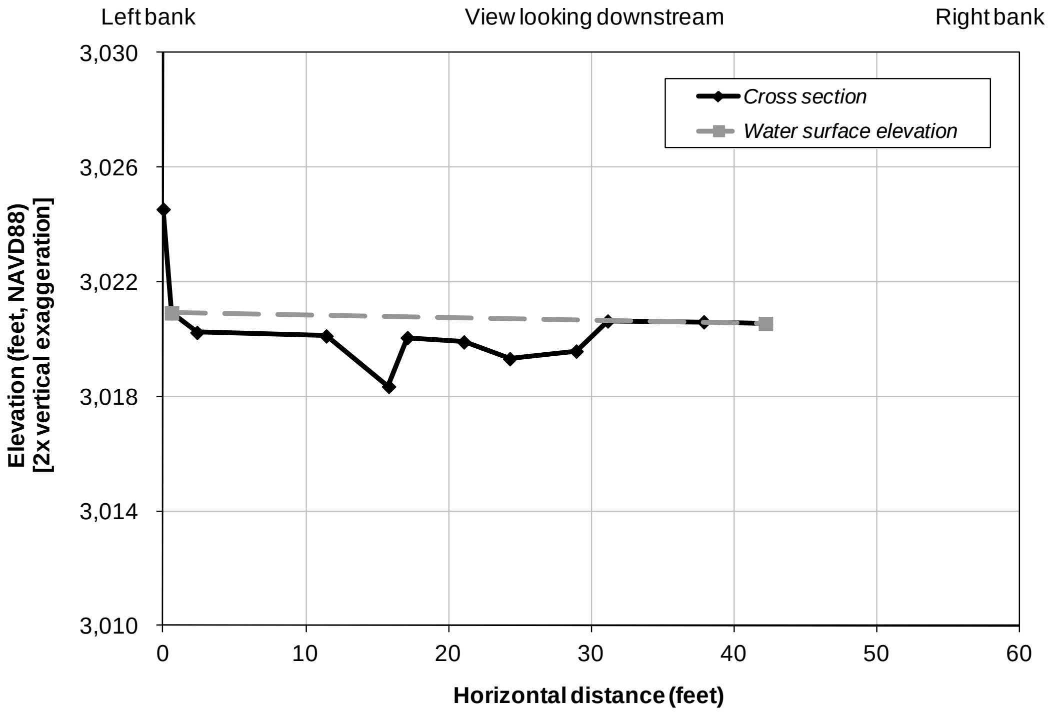


Figure A-15. Cross-section B of WBFR-X 27.5, located immediately upstream of the waterfall barrier

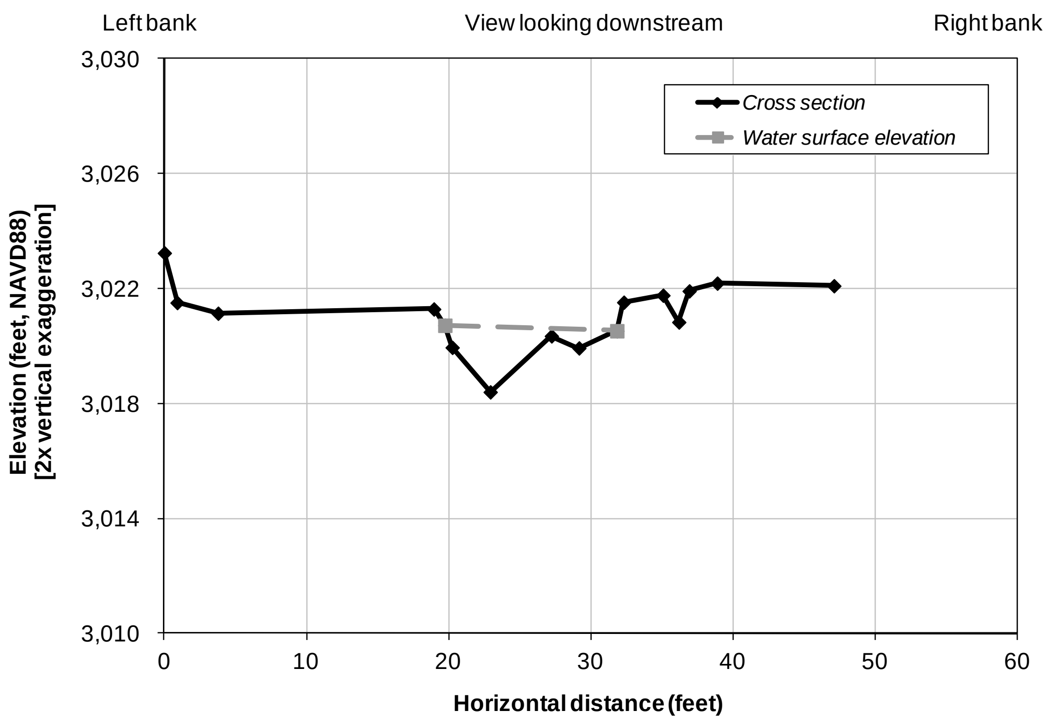


Figure A-16. Cross-section C of WBFR-X 27.5, located at the top of the waterfall barrier

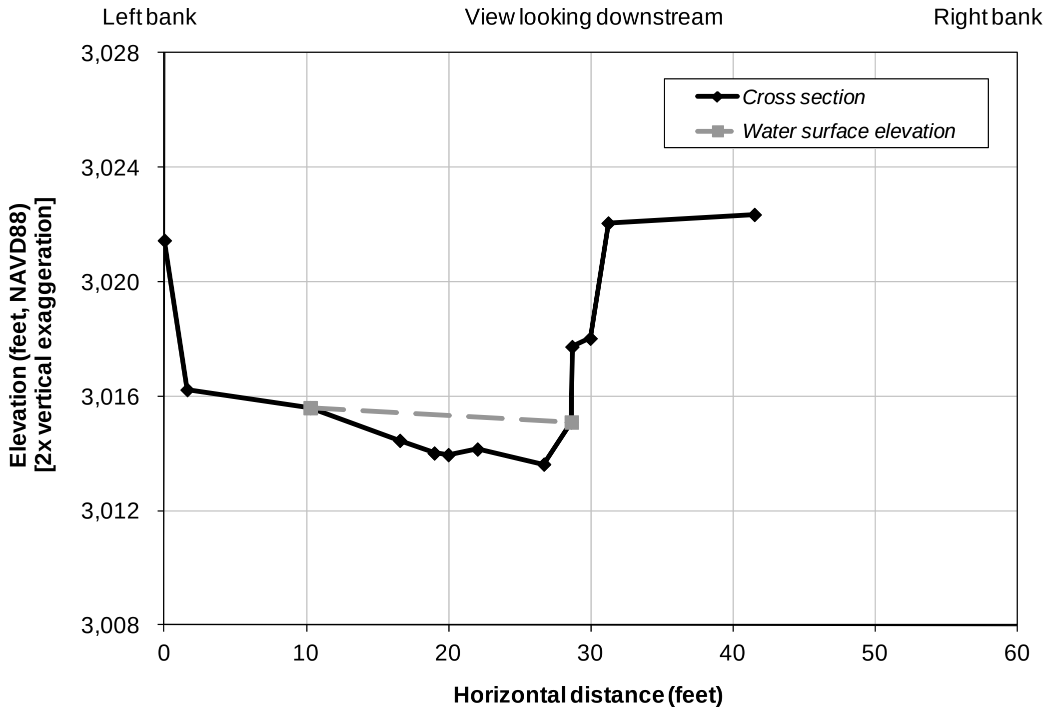


Figure A-17. Cross-section D of WBFR-X 27.5, located at the base of the waterfall barrier

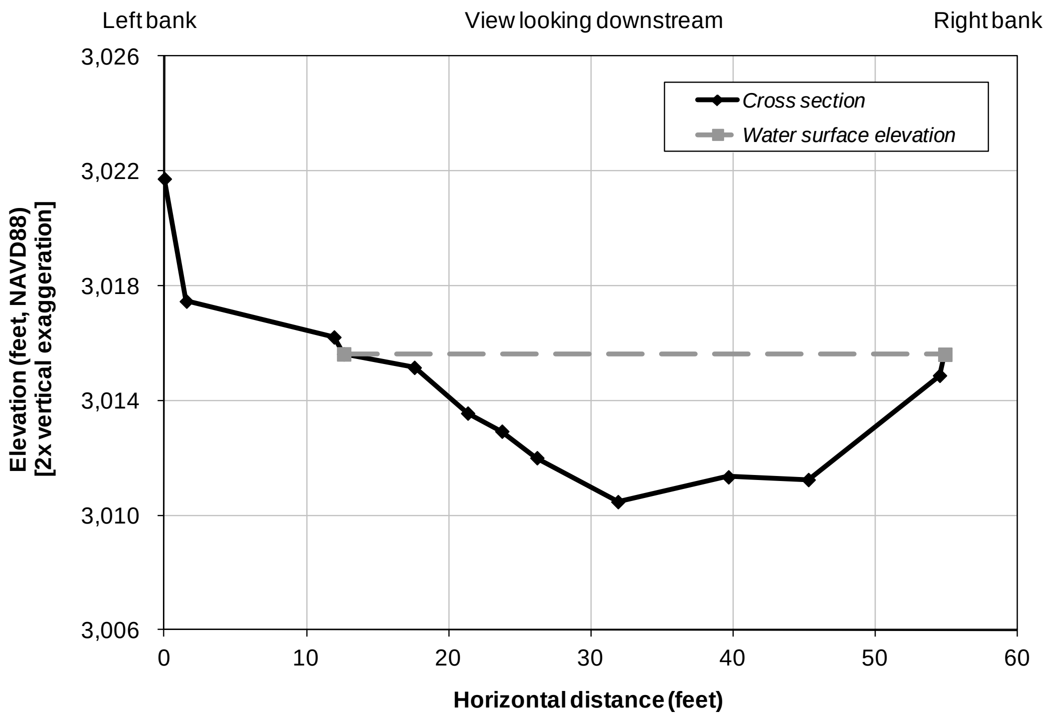


Figure A-18. Cross-section E of WBFR-X 27.5, located in the plunge pool immediately below the waterfall barrier

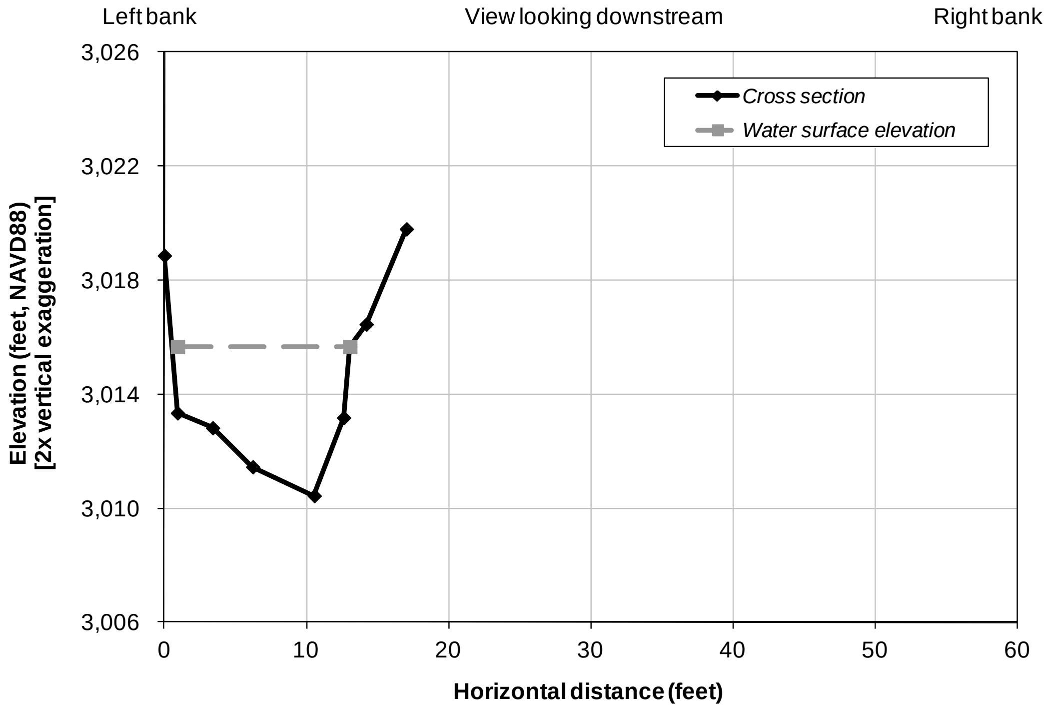


Figure A-19. Cross-section F of WBFR-X 27.5, located in the plunge pool below the waterfall barrier

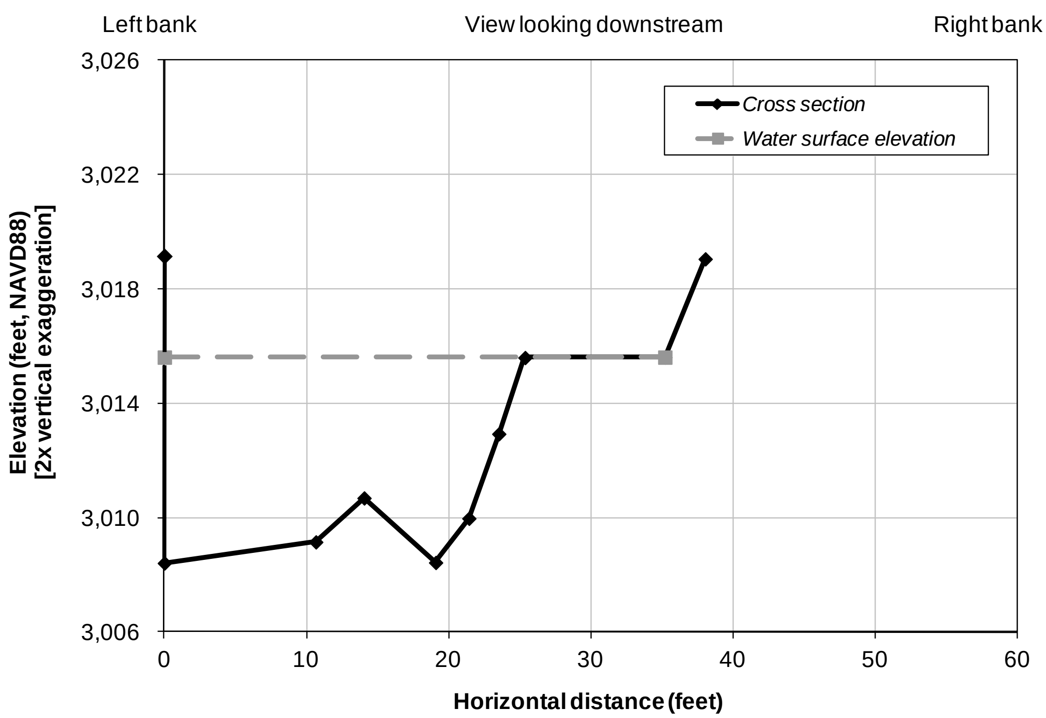
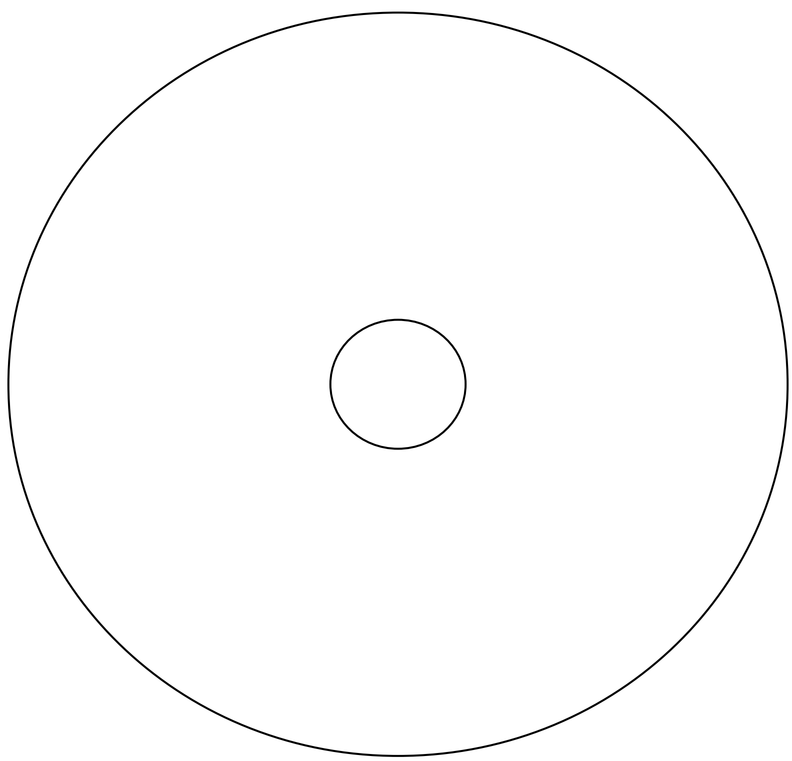


Figure A-20. Cross-section G of WBFR-X 27.5, located in the long pool far below the waterfall barrier

APPENDIX B

DVD containing: ArcGIS shapesfiles, *.kmz files, aerial and ground photographs



DVD CONTENTS

Contents	Description	File size (MB)	File type
Root folder			
BarriersSurvey_DataViewer.mxd	ArcGIS 10.0 ArcMap Document displaying the results of the October 2011 barriers survey.	2.5	ArcGIS ArcMap Document
SurveyData_Oct2011_v2.gdb	ESRI File Geodatabase housing all data collected in the field and displayed in the "BarriersSurvey_DataViewer.mxd" file and in the "DeSabla WBFR Barrier Survey 2011.kmz" file. In addition, this GeoDatabase contains several relationship tables used by the GeoDatabase to display the barrier photos and related information. Do not alter or delete the contents of this GeoDatabase.	91.4	ESRI File Geodatabase
basedata folder			
Barriers_Photo_Catalog.lyr	ArcGIS Layer showing the Barrier Photo Catalog data using the MXD's symbology.	< 1.0	ArcGIS Layer
Rivers.shp	ESRI Shapefile showing selected rivers\streams around the Project area.	< 1.0	ESRI Shapefile
World_Imagery.lyr	Dynamic ArcGIS Layer/Map Service. This map service presents high-resolution imagery for the United States. Use requires an internet connection and ESRI ArcMap software.	< 1.0	ArcGIS Layer
graphs folder			
"upper" and "lower" folders	These folders contain a collection of graphs representing the topographic cross-sections and longitudinal profiles surveyed at each topographic survey sites: WBFR-X 24.4 and 27.5.	< 1.0	GIF images
graphsGE folder			
"upper" and "lower" folders	These are the same graphs as above at a lower resolution. Created for the GeoDatabase and Google Earth file (.kmz).	< 1.0	GIF images

Contents	Description	File size (MB)	File type
imagery folder			
2005 GeoTiffs	1-ft resolution imagery of the barrier sites (2005 GeoTiffs) provided by PG&E. Do not alter the contents of this folder.	981.0	GeoTiffs (and ancillary files)
info folder			
GeoDatabase dependent files	GeoDatabase dependent files. Do not delete this folder or modify its contents.	< 1.0	Geo Database dependent files
kml folder			
DeSabla WBFR Barrier Survey 2011.kmz	Google Earth version of the "BarriersSurvey_DataViewer.mxd" file. This file was created to allow access to the GeoDatabase content outside ESRI ArcGIS software. It requires the latest version of Google Earth to be installed in your computer; available for download www.google.com/earth/index.html	28.1	Google Earth File
layers folder			
Barriers_Photo_Catalog.lyr	ArcGIS Layer showing the Barrier Photo Catalog data using the MXD's symbology. Note: layers have absolute paths to their sources. User will need to update the source to match local directory settings	< 1.0	ArcGIS Layer
Cross Sections Data Points (lower).lyr	ArcGIS Layer showing the "lower" Cross Sections Data Points (WBFR-X 24.4) using the MXD's symbology. Note: layers have absolute paths to their sources. User will need to update the source to match local directory settings	< 1.0	ArcGIS Layer
Cross Sections Data Points (upper).lyr	ArcGIS Layer showing the "upper" Cross Sections Data Points (WBFR-X 27.5) using the MXD's symbology. Note: layers have absolute paths to their sources. User will need to update the source to match local directory settings	< 1.0	ArcGIS Layer

Contents	Description	File size (MB)	File type
photos folder			
Original full resolution barrier photos	Original photographs showing the surveyed barriers.	93.8	JPG images
photosGE folder			
Barrier photos (lower resolution version)	Lower resolution versions of the barrier photographs.	22.4	JPG images
survey_shapefiles folder			
BarrierSurveyPhotos_vAll.shp	An ESRI Shapefile showing the location of all the surveyed barrier photographs.	< 1.0	ESRI Shapefile

Summary of Butte Creek Fish Barriers near
Lower Centerville Diversion Dam (DeSabla-Centerville Project, FERC 803)

Gene Geary
Senior Aquatic Biologist Pacific Gas and Electric Company
May 10, 2005

BACKGROUND

This summary has been prepared in response to a request by NOAA Fisheries Service during a conference call on April 27, 2005. At that time, representatives of NOAA Fisheries Service requested that Pacific Gas and Electric Company (PG&E) summarize the information that has been developed regarding natural barriers immediately upstream of the Lower Centerville Diversion Dam (LCDD), to help them understand why both the California Department of Fish and Game (CDFG) and PG&E are convinced that anadromous fish never were able to pass upstream of the vicinity of LCDD. This question arose in the context of determining if anadromous fish needed to be considered as target species by any instream flow study conducted for Butte Creek upstream of LCDD as part of the DeSabla-Centerville Project relicensing effort.

SUMMARY OF EXISTING REPORTS

The highest stream gradient in Butte Creek is found in the 1.4 miles immediately upstream of LCDD; in this section, Butte Creek has a gradient of 7.3 % (386 feet per mile) (USGS *Paradise West* and *Cohasset* 7.5' Quadrangles). Multiple natural barriers to fish passage have been documented in and above this section. Holtgrieve and Holtgrieve (1995) were unable to find any historical information suggesting that salmon were ever present in Butte Creek above LCDD, and they identified 35 potential barriers to fish passage between LCDD and Butte Creek Diversion Dam, eight of which were mapped within 0.8 miles of LCDD. They noted that the most difficult barriers occurred in the 3.5 miles upstream of LCDD, and recommended that "*Persons particularly expert in the capabilities of migrating salmon should evaluate barriers in this segment*." A second barrier survey was conducted in 1996 by Johnson and Kier (1998) to explore the potential for expanding spring-run Chinook habitat opportunities. Johnson and Kier identified 77 natural barriers between LCDD and Butte Creek Diversion Dam, with 22 barriers identified in the 1.4 miles upstream of LCDD. The largest barrier identified by Johnson and Kier was 35 high and occurred 0.58 miles upstream of LCDD. Johnson and Kier also identified five other barriers downstream of this point. Two were major barriers nearer to LCDD (a 16.5-foot compound barrier 0.54 miles above LCDD, and a 17-foot barrier 0.45 miles above LCDD), while the other 3 were smaller, between 6 and 7.4 feet high.

Both Holtgrieve and Holtgrieve (1995) and Johnson and Kier (1998) suggested that the natural barriers in Butte Creek could be modified to allow upstream access by salmon. In April, 1997 a proposal was initiated by the Institute for Fisheries Resources to open Butte Creek Canyon to salmon and steelhead production. According to Watanabe (2000), an analysis was performed and revisions to the project proposal were suggested in a paper prepared by a CDFG biologist. This analysis stated that there were significant environmental and engineering issues that needed to be addressed before developing a restoration plan for Upper Butte Creek. Subsequently, five barriers (Quartz Bowl, LCDD, and three major barriers upstream of LCDD) were briefly examined on July 12, 1999 by representatives from CDFG, NOAA Fisheries Service, the Bureau of Reclamation (BOR), and PG&E. The goal of the group was to see several barriers on Butte Creek to get an idea of the fish passage problems that exist and to begin doing a technical assessment of fish passage conditions. The data collected by this team were evaluated by Watanabe (2000). Watanabe's conclusions about the ease of modifying natural barriers to allow fish passage contradicted those of Holtgrieve and Holtgrieve (1995) and Johnson and Kier (1998). She concluded that while fish passage design criteria are frequently "stretched" in designing passage over natural barriers, this is usually in the context of a single barrier. However, she concluded that "*The significant difference at Butte Creek is that there is not just one barrier where the standards could be stretched, but 11 or more miles of potentially up to 77+ barriers. The goal should be to provide unimpeded passage at each barrier to allow the spring-run salmon to reach the Upper Butte Creek holding area in good condition so they can successfully hold over the summer and spawn in the fall. A second goal should be to avoid stranding salmon and steelhead in this stretch of river when the flow changes, where they may not be able access suitable holding pools or spawning sites. If passage were provided through this 11+ mile reach, adherence to the criteria listed for manmade structures in an effort to provide unimpeded upstream passage is required.*" Watanabe identified a series of detailed information requirements that would have to be met in order to design and estimate costs for a passage project such as was proposed by the Institute for Fisheries Resources in 1997. No further studies have been undertaken by the resource agencies after the initial survey in 1999 (Paul Ward, CDFG personal communication). At a Butte Creek Science Workshop in Chico on April 8, 2004, George Heise (CDFG's senior engineer) concluded that upper Butte Creek above LCCD did not make a good candidate for fish passage improvement because of the number of migration barriers and the overall high gradient of the channel.

In addition to the natural migration barriers upstream of LCDD, Watanabe documented an 11.1-foot high barrier at Quartz Bowl, approximately one mile downstream of LCDD, and an 11.4-foot bedrock cascade/falls that forms the foundation for the LCDD. Holtgrieve and Holtgrieve (1995) quoted one source that the Quartz bowl was a total barrier to salmon migration until the barrier was modified by blasting in the 1930s, which allowed some fish passage. Currently a few salmon are able to pass the Quartz Bowl barrier in very wet springs. In 1995 and 2003, 25 and 6 spring-run Chinook salmon, respectively, were observed between the Quartz Bowl barrier and the LCDD (P. Ward Personal communication), which equaled 0.3% and 0.1%, respectively of the observed population of spring-run Chinook in each year (7,500 fish in 1995 and 4,398 fish in 2003): no fish were observed in 1998, 2000 or 2004 when there were also specific surveys of this area completed). The natural bedrock falls at the site of LCDD is also likely to have been a significant impediment to anadromous fish migration before the dam was constructed. Yoshiyama et al (2001) concluded that historically the upstream limit of salmon migration on Butte Creek was the present vicinity of LCDD. NOAA Fisheries Service (Schick et al. 2005), followed this conclusion, and identified no change in current available habitat for Central Valley spring-run Chinook salmon in Butte Creek compared with historical conditions, and did not identify LCDD as a "keystone dam" restricting salmon migration. Schick et al. also developed a map inferred from Yoshiyama et al (2001) that suggests steelhead might have ascended further into Butte Creek. Yoshiyama et al. cites Flint and Meyer (1977), stating that steelhead are believed to have ascended as far upstream as Butte Meadows. In turn this reference in Flint and Meyer is based on an uncorroborated personal communication: "*Both species originally migrated far into the canyon – some steelhead probably going as far as Butte Meadows (R. Hallock, Citizens Advisory Committee, 1971, personal communication).*" No other reference has been located that would verify this remark, despite considerable research (Holtgrieve and Holtgrieve 1995, P. Ward, CDFG, Personal Communication). Given the best available technical information currently available on the barriers mapped in Butte Creek (see discussion below), the speculative remark repeated by Flint and Meyer (1977) was almost certainly in error.

BARRIER DETAILS

Butte Creek barrier information was presented in slightly different forms by Holtgrieve and Holtgrieve (1995), Johnson and Kier (1998), and Watanabe (2000). Holtgrieve and Holtgrieve noted barrier locations on USGS quads, with symbols noting barriers, but provided no information on individual barrier heights (Figure 1). Johnson and Kier identified barrier locations and characteristics relative to distance downstream of Butte Creek Diversion Dam (a.k.a. Butte Head Dam) (Table 1), but did not plot barrier locations on a map. Watanabe reported barrier measurements from the Quartz Bowl Barrier downstream of LCDD to the third major barrier upstream of LCDD (Table 2). Watanabe's draft report did not identify the specific locations of the barriers upstream of LCDD, but Paul Ward (CDFG) has identified the general locations of these barriers, which are noted on Figure 1. The distance of these barriers upstream of LCDD was estimated to the nearest 0.1 mile by comparing stream locations to the river mile designations in PG&E (2004) (Appendix D, map 8 of 11). Of the three Butte Creek barrier surveys, the most accurate and detailed barrier measurements are those in Watanabe (2000). These measurements will be used in the remainder of this discussion.

Evans and Johnston (1980) suggested that natural bedrock falls with a vertical drop of greater than 6 feet should be considered to be a total barrier for salmon and steelhead without further study. However, a detailed review by Powers and Orsborn (1985), concluded that falls where the change in water surface elevation is in excess of 11 feet can be considered a total barrier to all species of Pacific salmon and steelhead. The validity of Powers and Orsborn's conclusion for Butte Creek is confirmed by the fact that the Quartz Bowl barrier height is right at the criteria (measured at 11.1 feet) and is a confirmed barrier to salmon migration for all but a few fish in the wettest years, and that prior to blasting in the 1930s it was reportedly not passable at all. At 11.4 feet high, the natural barrier that forms the foundation for LCDD is also right at Powers and Orsborn total barrier criteria. Of the barriers summarized in Table 2 that occur upstream of LCDD, all three significantly exceed the Powers and Orsborn criteria for a total passage barrier, ranging in height from 13 feet to 23.8 feet with the first of these barriers (14.4

feet high) located only 0.3 miles upstream of LCDD. The furthest upstream (and most difficult) barrier reported by Watanabe is approximately 0.8 miles upstream of LCDD.

Spawning habitat is extremely limited in the vicinity of LCDD. Johnson and Kier (1998) reported 400 square feet of gravel (37 square meters) in the 0.54-mile section upstream of LCDD; this section extends up to the second total barrier (i.e. >11-foot) identified on that survey (this corresponds to the second total barrier identified by Watanabe 2000). Assuming a recommended Chinook salmon spawning area of 15.5 square meters[1] this amount of gravel could be sufficient for two pairs of spawning Chinook. Assuming an average steelhead redd size of 4.4 square meters (Bjorn and Rieser 1991), there may be enough spawning gravel in this section for eight pairs of steelhead.

CONCLUSIONS

Key conclusions from the above compilation can be summarized as follows:

- Holtgrieve and Holtgrieve (1995) found no historical records to indicate the pre-Project occurrence of salmon in Butte Creek upstream of the site of LCDD. One source indicated that all salmon were blocked at the Quartz Bowl barrier one mile downstream of LCDD until blasting modified this barrier sometime in the 1930s.
- The Quartz Bowl barrier was measured by resource agency engineers as 11.1 feet high. This is just at the 11-foot criteria proposed by Powers and Orsborn (1985) to delineate a total barrier for all species of Pacific salmon and steelhead. Observations of spring-run Chinook salmon above the Quartz Bowl barrier have confirmed this site to be a barrier to salmon migration for all but a few fish in only the wettest years. A slightly higher natural barrier (11.4 feet) forms the foundation of LCDD. Prior to the construction of LCDD, this barrier was arguably a total barrier to anadromous fish in most years.
- There are at least three locations from 0.3 - 0.8 miles upstream of LCDD with natural barriers significantly higher than 11 feet. Based on the Orsborn and Powers (1985) criteria, these locations can be considered to be total barriers to potential salmon and steelhead passage without further analysis.
- Yoshiyama et al (2001) and identified no change in current available habitat for Central Valley spring-run Chinook salmon in Butte Creek compared with historical conditions. Schick et al. (2005) did not identify LCDD as a "keystone dam" restricting salmon migration. Yoshiyama et al (2001) relied on a speculative remark repeated by Flint and Meyer (1977) to suggest that steelhead may once have migrated as far upstream as Butte Meadows. The subsequent surveys of barriers in Butte Creek prove that the communication cited by Flint and Meyer (1977) was in error.
- The quantity of spawning gravel reported by Johnson and Kier (1998) to be present between LCDD and the total barriers upstream, is adequate to support two pairs of spawning salmon and eight pairs of spawning steelhead.

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[1] An average of recommendations from Cramer and Hammack (1952), M. Gard (personal communication as cited in Ward and Kier 1999), and Needham et al. (1941)

From this evidence, it is clear that, before the development of the DeSabla-Centerville Project, the migration of both salmon and steelhead was blocked somewhere between one mile below and 0.3 miles above the current site of the LCDD. Therefore, it would be inappropriate to consider steelhead or salmon as a target species in an instream flow study upstream of LCDD.

REFERENCES CITED

Bjornn, T. C., and D. W. Reiser. 1991. Habitat requirements of salmonids in streams. *In* Influences of Forest and Rangeland Management on Salmonid Fishes and Their Habitats. W.R. Meehan ed. American Fisheries Society Special Publication 19:83-138.

California Department of Fish and Game (CDFG). 2003. Butte Creek spring-run Chinook salmon summary, September 24, 2003. [Informational handout distributed at Spring-run Chinook Workgroup meeting, September 24, 2003.] Chico, CA.

Cramer, F. K. and D. F. Hammack. 1952. Salmon research at Deer Creek, California. U. S. Fish and Wildlife Service Special Science Report Fisheries 67. 16 pp.

Evans, W.A. and B. Johnston (1980). Fish migration and fish passage. USDA Forest Service, EM-7100-12, Washington D.C.

Flint, R. A. and F. A. Meyer. 1977. The DeSabla-Centerville Project (FERC No. 803) and its impact on fish and wildlife. California Department of Fish and Game, Inland Fisheries.

Holtgrieve, D. G. and G. W. Holtgrieve. 1995. Physical stream survey - upper Butte Creek, Butte County, California. Department of Geography and Planning, California State University Chico, California.

Johnson, J. L. and W. M. Kier. 1998. A preliminary assessment of the salmon habitat potential of Butte Creek, a tributary of the Sacramento River, between the Butte Head Dam and Centerville Diversion Dam, Butte County, California. William M. Kier Associates, Sausalito, CA.

Needham, P.R., O.S. Smith, and H.A. Hanson. 1941. Salmon salvage problems in relation to Shasta Dam, California, and notes on the biology of the Sacramento River salmon. Trans. Amer. Fish. Soc. 70 (1940): 55-69.

Pacific Gas and Electric Company. 2004. DeSabla-Centerville Hydroelectric Project. FERC Project No. 803. Pre-application document. Volume 1: Public information.

Powers, P.D. and J.F. Orsborn. 1985. Analysis of barriers to upstream fish migration. An investigation of the physical and biological conditions affecting fish passage success at culverts and waterfalls. Part 4 of 4 of a BPA fisheries project on the development of new concepts in fishladder design. Contract DE-A179-82BP36523. Project No. 82-14.

Schick, R.S., A.L. Edsall, and S.T. Lindley. 2005. Historical and current distribution of pacific salmonids in the Central Valley, CA. NOAA Technical Memorandum NMFS. NOAA-TM-NMFS-SWFSC-369. February 2005.

Watanabe, C. 2000. Preliminary engineering requirements for fish passage on Upper Butte Creek: An assessment of the natural barriers-DRAFT. California Department of Fish and Game.

Ward, M. B. and W. M. Kier. 1999. Battle Creek salmon and steelhead restoration plan. Kier Associates, Sausalito, Ca.

Yoshiyama, R. M., E. R. Gerstung, F. W. Fisher, and P. B. Moyle. Historic and present distribution of Chinook salmon in the Central Valley drainage of California. In: R.L. Brown, editor, Fish Bulletin 179: Contributions to the biology of Central Valley salmonids, Volume 1, pages 71-176. California Department of Fish and Game, Sacramento, CA 2001.

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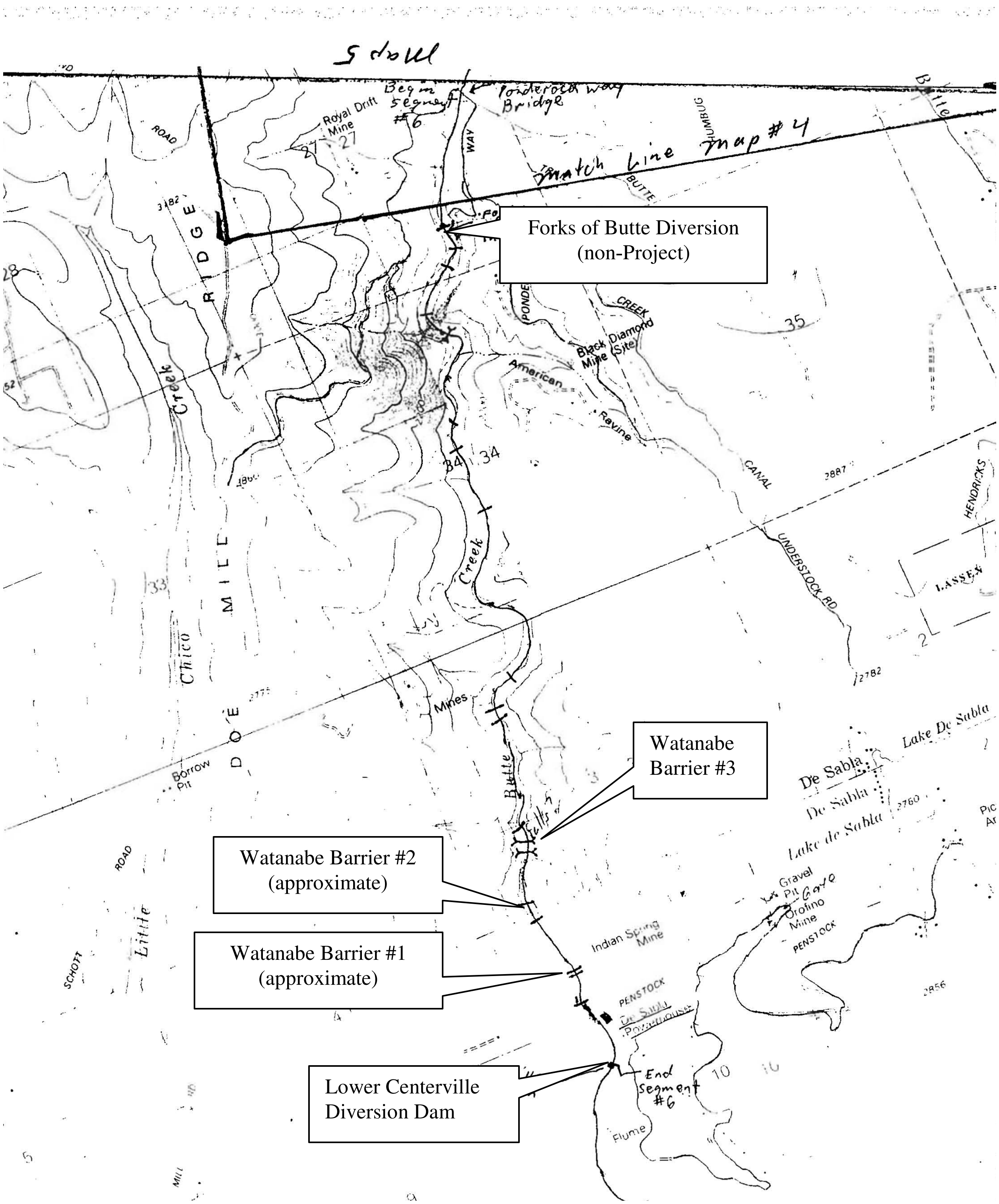


Figure 1. Holtgrieve and Holtgrieve (1995) Map of Fish Migration Barriers near LCDD. (Annotations of Watanabe (2000) barrier locations from P. Ward, CDFG)

Table 1. Migration Barriers within 1 Mile upstream of Lower Centerville Diversion Dam (LCDD) from Johnson and Kier (1998)*

Barrier Type	Distance d/s of Butte Creek Diversion (miles)	Distance from LCDD (miles) (Assuming LCDD at mile 11)	Vertical Ht. (ft)	Horizontal Distance (ft)	Plunge Pool Depth (ft)
[LCDD]	[11**]				
Single Wfall (1st Barrier above LCDD)	10.76	0.24 s	7.0	20.0	2.0
Single Waterfall	10.74	0.26	6.0	15.0	3.0
Single Waterfall	10.55	0.45	17.0	25.0	8.0
Single Waterfall	10.48	0.52	6.0	8.0	4.0
Single Waterfall	10.46	0.54	11.0	15.0	0.0
Single Waterfall	10.46	0.54	3 X 5.5	10.0	0.0
Single Waterfall	10.43	0.57	7.4	10.0	2.0
Single Waterfall	10.42	0.58	35.0	60.0	8.0
Single Waterfall	10.40	0.60	9.0	35.0	0.0
Single Waterfall	10.38	0.62	8.0	25.0	4.0
Single Waterfall	10.36	0.64	6.5	2.0	4.0
Single Waterfall	10.36	0.64	6.5	2.0	4.0
Single Waterfall	10.17	0.83	12.6	25.0	13.5
Single Waterfall	10.10	0.90	8.0	10.0	5.0
Single Waterfall	10.04	0.96	12.0	30.0	3.0
Single Waterfall	10.03	0.97	10.0	25.0	3.5

***Barrier Data extracted from Johnson and Kier (1998) on migration barriers within 1 mile of LCDD**

**** Johnson and Kier (1998) refer to LCDD as eleven miles downstream of the Butte Creek Diversion Dam, but did not specifically report the distance in their barrier location table.**

Table 2. Summary of Upper Butte Creek Field Trip Barrier Notes from Watanabe (2000)

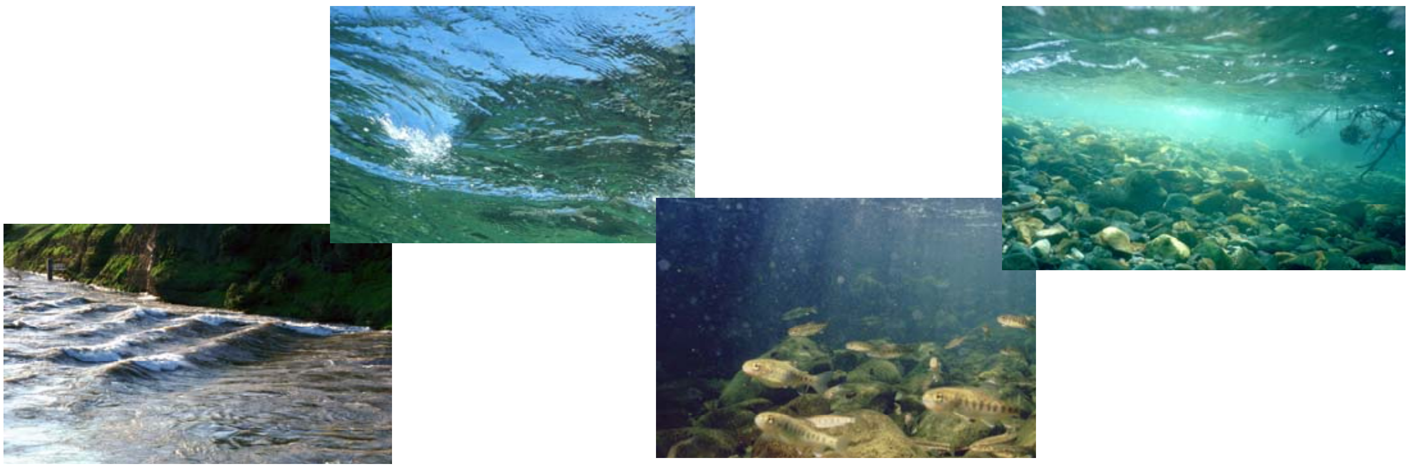
July 12, 1999

Flow: 47 cfs

Barrier Location	Fall Height	Downstream Pool length	Pool depth (base of falls)	Upstream Conditions	Alternate Routes around pool	Distance from LCDD (P. Ward CDFG, personal communication)
Quartz Bowl Pool Barrier	11.1 feet (Chute)	118 feet	16.5 feet	Small cascades, steep grade	None evident	1 mile Below LCDD
Lower Centerville Diversion Dam	14.2 feet (Dam Height)	28' wide 42' long 7.7' deep	No pool at base of dam	No information	None	0
	11.4 feet (Cascade)	42' wide 47'long				
Barrier 1 1st Barrier Above LCDD	14.4 feet	141 feet	No information	Pool 52' long 40' wide	Possible passage around bedrock outcropping on right bank	0.3 miles (approximate)
Barrier 2 2nd Barrier Above LCDD	13 feet	100 feet	11.8 feet	Pool 82' long	None, steep bedrock walls both sides	0.6 miles (approximate)
Barrier 3 3rd Barrier Above LCDD	12' 11.8' Total 23.8'	99 feet No defined pool, lots of big boulders/bedrock structures	4.3 feet	Cascades and 3'-4' deep Pools 4' rise over 40'	Possible ladder route on left bank	0.8 miles

Note - Holtgrieve reported that the Quartz Bowl barrier was dynamited in the 1930's, allowing occasional passage up to the Lower Centerville Diversion Dam.

FINAL WHITE PAPER
TURBIDITY AND SUSPENDED SEDIMENT EFFECTS ON SALMONIDS AND AQUATIC BIOTA IN FLOWING SYSTEMS



Prepared for:

ROCK CREEK-CRESTA PROJECT NO. 1962
ECOLOGICAL RESOURCES COMMITTEE

Sponsored by:

PACIFIC GAS & ELECTRIC COMPANY
San Francisco, CA

Prepared by:

ENTRIX, INC.
Sacramento, CA

Project No. 3066044

March 22, 2007

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March 22, 2007

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INTRODUCTION

This white paper was prepared to assist decision makers in understanding and addressing issues related to effects of turbidity and suspended sediment on fish and aquatic resources in California's Sierra Nevada rivers and streams. The approach used to prepare this paper was to compile available, relevant literature into one document and summarize the extensive amount of technical information available on this topic. An approach to interpreting the potential effects of suspended sediment and turbidity on fish, based on the literature, is presented in Section 8 of this paper.

There is an extensive literature on the effects of turbidity and suspended sediment on aquatic systems. The authors have compiled and reviewed over 200 documents obtained from biological database and Internet searches. These documents include peer-reviewed papers, theses, agency reports, gray, and published literature. We approached this paper without bias or any preconceived idea about how this paper would or could be used.

The effects of sedimentation or deposited sediment on physical habitat in streams are related to the effects of suspended sediment, but this topic is outside of the scope of this paper. A cursory review of literature related to sedimentation is provided in Appendix A.

Key questions addressed in this paper are as follows.

- What are the impact mechanisms for suspended sediment and turbidity to affect aquatic organisms or communities? Are effects behavioral, physical or physiological?
- What measures for suspended solids, turbidity, and water clarity provide the most accurate and informative data for purposes of assessing impacts to aquatic life? Given specific management objectives, which water quality measurement is most appropriate?
- How severe are the biological effects (short-term, acute or chronic, sublethal or lethal)? What potential thresholds or effect endpoints should be considered to assess effects on aquatic invertebrates, amphibians, fish and aquatic communities?
- To what extent can existing data and literature on exposure, duration, and event frequency relationships be used to guide impact assessment and management in California's Sierra Nevada streams and rivers?

This white paper is organized as follows.

Section 2 *Definitions, Properties and Measurement* defines the terms "turbidity," "water clarity" and "suspended sediment" and describes their respective properties. Methods used to measure these parameters are described. This section brings the reader up to current understanding of the issues associated with the measurement of water clarity.

Section 3 *Watershed Consideration*s summarizes watershed characteristics that influence the erosion and transport of sediment from watershed upland into streams. These processes characterize natural turbidity and suspended sediment formation and movement, and provide a context for considering changes from these conditions for regulatory purposes.

Section 4 *Recent Literature Reviews* summarizes several key, recent, peer-reviewed publications considering the effects of turbidity and suspended sediment on aquatic organisms and their habitat. The section provides a summary of literature reviews and models developed to evaluate the level of biological responses to these water quality parameters in the context of management objectives.

Sections 5, 6, and 7 summarize literature addressing macroinvertebrates, fish, and amphibians, respectively.

Section 8 *Summary and Recommendations* briefly summarizes the available information and provides some general recommendations to assessing effects of turbidity and suspended sediment on aquatic species and their habitat.

Section 9 contains the bibliography.

Because many of the tables and figures from the literature reviewed are relevant to our discussions, they were reproduced and included in this document.

Turbidity and suspended solids measurements are commonly used in assessments of stream water quality and, along with deposited sediments, evaluated in relation to their effects on aquatic biota. The concepts of turbidity, water clarity, and suspended sediment concentration are related, but distinct from each other. In order to measure turbidity and suspended solids and correctly interpret or apply the results, it is important that their definitions and physical properties are understood. This section defines these concepts and describes their respective properties.

Methods used to measure turbidity, water clarity, and suspended sediment also are described in this section, and the issues associated with these measurements are summarized. Measurements of these water quality parameters are sometimes used as surrogates for each other. Since different properties are measured, this approach is appropriate only in certain circumstances, and without cross-calibration can lead to misunderstanding of conditions and processes. An additional objective of this section is to summarize the definitions and properties of turbidity, water clarity, and suspended sediment as they apply to measurements of these parameters.

2.1 TURBIDITY AND WATER CLARITY

2.1.1 DEFINITION AND PROPERTIES

The American Society for Testing and Materials (ASTM) definition of turbidity was adopted during the April 30-May 2, 2002 Federal Interagency Workshop on Turbidity and Other Sediment Surrogates (USGS 2003), as follows:

> *Turbidity – an expression of the optical properties of a sample that causes light rays to be scattered and absorbed rather than transmitted in straight lines through a sample. (Turbidity of water is caused by the presence of suspended and dissolved matter such as clay, silt, finely divided organic matter, plankton, other microscopic organisms, organic acids, and dyes.)*

Turbidity is commonly used as a surrogate measure for either suspended sediment concentration (SSC) or water clarity. It is essential to understand how turbidity is measured and the limitations of its use for estimating SSC or water clarity. Davies-Colley and Smith (2001) provide a thorough review of the relationships among turbidity, suspended sediment, and water clarity, as well as an overview of light properties that affect measurements. The salient points from their review are included below.

Turbidity is caused by the intense scattering of light by fine suspended matter (composed of inorganic sediment and organic matter). Suspended solids contribute to both absorption and scattering of light. Light attenuation in water (reduced light transmission) by suspended particles is responsible for the “cloudiness” of turbid water and contributes to low visual clarity. Turbidity and water clarity, both optical properties, are inversely related, i.e. as turbidity increases, water clarity decreases. Davies-Colley and Smith (2001) distinguish between two main aspects of water clarity that relate to effects on the aquatic biota; these are "light penetration" and "visual clarity."

The photons in a light beam passing through water undergo both absorption (photon energy is ultimately converted to heat) and scattering (Figure 2-1). The sum of these two optical processes can be quantified as the beam attenuation coefficient. The absorption, scattering and beam attenuation coefficients are inherent optical properties of the water that do not depend on the incident light field. Beam attenuation can be directly measured with a beam transmissometer.

In contrast, the attenuation of diffuse light, quantified by the irradiation attenuation coefficient, depends not only on the inherent optical character of water, but also (weakly) on incident light conditions (such as the position of the sun or cloud cover). The euphotic depth, the depth at which photosynthetically available radiation (PAR) is reduced to 1 percent of its incident value, is proportional to the irradiance attenuation coefficient for the whole PAR waveband. The relationship between irradiance attenuation and suspended matter concentration or turbidity is generally non-linear (although it can be linear) and depends on the optical character of suspended particles. There is no universal relationship between light penetration and suspended sediment concentration, turbidity, or visual clarity; the relationship must be established empirically in a given water body (Davies-Colley and Smith 2001), which can vary with the flow and type and size of suspended matter.

