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Ecological Condition Assessments of California's Perennial Wadeable Streams (2000 through 2006)

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Ecological Condition Assessments of California's Perennial Wadeable Streams (2000 through 2006)

Report to the State Water Resources Control Board's
Non-Point Source Program

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These analyses were based on methodologies developed by the US EPA's Office of Research and Development (ORD) for its EMAP program. We are especially indebted to Tony Olsen, who developed the probabilistic sampling designs used for this assessment, and Tom Kincaid, who wrote the scripts for combining the five study designs and helped with interpretation of the output. Development of the California RIVPACS predictive model by Chuck Hawkins of Utah State University was funded in part by the Pacific Southwest Region of the USDA Forest Service (Cooperative Agreement #PSW-88-0011CA, Cost Reimbursable Agreement # 03-CR-11052007-100, and Contract # 53-9A63-00-1T52). Raphael Mazor (SCCWRP) helped with conversion of invertebrate taxa IDs to RIVPACS OTUs. Will Patterson (DFG-ITB) and the CSU Chico Geographic Information Center provided helpful assistance with GIS analytical tools.

The project owes much of its success to the efforts of staff at DFG's Aquatic Bioassessment Laboratory (ABL): James Harrington managed the ABL's collaboration with the EMAP and CMAP programs. Andy Rehn participated in CMAP project design. ABL field staff, led by Shawn McBride with assistance from Michael Dawson and Jennifer York, were responsible for the extensive field reconnaissance and data collection efforts, ABL taxonomic staff (Dan Pickard, Doug Post, Brady Richards and Joe Slusark) were responsible for all taxonomic data, and Glenn Sibbald contributed watershed delineations for the RIVPACS and GIS analyses.

INTRODUCTION

The struggle to adequately monitor the condition of waterbodies with limited financial resources is a challenge faced by water resource agencies worldwide. For a resource agency to adequately meet its obligations to monitor and assess the condition of its waterbodies it must provide information at both the site-specific scale and the scale of the entire resource. This dual obligation requires different monitoring strategies.

Traditional targeted monitoring (in which site locations are selected to meet specific monitoring goals) is essential for answering many key water quality monitoring questions (e.g., what is the condition of specific sites, how do site conditions vary seasonally and annually, where are the best and worst sites, what are the primary water quality problems at specific sites, what are the effects of specific watershed activities and/or BMPs). Although this approach can generate much valuable water quality data, it is unable to provide information about the overall condition of large populations of resources (e.g., all streams, lakes or wetlands in a state) unless the monitoring entity is prepared to perform a complete census of the targeted resource. Site specific monitoring approaches are also unable to provide an objective context for interpreting the data they generate. However, the perspective provided from this context is a necessary logical foundation for a sound monitoring program.

In the US, the need for a broader context for interpreting the results of targeted monitoring data has driven the development of alternate sampling designs at both the state and federal level. To meet this need, the U.S. EPA established its Environmental Monitoring and Assessment Program (EMAP), a long term research program designed to develop the tools and techniques needed for cost-effectively answering the fundamental status and trends questions in the Clean Water Act. The EMAP studies are based on a probabilistic survey design in which each sampling location represents a known proportion of the total resource of interest (e.g., percent of total stream length) with known statistical confidence. This design permits the inference of resource conditions for large geographic regions with a relatively small investment in sampling (Ringold et al. 1996, Olsen et al. 1999, Stevens and Olsen 2004). After completing assessments of the condition of Eastern lakes and the condition of wadeable streams in the Middle-Atlantic states, the EPA initiated a similar assessment of streams in the western states (WEMAP), which included a high density of sites in California.

For the first time in 2006, the state of California used data from a probability survey to derive the condition assessments of its perennial streams in its 305(b) report to the US EPA (Ode and Rehn 2005, California State Water Resources Control Board 2006). These reports were developed to meet California's obligation to monitor its compliance with the goal of biotic integrity stipulated under Clean Water Act §305(b). Prior to these report, the ABL has presented results of the Southern Coastal California intensification area (Rehn and Ode 2004), Northern Coastal California intensification area (Rehn et al. 2005) as demonstration projects.

As the EPA's Western EMAP sampling effort ended in 2003, the EPA strongly encouraged western states to continue the EMAP monitoring approach in their own state monitoring programs. Two monitoring programs in the California State Water Resources Control Board, the recently established Surface Water Ambient Monitoring Program (SWAMP) and the Non-Point Source Program (NPS) expressed interest in probabilistic sampling. In concert with (and with funding from) their counterparts in the EPA Region 9 Non-point Source Programs and Office of Water, NPS and SWAMP developed the California Monitoring and Assessment Program (CMAP) to provide an additional four years of statewide probabilistic sampling to support continued ecological condition assessments. The study designs used for these condition assessments were developed by the EPA's Office of Research and Development. Since the state and federal NPS programs brought a strong interest in the relationship between non-point source pollution sources and aquatic life use, the sampling design was adapted to include an explicit stratification by landuse/ landcover. The CMAP project goals are to extend the benefits of the EMAP sampling effort to enable long term trend monitoring and condition assessments both at the statewide scale and for three several major NPS classes: agricultural lands, urban lands and forested lands.

This report presents the combined results of six years of probabilistic sampling in California, four years of EMAP (2000-2003) and the first two years of CMAP (2004, 2005). The results presented here are focused on the condition assessments, but we include some initial assessments by NPS categories when the sample sizes are sufficient. A full report of ecological condition assessments for each of the NPS categories will follow the completion of the final CMAP sampling (2008).

METHODS

Study Design/ Site Selection

This report combines the results from two large surveys (Figure 1, Figure 2, Table 1), the EMAP West study (2000 to 2003) and the first two years of the CMAP study (2004-2005). All surveys were based on a