Visual clarity, when measured as the hydrologic range (the maximum sighting distance [generally horizontal] of a perfectly black target), depends only on beam attenuation. The hydrologic range does not depend on lighting conditions, and therefore visual clarity measured as "black disc visibility" may have practical advantages for water quality management (Davies-Colley and Smith 2001).

The correlation of turbidity (scatter and absorption of light) and SSC depends on the optical character of the suspended matter being measured. The optical character of suspended matter varies widely, even in the same water body in different seasons (and flows) and at different locations. Hence, light penetration and water clarity will also vary. The important attributes of suspended particles are size, shape, and composition, with light attenuation depending most on particle size (geometrical cross-section per unit volume). Mineral particles in the size range of 0.2 to 5 µm and organic particles in the size range of 1 to 20 µm tend to dominate light attenuation, which in many natural waters are composed of clay minerals and phytoplankton cells (Davies-Colley and Smith 2001). Due to their highly aspherical shape, clay mineral crystals have a much higher light scattering effect than other more spherical particles. Particle composition also affects light attenuation when pigments within or on the particle's surface absorb light.

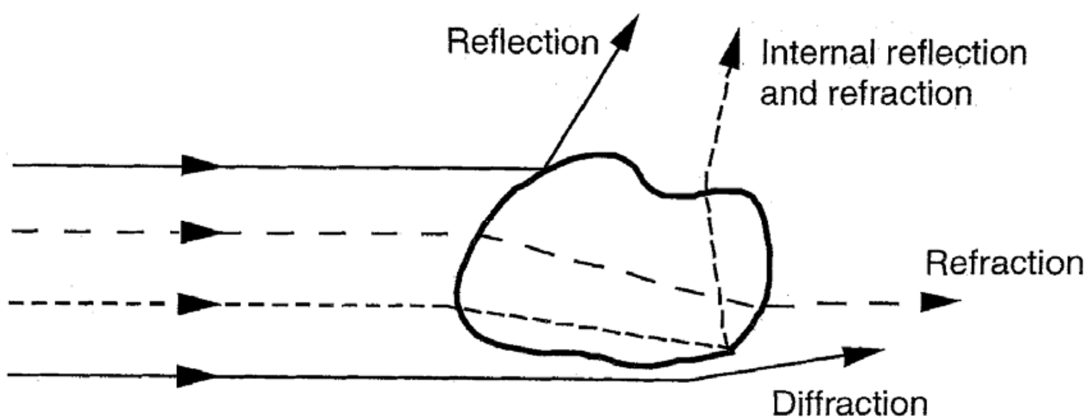


Figure 2-1. Schematic of the Scattering of Light by a Suspended Sediment Particle via the Process of Reflection, Refraction, and Diffraction (Davies-Colley and Smith 2001).

Similarly, organic matter often complexes with iron and aluminum hydrous oxide, which contributes significantly to light absorption.

The size of sediment particles plays an important role in their contribution to water clarity and turbidity. Except under very large flows, mineral particles generally only remain suspended if they are smaller than 63 µm. Larger particles tend to settle out very quickly and do not contribute significantly to the sustained reduction of water clarity. The disc-shaped clay minerals settle out at half the speed of similar-sized spherical particles, thus clays tend to be more persistent as suspended particles. Organic particles, or flocculated aggregates of organic and mineral particles that contain trapped water, have relatively low settling velocities, and thus remain suspended for long periods. The source of water may play an important role in the type of materials present and their persistence, as well. Streams receiving outflows from lakes frequently contain particulate organic matter originating from plankton and bacteria in lake waters. This source may represent a persistent seasonal source of suspended materials.

2.1.2 Measurement

2.1.2.1 Turbidity

Historically, the standard method for determination of turbidity was based on the Jackson candle turbidimeter. Results were reported as Jackson turbidity units, or JTUs. However, the lowest turbidity value that can be measured by this method is 25 units, which is higher than ambient values in many situations where measurement is needed. Due to this shortcoming, other methods were developed. Among the many methods that are available, nephelometry has become the most popular and widely used. The nephelometric method compares the intensity of light scattered by the sample under defined conditions with the intensity of light scattered by a standard reference suspension under the same conditions. The greater the scattering of light, the higher the turbidity values (APHA 1998). Results are reported as nephelometric turbidity units (NTU).

Nephelometric turbidity meters are designed to measure a portion of the scattered light from an incident light beam projected from the instrument. Because side scatter has a roughly constant ratio to total scatter, this relative index can produce consistent results. Typically, light scattered at a 90-degree angle to the incident beam is measured, although meters vary. Turbidity measured in NTUs is not an absolute quantity. Turbidity meters provide only a relative measure or an index of the light scattering that is actually occurring in a water body.

Due to the wide variety of instruments with different detector geometries and light sources, turbidity measurements of the same water are not equivalent among different meters (USGS 2004). Different types of meters are known to provide very different readings, often using different physical principles for measurement (e.g., transmissivity vs. reflectance). Comparisons of different meters have resulted in measurements differing by as much as 125 NTUs (Gray and Glysson 2003). For this reason, the USGS has recently implemented a new turbidity data reporting procedure for USGS databases and products, which specifies the make and model of turbidity instrument, as well as the light source, light wavelength, and detector geometry used by the meter (USGS 2004). In addition, they have revised Chapter 6, Section 6.7 Turbidity, of their *National Field Manual for the Collection of Water Quality Data*, which documents the USGS protocols for the collection, quality assurance, storage, and publication of water quality data. The purpose of these protocols is to improve the quality and comparability of reported turbidity data (Anderson 2004).

2.1.2.2 Water Clarity

Historically, water clarity has been measured using a Secchi disc, a white or black-and-white disc lowered into water until the image can no longer be seen. The depth at which it disappears is the Secchi depth. Secchi depth is inversely proportional to beam attenuation and irradiance attenuation (Davies-Colley and Smith 2001).

The maximum sighting distance of a black target (or disc), viewed horizontally, depends only on the beam attenuation coefficient and is therefore independent of lighting conditions. Black disc visibility can be observed by a snorkel diver. Observations from above water can be made using a viewer with a 45-degree mirror, or with a trough constructed of reflective material. If water is so turbid that short visual ranges make direct measurements difficult, observations can be made in a trough on a sample diluted with clear water of known clarity. A more recent development is the transparency tube, a clear tube with a Secchi disc or black disc painted on the bottom. The depth of water at which this image is no longer seen is measured (Davies-Colley and Smith 2001). Alternatively, beam transmissometers, which have not been widely adopted for work in freshwaters, but have been used extensively in marine waters, can be used to measure beam attenuation.

2.2 SUSPENDED SEDIMENT

2.2.1 DEFINITION AND PROPERTIES

Biologists recognize two components of sediment load by their respective effects on stream organisms: suspended sediment and deposited sediment. With regard to deposited sediment, stream biologists are interested in the degree of sedimentation on the streambed (more closely related to bedload transport). A brief review of literature related to sedimentation is provided in Appendix A.

Suspended sediments, along with other constituents such as organic matter, affect light attenuation and can have direct physiological or behavioral effects on aquatic organisms. The optical characteristics and settling velocities of organic and inorganic particles in the water column depend of particle size, shape, and composition (Davies-Colley and Smith 2001).

Usually the size of suspended sediment is 63 μm or less; suspended sediment primarily is comprised of silt and clay particles, but under certain stream conditions, higher flows with greater transport velocities, fine- to medium-sized sand may also be entrained in the suspended fraction as well (Waters 1995). The two categories of suspended and deposited sediments should not be considered as mutually exclusive to particle size, because they will overlap depending on stream conditions of water velocity and turbulence (Waters 1995). In a specific stream, these conditions may vary with flow and consequent initiation of motion and subsequent suspension of bed particles.

The size of sediment particles is an extremely important attribute in the effect produced upon stream communities (Waters 1995). A scale developed by the American Geophysical Union comprises 24 categories of sizes and class names (Lane 1947). Cummins (1962) provided stream ecologists with a classification of sediment that is more commonly used today. The scale has 11 particle sizes and names (e.g. clay, silt, sand, etc.) and is based on the Wentworth (1922) scale.

Sediment in the water column is estimated in three ways: total suspended solids, turbidity, and water clarity. However, the latter two categories actually are relevant to effects on light rather than sediment mass. The correlation between these measures is examined in Section 2.3.

2.2.1.1 Inorganic and Organic Components

Suspended solids measured as Total Suspended Sediment (TSS) or SSC consist of both inorganic and organic particles. Turbidity measurements include the integrated influence of both components on the scatter of a beam of light. Both inorganic and organic particles influence this measurement but their proportions may vary dramatically from one watershed to another, from one season to another, and from one flow level to another.

The proportions and physical characteristics of organic and inorganic particles may have a strong influence on the potential for adverse effects on aquatic biota. The two factors reported to have the most influence on the degree of aquatic impacts are the size and angularity of inorganic particles. The vast majority of the laboratory research on the adverse effects of suspended sediment on aquatic organisms has focused on the impacts associated with fine inorganic suspended sediment and largely ignores the potential mediating influences of the organic fraction. The composition of the suspended solids, proportions of organic and inorganic fractions, influences the effects on aquatic fauna (Duchrow and Everhart 1971).

Madej et al. (2002) reported that the general failure to distinguish between the organic and inorganic components hinders a full understanding of the effects of the particles on stream health. It has been shown that the organic component can account for more than half of the suspended load (Madej et al. 2002). In general, organic particles remain in suspension longer and contribute more to turbidity than mineral particles due to their size and composition. If there are higher proportions of organic matter in the suspended load, elevated turbidity may last longer due to the tendency of the organic particles to remain in suspension longer. This could result in a greater decrease in photosynthesis, and hence primary productivity, compared with turbidity that is caused primarily by inorganic particles. This has the potential to contribute to the loss of the invertebrate scrapers that feed on periphyton (the community of tiny organisms that live on the tops of rooted aquatic plants). Conversely, more organic matter could benefit filter-feeding invertebrates. The net effect of the ratio of organic to inorganic components of suspended load is unknown. Madej et al. (2002) recommend that suspended sediment measurements include a separation of the load into inorganic and organic fractions and into particle size distributions to better understand the watershed conditions.

2.2.1.2 Inorganic Particle Size and Angularity

Waters (1995) makes the point that suspended sediment is transported and dispersed in the stream channel depending upon the flow magnitude, and the velocity and size of the particles. Fine suspended sediment is typically less then 63 µm in diameter. Fine sediment particles tend to remain suspended in the water column and contribute to the turbid conditions. Coarser sand particles tend to settle from the flow and only make a significant contribution to suspended sediment during higher flows. Organic particles that typically have a lower density will remain with the fined suspended sediment and be transported downstream or to the stream margins.

Several authors have indicated that the angularity of the inorganic sediment particles may make a significant difference in the amount of injury that fish experience during high flows (Noggle 1978, Redding et al. 1987, Newcombe 2003). Angularity has also been shown to influence the degree to which fine particles enter into and clog the interstitial spaces in gravel substrate.

2.2.2 MEASUREMENT

The most reliable and consistent method of measurement for suspended solids is considered to be as a mass per unit volume (mg/L) (Noggle 1978). This standardized approach is not subject to the potential sources of error that can complicate mass per mass (gram of solids per gram of water) comparisons, often reported as parts per million (ppm). Because the density of water changes with temperature, the volumetric relationship between mass of solids and mass of water may vary from one set of measurements to another.

The investigator must first determine the time base for measurements. In increasing time spans, these time bases include event-based (point or grab) sampling and continuous sampling (Hicks and Gomez 2003). Continuous sampling typically yields a superior picture of sediment profiles, but because continuous sampling requires that expensive equipment remain in place for extended times, point sampling may be more practical in publicly accessible areas. Water velocity bears directly on flow competence, so it is measured at the same time that water samples are collected for sediment measurements.

Suspended sediment and bedload are sampled and analyzed differently, as described below and in Section 2.3.

2.2.2.1 Suspended Solids Sampling

Point sampling. Although sediment can theoretically be measured anywhere along a stream, in practice it is simplest to measure in transects across smooth runs and riffles of moderate depth (Beschta 1996). Such stream regions are usually safer for workers and for equipment, and it is usually straightforward to designate uniform sampling points if these are necessary. Point samples for suspended load are obtained with special sediment samplers suspended in the current by cables at specific depths. These samplers resemble small torpedoes and are hydrodynamically designed to minimize turbulence at the intake and to sample isokinetically (at ambient flows) so that they do not exclude or favor specific particle radii ranges. A removable flask inside the device collects the sample. A solenoid-operated valve controls the water/sediment collection, and sample volumes typically span 100-500 ml. The flask is removed and stored under refrigeration for laboratory analysis.

Depth-integrated sampling. Inconsistencies in the sediment concentration with depth in the water column arise from changes in distance from the sediment source, water depth, flow variation, and streambed topography variability. Thus, single point sampling does not accurately characterize the suspended sediment profile, except perhaps in special cases such as very small and shallow streams. In practice, sampling should be done at various depths (depth-integrated), to generate a profile of sediment load as a function of water depth and velocity (Hicks and Gomez 2003). Sampling can be depth-integrated by collecting point samples at several depths and plotting the sediment burden versus water velocity across the sampling range. However, standard point samplers such as the US P-61 may inadvertently collect bedload, which will distort and invalidate measurements of suspended sediment, and in any case multiple point sampling is tedious and labor-intensive (Edwards and Glysson 1999). Special depth-integrating samplers such as the DH-48 are designed to sample only to depths above typical bedload, and can yield substantial data that better reflects stream-wide sediment profiles. The sampling method for these devices is quite different from standard point sampling. Depth integrating samplers are operated by moving the sampler steadily and at uniform velocity from surface to bed and back to the surface with the collection valve continuously open. Sample flasks are refrigerated for later laboratory analysis.

Continuous sampling. Continuous sampling refers to repeated, single collections of water with entrained suspended sediment over time periods that can range from minutes to months. Such a program can yield accurate suspended sediment profiles, but the equipment for this methodology is typically far more elaborate than that required for point or depth-integrated sampling. The "pumping sampler" used for continuous sampling consists of an intake system, a pump, a sampling frequency control system, and a power source. Although the units are compact and portable, their drawbacks include expense, inability to sample isokinetically, and easily clogged intake ports (Beschta 1980, Hicks and Gomez 2003). Samples are collected from the pumping unit at appropriate intervals and refrigerated for laboratory analysis.

2.2.2.2 Suspended Sediment Analysis

Water samples with entrained sediment are analyzed directly for suspended sediment by recovering and weighing the suspended sediment particles and calculating mass per unit sample volume. To recover the sediment a predetermined sample volume is vacuum filtered through pre-weighed filter paper, and the filter paper with filtered sediment is oven-dried for 24 hours at 105°C. The paper is then weighed and the tare weight subtracted, and the resulting sediment weight is divided by the initial water volume to calculate sediment concentration (mg/L). Sediment concentration is plotted against streamflow to yield a sediment-rating curve for the stream (Beschta 1996).

2.3 CORRELATION OF SUSPENDED SEDIMENT CONCENTRATION, TURBIDITY, AND VISUAL CLARITY

The relationship between turbidity and the concentration of suspended matter depends on the variability in the optical characteristics of the suspended matter, as determined by the size, shape, and refractive index of the particles. Optically black particles, such as those of activated carbon, may absorb light and effectively increase turbidity measurements (APHA 1998). Several authors have reported very strong relationships between suspended sediment and turbidity for specific rivers and laboratory conditions (Lloyd et al 1987, Gregory 1993, Redding et al. 1987). In many of these instances the sediment has been derived from a relatively uniform source and used in a laboratory setting (Gregory 1993 and Redding et al. 1987).

Other researchers have indicated that turbidity alone is not a reasonable surrogate measurement for suspended solids because the particle size, optical properties, organic components, and angularity of the inorganic particles greatly influence turbidity measurements (Bash et al. 2001, Anderson 1996). The relationship between these properties and turbidity vary with changes in flow and substrate type (Noggle 1978, Bash et al. 2001). The relationship will also vary from one reach of a stream to another as conditions change (e.g., gradient, substrate type, and tributary inflow).

The relationship between turbidity and suspended matter is a function of watershed-specific factors and temporal trends within storms and across seasons. Additionally, it is likely that suspended matter and turbidity measurements will differ on the rising limb of the hydrograph compared with similar discharges on the falling limb (known as hysteresis). This was observed on a multi-year and multi-watershed study in a series of Sierra foothill streams (Lewis et al. 2002). Regression slopes differed significantly between watersheds and TSS and turbidity values increased during the transition from the wetting to saturation phases of the seasonal rainfall pattern. TSS and turbidity were variable across seasons and storms. Lewis et al. (2002) recommended intensive sampling across multiple storms and for a full range of flows for establishing valid storm-based turbidity relationships. Davies-Colley and Smith (2001) also point out that turbidity is a relative measure, and that to be used, turbidity must be calibrated to an absolute measure.

Davies-Colley and Smith (2001) suggest that turbidity may be useful when a relative index of water cloudiness is sufficient to meet management objectives. However, they suggest visual water clarity measurement is preferred over turbidity and SSC measurements, particularly when the optical effect of the suspended sediment is of primary concern. They indicate water clarity is an absolute quantity (unlike turbidity), can be measured with better precision, can be measured more cheaply, and has a more immediate environmental relevance (Table 2-1).

Water quality management objectives are a key consideration when deciding which water quality measurement is most appropriate. Where sediment loading is the concern, mass concentration measures (suspended sediment concentration) would be appropriate. If biological effects related to changes in light penetration or water clarity are the main concern, then measurement of turbidity or water clarity would be more relevant than measurement of sediment concentrations (Davies-Colley and Smith 2001). Specifically, they recommend measures of visual clarity rather than turbidity.

Table 2-1. Comparison of Suspended Sediment Concentration, Nephelometric Turbidity, and Visual Clarity Measurement for Water Quality Assessment (Davies-Colley and Smith 2001).

Attribute	Suspended Sediment Concentration (SSC) (non-filterable residue - NFR)	Turbidity	Visual Clarity (or alternatively, beam attenuation coefficient)
Principle and Procedure	Weight of particulates captured on a glass fiber filter through which a known volume of water sample has been filtered.	Side scattering of light by nephelometry (relative scale).	Sighting range of a black disc viewed horizontally through water (or Secchi depth) (alternatively, beam transmittance measurement).
Equipment	Filter assembly with vacuum pump, oven, weigh balance, desiccator, glass fiber filters.	Nephelometer and standards.	Underwater viewer and visual target, tape measure (beam transmissometer).
Calibration?	None	Arbitrary calibration to formazin	None
Scientific Measurement? (units)	Yes (g m^{-3}).	No, arbitrary, relative measurement (in NTU).	Yes (m) (beam attenuation coefficient, m^{-1}).
Cost (and difficulty)	Simple, but involved and consumptive of technician time - hence expensive. $23/sample (NZ$ 18/sample)	Fairly simple (standards required for calibration). $5/sample (NZ$6/sample)	Simple, but does require access to water body. $4/observation[1] (NZ$4/observation)
Precision (typical standard error)	10 percent[2]	10 percent[3]	4 percent[4,5]
Sample Size	Depends on sediment concentration. For best precision, 100 mg is required (i.e., 10L volume at 10 g m^{-3}).	100 ml or less for a laboratory measurement.	Not applicable (usually done in situ).
Stability of Samples (and storage)	Stable for several days (store chilled, dark).	Unstable (store chilled, dark, and measure within 24-hours of collection).	Not applicable (usually done in situ).
On Site or In situ Measurement?	No (must be done in a laboratory).	Yes (portable models).	Yes (usual procedure).
Continuous Monitoring?	No.	Yes (in situ turbidity monitors).	Yes, as beam transmittance (from which visibility may be calculated).
Environmental Relevance	Relevant to sediment yields (in geomorphology, agronomy), and benthic effects of sedimentation. Less relevant to optical effects.	Indirectly relevant - because the measurement is relative to arbitrary standards. Requires calibration (e.g., to suspended solids or visual clarity).	Relevant to aesthetic quality of water and habitat for sighted aquatic animals. Less relevant to sediment mass-related impacts.

[1]Assuming ten minutes *extra* on site per observation (at US$20/hr), and allowing $0.60/observation for equipment depreciation.

[2]McGirr (1974), APHA (1998)

[3]McGirr (1974), ASTM (1996), U.S. EPA (1999)

[4]Davies-Colley and Close (1990)

[5]Smith and Hoover (1999)

This section summarizes watershed characteristics that influence the transport of sediment from watershed uplands into streams. Sediment transport is controlled by hydrologic conditions, geologic and soils conditions, topography, vegetation and other surface cover, and the occurrence of natural and anthropogenic disturbances. Each of these factors has several components that will increase or decrease the loading of sediment and solids to a stream, and will influence the character of the suspended solid load under different conditions. As such, these factors control the nature and occurrence of suspended sediment in streams and rivers. The development of ambient water quality criteria for turbidity and suspended sediment relies on knowledge of natural background conditions in the watershed. Due to the high degree of variability in suspended sediment loading and numerous natural and anthropogenic influences on watershed conditions, however, characterizing background turbidity and suspended solids levels is a daunting task (Bash et al. 2001).

3.1 HYDROLOGY AND EROSION

Sediment is transported from watershed surfaces to channels in three main ways: soil loss and erosion from upland areas, mass movement such as landslides (Swanston 1991, Beschta 1996), and streambank or channel erosion (Flosi et al. 1998). The interplay between hydrology and watershed characteristics determines the dominant erosion process on a seasonal basis.

3.1.1 HYDROLOGIC CYCLE

Climate affects the amount and form of precipitation (rain, snow or fog) that enters a watershed, and the timing, duration, and intensity of precipitation events.

Precipitation falls to the surface of the soil or is intercepted by vegetation or litter. Intercepted precipitation eventually either continues to the soil surface or evaporates. Precipitation that reaches the soil surface infiltrates the soil, or, if the soil is saturated or the infiltration rate is lower than the precipitation rate, moves over the surface of the ground as runoff (Fredriksen and Harr 1981, Swanston 1991). When the amount of moisture produced by a precipitation event exceeds the infiltration capacity of the soil, non-perennial source channels form. These provide new source areas for sediment (Fredriksen and Harr 1981, Swanston 1991). The type of precipitation affects the rate and timing of delivery to the channel, and the timing of surface erosion affects the type and volume of sediment (Fredriksen and Harr 1981).

If water infiltrates soil, it either evaporates, is withdrawn by plants for transpiration, is held in the soil, moves into bedrock, or flows, eventually, into a stream as baseflow (Swanston 1991, Fredriksen and Harr 1981).

3.1.2 Erosion and Sediment Transport

In a rain-dominated hydrologic system, the streamflow regime follows the rainy season with increased streamflow following precipitation events (Swanston 1991). The Sierra Nevada receives the majority of total precipitation as snow, which is stored in a snowpack. This affects the timing and rate of water delivery to streams. Moisture stored in snowpack generally melts in a relatively short period of time during spring (April to June, or later depending upon the elevation, water year and snow pack), which results in high flows that have the capacity to transport high levels of sediment into streams (Swanston 1991).

The steepness, gradient, and length of a slope influence the rate that sediment is transported into a channel and the predominant erosion process (Swanston 1991, Beschta 1996). The aspect of a slope also can affect the timing of snowmelt (Swanston 1991), which may affect the level of erosion from high flows.

The rock and soil types that constitute the sedimentary parent material control an area's susceptibility to erosion. The parent material affects the composition and structure of the soil, which determines how much water can infiltrate, in comparison with how much runs off and erodes during a precipitation event (Swanston 1991). It also influences the strength of the aggregate that a soil will form, another characteristic that can affect erosion (Fredriksen and Harr 1981), and soil potential for slumps or earthflows (Swanston 1991).

The amount and type of vegetation and surface cover in a watershed influence the amount of water that reaches the surface of the soil and the amount of water that moves through soil as subsurface flow into a stream. Vegetation and other surface cover impede erosion by dissipating the energy in raindrops through interception, protecting the soil surface (litter) and stabilizing soil structure (roots) (Fredriksen and Harr 1981, Beschta et al. 1995, cited in Spence et al. 1996).

Once sediment is delivered to a stream, the two dominant mechanisms of sediment transport in streams are bedload (sliding, rolling, or bouncing along the bottom) and suspended load. Consideration of bedload transport is beyond the scope of this paper, and principally affects the instream habitat characteristics of the channel bed. Suspended load is typically limited to clay and silt-sized particles, which can be moved under most flows. Once entrained, suspended sediment can remain in suspension until a large reduction in stream velocity and energy occurs. During high flows, streams can carry very high levels of suspended load, affecting the viscosity and flow properties of the water. In practice, the available source, rather than the transport capacity of the stream itself governs the amount of suspended load. That is, streams typically carry the volume of suspended load available to them, only depositing in very low energy reaches such as pools (Simons and Senturk 1992).

3.2 NATURAL EVENTS

3.2.1 SOIL MASS MOVEMENTS

Soil mass movements, as slumps, landslides, and debris flows, are major contributors to sediment in streams (Swanston 1991). Frictional resistance of soil material and vegetation restricts these movements, and is influenced by the soil moisture level (Fredriksen and Harr 1981). Both the resistance to flow provided by bedrock and the characteristics of the soil determine which type of mass movement it will be (Fredriksen and Harr 1981). Vegetation, which removes water from soil and reinforces soil structure with roots, can decrease the likelihood of some mass movements (Fredriksen and Harr 1981).

3.2.2 FIRE

Fire can remove a large amount of vegetation and litter within a watershed (Swanston 1991), resulting in decreased protection for the soil surface and an increase in the amount of water that reaches the surface, both of which increase surface erosion (Swanston 1991). Reduced vegetation may also decrease the amount of water transferred from the soil for transpiration, another factor that increases soil moisture level. Roots, which aid in stabilizing slopes and stream banks, are often destroyed, which can lead to increased risk of landslides (Swanston 1991). Fire may also create a hydrophobic layer near the soil surface, decreasing the infiltration capacity and thereby increasing surface runoff and erosion (Swanston 1991). Fire-affected watersheds recover with time, at a rate dependant on the soil, climate, and vegetation succession.

3.3 ANTHROPOGENIC SOURCES OF SEDIMENT

3.3.1 MINING, TIMBER HARVEST, ROADS, AND GRAZING

Human uses of the land have a significant influence on the sources, types, and amount of sediment contributed to streams. Watershed activities that are known to contribute to erosion and stream sedimentation include mining, timber harvest, road construction and maintenance, fire, and reservoir operation. Mining and suction dredging have occurred in the mountains of Western North America for well over 100 years (Sumner and Smith 1940), and have caused increased levels of sedimentation in streams (Nelson et al. 1991). Grazing influences on stream channelization can have a significant influence on sediment transport in small streams and habitat conditions for salmonids (Chapman and Knudsen 1980).

Several authors have discussed the increased sediment impact to streams associated with timber harvest such as clear cutting, skidding, slash and burn, road development, and hauling (Anderson 1996, Holtby 1988, Holtby et al. 1989). Road construction, use and maintenance are considered to be some of the most significant sources contributing sediment to streams (Furniss et al. 1991). Roads increase the vulnerability of an area to erosion for a long period after the road is built and can increase the frequency and severity of mass soil movements (Furniss et al. 1991).

3.4 ANNUAL VARIABILITY

The input of sediments to streams is highly variable. In most western mountain watersheds, it generally occurs during episodic flows associated with storm water runoff events in the wet season and large magnitude flow releases sometimes associated with reservoir operations. These can occur during any time of the year, but are generally associated with runoff, generation, channel maintenance, or recreational flows. Suspended solids loading under natural flow conditions occurs predominantly during the winter wet season, spring snowmelt runoff period, or summer thunderstorms. In some instances the highest loading of fine suspended solids occurs during the first flush of the watershed in the fall or early winter. The first storm water runoff of the season will transport fine particulate material that has accumulated on surfaces throughout the dry season to local streams and rivers. In regulated systems, the highest loading can occur due to these same precipitation-induced processes, or by releases from reservoirs. In some cases, the reservoir release affects the timing of sediment loading to a river that would otherwise occur during precipitation events.

Road construction and repair is most likely to occur during the summer dry season and may result in the discharge of sediment to local streams (Bash et al. 2001). Individual construction activities may be short in duration, but the potential for several construction projects to take place in a single watershed exists, potentially leading to a series of sediment releases. Depending on the frequency and magnitude of these releases, the potential for a significant cumulative impact exists (Bash et al. 2001).

3.5 REGULATED FLOW

Natural flows in the watersheds of Sierra Nevada and Cascade Mountains of the western United States are in many cases typified by relatively low winter flows, except during storm water runoff events, and then higher flows during spring snowmelt. Rivers that are sustained by relatively consistent sources of emergent groundwater may exhibit sustained flow throughout what otherwise would be the dry season. Watersheds dominated by shallow bedrock conditions may have limited groundwater storage capacity and may exhibit high peak runoff in the spring followed by much lower flows during the dry season (remainder of the year).

Reservoirs capture winter rains and spring snowmelt and release stored water during the dry season when demand for water supply and power generation is higher. This has the effect of dampening winter storm and spring runoff peak flows in the rivers downstream of the reservoirs. During the dry season, flows below hydroelectric project bypass reaches tend to be higher and more consistent as a result of instream flow releases for aquatic resources. The subset of hydroelectric projects with reaches subject to peaking operations typically experience more variable releases during the summer months than would have occurred naturally. Figure 3-1 shows daily mean streamflow for two rivers: the Cosumnes River, an unregulated river, and Stanislaus River, a regulated river.

Reservoirs can act as sediment traps, reducing the amount of sediment in downstream reaches (Williams and Wolman 1984). Some dams are designed to flush sediment continuously, while others flush periodically (Williams and Wolman 1984), and others not at all. The amount of sediment released during a flush depends on variables that include: the volume of water released, the volume of sediment in the reservoir, the release rate, the type and location of outlet gates, and the size of the sediment particles (Williams and Wolman 1984).

By trapping sediment, reservoirs can affect the timing of higher turbidity flows. When initial high releases from reservoirs occur, they can transport sediment accumulated during the previous months that would otherwise have been transported naturally during precipitation events or snowmelt.

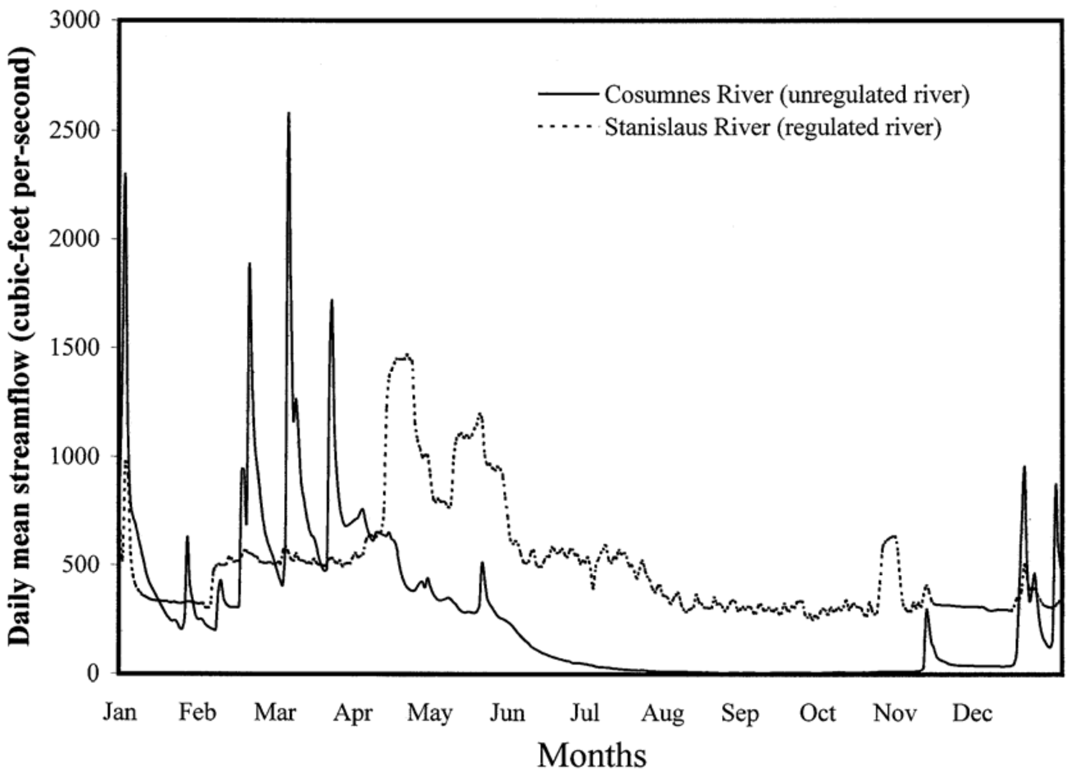


Figure 3-1. Mean Daily Streamflow (January 1 - December 31, 2002) of an Unregulated (Cosumnes River USGS gage 11335000) (Drainage Area 536 mi^2) and Regulated River (Stanislaus River USGS gage 11303000) (Drainage Area 1,075 mi^2) (USGS National Water Information System).

Several key scientific papers published since the late 1980s have compiled field and laboratory study results and developed the concepts that relate the effects of suspended sediment to biological responses in fish, macroinvertebrates and aquatic ecosystems (Lloyd et al. 1987, Anderson 1996, Newcombe and Jensen 1996, Bash et al. 2001). More recent papers elucidate the relationships between turbidity, water clarity, and suspended sediment, and propose ways to measure these parameters and evaluate the level of biological responses (Davies-Colley and Smith 2001, Newcombe 2003). Collectively, these papers provide an overview of the current level of understanding of turbidity (and related effects on water clarity) and suspended sediment and their effects on aquatic ecosystems. The review papers compile and summarize relevant information that can be applied by managers to regional issues.

The literature reviewed addresses physiological and optical effects of suspended sediment and turbidity. It provides the foundation for the development of a framework and models to evaluate the severity of adverse effects on clear-water fishes in relation to visual clarity of water and duration of exposure (Newcombe and Jensen 1996, Newcombe 2003). A discussion of these models is provided at the end of this section. The available literature does not generally address the relationship between the frequency of exposure to increased turbidity or suspended sediment events with the severity of ill effects.

Suspended solids can have a range of effects on aquatic organisms. These effects may be physiological (e.g. irritation or damage of fish gills). These effects also may be optical in nature. Light attenuation, the magnitude of which depends on the optical character of suspended matter, has two main biotic effects: a) reduced penetration of light into water for photosynthesis, and b) reduced visual range for sighted organisms (water clarity) (Davies-Colley and Smith 2001). Light attenuation may affect multiple trophic layers of an aquatic community, including algae, and other plants as well as the invertebrate community that supports fish, amphibians, and other wildlife species. This is discussed in greater detail in Sections 5, 6, and 7, which review studies addressing invertebrates, fish, and amphibians, respectively.

4.1 LLOYD 1987

Lloyd (1987) conducted a literature review to support an evaluation the State of Alaska's water quality standards. The paper includes a two-page summary table of published studies on the effects of SSC and/or turbidity on life stages of salmon and trout species organized from severe lethal effects to minor behavioral effect. (This table was updated by Bash et al. 2001, see Section 4.4). The review addresses trophic level changes related to reduction in light penetration, as well as direct effects of sediment and turbidity on aquatic organisms.

Lloyd (1987) summarizes the turbidity standards of nine states, including California. He also summarizes recommended levels of SSC for the protection of fish habitat (which offer high or moderate levels of protection) from several literature sources. Material reviewed from the European Inland Fisheries Advisory Commission (EIFAC) noted that waters containing 0 to 25 ppm of chemically inert solids should not adversely affect freshwater fisheries. Waters containing SSC of 25 to 80 ppm may lower the production of fish; waters containing SSC of greater than 80 ppm are unlikely to support good fisheries (EIFAC 1964 as cited in Lloyd 1987). Justification to apply these levels to the US is questioned based on effects seen at lower levels. For instance, a turbidity level of 10 NTUs can significantly reduce feeding rate, food assimilation and reproductive potential of *Daphnia pulex*, (McCabe and O'Brien 1983 as cited in Lloyd 1987). References to other studies show that turbidities in the 10 to 25 NTU range and SSC near 35 ppm had deleterious effects on fish. Lloyd (1987) discusses light penetration and productivity in streams, but cautions that assumptions about the importance of aquatic primary production in streams may not be completely applicable to vegetated watersheds, which likely rely more on organic material from terrestrial sources (Chapman and Demory 1963, Chapman 1966 cited in Lloyd 1987). However, even in heavily forested watersheds of the Pacific Northwest, increased light to stream bottoms can translate to more salmonids (Chapman and Knudsen 1980 and Murphy et al. 1981, cited in Lloyd et al. 1987). Lloyd (1987) notes that turbidities of 4 to 8 NTUs may hamper efficient management of fisheries in Alaska because aerial observers cannot see into the streams and estimate returns of adult salmon. Furthermore, absolute turbidities of 8 NTUs or higher have been shown to reduce sport fishing in fish bearing waters of Alaska.

The paper notes that an effective turbidity standard must prevent a loss of aquatic productivity and cause no lethal or chronic sublethal effects on fish and wildlife. It also notes that to be effective the standard should not be constrained by language requiring extensive work by the resource agencies to establish background levels. Alaska's standard is 25 NTUs above natural background in streams and 5 NTUs in lakes. An increase of 5 NTU in a clear, shallow (< 0.5 m) stream can decrease primary productivity by about 3 to 13 percent, and a 25 NTU increase in a shallow stream may reduce plant production by about 13 to 50 percent depending on stream depth (Lloyd et al., 1987). In a clear-water lake, a 5 NTU increase may reduce the productive volume of that lake by about 80 percent (Lloyd et al., 1987). Lloyd (1987) recognizes that even stricter standards may be necessary to protect extremely clear waters due to the dramatic initial impact of turbidity on light penetration. The paper concludes that Alaska's standards can be expected to provide a moderate level of protection for clear coldwater habitats. A higher level of protection may be required to protect extremely clear waters because of the dramatic initial effect of turbidity on light penetration.

4.2 NEWCOMBE AND JENSEN 1996

Newcombe and Jensen (1996) address the need for a reliable metric to quantify the assessment of risk and impact for fishes subjected to excess suspended sediment pollution. This work lays the foundation for subsequent development of the Newcombe

(2003) impact assessment model (see Section 4.7). A meta-analysis of 80 published and adequately documented reports (previously summarized by Lloyd 1987) on fish responses to suspended sediments in streams and estuaries yielded six empirical equations that relate biological response to the duration of exposure and SSC. This paper uses an exposure and duration concept for assessing suspended sediment effects on fish and includes a link to the effect of sediment grain size. The relationship is characterized by three variables: 1) the duration of exposure as the natural log of hours; 2) the natural log concentration of sediment in mg/L, and 3) a stress index value that is based on value of exposure level times duration of exposure.

Newcombe and Jensen (1996) developed tables that represent a method to assess the empirical relationship between the response of fishes to suspended sediment pollution (Table 4-1). The empirical relationship is augmented by the severity-of-ill-effect (SEV) scores (Table 4-2) in the table cells. These are a semi-quantitative scoring based on a 15-point scale and on which is superimposed four decision categories: no effect, behavioral, sub-lethal and lethal effects. The table cells are organized along a logarithmic scale of hours on the X-axis and a logarithmic scale of SSC in mg/L along the Y-axis. The paper presents SEV scores for various combinations of seven different fish groups including adult salmonids, juvenile salmonids, eggs and larvae of salmonids and non-salmonids, adult estuarine salmonids, adult estuarine non-salmonids and adult freshwater non-salmonids.

4.3 ANDERSON 1996

Anderson (1996) reviewed literature on the effects of increased sediment load and sedimentation relating to forestry operations on aquatic ecosystems. The review briefly addresses suspended sediment effects on fish (behavioral, physiological and population) and suspended sediment and sedimentation on freshwater aquatic habitat. Anderson used the exposure and duration tables developed by Newcombe and Jensen (1996) to introduce a similar severity-of-ill-effects rankings for physical habitat in freshwater systems (Table 4-2).

4.4 BASH ET AL. 2001

This paper was prepared by the Center for Streamside Studies, University of Washington, for Washington State to assist agencies that implement transportation projects. The paper is a comprehensive review of literature up to 2001. There is a concise discussion of the differences between suspended solids, suspended sediment, and the limitations associated with turbidity and water clarity measurements. The paper addresses the adverse effects of suspended solids on coldwater fishes including physiological effects, behavioral effects and habitat effects, and acknowledges the importance of the organic fraction in the suspended solids as a direct impact on dissolved oxygen levels. The paper provides useful summary tables updated from Lloyd (1987) that summarizes important attributes of the covered studies (Tables 4-3, 4-4 and 4-5).

Table 4-1. Calculated Severity of Ill-Effects Scores in the Matrix of Suspended Sediment (SS) Concentration and Duration of Exposure for Different Salmonid Life Stages (Newcombe and Jensen 1996).

Adult Salmonids

Duration of exposure to SS ($\log_e$ hours)

A

Average severity-of-ill-effect scores (calculated)

Concentration (mg SS/L)	0	1	2	3	4	5	6	7	8	9	10	($\log_e$ mg SS/L)
162755	11	11	12	12	13	13	14	14	-	-	-	12
59874	10	10	11	11	12	12	13	13	14	14	-	11
22026	9	10	10	11	11	12	12	13	13	14	14	10
8103	8	9	9	10	10	11	11	12	12	13	13	9
2981	8	8	9	9	10	10	11	11	12	12	13	8
1097	7	7	8	8	9	9	10	10	11	11	12	7
403	6	7	7	8	8	9	9	10	10	11	11	6
148	5	6	6	7	7	8	8	9	9	10	10	5
55	5	5	6	6	7	7	8	8	9	9	9	4
20	4	4	5	5	6	6	7	7	8	8	9	3
7	3	4	4	5	5	6	6	7	7	7	8	2
3	2	3	3	4	4	5	5	6	6	7	7	1
1	2	2	3	3	4	4	5	5	5	6	6	0
	1	3	7	1	2	6	2	7	4	11	30	
	Hours			Days			Weeks		Months			

Juvenile and Adult Salmonids

Duration of exposure to SS ($\log_e$ hours)

C

Average severity-of-ill-effect scores (calculated)

Concentration (mg SS/L)	0	1	2	3	4	5	6	7	8	9	10	($\log_e$ mg SS/L)
162755	10	11	11	12	12	13	14	14	-	-	-	12
59874	9	10	10	11	12	12	13	13	14	-	-	11
22026	8	9	10	10	11	11	12	13	13	14	-	10
8103	8	8	9	10	10	11	11	12	13	13	14	9
2981	7	8	8	9	9	10	11	11	12	12	13	8
1097	6	7	7	8	9	9	10	10	11	12	12	7
403	5	6	7	7	8	9	9	10	10	11	12	6
148	5	5	6	7	7	8	8	9	10	10	11	5
55	4	5	5	6	6	7	8	8	9	9	10	4
20	3	4	4	5	6	6	7	8	8	9	9	3
7	3	3	4	4	5	6	6	7	7	8	9	2
3	2	2	3	4	4	5	5	6	7	7	8	1
1	1	2	2	3	3	4	5	5	6	7	7	0
	1	3	7	1	2	6	2	7	4	11	30	
	Hours			Days			Weeks		Months			

Juvenile Salmonids

Duration of exposure to SS ($\log_e$ hours)

B

Average severity-of-ill-effect scores (calculated)

Concentration (mg SS/L)	0	1	2	3	4	5	6	7	8	9	10	($\log_e$ mg SS/L)
162755	9	10	11	11	12	13	14	14	-	-	-	12
59874	9	9	10	11	11	12	13	14	14	-	-	11
22026	8	9	9	10	11	11	12	13	13	14	-	10
8103	7	8	9	9	10	11	11	12	13	13	14	9
2981	6	7	8	9	9	10	11	11	12	13	13	8
1097	6	6	7	8	9	9	10	11	11	12	13	7
403	5	6	6	7	8	9	9	10	11	11	12	6
148	4	5	6	6	7	8	9	9	10	11	11	5
55	4	4	5	6	6	7	8	8	9	10	11	4
20	3	4	4	5	6	6	7	8	8	9	10	3
7	2	3	4	4	5	6	6	7	8	8	9	2
3	1	2	3	4	4	5	6	6	7	8	8	1
1	1	1	2	3	4	4	5	6	6	7	8	0
	1	3	7	1	2	6	2	7	4	11	30	
	Hours			Days			Weeks		Months			

Eggs and Larvae of Salmonids and Nonsalmonids

Duration of exposure to SS ($\log_e$ hours)

D

Average severity-of-ill-effect scores (calculated)

Concentration (mg SS/L)	0	1	2	3	4	5	6	7	8	9	10	($\log_e$ mg SS/L)
162755	7	9	10	11	12	13	14	-	-	-	-	12
59874	7	8	9	10	12	13	14	-	-	-	-	11
22026	7	8	9	10	11	12	13	-	-	-	-	10
8103	7	8	9	10	11	12	13	14	-	-	-	9
2981	6	7	8	10	11	12	13	14	-	-	-	8
1097	6	7	8	9	10	11	12	14	-	-	-	7
403	6	7	8	9	10	11	12	13	14	-	-	6
148	5	6	7	9	10	11	12	13	14	-	-	5
55	5	6	7	8	9	10	12	13	14	-	-	4
20	5	6	7	8	9	10	11	12	13	-	-	3
7	4	5	7	8	9	10	11	12	13	14	-	2
3	4	5	6	7	8	10	11	12	13	14	-	1
1	4	5	6	7	8	9	10	11	13	14	-	0
	1	3	7	1	2	6	2	7	4	11	30	
	Hours			Days			Weeks		Months			

Note: Diagonal terraced lines denote thresholds of sublethal effects (lower left) and lethal effects (middle diagonal). Shaded areas represent extrapolations beyond empirical data.

Table 4-2. Severity of Ill-Effects Table for Impacts to Physical Habitat in Freshwater Systems (Anderson 1996, after Newcombe and Jensen 1996).

	Description of effect	
Rank	Effects on fish behavior, physiology, and survival	Effects on aquatic habitat
	SEV (Newcombe and Jensen 1996)	SE (Anderson 1996)
	Nil effect	
0	No behavioral effects	
	Behavioral effects	
1	Alarm reaction	
2	Abandonment of cover	
3	Avoidance response	Measured change in habitat preference
	Sublethal effects	
4	Short-term reduction in feeding rates or feeding success	
5	Minor physiological stress:	
	increase in rate of coughing	
	increased respiration rate	
6	Moderate physiological stress	
7	Moderate habitat degradation; impaired homing	Moderate habitat degradation
8	Indications of major physiological stress:	
	long-term reduction in feeding rate or feeding success	
	poor condition	
	Lethal and paralethal effects	
9	Reduced growth rate:	
	delayed hatching	
	reduced fish density	
10	0-20% mortality	Moderately severe habitat degradation
	increased predation	
	moderate to severe habitat degradation	
11	>20-40% mortality	
12	>40-60% mortality	Severe habitat degradation
13	>60-80% mortality	
14	>80-100% mortality	Catastrophic or total destruction of habitat in the receiving environment

Table 4-3. Some Reported Effects of Turbidity and Suspended Sediment Concentrations on Salmonids outside Alaska (Bash et al. 2001, updated from Lloyd 1987).

Effect	Species[a] (life stage)	*Location*	Reported turbidity[b] or suspended sediment concentration	Reference
Fatal (96-h LC50)	Coho salmon (juveniles)	Washington	1,200 mg/l	Noggle (1978)
Fatal (96-h LC50)	Coho salmon (juveniles)	Washington	509; 1,217 mg/l	Stober et al. (1981)
Fatal (96-h LC50)	Chinook salmon (juveniles)	Washington	488 mg/l	Stober et al. (1981)
Reduced survival (marked)	Chum salmon (eggs)	British Columbia	97 mg/l	Langer (1980)
Reduced survival (marked)	Rainbow trout (eggs)	Great Britain	110 mg/l	Scullion and Edwards (1980)
Reduced survival (marked)	Rainbow trout (eggs)	Oregon	1,000-2,500 ppm	Campbell (1954)
Reduced survival (marked)	Rainbow trout (juveniles)	Great Britain	270 ppm	Herbert and Merkens (1961)
Reduced survival (marked)	Rainbow trout (juveniles)	Great Britain	200 ppm	Herbert and Richards (1963)
Reduced survival (marked)	Rainbow trout (juveniles)	Oregon	1,000-2,500 ppm	Campbell (1954)
Reduced survival (marked)	Rainbow trout (juveniles)	Great Britain	90 ppm	Herbert and Merkens (1961)
Reduced survival (marked)	Coho salmon (juveniles)	Pennsylvania	6; 12 mg Fe/l (15-27 JTU)	Smith and Sykora (1976)
Reduced survival (marked)	Coho salmon (adults)	Washington	1,400-1,600 mg/l	Stober et al. (1981)
Reduced abundance (marked)	Brown trout	Great Britain	1,000; 6,000 ppm	Herbert et al. (1961)
Reduced abundance (marked)	Lake trout	Northwest Territories	<10 FTU	McCart et al. (1980)
Reduced growth (marked)	Brook trout (juveniles)	Pennsylvania	50 mg Fe/l (86 JTU)	Sykora et al. (1972)
Reduced growth (slight)	Brook trout (juveniles)	Pennsylvania	12 mg Fe/l (32 JTU)	Sykora et al. (1972)
Reduced growth (slight)	Rainbow trout (juveniles)	Great Britain	50 ppm	Herbert and Richards (1963)
Reduced growth	Coho salmon (juveniles)	Idaho	25 NTU	Sigler et al. (1984)
Reduced growth (marked)	Arctic grayling (juveniles)	Yukon	1,000 mg/l	McLeay et al. (1984)
Reduced growth (slight)	Arctic grayling (juveniles)	Yukon	100; 300 mg/l	McLeay et al. (1984)

a Arctic grayling (*Thymallus arcticus*)
Brook trout (*Salvelinus fontinalis*)
Brown trout (*Salmo trutta*)
Chinook salmon (*Oncorhynchus tshawytscha*)
Chum salmon (*Oncorhynchus keta*)
Coho salmon (*Oncorhynchus kisutch*)
Cutthroat trout (*Salmo clarki*)
Lake trout (*Salvelinus namaycush*)
Rainbow trout (*Salmo gairdneri*)
Steelhead (anadromous *S. gairdneri*)

b Formazin (FTU), Jackson (JTU), and nephelometric (NTU) turbidity units.

c Information not available.

Table 4-3. Some Reported Effects of Turbidity and Suspended Sediment Concentrations on Salmonids outside Alaska (Bash et al. 2001, updated from Lloyd 1987) (continued).

Effect	Species[a] (life stage)	*Location*	Reported turbidity[b] or suspended sediment concentration	Reference
Reduced food conversion	Rainbow trout (juveniles)	Arizona	< 70 JTU	Olson et al. (1973)
Reduced feeding (cessation)	Coho salmon (juveniles)	Washington	300 mg/l	Noggle (1978)
Reduced feeding	Coho salmon (juveniles)	Washington	100 mg/l	Noggle (1978)
Reduced feeding	Coho salmon (juveniles)	British Columbia	10-60 NTU	Berg (1982), Berg and Northcote (1985) Bachmann (1958)
Reduced feeding (cessation)	Cutthroat trout	Idaho	35 ppm	Bachmann (1958)
Reduced feeding	Brown trout	Pennsylvania	7.5 NTU	Bachman (1984)
Reduced feeding	Rainbow trout (juveniles)	Arizona	70 JTU	Olson et al. (1973)
Reduced feeding	Arctic grayling (juveniles)	Yukon	100; 300; 1,000 mg/L	McLeay et al. (1984)
Reduced condition factor	Rainbow trout (juveniles)	Great Britain	110 mg/l	Scullion and Edwards (1980)
Altered diet (terrestrial instead of aquatic)	Rainbow trout (juveniles)	Great Britain	110 mg/l	Scullion and Edwards (1980)
Stress (increased plasma cortisol, hematocrit, and susceptibility to pathogens)	Coho salmon (juveniles) Steelhead (juveniles)	Oregon	500 mg/l 2,000 mg/l	Redding and Schreck (1980)
Stress (increased metabolic rate, susceptibility to toxicants)	Arctic grayling	Yukon	300 mg/l	McLeay et al. (1984)
Stress (increased plasma glucose)	Arctic grayling (juveniles)	Yukon	50 mg/l	McLeay et al. (1983)
Stress (respiratory distress)	Coho salmon (juveniles)	Pennsylvania	6; 12 mg Fe/l (15-27 JTU)	Smith and Sykora (1976)
Stress (increased ventilation)	Brook trout	Lake Superior	231 NTU	Carlson (1984)
Disease (fin rot)	Rainbow trout (juveniles)	Great Britain	270 ppm	Herbert and Merkens (1961)
Disease (fin rot)	Rainbow trout (juveniles)	Great Britain	100; 200 ppm	Herbert and Merkens (1961)

a Arctic grayling (*Thymallus arcticus*)
Brook trout (*Salvelinus fontinalis*)
Brown trout (*Salmo trutta*)
Chinook salmon (*Oncorhynchus tshawytscha*)
Chum salmon (*Oncorhynchus keta*)
Coho salmon (*Oncorhynchus kisutch*)
Cutthroat trout (*Salmo clarki*)
Lake trout (*Salvelinus namaycush*)
Rainbow trout (*Salmo gairdneri*)
Steelhead (anadromous *S. gairdneri*)

b Formazin (FTU), Jackson (JTU), and nephelometric (NTU) turbidity units.

c Information not available.

Table 4-3. Some Reported Effects of Turbidity and Suspended Sediment Concentrations on Salmonids outside Alaska (Bash et al. 2001, updated from Lloyd 1987) (continued).

Effect	Species[a] (life stage)	*Location*	Reported turbidity[b] or suspended sediment concentration	Reference
Avoidance	Chinook salmon (adults)	California	"Natural turbidity"	Sumner and Smith (1940)
Avoidance	Chinook salmon (adults)	Washington	650 mg/l	Whitman et al. (1982)
Avoidance	Chinook salmon (adults)	Washington	350 mg/l	Brannon et al. (1981)
Avoidance (sensitivity)	Lake trout	Lake Superior	6 FTU	Swenson (1978)
Avoidance	Coho salmon (juveniles)	Washington	70 NTU	Bisson and Bilby (1982)
Avoidance	Coho salmon, steelhead (juveniles)	Idaho	22-265 NTU	Sigler (1980), Sigler et al. (1984)
Displacement	Coho salmon, steelhead (juveniles)	Idaho	40-50 NTU	Sigler (1980)
Displacement	Arctic grayling (juveniles)	Yukon	300; 1,000 mg/l	McLeay et al. (1984)
Displacement	Rainbow trout (juveniles)	Great Britain	110 mg/l	Scullion and Edwards (1980)
Altered behavior (feeding)	Trout	c	25 JTU	Langer (1980)
Altered behavior (less use of overhead cover)	Brook trout	Wisconsin	7 FTU	Gradall and Swenson (1982)
Altered behavior (visual)	c	c	25-30 JTU	Bell (1984)
Altered behavior (visual)	Coho salmon (juveniles)	British Columbia	10-60 NTU	Berg (1982), Berg and Northcote (1985)
Altered behavior (loss of territoriality)	Coho salmon (juveniles)	British Columbia	10-60 NTU	Berg (1982), Berg and Northcote (1985)
Altered behavior (listlessness)	Coho salmon (juveniles)	Pennsylvania	6; 12 mg Fe/l (15-27 JTU)	Smith and Sykora (1976)
Change in body color	Arctic grayling (juveniles)	Yukon	300; 1,000 mg/l	McLeay et al. (1984)
Change in body color	Coho salmon (juveniles)	Pennsylvania	6; 12 mg Fe/l (15-27 JTU)	Smith and Sykora (1976)
Reduced tolerance to saltwater	Chinook salmon (juveniles)	Washington	3,109 mg/l	Stober et al. (1981)

a Arctic grayling (*Thymallus arcticus*)
Brook trout (*Salvelinus fontinalis*)
Brown trout (*Salmo trutta*)
Chinook salmon (*Oncorhynchus tshawytscha*)
Chum salmon (*Oncorhynchus keta*)
Coho salmon (*Oncorhynchus kisutch*)
Cutthroat trout (*Salmo clarki*)
Lake trout (*Salvelinus namaycush*)
Rainbow trout (*Salmo gairdneri*)
Steelhead (anadromous *S. gairdneri*)

b Formazin (FTU), Jackson (JTU), and nephelometric (NTU) turbidity units.

c Information not available.

Table 4-4. Some Reported Effects of Turbidity and Suspended Sediment Concentrations on Salmonids: 2001 Update (Bash et al. 2001, updated from Lloyd 1987).

Effect	Species (life stage)	Location	Reported turbidity or suspended sediment concentration	Reference
Activity	Creek Chubs, Brook Trout	Wisconsin	Increase in moderately turbid waters	Gradall and Swenson (1982)*
Avoidance	Coho salmon (underyearling)	British Columbia	After 60 NTU pulse, fish move to substrate	Berg (1982)*
Avoidance	Coho salmon (underyearling)	British Columbia	Approx 25% at 7,000 mg/l – estimated that the threshold for avoidance in the vertical plane was 37 NTU	Servizi and Martens (1992)*
Avoidance	Creek Chubs	Wisconsin	Preferred 56.6 FTU	Gradall and Swenson (1982)*
Blood Sugar	Coho salmon (underyearling)	British Columbia	Elevated, proportional to SS exposure	Servizi and Martens (1992)*
Capture success per strike	Coho salmon (juvenile)	British Columbia	30 and 60 NTU	Berg and Northcote (1985)*
Cough Frequency	Coho salmon (underyearling)	British Columbia	Elevated eightfold over control levels at 240 mg/l	Servizi and Martens (1992)*
Feeding rates	Pacific herring (larval stage)	Oregon	Maximum feeding potential at 500 and 1000 mg/l	Boehlert and Morgan (1985)*
Feeding rates	Coho salmon (juvenile)	British Columbia	Prey consumption only 35% of feeding in clear water at 60 NTU	Berg (1982)*
Feeding rates	Coho salmon and steelhead (yearlings)	Oregon	When exposed to 2,000-3,000 mg/l of topsoil, kaolin clay, volcanic ash, 7-8 days	Redding et al. (1987)*
Feeding rates	Chinook salmon (juvenile)	British Columbia	Reduced at higher turbidities, highest rates at intermediate turbidity 35-150 NTU for surface and benthic prey	Gregory and Northcote (1993)*

* laboratory study
** field study

Table 4-4. Some Reported Effects of Turbidity and Suspended Sediment Concentrations on Salmonids: 2001 Update (Bash et al. 2001, updated from Lloyd 1987) (continued).

Effect	Species (life stage)	Location	Reported turbidity or suspended sediment concentration	Reference
Feeding rates	Chinook salmon (juvenile)	British Columbia	Increased rates on surface and benthic prey in conditions of moderate turbidity (18-150 NTU) compared with lower (<1 NTU) or higher 370-810 NTU	Gregory (1992)*
Feeding rates	Chinook salmon (juvenile)	British Columbia	Above 150 NTU, juvenile chinook exhibit reduced feeding regardless of prey type and forager size	Gregory (1992)*
Feeding rates	Bluegills	North Carolina	14 prey per minute in clear water to 1, 10, 7 per minute in pools of 60, 120, and 190 NTU. Size selectivity independent	Gardner (1981)*
Gill trauma	Sockeye salmon (underyearling)	British Columbia	3,148 mg/l or 0.2 of the 96 h LC50 Value	Servizi and Martens (1987)*
Homing	Chinook salmon (adult)	Washington	Strong baseline preference for clean (ash-free) home water over a clean non-natal water source	Whitman et al. (1982)**
Impairment in hypo-osmoregulatory capacity	Sockeye salmon (underyearling)	British Columbia	Exposed 96 h to 14,407 mg/l of fine sediment	Servizi and Martens (1987)*
Percentage of prey ingested	Coho salmon (juvenile)	British Columbia	30 and 60 NTU	Berg and Northcote (1985)*
Plasma glucose increase	Sockeye salmon (underyearling)	British Columbia	Increased 150 and 39% from exposure to 1,500 and 500 mg/l of fine sediment	Servizi and Martens (1987)*

* laboratory study
** field study

Table 4-4. Some Reported Effects of Turbidity and Suspended Sediment Concentrations on Salmonids: 2001 Update (Bash et al. 2001, updated from Lloyd 1987) (continued).

Effect	Species (life stage)	Location	Reported turbidity or suspended sediment concentration	Reference
Predation rates	Chinook salmon (juvenile), chum, sockeye, cutthroat trout	British Columbia	Mean predation rates were 10-75% lower than those in controls (no vegetation and clear water); addition of turbidity reduced effect	Gregory and Levings (1996)*
Predator avoidance	Chinook salmon (juvenile)	British Columbia	In absence of risk, juvenile chinook were distributed randomly in 23 NTU, at bottom in clear water– with risk, all at bottom, and responses less marked and of shorter duration	Gregory (1993)*
Prey abundance	N/A	Columbia River Estuary	Reduction in amphipods in substrate with surface layer of ash	Brzezinski and Holton (1981)**
Prey abundance	N/A	Northwest Territories	Sediment addition increased total drift of invertebrates (avoidance reaction)	Rosenberg and Wiens (1978)**
Reaction distance	Coho salmon (juvenile)	British Columbia	30 and 60 NTU	Berg and Northcote (1985)*
Reaction distance	Chinook salmon (juvenile)	British Columbia	Decline with increasing turbidity	Gregory and Northcote (1993)*

* laboratory study

** field study

Table 4-4. Some Reported Effects of Turbidity and Suspended Sediment Concentrations on Salmonids: 2001 Update (Bash et al. 2001, updated from Lloyd 1987) (continued).

Effect	Species (life stage)	Location	Reported turbidity or suspended sediment concentration	Reference
Reaction distance	Adult lake trout	Utah	Reaction distance increased w/ increasing light - <25 cm at .17 1x to about 100 cm at light threshold of 17.8 1x., declined with turbidity - > 80% of decline in reaction distance occurred over 0-5 NTU	Vogel and Beauchamp (1999)*
Reactive Distance	Rainbow Trout	Georgia	Reactive distances in 15 and 30 NTU treatments were only 80 and 45% respectively of those observed at ambient turbidities 4-6 NTU.	Barrett and Rosenfeld (1992)*
Reduced Growth	Coho salmon (juvenile)	Oregon	Significant decrease in fish production when fine sediments were 26-31% by volume	Crouse et al. (1981)*
Reduction in prey	Chinook salmon (juvenile)	Washington	Reduced appearance of highly utilized amphipod Corophium salmonis.	McCabe et al. (1981)**
Relation of turbidity and suspended solids	N/A	Alaska	Depth to which 1% of subsurface light penetrates has inverse correlation with sediment-induced turbidity	Lloyd et al. (1987)**
Stress (Gill Flaring)	Coho salmon (juvenile)	British Columbia	Increased at 30 and 60 NTU	Berg and Northcote (1985)*

* laboratory study
** field study

Table 4-4. Some Reported Effects of Turbidity and Suspended Sediment Concentrations on Salmonids: 2001 Update (Bash et al. 2001, updated from Lloyd 1987) (continued).

Effect	Species (life stage)	Location	Reported turbidity or suspended sediment concentration	Reference
Stress (increased plasma cortisol)	Coho salmon and steelhead (yearlings)	Oregon	When exposed to 2-3 g/L of topsoil, 7-8 days	Redding et al. (1987)*
Stress (blood hematrocrits and plasma cortisol)	Coho salmon and steelhead (yearlings)	Oregon	Increased in fish exposed to high concentrations for two days, topsoil, kaolin clay, or ash.	Redding et al. (1987)*
Stress (resistance to bacterial pathogen)	Yearling steelhead and coho	Oregon	Vibrio anguillarum	Redding et al. (1987)*
Territoriality	Coho salmon (juvenile)	British Columbia	Territoriality ceases with 60 NTU pulse – re-established at 20 NTU – lateral displays minimized	Berg (1982)*

* laboratory study

** field study

Table 4-5. Summary of Suspended Sediment Effects on Selected Salmonids Commonly Present in the Yakima River Basin (Bash et al. 2001, from Newcombe and McDonald 1991).

Species	Concentration (mg/1)	Duration (hours)	Effect
Chinook Salmon	1400*	36	10% mortality of juveniles
	488	96	50% mortality of smolts
	82,000	6	60% mortality of juveniles
	19,364	96	50% mortality of smolts
	1.5-2.0	1,440	Gill hyperplasia, poor condition of fry
	6	1,440	Reduction in growth rate
	75	168	Harm to quality of habitat
	84	336	Reduction in growth rate
	1,547	96	Histological damage to gills
	650	1	Homing performance disrupted
Whitefish	16,613	96	50% mortality of juveniles
	.7	1	Overhead cover abandoned
Salmon (general)	8	24	Sport fishing declines
Steelhead	84	336	Reduction in growth rate
Rainbow Trout	19,364	96	50% mortality of smolts
	157	1728	100% mortality of eggs
	21	1152	62% reduction in egg to fry survival
	37	1440	46% reduction in egg to fry survival
	7	1152	17% reduction in egg to fry survival
	90	456	5% mortality in sub-adults
	171	96	Histological damage
	50	1848	Reduction in growth rate
	100	1	Avoidance response

Compiled by the Washington State Department of Ecology for “A Suspended Sediment and DDT Total Maximum Daily Load Evaluation Report for the Yakima River.”

(*) indicates estimated concentation

The paper recognizes potential limitations associated with extrapolation of study findings from the laboratory to the field. Most experiments fail to report key information on spatial and temporal factors that may influence interpretation of the data (e.g., distribution, abundance and availability of suitable habitat, time of year, frequency, duration, and magnitude of events, cumulative or synergistic effects).

4.5 DAVIES-COLLEY AND SMITH 2001

Davies-Colley and Smith (2001) provide a thorough review, explanation and discussion of the relationships among turbidity, suspended sediment and water clarity, as well as an overview of light properties that affect measurements (discussed in greater detail in Section 2). A comprehensive discussion of the technical basis for nephelometric turbidity measurements is presented along with explanations of their limitations as measures of water clarity and suspended sediment. There is a summary of important physical properties of suspended particles (e.g., size, shape, composition and angularity) as they effect turbidity and water clarity measurements. The problems encountered when attempting to relate turbidity and suspended sediment measures are discussed. This paper makes several recommendations regarding the type of water clarity measurements appropriate to the nature of the study or for establishing environmental standards, as follows:

- When the optical effect of the suspended sediment is of primary concern to water resource and fishery managers, light penetration or visual clarity is usually the most appropriate measures rather than SSC.
- Turbidity measures may be appropriate where a relative index of water cloudiness is sufficient, or if there is some inherent advantage of a laboratory assay.
- Ideally, environmental standards should not be based on nephelometric turbidity, due to the general lack of correlation with suspended matter, although site-specific conditions may justify its use. Turbidity is not an absolute scientific measure and needs to be calibrated to an objective measure such as visual clarity to be meaningful.
- Davies-Colley and Smith recommend formulation of environmental water quality standards in terms of visual water quality, recognizing its environmental relevance and significant practical advantages over both SSC and turbidity.

4.6 NEWCOMBE 2003

While not technically a review paper, this resource provides further refinement of the relationship between turbid water and effects on fish presented in Newcombe and Jensen (1996). The empirical model presented in Newcombe (2003) has several modifications from previous work. The model is presented as an assessment tool for impacts to clear water fishes where loss of visual clarity is the primary mode of harmful effect in systems that are relatively clear and relatively free of excessive suspended sediment (e.g. where black disk sighting range exceeds 0.55 m). It has been developed as a single table to estimate Severity-of-ill-effect Scores (SEV) on rearing success of clear water fish (Table 4-6). The table provides an estimate of the level of effect in relation to the level and duration of exposure. This model also includes an assessment of impact to juvenile and adult life history phases through calibration for reactive distance of trout.

This empirical model relates water cloudiness as a function of reduced visual clarity and duration of exposure in the SEV table (Table 4-6). The 15-point SEV scores are set in the table cells along the same logarithmic scale of hours on the X-axis and an expanded logarithmic scale calibrated to black disk sighting distance in meters along the Y-axis. Black disk sighting distance or beam attenuation is recommended as the preferred method to measure visual clarity. Alternative scales of Secchi disk sighting range and turbidity in NTUs are also provided since these methods have been frequently used for measurement in the field. The expansion of the vertical axis allows for some refinement of the SEV scaling within the table.

4.7 META-ANALYSES AND MANAGEMENT

The meta-analyses of Newcombe and MacDonald (1991), Newcombe and Jensen (1996), and Newcombe (2003) synthesized much of the existing data on the effects of sediment on fishes. These analyses were used to develop a semi-quantitative series of SEV related to SSC and loss of visual clarity with duration of exposure (Tables 4-1, 4-2, and 4-6).

Newcombe and Jensen (1996) constructed matrices from existing literature and reports to develop models to predict probable SEV to fish based on taxon, life history stage, sediment concentration, and duration of exposure. From these matrices and accompanying statistics they derived a function of the general form:

$$z = a + b(\log_e x) + c(\log_e y)$$

> where: z = calculated SEV, a = y-intercept, b and c = slope coefficients, x = estimated duration of exposure to sediments (hours), and y = concentration of the (estimated) predominant suspended sediment size (mg SS/L).

They then calculated SEV for a series of sediment concentrations and exposure durations to yield profiles of the potential effects of significant suspended sediments on various fish taxa and size/age cohorts.

The calculated threshold values of SSCs and durations of exposure above which sublethal (SEV = 3) and lethal (SEV = 8) effects occur on the various fish cohorts varied somewhat between adult and juvenile salmonids. In general, sublethal effects (SEV 4-8) began to appear with only one-hour exposure to sediment concentrations of only 20-55 mg/L, but lethal and "paralethal" effects (SEV 9-14) at one hour of exposure did not appear until sediment concentrations reached 8,100-22,000 mg/L (Table 4-1). Adult salmonids were actually somewhat more sensitive to short-term exposure than were juveniles--calculated

Table 4-6. Impact Assessment Model for Clear Water Fishes Exposed to Conditions of Reduced Water Clarity. A Model Based on Literature Reports and Consultation with Scientific Experts to Estimate Severity of Impact on Rearing Success of Clear Water Fish as a Function of Reduced Visual Clarity of Water (m) and Duration of Exposure (h), for Juvenile and Adult Life History Phases, Including Calibration for Reactive Distance of Trout (Newcombe 2003).

Visual clarity of water (*y* BD) and related variables				Duration of exposure to conditions of reduced VISUAL CLARITY ($\log_e$) hours											Fish reactive distance:	
alternate		*preferred*		0	1	2	3	4	5	6	7	8	9	10	*calibrated for trout*	
NTU	*z* SD (m)	BA (m^{-1})	*y* BD (m)	Severity-of-ill-effect Scores (SEV) - Potential SEV = -4.49 + 0.92 ($\log_e$ h) - 2.59 ($\log_e$ yBD)											ψ_{BD} (cm)	*x* RD (cm)
1100	0.01	500	0.010	7	8	9	10	11	12	13	14				1	-
			0.014	7	7	8	9	10	11	12	13	14			1	-
400	0.03	225	0.020	6	7	7	8	9	10	11	12	13	14		2	-
			0.030	4	5	6	7	8	9	10	11	12	13	14	3	-
150	0.07	100	0.050	3	4	5	6	7	8	9	10	11	12	13	5	-
			0.070	2	3	4	5	6	7	8	9	10	11	11	7	-
55	0.05	45	0.110	1	2	3	4	5	6	7	8	9	10	10	11	6
			0.160	0	1	2	3	4	5	6	7	8	9	9	16	17
20	0.34	20	0.240	0	0	1	2	3	4	5	6	7	8	8	24	30
			0.360	0	0	0	1	2	3	4	5	6	6	7	36	42
7	0.77	9	0.550	0	0	0	0	1	2	3	4	5	5	6	55	55
			0.770	0	0	0	0	0	1	2	3	4	4	5	77	66
3	1.53	4	1.090	0	0	0	0	0	0	1	2	3	4	5	109	77
			1.690	0	0	0	0	0	0	0	1	2	2	3	169	90
1	3.68	2	2.630	0	0	0	0	0	0	0	0	0	1	2	263	104
Duration of exposure				1	3	7	1	2	6	2	7	4	11	30		
				Hours			Days			Weeks		Months				

Key: **yBD** = Black disc sighting range (m): horizontal measurement in water of any depth (reciprocal of beam attenuation)

y_{BD} = Black disc sighting range (cm): a convenient calibration for measurements made in very cloudy water.

BA = Beam attenuation (m-1): measures absorption and scattering of light by "water constituents" –clay and color; reciprocal of black disc sighting range.

$_Z$SD = Secchi disk sighting range (m): a vertical measurement, usually in deep water.

$_X$RD = Reactive distance of adult trout (pooled data for rainbow, lake, and brook trout) to fish prey as a function of visual clarity.

NTU = Nephelometric turbidity units: a measure of light scattering by suspended clay particles (0.2 to 5 mm diameter).

SEV = Severity of Ill Effect scale: 0≤nil effect≤0.5 (lower left diagonal line, nil effects threshold); 0.5≤minor≤3.5; (next diagonal line to the right, minor effects threshold) 3.5≤moderate≤8.5 (next diagonal line to the right, sublethal effects threshold); 8.5≤severe≤14.5 (next diagonal line to the right, lethal and paralethal effects threshold).

paralethal effects after one hour were reached at a SSC concentration of about 22,000 mg/L for adult salmonids (Table 4-1), but for juveniles were not reached until concentrations reached almost 60,000 mg/L (Table 4-1). However, when the adult and juvenile data were pooled the threshold concentration for one hour of exposure rose to about 60,000 mg/L (Table 4-1). Early life history stages (egg development through young juveniles) are more susceptible than older juveniles and adults. For adult and juvenile salmonids combined, a one-day exposure of about 3,000 mg/L SSC reached lethal values (Table 4-1), and only two days exposure to just 7.0 mg/L suspended sediments was lethal to salmonid eggs and larvae (Table 4-1).

Anderson (1996) adapted the Newcombe and Jensen (1996) model to predict the effects of various sediment concentrations and exposure durations on aquatic habitat and habitat selection by aquatic organisms. He used the same meta-analytic approach to relate the Newcombe and Jensen (1996) SEV scale to previously published studies. He developed a simplified "severity of ill effects" (SE) scale of habitat impacts (Table 4-2), with the intent of determining the relative importance of sediment concentration and duration of exposure in habitat impacts. By multiple regression techniques similar to those employed by Newcombe and Jensen (1996), Anderson (1996) showed that these factors affect the extent of habitat alteration in dissimilar ways (because each factor has a different slope).

Newcombe (2003) modified the Newcombe and Jensen (1996) SEV ranking model so that it could be used for impact assessment in clear water systems where the ill effects of interest are caused by changes in visual water clarity. The model also incorporates fish "reactive distance," a key variable in predator-prey interactions. Another key variable, compensation depth (the depth where attenuation of sunlight reduces the rate of photosynthesis to levels equivalent to the rate of respiration, the depth limit of green plant life), was not incorporated in the model because the assumption that compensation depth varies inversely with water cloudiness is not universally applicable (Davies-Colley and Smith 2001). Newcombe (2003) developed this model largely through consultation with peers and limited synthesis of published data. Although the model describes a relationship between visual clarity and biological effects on clear water fishes (for which data were available in sufficient quantity), relatively few of the studies that support this model directly address direct measurement of water clarity as recommended by Davies-Colley and Smith (2001). Rather most studies address effects of suspended sediment concentrations and to a lesser extent, nephelometric turbidity measurements.

The modification used to evaluate visual water clarity takes the general form:

$$z = a + b(\log_e x) + c(\log_e y\mathrm{BD})$$

where: z = calculated SEV, a = y-intercept, b and c = slope coefficients, x = estimated duration of exposure to sediments (hours), and yBD = Black Disk reading.

The 15-level SEV scale is the same as used in the Newcombe and Jensen (1996) model (Table 4-2). The Newcombe (2003) model (Table 4-6) develops a diagonal exposure and duration matrix similar to those specified for the Newcombe and Jensen (1996) model (Table 4-1). The model was designed to accomplish the following:

> *"…(a) identifies the threshold of the onset of ill effects among clear water fishes; (b) postulates the rate at which serious ill effects are likely to escalate as a function of reduced visual clarity and persistence; (c) provides a context (the "visual clarity" matrix, with its cell coordinates) to share and compare information about impacts as a function of visual clarity "climate;" (d) demonstrates changes in predator prey interactions at exposures greater than and less than the threshold of ill effects (e) calibrates trout reactive distance (cm) as a function of water clarity in the form*
>
> $$y = a + b\,ln(x)$$
>
> *where: y represents reactive distance (cm) and x represents visual clarity (black disk sighting range, cm) and a and b are intercept and slope respectively, such that*
>
> $$y = -68.0546 + 30.8307\ ln(x);$$
>
> *(f) identifies the black disk sighting range, in meters, and its reciprocal, beam attenuation, as preferred monitoring variables, and (g) provides two additional optical quality variables (Secchi disk extinction distance and turbidity) which, suitably calibrated as they have been in this study, expand the range of monitoring options in situations in which the preferred technology – beam attenuation equipment or black disk sighting equipment – is unavailable or impractical to use." (Newcombe 2003 p. 529)*

Not unexpectedly, this model matrix relates turbidity exposure to black disk assessment in a pattern very similar to the matrices generated by the Newcombe and Jensen (1996) model for SSC.

The information presented Newcombe and Jensen (1996), Anderson (1996), and Newcombe (2003) serves best as a set of indicators and guidelines, not as a rigid set of management tools *per se*.

This section reviews literature that evaluates direct physiological and behavioral effects of turbidity (including water clarity) and suspended sediment on aquatic invertebrates. It also reviews studies that address indirect effects that occur through changes in primary production (trophic effects). Studies that evaluate invertebrate community responses are summarized.

In many of these studies, the effects of turbidity and suspended sediment are not always distinguished from the effects of sediment deposition. A cursory review of literature related to sedimentation is provided in Appendix A.

5.1 PHYSIOLOGICAL EFFECTS

Suspended sediment and turbidity both can have direct adverse effects on benthic macroinvertebrates. The effects of suspended sediment and turbidity are not as well-documented as the effects of deposited sediment, but the information that is available suggests that the effects of deposited sediment are generally more severe and long-lasting. Even so, suspended sediment and turbidity effects may be significant by causing stress that reduces feeding, growth and reproductive abilities. Inorganic sediment may accumulate on body surfaces and respiratory structures (gills), thus causing respiratory stress (Lemly 1982). In turn, this causes a disruption in feeding efficiency, particularly for filter feeding organisms, and may result in reduced growth or mortality. A loss in visual efficiency during feeding also may occur, although this has not been proven (Waters 1995). Invertebrates that inhabit exposed streambeds are subject to scouring by velocities associated with high flows, which can damage their integument (protective, shell-like covering) and their respiratory organs (Newcombe and MacDonald 1991). Newcombe and MacDonald (1991) summarize data on the effects of suspended sediment on aquatic invertebrates using a Ranking developed by multiplying the level or exposure with the duration of exposure, then developing a 14 point SEV scale. This is a scoring system that is similar in many respects to the approach Newcombe and Jensen (1996) and Newcombe (2003) discussed previously. However, it has not been developed further in more recent literature. The summary provided by Newcombe and MacDonald (1991) still provides a useful collection of effect studies (Table 5-1).

5.2 BEHAVIORAL EFFECTS

One of the most important behavioral effects on aquatic invertebrates associated with suspended sediment is an increase in their drift density. Drift is the downstream transport of aquatic insects with the current (Cereghino et al. 2004). Drift is a natural phenomenon that is assumed to be an active behavioral process allowing the regulation of benthic

Table 5-1. Summary of Data on the Effects of Suspended Sediment on Aquatic Invertebrates (Newcombe and MacDonald 1991).

Taxon	Exposure C	Exposure D	Stress index ($\log_e$ [C × D])	Effect	Rank of effect	Source
Zooplankton	24[a]	0.15	1.281	Reduced capacity to assimilate food	4	McCabe and O'Brien (1983)
Benthic invertebrates	8	2.5	2.996	Lethal: increased rate of drift	10	Rosenberg and Wiens (1978)
Macro invertebrates	53–92	24[a]	7.462	Lethal: reduction in population size	10	Gammon (1970)
Benthic invertebrates	1,700	2	8.132	Lethal: alteration in community structure and drift patterns	10	Fairchild et al. (1987)
Zoobenthos	10–15	720[a]	9.105	Lethal: reduction in standing crop	10	Rosenberg and Snow (1977)
Benthic invertebrates	8	1,440	9.352	Lethal: up to 50% reduction in standing crop	12	Rosenberg and Wiens (1978)
Cladocera	82–392	72[a]	9.745	Lethal: survival and reproduction harmed	12	Robertson (1957); from Alabaster and Lloyd (1982)
Benthic fauna	29	720[a]	9.947	Lethal: populations of Trichoptera, Ephemeroptera, Crustacea, and Mollusca, disappear	14	M.P. Vivier, personal communication in Alabaster and Lloyd (1982)
Benthic invertebrates	16	1,440	10.045	Lethal: reduction in standing crop	12	Slaney et al. (1977b)
Cladocera and Copepoda	300–500	72	10.268	Lethal: gills and gut clogged	14	Stephan (1953) cited in Alabaster and Lloyd (1982)
Benthic invertebrates	32	1,440	10.738	Lethal: reduction in standing crop	12	Slaney et al. (1977b)
Zoobenthos	>100	672[a]	11.115	Lethal: reduction in standing crop	12	Rosenberg and Snow (1977)
Benthic invertebrates	62	2,400	11.910	Lethal: 77% reduction in population size	13	Wagener and LaPerriere (1985)
	77	2,400	12.127	Lethal: 53% reduction in population size	12	Wagener and LaPerriere (1985)
Bottom fauna	261–390	720[a]	12.365	Lethal: reduction in population size	12	Tebo (1955)
Benthic invertebrates	390	720[a]	12.545	Lethal: reduction in population size	12	Tebo (1955)
	278	2,400	13.411	Lethal: 80% reduction in population size	13	Wagener and LaPerriere (1985)
Stream invertebrates	130[b]	8,760	13.945	Lethal: 40% reduction in species diversity	14	Nuttall and Bielby (1973)
Benthic invertebrates	743	2,400	14.394	Lethal: 85% reduction in population size	14	Wagener and LaPerriere (1985)
	5,108	2,400	16.322	Lethal: 94% reduction in population size	14	Wagener and LaPerriere (1985)
Stream invertebrates	25,000[b]	8,760	19.204	Lethal: reduction or elimination of populations	14	Nuttall and Bielby (1973)

[a] Estimated.
[b] China clay.

production and downstream colonization (Cereghino et al. 2004). This behavior also shows a diel periodicity, with higher drift density occurring during the night than the day and larger individuals entering the drift at night.

Some natural or man-made disturbances, such as very sudden increases in flow (up to and including those flows which may mobilize a substantial portion of the bedload) or a large sediment influx, can cause 'catastrophic drift'. This sudden and large-scale drift may be the result of dislodgment caused by rapid flow increases or by triggered avoidance behavior by the organisms due to the effects of transported sediment. Other flow increases or sediment movements may cause less dramatic increases in drift rates. Invertebrates also can become dislodged due to the effect of rolling or saltating particles. Culp et al. (1986) concluded that saltating sediments were the primary factor causing a reduction in benthic densities of more than 50 percent in 24 hours in their field study of fine sediment additions. They also found that there was both a distinct immediate effect (in the form of increased drift) and delayed responses. Macroinvertebrates having a delayed response were initially present below the surface but became exposed to sediment effects during their vertical shift in distribution 6 to 9 hours after the sediment additions. The abundance and composition of the benthic assemblage is likely to be altered due to differing responses of species based on differences in the sizes and tolerances of the species and life stages present (Cereghino et al. 2004). These effects will vary with season, reflecting differences in life stages and species present.

Rosenberg and Weins (1978) performed an instream study of the effects of sediment additions on benthic invertebrates during two periods. They constructed a study channel and a control channel and added sediment to the study channel for five hours at a rate that produced a SSC of 7.76 mg/l (August) and 7.42 mg/l (September). Flows were constant during the experiments and slightly lower in the study channel. The addition of sediment caused the number of invertebrates drifting from the study channel to increase 3-fold in August and 2-fold in September (Table 5-2). Generally, sediment addition caused higher numbers of Oligochaeta and Simuliidae to drift during both study periods. During August a higher proportion of Hydracarina and Chironomidae drifted and during September a higher proportion of Plecoptera and Ephemeroptera drifted. Differences observed between the two study periods are probably due to the normal seasonal differences in taxon composition. The authors concluded that measures of settled sediment rather than suspended sediment would have a more direct relationship with invertebrate response.

Net making species could be affected by fouling or ripping of their nets (Strand and Merritt 1997). In turn, this could decrease food acquisition and result in a reduction in adult reproductive success due to time and energy costs. Strand and Merritt (1997) studied the effects of daily exposure to moderate levels of sediment on the net-spinning Trichoptera *Hydropsyche betteni* and *Ceratopsyche sparna*. Nets became clogged with sediment and were cleaned or replaced by the organisms each day following exposure. Although this study was focused on the effects of deposited sediment (Appendix A), the effect of clogging the caddisfly nets is also a suspended sediment effect. The sediment

Table 5-2. Total Numbers of Macrobenthic Invertebrates Drifting in 5 h from Experimental Channels in 1 August and 9 September, 1974 Sediment Additions (Rosenberg and Wiens 1978).

		Control Channel		Sediment Additional Channel	
Date	Sampling periods	Number of Invertebrates Drifting	Number of Invertebrates per m^2 Drifting	Number of Invertebrates Drifting	Number of Invertebrates per m^2 Drifting
1 August	1 & 2	146.6	9.8	925.9	61.7
	3 & 4	2200	14.7	493.2	329
	5 & 6	200.0	13.3	420.0	280
		Σ = 566.6	37.8	1839.1	1226
9 September	1 & 2	196.6	13.1	1781.0	118.7
	3 & 4	426.8	28.5	526.8	35.1
	5 & 6	726.8	48.5	753.2	50.2
		Σ = 1350.2	90.1	3061.0	204.0

treatments reduced larval survival of both species, but *H. betteni* was less tolerant and had lower survival. Because sediment treatments had no effect on relative growth rates or final mass, net maintenance costs were apparently negligible over the 16-day study. However, if younger larvae had been studied, they may have responded differently. Table 5-3 presents an overview of literature that relates turbidity and suspended sediment effects on macroinvertebrates.

5.3 TROPHIC EFFECTS AND ORGANIC MATTER PROCESSING

The effects of suspended sediment and turbidity likely go beyond changes in the invertebrate community. As discussed in earlier sections, suspended sediment may decrease light penetration into the water column, which reduces primary production and consequently decreases food resources, particularly for grazers and filter feeders. Studies where this type of response was observed looked at elevated turbidity over a period of months or longer (Van Nieuwenhuyse and LaPerriere 1986, Waters 1995, Lloyd et al. 1987). The distribution of grazing invertebrates can be affected by the abrasion or scouring of cells (Vuori and Joensuu 1996). In turn, the numbers of predatory invertebrates and other secondary consumers in the system may decrease due to lack of food resources. Ultimately, fish and other wildlife dependent on the aquatic system may be affected. However, the biological consequences of reducing photosynthesis in a stream will depend on the relative contribution of autochthonous food sources (i.e., algal) and allochthonous material (i.e., particulate organic matter from outside the stream) in supporting the invertebrate community (Ryan 1991).

Waters (1995) presented a review of research on the effects of suspended sediment on primary production. Very little quantitative work was done on this topic until the 1980s. Lloyd et al. (1987) developed a model to relate turbidity (in NTUs) to primary production in Alaskan streams. They found that in clear (<1 NTU), shallow streams increases of 5 NTU reduced production by 3 to 13 percent, while an increase of 25 NTU reduced production by 13 to 50 percent. Graham (1990) examined the effects of clay-size inorganic particles on algal periphyton and found that the clay attached to sticky surfaces and reduced the organic proportion in the periphyton mat. This results in a loss in food value for grazing invertebrates.

In a study of the effects of placer gold mining in subarctic Alaskan streams, Van Nieuwenhuyse and LaPerriere (1986) found that primary productivity was significantly reduced when turbidity and settleable solids levels increased. During active mining in a moderately mined stream, turbidity averaged 170 NTU and in a heavily mined stream turbidity averaged 2,200 NTU. Undisturbed streams had turbidities less than 1 NTU. Primary productivity (standing crop of periphyton measured as chlorophyll α) was undetectable in heavily mined streams, up to 3.8 mg/m^2 in a moderately mined stream, and up to 11.8 mg/m^2 in an undisturbed stream (Van Nieuwenhuyse and LaPerriere 1986).

Table 5-3. Literature Relating the Effects of Suspended Sediment and Turbidity on Macroinvertebrates and Primary Productivity.

Source	Concentration or Turbidity	Exposure Duration	Description of Effect(s)	Taxa	Comment
Lemly, A.D. 1982	1.5 – 8 JTU	8 months continuous	Decrease in species richness and diversity; inorganic particles adhering to body and gills	Ephemeroptera, Plecoptera, Trichoptera, Diptera	Sedimentation effects may have been more significant than suspended
Culp et al. 1986	Not given	Sediment addition took place in less than 1 hour	Density reduction and change in community composition	Chironomidae, Baetidae, Alloperla, Cinygmula, Paraleptophlebia, Zapada	Results sampled for 24 hours – mostly due to saltating particles
Rosenberg and Wiens 1978	7.76 and 7.42 mg/L	5 hours	Drift of macrobenthos increased significantly	Hydracarina, Simuliidae, Oligochaeta, Chironomidae, Plecoptera, Ephemeroptera	Could not determine if standing crop was effected
Strand and Merritt 1998	5-23 NTU	3-6 hours	Diminished survival; interspecific behavioral effects	*Hydropsyche betteni* and *Ceratopsyche sparna*	Growth rate was not altered; higher mortality of *C. sparna*
Van Nieuwenhuyse and LaPerriere 1986	0.5-3400 NTU	~1-4 months	Reduced to eliminated primary productivity	Algae	Heavy metals from mining probably an additional factor
Wagener and LaPerriere 1985	1-2,500 NTU	Varies, months to years	Decreased density and biomass	Orthocladiini and Chloroperlid stoneflies	
Shaw and Richardson 2001	695 to 705 mg/L	0-6 hours, every other day for 19 days	Decreasing abundance and family richness with increasing duration due to increasing drift	Simuliidae, Elmidae, Nemouridae, Baetidae, Leptophlebiidae, Heptabeniidae	

5.4 COMMUNITY RESPONSE

Changes in invertebrate community composition due to sediment have been detected by many investigators (Zweig 2000, Relyea et al. 2000, Wagener and LaPerriere 1985, McClelland and Brusven 1980). However, most studies were focused on deposited sediments (Appendix A). The diversity of species is often reduced. This includes reduction in sensitive species and life stages. Filter feeders and grazers are often reduced. For example, Lemley (1982) observed a reduction in filter feeding and sensitive taxa and an increase in burrowing macroinvertebrates from exposure to sedimentation and nutrients (Table 5-4). This in turn may cause reductions in predaceous insect larvae.

A study of the effects of placer mining on invertebrate communities in nine similar Alaska streams found that increased turbidity, settleable solids, and suspended sediment were associated with decreased density (Table 5-5) and biomass (Table 5-6) of invertebrates (Wagener and LaPerriere 1985). Orthocladiini (Chironomidae) and Chloroperlid stoneflies decreased in abundance in a mined stream, but not in a nearby unmined stream. Interestingly, increased turbidity was the strongest descriptor of reduced invertebrate density.

5.5 DURATION, MAGNITUDE, AND FREQUENCY OF DISTURBANCE

A number of studies address the effects of turbidity and suspended sediment in relation to degree and duration of disturbance events. However, only two studies addressed the frequency of disturbance events: Robinson and Minshall (1986) and Shaw and Richardson (2001).

The degree and duration of disturbance are important factors in the response of invertebrates to sediment pollution (Henley et al. 2000, Strand and Merritt 1997, Robinson and Minshall 1986). For example, acute disturbances such as suction mining, road construction, or riparian clear-cutting have dramatic, but temporary effects on community composition due to differences in species response (Strand and Merritt 1997).

Shaw and Richardson (2001) studied the effects of fine sediment pulse duration on drifting and benthic invertebrate assemblages using 14 streamside flow-through experimental channels. The channels were exposed to fine sediment pulses of constant concentration but varying pulse duration of 0 to 6 hours every other day over a period of 19 days. Total abundance and family richness of invertebrates declined significantly with increasing pulse duration (Shaw and Richardson 2001) (Figure 5-1). The abundance of drift organisms also increased as pulse duration increased (Figure 5-2). Treatment effects became stronger as the number of sediment pulses increased over the period of the experiment (Shaw and Richardson 2001). The timing of the response was delayed and was not detected until the fifth sediment pulse (day nine). Some taxa were disproportionately affected, including Simuliidae, Elmidae, Nemouridae, Baetidae, Leptophlebiidae, and Heptageniidae (Shaw and Richardson 2001). Plecoptera are also

Table 5-4. Mean Standing Crop, Biomass, and Diversity of Major Stream Insect Taxa from Cullowhee Creek, March-October 1978 (Lemley 1982).

Taxa and parameter	Zone 1		Zone 2		Zone 3	
	Mar-Jun	Jul-Oct	Mar-Jun	Jul-Oct	Mar-Jun	Jul-Oct
Plecoptera						
Density(No./m^2)	69	225	36	24	17	43
Dry weight (g/m^2)	0.260	1.534	0.116	0.119	0.188	0.141
Species richness	10	12	6	7	3	4
Species diversity (đ)	3.86	3.26	1.99	2.66	1.91	2.47
Trichoptera						
Density(No./m^2)	57	348	60	190	44	112
Dry weight (g/m^2)	0.061	0.941	0.130	0.172	0.076	0.103
Species richness	18	14	12	12	7	9
Species diversity (đ)	3.99	3.06	3.10	3.00	3.03	2.80
Ephemeroptera						
Density(No./m^2)	412	275	280	305	270	314
Dry weight (g/m^2)	1.431	0.277	0.280	0.316	0.338	0.364
Species richness	14	17	11	12	10	10
Species diversity (đ)	2.98	3.61	2.11	2.27	1.93	1.74
Diptera						
Density(No./m^2)	369	98	380	707	314	935
Dry weight (g/m^2)	0.417	0.028	0.436	1.530	0.403	1.680
Species richness	8	10	10	10	8	8
Species diversity (đ)	2.42	3.77	2.39	2.78	1.66	1.10
Totals						
Density(No./m^2)	818	721	756	1226	625	1404
Dry weight (g/m^2)	2.169	2.770	0.962	2.137	1.005	2.288
Species richness	50	53	39	41	28	31
Species diversity (đ)	3.32	3.43	2.40	2.68	2.13	2.03

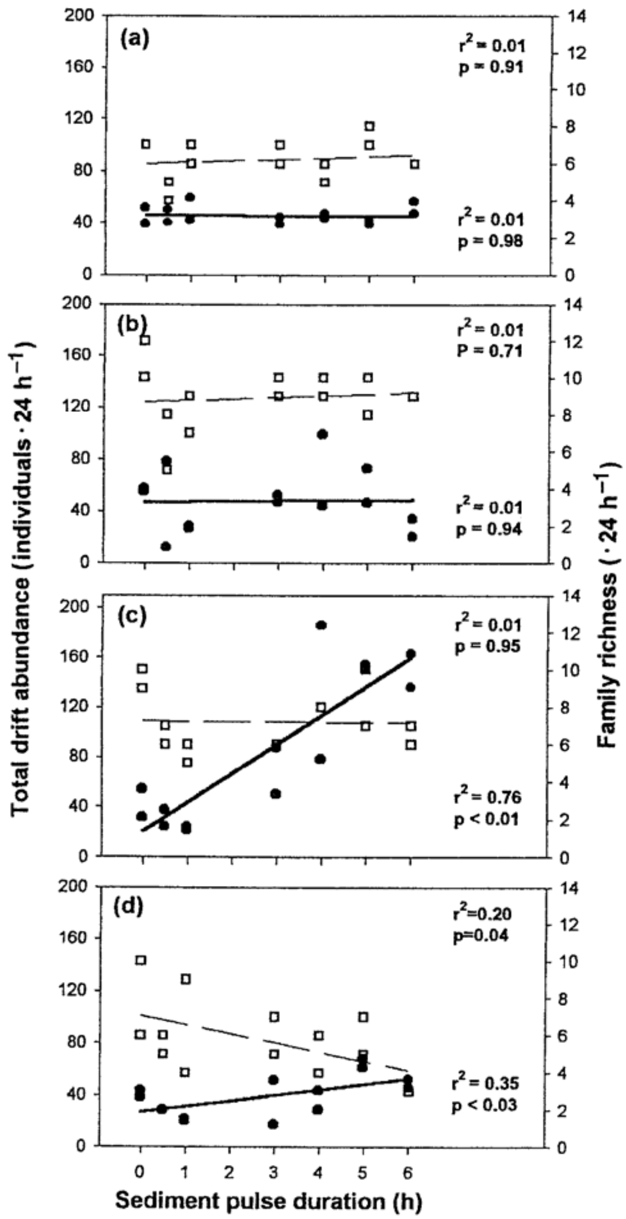
Zone 1 = no sedimentation or nutrient enrichment. Zone 2 = sedimentation only. Zone 3 = sedimentation and nutrient enrichment

Table 5-5. Seasonal Means, Standard Deviations, and Medians of Invertebrate Densities (No./0.1 m^2) Summer, 1983 (n = 20 except Chatanika River n = 19) (Wagener and LaPerriere 1985).

Category of Stream and Stream	Invertebrate Densities (No. /0.1 m^2)		
	Mean	SD	Median
Reference			
McManus	68	32	65.5
Twelvemile	56	48	38.5
Boulder	37	38	31.5
Mined			
Chatanika	25	15	27.0
Faith	18	16	17.0
Ketchem	11	10	6.5
Birch	8	5	7.0
Mammoth	3	4	3.0

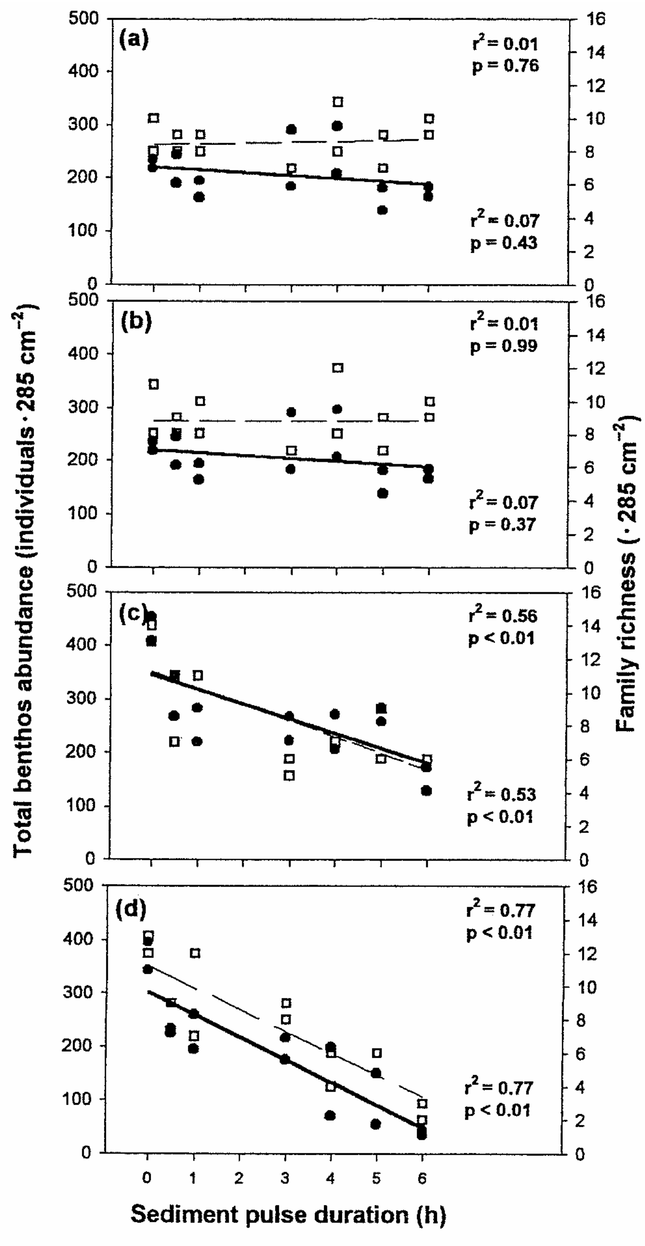
Table 5-6. Seasonal Means, Standard Deviations, and Medians of Invertebrate Biomass (Ash-free Dry Weight, mg/0.1 m^2) Summer, 1983 (Wagener and LaPerriere 1985).

Category of Stream and Stream	n	Invertebrate Biomass Mean	SD	Median
Reference				
McManus	20	4.5	4.8	3.0
Twelvemile	19	3.7	3.2	3.0
Boulder	20	1.9	1.8	1.0
Mined				
Chatanika	16	1.9	3.7	1.0
Faith	20	2.5	2.4	3.0
Ketchem	19	1.9	3.3	1.0
Birch	20	0.7	1.8	<1.0
Mammoth	20	0.5	0.8	<1.0



Note: Solid lines represent least squared regression lines for total abundance (●) and broken lines for family richness (□). Regression statistics for each line are shown in the upper (abundance) and lower (richness) right corner. In each panel $n = 14$.

Figure 5-1. Relationship between Sediment Treatments and Drift Total Abundance or Family Richness at Pretreatment (a), Day +1 (b), Day +9 (c), and Day +19 (d) (Shaw and Richardson 2001).



Note: Solid lines represent least squared regression lines for total abundance (●) and broken lines for family richness (□). Regression statistics for each line are shown in the upper (abundance) and lower (richness) right corner. In each panel n = 14.

Figure 5-2. Relationship between Sediment Treatments and Benthos Total Abundance or Family Richness at Pretreatment (a), Day +1 (b), Day +9 (c), and Day +19 (d) (Shaw and Richardson 2001).

known to be very sensitive to elevated fine sediments. Some Diptera taxa, such as Chironomidae, are known to sometimes increase with elevated sediment levels.

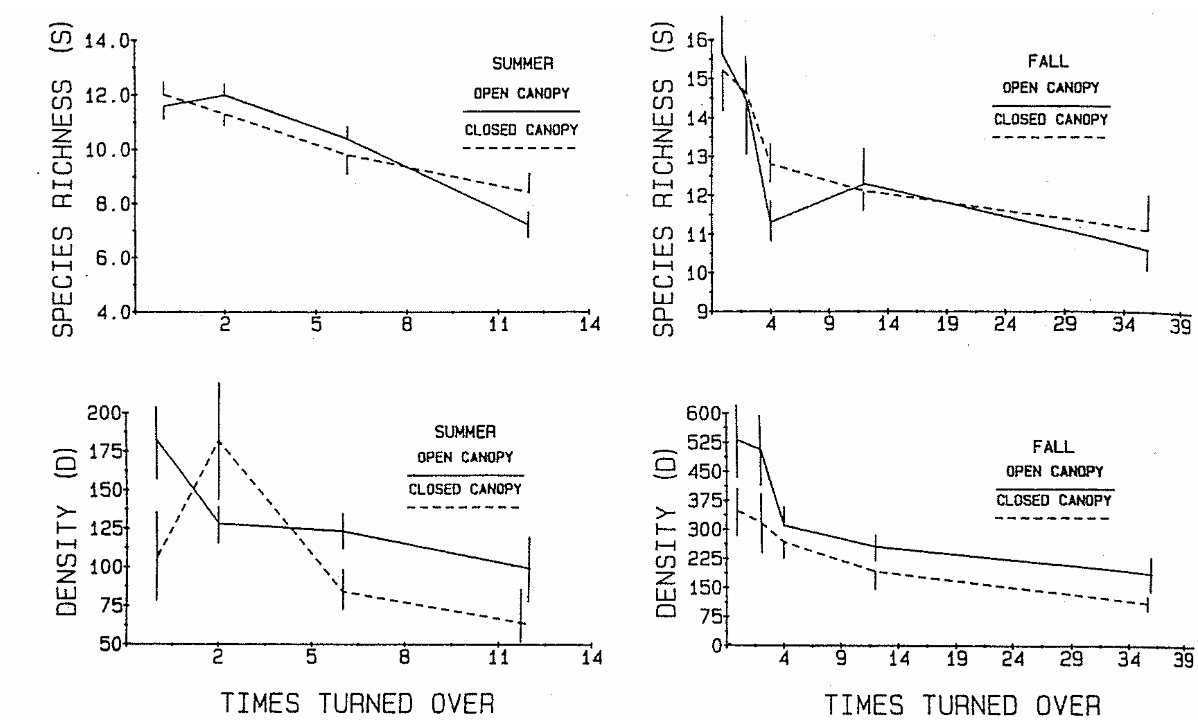
Robinson and Minshall (1986) performed a field experiment on the effects of disturbance frequency on invertebrates and periphyton in a third order Rocky Mountain stream. Previously colonized bricks were turned at intervals of 0, 3, 9, 27, or 54 days. Although this type of disturbance is not equivalent to increased flows or suspended sediments, the results suggest that macroinvertebrate communities are generally maintained when disturbance is infrequent (not more than once per month) and macroinvertebrates have the ability to recolonize. Robinson and Minshall (1986) found that invertebrate species richness and density were reduced as disturbance frequency increased (Table 5-7). Periphyton was found to decrease with disturbance frequency in an open canopy site, but not a closed canopy site (Figure 5-3). Robinson and Minshall (1986) found that invertebrate species richness was maintained when disturbance frequency was greater than every 27 days. As the frequency increased to every nine, three, and zero days, a concomitant decrease in species richness and invertebrate density was observed during both the summer and fall experiments.

Ultimately, the time that it takes for an invertebrate community to recover from a single sediment event relative to the frequency of such events will determine whether there is a long-term alteration in the community. If streamflows are able to flush out fine deposited sediments on a regular basis, the potential for rapid recolonization is good (Ryan 1991).

Table 5-7. Effects of Disturbance on Absolute and Relative Numbers (per brick) of the 10 Most Abundant Taxa, by Site and by Season (Robinson and Minshall 1986).

	Disturbance Interval (days)									
	3		9		27		54		108	
Species	No.	%	No.	%	No.	%	No.	%	No.	%
OPEN CANOPY SITE—SUMMER										
*Baetis tricaudatus**	37.6	38	44.2	36	53.6	42	78.6	43		
Chironomidae	33.0	33	40.2	33	34.5	27	53.0	29		
Glossosoma spp.	18.0	18	17.8	14	15.7	12	3.2	2		
Alloperla spp.**	2.6	3	3.4	3	2.7	2	19.8	12		
Isoperla spp.**	0.4	0	3.6	3	1.9	1	0.4	0		
*Drunella grandis***	1.0	1	3.6	3	1.0	1	8.4	5		
*Serratella tibialis***	1.0	1	3.0	2	6.5	5	9.8	5		
Cleptelmis spp.	1.2	1	3.1	2	5.2	4	1.6	1		
Hydropsyche spp.	1.0	1	0.6	0	1.0	1	0.9	0		
Simulium spp.	0	0	0	0	0	0	0	0		
Others	3.0	3	4.0	3	5.0	4	4.0	2		
CLOSED CANOPY SITE—SUMMER										
Baetis tricaudatus	49.8	79	52.6	63	117	64	65.5	61		
Chironomidae**	3.4	5	10.6	12	22.8	12	12.8	12		
Glossosoma spp.	0	0	0	0	0	0	0	0		
Alloperla spp.	1.6	2	1.0	1	3.2	2	1.5	1		
Isoperla spp.	1.0	1	0.8	1	1.0	0	1.8	1		
*Drunella grandis**	0	0	0	0	0	0	0.75	1		
*Serratella tibialis**	0	0	3.4	4	3.8	2	5.5	5		
Cleptelmis spp.	0	0	1.6	2	1.2	1	1.0	1		
Hydropsyche spp.*	1.4	2	5.2	6	8.8	5	8.0	7		
Simulium spp.*	0	0	1.6	2	11.5	6	4.5	4		
Others	7.0	11	7.0	9	11.0	6	5.0	5		
OPEN CANOPY SITE—FALL										
*Baetis tricaudatus**	50	27	59	23	114	35	126	25	145	27
Chironomidae**	56	30	67	26	41	13	114	23	141	27
Glossosoma spp.	32	17	49	19	100	31	123	24	86	16
Alloperla spp.	12	6	19	7	22	7	34	7	33	6
Isoperla spp.*	13	7	19	7	18	5	34	7	33	6
*Drunella grandis***	1	0	2	1	2	1	4	1	8	2
*Seratella tibialis***	7	4	19	7	6	2	28	5	40	7
Hydropsyche spp.*	1	0	1	0	2	2	4	1	4	1
Capnia spp.	8	4	4	2	2	1	4	1	6	1
Cinygmula spp.	2	1	2	1	7	2	9	2	9	2
Others	7	4	18	7	6	2	20	4	21	4
CLOSED CANOPY SITE—FALL										
Baetis tricaudatus	53	49	102	53	143	53	165	52	211	61
Chironomidae**	5	5	8	4	10	4	17	5	16	5
Glossosoma spp.**	1	1	2	2	2	1	3	1	1	0
Alloperla spp.	5	5	7	4	9	3	9	3	11	3
Isoperla spp.**	5	5	6	3	12	4	11	3	7	2
Drunella grandis	0	0	0	0	0	0	0	0	0	0
Serratella tibialis	5	5	7	4	10	4	9	3	13	4
Hydropsyche spp.**	1	1	3	2	6	2	7	2	15	4
Capnia spp.	8	7	8	4	10	4	10	3	7	2
Cinygmula spp.**	9	8	11	6	21	8	28	9	24	7
Others	9	9	36	19	45	17	60	19	42	12

Note: Asterisks mark taxa whose absolute numbers were significantly different at different disturbance frequencies: * significant at p=0.10, ** significant at p=0.05. Relative abundances were not significantly different at p=0.05.



Note: Bars represent standard error of the mean. n=7.

Figure 5-3. Effects of Disturbance Frequency (Times Bricks Turned Over) on Invertebrate Species Richness (S) and Animal Density (D) (Robinson and Minshall 1986).

Literature that evaluates behavioral and physiological effects of turbidity and suspended sediment on fish is summarized. Turbidity and suspended sediment affect fish by impairing vision and altering behavior associated with feeding, the perceived risk of predation and social interaction with other fishes. High levels of suspended sediment can cause physical harm to gill tissues and cause physiological effects that ultimately can result in injury or death. Recent research has investigated physiological effects such as suspended sediment effects on blood glucose, "stress" hormone secretion, packed red blood cell volume, and impaired disease resistance.

The meta-analyses of Newcombe and MacDonald (1991), Newcombe and Jensen (1996), and Newcombe (2003) presented in Section 4 synthesized much of the existing data on the effects of sediment on fish. This section reviews some of the studies that supported those analyses.

6.1 BEHAVIORAL EFFECTS OF TURBIDITY AND SUSPENDED SEDIMENT

Fish use their eyesight to orient to their physical surroundings, find prey items, interact with neighbors and avoid predators. Many fish species are adapted to living in turbid waters and have developed non-visual methods for prey detection and capture. Catfish, sturgeon and carp that typically inhabit turbid waters all use non-visual methods to find food including barbels and their lateral line systems. Species that are sight feeders and are typically associated with clear water include trout, some of the native minnows such as hardhead, (*Mylopharodon conocephalus*) and Sacramento pikeminnow (*Ptychocheilus grandis*), as well as prickly sculpin and riffle sculpin (*Cottus asper* and *C. gulosus*). All of these species are regularly exposed to turbid water conditions during high flow events.

Most behavioral effect studies have been conducted in controlled laboratory systems, either in completely artificial environments in tanks with circulating water or in artificial streams. Other variables are strictly controlled for these tests with the expectation that the effect of turbidity will be isolated in the study design. Parallel control studies are typically run in clear water. In most study designs the water is re-circulated by a pump and filter system, provided in a flow through system or is kept in suspension in tanks by aerators. Turbidity in the lab is generated from soils, clays or other products that are mixed with water, then allowed to settle for a specified period of time prior to mixing the elutriate into the tank or stream through a mixing box. Most studies have evaluated exposure to near constant turbidity levels. A few studies have evaluated response to a pulse of turbidity (e.g. Berg and Northcote 1985).

Most of the laboratory studies examining turbidity and suspended sediments have used juvenile fish as test subjects. This is partly a result of the economy of scale, but also because juvenile fish are commonly believed to be more sensitive than subadult or adult life history phases (but see Newcombe and Jensen [1996] discussed in Section 4), so the thinking appears to be that effects will be more easily detected. However, the behavior of juvenile hatchery fish may differ from naturally spawned fish. Investigators also have used small species such as shiners from US Midwestern streams. Exceptions to this practice include the use of adult male Chinook salmon (*Oncorhynchus tshawytscha*) in sediment avoidance tests and the use of subadult largemouth bass (*Micropterus salmoides*) and adult rainbow (*Oncorhynchus mykiss*) and lake trout (*Salvelinus namaycush*) in feeding rate and reaction distance studies.

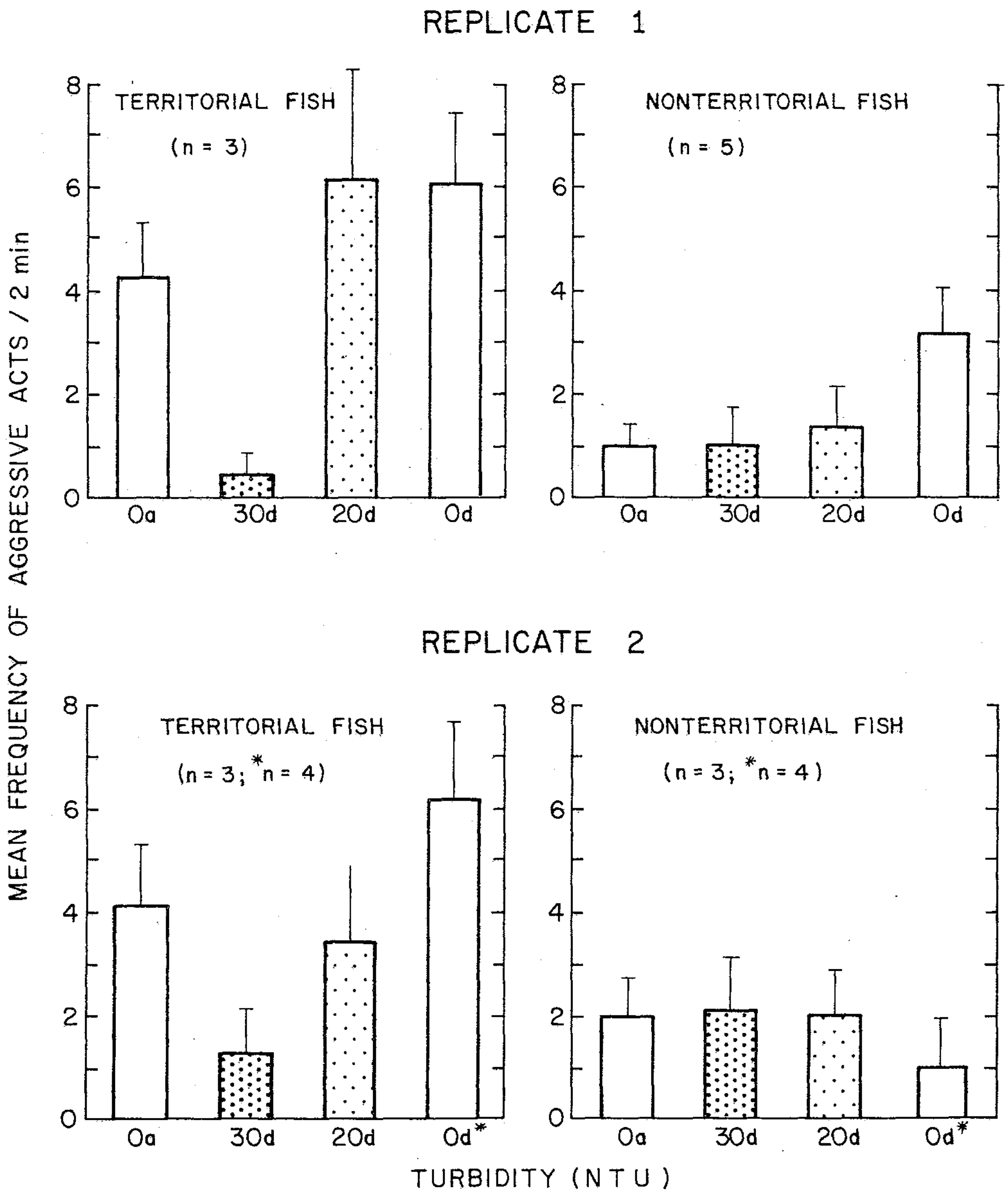
This section summarizes the known effects of turbidity on fish behavior related to the ability of fish to see their surroundings, interact with other fish in territorial displays, obtain prey and avoid predators.

6.1.1 TERRITORIALITY

Berg and Northcote (1985) recorded the behavior of juvenile coho salmon (*Oncorhynchus kisutch*) in response to short-term pulses of suspended sediment in an otherwise clear water test chamber. Two groups of fish were allowed to occupy the test stream and establish a social hierarchy prior to exposure to a rapid increase in turbidity (60 NTU in an hour). A third group was exposed to a gradual increase (over 2 days) to the same level. Juvenile coho salmon were captured by seine from the wild and held in tanks for several weeks prior to beginning the study. Young of the year coho salmon averaged 5.3 mm and ranged from 4.7 to 6.0 mm, with a mean weight of 1.7 gram (g) and range of 1.1 to 2.8 g. The lengths are off by a factor of 10 and should be reported as 53 mm and 47 to 60 mm. The paper does not indicate if the lengths are standard, fork or total lengths.

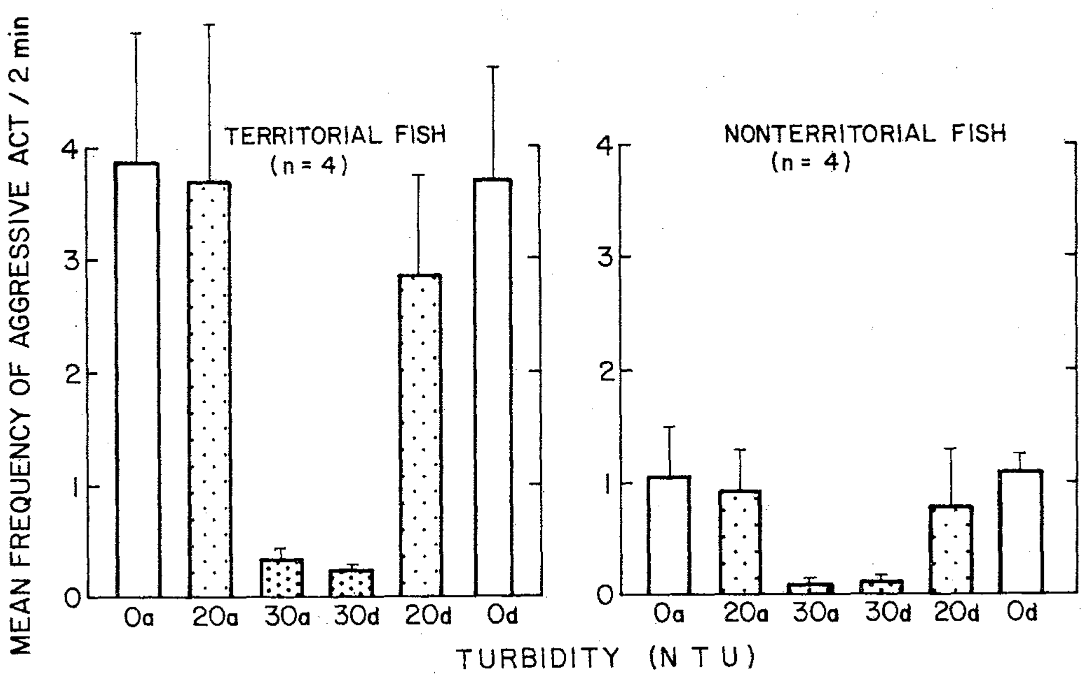
Fish response differed between the rapid and gradual increase in turbidity. The rapid increase of turbidity completely disrupted established behavior (Figure 6-1). Most fish swam upstream to the turbidity front then followed the front downstream staying in clear water until they reached the downstream end of the test stream. A few fish displayed an alarm reaction, darting about the test stream, while others entered the gravel and stayed there for several hours. In comparison, the gradual increase in turbidity elicited no fright reaction even at 60 NTUs (Figure 6-2). Territories continued to be defended up to 30 NTU, when it was largely disrupted. Fish changed their holding position to the lower part of the water column and maintained that holding position even when turbidities increased to 60 NTU.

Observation of fish behavior at 60 NTU was difficult, but few interactions were observed. Formerly dominant individuals no longer exerted their dominance, and no fish was observed to defend its territory. In both the rapid and gradual pulse tests, fish did not change this behavior until turbidity dropped from 60 to 30 NTU, at which time the dominance hierarchy was reestablished but less structured. Only a few fish retained their social rank and no territory defense was observed. Fish frightened from the rapid



Note: Vertical lines show ± 1 SE; a = ascending limb of pulse; d = descending limb of pulse; numbers of fish given in parentheses.

Figure 6-1. Frequency of Aggression by Juvenile Coho Salmon during Sudden Pulse Experiments (Berg and Northcote 1985).



Note: Vertical lines show ± 1 SE; a = ascending limb of pulse; d = descending limb of pulse; numbers of fish given in parentheses.

Figure 6-2. Frequency of Aggression by Juvenile Coho Salmon during the Gradual Pulse Experiment (Berg and Northcote 1985).

increase in turbidity moved out of the gravel at 30 NTU but remained within the lower part of the water column. When turbidity decreased to 20 NTU the dominance hierarchy was more structured and territories were reestablished. A social organization similar to what existed pretreatment reestablished. As turbidity was reduced to nearly 0 NTU, fish extended their holding positions to locations higher in the water column. Fish behavior in the post-treatment phase was similar to the pre-treatment phase.

6.1.2 AVOIDANCE

Several studies have evaluated the effects of turbidity and suspended sediment in relation to fish avoidance behavior. Periodically high levels of turbidity and SSC are a natural occurrence during some portion of the year, and fish may have evolved to tolerate, or perhaps even utilize, low to moderate levels of turbidity. These studies have attempted to characterize the turbidity and SSC levels at which avoidance responses occur, as well as factors that may affect these threshold levels.

Noggle (1978) found no significant avoidance reaction or downstream movement in response to suspended sediment concentrations as high as 2,500 mg/L in artificial, experimental streams. He exposed juvenile coho salmon (size range 76-133 mm) and steelhead (*O. mykiss*) (136 mm) in initially clear water to variable suspended sediment levels of up to 2,500 mg/L. Fish avoiding these levels were quantified by counting the number of fish in the clear and turbid tributaries and in the net-lined sumps at half-hour intervals over periods of three to ten hours. No statistically significant avoidance or preference for the tributary of the turbid or clear stream was observed and no significant downstream movement occurred during any of the 10 trials.

Noggle (1978) also exposed juvenile coho salmon (73 mm) to suspended sediment in Y-trough experiments. Test SSC concentrations were low (clear water, 0 mg/L), medium (1,000 to 4,000 mg/L) and high (4,000 to 12,000 mg/L). The downstream arm of the Y-trough contained a mixture of water from the two upstream troughs. There was a statistically significant difference in proportions of coho salmon choosing the three arms between low, medium and high concentrations. Coho salmon in clear-water trials were about equally distributed between the three arms of the Y-trough. In the tests with medium concentrations, coho salmon favored the downstream, mixed arm of the Y-channel and more chose the turbid arm than the clear arm ($p = 0.05$). At high concentrations, coho salmon showed a shift to the clear arm ($p = 0.05$) and fish finally choosing the clear arm showed rapid opercular and cough rates. Noggle concluded that juvenile coho salmon do not avoid suspended sediment concentrations normally encountered in nature, but that avoidance was observed at suspended sediment concentrations (4,000-8,000 mg/L) well above the 96-hr LC50 value.

Bisson and Bilby (1982) found that young-of-the-year coho salmon (no length data provided, average weight 0.7 to 2.0 g) acclimated in clear water (< 0.3 NTU) did not exhibit significant avoidance of turbid water until turbidities reached 70 NTU. However, coho salmon acclimated in slightly turbid water (for three weeks) did not exhibit avoidance until turbidities reached 100 NTU (Table 6-1). Following their acclimation period in clear or slightly turbid water, fish were placed in an aquarium with clear water for the first 30 minutes of the trial, then suspended sediment was introduced into one-half of the chamber and fish were allowed to choose between the turbid and clear halves for an additional 30 minutes. Fish initially avoided the onset of turbid water, but after about 5 minutes they began to pass into and out of the turbid and clear water sections of the chamber and then selected a site either on the clear or turbid side of chamber. Responses to turbidity increases were not always in direct proportion to the sediment concentrations. For example, greater avoidance was observed at 158 NTU than at 184 NTU. Neither test group showed a response to slightly turbid water in the 10 to 20 NTU range.

Bisson and Bilby (1982) noted a fright behavior in some of their trials with juvenile coho salmon that had been acclimated to slightly turbid water, then exposed to turbidity levels of 42, 99, 104 and 195 NTU in the test chamber. The authors speculate the fright response was related to the sudden transfer from turbid water into clear water where cover was lacking. The fright response included rapid darting movements, a "fright huddle" and attempts to hide in the corners of the tank. In trials where the fright response occurred, the test subjects preferred the turbid portion of the chamber and this preference increased with higher turbidities. Bisson and Bilby (1982) mention that all the fish in the four tests in question exhibited the fright response. They conclude that there was no evidence of a significant preference for slightly turbid water (10-20 NTU), but that in certain instances, water having higher turbidity was sought for cover when the fish were frightened.

When Berg and Northcote (1985) examined behaviors of juvenile coho salmon following short-term pulses of suspended sediment in a laboratory stream, they found no fright response to a gradual increase in turbidity (over two days). However, juvenile coho salmon avoided a sudden increase (over one hour) in turbidity to 60 NTU, with some fish exhibiting a fright response entering the gravel at the bottom of the tank.

Experiments by Sigler et al. (1984) found that juvenile steelhead and coho salmon subjected to continuous clay turbidities were more likely to emigrate from experimental channels than fish in clear water, and that they exhibited less growth. Hatchery juvenile steelhead (mean length of subsampled test groups at the start of study ranged from 26.8 mm to 45.7 mm) and coho salmon (mean length of subsampled test groups at the start of the study ranged from 33.4 to 45.2 mm) were introduced into experimental channels, and test duration ranged from 11 to 31 days. When experimental turbidity levels were between 167 to 265 NTUs, most of the steelhead left the turbid channels in the first two to three days and almost no fish were found in the channels after 14 days. During initial tests with turbidity levels of 57 to 77 NTUs, most small steelhead survived, and therefore subsequent tests were conducted with turbidities of approximately 80 NTU or less. Although not always statistically significant, density of coho and steelhead in clear water channels was always higher than for turbid channels. Most fish emigrated from the channel with turbid water during the first two diel cycles in each test, indicating the

Table 6-1. Behavioral Response by Juvenile Coho Salmon to the Introduction of Suspended Sediment, as Measured by the Change in Number of Fish Observed in the Treated Half of the Test Chamber before and after Sediment Addition. Positive Numbers Denote Increased Preference for the Treated Portion after Sediment Addition; Negative Numbers Denote Avoidance (Bisson and Bilby 1982).

Turbidity (NTU)	Average per cent change
Clear-water acclimation	
10	+9
16	-5
19	-9
41	-6
42	+1
53	-7
70	-13[a]
97	-16[a]
158	-26[a]
184	-12[a]
Turbid-water acclimation (normal behavior)	
10	+1
16	+3
81	-3
92	+3
106	-15[a]
124	-34[a]
126	-26[a]
160	-19[a]
179	-15[a]
Turbid-water acclimation (fright behavior)	
42	+13[a]
99	+15[a]
104	+26[a]
195	+37[a]

[a] $P < 0.05$.

turbidity was stressful to the fish. Small fish (< 40 mm) were less likely to stay in turbid-water channels than larger fish. The study design placed the fish directly in turbid channels and therefore the sudden transfer of fish to turbid water could have initiated the movement. The small size of some of the steelhead used in the tests also could have been a factor in the avoidance reaction.

Servizi and Martens (1992) observed the tendency of hatchery coho salmon (1.0 g ± 0.1 g) to swim toward the surface to avoid suspended sediment. In clear water, less than 1 percent of the coho salmon swam at the surface, usually swimming at the bottom (depths of approximately 20 cm) in the test vessel. At SSC of 0.3 g/L (37 NTU) over a period of 96 hours, coho salmon exhibited the first sign of avoidance by swimming near the surface for minutes at a time (then submerging) to avoid what the authors concluded was the higher SSCs present at depth. This SSC was close to 0.24 g/L, at which significant elevation of cough frequency occurred. Mean avoidance was less than 5 percent up to the inflection point at 2.55 g/L (270 NTU), but rose to approximately 25 percent at 7.0 g/L (Figure 6-3). Continuous increase in the slope of the avoidance curve beyond the inflection point may be an indication that the need for relief from suspended sediments stress increasingly overcomes the preference to stay at greater depth. As a comparison, Servizi and Martens (1992) noted that Bisson and Bilby (1982) reported at 70 NTU, around 13 percent of the coho salmon avoided suspended sediment in the horizontal plane while only about 2 percent of the fish showed avoidance in the vertical plane. This suggested that coho salmon might be more inclined to move laterally than to the surface to avoid suspended sediment. Steelhead and coho salmon avoidance in the horizontal plane (by leaving the artificial stream) in Sigler et al. (1984) occurred at turbidity levels as low as 11 to 49 NTU.

6.1.3 Risk to Predation

Laboratory studies indicate turbidity and suspended sediment conditions appear to influence perceived risk of predation. Several laboratory studies have identified a change in feeding rates and in feeding behavior when test fish are exposed to higher turbidities (Gradall and Swenson 1982, Gregory 1990, 1993, Gregory and Northcote 1993, Gregory and Levings 1996). This behavior is closely tied to feeding or foraging behavior and was recognized during feeding rate studies. In three laboratory studies, juvenile salmonids exposed to moderate levels of suspended sediment spent more time up in the water column rather than near the bottom or near cover (Gradall and Swenson 1982, Gregory and Northcote 1993, Gregory 1993).

Gradall and Swenson (1982) exposed wild brook trout (*Salvelinus fontinalis*) (average size 16.4 cm, range: 12-23 cm) and creek chubs (*Semotilus atromaculatus*) (average size 5.3 cm, range 3.2-7.5 cm) to turbidity gradients within continuous-flow chambers. Red-clay turbidity SSCs ranged from an average of 6.4 formazin turbidity units (FTU) (equivalent to 4.1 mg/L) to 59 FTU (36.2 mg/L). Creek chub were concentrated in areas of the highest turbidity within the tanks while brook trout showed no preference relative to turbidity (average range of 7.1-61.1 FTU or 4.5-37.4 mg/L). Moderate turbidity levels

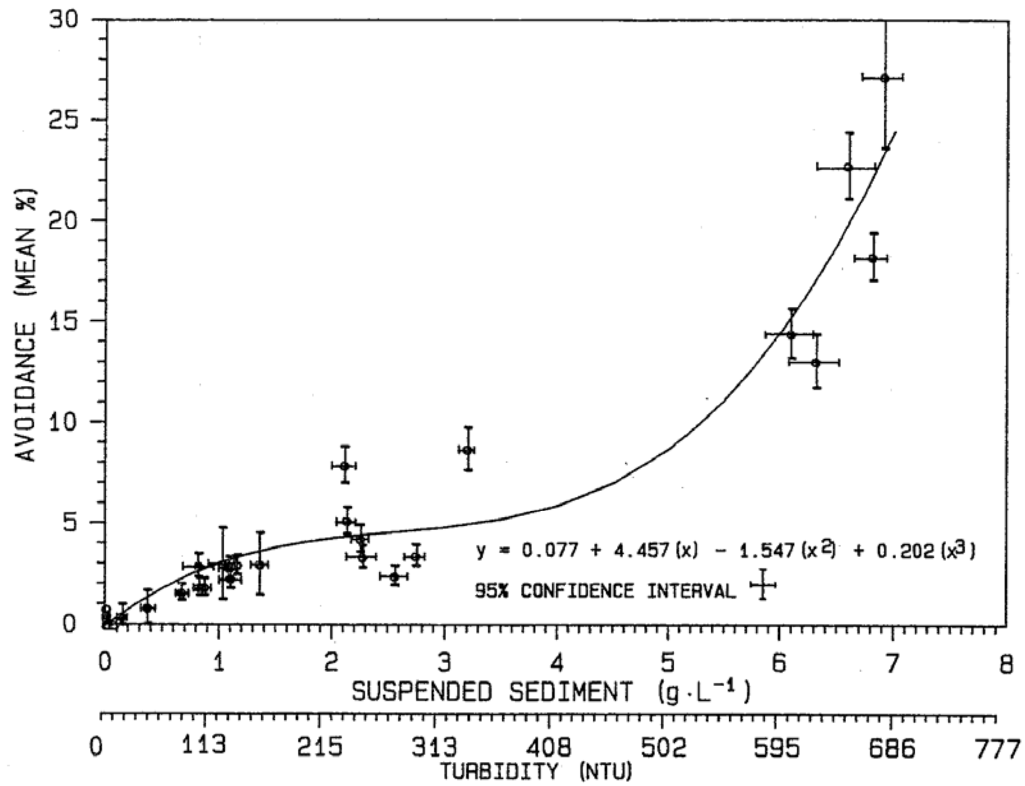


Figure 6-3. Avoidance of Suspended Sediment by Underyearling Coho Salmon Calculated from the Percentage of the Population Observed at the Surface during 96 h (Servizi and Martens 1992).

(average 7.1 FTU) did affect brook trout behavior in that trout used overhead cover less, spent less time in association with the bottom, and were more active than in clearer water (average 2.3 FTU or 1.6 mg/L). Creek chubs exposed to turbidities of 5.8 FTU (3.7 mg/L) demonstrated a statistically significant decrease in the use of overhead cover and increased activity when compared to clear water control tests (Table 6-2). (Gravimetric measurements taken on some samples were used for regression analysis, yielding the following relationship between turbidity measurements and SSC: mg/liter = 0.61 FTU + 0.16.)

Gregory (1990) found that a model bird predator elicited a significant response in juvenile Chinook salmon (65-70 mm fork length [FL]) in both clear and turbid waters, but the reaction in the turbid condition (25 mg/L or 18 NTUs) was less pronounced. Furthermore, the change in spatial distribution of fish, expressed as the ratio of the number of observed fish in the deep areas of the test tank to the number expected from a random distribution, was much lower in turbid than in clear water. The behavior of test fish in conditions of elevated turbidity indicated that the perceived risk to predation declined inversely with turbidity (Figure 6-4) (Gregory 1990).

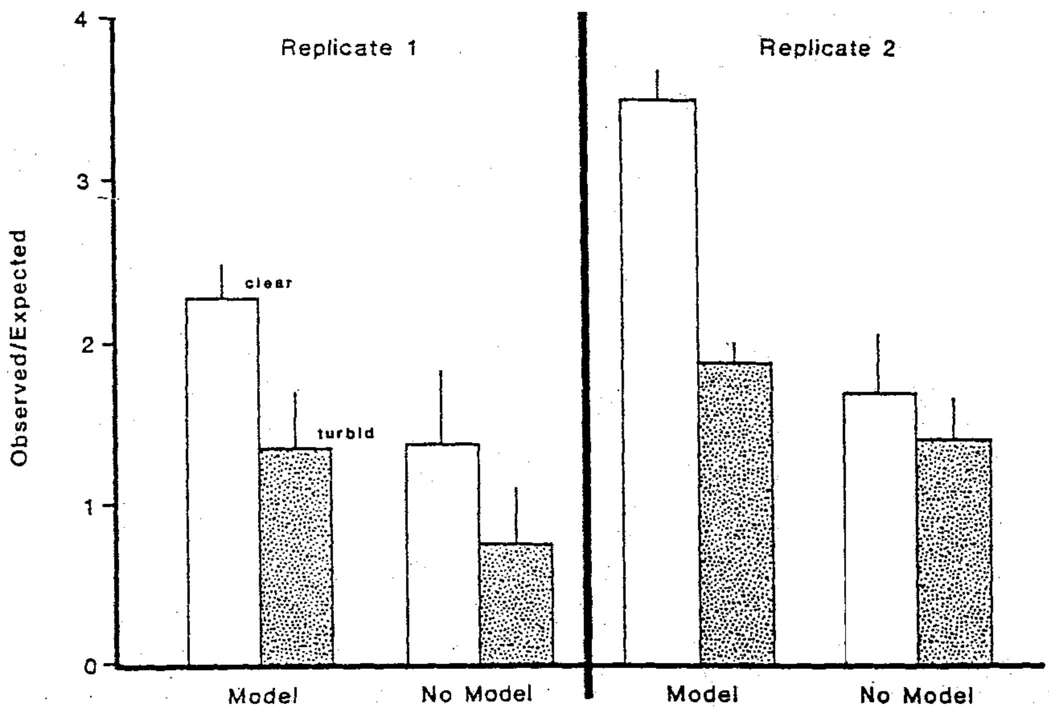
In another study, Gregory (1993) observed juvenile Chinook salmon (average size for two test groups 83.4 ± 02.7 and 78.8 ± 3 mm) were dispersed throughout a test chamber in turbid water (approximately 23 NTU) but were primarily located in the deepest region of the chamber in clear water (< 1 NTU). With the introduction of bird and fish predator models, fish altered their distribution in both turbid and clear water, but in turbid conditions significantly reduced their response. In clear water, juvenile fish subject to the fish or bird predator model rapidly moved to the chamber bottom (Figure 6-5). In turbid conditions, the "startle response" to each predator model was significantly reduced and the recovery time was shorter. In instances with the bird model in clear water, Chinook salmon exhibited long recovery times and in some cases remained at the bottom for as long as 10 minutes. Recovery time was seven times longer in clear water than in turbid water, when averaged across all treatments (Figures 6-6 and 6-7). Recovery occurred much faster with the fish model than with the bird model.

Gregory and Levings (1996) examined the combined effects turbidity and cover on predation by adult coastal cutthroat trout (*Oncorhynchus clarki clarki*) on underyearling juvenile salmon. Prey species included Chinook salmon, chum salmon (*O. keta*), sockeye salmon (*O. nerka*), and coastal cutthroat trout in separate trials. Prey size (mm FL ± SD) for chum salmon ranged from 38.8 ± 2.2 to 41.1 ± 3.1; for Chinook salmon 70.8 ± 3.0 to 79.8 ± 2.6; for sockeye salmon 62.8 ± 4.9 to 64.1 ± 6.3 and for coastal cutthroat 37.4 ± 2.0. Adult coastal cutthroat trout were 35.1 ± 2.7 cm FL and 411 ± 62 g. Treatment conditions in outdoor, concrete ponds included with or without cover (artificial vegetation) and with or without turbidity. Turbidity in clear water ponds ranged from 0.5 to 2.4 NTU and in turbid water ponds ranged from 13 to 87 NTU. Prey species were introduced into the different experimental tanks and survivors were counted at the end of the test (0.75 day for cutthroat trout, 6.75 days for Chinook salmon and 1.75 days for all other species). The number of fish missing at the end of the tests was then expressed as

Table 6-2. Test Conditions in Two Sections of Turbidity-gradient Chamber, and Behavioral Responses of Brook Trout and Creek Chubs to Red-clay Turbidity (Gradall and Swenson 1982).

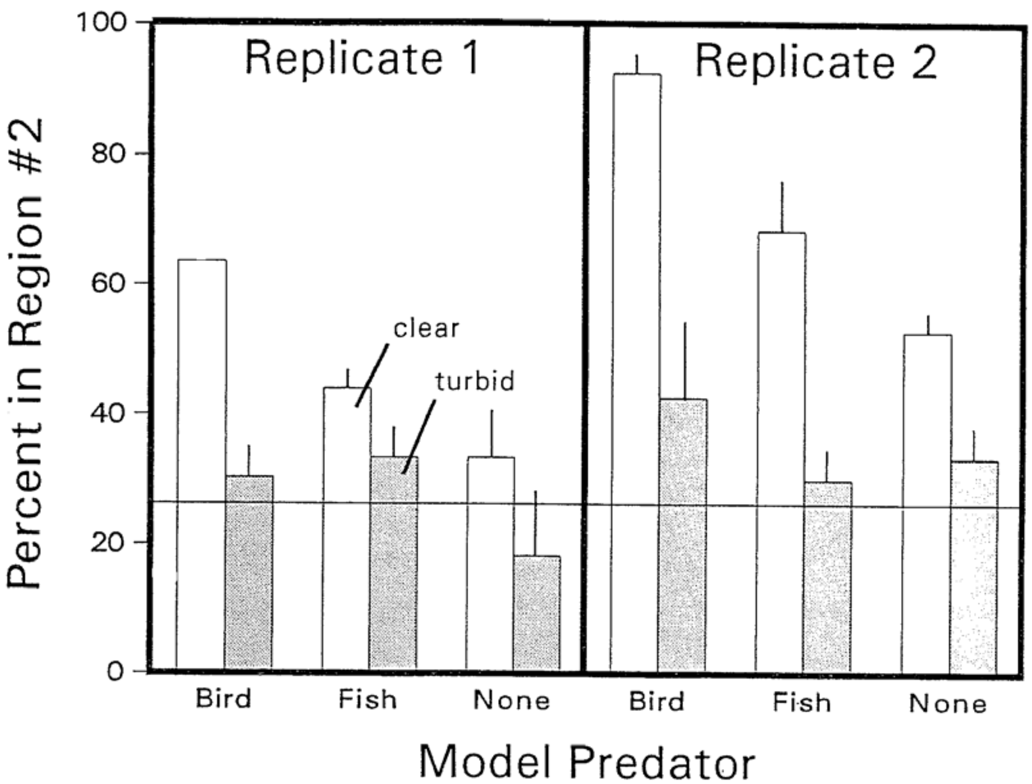
		Section I		Section II				
Date	Temperature (C)	Turbidity (FTU[a])	Fish observed[b]	Turbidity (FTU[a])	Fish observed[b]	Cover-associated (%[c])	Bottom-associated (%[c])	Active (%[c])
				Brook trout—gradient tests				
Nov 17–18, 1977	10.7	59.7	43	11.9	13	69	92	0
Dec 5–6, 1977	8.1	56.8	31	4.5	25	44	72	24
Jun 11–12, 1978	13.0	77.0	35	8.2	21	48	72	0
Jun 23–24, 1978	13.7	50.8	29	5.1	27	33	70	15
Mean	11.3	61.1	35	7.1	22	49	77	10
				Brook trout—control tests				
Nov 28–29, 1977	8.8	1.0	41	1.0	15	95	100	0
Jun 8–9, 1978	13.1	2.0	31	2.0	25	70	84	2
Jun 15–16, 1978	13.2	2.0	28	3.0	28	52	79	2
Jun 28–29, 1978	14.7	3.5	28	3.3	28	92	100	0
Mean	12.5	2.4	32	2.3	24	77	91	1
				Creek chubs—gradient tests				
Dec 17–18, 1977	7.0	54.5	216	7.2	44	4		68
Jan 10–11, 1978	5.5	49.0	229	6.6	71	11		51
Jul 24–25, 1978	16.3	68.8	175	4.8	105	26		24
Aug 3–4, 1978	16.0	54.2	134	4.4	146	33		61
Mean	11.2	56.6	189	5.8	91	19		51
				Creek chubs—control tests				
Dec 11–12, 1977	7.2	2.0	168	2.0	112	30		49
Jan 4–5, 1978	6.4	2.0	70	2.0	230	15		65
Jul 19–20, 1978	15.8	2.0	113	2.0	167	75		27
Aug 9–10, 1978	15.6	3.0	130	3.0	130	95		4
Mean	11.3	2.3	122	2.3	160	54		36

[a] Formazin turbidity units.
[b] Number of fish in the section summed over all observation periods.
[c] Percent of all fish observed in Section II.



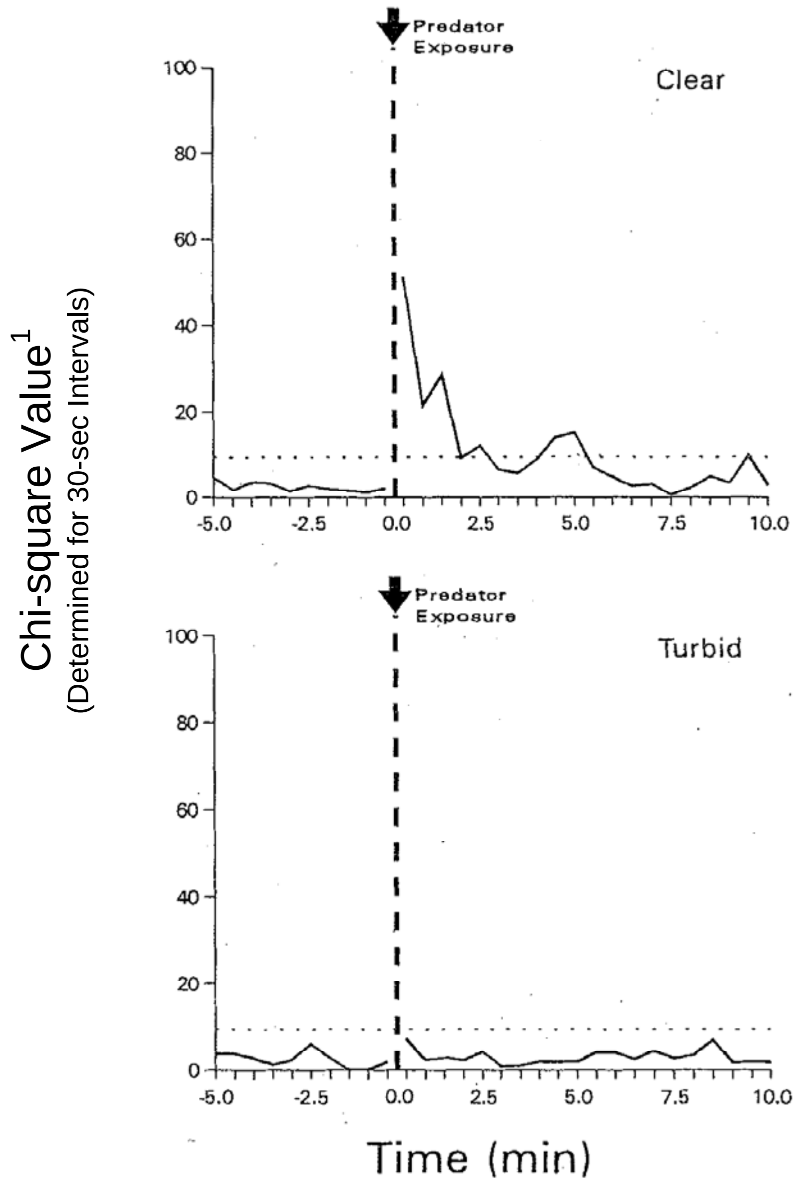
Note: Expected number of fish was based on a random distribution of the 22 fish used in each of the two replicates; vertical bars denote standard error.

Figure 6-4. Observed/Expected Number of Fish in the Deepest Region of an Experimental Arena in Response to a Model Gull Predator in Clear Water and 25 mgL^{-1} (18 NTUs) Turbidity (Gregory (1990).



Note: Horizontal line represents proportion expected by chance; vertical bars represent SD.

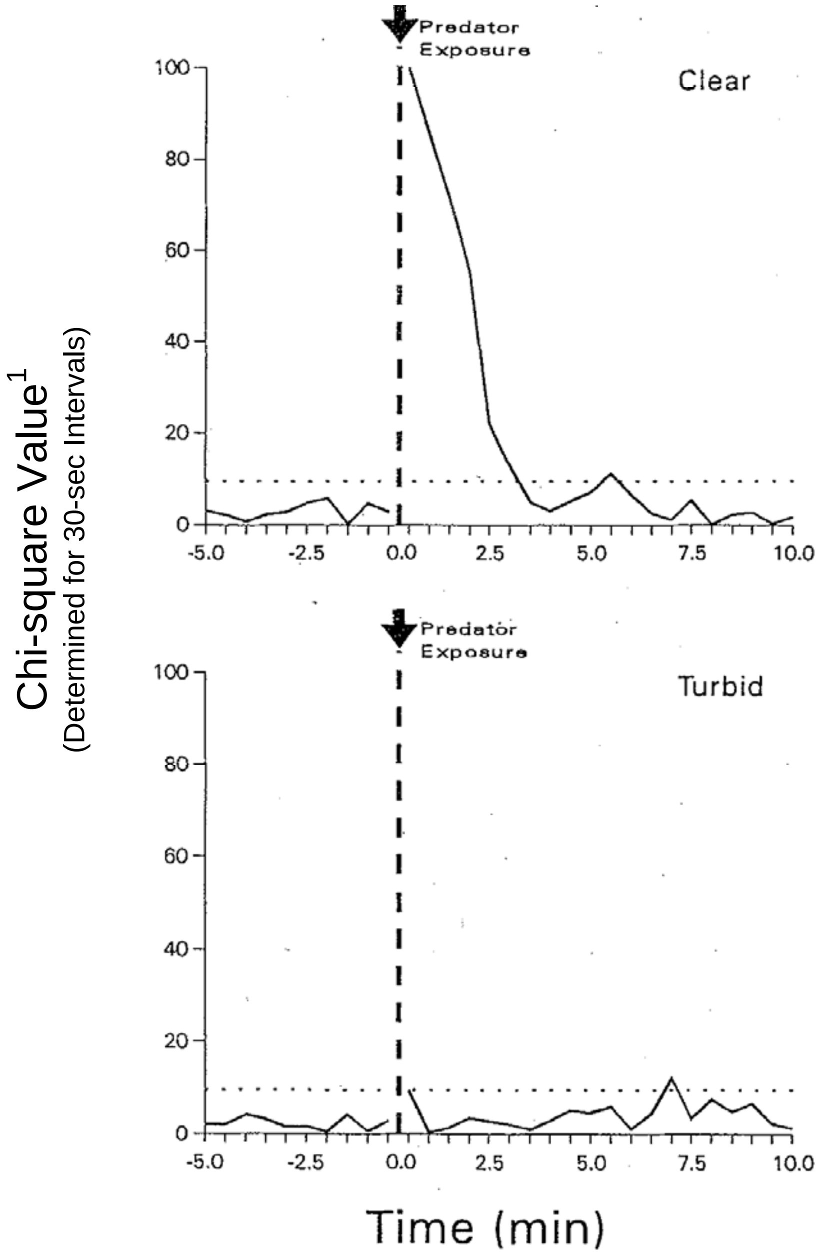
Figure 6-5. Response of Juvenile Chinook to Exposure to Predator Models in Clear and Turbid (23 NTU) Conditions; Proportion of Chinook in the Deepest Region (2) of the Experimental Arena (Gregory 1993).



Note: First of two replicates is shown; horizontal line represents critical chi-square value.

[1] Chi-square was determined for each 30-s time interval before and after exposure to the models. The first occurrence of three consecutive time intervals after exposure to a model that fell below the critical value of the chi-square ($X^2_{(a = 0.05;\ df = 4)} = 9.488$) was arbitrarily identified as fish recovery from the model's effects. The time elapsed before this occurrence was the "recovery time."

Figure 6-6. Changes in Distribution in Juvenile Chinook at 30-s Intervals, before and after Exposure to a Model Fish Predator, in Clear and Turbid Water (Gregory 1993).



Note: First of two replicates is shown; horizontal line represents critical chi-square value.

[1] Chi-square was determined for each 30-s time interval before and after exposure to the models. The first occurrence of three consecutive time intervals after exposure to a model that fell below the critical value of the chi-square ($X^2_{(a = 0.05;\ df = 4)} = 9.488$) was arbitrarily identified as fish recovery from the model's effects. The time elapsed before this occurrence was the "recovery time."

Figure 6-7. Changes in Distribution in Juvenile Chinook at 30-s Intervals, before and after a 3-s Exposure to a Model Bird Predator, in Clear and Turbid Water (Gregory 1993).

the daily instantaneous per capita predation rate. Mean daily instantaneous per capita predation rates in non-vegetated compartments in turbid conditions were 19 to 41 percent lower than those in controls for Chinook, chum and sockeye salmon prey, but the effect of turbidity alone was not statistically significant. In contrast, the mean daily instantaneous per capita predation rates in clear water compartments with vegetation were 29 to 75 percent lower than those in controls and the difference was statistically significant. The effect of vegetation and turbidity combined was tested for significance for sockeye and Chinook salmon, but only the presence or absence of vegetation had a significant effect on daily instantaneous predation rate (Figure 6-8). The authors acknowledge the difficulty in evaluating the species specific and size specific effects on predation rates because there was little overlap in size ranges of species used. However, small juvenile Chinook salmon were 75 percent more vulnerable to predation than larger juvenile Chinook salmon that were only 13 percent longer.

These studies suggest that moderately turbid conditions could provide potential cover for juvenile fish, which may reduce perceived predation risk and allow for greater freedom of movement and foraging.

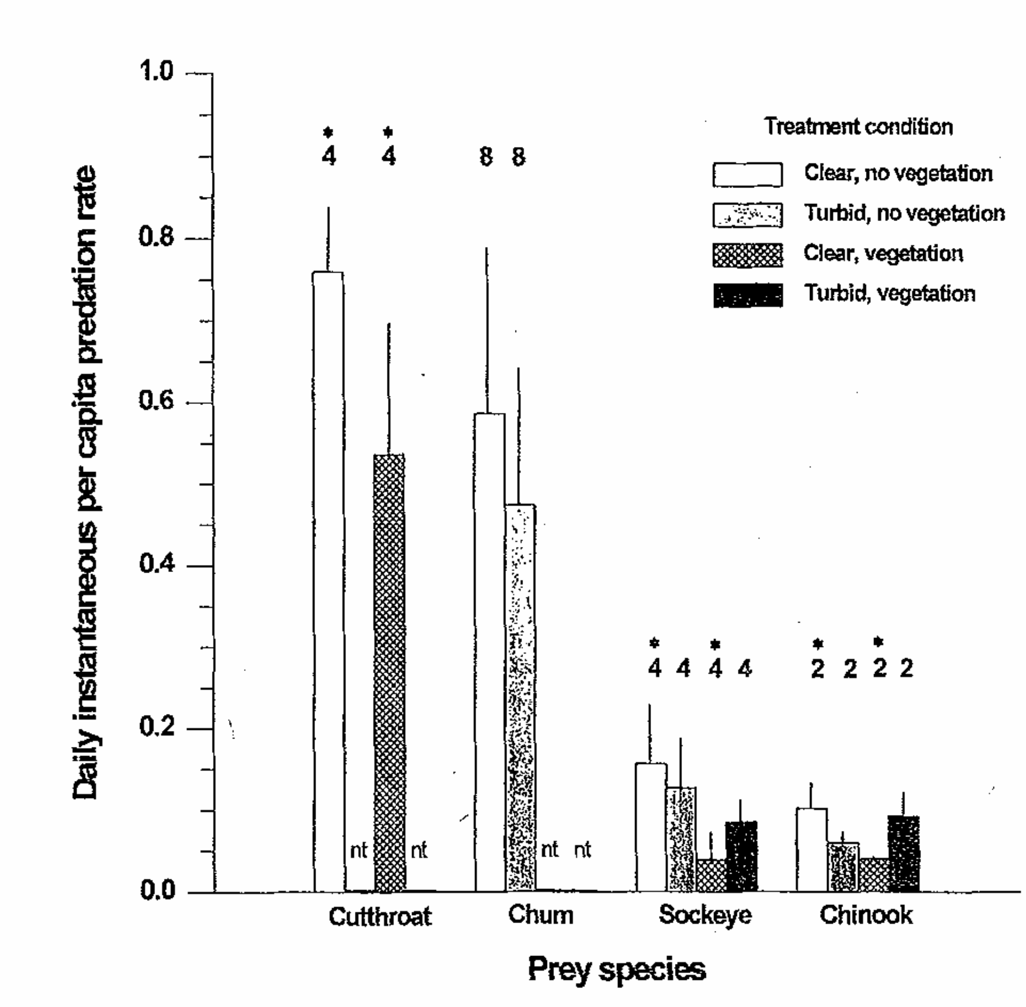
6.1.4 FEEDING

Turbidity can effect feeding success by reducing the ability of the predator to detect or catch its prey. Underwater vision also is influenced by light level. At certain intensities, there is no longer enough light to discern shapes, even in clear water.

To the predator and prey, the effect of turbidity on feeding success is a relative scale that is based upon the size of the animals. The distance that the predator sees and orients to the prey is termed the reaction distance. Large, predatory fish require reaction distances in the 80- to 100-cm range to detect, pursue and capture prey. Fish of this size require low turbidities to successfully sight feed. The reaction distance is shorter for smaller fish (larvae and fry) than large fish (juvenile or adult fish). Similarly, this distance would be even shorter for visual feeding invertebrates. Consequently, the visual feeding ability of larger animals will be affected at lower turbidity levels than of smaller fish and invertebrates. This means that all else being equal, small fish would be able to continue to feed in water of higher turbidity levels than larger fish if they are using vision alone as the main sense to detect and capture prey.

Most feeding rate studies have been carried out on juvenile fish using live or dead invertebrates as prey items. A few studies have used adult trout as predators and juvenile salmonids as the prey or largemouth or smallmouth bass (*M. dolomieu*) at subadult sizes with minnows or crayfish as prey. No study was found addressing feeding behaviors or predation rates of native California minnows.

Vogel and Beauchamp (1999) studied the effects of light, prey size and turbidity on the reactive distance of adult lake trout (330-456 mm TL [total length]). Hatchery juvenile rainbow trout (*O. mykiss*) and juvenile cutthroat trout (*O. clarki*) were used as prey



Note: Vertical lines on bars are SD; nt = no treatment; * = treatments significantly different from each other within species, t-test; numbers above each bar indicate number of replicates.

Figure 6-8. Mean Daily Instantaneous Per Capita Predation Rate by Adult Cutthroat Trout Predators on Juvenile Cutthroat Trout, Chum Salmon, Sockeye Salmon, and Chinook Salmon in Clear and Turbid Water, in the Presence and Absence of Artificial Vegetation (Gregory and Levings 1996).

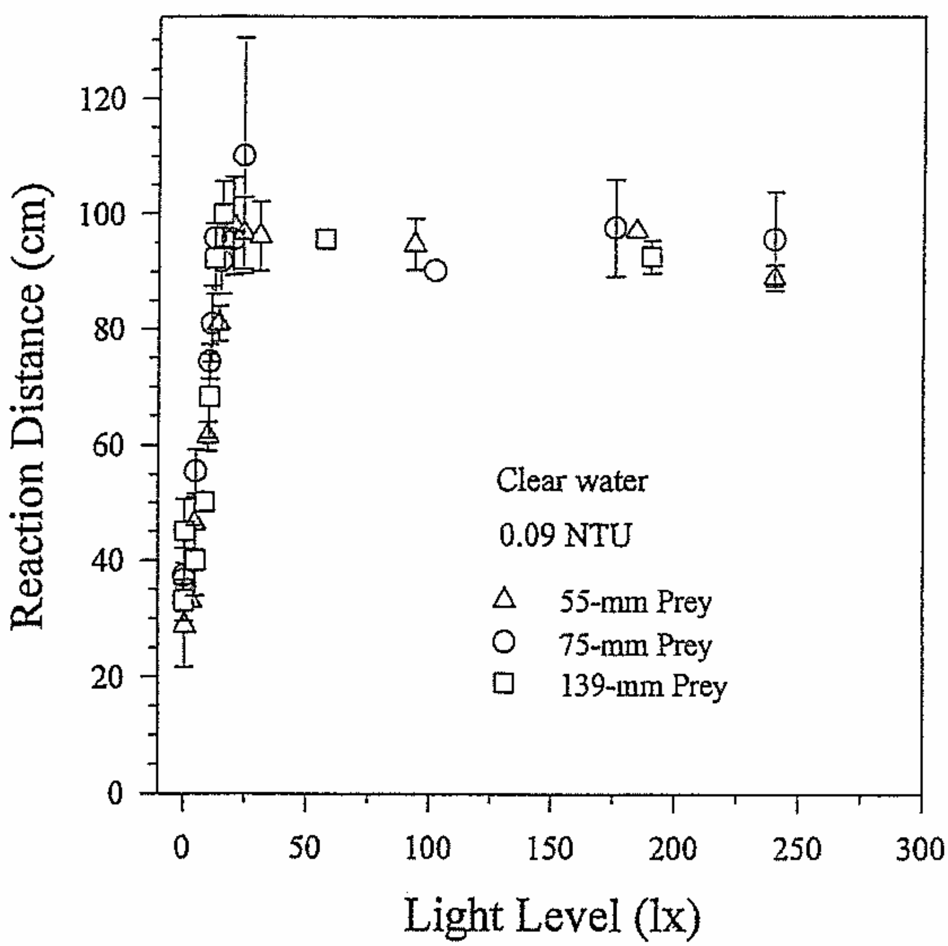
species (lengths of 55 ±1 mm, 75 ± 2 mm and 139 ± 2 mm). Large holding tanks were exposed to different light levels from 0.1 Lux (lx) to 261 lx, and turbidities of 0.09, 3.18 and 7.40 NTU. As a reference condition, midday light levels in natural lake, reservoir or marine systems are 200 to 20 lx at depths of 25 to 40 meters between midcrepuscular (mid-twilight) periods (when light levels are near 0.17 lx). All prey were considered to be visually similar for the purpose of the study and no mention was made of any behavioral differences between the two species. The study found that reaction distance increased from < 25 cm at 0.17 lx to about 100 cm at threshold light intensity of 17.8 lx. The threshold light intensity, called the saturation intensity threshold (SIT), is significant because it sets a maximum reaction distance to prey (the distance where there is no further advantage to the predator and no increased risk to prey) (Figure 6-9). Above this threshold, reaction distance declined as a decaying power function of turbidity (Figure 6-10). Turbidity was found to be a significant factor in reducing the reaction distance of lake trout. The authors note that Miner and Stein (1996) examined reaction distances between prey fish and piscivores over a much larger turbidity range (0.3-91.0 NTU), but 80 percent of the observed decline in reaction distance occurred between 0 and 5 NTU.

Noggle (1978) fed shelled caddisfly to coho salmon smolts (131-145 mm) exposed to constant levels of sediment concentrations in six different tests and counted the number eaten to determine feeding efficiency. The number of caddis fly larvae eaten decreased from a maximum in clear water to no feeding at SSCs above 300 mg/L.

Ginetz and Larkin (1976) measured the rate of predation by wild rainbow trout (25-35 cm) on newly emerged sockeye salmon fry under simulated moonlight and cloudy night light intensities in turbid and clear water. Turbid conditions were generated by a solution of Bismark Brown dye (0.123 g/L – no NTU equivalent provided). Experiments were run at two different water velocities under the two different light intensities in clear and turbid water with either button-up fry or sac-fry. Predation rates were higher for the earlier fry developmental stage, and at the higher light intensity (moonlight), the lesser turbidity and the slower water velocity (Figure 6-11).

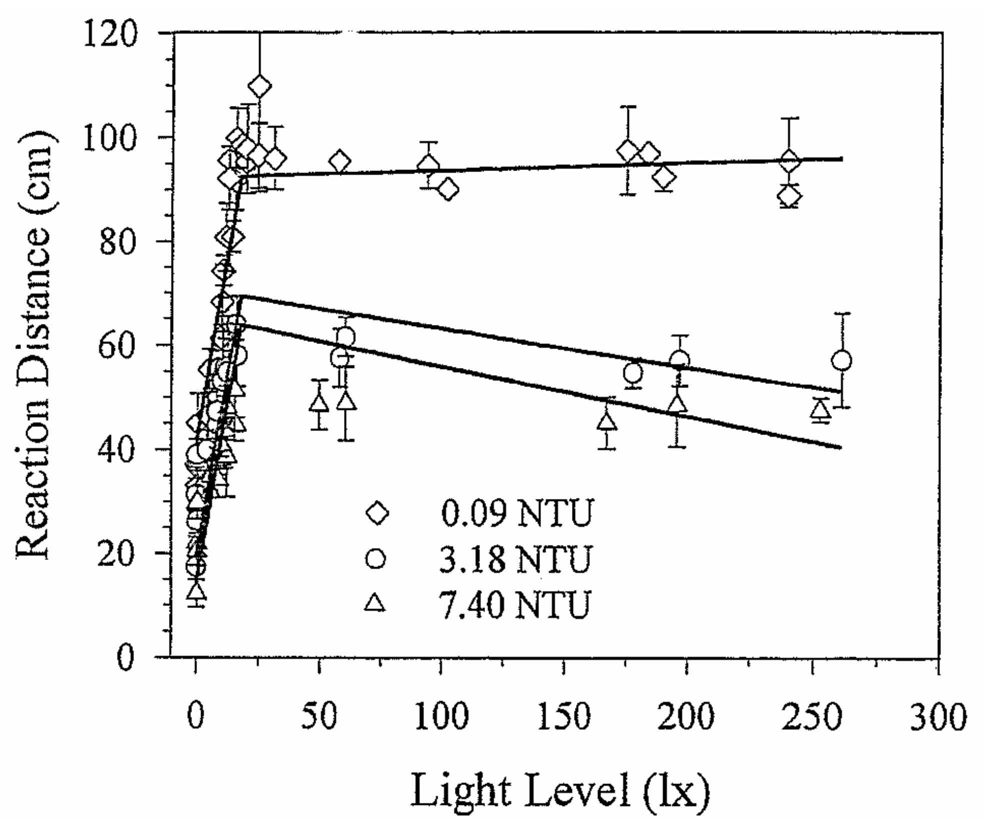
Gregory (1990) conducted feeding studies of juvenile Chinook salmon (no size data specified) on *Artemia* (brine shrimp) and *Tubifex* (worms) at SSCs of 0, 12.5, 25, 50, and 100 mg/L and with one fish at 200 and 400 mg/L. *Artemia* represented midwater prey and *Tubifex* represented benthic prey. Reaction distance to midwater prey decreased exponentially with increase in SSC, from approximately 30 to 40 cm at 0 NTU to about 20 cm at 12 NTU and 10 cm at about 40 NTU. Benthic foraging rates of juvenile Chinook salmon were highest at intermediate SSCs of 50 to 100 mg/L, and lowest at 0 mg/L and 800 mg/ (Figure 6-12).

In a follow-up study, Gregory and Northcote (1993) assessed foraging rates of juvenile Chinook salmon (60-70 mm FL) exposed to surface (frozen *Drosophila*), planktonic (dead *Artemia*) and benthic (*Tubifex*) prey in turbid laboratory conditions (< 1, 18, 35, 70, 150, 370, 810 NTU). The effect of turbidity (seven levels ranging from 0.5 to 243 NTU) on reactive distance to planktonic prey (*Artemia*) also was determined using video



Note: Data points represent the means of ± 2 SE of individual reaction distances for 4-11 lake trout at each combination of light and prey size.

Figure 6-9. Reaction Distances of Lake Trout (330-456 mm Total Length) to 55-, 75-, and 139-mm Prey as a Function of Light (0.17-240 lx) in Clear Water (0.09 NTU) (Vogel and Beauchamp 1999).



Note: Data points represent the means ± 2 SE of individual average reaction distances for 4-11 lake trout at each combination of light and turbidity.

Figure 6-10. Unified Model of Lake Trout Reaction Distances as a Function of Light (0.17-261 lx) and Turbidity (0.09, 3.18, and 7.4 NTU) Pooled over the Three Prey Sizes (Vogel and Beauchamp 1999).

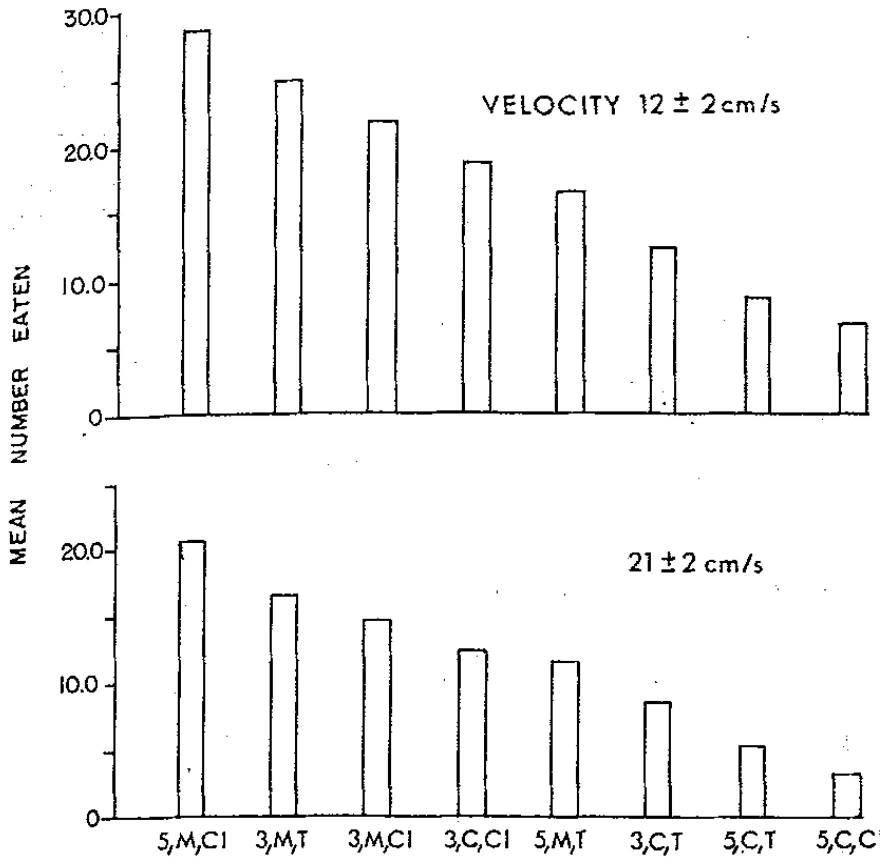
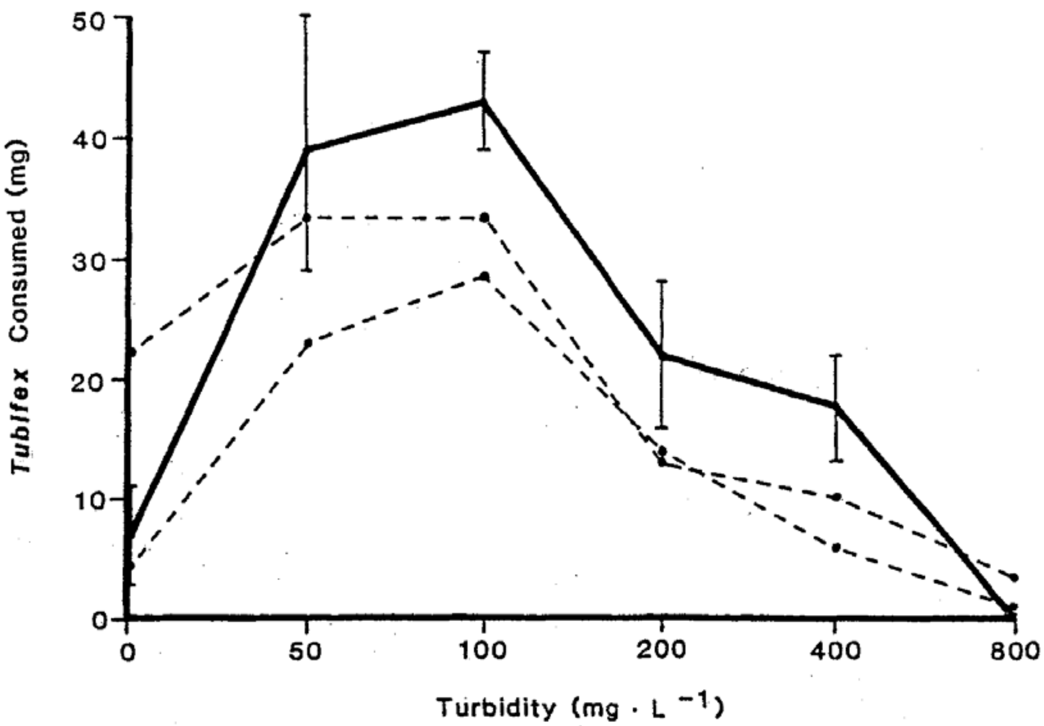


Figure 6-11. Mean Numbers (8 Replicates) of Sockeye Salmon Fry (*Oncorhynchus nerka*) Eaten when 50 were Exposed to 5 Rainbow Trout (*Salmo gairdneri*) in Experimental Streams at Two Levels of Each of Light Intensity (moonlit = M, Cloudy = C), Water Turbidity (Turbid = T, Clear = Cl), Fry Development (Yolk Partially Absorbed = 3, Yolk Fully Absorbed = 5) at each of Two Water Velocities (Ginetz and Larkin 1976).



Note: Vertical bars represent standard error bars on Replicate 1.

Figure 6-12. Feeding Rates of Juvenile Chinook Salmon on *Tubifex* Prey Under Laboratory Conditions at Different Turbidity Levels (Gregory 1990).

cameras. Chinook salmon were found to exhibit a log-linear decline in reaction distance with increasing turbidity, given as

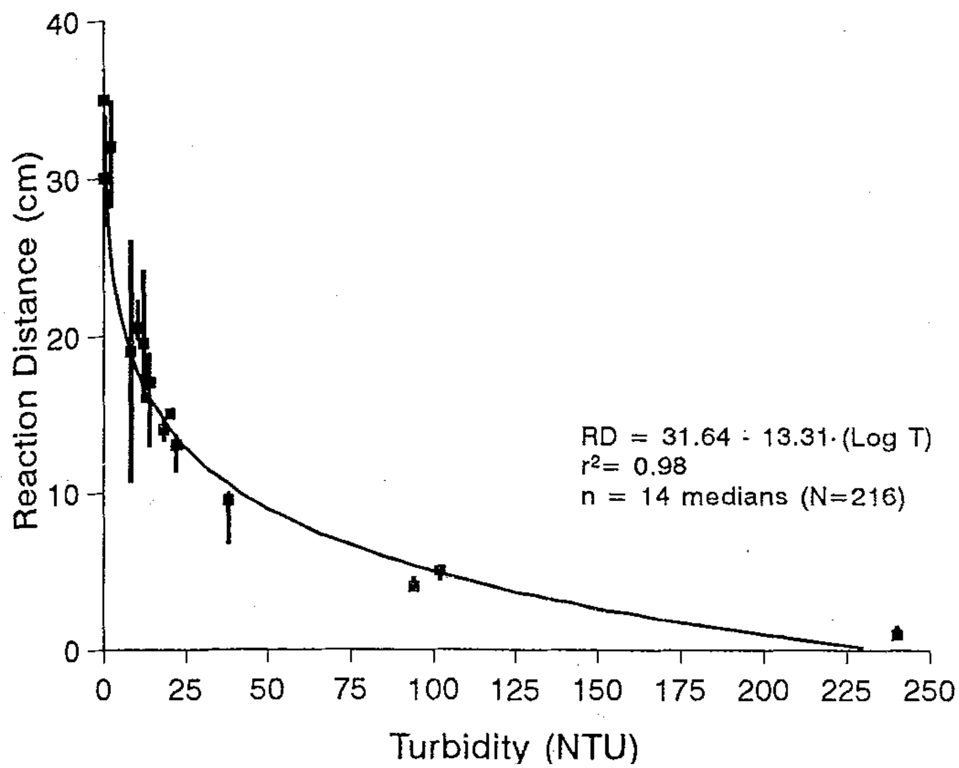
$$RD = 31.64 - 13.31 X \log T$$

where: T = turbidity (in NTUs) and RD = reaction distance (in cm).

For clear water, the reaction distance was about 35 cm and declined about 10 cm around 35 NTU (Figure 6-13). Foraging rates were reduced at higher turbidities for all three prey species. For surface and benthic prey, foraging rates also were low in clear water. Benthic and surface foraging rates followed the same general pattern, with the highest rates found at intermediate turbidity levels of 35 to 150 NTU and reduced rates at the lowest and highest treatment levels (Figures 6-14, 6-15). High variability was exhibited in the surface feeding experiments with some fish not feeding at all and others consuming 40 or more prey items. The effect of turbidity on planktonic foraging also was significant. Foraging rates were high at turbidities up to 70 NTU, then declined rapidly (Figure 6-16). The authors reflect on the large amount of literature concluding that turbidity is harmful to salmonids, but suggest that, for some juvenile salmonids, turbidity may be required to successfully feed. They note that the effect of turbidity on foraging behavior is inconsistent in the literature.

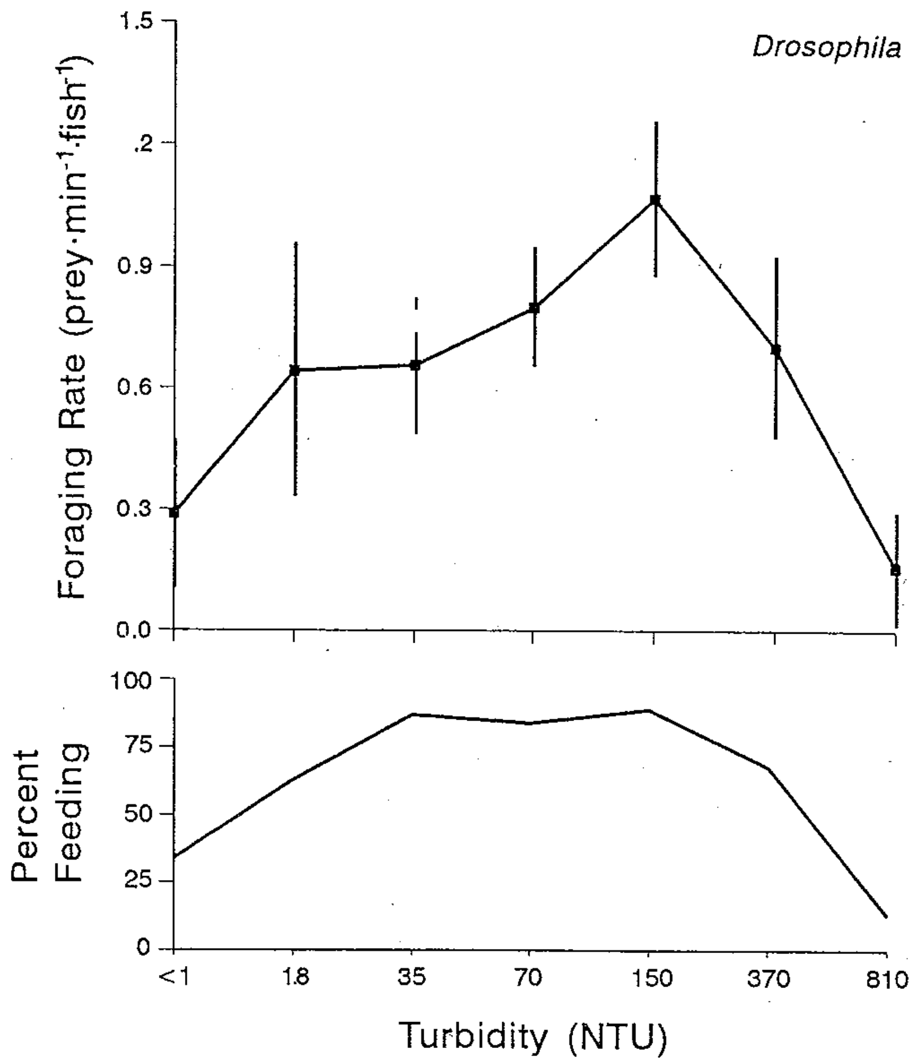
Rowe et al. (2003) investigated the effects of turbidity on the ability of juvenile rainbow trout to feed on limnetic prey (*Daphnia spp.)* and larval benthic prey (Chironomid and *Deleatidium spp*, a large mayfly). The authors used live prey in all experiments because they believe that movement of prey influences probability of capture. Feeding experiments were carried out in laboratory tanks at turbidities of 10, 20, 40, 80, 160 and 320 NTUs. Turbidity levels of 0 (control), 20, 40, 80, 160 and 320 NTUs were used to test the ability of trout to select benthic prey on the basis of size (small, medium and large). Feeding rates were maintained up to 160 NTU on *Daphnia* and up to 320 NTU on benthic prey. Trout selected large mayflies and Chironomid prey and rejected small prey in clear water, but the ability to select large prey and reject small ones declined as turbidities increased. Positive selection for large mayflies was evident up to turbidities of 80 NTU, but decreased markedly by 160 NTU and there was no size selection at 320 NTU. Trout were also able to feed on Chironomid larvae in the complete absence of light in one experiment, with feeding rates slightly less than under normal lighting conditions. It was surmised that trout use non-visual methods to detect and capture prey. The study results seem to contradict studies by Sigler et al. (1984) that documented decreased growth rates at turbidities as low as 38 NTU. The authors attribute this to the fact that Sigler used frozen brine shrimp. Another explanation may be that this study exposed test fish held in small areas with relatively high concentrations of food in turbid conditions for only a 2 to 3 hour period.

Chronic turbidity levels may affect trout feeding methods and growth rates. In a field study in the chronically turbid water of the McCloud river, Tippets and Moyle (1978) documented invertebrate abundance on stream substrate and in the drift and compared the



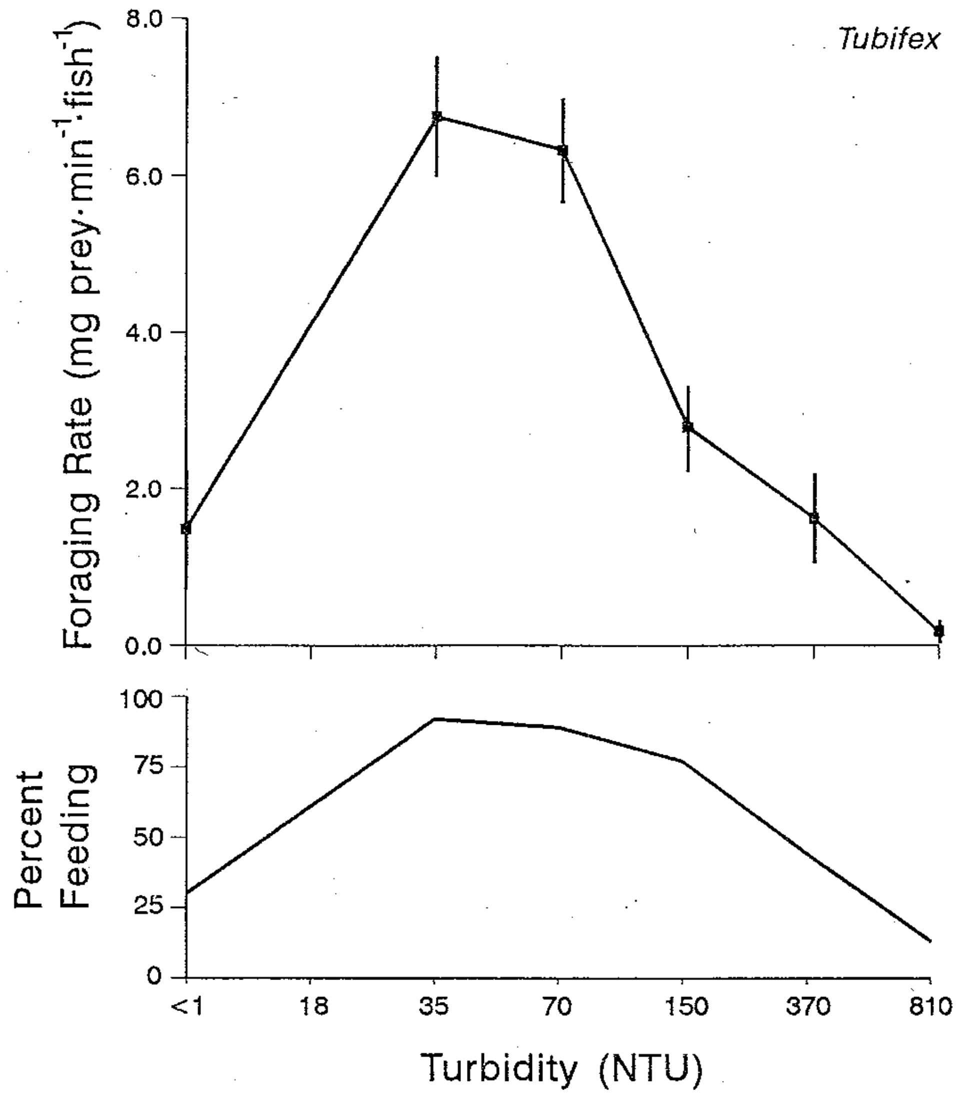
Note: Regression based on median values; bars include 50% of data points.

Figure 6-13. Effect of Turbidity on the Reaction Distance of Juvenile Chinook for *Artemia* Prey (Gregory and Northcote 1993).



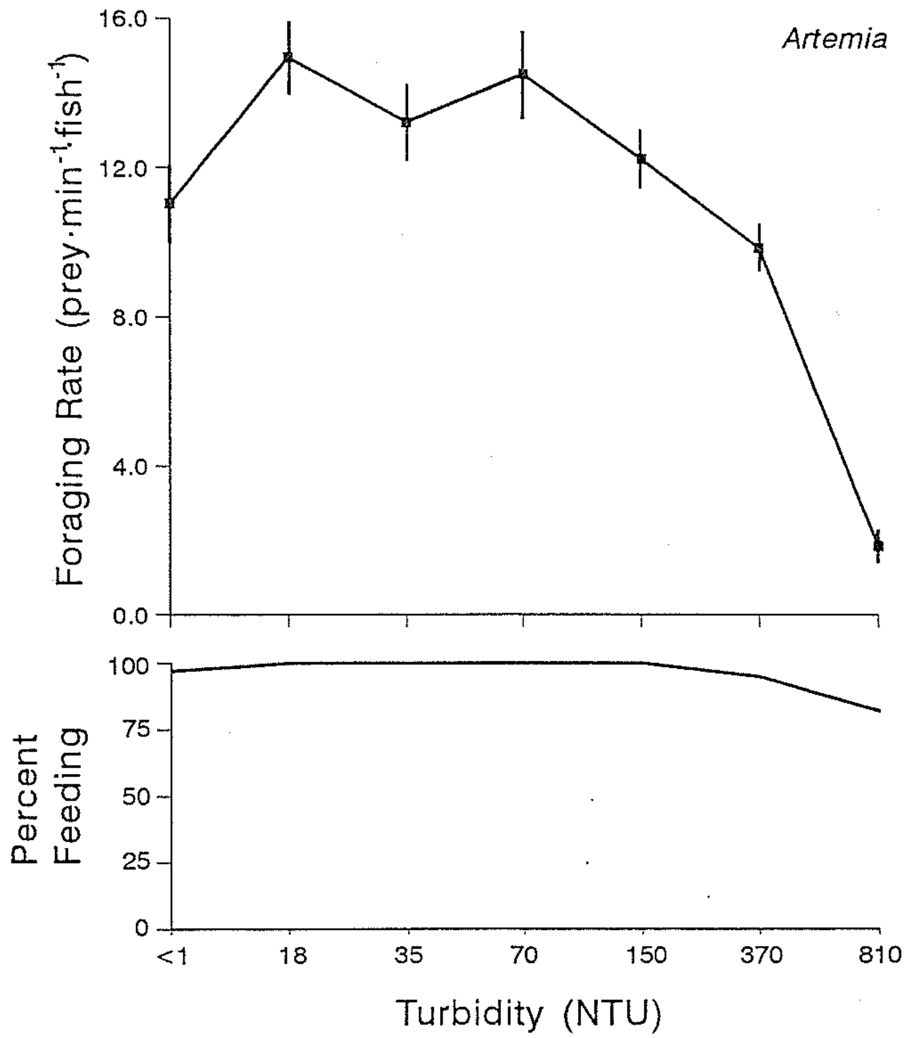
Note: Vertical bars indicate the standard error of means from five trials.

Figure 6-14. Effects of Turbidity on Mean Foraging Rate of Juvenile Chinook Feeding on Surface Prey (*Drosophila*) and the Percentage of Salmon Foraging in 70-L Aquaria in 10.0-min Trials (Gregory and Northcote 1993).



Note: Vertical bars indicate the standard error of means from five trials.

Figure 6-15. Effects of Turbidity on Mean Foraging Rate of Juvenile Chinook Feeding on Benthic Prey (*Tubifex*) and the Percentage of Salmon Foraging in 70-L Aquaria in 5.0-min Trials (Gregory and Northcote 1993).



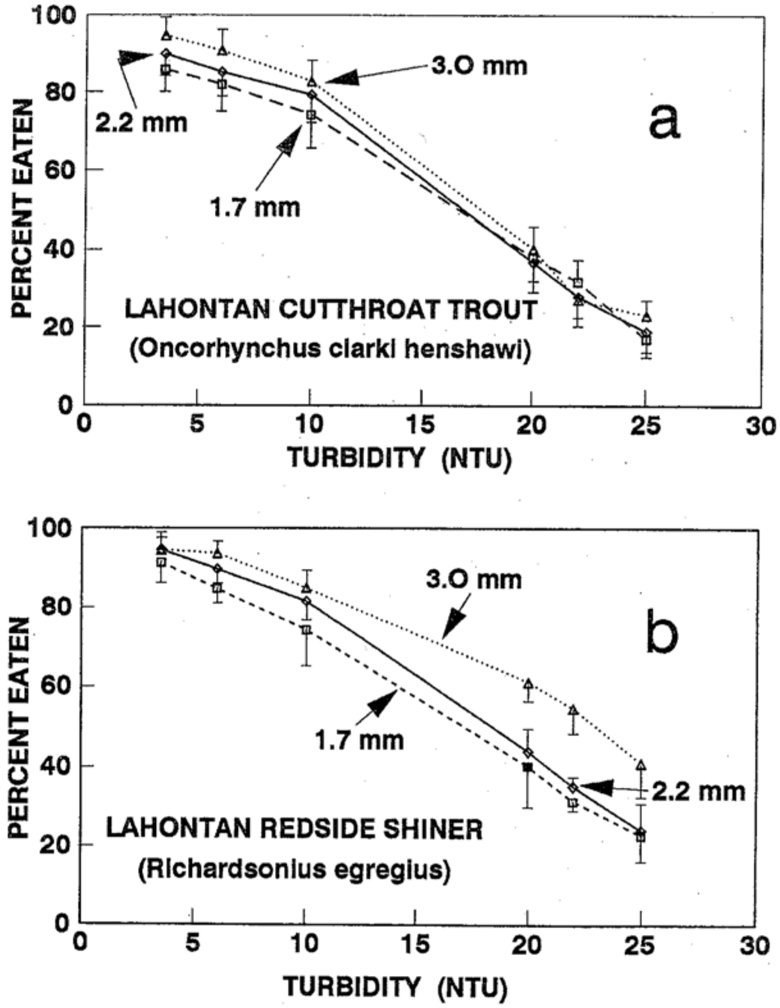
Note: Vertical bars indicate the standard error of means from five trials.

Figure 6-16. Effects of Turbidity on Mean Foraging Rate of Juvenile Chinook Feeding on Planktonic Prey (*Artemia*) and the Percentage of Salmon Foraging on 70-L Aquaria in 1.0-min Trials (Gregory and Northcote 1993).

collected samples to gut samples of juvenile and adult rainbow trout. The study showed age 0+ trout depended on drift aquatic invertebrates and age 1+ trout utilized an intermediate pattern of drift and benthic invertebrates. However, gut contents of adult trout were primarily comprised of active benthic invertebrates (that had low drift rates) and drifting terrestrial invertebrates, and they contained large amounts of indigestible material from the stream bottom. The gut contents of adults were full during the day with no indication of morning or evening feeding peaks typical of river dwelling rainbow trout. The evidence strongly suggests that adult McCloud River rainbow trout are actively feeding on invertebrates picked off the bottom substrate during daylight. Most drift occurs at night, and typically, rainbow trout drift feeding peaks in the early morning and evening hours. This strategy does not work in the McCloud River because the high turbidity does not provide sufficient reaction distance for fish to drift and surface feed successfully. The extra energy cost of the fish actively feeding on the bottom and the large volume of indigestible material in the gut are probable reasons for the slow growth of fish in the McCloud River.

Turbidity can also be a factor in competition for resources, providing some species with a greater competitive advantage under turbid conditions. In Summit Lake in Nevada, the spawning population of Lahontan cutthroat trout (*O. clarki henshawi*) has declined and abundance of Lahontan redside shiners (*Richardsonius egregius*), probably introduced as bait fish, has increased. Land use in the watershed has made Summit Lake highly turbid (Vinyard and Yuan 1996). Vineyard and Yuan (1996) studied the effect of turbidity levels commonly observed in Summit Lake on feeding rates of Lahontan cutthroat trout and Lahontan redsides (fish lengths 70 to 93 mm SL). *Daphnia*, sorted by size, were introduced into tanks with six levels of turbidity from 3.5 to 25 NTU. Fish were acclimated in the tanks for 24 hours, allowed to feed for 2 hours, then removed and the tank was drained and the remaining *Daphnia* were counted. It was found that feeding rates of both species varied inversely with turbidity, but feeding rates of shiner were greater than trout at all turbidity levels. The decrease in feeding rates was linear and at 25 NTU had decreased by 80 percent for trout and 60 to 80 percent for redsides (Figure 6-17). Redsides consumed 3 percent more prey than cutthroat trout at low turbidities (5 NTU) and 10 percent more at high turbidities. In general, both species consumed large prey at higher rates, and at low turbidity (3.5 NTU) 90 percent of prey was consumed. However, redsides showed increasing predation on large *Daphnia* at NTUs of 20 and higher, while cutthroat trout showed no prey size selection at high turbidity (≥20 NTU). Observations during the laboratory study also showed that redsides searched faster and more widely at high turbidities compared to their search patterns in clear water. These results suggest that Lahontan redside shiner may be more effective as zooplankton predators at high turbidity levels than Lahontan cutthroat trout.

No information on foraging behavior of Sacramento pikeminnow and hardhead was found in the published literature. Both fish species are visual feeders and typically inhabit clear water streams. Both species can attain large sizes (300-600 mm F/L) as adults, and therefore may require relatively large reaction distances to successfully feed.



Note: Four fish of each species were exposed to prey of a single size for 2-h feeding trials. *Daphnia magna* prey sizes are as indicated. Vertical bars indicate 1 standard deviation.

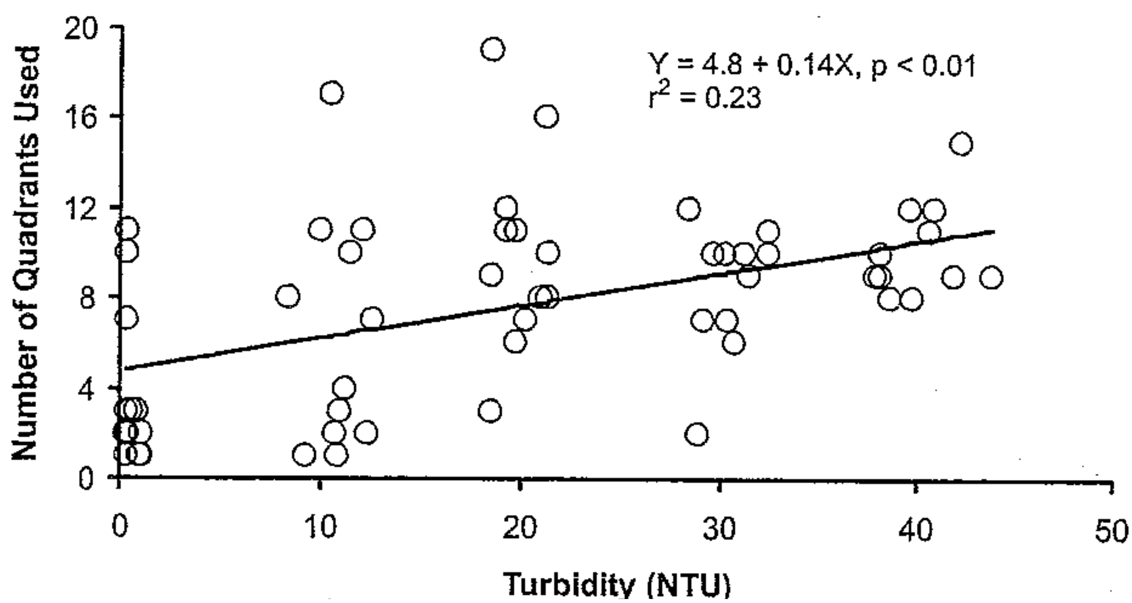
Figure 6-17. Mean Percent Prey Consumed in Relation to Turbidity. Upper Panel (a) Shows Results from Feeding Trials with Lahontan Cutthroat Trout (*Oncorhynchus clarki henshawi*) and Lower Panel (b) Shows Results from Lahontan Redside Shiner (*Richardsonius egregius*) (Vinyard and Yuan 1996).

In the absence of published studies on the native Sacramento pikeminnow and hardhead, the results of studies on other visual predators (non-native brook trout, largemouth and smallmouth bass) are reviewed to expand our discussion of how turbidity affects reaction distance and feeding behavior of fish.

Sweka and Hartman (2001a) investigated brook trout reactive distance and foraging behavior in turbid water. They videotaped the reaction distance of brook trout (mean length 126 mm, range 77-153 mm) preying on live house fly larvae in an artificial stream at turbidities of ranging from 0 to 43 NTU. Reaction distance was about 80 cm in clear water (<3.0 NTU) and showed a nonlinear decrease to about 40 cm at 10 NTU, 20 cm at 20 NTU, and 12 cm at >30 NTU. The largest percent change in reactive distance occurred between 0 and 15 NTU. Increasing turbidity had the effect of masking detection of the prey by the predator. As turbidity increased, the probability of reacting to prey decreased. Brook trout adjusted their feeding behavior to compensate for the reduced detection distance by foraging more actively. A movement study conducted as part of the experiment showed that the number of quadrants within the artificial stream used to forage increased with increasing turbidity (Figure 6-18). Mean daily consumption was not affected by turbidity (Figure 6-19), but specific growth rates decreased with increasing turbidities (Figure 6-20). Foraging success was governed by the ability to detect and react to prey, but success after detection was not influenced by turbidity. Higher turbidities had no influence on the probability of attack once the prey had been detected, on the probability of capture once the attack had been initiated, or on the probability of ingestion once the prey had been captured.

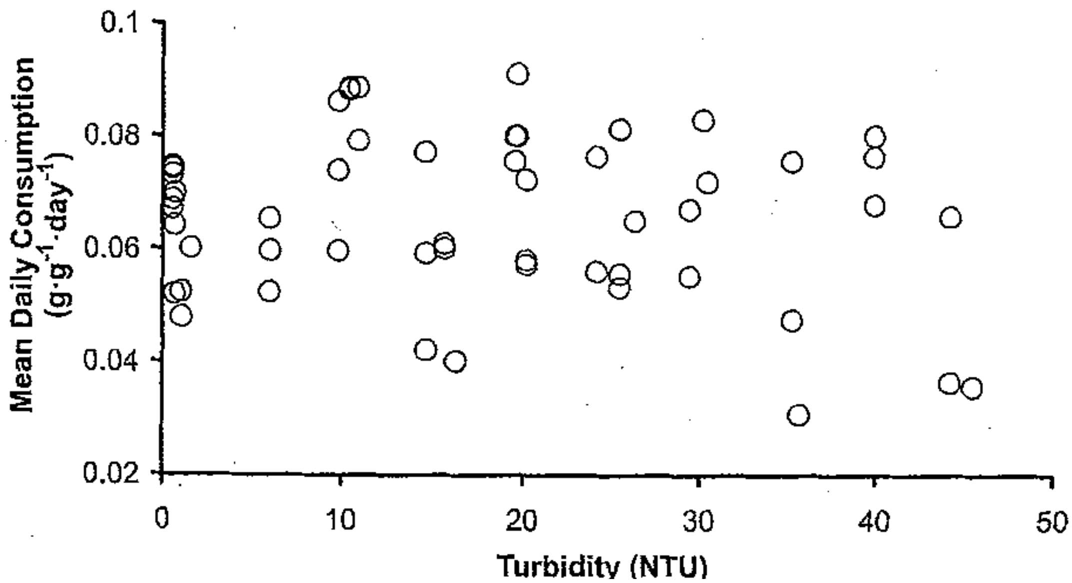
Brook trout mean daily consumption and growth rates were evaluated at turbidities from of <3.0 to 50 NTU in 5-day trials (Sweka and Hartman 2001b). There was a 62 percent change in growth rate at the highest turbidities compared to clear water, but there was no significant change in growth rate at turbidities below 25 NTU. There was no relationship between mean daily consumption and NTU. Brook trout shifted from drift feeding to an active searching behavior between 10 and 20 NTU. As has been suggested for rainbow trout (Tippets and Moyle 1978), these experimental results suggest that brook trout in turbid conditions have higher energetic costs and would likely experience lower growth rates.

Crowl (1989) carried out experiments in clear and turbid water to examine effects of prey (crayfish) size, orientation, and movement on adult largemouth bass reaction distance. Adult largemouth bass (280-300 mm total length [TL]) and crayfish (*Procambarus acutus*) (16.6 to 28.5 mm carapace length) were used as predator and prey, respectively, in clear (1-3 JTU) and moderately turbid waters (17-19 JTU). In clear water, there was a statistically significant, positive relationship between reactive distance and prey size, and a significant increase in reaction distance when the prey was moving, but no difference in reaction distance to orientation of the prey. In turbid water, reactive distance was independent of prey size and, contrary to theoretical predictions, prey movement did not affect reactive distance. Reactive distance ranged from about 70 to 270 cm in clear water



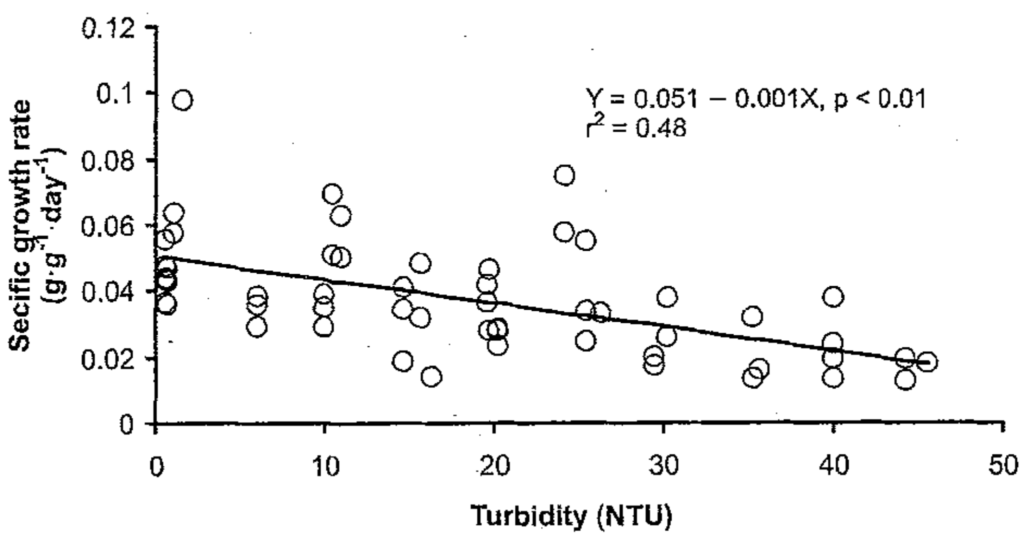
Note: Each point represents the number of quadrants used by brook trout to forage during one videotaping session. The number of quadrants used to forage within the artificial stream increased significantly as turbidity increased ($F = 28.41$, $p < 0.01$). Individual fish also differed from one another in the number of quadrants used ($F = 14.83$, $p < 0.01$).

Figure 6-18. Regression of the Number of Quadrants Used to Forage (*Y*) on Turbidity (*X*) by Brook Trout (Sweka and Hartman 2001a).



Note: Although individual fish differed from each other ($F = 5.10$, $p < 0.01$), turbidity did not have a significant influence on mean daily consumption ($F = 0.49$, $p = 0.48$).

Figure 6-19. Regression of Standardized Mean Daily Consumption Values on Mean Turbidity Level for Each Trial (Sweka and Hartman 2001a).



Note: Specific growth rates decreased significantly with increasing turbidity (F = 33.87, p < 0.01). Individual fish did not differ from each other (F = 0.61), p < 0.78).

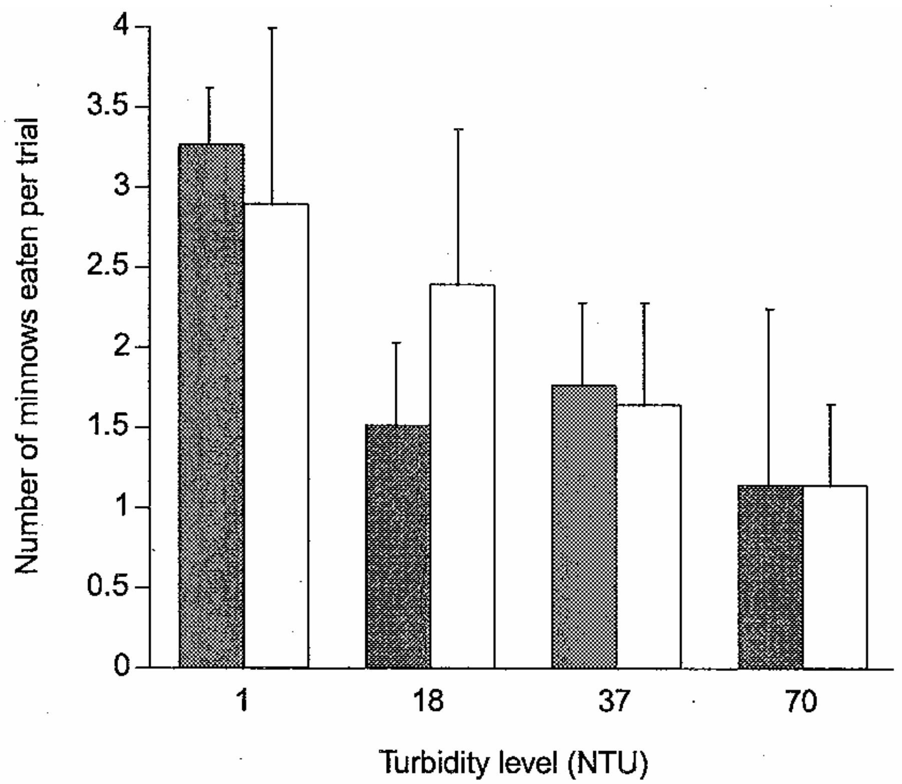
Figure 6-20. Regression of Specific Growth Rate (*Y*) on Mean Trial Turbidity (*X*) (Sweka and Hartman 2001a).

and 25 to 40 cm in turbid water. There were behavioral differences in turbid water; 38 out of 40 predators struck at a rock offered as prey in turbid water while no strikes occurred in clear water. The authors suggest that when prey are highly visible, predators attack after prey recognition, but when prey are less visible, predators attack immediately upon sighting. They suggest this "switching tactic as a function of water clarity" may benefit visual predators, such as largemouth bass, which consume few, relatively large prey, but differs from predators that must consume large numbers of small prey (such as planktivorous fish).

During *in situ* feeding trials in Lake Ontario, Reid et al. (1999) found no significant differences in capture success for largemouth bass (195-245 mm FL) predation on red belly dace (*Phoxinus eos*) between different coastal wetland areas, where turbidities were 2.3 and 20 NTU. A comparison of gut contents of wild bass (from clear and turbid habitats) with local prey populations suggest that availability of prey, rather than water clarity, is the primary factor in determining diet. Reid et al. (1999) also conducted one-hour laboratory feeding trials with juvenile largemouth bass (83-130 mm FL) as the predator and fathead minnows (*Pimephales promelas*) as the prey. As turbidity increased, the number of fathead minnows captured (with four size classes combined) generally decreased (Figure 6-21), but among pairwise comparisons, the difference was statistically significant only between one and 70 NTU ($p = 0.004$), the highest and lowest turbidities studied. Largemouth bass selected the smallest size class of fathead minnow at the lowest turbidities, but capture rates of small fathead minnows decreased with increasing turbidity, with little change in capture rate of large minnows (Figure 6-22).

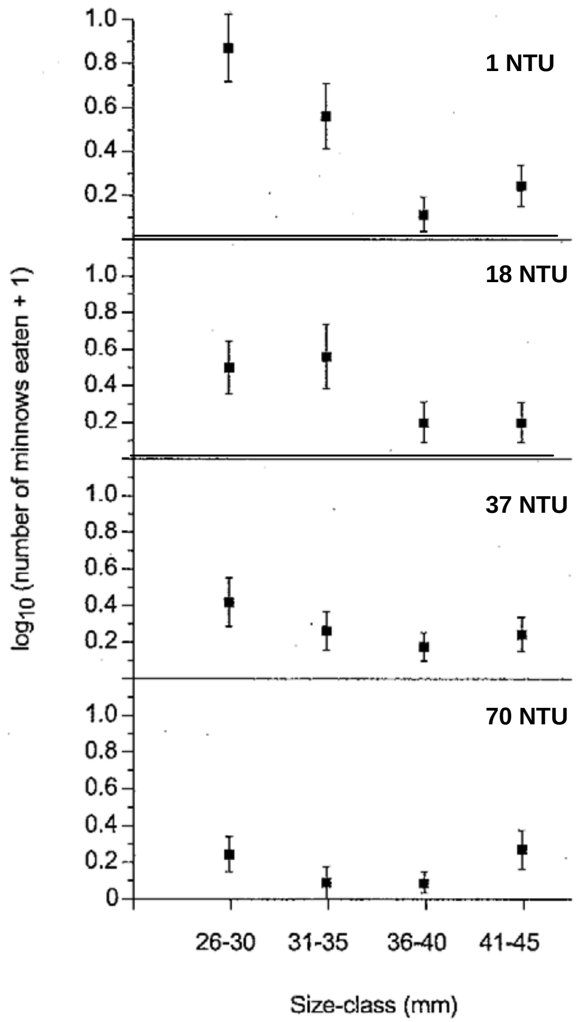
Sweka and Hartman (2003) studied reaction distance and foraging success of wild and hatchery smallmouth bass (mean 99 mm TL, range 87-155 mm) in turbidity conditions of <5 NTU to 40 NTU. The reaction distance was about 65 cm in clear water and decreased non-linearly with increasing turbidity to 10 cm at the highest turbidity, with the greatest rate of decrease from 0 to approximately 25 NTU. Turbidity significantly reduced the probability that a fish would react to a prey item. However, once the prey was recognized, there was no decrease in initiating an attack, capturing, or ingesting prey. Turbidity decreased foraging success by reducing the probability of encountering prey.

Bonner and Wilde (2002) note that the naturally high suspended sediment loads in prairie streams in the central US may be important in maintaining the integrity of fish assemblages in these systems and may also be a factor in the low abundance of predatory fish. They investigated the effect of turbidity on feeding of small minnows in these prairie streams. Some of these species appear to be well adapted to feeding in low visibility conditions. For six species of minnows from these streams, there was a general relationship of decreased prey consumption at higher turbidity levels. All species consumed prey at all experimental turbidity levels of 0, 1,000, 2,000 and 4,000 NTUs. Prey consumption was generally highest at 0 NTU. Prey consumption for peppered chub (*Macrhybopsis tetranema*) (50-65 mm) and flathead chub (*Platygobio gracilis*) (72-100 mm) decreased only slightly, by 21 percent and 26 percent, respectively, as turbidity



Note: Vertical bars represent ±1 SE.

Figure 6-21. Comparison of the Mean Number of Fathead Minnows Eaten by Cootes Paradise (Shaded Bars) and Rice Lake (Open Bars) Juvenile Largemouth Bass during 1-h Feeding Trials across Four Levels of Turbidity (Reid et al. 1999).



Note: Vertical bars represent ±1 SE.

Figure 6-22. Comparison of the Mean Number of Fathead Minnows Eaten of Four Size-Classes across Four Turbidity Levels by Juvenile Largemouth Bass (Cootes Paradise and Rice Lake Largemouth Bass Combined) during 1-h Feeding Trials (Reid et al. 1999).

increased, but the difference was not statistically significant. The negative relationship between prey consumption and turbidity was statistically significant for the Arkansas river shiner (*Notropis girardi*) (56-71 mm), emerald shiner (*N. atherinoides*) (62-72 mm), red shiner (*Cyrinella lutrensis*) (56-64 mm), and sand shiner (*N. stramineus*) (65-75 mm) decreasing by 59, 73, 84 and 89 percent respectively for turbidities between 0 and 4,000 NTU. Emerald and red shiners showed a linear decline in prey consumption, while Arkansas River and sand shiners showed higher consumption rates at the mid-level turbidities tested and lower consumption rates at 0 and 4,000 NTUs. The near linear decrease in prey consumption by emerald and red shiners suggests a simple attenuation of vision as a function of increased turbidity. Explanations advanced for the quadratic relationships between prey consumption and turbidity for Arkansas River and sand shiners include improved discrimination of prey at mid-turbidity ranges or increased feeding activity at middle turbidities through reduced risk of predation. The explanation for the lower effects on prey consumption by flathead chub and peppered chub was attributed to morphological adaptations these species possess for feeding in turbid water, including barbels, large numbers of olfactory lamellae and numerous, cutaneous taste buds.

6.1.5 Homing and Migration

Whitman et al. (1982) evaluated sediment avoidance and home water preference of adult Chinook salmon in flow-through artificial streams offering the choice between turbid and clear waters from the fishes' natal stream and non-natal water (city water). Adult male Chinook salmon (mean FL 596 mm, weight 2.5 kg) were used to examine preference or avoidance of water in a Y-maze with and without the addition of ash from Mt. St. Helens. Suspended sediment levels ranged from 373 to 328 mg/L at the entry point of water to the Y-maze to 30 to 295 mg/L at the lower end of the Y-maze. Fish could not be observed at SSC greater than 350 mg/L. Three treatments were evaluated, providing the fish a choice between clear home water and city water; turbid home water and clear city water; and turbid home and city water. The fish selected the home water preferentially in the first and third treatments, but showed no significant preference between the turbid home water and clear city water (Table 6-3). The authors suggest that these results indicate Chinook salmon will stray into non-natal streams when degraded water quality conditions exist.

Whitman et al. (1982) ran another experiment to test the proportion and rate of return of control and ash-exposed groups of 25 control and 25 experimental adult male Chinook salmon (mean FL 596 mm, weight 2.5 kg). After the salmon migrated up the Lake Washington Ship Channel, they were held for seven days in 12,000-liter circular tanks supplied with water from the Lake Washington Ship Channel. Control fish were held in flowing water and experimental fish in aerated water that had about 650 mg/L ash added to it. The fish were marked according to their treatment and then trucked back downstream and released at the near the entrance of the Lake Washington Ship Channel. When return frequencies were analyzed, exposure to ash had resulted in no discernable difference on the return of the fish to the recovery location (Table 6-4).

Table 6-3. Choices of Chinook Salmon between Home Water (HW) and Seattle City Water (CW) in the Presence and Absence of Volcanic Ash (A) (Whitman et al. 1982).

	Treatment I			Treatment II			Treatment III		
Measure	**HW**	**CW**	**No score**	**HWA**	**CW**	**No score**	**HWA**	**CWA**	**No score**
Number of fish	20	5	7	9	11	15	17	2	36
Proportion of releases	62%	16%	22%	26%	31%	43%	31%	4%	65%
Proportion of scores	80%	20%		45%	55%		89%	11%	

Table 6-4. Proportion and Rate of Return of Control and Ash-Exposed Chinook Salmon after Downstream Displacement (Whitman et al. 1982).

Release date (1980) and statistic	Control	Ash-exposed
October 13		
Number released	20	20
Number (%) recovered	13 (65%)	4 (20%)
Average days to return	7	2
October 20		
Number released	20	20
Number (%) recovered	9 (45%)	14 (70%)
Average days to return	2	3
October 29		
Number released	18	20
Number (%) recovered	11 (61%)	15 (75%)
Average days to return	3	2
All dates		
Number released	58	60
Number (%) recovered	33 (57%)	33 (55%)
Average days to return	4	2

6.2 EFFECTS OF SUSPENDED SEDIMENT ON FISH PHYSIOLOGY

Substantial literature has accumulated since the 1930s that suggests that suspended sediment negatively affects fish physiology. Much of this research has focused on suspended sediment-related respiratory impairment by direct impairment of gas exchange across the gills, by gill injury, or by morphologic responses of gill tissue to sediment coating. Recent research has involved further work on that and on certain downstream physiological parameters such as suspended sediment effects on blood glucose, “stress” hormone secretion, packed red blood cell volume (hematocrit), and impaired disease resistance. Many authors also have reported decreased feeding efficiency associated with high levels of suspended sediment. This effect is probably attributable primarily to a combination of decreased prey availability and decreased prey visibility in turbid water (and it was discussed in this context in section 6.1.4), but decreased feeding efficiency also may be related to physiological effects.

Reviews by Cordone and Kelley (1961), Sorensen et al. (1977), Langer (1980), Alabaster and Lloyd (1982), and Waters (1995) discuss these topics. Newcombe and MacDonald’s (1991) review is significant because it laid the foundation for their attempt to model and predict responses of salmonid fish to specific SSCs and duration. Newcombe and MacDonald developed a “Stress Index,” which was further developed in subsequent work (e.g. Newcombe and Jensen [1996], see Section 4.7). Newcombe (2003) modified the Newcombe and Jensen (1996) model to include terms to measure visual water clarity, which is particularly relevant to natural history concerns such as spawning behavior and foraging ecology.

With the exception of altered disease resistance, the reported specific physiological effects of suspended sediment on salmonids have been sublethal. However, direct lethal effects of very high SSCs or extended duration of lower concentrations also have been commonly reported for juvenile and pre-smolt salmonids. The data are difficult to compare among the various studies because of variation in studied natural systems and lack of experimental controls, and because of failure to tabulate central parameters such as SSC or exposure time duration. Newcombe and MacDonald (1991) used the “best” of the available data in a meta-analysis to model severity effects of suspended sediment as functions of concentration and time duration to attempt to predict lethal and sublethal effects of suspended sediment exposure events of known intensity on salmonids. Servizi and Martens (1992) found Newcombe and MacDonald’s model to be unreliable, primarily because it lacked compensation for threshold effects and for other physical factors such as temperature. However, Newcombe and MacDonald (1991) acknowledged those difficulties and stressed that future studies of suspended sediment effects in fish should include accurate measurements of SSCs, time duration of exposure, and organismal response. Newcombe and MacDonald (1991) also recommended that future studies should disassociate observational and experimental systems from confounding variables such as temperature, sediment composition, and ancillary toxic properties of sediments, which plagued many of the earlier studies. Newcombe and Jensen (1996) indicate many data gaps remain; that age-specific and size-specific dose-response profiles should be developed for each developmental stage; thresholds of sublethal and lethal effects should be known more precisely; and that research is needed on effects of particle quality, particle toxicity, and temperature.

Most of the studies discussed in this chapter involve several species of salmonid fish. The physiological effects of suspended sediment on warm water fish are far less studied but the assumption has long been that the muddy water in which these species are frequently found indicates that warm water fish are more resistant to turbidity and suspended sediments. The following sections discuss documented physiological effects of sediment exposure in salmonids.

6.2.1 Effects on Gills and Gas Exchange

Far more studies of the effects of suspended sediments on fish have addressed gill structure and function than any other physiological parameter, presumably because the filamentous structure of gill tissue is clearly capable of trapping fine sediment and because impaired gill function has sublethal and lethal consequences. Gill tissue also is easy to examine in living specimens, and it is relatively easy to detect gross morphological changes in gill tissue with basic histological techniques. Cordone and Kelley (1961) cited several such studies that dated from as early as 1937, about 50 years before any other type of physiological study of suspended sediment effects was published.

The three most frequently reported effects of suspended sediment on salmonid gills are as follows. First, fine suspended sediment "coats" or "clogs" gill filaments and thus impedes gas exchange; second, suspended sediment injures gill tissue directly through abrasion; and third, gill tissue responds morphologically to elevated suspended sediment by thickening or other hyperplasia. However, none of these effects has been reported consistently, apparently in part because of inconsistencies in sediment concentrations and exposure duration across the various natural and experimental systems studied, and in part because multiple fish species and age ranges were studied. As Newcombe and MacDonald (1991) and Waters (1995) indicate, these inconsistencies make direct comparison of the various studies and their results impossible. Nevertheless, each of these pathogenic effects has been reported often enough to suggest strongly that elevated suspended sediment may have all three of the effects discussed above on salmonid gills.

With the exception of Wallen (1951, cited by Cordone and Kelley 1961), early studies did not explicitly record sediment types, particle sizes, or concentrations; they were instead based primarily on studies of fish mortality in natural systems affected primarily by anthropogenic sediment influx. Thus, for the most part, threshold significant sediment levels cannot be derived from that work. Wallen (1951) experimented with various sediment concentrations on small, warm water fish in aquaria, and found that experimental subjects could withstand 100,000 ppm of montmorillonite clay for one week, but that they quickly succumbed when concentrations reached 175,000 to 200,000 ppm. He noted that all of the fish that succumbed had fine sediment coating the gill filaments and that water aeration in sublethal conditions helped the fish to "clear" their gills.

The most detailed available analysis of morphological effects (detected by histological examination) on gill tissue was apparently that of Noggle (1978), who exposed juvenile coho and Chinook salmon and steelhead experimentally to various concentrations of suspended sediment. Noggle (1978) provided micrographs of gill tissue recovered from six experimental animals and one control (zero exposure). The analysis was limited because just two specimens in each species-specific experimental group were used and because sediment concentrations varied widely among and within the groups (Table 6-5). Noggle's results also were somewhat inconsistent. For example, a 50 mm Chinook salmon died after 72.5 hours exposure to 4,266 mg/L suspended sediment and showed gross branchial necrosis, yet a 52 mm Chinook salmon exposed to the same sediment concentration and composition survived to experimental termination at 96 hours and its gill tissue was "normal" upon histological examination (Table 6-5).

However, the histopathological differences between the single control and most of the experimental group show clearly that suspended sediment is associated with significant pathological change in salmonid gill tissue. Impaired gill function (gas exchange), as such, is difficult to demonstrate *in vivo* or *post mortem*, but can be reasonably assumed given the pathologic changes in gill tissue. Elevated packed red blood cell volumes (hematocrit), which have been reported in several suspended sediment effects studies (e.g. Noggle 1978, Redding et al. 1987), may indicate gas exchange impairment (Table 6-6). Noggle's (1978) results also support a "mechanical gill injury" interpretation, that is, the changes documented are consistent with mechanical injury by abrasion, but the experiments were of insufficient duration to demonstrate sublethal tissue healing response (such as thickening/hyperplasia) to injury.

Sediment particle angularity and size appear to be factors in salmonid gill damage. Servizi and Martens (1987, cited by Bash et al. 2001) reported that gill trauma in Fraser River (British Columbia) underyearling sockeye salmon increases with sediment particle angularity and size, as well as concentration. They demonstrated gill trauma in these juvenile salmonids at angular sediment concentrations of 3,143 mg/L, levels that occur naturally in the Fraser River. Lake and Hinch (1999) found a significant difference in hematocrit and leukocrit (white blood cell volumes) values between control fish (unexposed to sediments) and fish exposed to angular suspended sediments at 1,000 to 40,000 mg/L. Fish exposed to 41,000 to 80,000 mg/L angular and smooth sediments (in separate experiments involving different fish) also differed significantly in leukocrit values from control fish, but hematocrit values at that concentration did not differ from control fish for either particle shape. However, the LC_{50} (the concentration at which 50 percent of test organisms die within a given time period) for either sediment particle shape was the same (164,500 mg/L after 96 hours of exposure), more than seven times the LC_{50} recorded by Servizi and Martens (1991). These inconsistencies are probably related to differences in experimental conditions (e.g. sediment sources may differ in levels of contaminants) and possibly to genetic differences in source fish populations.

Table 6-5. Effects of Experimental 96-Hour Sediment Exposure on Gill Tissue of Salmonids (From Noggle 1978)

Damage to gills	Species	Length (mm)	Suspended Sediment Concentration (mg/L)	Comments	Diagnosis
Little	Steelhead	134	12,936	Only survivor in tank	No visible lesions
Heavy	Steelhead	157	8,430	Alive	Branchial necrosis, branchial aneurysm, branchial hemorrhage
Fairly heavy	Chinook	50	4,266	Dead at 72.5 hours	Branchial necrosis
Little	Chinook	52	4,266	From 8% survivors	No visible lesions
Some	Coho	58	1,547	Dead at 45.5 hours	Diffuse branchial edema[1]
Little	Coho	63	5,346	Only survivor in tank	Focal lamellar fusion[2]
None	Coho	75	0	Control	No visible lesions

[1]Edema is one of the early stages of response to injury.

[2]Limited changes, representing only minimal damage.

Table 6-6. Blood Hematocrits as Percent Packed Red Cells in Yearling Steelhead Continuously Exposed to High (2-4 g/L) or Low (0.4-0.6 g/L) Concentrations of Suspended Topsoil, Kaolin Clay, or Volcanic Ash (Redding et al. 1987).

Exposure (h)	Topsoil				Clay			Ash		
	Control	Low	High	Acute	Control	Low	High	Control	Low	High
0	47±1 (17)				32±2 (12)			37±2 (17)		
3	48±2 (10)	47±2 (12)	47±1 (9)	47±1 (11)	44±2 (9)	45±2 (9)	44±2 (12)			
9	43±2 (8)	50±2* (8)	53±2* (9)	49±2 (6)	38±2 (11)	49±1* (12)	46±2* (11)	37±3 (6)	40±1 (9)	47±2 (11)
24	44±1 (12)	47±4 (5)	55±3* (7)	46±3 (11)	40±2 (6)	50±2* (10)	46±2* (11)	42±4 (6)	46±2 (11)	46±2 (7)
48	47±2 (11)	50±4 (7)	53±2 (9)		37±2 (6)	45±2* (11)	39±3 (9)	37±4 (5)	43±1 (11)	45±2 (11)

Note: Data are presented also for fish acutely exposed to 3 g/L suspended topsoil. Data are means ± SE; sample sizes are in parentheses. Asterisks denote significant differences from control values at $P < 0.05$. Data at hour 0 were pooled for all groups within an experiment.

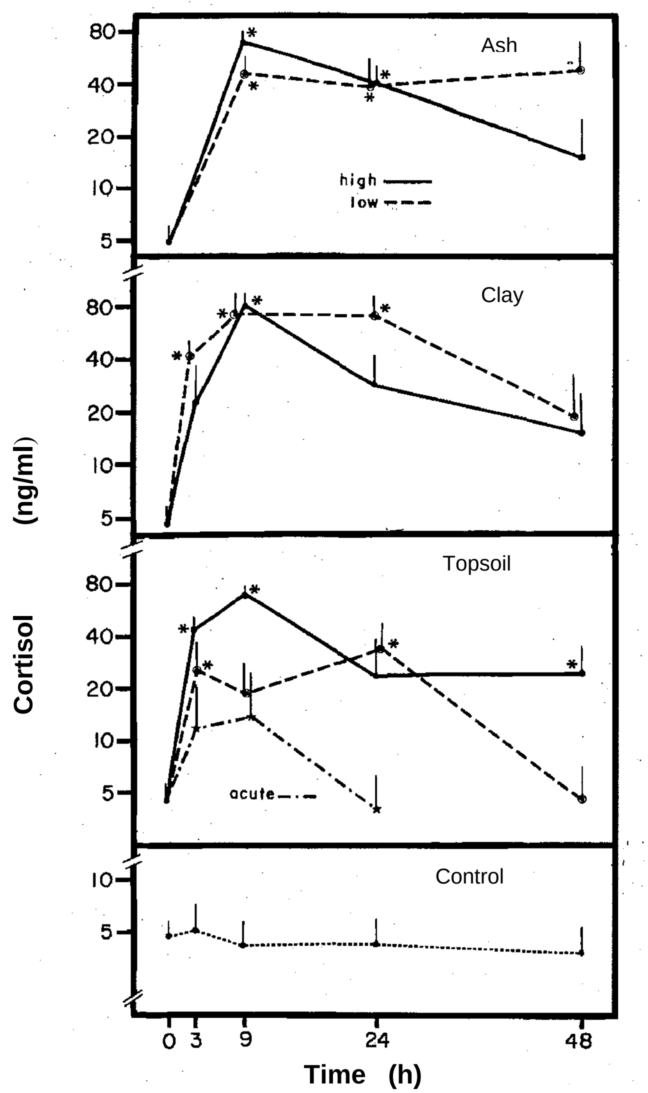
However, these studies indicate that sediment structure is clearly a factor in gill pathogenesis.

6.2.2 STRESS EFFECTS

Physiologic response to stress is manifested in most vertebrates by a rapid rise in serum cortisol. Cortisol is a glucocorticoid (steroid) hormone that is secreted by the adrenal cortical cells ("interrenal" cells in teleosts [most bony fishes]). Cortisol secretion is stimulated physiologically by adrenocorticotropic hormone (ACTH), one of several stimulating hormones secreted by the pituitary gland. ACTH release is in turn stimulated by corticotropin releasing hormone (CRH) secreted by the hypothalamus, which links the endocrine system to the central nervous system and thus bridges the sensory effects of stress to a hormonally mediated physical response (Willmer et al. 1999).

Among the numerous downstream effects of elevated cortisol is elevated blood glucose, which cortisol and other adrenocortical hormones stimulate by a variety of mechanisms. Hence, elevated blood glucose in company with sustained elevated serum cortisol suggests strongly that stress is operating as a metabolic factor at the time the fluid samples were obtained for analysis. Even at peak levels, all of these hormones circulate in extremely low concentrations (typically nanograms or picograms per milliliter of blood serum). They are assayed in serum by various competitive binding assay methods, including radioimmunoassay (RIA) with tritium- or radioiodine-labeled tracer, and enzyme-linked immunosorbent assay (ELISA), with peroxidase-conjugated tracer (Wheeler and Fraser 2004). Experimental systems that involve hormone assay are best evaluated "longitudinally" so that baseline samples can be obtained from the experimental animals prior to experimental manipulation and several times thereafter. When this is not possible the usual approach is to sample "cross-sectionally" from at least three animals in each group to yield a mean ± standard deviation for normal and post-manipulation serum hormone values (Wheeler and Fraser 2004).

Redding et al. (1987) showed, in a longitudinal study with separate controls, that "low" (300-600 mg/L) or "high" concentrations (2,000-3,000 mg/L) of topsoil-derived sediment stimulated serum cortisol in steelhead and coho salmon so that it peaked from 24 hours (coho, low concentration) to 48 hours (steelhead, low concentration) past time zero of a 192-hour exposure (Figure 6-23). Peak cortisol levels were 2 to 7 times the control levels for the same time point (low concentration, $p<0.05$ or $p<0.01$) and 10 to 14 times baseline levels at high sediment concentrations ($p<0.05$ for steelhead, but not significant for coho salmon, due to abnormal response from single individuals in the experimental series). However, cortisol levels dropped off rapidly after "peaking" to levels somewhat elevated above those in the control group. This observation held true for exposure to three types of sediment (topsoil, volcanic ash, and kaolin clay) (Figure 6-23). These authors concluded that exposure to suspended solids at the concentrations and duration used in their study was not severely stressful for yearling salmonids.



Note: Results from fish exposed acutely to 3 g/L suspended topsoil also are shown. Data are shown as antilogs of transformed ($\log_{10}X$) data, but the log scale is maintained. Asterisks indicate significant differences ($P < 0.05$) between treatment ($N = 11$-13) and pooled control groups ($N = 24$-29). Data from all groups were pooled for the hour-0 sample ($N = 60$).

Figure 6-23. Plasma Cortisol Concentrations (Mean + SE) in Yearling Steelhead during Continuous Exposures to High (2-4 g/L) or Low (0.4-0.6 g/L) Concentrations of Suspended Volcanic Ash, Kaolin Clay, or Topsoil (Redding et al. 1987).

In a cross-sectional study, Servizi and Martens (1992) demonstrated elevated blood glucose in acute samples obtained 96 hours after continuous exposure to 400 to 1,700 mg/L of natural Fraser River sediments. They used linear regression to predict the relationship between sediment concentration and blood glucose response in under yearling coho and adult sockeye salmon (Figure 6-24). Their results reflect a probable physiological delay in blood glucose increase after plasma cortisol spike, but cortisol was not assayed in the Servizi and Martens experiment. Other studies (e.g. Carruth et al. (2002) and Barton (2002)) indicate that cortisol is frequently elevated in oceangoing anadromous fish that are seeking their ancestral rivers, and these authors suggest that cortisol somehow mediates the homing olfactory phenomenon in these fish. Thus the fish used in the various experiments discussed above may have been predisposed to elevated cortisol and downstream blood glucose release. The physiological response to turbidity and suspended sediment exposure may be an environmental trigger rather than a hazard, at least in moderate concentrations. Further research may clarify the potential difference between adaptive and pathological response to suspended sediment in salmonid fish.

6.2.3 IMMUNE SYSTEM EFFECTS

The immune system includes various tissue, cellular, and molecular components, all of which interact to defend against invading bacteria, fungi, viruses, foreign proteins, and other antigens. Immune system compromise may follow various types of systemic "insult," possibly including exposure to suspended sediments. As part of their longitudinal study detailed in Section 6.2.2, Redding et al. (1987) exposed yearling steelhead to 2,500 mg/L suspended topsoil for two days, and then "challenged" them with injections of the pathogenic bacteria *Vibrio anguillarum* (Table 6-7). The quantities injected had previously been shown to cause the deaths of approximately 50 percent of control fish. Of the sediment-exposed group, 74 to 80 percent died of frank *Vibrio* infection, a statistically significant result ($p<0.05$). Although they apparently did not assay serum cortisol in the *Vibrio*-challenged group, Redding et al. (1987) speculated that elevated serum cortisol induced by suspended sediment exposure predisposed the fish to infection. Significantly, the sediment exposure itself caused about 25 percent mortality in control and experimental groups prior to inoculation with the bacteria challenge (Table 6-7). Redding et al. (1987) cite other published studies to support the hypothesis that exposure to suspended sediment or to exogenous cortisol can increase fish susceptibility to fin rot and increase mortality rate from "artificial" *Vibrio anguillarum* infection. Further research on this topic is indicated.

6.2.4 EFFECTS ON GROWTH AND DEVELOPMENT

Any of the physiological effects discussed in Sections 6.2.1 through 6.2.3 can in turn affect growth and development, but the various studies cited in those sections are deficient in that respect because none followed the experimental subjects beyond acute or chronic experimental manipulation. However, growth reduction as a consequence of suspended sediment exposure is well studied across a range of salmonid species and exposures (*sensu* Newcombe and MacDonald [1991]). The loss of visual capacity can

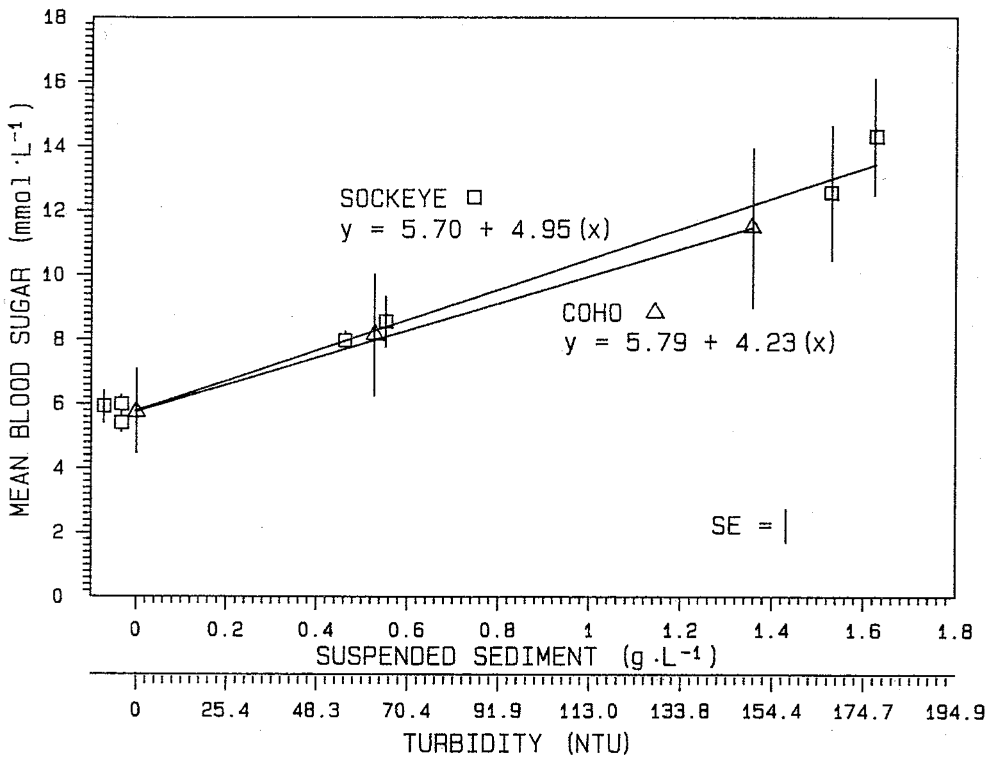


Figure 6-24. Responses of Blood Sugar to Suspended Sediments Exposure for Underyearling Coho (this Study) and Adult Sockeye (from Servizi and Martens 1987) (Servizi and Martens 1992).

Table 6-7. Mortality of Yearling Steelhead Exposed to 2.5 g/L Suspended Topsoil for Two Days, then Challenged with *Vibrio anguillarum* (Redding et al. 1987).

	Suspended solid exposure			Control		
	Vibrio-challenged fish		Unchallenged fish	*Vibrio*-challenged fish		Unchallenged fish
	(1)	(2)		(1)	(2)	
Total number of fish at start	25	25	25	25	25	25
Number dead before *Vibrio* challenge	6	5	6	2	1	0
Deaths caused by *Vibrio*	14	16	0	10	15	0
Percentage of challenged fish that died from *Vibrio*	74*	80*	0	43	63	0
Mean days to death	4.5	3.7	0	4.5	4.8	0

Note: The two columns for groups challenged with *Vibrio* represent results of replicate experiments (1) and (2).

* Asterisks denote significant differences from control value at $P < 0.05$.

lead to reduced feeding and depressed growth rate (Waters 1995), as discussed in Section 6.1.4. Of the sublethal effects of suspended sediment shown in 15 suspended sediment impact studies cited by Newcombe and MacDonald (1991), decreased growth or feeding rate was cited six times more often than any other factor. Growth rate reduction also was ranked just behind sediment-induced mortality in Newcombe and MacDonald's (1991) impact severity table (Table 6-8). In the absence of long-term studies of fish affected by episodic or chronic suspended sedimentation at significant concentrations, it is unclear whether the feeding and growth responses noted in the literature represent true impacts or short-term, potentially fully recoverable sequelae.

6.2.5 Temperature and Suspended Sediment

The SSCs at which lethal and sublethal effects occur may be affected by stress from other factors, such as temperature or disease. Servizi and Martens (1991) found that underyearling coho salmon (mean length range from 3.8 [swim-up fry] to 7.3 cm [fingerlings]) tolerance to suspended sediment differed at temperatures other than long-term rearing temperature, and was reduced among coho salmon with a viral kidney infection. Hatchery coho salmon were reared at 7°C and acclimated to test temperatures prior each experiment. Fish were tested at temperatures from 1 to 18°C and at five experimental SSCs (1 to 40 g/L) and a control (no sediment). The 96-h LC_{50} value for SSC was highest at 7°C, and lower at temperatures warmer and cooler (Figure 6-25). The 96-h LC_{50} value at 7°C rose sharply as fish length increased to 4.6 cm, after which SSC tolerance was independent of fish size (Figure 6-26). The authors suggest that small fish may be less able to clear sediments via the cough reflex. It is unclear whether the 7°C rearing temperature represented an optimum temperature for tolerance to SSC or whether the length of time rearing at this temperature may have affected the outcome of the experiments at warmer and cooler temperatures.

The response to SSC may differ between species. Servizi and Martens (1991) compared LC_{50} values found for coho salmon with values for other juvenile salmon under similar test conditions (at temperatures of 7 to 8.3°C). They found that within this narrow temperature range, Chinook salmon were most tolerant (96-h LC_{50} value of 31 g/L), followed by coho (22.7 g/L) and sockeye salmon (17.6 g/L) (Servizi and Martens 1987, Servizi and Gordon 1990, cited in Servizi and Martens 1991).

Table 6-8. Summary of Data (*in situ* Observations) on Exposures to Suspended Sediment that Resulted in Lethal Responses in Salmonid Fishes (Newcombe and MacDonald 1991).

Species[a]	Exposure C	Exposure D	Stress index ($\log_e$ [C × D])	Effect	Rank of effect	Source
				Arctic grayling		
Arctic grayling	25	24	6.397	6% mortality of sac fry	10	Reynolds et al. (1988)
	23	48	7.007	14% mortality of sac fry	10	Reynolds et al. (1988)
	65	24	7.352	15% mortality of sac fry	10	Reynolds et al. (1988)
	22	72	7.368	15% mortality of sac fry	10	Reynolds et al. (1988)
	20	96	7.560	13% mortality of sac fry	10	Reynolds et al. (1988)
	143	48	8.834	26% mortality of sac fry	11	Reynolds et al. (1988)
	185	72	9.497	41% mortality of sac fry	12	Reynolds et al. (1988)
	230	96	10.002	47% mortality of sac fry	12	Reynolds et al. (1988)
	20,000	96	14.468	10% mortality of age-0 fish	10	McLeay et al. (1987)
	100,000	96	16.077	20% mortality of age-0 fish	10	McLeay et al. (1987)
				Salmons		
Chinook salmon	488	96	10.755	50% mortality of smolts (high T°C)	12	Stober et al. (1981)
Coho salmon	509	96	10.797	50% mortality of smolts (high T°C)	12	Stober et al. (1981)
Chinook and sockeye salmon	1,400[b]	36	10.827	10% mortality of juveniles	10	Newcomb and Flagg (1983)
Coho salmon	1,200	96	11.654	50% mortality of juveniles	12	Noggle (1978)
	1,217	96	11.668	50% mortality of pre-smolts (high T°C)	12	Stober et al. (1981)
Chinook and sockeye salmon	207,000[b]	1	12.240	100% mortality of juveniles	14	Newcomb and Flagg (1983)
	9,400	36	12.732	50% mortality of juveniles	12	Newcomb and Flagg (1983)
Chum salmon	97	3,912[b]	12.847	77% mortality of eggs and alevins	13	Langer (1980)
	111	3,912[b]	12.981	90% mortality of eggs and alevins	14	Langer (1980)
Chinook and sockeye salmon	82,000	6	13.106	60% mortality of juveniles	12	Newcomb and Flagg (1983)
Coho salmon	18,672	96	14.400	50% mortality of presmolts	12	Stober et al. (1981)
Chinook salmon	19,364	96	14.436	50% mortality of smolts	12	Stober et al. (1981)
Chum salmon	28,000	96	14.804	50% mortality of juveniles	12	Smith (1939)
Coho salmon	28,184	96	14.811	50% mortality of smolts	12	Stober et al. (1981)
	29,580	96	14.859	50% mortality of smolts	12	Stober et al. (1981)
	35,000[b]	96	15.027	50% mortality of juveniles	12	Noggle (1978)
Chinook and sockeye salmon	39,400	36	15.145	90% mortality of juveniles	14	Newcomb and Flagg (1983)
Chum salmon	55,000	96	15.479	50% mortality of juveniles	12	Smith (1939)
				Whitefish		
Whitefish	16,613	96[b]	14.282	50% mortality of juveniles	12	Lawrence and Scherer (1974)
				Trouts		
Rainbow trout	200[c]	24	8.476	5% mortality of fry	10	Herbert and Richards (1963)
	7	1,152	8.995	17% reduction in egg-to-fry survival	10	Slaney et al. (1977b)
	21	1,152	10.094	62% reduction in egg-to-fry survival	13	Slaney et al. (1977b)
	200[c]	168	10.422	8% mortality of fry	10	Herbert and Richards (1963)
	90	456	10.622	5% mortality of subadults	10	Herbert and Merkens (1961)

Note: Within species groups, stress indices are arranged in increasing order. For exposure, C = concentration (mg/L) and D = duration (h).

Table 6-8. Summary of Data (*in situ* Observations) on Exposures to Suspended Sediment that Resulted in Lethal Responses in Salmonid Fishes (Newcombe and MacDonald 1991) (continued).

Species[a]	Exposure C	Exposure D	Stress index ($\log_e \cdot [C \times D]$)	Effect	Rank of effect	Source
	68	720[b]	10.799	25% reduction in population size	11	Peters (1967)
	37	1,440	10.883	46% reduction in egg-to-fry survival	12	Slaney et al. (1977b)
	47	1,152	10.889	100% mortality of incubating eggs	14	Slaney et al. (1977b)
	57	1,440	11.315	23% reduction in egg-to-fry survival	11	Slaney et al. (1977b)
	270[d]	456	11.721	10–35% mortality of sub-adults	11	Herbert and Merkens (1961)
	270[e]	456	11.721	80% mortality of sub-adults	13	Herbert and Merkens (1961)
	101	1,440	11.888	98% mortality of eggs (high metals and NH_3 levels)	14	Turnpenny and Williams (1980)
Brown trout	110	1,440	11.973	98% mortality of eggs	14	Scullion and Edwards (1980)
Rainbow and brown trout	300	720[b]	12.283	97% reduction in population size	14	Peters (1967)
Rainbow trout	1,000–2,500	144	12.437	100% mortality of eggs	14	Campbell (1954)
	157	1,728	12.511	100% mortality of eggs	14	Shaw and Maga (1943)
	810[d]	456	12.820	5–80% mortality of sub-adults	13	Herbert and Merkens (1961)
	810[e]	456	12.820	80–85% mortality of sub-adults	14	Herbert and Merkens (1961)
	200[c]	2,352	13.061	50% mortality of fry	12	Herbert and Richards (1963)
	1,000–2,500	480	13.641	57% mortality of fingerlings	12	Campbell (1954)
	4,250	588	14.731	50% mortality (life stage not specified)	12	Herbert and Wakeford (1962)
	160,000	24	15.161	100% mortality (life stage not specified)	14	D. W. Herbert, personal communication in Alabaster and Lloyd (1982)
	49,000	96	15.363	50% mortality of juveniles	12	Lawrence and Scherer (1974)
	1,000–6,000	1,440[b]	15.432	85% reduction in population size	14	Herbert and Merkens (1961)
Brown trout	1,040	8,670	16.024	85% reduction in population size	14	Herbert et al. (1961)
	5,838	8,670	17.750	85% reduction in population size	14	Herbert et al. (1961)

[a] Scientific names: Arctic grayling, *Thymallus arcticus*; chinook salmon, *Oncorhynchus tshawytscha*; coho salmon, *O. kisutch*; sockeye salmon, *O. nerka*; chum salmon, *O. keta*; whitefish, *Coregonus* sp.; rainbow trout, *Oncorhynchus mykiss*; brown trout, *Salmo trutta*.

[b] Estimated.

[c] Wood fiber.

[d] Kaolin.

[e] Diatomaceous earth.

Note: Within species groups, stress indices are arranged in increasing order. For exposure, C = concentration (mg/L) and D = duration (h).

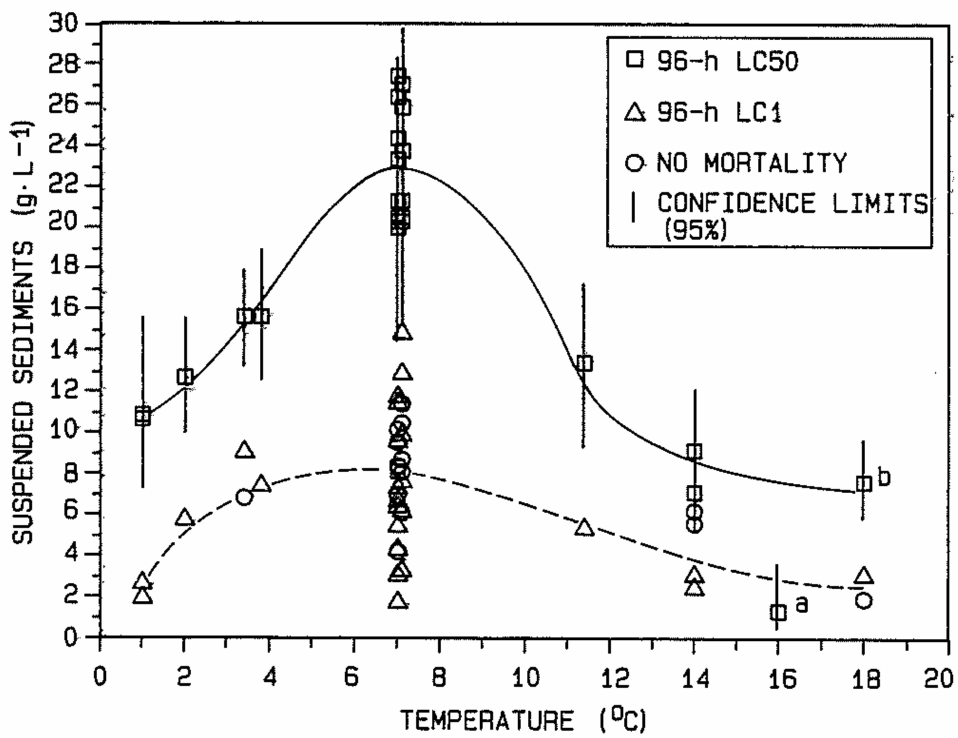


Figure 6-25. Tolerance of Underyearling Coho to Suspended Sediments at Temperatures between 1 and 18°C. a, fish were stressed; b, fish were partially confined (Servizi and Martens 1991).

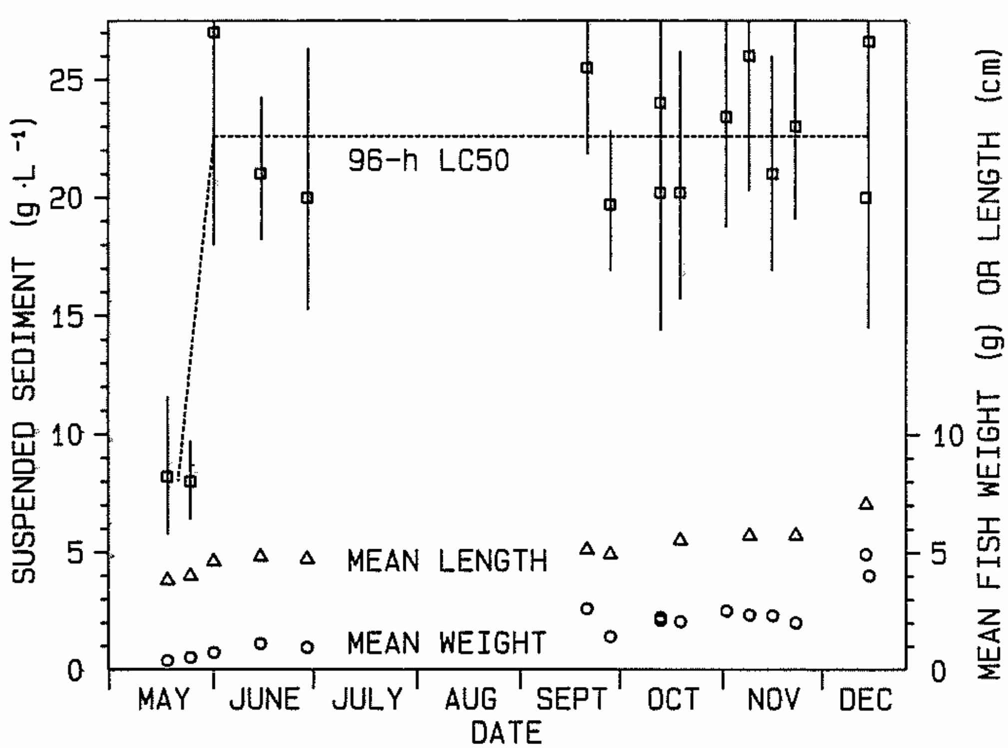


Figure 6-26. Mean Weights and Lengths of Underyearling Coho and Acute Lethalities of Suspended Sediments (96-h LC50) at 7°C. Vertical lines indicate 95% confidence limits (Servizi and Martens 1991).

Information is not available on the effects of turbidity and suspended sediment concentrations on any life history stage of amphibians. The literature reviewed in this section address potential effects of sedimentation on amphibian life history stages.

The only California amphibian species that breed in wide shallow rivers such as the Feather River are the foothill yellow-legged frog (*Rana boylii*), the Pacific chorus frog (*Pseudacris (=Hyla) regilla*), and the western newt (*Taricha torosa*) (Stebbins 1951). However, other species may be found in streams at higher elevations, such as the mountain yellow-legged frog (*Rana muscosa*). The potential harmful effects of turbidity and suspended sediments on eggs, larvae, and adults of these species are unknown.

7.1 POTENTIAL FOR EFFECTS

Egg masses of all of these species are potentially vulnerable to smothering by natural or anthropogenic sedimentation during springtime high water. River populations of the western newt and Pacific chorus frog avoid these hazards by spawning in backwater pools and very shallow parts of mainstem rivers (Storer 1925, Stebbins 1951). Larvae of these species are adapted to lentic water, and adults and larvae of both species usually avoid lotic water. However, the foothill yellow-legged frog characteristically inhabits the margins of moderately fast water along cobble riffles, particularly downstream of confluences (Zweifel 1955, Kupferberg 1996). It avoids high water impacts on its egg masses, presumably including elevated bedload and suspended sediment levels, by delaying spawning until after the springtime flows have receded to near minimum levels that flow gently just a few inches over cobble substrata (Zweifel 1955, Kupferberg 1996, Lind et al. 1996). Newly hatched tadpoles often linger in crevices and other secure shelters near the hatching sites until they are large enough to swim in stronger currents (S. Barry, pers. obs.). This behavior protects the tadpoles from being swept downstream, but it renders them vulnerable to any physical perturbations that affect such lentic microhabitat.

Absent from the literature are data on the potential effects of turbidity and precipitated sediments on foothill yellow-legged frog egg masses. Moderate turbidity and sediment transport are normal characteristics of late spring stream water in the Sierra Nevada, and foothill yellow-legged frog egg masses are usually covered with fine silt soon after deposition. Storer (1925) first made this observation on un-dammed, un-mined coastal rivers during the 1920's, so the silt covering is clearly a natural phenomenon. Storer (1925) speculated that the thin silt coating protects the eggs from excessive solar radiation and camouflages them from predators, and various authors have suggested that any such silt coating of amphibian eggs may help to protect the eggs from ultraviolet radiation exposure. However, although light silt coatings apparently do no harm and may be beneficial, a threshold limit of sediment covering above which foothill yellow-legged frog eggs cannot survive undoubtedly exists. These threshold limits are apparently unknown for any amphibian species, but a massive, sudden sediment dump sufficient to fill cobble interstices in favored foothill yellow-legged frog riffle spawning habitat will obviously render that habitat at least temporarily unsuitable for spawning. Such an event would also smother and kill any eggs or newly hatched tadpoles already present.

7.2 SEDIMENT DEPOSITION

Recent research on the effects of inorganic non-toxic sediments on stream or river-dwelling amphibians has typically examined opportunistically the effects of unique sedimentation events on stream and spring dwelling amphibian species. For example, studies following accidental sedimentation from road construction (Welsh and Ollivier 1998) and from logging (Corn and Bury 1989) in northwestern California and west-central Oregon found that sediment dumps in headwaters and other small streams exert profound deleterious effects on amphibian reproduction and survival. Individual species densities and distributions within some mesohabitats along these massively impacted streams and rivulets changed in ways that indicated mass population movement to suboptimal habitat, population declines, or extirpation in response to the sediment dumps.

A study of the interaction of logging-related sediment incursions and trout predation on larval and adult dusky salamanders (*Desmognathus porphyriticus*) in headwater streams in the Appalachian Mountains (Lowe et al. 2004) revealed differential vulnerability to each perturbation. Abundance of larvae was negatively related to brook trout abundance and unrelated to substrate embeddedness, but abundance of adults was primarily related to substrate embeddedness. Growth and survival of larvae also were negatively affected by brook trout. The authors suggest that in streams where brook trout are present and larval abundance is low, an unnatural increase in sedimentation could reduce adult abundance, which could jeopardize entire stream populations of these amphibians. Although the circumstances that provided the backdrop for these studies were unusual, the study results indicate that sediment transport and deposition are indeed potential hazards for amphibians and probably for exposed amphibian eggs in any lotic habitat. The trout-sediment-salamander interaction study also suggests that foothill yellow-legged frog populations (which often occur with trout) might be rendered more vulnerable to extirpation by trout predation pressure subsequent to sediment releases.

8.1 SUMMARY AND RECOMMENDATIONS

This section summarizes the material discussed in the previous chapters, in the context of its relevance and application to Sierra Nevada stream fish and their habitats. Our objectives are:

- to identify the most appropriate operational turbidity and suspended sediment measurement and evaluation methods;
- to identify the components and parameters of turbidity and suspended sediment that are most likely to adversely affect aquatic biota;
- to evaluate the utility of current sediment and turbidity models in the context of pulse-flow releases;
- to identify and discuss significant scientific disagreement on any of the topics covered in this white paper; and
- to offer general recommendations for future appropriate studies and study methods.

8.2 TURBIDITY AND SUSPENDED SEDIMENT MEASUREMENT EVALUATION

Two issues related to the appropriate metric for evaluation of turbidity and suspended sediment emerged from our review of methodologies and analyses from the scientific literature. These are:

- the appropriateness of measuring turbidity (nephelometry) versus water clarity as an indicator of light penetration and optical effects on aquatic biota; and
- the appropriateness of using turbidity and suspended sediment measurements as surrogates for each other when examining both potential optical effects and effects due to mass sediment concentration (e.g. physiological effects) on aquatic biota.

The principal advantages of using nephelometry to evaluate suspended sediment (as turbidity) are that nephelometry instrumentation is widely available and inexpensive and that NTUs have been used in a wide body of literature. It also is clear that nephelometry instrumentation has had decades of use. Continuity of methods and units is usually regarded as beneficial. Additionally, current California water quality regulations specify required turbidity limits in NTUs, so nephelometry will continue to be the only acceptable compliance method until the regulatory standards units are changed or broadened to include water clarity data or other measures. However, nephelometry as a research, management, or monitoring tool has several important drawbacks (Davies-Colley and Smith 2001, Ziegler 2002) as discussed below.

- The variability of results related to inter-sample variation in particle composition, size, and angularity (the nephelometer may deliver the same reading for two samples with widely different suspended sediment particle classes and concentrations).
- Stream and seasonal variation in the relationship between turbidity and suspended sediments.
- The inconsistency of same-sample readings among various meter brands and models (design and optics differences may cause different instrument brands to report different NTU values or slopes for the same water sample series).
- NTUs are an arbitrary scale, which cannot be related to any real physical quantity.

Further, because of the arbitrary scale of NTUs, nephelometers can only be calibrated by arbitrary standards such as formazin, not by any independent method of measuring turbidity, water clarity, or suspended sediment. Thus, NTU data are potentially misleading and likely to lack consistency among studies, seasonally, or longitudinally within a single management program (Davies-Colley and Smith 2001). Consequently, a minimum requirement for any work with nephelometry is that the brand, model, and serial number, light source, wave-length and defector geometry of the nephelometer always be noted with the data (Davies-Colley and Smith 2001, Ziegler 2002).

Davies-Colley and Smith (2001) argue that "water clarity," the inverse of turbidity, is a far more useful and meaningful measure of the optical component of suspended sediment than is nephelometry. For water clarity ("visual acuity") measurements, the horizontal black disk viewer offers a good (if somewhat cumbersome) alternative in that it reports reproducible scientific units (in meters) with acceptable precision, and is seemingly easy to learn to use (Davies-Colley and Smith 2001). Further, beam attenuation (m^{-1}) as determined in individual water samples by a beam transmissometer is the reciprocal of the horizontal black disk measurement and could potentially serve as a useful laboratory quality control to confirm the accuracy of in situ black disk readings, or as an alternative field measurement. This is a fairly standard field method for measurement of light attenuation in marine studies. Although the nephelometer (standardized to a single brand and model for baseline and subsequent recording) may continue to be the instrument of choice for regulatory compliance in California, the horizontal black-disc viewer and beam transmissometer may be better alternatives for research and perhaps for management decision-making and monitoring.

The primary difficulty of using turbidity or water clarity measurements as correlation surrogates for suspended sediment concentrations is the differential contribution of various types of suspended sediment particles to light attenuation. Waterways with seasonally or temporally variable sediment particle size and composition profiles will be more difficult in this regard than waterways with a single source and type of sediment (Davies-Colley and Smith 2001). The only way to construct a reliable correlation is to gather appropriate water clarity data across the time periods and flows of interest, accompanied by accurate suspended sediment determinations sampled during the same times. The recommendations of recent literature suggest that where suspended sediment concentration effects are the focus of study, it is advisable to measure them directly rather than by using surrogates. Where water clarity is of concern, then it should be measured directly by an objective technique such as a black disc rather than by a surrogate measure such as SSC or an arbitrary scale such as turbidity.

8.2.1 Summary of Effects on Aquatic Biota Associated With Turbidity and Suspended Sediment

This section summarizes documented effects of turbidity and suspended sediments on benthic macroinvertebrates, fish, and aquatic amphibians. These effects are poorly documented for stream-dwelling benthic macroinvertebrates, fairly well documented for trout and salmonids (trout and salmon), and undocumented for amphibians and non-salmonid native California stream fish. All three groups are worthy of more intensive field study.

8.2.2 Invertebrates

We reviewed reports of the effects of turbidity and suspended sediment on benthic macroinvertebrate physiology, behavior (including drift and net construction), trophic structure, and community response. Our review indicated that the effects of turbidity and suspended sediment on benthic macroinvertebrates are poorly understood and thus largely speculative. The effects of deposited sediment that can bury invertebrate colonies and alter benthic habitat may, in fact, be more significant.

Seven studies included enough data to suggest conclusions. The most speculative were the effects on benthic macroinvertebrate physiology, although the data suggested that suspended sediment could affect respiration by adhering to gills and could possibly affect the dermal cuticle when coarse sediment scours benthic habitat (Lemly 1982). The most pronounced experimental effects of suspended sediment were observed on community size, which decreased in rough proportion to duration of exposure, but not to increasing concentration. Another experiment tested a suspended sediment concentration increase from 0.0 to only ~7.5 mg/L with a five-hour exposure duration, which showed a significant increase in macroinvertebrate drift (Rosenberg and Wiens 1978). A second duration test study (Shaw and Richardson 2001) of increasing exposure duration (0-6 hours daily for 19 days) to about 700 mg/L suspended sediment also yielded a similar pattern of increased drift, along with a decreased diversity of the remaining benthic component. The results of these two studies suggest that threshold concentrations of suspended sediment sufficient to affect macroinvertebrate community size are quite low, but that significantly elevated concentrations may have little additional effect. Finally, Van Niewenhuyse and LaPerriere (1986) showed that very high turbidity (to 3,400 NTU) virtually eliminated primary productivity in benthic ecosystems, although these results were confounded by the presence of heavy metal contamination.

These studies indicate that relatively low concentrations of suspended sediment may dramatically affect benthic community structure and ecology. It remains difficult to separate the effects of suspended sediment from those of deposited sediment. Future work should strive to eliminate deposited sediment from experimental systems and to examine more closely the effects of exposure duration, particle size and composition, and a wider spectrum of suspended sediment concentrations.

8.2.3 FISH (SALMONIDS)

Studies of the effects of suspended sediment and turbidity on salmonids dominate the literature reviewed. Experimental studies have been devoted primarily to the effects on salmonid behavior, particularly avoidance, the effects on foraging, and risk of predation. Collectively these studies indicate that salmonids tend to avoid turbid water above about 25 NTUs, especially when they are exposed to it suddenly rather than gradually. Yet, salmonids also will hide in moderately turbid water to avoid predation.

A potential research bias is that juvenile fish, the most common study subjects, are inherently more vulnerable to predation than are adult fish and turbidity may offer an important source of cover. In addition, juvenile salmonids also may be less vulnerable to suspended sediment in moderate concentrations as an adaptation that allows them to find shelter within such water. Noggle's (1978) study of salmonid morbidity and mortality at moderate to high suspended sediment concentrations suggests that some juvenile fish are far more resistant to the physiological effects of turbidity than larger fish, although only a few survived exposure to >1000 mg/L suspended sediment for 96 hours. This observation may explain Newcombe and Jensen's (1996) modeled finding that juvenile salmonids overall are less vulnerable to suspended sediment than adult salmonids. Even moderate gill damage, a very commonly cited consequence of exposure to suspended sediment, was not consistently associated with mortality or correlated with sediment concentration within Noggle's work. In that work, a 134 mm steelhead survived 96 hours of almost 13,000 mg/L suspended sediment with "little" gill pathology, while a 58 mm coho salmon that died after 45.5 hours exposure to only about 1,500 mg/L showed "some" gill damage.

Most of the physiologic parameters discussed in this review were studied in only one or a few of the papers, but together they suggest that turbidity and suspended sediment exert moderate physiologic effects, primarily through the hypothalamic-pituitary-adrenal ("stress") axis. The documented effects of exposure to elevated turbidity and suspended sediment on growth and development also suggest a pituitary-thyroid axis involvement. This and other pathways collectively suggest a cascade of pituitary hormones in response to elevated turbidity/suspended sediment. However, the various studies indicate that although salmonids may prefer clear water from a strictly physiological viewpoint, they are capable of surviving moderate turbidity or suspended sediment for extended time periods. In fact, some individuals can withstand extremely high sediment concentrations with seemingly little adverse physiologic effect. This conclusion is not surprising since turbid water is a frequent and widespread feature of the rivers and streams that these fish inhabit or use for reproductive activity. An objective of assessing effects on fish will continue to focus on defining the threshold limits of exposure-both in time and concentration. The exposure models discussed later in this chapter offer insight toward that goal.

8.2.4 AMPHIBIANS

We included evaluation of turbidity and suspended sediment effects on amphibians in this review, primarily because the ecologically-unique foothill yellow-legged frog (*Rana boylii*) inhabits and spawns within main channel runs and riffles along many Sierra Nevada streams and rivers, including river sections downstream of several dams and diversions. No other native frog species utilizes main channel riffle habitat for foraging or for reproduction. Unfortunately, our review yielded little information on the effects of lotic turbidity or suspended sediments on any amphibian species, although Storer's (1925) observations suggest that eggs of the foothill yellow-legged frog are adapted to moderate sediment deposition with an unknown "damage" threshold. Lind et al. (1996) offer useful information about the effects of dam-mediated water level manipulation on this frog species, but sediment effects are not included in their analysis. Clearly, additional information from both field and laboratory study is needed to assist in evaluating potential effects of sediment and turbidity.

8.3 THE UTILITY OF AVAILABLE EXPOSURE EFFECTS MODELS

Newcombe and Jensen (1996) summarized and synthesized the highly diverse literature on suspended sediment effects on fish into a manageable series of six tabular effects-model interpretations. Each of the six applies to a different fish taxon or life history stage and in some cases to a different series of sedimentation particle sizes. Important features of the Newcombe and Jensen (1996) model are that it addresses the dose-response effect of fish to the interaction of suspended sediment concentration and duration of exposure. The model accounts for the information derived from the literature that different fish taxa and life stages react differently to suspended sediment concentration and duration of exposure. The model is structured into a tiered series of effect categories resulting from the interaction of concentration and duration of exposure with the following major tiers:

- "Nil" (no discernable) effects;
- Behavioral effects (presumably not damaging);
- Sublethal effects, but damaging; and
- Lethal and paralethal effects.

Newcombe (2003) developed a similar "duration of exposure" model for the effects of turbidity (water clarity) on fishes, based primarily on peer discussion rather than the type of meta-analysis used by Newcombe and Jensen (1996). The focus of this model is on effects related to water clarity. This model also incorporates "fish reactive distance" (calibrated for trout only), correlated roughly to turbidity and by inference to water clarity, as an alternative to biologically based calibration.

The subsequent scientific literature does not suggest that either model has been applied widely as a research tool. However, the Newcombe and Jensen (1996) model has been applied in a number of "real world" applications. It has been recommended for use in helping to set total maximum daily loads (TMDLs) of sediment in a number of states including California, and in assessing management actions (e.g., Central Coast RWQCB 2004, IDEQ 2003, USEPA 2004). For example, TMDL numeric sediment targets for the Pajaro River Watershed (located in Monterey, San Benito, Santa Cruz and Santa Clara counties) were based on the concentration/duration model of Newcombe and Jensen (1996) (Central Coast California RWQCB 2004). Modeled TMDL targets were based on Newcombe and Jensen's SEV 8 exposure ranges (indications of major physiological stress). Targets were specified for specific subwatershed areas to account for variation in sediment-loading characteristics.

The Newcombe and Jensen (1996) model also was used to assess potential effects of the USEPA Superfund Program clean-up plan for the Milltown Reservoir/Clark Fork River Superfund site in Montana (USEPA 2004, USEPA and FERC 2004). The U.S. Fish and Wildlife Service used the Newcombe and Jensen (1996) model in their biological opinion for this USEPA Superfund clean-up plan to evaluate project effects on ESA-protected bulltrout. The Idaho Department of Environmental Quality (IDEQ) based their TMDL guidelines on the model of Newcombe and Jensen (1996) for suspended sediment targets in the Snake River (IDEQ 2003). The model has also been referenced in other reviews and in the gray literature (Reid and Anderson no date, Clark and Wilbur 2000).

The Newcombe (2003) model is as amenable as the earlier model to use for simple assessment of the effects of turbidity as a duration and measurement dose-response (particularly as measured by objective measures such as water clarity measures such as horizontal black disc visibility). Notably, Newcombe (2003) strongly criticizes the appropriateness of nephelometric turbidity measurements in turbidity studies, paralleling similar criticism by Davies-Colley and Smith (2001). His model states that water clarity measures including horizontal black disk and beam attenuation measurements are "preferred," and that nephelometric units should be considered as an "alternate" method to measure turbidity.

8.3.1 MODEL CAVEATS

The Newcombe and Jensen (1996) and Newcombe (2003) models can be used as tools to assess the effects of suspended sediment or turbidity on fish in streams. These models can also assist in identifying potential suspended sediment or turbidity objectives for stream management. However, it is important to appreciate the limitations of each. The most important limitation of both models is that neither has been specifically validated for any locality, let alone for the broad geographic region and range of hydrologic conditions. Newcombe and Jensen (1996) stated that validation would [only] come from further studies, and is "…bound to be a slow process." Given the diversity of published reports that form the foundation of the Newcombe and Jensen (1996) model and the potentially non-objective nature of the input to the Newcombe (2003) model, validation (i.e., calibration) for any locality where either is to be applied is an important consideration. The Newcombe and Jensen (1996) model was constructed from data collected from 80 published papers, representing at least 50 fish species and a large number of different sets of experimental and observational conditions. This is not necessarily a drawback, but it does suggest that further corroboration for the values of the individual cells within the model matrix would be desirable for species of interest. It is recognized that application of these models for use with salmonids is based on a broader literature base than for other taxa.

Other factors not addressed in either models include: overall particle composition, water temperature, water velocity, physical characters of the streams, rate of increase of sediment concentration over background level, and frequency (not duration) of acute exposure. Variability among these factors between the study conditions that contributed to the models and later experimental conditions applied against the models are likely to affect the reliability of the results. For those reasons and because of residual uncertainty of these models to specific locations and species, they should be considered to offer guidelines, not rules, for turbidity and suspended sediment assessment and management.

Despite their shortcomings, the Newcombe and Jensen (1996) and Newcombe (2003) suspended sediment and turbidity effects models remain the best available interpretations of sediment dose-response effects on fish. Local evaluation or calibration of these models is an important aspect to increasing their reliability and the certainty of their results.

8.4 GENERAL RECOMMENDATIONS

Based on review of the relevant literature, we suggest the following recommendations in the design of future field studies where the effects of water clarity and suspended sediment on aquatic biota are a main concern.

- The Newcombe (2003) and Newcombe and Jensen (1996) models can be used to provide a reasonable framework for assessing potential effects on fish, particularly salmonids.
- To reduce uncertainty, inputs to the Newcombe and Jensen (1996) model preferably should be determined by direct measurement of suspended sediment.
- Input to the Newcombe (2003) model preferably should be determined by water clarity measurement. This model should be used when water clarity effects on fish are the main concern.
- If a turbidity measure (nephelometry) is used, there should be field calibration of turbidity with water clarity and/or suspended sediment.

- Prior to the routine use of the Newcombe and Jensen (1996) or Newcombe (2003) models as management tools, additional corroboration of duration and concentration effect levels should be made for selected species of concern.
- Studies should be conducted on lifestages of non-salmonid native fish and amphibians, which may be of particular concern to determine SEV on these species and their habitat.

Alabaster, J.S., and R. Lloyd. 1982. Finely divided solids. *In:* J.S. Alabaster and R. Lloyd (editors), Water Quality Criteria for Freshwater Fish (2nd ed). Butterworth, London, England.

Alexander, G.R., and E.A. Hansen. 1983. Effects of sand bedload sediment on a brook trout population. Michigan Department of Natural Resources Fisheries Research Report 1906.

American Public Health Association (APHA), American Water Works Association, and Water Pollution Control Federation. 1998. Standard Methods for the Examination of Water and Wastewater (20th ed.). Washington D.C. 1,162 pp.

Anderson, C.W. 2004. Chapter 6, Section 6.7 – Turbidity. *In:* USGS National Field Manual USGS TWRI Book 9-A6 - Turbidity (version 2.0): 64 pp.

Anderson, P.G. 1996. Sediment generation from forestry operations and associated effects on aquatic ecosystems. Proceedings of the Forest-Fish Conference: Land Management Practices Affecting Aquatic Ecosystems, May1-4: 22 pp.

ASTM. 1996. Water and environmental technology. *In:* 1996 Annual Book of ASTM Standards, Section 11. R.A. Storer (editor). American Society for Testing and Materials. West Conshocken, Pennsylvania.

Barton, B.A. 2002. Stress in fishes: a diversity of responses with particular references to changes in circulating corticosteroids. Integrative and Comparative Biology 42: 517-525.

Bash, J., C. Berman, and S. Bolton. 2001. Effects of turbidity and suspended solids on salmonids. Center for Streamside Studies University of Washington, Seattle, Washington. 1-74. http://depts.washington.edu/cwws/Outreach/Publications/Salmon%20and%20Turbidity. pdf

Berg, L. and T.G. Northcote. 1985. Changes in territorial, gill-flaring, and feeding behavior in juvenile coho salmon (*Oncorhynchus kisutch*) following short-term pulses of suspended sediment. Canadian Journal of Fisheries and Aquatic Science 42: 1410-1417.

Beschta, R.L. 1980. Modifying automated pumping samplers for use in small mountain streams. Water Resources Bulletin 16: 137-138.

Beschta, R.L. 1981. Increased bag size improves Helley-Smith bedload sampler for use in streams with high sand and organic matter transport. *In:* Erosion and Sediment Transport Measurement Symposium Proceedings. International Association of Hydrologic Sciences, Publication 133, Washington, D.C.

Beschta, R.L. 1996. Suspended Sediment and Bedload. *In:* Methods in Stream Ecology, F.R. Hauer and G.A. Lamberti (editors) Academic Press, San Diego, California 123-143.

Beschta, R.L., J.R. Boyle, C.C. Chambers, W.P. Gibson, S.V. Gregory, J. Grizzel, J.C. Hagar, J.L. Li, W.C. McComb, M.L. Reiter, G.H. Taylor, and J.E. Warila. 1995. Cumulative effects of forest practices in Oregon. Oregon State University, Corvallis. Prepared for the Oregon Department of Forestry, Salem, Oregon.

Bisson, P.A. and R.E. Bilby. 1982. Avoidance of suspended sediment by juvenile coho salmon. North American Journal of Fisheries Management 4: 371-374.

Bjornn, T.C., and D.W. Reiser. 1991. Habitat requirements of salmonids in streams. *In:* W.R. Meehan, editor. Influences of Forest and Rangeland Management on Salmonid Fishes and their Habitats. American Fisheries Society, Bethesda, Maryland.

Bjornn, T.C., and seven coauthors. 1974. Sediment in streams and its effects on aquatic life. Water Resources Research Institute, Research Technical Completion Report Project B-025-IDA, Moscow, Idaho.

Bjornn, T.C., M.A. Brusven, M.P. Molnau, J.H. Milligan, R.A. Klamt, E. Chacho, and C. Schaye. 1977. Transport of granitic sediment in streams and its effects on insects and fish. University of Idaho, College of Forestry, Wildlife and Range Sciences, Bulletin 17, Moscow, Idaho.

Bonner, T.H. and G.R. Wilde. 2002. Effects of turbidity on prey consumption by prairie stream fishes. Transaction of the American Fisheries Society 131: 1203-1208.

Brusven, M.A. and K.V. Prather. 1974. Influence of stream sediments on distribution of macrobenthos. Journal of the Entomological Society of British Columbia 71: 25-32.

California Regional Water Quality Control Board Central Coast Region (RWQCB). 2004. Technical Support Document for Establishment of a Suspended Sediment Total Maximum Daily Load for the Pajaro River Watershed. May 2004. San Luis Obispo, California.

Carruth, L.L., R.E. Jones, and D.O. Norris. 2002. Cortisol and pacific salmon: a new look at the role of stress hormones in olfaction and home-stream migration. Integrative and Comparative Biology 42: 574-581.

Cereghino, R., M. Legalle, and P. Lavandier. 2004. Drift and benthic population structure of the mayfly *Rhithrogena semicolorata* (Heptageniidae) under natural and hydropeaking conditions. Hydrobiologia 519: 127-133.

Chapman, D.W. 1966. The relative contributions of aquatic and terrestrial primary producers to the trophic relations of stream organisms. *In:* K.W. Cummins et al., (editors). Special publications of the Pymatuning Laboratory of Field Biology, Vol 4:116-130. University of Pittsburgh, Pittsburgh, Pennsylvania.

Chapman, D.W. 1988. Critical review of variables used to define effects of fines in redds of large salmonids. Transactions of the American Fisheries Society 117:1-21.

Chapman, D.W. and R.L. Demory. 1963. Seasonal changes in the food ingested by aquatic insect larvae and nymphs in two Oregon streams. Ecology 44:140-146.

Chapman, D.W. and E. Knudsen. 1980. Channelization and livestock impacts on salmonid habitat and biomass in western Washington. Transactions of the American Fisheries Society 109: 357-363.

Clarke, D.G., and D.H. Wilber. (2000). "Assessment of potential impacts of dredging operations due to sediment resuspension," DOER Technical Notes Collection (ERDC TN-DOER-E9), U.S. Army Engineer Research and Development Center, Vicksburg, MS. www.wes.army.mil/el/dots/doer

Collier, M., R.H. Webb and J.C. Schmidt. 1996. Dams and rivers, a primer on the downstream effects of dams. U.S. Geological Survey, circular 1126.

Cooper, A.C. 1965. The effect of transported stream sediments on survival of sockeye and pink salmon eggs and alevin. International Pacific Salmon Fisheries Commission Bulletin 18.

Cordone, A.J. and D.W. Kelley. 1961. The influences of inorganic sediment on the aquatic life of streams. California Fish and Game 47: 189-228.

Corn, P.S. and R.B. Bury. 1989. Logging in western Oregon: responses of headwater habitats and stream amphibians. Forest Ecology and Management 29: 39-57.

Crowl, T.A. 1989. Effects of crayfish size, orientation, and movement on the reactive distance of largemouth bass foraging in clear and turbid water. Hydrobiologia 183: 133-140.

Culp, J.M., F.J. Wrona, and R.W. Davies. 1986. Response of stream benthos and drift to fine sediment deposition versus transport. Canadian Journal of Zoology 64: 1345-1351.

Culp, J.M., S.J. Walde, and R.W. Davies. 1983. Relative importance of substrate particle size and detritus to stream benthic macroinvertebrate microdistribution. Canadian Journal of Fisheries and Aquatic Sciences 40: 1568-1574.

Cummins, K.W. 1962. An evaluation of some techniques for the collection and analysis of benthic samples with special emphasis on lotic waters. The American Midland Naturalist 67(2): 477-504.

Cummins, K.W. and M.J. Klug. 1979. Feeding ecology of stream invertebrates. Annual Review of Ecology and Systematics 10: 147-172.

Davies-Colley, R.J. and M.E. Close. 1990. Water colour and clarity of New Zealand rivers under baseflow conditions. New Zealand Journal of Marine and Freshwater Research 24:357-365.

Davies-Colley, R.J. and D.G. Smith. 2001. Turbidity, suspended sediment, and water clarity: a review. Journal of the American Water Resources Association 37(5): 1085-1101.

Duchrow, R.M. and W.H. Everhart. 1971. Turbidity measurement. Transactions of the American Fisheries Society No. 4: 682-690.

Edwards, T.K. and D.G. Glyson. 1999. Field Methods for Measurement of Fluvial Sediment. US Geological Survey Techniques of Water-Resources Investigations, Book 3, Chapter C2: US Geological Survey.

Elliott, J.M. 1989. The critical-period concept for juvenile survival and its relevance for population in young sea-trout, *Salmo trutta*. Journal of Fish Biology 35:91-98.

European Inland Fisheries Advisory Commission (EIFAC). 1964. Water quality criteria for European freshwater fish. Report on finely divided solids and inland fisheries. EIFAC Technical Paper 1.

Flosi, G., S. Downie, J. Hopelain, M. Bird, R. Coey, and B. Collins. 1998. Sediment Sources. *In:* California Department of Fish and Game California Salmonid Stream Habitat Restoration Manual (3rd ed): II-12 - II-14. http://www.dfg.ca.gov/nafwb/manual.html

Fredriksen, R.L. and R.D. Harr. 1981. Soil, Vegetation, and Watershed Management. *In:* P.E. Heilman, H.W. Anderson, and D.M. Baumgartner (editors), Forest Soils of the Douglas Fir Region. Washington State University Co-op Extension Service: 231-260.

Furniss, M.J., T.D. Roelofs, and C.S. Yee. 1991. Road construction and maintenance. *In:* W.R. Meehan (ed.), Influences of Forest and Rangeland Management on Salmonid Fishes and their Habitats. American Fisheries Society Special Publication 19: 297-323.

Ginetz, R.M. and P.A. Larkin. 1976. Factors affecting rainbow trout (*Salmo gairdneri*) predation on migrant fry of sockeye salmon (*Oncorhynchus nerka*). Journal of the Fisheries Research Board of Canada 33: 19-24.

Gottschalk, L.C. 1964. Reservoir sedimentation In: V.T. Chow (ed.), Handbook of Applied Hydrology. New York, New York, USA: McGraw-Hill.

Gradall, K.S. and W.A. Swenson. 1982. Responses of brook trout and creek chubs to turbidity. Transactions of the American Fisheries Society 111: 392-395.

Graham, A.A. 1990. Siltation of stone-surface periphyton in rivers by clay-size particles from low concentrations in suspension. Hydrobiologia 199: 107-115.

Grant, G.E., J.C. Schmidt and S.L. Lewis. 2003. A geological framework for interpreting downstream effects of dams on rivers. Geology and Geomorphology of the Deschutes River, Oregon. Water Science and Application 7, American Geophysical Union.

Gray, J. (ed) and G.D. Glysson (ed). 2003. Proceedings of the Federal Interagency Workshop on Turbidity and other Sediment Surrogates, April 30-May 2, 2002 USGS Circular, 1250: 56pp.

Gregory, R.S. 1990. Effects of turbidity on benthic foraging and predation risk in juvenile chinook salmon. *In:* C.A. Simenstad (editor), Effects of Dredging on Anadromous Pacific Coast Fishes - Workshop proceedings, Sept 8-9, 1988: 64-73.

Gregory, R.S. 1993. Effect of turbidity on the predator avoidance behaviour of juvenile chinook salmon (*Oncorhynchus tshawytscha*). Canadian Journal of Fisheries and Aquatic Sciences 50: 241-246.

Gregory, R.S. and C.D. Levings. 1996. The effects of turbidity and vegetation on the risk of juvenile salmonids *Oncorhynchus* spp., to predation by adult cutthroat trout, *O. clarki*. Environmental Biology of Fishes 47: 279-288.

Gregory, R.S. and T.G. Northcote. 1993. Surface, planktonic, and benthic foraging by juvenile chinook salmon (*Oncorhynchus tshawytscha*) in turbid laboratory conditions. Canadian Journal of Fisheries and Aquatic Sciences 50: 233-240.

Gregory, K.J. and D.E. Walling. 1973. Drainage Basin Form and Process, a geomorphological approach. Halsted Press, a Division of John Wiley & Sons, New York.

Groot, C., and L. Margolis. 1991. Pacific Salmon Life Histories. University of British Columbia Press, Vancouver, B.C.

Hall, J.D., and R.L. Lantz. 1969. Effects of logging on the habitat of coho salmon and cutthroat trout in coastal streams. *In* T.G. Northcote, editor. Symposium on Salmon and Trout in Streams. H.R. MacMillan Lectures in Fisheries, University of British Columbia, Vancouver, B.C.

Henley, W.F., M.A. Patterson, R.J. Neves, and A.D. Lemly. 2000. Effects of sedimentation and turbidity on lotic food webs: a concise review for natural resource managers. Reviews in Fisheries Science 8(2): 125-139.

Hicks, D.M. and B. Gomez. 2003. Sediment transport *In:* G.M. Kondolph, H. Piegay, (editors.), Tools in Fluvial Geomorphology. West Sussex, England: John Wiley & Sons.

Holtby, L.B. 1988. Effects of logging on stream temperatures in Carnation Creek, British Columbia, and associated impacts on the coho salmon *(Oncorhynchus kisutch*). Canadian Journal of Fisheries and Aquatic Sciences 45: 502-515.

Holtby, L.B., T.E. McMahon, and J.C. Scrivener. 1989. Stream temperatures and inter-annual variability in the emigration timing of coho salmon (*Oncorhynchus kisutch*) smolts and fry and chum salmon (*O. keta*) fry from Carnation Creek, British Columbia. Canadian Journal of Fisheries and Aquatic Sciences 46: 1396-1405.

Idaho Department of Environmental Quality (IDEQ). 2003. Mid Snake River/Succor Creek Subbasin Assessment and TMDL April 2003.

Kondolph, G.M. 2000. Assessing salmonid spawning gravel quality. Transactions of the American Fisheries Society 129:262-281.

Koski, V. 1966. The survival of coho salmon (*Oncorhynchus kisutch*) from egg deposition to emergence in three Oregon coastal streams. Thesis, Oregon State University.

Kupferberg, S.J. 1996. Hydrologic and geomorphic factors affecting conservation of a river-breeding frog (*Rana boylii*). Ecological Applications 6: 1332-1344.

Lake, R.G., and S.G. Hinch. 1999. Acute effects of suspended sediment angularity on juvenile coho salmon (*Oncorhynchus kisutch*). Canadian Journal of Fisheries and Aquatic Sciences 56:862-867.

Lane, E.W. 1947. Report of the subcommittee on sediment terminology. Transactions of the American Geophysical Union 28: 936-938.

Langer, O.E. 1980. Effects of sedimentation on salmonid stream life. Department of Indian Affairs and Northern Development, Whitehorse, Yukon Territory.

Lemly, A.D. 1982. Modification of benthic insect communities in polluted streams: combined effects of sedimentation and nutrient enrichment. Hydrobiologia 87: 229-245.

Lewis, J. 2002. Estimation of suspended sediment flux in streams using continuous turbidity and flow data coupled with laboratory concentrations. Turbidity and Other Surrogates Workshop - Reno, NV April 30-May 2: 3pp.

Lewis, D.J., K.W. Tate, R.A. Dahlgren and J. Newell. 2002. Turbidity and total suspended solid concentration dynamics in streamflow from California oak woodland watersheds. USDA Forest Service Gen. Tech. Rep. PSW-GTR-184.

Lind, A.J., H.H. Welsh, and R.A. Wilson. 1996. The effects of a dam on breeding habitat and egg survival of the foothill yellow-legged frog (*Rana boylii*) in northwestern California. Herpetological Review 27: 62-67.

Lisle, T.E. 1989. Sediment transport and resulting deposition in spawning gravels, north coastal California. Water Resources Research 25:1303-1319.

Lloyd, D.S. 1987. Turbidity as a water quality standard for salmonid habitats in Alaska. North American Journal of Fisheries Management 7:34-45.

Lloyd, D.S., J.P. Koenings, J.D., and LaPerriere. 1987. Effects of turbidity in fresh waters of Alaska. North American Journal of Fisheries Management 7: 18-33.

Lotspeich, F.B., and F.H. Everest. 1981. A new method for reporting and interpreting textural composition of spawning gravel. USDA Forest Service Pacific Northwest Forest and Range Experimental Station, Research Note PNW-369.

Lowe, W.H., K.H. Nislow, and D.T. Bolge. 2004. Stage-specific and interactive effects of sedimentation and trout on a headwater stream salamander. Ecological Applications 14: 164-172.

Madej, M.A., M. Wilzbach, K. Cummins, C. Ellis, and S. Hadden. 2002. The contribution of suspended organic sediments to turbidity and sediment flux. Abstract from: Turbidity and Other Surrogates Workshop - Reno, NV April 30-May 2.

McCabe, G.D. and W.J. O'Brien. 1983. The effects of suspended silt on feeding and reproduction of *Daphnia pulex*. American Midland Naturalist 110:324-337.

McClelland, W.T. and M.A. Brusven. 1980. Effects of sedimentation on the behavior and distribution of riffle insects in a laboratory stream. Aquatic Insects 2(3): 161-169.

McCullough, D.A., G.W. Minshall, C.E. Cushing. 1979. Bioenergetics of lotic filter-feeding insects *Simulium* spp. (Diptera) and *Hydropsyche occidentalis* (Trichoptera) and their function in controlling organic transport in streams. Ecology 60(3): 585-596.

McGirr, D.J. 1974. Turbidity and filterable and non-filterable residue. Interlaboratory quality control study no. 10. Canada Centre for Inland Waters Report Series No. 37. Burlington, Ontario, 9 pp.

McNeil, W.J., and W.H. Ahnell. 1964. Success of pink salmon spawning relative to size of spawning bed materials. United States Fish and Wildlife Service Special Scientific Report--Fisheries No. 469.

Miner, J.G. and R.A. Stein. 1996. Detection of predators and habitat choice by small bluegills: effects of turbidity and alternative prey. Transactions of the American Fisheries Society 125:97-103.

Monaghan, M.T., S.A. Thomas, G.W. Minshall, J.D. Newbold and C.E. Cushing. 2001. The influence of filter-feeding benthic macroinvertebrates on the transport and deposition of particulate organic matter and diatoms in two streams. Limnology and Oceanography 46(5): 1091-1099.

Mount, J.F. 1995. California Rivers and Streams. University of California Press, Berkeley, and Los Angeles, California.

Murphy, M.L., C.P Hawkins, and N.H. Anderson. 1981. Effects of canopy modification and accumulated sediment on stream communities. Transactions of the American Fisheries Society 110:469-478.

Nelson, R.L., M.L. McHenry, and W.S. Platts. 1991. Mining. *In:* W.R. Meehan (ed.), Influences of Forest and Rangeland Management on Salmonid Fishes and their Habitats. American Fisheries Society Special Publication 19: 425-457.

Newcombe, C.P. 2003. Impact assessment model for clear water fishes exposed to excessively cloudy water. Journal of the American Water Resources Association 39: 529-544.

Newcombe, C.P. and D.D. MacDonald. 1991. Effects of suspended sediments on aquatic ecosystems. North American Journal of Fisheries Management 11: 72-82.

Newcombe, C.P. and J.O.T. Jensen. 1996. Channel suspended sediment and fisheries: A synthesis for quantitative assessment of risk and impact. North American Journal of Fisheries Management 16(4): 693-727.

Noggle, C.C. 1978. Behavioral, physiological and lethal effects of suspended sediment on juvenile salmonids. Masters Thesis, University of Washington, Seattle. 87 pp.

Platts, W.S., M.A. Shirazi, and D.H. Lewis. 1979. Sediment particle sizes used by salmon for spawning with methods for evaluation. U.S. Environmental Protection Agency, EPA Report EPA-600/3-79-043.

Redding, J.M., C.B. Schreck, and F.H. Everest. 1987. Physiological effects on coho salmon and steelhead of exposure to suspended solids. Transactions of the American Fisheries Society 116: 737-744.

Reid, S.M. and P.G. Anderson. Suspended sediment and turbidity restrictions associated with instream construction activities in the United States: an assessment of biological relevance. http://aplwww.alliance-pipeline.com/contentfiles/30____TSS_Criteria.pdf.

Reid, S.M., M.G. Fox, T.H. Whillans. 1999. Influence of turbidity on piscivory in largemouth bass (*Micropterus salmoides*). Canadian Journal of Fisheries and Aquatic Sciences 56: 1362-1369.

Reiser, D.W. and M.P. Ramey, and T.R. Lambert. 1985. Review of flushing flow requirements in regulated streams. Dep. Eng. Res. Pacific Gas and Electric Co., San Ramon, CA. 97 pp.

Reiser, D.W., and R.G. White. 1988. Effects of two sediment size-classes on survival of steelhead and chinook salmon eggs. North American Journal of Fisheries Management 8:432-437.

Reiser, D.W. and T.A. Wesche. 1977. Determination of physical and hydraulic preferences of brown and brook trout in the selection of spawning locations. Wyoming Water Resources Research Institute, Water Resources Series 64, Laramie, Wyoming.

Relyea, C.D., G.W. Minshall, and R.J. Danehy. 2000. Stream insects as bioindicators of fine sediments. *In:* Proceedings Watershed 2000, Water Environment Specialty Conference. Vancouver, British Columbia, Canada.

Richards, C. and K.L. Bacon. 1994. Influence of fine sediment on macroinvertebrate colonization of surface and hyporheic stream substrates. Great Basin Naturalist 54(2): 106-113.

Robinson, C.T. and G.W. Minshall. 1986. Effects of disturbance frequency on stream benthic community structure in relation to canopy cover and season. Journal of the North American Benthological Society 5(3): 237-248.

Rosenberg, D.M. and A.P. Wiens. 1978. Effects of sediment addition on macrobenthic invertebrates in a northern Canadian river. Water Research 12: 753-763.

Rosetta, T. 2004. Oregon Department of Environmental Quality, Technical basis for revising turbidity criteria – Draft, February 2004: 1-85.

Rowe, D.K., T.L. Dean, E. Williams, J.P. Smith. 2003. Effects of turbidity on the ability of juvenile rainbow trout, *Oncorhynchus mykiss*, to feed on limnetic and benthic prey in laboratory tanks. New Zealand Journal of Marine and Freshwater Research 37: 45-52.

Ryan, P.A. 1991. Environmental effects of sediment on New Zealand streams: a review. New Zealand Journal of Marine and Freshwater Research 25: 207-221.

Servizi, J.A. and D.W. Martens. 1987. Some effects of suspended Fraser River sediments on sockeye salmon, *Oncorhynchus nerka*. *In* Smith, H.D., L. Margolis, and C.C. Wood, Editors, Sockeye salmon (*Oncorhynchus nerka*), population biology and future management. Canadian Special Publications in Fisheries and Aquatic Sciences No. 96, 254-264.

Servizi, J.A. and D.W. Martens. 1991. Effect of temperature, season, and fish size on acute lethality of suspended sediments to coho salmon (*Oncorhynchus kisutch*). Canadian Journal of Fisheries and Aquatic Sciences. 48(3):493-497.

Servizi, J.A. and D.W. Martens. 1992. Sublethal responses of coho salmon (*Oncorhynchus kisutch*) to suspended sediments. Canadian Journal of Fisheries and Aquatic Science 49(7): 1389-1395.

Shaw, E.A. and J.S. Richardson. 2001. Direct and indirect effects of sediment pulse duration on stream invertebrate assemblages and rainbow trout (*Oncorhynchus mykiss*) growth and survival. Canadian Journal of Fisheries and Aquatic Sciences 58: 2213-2221.

Sigler, J.W. 1990. Effects of chronic turbidity on anadromous salmonids: recent studies and assessment techniques perspective. *In:* C.A. Simenstad (editor), Effects of Dredging on Anadromous Pacific Coast Fishes. Workshop proceedings, University of Washington and WA Sea Grant Program, Sept 8-9, 1988: 26-37.

Sigler, J.W., T.C. Bjornn and F.H. Everest. 1984. Effects of chronic turbidity on density and growth of steelheads and coho salmon. Transactions of the American Fisheries Society 113: 142-150.

Simenstad, C.A. 1990. Effects of Dredging on Anadromous Pacific Coast Fishes - Workshop Proceedings, Seattle, September 8-9, 1988. C.A. Simenstad, editor. University of Washington and WA Sea Grant Program, Sept 8-9, 1988: 26-37.

Simons, D.B. and F. Senturk. 1992. Sediment Transport Technology - Water Resources Publications, Littleton, Colorado.

Smith, D.G. and C.M. Hoover. 1999. Use of a viewer box in Secchi disk measurements. Journal of the American Water Resources Association 35:1183-1189.

Snyder, D.E. 1983. Fish eggs and larvae. *In:* L.A. Nielsen and D.L. Johnson, Editors. Fisheries Techniques. American Fisheries Society, Bethesda, Maryland.

Sorenson, D.L., M.M. McCarthy, E.J. Middlebrooks, and D.B. Porcella. 1977. Suspended and dissolved solids effects on freshwater biota: a review. U.S. Environmental Protection Agency, EPA-60Q/3-77-042, Corvallis: 64 pp.

Spence, B.C., G.A. Lomnicky, R.M. Hughes, and R P. Novitzki. 1996. Effects of Human Activities. *In:* An Ecosystem Approach to Salmonid Conservation TR-4501-96-6057. ManTech Environmental Research Services Corp., Corvallis, OR. (Available from the National Marine Fisheries Service, Portland, Oregon): 61pp. http://www.nwr.noaa.gov/1habcon/habweb/habguide/ManTech/front.htm

Stebbins, R.C. 1951. Amphibians of Western North America. University of California Press, Berkeley, California.

Storer, T.I. 1925. A synopsis of the amphibia of California. University of California Publications in Zoology 27: 1-342.

Strand, R.M. and R.W. Merritt. 1997. Effects of episodic sedimentation on the net-spinning caddisflies *Hydropsyche betteni* and *Ceratopsyche sparna* (Trichoptera: Hydropsychidae). Environmental Pollution 98(1): 129-134.

Sumner, F.H. and O.R. Smith. 1940. Hydraulic Mining and Debris Dams in Relation to Fish Life in the American and Yuba Rivers of California. California Fish and Game Vol. 26, No. 1: 2-22.

Swanston, D.N. 1991. Natural Processes. *In:* W.R. Meehan (editor), Influences of Forest and Rangeland Management on Salmonid Fishes and their Habitats. American Fisheries Society Special Publication 19: 139-179.

Sweka, J.A. and K.J. Hartman. 2001a. Influence of turbidity on brook trout reactive distance and foraging success. Transactions of the American Fisheries Society 130: 138-146.

Sweka, J.A. and K.J. Hartman. 2001b. Effects of turbidity on prey consumption and growth in brook trout and implication for bioenergetics modeling. Canadian Journal of Fisheries and Aquatic Sciences 58: 386-393.

Sweka, J.A. and K.J. Hartman. 2003. Reduction of reactive distance and foraging success in smallmouth bass, *Micropterus dolomieu*, exposed to elevated turbidity levels. Environmental Biology of Fishes 67: 341-347.

Tappel, P.D., and T.C. Bjornn. 1983. A new method of relating size of spawning gravel to salmonid embryo survival. North American Journal of Fisheries Management 3:123-135.

Tippets, W.E. and P.B. Moyle. 1978. Epibenthic feeding by rainbow trout (*Salmo gairdneri*) in the McCloud River. California Journal of Animal Ecology 47: 549-559.

United States Environmental Protection Agency (USEPA). 1999. Guidance Manual for Compliance with the Interim Enhanced Surface Water Treatment Rule. Turbidity Provisions. U.S. Environmental Protection Agency, Office of Water.

USEPA. 2004. The Milltown Reservoir Record of Decision. United States Environmental Protection Agency Region 8, Montana Office Helena, Montana 59626.

USEPA and Federal Energy Regulatory Commission (FERC). 2004. Biological assessment, bull trout, of the Milltown Reservoir Sediments Operable Unit revised proposed plan and for the Milltown Hydroelectric Project. Prepared by CH2MHILL and CFBLLC. 176p.

United States Fish and Wildlife Service (USFWS). 2004. Biological Opinion for threatened and endangered species USEPA Superfund Program Revised Proposed Clean-up Plan for the Milltown Reservoir Sediments Operable Unit of the Milltown Reservoir/Clark Fork River superfund site. December 17, 2004.

United States Geological Survey (USGS). Variously dated. National field manual for the collection of water-quality data: U.S. Geological Survey Techniques of Water-Resources Investigations, book 9, chaps. A1-A9, available online at http://pubs.water.usgs.gov/twri9A.

USGS. 2003. Proceedings of the Federal Interagency Workshop on Turbidity and other Sediment Surrogates, April 30 – May 2, 2002. USGS Circular, 1250, 56 pp.

USGS. 2004. Office of Water Quality Technical Memorandum 2004.03. Revision of NFM Chapter 6, Section 6.7 – Turbidity: 14 pp.

Van Nieuwenhuyse, E.E. and J.D. LaPerriere. 1986. Effects of placer gold mining on primary production in subarctic streams of Alaska Water Resources Bulletin 22(1): 91-99.

Vinyard, G.L. and A.C. Yuan. 1996. Effects of turbidity on feeding rates of Lahontan cutthroat trout (*Oncorhynchus Clarki Henshawi*) and Lahontan redside shiner (*Richardsonius egregius*). Great Basin Naturalist 56(2): 157-161.

Vogel, J.L. and D.A. Beauchamp. 1999. Effects of light, prey size, and turbidity on reaction distances of lake trout (*Salvelinus namaycush*) to salmonid prey. Canadian Journal of Fisheries and Aquatic Sciences 56: 1293-1297.

Vuori, K.M. and I. Joensuu. 1996. Impact of forest drainage on the macroinvertebrates of a small boreal headwater stream: do buffer strips protect lotic biodiversity? Biological Conservation 77: 87-95.

Wagener, S.M. and J.D. LaPerriere. 1985. Effects of placer mining on the invertebrate communities of interior Alaska streams. Freshwater Invertebrate Biology 4(4): 208-214.

Walkotten, W.J. 1973. A freezing technique for sampling streambed gravel. USDA Forest Service Research Note PNW-205.

Wallen, E.I. 1951. The direct effect of turbidity on fishes. Oklahoma Agricultural and Mechanical College, Arts and Science Studies, Biology Series no. 2.

Waters, T.F. 1995. Sediment in Streams: Sources, Biological Effects, and Control. American Fisheries Society Monograph 7, Bethesda, Maryland.

Waters, T.F. 1972. The drift of stream insects. Annual Review of Entomology 17: 253-272.

Welsh, H.H. and L.M. Ollivier. 1998. Stream amphibians as indicators of ecosystem stress: a case study from California's redwoods. Ecological Applications 8: 1118-1132.

Wentworth, C.K. 1922. A scale of grade and class terms for clastic sediments. Journal of Geology 30: 377-392.

Wheeler, M.J. and W.D. Fraser. 2004. Hormone Assays in Biological Fluids (Methods in Molecular Biology). Humana Press: Totowa, New Jersey.

Whitman, R.P., T.P. Quinn, and E.L. Brannon. 1982. Influence of suspended volcanic ash on homing behavior of adult chinook salmon. Transactions of the American Fisheries Society 111: 63-69.

Williams, G.P. and M.G. Wolman. 1984. Sediment Loads. In: Downstream Effects of Dams on Alluvial Rivers. Geological Survey Professional Paper 1286: 8-14. http://dlib.stanford.edu:6520/text1/dd-ill/downstream.pdf

Willmer, P., G. Stone, and I.A. Johnston. 1999. Environmental Physiology of Animals Blackwell: Oxford, England.

Young, M.K. and W.A. Hubert. 1989. Substrate alteration by spawning brook trout in a southeastern Wyoming stream. Transactions of the American Fisheries Society 118:379-385.

Young, M.K., W.A. Hubert, and T.A. Wesche. 1990. Fines in redds of large salmonids. Transactions of the American Fisheries Society 119:156-162.

Young, M.K., W.A. Hubert, and T.A. Wesche. 1991. Selection of measures of substrate composition to estimate survival to emergence of salmonids and to detect changes in stream substrates. North American Journal of Fisheries Management 11:339-346.

Ziegler, A.C. 2003. Issues related to use of turbidity measurements as a surrogate for suspended sediment. *In:* Gray, J.R., and Glysson, G.D., eds., Proceedings of the Federal Interagency Workshop on Turbidity and Other Sediment Surrogates, April 30-May 2, 2002, Reno, Nevada: U.S. Geological Survey Circular 1250, p. 16-18. Available online http://ks.water.usgs.gov/kansas/pubs/biblio/author.html.

Zweifel, R. G. 1955. Ecology, distribution, and systematics of frogs of the *Rana boylii* group. University of California Publications in Zoology 54: 207-292.

Zweig, L.D. 2000. Effects of deposited sediment on stream benthic macroinvertebrate communities. Masters Thesis, University of Missouri-Columbia, 164 pp.

APPENDIX A

SEDIMENTATION EFFECTS ON AQUATIC ORGANISMS

1 SEDIMENTATION EFFECTS ON AQUATIC ORGANISMS

Appendix A provides a brief overview of literature related to deposited sediment on aquatic organisms and their habitat. Definitions and properties of sediment are provided, as well as a description of common techniques used for measurement. Literature related to effects of sediment deposition on invertebrates and fish is reviewed.

1.1 DEFINITION AND PROPERTIES

The US Geological Survey (Edwards and Glysson 1999) defines fluvial sediment as fragmentary material that originates mostly from weathering of rocks, which is transported by or deposited from water. Fluvial sediment includes inorganic, biological, and decomposed organic material.

Sediment is transported from watershed surfaces to channels in three main ways: soil loss and erosion from upland areas, mass movement such as landslides (Swanston 1991, Beschta 1996), and streambank or channel erosion. Once sediment is delivered to a stream, the two dominant mechanisms of sediment transport in streams are bedload (sliding, rolling, or bouncing along the bottom) and suspended load. Bedload transport principally affects the instream habitat characteristics of the channel bed. “Bed material load,” primarily coarse material such as sand and gravel, is a primary constituent of anadromous fish spawning habitat. Suspended load is typically limited to clay and silt-sized particles, which can be moved under most flows. The material that settles from the flowing water to the stream bed includes some proportion of each type, but is primarily bed material because of its greater average particle radius (Hicks and Gomez 2003).

Particle sorting in the water column is a function of time and distance from the initial disturbance. The coarsest particles sink quickly and travel short distances. Fine-grained sediments, which are mostly responsible for water cloudiness, remain in suspension for long periods of time. Therefore, coarse and fine sediments have different fates and different modes of effect on biota (Newcombe 2003). Sand particles (< 2.0 mm diameter) are moved as flows increase, but remain on the streambed between storms. During storm flows, larger sediments can be moved (Swanston 1991). In riffles and cascades, flows are shallow and fast and suspended sediment may be transported through these habitats. In deep pools with low velocities, suspended and bed-load sediments are more likely to be deposited.

The mode of transport influences the rate and profile of settled material and also dictates the methodology for measuring sediment load. Additional concepts include the stream’s transport capacity, which is a measure of the stream’s ability to transport bed load, and flow competence, which relates to the maximum size (particle radius) of sediment that can be moved by a specified flow condition (Hicks and Gomez 2003). In general, fine suspended material tends to be concentrated uniformly in flowing water, but coarser suspended material is concentrated near the streambed (Edwards and Glysson 1999).

1.2 SEDIMENT MEASUREMENT

1.2.1 BEDLOAD SAMPLING

Bedload composition varies across short distances because of streambed topography, variation in source material composition, and variable hydraulic conditions. Therefore, bedload should be sampled at evenly-spaced points across the stream channel. Subsamples of a single transect are combined to yield a single sample (one data point). Multiple transects of the same stream should be sampled similarly to yield a profile of the stream bedload (Beschta 1996). The Helley-Smith pressure-difference sampler is widely used because it is a relatively compact, hand-held unit and it appears to minimize hydraulic interference with stream flow and transported bed material (Hicks and Gomez 2003). The device consists of a rigid, square, 77 mm orifice and surrounding rigid frame, with a triangular or much larger cylindrical mesh bag that collects the bed load sediment as it enters the orifice. The frame is held against the streambed for a (recorded) standard time period, and bedload is collected into the net. At the end of the sampling period the sediment is transferred to an appropriate container for later analysis in the field or laboratory. The sampling efficiency and accuracy declines substantially if the bag is overfilled or if the mesh becomes clogged with fine particles, and it is essential to use that same type of bag for all samples within a single stream (Beschta 1981, Hicks and Gomez 2003).

Some doubt remains about sampling accuracy with the Helley-Smith sampler, but Hicks and Gomez (2003) suggest that calibration for each stream reduces or eliminates accuracy problems with this device. Conversely, the "bedload trap" sampling system is regarded as 100 percent accurate. This system consists of a chamber or box sunk into the stream bed, with an upstream lip that putatively intercepts 100 percent of the bedload if the opening is wide enough to catch saltating particles (Hicks and Gomez 2003). However, bedload traps are difficult and expensive to install, and depending on design, they may require extensive maintenance to preserve their functionality (Hicks and Gomez 2003).

1.2.2 BEDLOAD ANALYSIS

Bedload samples are weighed in the field or laboratory, and weight (M_b) is combined in a simple formula with subsample time duration (T), the number of subsamples (N), the wetted width of the stream channel (W), and an empirically derived unitless constant to yield instantaneous bedload transport rate (Q_b(kg/s)):

$$Q_b = \frac{M_b}{T} x \frac{1}{N} x \frac{W}{0.076}$$

Volumetric determination can be substituted for weight in the field if a sufficiently large balance is not available (Beschta 1996). The formula for total sediment mass (M_b) calculated from volumetric determination includes terms for Specific Weight (SW_b) of different types and mixtures of sediment (Gottschalk 1964) and sediment volume (V_b):

$$M_b = V_b \; x \; SW_b$$

If the sample is weighed in the laboratory, it should first be cooked in a muffle oven at 550°C for 24 hours to remove any organic material. If information on particle size distribution in the sample is desired, the sample can be sieved through a series of standard meshes and the yield from each sieve weighed to yield a particle size profile, usually plotted as a cumulative frequency on log paper (Beschta 1996).

1.3 EFFECTS OF DEPOSITED SEDIMENT ON MACROINVERTEBRATES

1.3.1 BEHAVIORAL EFFECTS

One of the most important behavioral effects of deposited sediment on aquatic invertebrates is an increase in drift density. Drift is the downstream transport of aquatic insects with the current (Cereghino et al. 2004). Drift is a natural phenomenon that is assumed to be an active, behavioral process allowing the regulation of benthic production and downstream colonization (Cereghino et al. 2004). This behavior also shows a diel periodicity, with higher drift density occurring during the night than the day and larger individuals entering the drift at night.

Natural or man-made disturbances, such as very sudden increases in flow (which may mobilize a substantial portion of the bedload) or a large sediment influx, can cause 'catastrophic drift.' This sudden and large-scale drift may be the result of dislodgment caused by rapid flow increases or avoidance behavior by the organisms due to the effects of transported sediment. Invertebrates also can become dislodged due to the effect of rolling or saltating particles. Culp et al. (1986) concluded that saltating sediments were the primary factor causing a reduction in benthic densities of more than 50 percent in their field study of fine sediment additions. They also found that there was both a distinct, immediate effect (in the form of increased drift) and delayed responses. Macroinvertebrates having a delayed response were initially present below the surface but became exposed to sediment effects during their vertical shift in distribution 6 to 9 hours after the sediment additions. The abundance and composition of the benthic assemblage is likely to be altered due to differing responses of species based on differences in the sizes and tolerances of the species and life stages present (Cereghino et al. 2004). These effects vary with season, reflecting differences in life stages and species present.

Net making species can be affected by fouling, ripping or burying of their nets (Strand and Merritt 1997). In turn, this could decrease food acquisition and result in a reduction in adult reproductive success due to time and energy costs. Strand and Merritt (1997) studied the effects of daily exposure to moderate levels of sedimentation on the net-spinning Trichoptera *Hydropsyche betteni* and *Ceratopsyche sparna*. In this experiment, daily additions of sediment were made over a 16-day period. Larval survival was reduced, although growth rates were not altered. Sedimentation treatments reduced larval survival of both species, but *H. betteni* was less tolerant and had lower survival. Non-lethal, behavioral effects were observed, although declines in abundance were not detected. Nets became clogged with sediment after exposure and were cleaned or replaced by the organisms prior to the onset of the next trial. Net maintenance costs were apparently negligible over the 16-day study. However, if younger larvae had been studied, they may have responded differently.

1.3.2 Habitat Impacts

Deposited sediment affects benthic invertebrates by directly altering the condition of the substrate they inhabit and indirectly by smothering periphyton. Periphyton is an important food source for many taxa, particularly grazers.

Various invertebrate taxa inhabit either the substrate surfaces (i.e., the top, bottom or sides of cobbles and boulders), the interstices of coarser material, or both the surface and interstices, depending on their life stage and habits (Brusven and Prather 1974). Additionally, those that inhabit the interstices may do so at great depth, thus occupying what is referred to as the hyporheic zone. When fine sediments embed cobbles, access to the interstitial and hyporheic habitats is restricted to a few specialized burrowing taxa (Brusven and Prather 1974). Sand size particles may be a more serious pollutant than silt in some streams because they remain settled during lower flows, whereas silts may be suspended and carried to slower portions of the river where they may settle. When a substantial amount of sediment settles on and around the coarser substrata, an impermeable sediment barrier may form, causing reductions in hyporheic oxygen levels. Ryan (1991) reported that a 12 to 17 percent increase in interstitial fine sediment caused a 16 to 40 percent reduction in invertebrate abundance in New Zealand streams.

Richards and Bacon (1994) found that macroinvertebrate colonization of the hyporheos was distinctly affected by the quantity of fine sediment that had filled the interstitial spaces, particularly sediment smaller than 1.50 mm. Not only does this size range of particles clog the interstitial spaces and thus alter subsurface flow, but also reduces the availability of dissolved oxygen. In their study, the total numbers of invertebrates were only 22 percent of those observed at the surface (Richards and Bacon 1994). This difference is not typically found in streams without a high proportion of fine sediment in the hyporheos. The ultimate effect of this may be a significant reduction in secondary production and food production for fish.

1.3.3 Community Response

The distribution of grazing invertebrates can be affected by the smothering of algal habitat and abrasion or scouring of cells (Vuori and Joensuu 1996). In turn, the numbers of predatory invertebrates and other secondary consumers in the system may decrease due to lack of food resources.

Studies have demonstrated that macroinvertebrate distribution is correlated with particle size and heterogeneity, as well as detritus in the substratum (Waters 1995, Culp et al. 1983). The importance of invertebrates as processors of organic matter and in the transfer of energy in aquatic systems is well-established (McCullough et al. 1979, Cummins and Klug 1979). If organic materials become buried by sediment it would be expected to impact the benthic community and interfere with organic matter processing. The deposition of organic material requires streams with rocky bottoms and available interstitial area. Excessive sediments and sedimentation can adversely affect the periphyton community and interfere with organic matter processing (McCelland and Brusven 1980).

Waters (1995) summarized research on the effects of deposited fine sediments on benthic invertebrates in streams. The diversity of species is often reduced. This includes reduction in sensitive species and life stages. Filter feeders and grazers are often reduced. This in turn may cause reductions in predaceous insect larvae.

The degree of substrate embeddedness within fine sediments is related to invertebrate composition and abundance (Waters 1995). Bjornn et al. 1977 found that when embeddedness was from zero to one-third, invertebrate communities were maintained. However, when it exceeded one-third, abundance declined by 50 percent. Research has found that the Ephemeroptera, Plecoptera, and Trichoptera (EPT) taxa are the least tolerant to sediment and are readily lost (McClelland and Brusven 1980), although burrowing types such as oligochaetes and Diptera may actually increase.

Zweig (2000) investigated the relation of benthic invertebrate communities to deposited sediment in four Missouri Ozark streams. Several community measures, including taxa richness, density, EPT richness, and EPT density, were related to percent cover and embeddedness. Deposited sediment level was shown to be related to community structure, with increasing sediment resulting in fewer taxa, lower densities, a greater proportion of shredders and lower proportion of gatherers, scrapers and filterers. It was also found that the proportion of burrowing taxa increased. Invertebrate density decreased substantially once 30 percent sediment cover was reached. This paper also demonstrated that the relationship of invertebrate response and tolerance to sediment must be evaluated at the genus or species levels due to the diversity that occurs at the family level.

Relyea et al. (2000) found that invertebrate species tolerances to fine sediment differed notably in an evaluation of data from 562 stream segments from Idaho, Oregon, Washington, and Wyoming. The objective of their study was to determine which taxa, functional feeding groups, or commonly used bioassessment metrics respond to increased fine inorganic sediment and might be used to develop a fine sediment bioassessment index. Their analysis revealed species-specific responses to the amount of fine sediment in a streambed. For example, the mayfly *Drunella doddsi* did not occur in streams with more than 37 percent fine sediment. In contrast, *Tricorythodes minutus* preferred fines and was found in high numbers where large amounts (up to 60 percent) of fine sediment occurred. Species from other taxa groups are also known to be fine sediment intolerant (Trichoptera and Plecoptera) or tolerant (Diptera). This study demonstrates the usefulness of identifying invertebrate tolerances to fine sediment for a given region as a tool for assessing fine sediment impacts.

McClelland and Brusven (1980) examined the effects of three levels of sedimentation on the behavior and distribution of three orders of aquatic insects (Ephemeroptera, Plecoptera and Trichoptera). Introduced sediment in laboratory streams caused the filling of substrate interstices and reduced the “effective” size of surface cobbles, which resulted in a decrease of macroinvertebrate density in the examined regions.

McCelland and Brusven (1980) attribute differences in macroinvertebrate response to differences in morphology, food requirements, and mode of respiration. As sediment level increased, fewer organisms were present. Ephemeroptera were the least sensitive to introduced quantities of sediments. *Rhithrogena robusta* is dorso-ventrally flattened and utilized the small spaces beneath and on the sides of sealed cobble substrates. *Ephemerella doddsi* possesses a ventral suction disk and was found to utilize exposed surfaces of cobbles. Differences between the

Trichopteran species *Rhyacophila acropedes* and *Arctopsyche grandis* were observed. *R. acropedes* is a free living type of Trichoptera and was more sensitive to increased sediment than was the net building *A. grandis*. However, the authors did note that this was a short duration study and over time sediment scour and deposition could reduce the available sites for net attachment. Plecopteran species (*Pteronarcys californica, Hesperoperla pacifica, Cultus sp. and Skwala sp.)* were the most sensitive to sediment increases of the macroinvertebrates examined.

Behavioral observations revealed that few of the macroinvertebrates attempted to gain access beneath cobbles sealed with fine sediment, even though the sediment remained loose and could have been excavated (McCelland and Brusven 1980). In natural streambeds, fine sediment is often composed of silt and clay which is highly cohesive and over time can cement the stream bottom and reduce habitat available for non-burrowing species. Therefore, it is likely that excessive sedimentation may be more harmful to deep-living than surface populations. The streambed surface may be periodically scoured during times of increase flow, whereas subsurface accumulations are more permanent (McCelland and Brusven 1980).

Persistent sediment pollution can result in permanent replacement of the assemblage with those that can tolerate high silt and burrow in the silt. The effects of moderate levels of chronic sedimentation (that does not cause total habitat transformation) are less well-known (Strand and Merritt 1997).

1.4 EFFECTS OF DEPOSITED SEDIMENT ON SALMONID FISH HABITAT

1.4.1 KEY LIFE HISTORY FEATURES

Most salmonid fish spawn in streams and rivers during various seasons specific to species, localities, and subpopulations. Anadromous fish migrate into rivers and streams from the ocean. Freshwater salmonids may move into streams from lentic habitat or move into suitable spawning habitat from elsewhere in streams and rivers. Spawning fish excavate egg-deposition sites known as “redds” in gravel stream beds, deposit eggs in excavated pockets, and cover the eggs with gravel to a depth of as much as 40 cm. The eggs hatch into alevin (egg sac stage larvae) in a few days to a few weeks. The alevin remain in the interstitial spaces within redds and survive and mature exclusively on yolk sac contents. Some time after the yolk sac is absorbed the fry emerge from the redds as free-swimmers, as much as several months after hatching (Groot and Margolis 1991). Depending on species, anadromous salmonid fry migrate downstream to estuarine or oceanic habitat or remain in fresh water, typically along riffles and pools (Groot and Margolis 1991). Fry of strictly freshwater species inhabit riffles during the winter and pools during the summer (Alexander and Hansen 1983). Some anadromous salmonid species perish after a single spawning run, and some others survive and return to the ocean.

1.4.2 SEDIMENT EFFECTS ON SPAWNING HABITAT

Bedload sediments – Heavy bedload sediment incursion into stream riffles and pools has been shown to have highly deleterious impacts on fish spawning habitat. As early as 1870 the Commissioners of Fisheries of California lamented that salmon had vanished from the Yuba and American Rivers of California because hydraulic mining sediments had covered all of the spawning beds—the Commission also noted that salmon readily swim through muddy water

"…if beyond they find clear water and clean gravelly bottoms" (Sumner and Smith 1940). Clearly, massive sedimentation subsequent to fire, road construction, deforestation, or land use changes may temporarily or permanently eliminate spawning habitat altogether and destroy embryos and alevin within existing redds, and thus reduce or extirpate the local year-class salmonid population. A threshold depth of overlaid sediment beyond which salmonids cannot spawn successfully undoubtedly exists but is apparently undocumented.

Fine sediments – Sufficient flowing water must always be available to developing embryos and alevin within redds in order to supply oxygen and remove metabolic wastes. "Permeability" of the redd is the empirical index of that availability, and it is governed by the availability of interstitial spaces within the redd gravel (Chapman 1988, Bjornn and Reiser 1991, Waters 1995). Salmonid fish tend to spawn where subsurface upwelling or downwelling currents exist (such as at pool tailwaters or at the upstream ends of riffles), which are usually associated with aggregate beds and which suggests that sufficient water influx is probably assured at the time of spawning (Reiser and Wesche 1977, cited by Bjornn and Reiser 1991, Kondolf 2000). As long as subsurface currents are present, the moderate presence of fine sediments within potential redd sites probably does not discourage spawning. Spawning female salmonids also excavate redd sites very energetically and they effectively clean most such fine sediment out of the redd gravel (Chapman 1988, Kondolf 2000). Fine, entrapped sediments also may already be less prevalent in traditional spawning areas than elsewhere in the streambed because repeated spawning by many fish over many years tends to maintain the gravel in favored redd areas in coarser configuration (Chapman 1988). Thus, although gravel or cobble substrate is critically important for spawning (Bjornn and Reiser 1991), spawning habitat selection may be initiated by environmental cues other than substrate composition, such as the presence of subsurface currents. Further, spawning habitat may not be particularly vulnerable to permanent damage or loss by moderate turbidity/suspended sediments as long as historic flow velocities are maintained. For example, substantial sedimentation consequent to moderate storm events does not usually alter redd site selection by spawning fish, even though it poses a threat to embryos if these events occur after spawning (Lisle 1989).

1.4.3 EFFECTS ON REDDS AND INCUBATION

Although redds offer security from predation and scouring, they are potential sediment traps and their distinctive shape may promote sediment-bearing water influx; suspended sediment load is very likely to be transported directly into redds (Cordone and Kelley 1961, Cooper 1965).

Sedimentation impacts to salmonid eggs and alevins have been studied directly in nature by trapping and counting all of the emerging fry from redds and comparing that number to the estimated redd clutch size (Snyder 1983, Chapman 1988). Sedimentation impacts on embryo survival also have been studied experimentally by inserting "eyed" salmon eggs into river gravel or existing redds (e.g., Reiser and White 1988). Laboratory studies involved constructing artificial egg pockets and measuring embryo survival as a function of sediment particle diameter percentage or percent distribution (Chapman 1988). Observational and experimental approaches have yielded substantial information on critical sedimentation impacts related to particle size and profile within the redd.

Subsequent studies revealed that significant egg or alevin mortality occurred in redds where sediments smaller than 0.8 mm diameter exceeded 20 percent of the redd composition, as sampled by various coring techniques (Waters 1995) (Figures A-1, A-2, A-3).

Other authors claim that a more sophisticated analysis of redd substratum characterization than simple percentage of fines yields a clearer, far less variable picture of salmonid embryo and alevin survival as a function of sediment diameter. Platts et al. (1979) suggested that the geometric mean particle size of a redd aggregate sample predicts spawning success far more precisely than does percentage of fines in the redd. Tappel and Bjornn (1983) used an experimentally-derived, log-normal distribution of particle sizes to describe redd gravel size composition. Lotspeich and Everest (1981) developed the "fredle index," which expresses substratum diameter composition as a variance (Waters 1995) (Figure A-4). The fredle index seems to be the most favored of these approaches (Chapman 1988, Waters 1995, Kondolf 2000), although Young et al. (1991) conducted experiments that showed the fredle index and geometric mean calculations yielded similar results. Kondolf (2000) maintains that sediments and gravel are too complex to expect any single-variable descriptor to be a good index. Young et al. (1989) also made the important point that overall substrate composition (including fine sediments) in the redd is of less significance to developing embryos than is the distribution of sediments. Fine sediments in the area immediately surrounding the egg pocket are of greater detriment to embryo survival than they are elsewhere in the redd. Measurement of sediment distribution within the redd requires special sampling techniques that are discussed below.

In a pivotal review, Chapman (1988) criticized most of the embryo mortality laboratory studies because they oversimplified the conditions that surround the egg clutch. He also criticized most of the field studies because they did not address all of the relevant environmental parameters that surround the egg pocket (fredle index, oxygenation, permeability, flow, etc). Chapman recommended that future work emphasize fry-trapping and multi-parameter studies of egg pockets under natural conditions. Young et al. (1991) criticized fry-trapping because it assumes an initial, near 100 percent egg fertilization and viability, it fails to account for known wide variation in fecundity as correlated with female length, and it fails to consider redd overlap or superimposition by multiple females.

Sampling redd gravel and egg pockets is also problematic, particularly where the objective is to avoid disturbing gravel stratification so that its structure can be studied directly. Although several techniques can be used (Walkotten 1973), only the freeze-core technique (e.g., Young and Hubert 1989) has allowed direct investigation of natural egg pockets and their surroundings. Direct study of egg pockets by freeze-core sampling in conjunction with the fredle index gravel descriptor has confirmed that female salmonids deliberately expel fine sediments from gravel that surrounds the egg pocket (Chapman 1988, Waters 1995). This has helped to differentiate permeability reduction-related mortality (which kills embryos) from emergence failure (which kills fry) (Waters 1995). Despite the inconsistencies among studies and the criticisms of methodologies, the preponderance of evidence derived from numerous studies across many sites, circumstances, species, and spawning systems indicates clearly that fine sediments ($\leq$0.833 mm) can accumulate in interstitial spaces of redd gravel, reduce redd permeability (McNeil and Ahnell 1964, Table A-1), and increase embryo mortality.

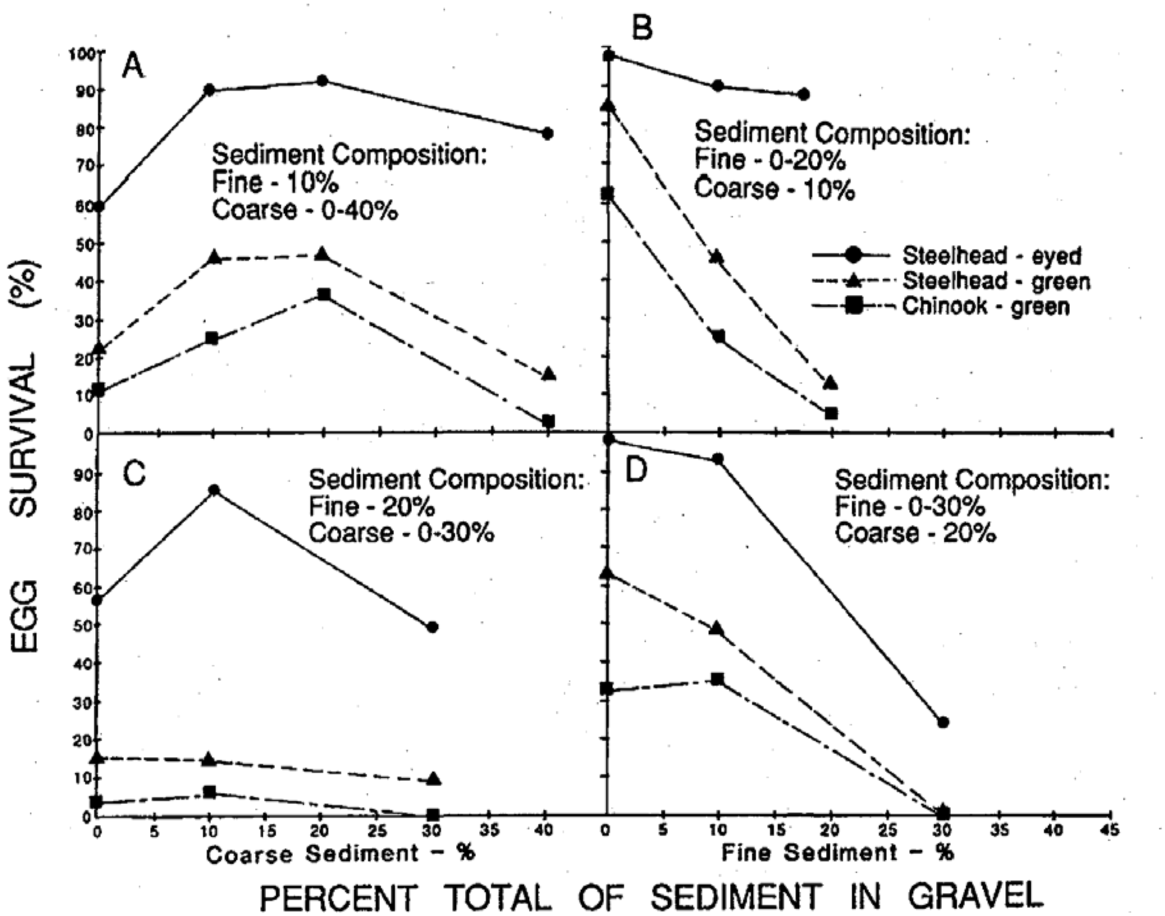


Figure A-1. Survival of Chinook Salmon and Steelhead Eggs in Relation to Combination of Fine (<0.84 mm) and Coarse (0.84-4.6 mm) Sediments in the Incubation Gravel (Reiser and White 1988).

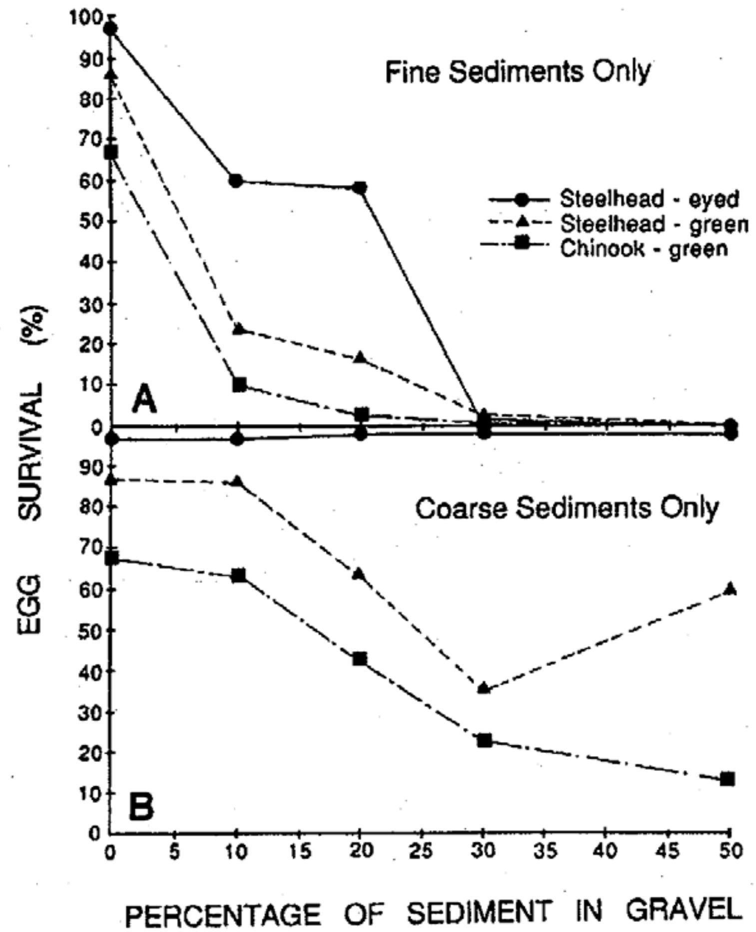


Figure A-2. Survival of Chinook Salmon and Steelhead Eggs in Relation to Proportions of (A) Fine (<0.84 mm) and (B) Coarse (0.84-4.6 mm) Sediments in the Incubation Gravel (Reiser and White 1988).

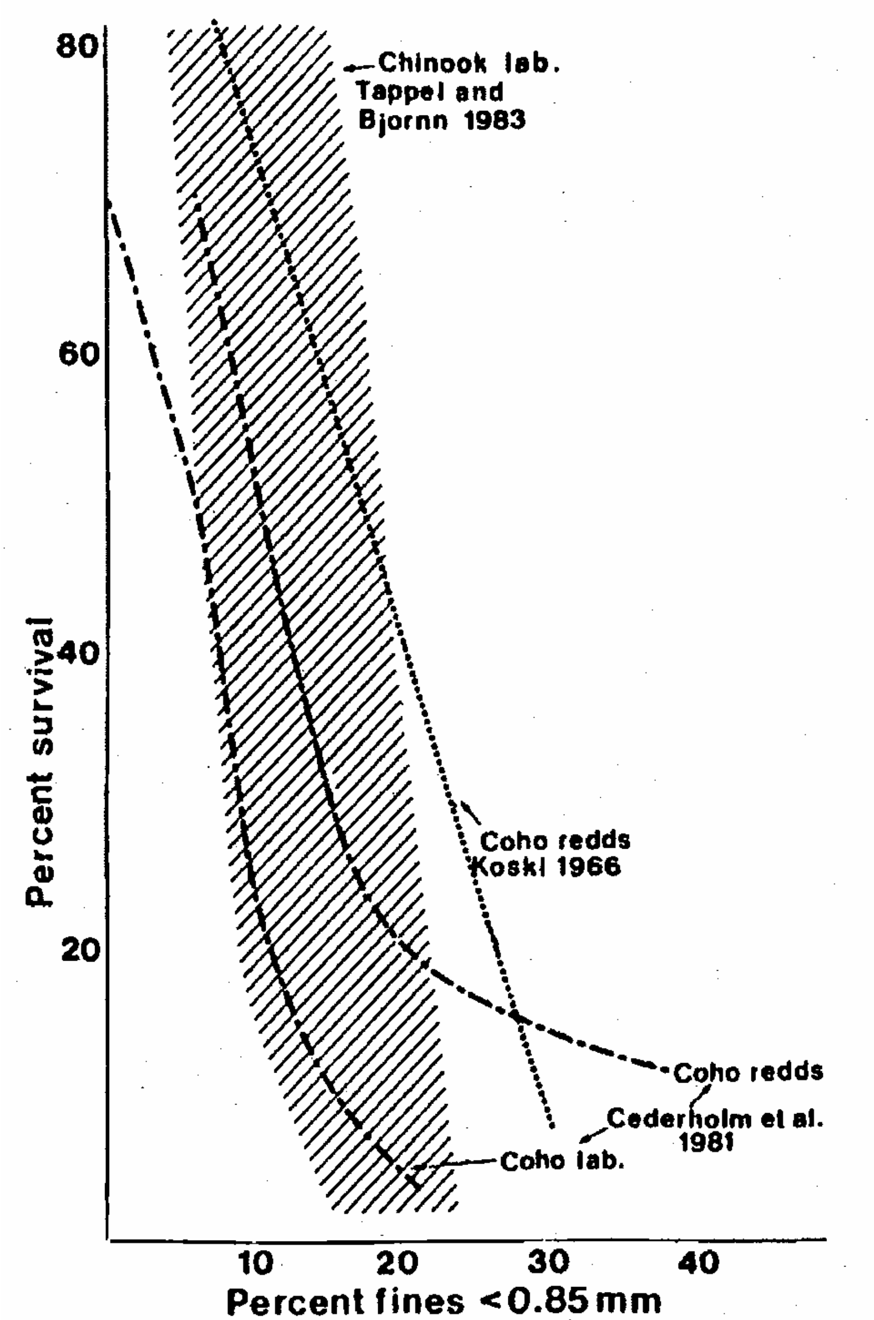


Figure A-3. Survival of Salmonid Embryos to Emergence in Relation to Fines Smaller than 0.85 mm in Diameter (Chapman 1988).

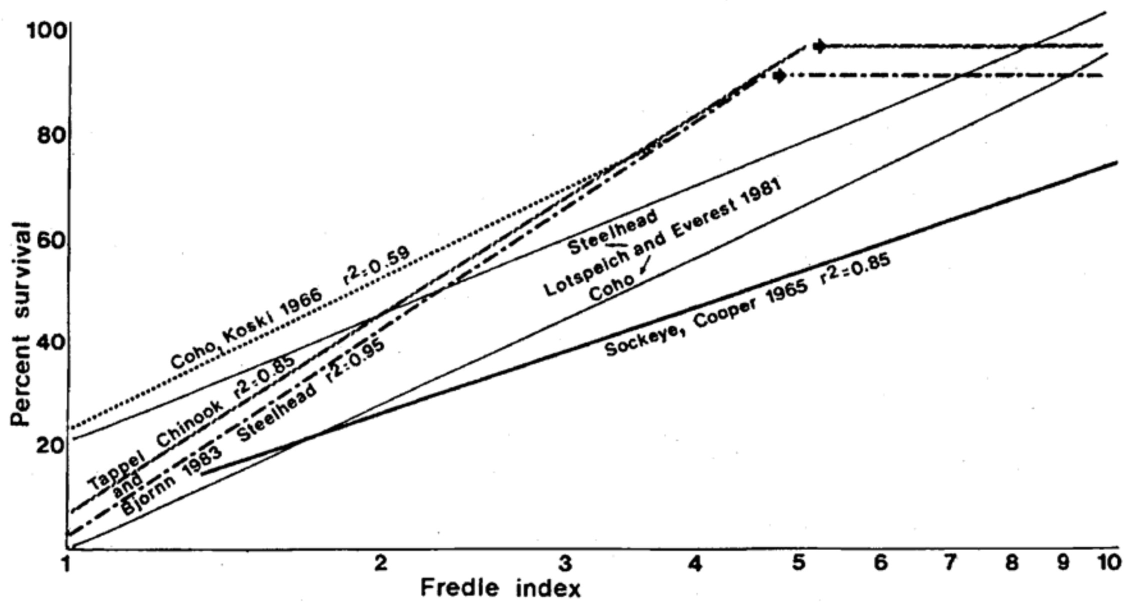


Figure A-4. **Survival to Emergence in Relation to the Fredle Index for Natural Coho Salmon Redds (Koski 1966) and for Chinook Salmon and Steelhead (Tappel and Bjornn 1983) and Sockeye Salmon (Cooper 1965) in Laboratory Gravel Mixes (Chapman 1988).**

Table A-1. Decrease in permeability of bottom materials from salmon spawning beds with addition of fine particles. From McNeil and Ahnell (1964).

No fine particles added		Fine particles added	
Percent passing through 0.833-mm sieve	**Permeability (cm/min)**	**Percent passing through 0.833-mm sieve**	**Permeability (cm/min)**
6.1	270	8.8	80
4.5	510	7.3	362
14.7	29	17.1	14
10.3	177	12.8	99
10.9	58	13.4	40
9.7	163	12.4	57
12.7	43	15.3	10
5.6	313	8.1	173

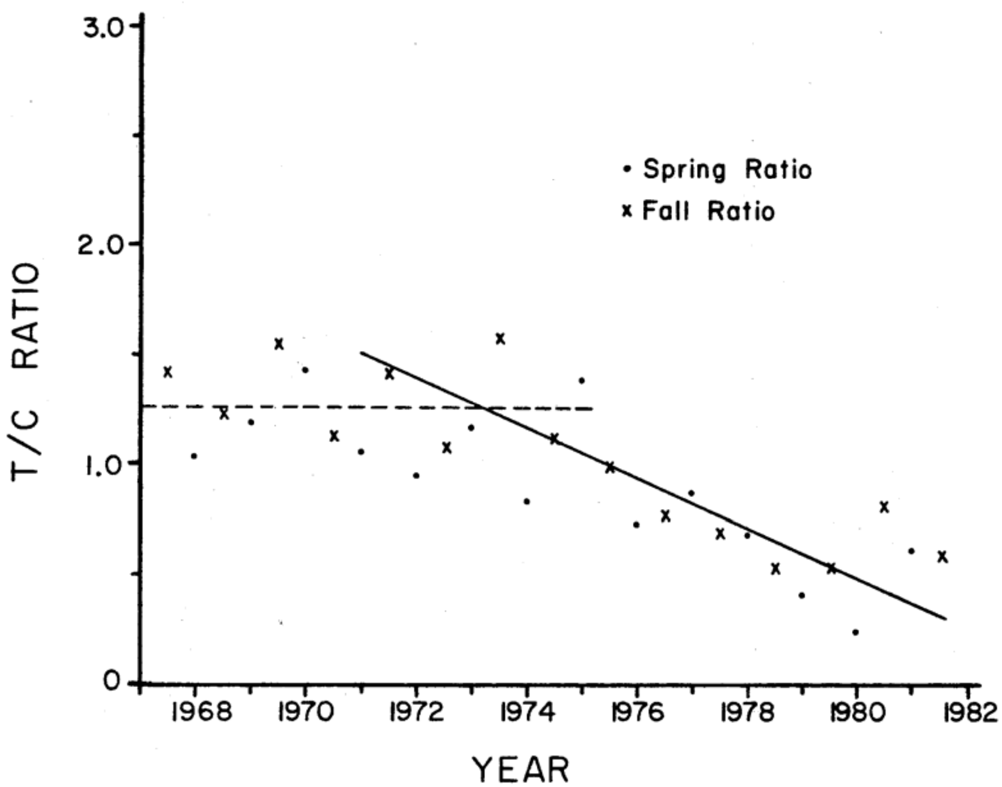
1.4.4 EFFECTS ON REDDS AND FRY EMERGENCE

Another potential impact of sedimentation on redds is entrapment or entombment of fry as they attempt to emerge from the redd to the free-swimming stage. Koski (1966) was the first to show that fry entombment by sediment accumulation is a significant, negative impact to salmonid recruitment. Although very fine sediments (0.8 mm or less) seemingly exert negative influence on incubation, the sediment particle diameter range that tends to entomb fry in redds is somewhat larger but it undoubtedly incorporates fines of the size range that smothers embryos. Commonly cited sediment particle diameters that impede fry emergence range from 1.0 to 6.0 mm, but fry survival to emergence begins to decline when these particle sizes reach only 10 percent of the redd composition (Hall and Lantz 1969, cited by Waters [1995], Bjornn and Reiser 1991). In several studies cited by Bjornn and Reiser (1991) just 50 percent of fry survived to emergence when particles of 1 to 3 mm in diameter reached 30 to 40 percent of the total redd composition.

1.4.5 EFFECTS ON REARING HABITAT

Recent research on this topic has highlighted sediment damage to fish stream rearing habitat. Very small juvenile fish are particularly vulnerable to such habitat perturbation because they depend on aggregate interstices for winter refuge and on relatively deep pools for summer cover. Both of these habitat types are particularly vulnerable to alteration by deposited sediment. Elliott (1989, cited by Waters 1995) showed that the "strength" of a salmonid year-class typically depends on critical events during juvenile stages, so massive die-off of anadromous fry or juveniles within single years has multi-generational consequences. Bjornn et al. (1974, cited by Waters 1995) were the first to show experimentally and by observation that winter fry habitat in gravel beds is severely impacted by the addition of sediments smaller than 6.4 mm in diameter because such sediments fill interstices and block fry from secure refugia. The effect is most pronounced at water temperatures less than 5°C, the range where fry normally remain in interstices. The relationship between fry population reduction and the degree of gravel bed embeddedness by sediments was linear. The sediment incursion that exerts the greatest negative impact on rearing habitat presumably originates from bedload because suspended load is probably not sufficiently dense or concentrated to fill large interstitial volumes.

Likewise, heavy bedload sediment incursion destroys summer pool rearing habitat along streams, not only by filling pools and blanketing structural cover but also by consequent alterations to channel morphology and fluvial processes. Waters (1995) cited numerous studies that correlated salmonid population reductions or extirpations with pool sedimentation from various sources. For example, Alexander and Hansen (1983) experimentally added over 4,200 cubic yards of sand bed load to a Michigan brook trout stream over five years (stream discharge = 20 ft^3/s), which increased normal bedload fourfold, widened and "shallowed" the stream, and filled pools. The stream channel became a shallow continuous sandy run without pools or riffles, water velocity and mean temperature increased, and the brook trout population/abundance declined by half (Figure A-5). Growth rate of individual fish did not change, but the brook trout population adjustment occurred primarily in the egg-to-fry or fry-to-fingerling stages, presumably attributable to decrease in food supply (Figure A-6) and possibly increase in predation. The fish population did not rebound until five years after the end of bedload supplementation.



Note: Dashed regression line is for pretreatment ratios (1968-1975) and the solid line is for treatment ratios (1971-1981).

Figure A-5. Ratio of the Total Number of Trout Present in the Treated Area (sand added to a Michigan trout stream), Divided by the Total Number of Trout Present in the Control Area, for each Spring and Fall (Alexander and Hansen 1983).

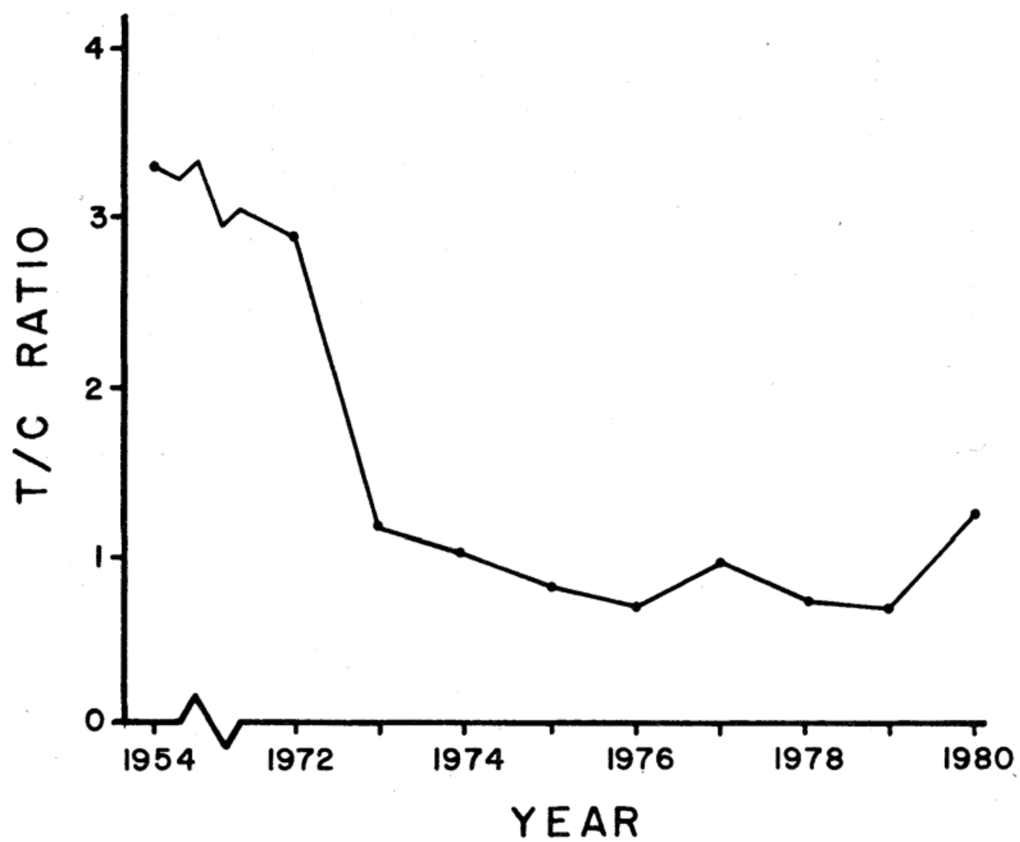


Figure A-6. Ratio T/C of Number of Invertebrates Per Square Foot of Stream Bottom (Alexander and Hansen 1983).

1.4.6 Summary: Effects of Turbidity/Sediments on Fish Spawning and Rearing Habitat

Effects on spawning. Moderate sedimentation is not known to affect salmonid spawning habitat or behavior. However, chronic, heavy bed load and possibly suspended sediments can blanket spawning gravel and disrupt characteristic subsurface flows so that salmonid spawning is disrupted or precluded.

Effects on redds and incubation. Although there is widespread agreement that substantial fine sediment (particle diameters <0.833 mm) infiltration into redd gravel reduces embryo survival, much disagreement remains about appropriate ways to investigate these impacts. The number and variety of pertinent studies, nearly all of which point to correlation between unusual embryo mortality and sediment infiltration into redd gravel, clearly indicate that fine sediment is detrimental to embryos in redds. Much work remains to elucidate the mechanisms of infiltration and subsequent reduction in permeability, the threshold sediment inflows that begin to cause embryo mortality, the relationship of turbidity to subsequent fines deposition in redds, and the long-term effects of chronic sedimentation on salmonid populations.

Effects on redds and fry emergence. Relatively fine suspended and bedload sediments (1-6 mm diameter) are known to fill gravel interstices sufficiently to entomb salmonid fry prior to emergence to the free-swimming stage. An impact threshold appears to occur when moderately fine sediments reach approximately 10 percent of redd composition. Complete entombment occurs when these sediments reach 40 percent of redd composition.

Effects on rearing habitat. Transported bedload sediments affect salmonid rearing habitat when the sediments are deposited over riffle aggregates that normally offer winter refuge habitat for salmonid fry, and when the sediments fill deep pools that normally offer summer foraging and refuge habitat. Sediments of diameter smaller than 6.2 mm (fine gravel and sand) most significantly affect riffle habitat; similar sediment sizes also entomb emerging fry. Summer refuge pools and associated cover can be obliterated by substantial increases of deposited sediment. Experiments have shown that a fourfold increase over natural transport can damage pool and riffle habitat for years, and the impacts are reflected in greatly reduced salmonid fish population levels. Affected waterways can recover when the excessive sediment transport ceases, but recovery can take many years.

REFERENCES

Alexander, G.R., and E.A. Hansen. 1983. Effects of sand bedload sediment on a brook trout population. Michigan Department of Natural Resources Fisheries Research Report 1906.

Beschta, R.L. 1981. Increased bag size improves Helley-Smith bedload sampler for use in streams with high sand and organic matter transport. *In*: Erosion and Sediment Transport Measurement Symposium Proceedings. International Association of Hydrologic Sciences, Publication 133, Washington, D.C.

Beschta. R.L. 1996. Suspended sediment and bedload. *In:* Methods in Stream Ecology, F.R. Hauer and G.A. Lamberti (editors). Academic Press, San Diego, CA, p. 123-143.

Bjornn, T.C., and D.W. Reiser. 1991. Habitat requirements of salmonids in streams. *In:* W.R. Meehan (editor). Influences of Forest and Rangeland Management on Salmonid Fishes and their Habitats. American Fisheries Society, Bethesda, Maryland.

Bjornn, T.C., M.A. Brusven, M.P. Molnau, J.H. Milligan, R.A. Klamt, E. Chacho, and C. Schaye. 1977. Transport of granitic sediment in streams and its effects on insects and fish. University of Idaho, College of Forestry, Wildlife and Range Sciences, Bulletin 17, Moscow, ID.

Bjornn, T.C., and seven coauthors. 1974. Sediment in streams and its effects on aquatic life. Water Resources Research Institute, Research Technical Completion Report Project B-025-IDA, Moscow, Idaho.

Brusven, M.A. and K.V. Prather. 1974. Influence of stream sediments on distribution of macrobenthos. Journal of the Entomological Society of British Columbia 71:25-32.

Cereghino, R., M. Legalle, and P. Lavandier. 2004. Drift and benthic population structure of the mayfly *Rhithrogena semicolorata* (Heptageniidae) under natural and hydropeaking conditions. Hydrobiologia 519:127-133.

Chapman, D.W. 1988. Critical review of variables used to define effects of fines in redds of large salmonids. Transactions of the American Fisheries Society 117:1-21.

Cooper, A.C. 1965. The effect of transported stream sediments on survival of sockeye and pink salmon eggs and alevin. International Pacific Salmon Fisheries Commission Bulletin 18.

Cordone, A.J., and D.W. Kelley. 1961. The influences of inorganic sediment on the aquatic life of streams. California Fish and Game 47:189-228.

Culp, J.M., S.J. Walde, and R.W. Davies. 1983. Relative importance of substrate particle size and detritus to stream benthic macroinvertebrate microdistribution. Canadian Journal of Fisheries and Aquatic Sciences 40:1568-1574.

Culp, J.M., F.J. Wrona, and R.W. Davies. 1986. Response of stream benthos and drift to fine sediment deposition versus transport. Canadian Journal of Zoology 64:1345-1351.

Cummins, K.W. and M.J. Klug. 1979. Feeding ecology of stream invertebrates. Annual Review of Ecology and Systematics 10:147-172.

Edwards, T.K. and D.G. Glyson. 1999. Field Methods for Measurement of Fluvial Sediment. US Geological Survey Techniques of Water-Resources Investigations, Book 3, Chapter C2: US Geological Survey.

Elliott, J.M. 1989. The critical-period concept for juvenile survival and its relevance for population in young sea-trout, *Salmo trutta*. Journal of Fish Biology 35:91-98.

Gottschalk, L.C. 1964. Reservoir sedimentation. *In:* V.T. Chow (editor), Handbook of Applied Hydrology. McGraw-Hill, New York, New York.

Groot, C., and L. Margolis. 1991. Pacific Salmon Life Histories. University of British Columbia Press, Vancouver, B.C.

Hall, J.D., and R.L. Lantz. 1969. Effects of logging on the habitat of coho salmon and cutthroat trout in coastal streams. *In*: T.G. Northcote (editor). Symposium on Salmon and Trout in Streams. H.R. MacMillan Lectures in Fisheries, University of British Columbia, Vancouver, B.C.

Hicks, D.M. and B. Gomez. 2003. Sediment transport. In: G.M. Kondolph, H. Piegay, (editors) Tools in Fluvial Geomorphology. John Wiley & Sons, West Sussex, England.

Kondolf, G.M. 2000. Assessing salmonid spawning gravel quality. Transactions of the American Fisheries Society 129: 262-281.

Koski, V. 1966. The survival of coho salmon (*Oncorhynchus kisutch*) from egg deposition to emergence in three Oregon coastal streams. Thesis, Oregon State University.

Lisle, T.E. 1989. Sediment transport and resulting deposition in spawning gravels, north coastal California. Water Resources Research 25:1303-1319.

Lotspeich, F.B., and F.H. Everest. 1981. A new method for reporting and interpreting textural composition of spawning gravel. USDA Forest Service Pacific Northwest Forest and Range Experimental Station, Research Note PNW-369.

McClelland, W.T., M.A. Brusven. 1980. Effects of sedimentation on the behavior and distribution of riffle insects in a laboratory stream. Aquatic Insects 2(3):161-169.

McCullough, D.A., G.W. Minshall, and C.E. Cushing. 1979. Bioenergetics of lotic filter-feeding insects *Simulium* spp. (Diptera) and *Hydropsyche occidentalis* (Trichoptera) and their function in controlling organic transport in streams. Ecology 60(3):585-596.

McNeil, W.J., and W.H. Ahnell. 1964. Success of pink salmon spawning relative to size of spawning bed materials. United States Fish and Wildlife Service Special Scientific Report--Fisheries No. 469.

Newcombe, C.P. 2003. Impact assessment model for clear water fishes exposed to excessively cloudy water. Journal of the American Water Resources Association 39:529-544.

Platts, W.S., M.A. Shirazi, and D.H. Lewis. 1979. Sediment particle sizes used by salmon for spawning with methods for evaluation. U.S. Environmental Protection Agency, EPA Report EPA-600/3-79-043.

Reiser, D.W., and T.A. Wesche. 1977. Determination of physical and hydraulic preferences of brown and brook trout in the selection of spawning locations. Wyoming Water Resources Research Institute, Water Resources Series 64, Laramie, Wyoming.

Reiser, D.W., and R.G. White. 1988. Effects of two sediment size-classes on survival of steelhead and chinook salmon eggs. North American Journal of Fisheries Management 8:432-437.

Relyea, C.D., G.W. Minshall, and R.J. Danehy. 2000. Stream insects as bioindicators of fine sediments. *In:* Proceedings Watershed 2000, Water Environment Specialty Conference. Vancouver, British Columbia, Canada.

Richards, C. and K.L. Bacon. 1994. Influence of fine sediment on macroinvertebrate colonization of surface and hyporheic stream substrates. Great Basin Naturalist 54(2):106-113.

Ryan, P.A. 1991. Environmental effects of sediment on New Zealand streams: a review. New Zealand Journal of Marine and Freshwater Research 25:207-221.

Snyder, D.E. 1983. Fish eggs and larvae. In L. A. Nielsen and D. L. Johnson, editors. Fisheries Techniques. American Fisheries Society, Bethesda, Maryland.

Strand, R.M. and R.W. Merritt. 1997. Effects of episodic sedimentation on the net-spinning caddisflies *Hydropsyche betteni* and *Ceratopsyche sparna* (Trichoptera: Hydropsychidae). Environmental Pollution 98(1):129-134.

Sumner, F.H., and O.R. Smith. 1940. Hydraulic mining and debris dams in relation to fish life in the American and Yuba rivers of California. California Fish and Game 26:2-22.

Swanston, D.N. 1991. Natural processes. In: W. R. Meehan (editor), Influences of Forest and Rangeland Management on Salmonid Fishes and Their Habitats. American Fisheries Society Special Publication 19: 139-179.

Tappel, P.D., and T.C. Bjornn. 1983. A new method of relating size of spawning gravel to salmonid embryo survival. North American Journal of Fisheries Management 3:123-135.

Vuori, K.M. and I. Joensuu. 1996. Impact of forest drainage on the macroinvertebrates of a small boreal headwater stream: do buffer strips protect lotic biodiversity? Biological Conservation 77:87-95.

Walkotten, W.J. 1973. A freezing technique for sampling streambed gravel. USDA Forest Service Research Note PNW-205.

Waters, T.F. 1995. Sediment in Streams: Sources, Biological Effects, and Control. American Fisheries Society Monograph 7, Bethesda, Maryland.

Young, M.K., and W.A. Hubert. 1989. Substrate alteration by spawning brook trout in a southeastern Wyoming stream. Transactions of the American Fisheries Society 118:379-385.

Young, M.K., W.A. Hubert, and T. A. Wesche. 1990. Fines in redds of large salmonids. Transactions of the American Fisheries Society 119:156-162.

Young, M.K., W.A. Hubert, and T.A. Wesche. 1991. Selection of measures of substrate composition to estimate survival to emergence of salmonids and to detect changes in stream substrates. North American Journal of Fisheries Management 11:339-346.

Zweig, L.D. 2000. Effects of deposited sediment on stream benthic macroinvertebrate communities. Masters Thesis, University of Missouri-Columbia, 164pp.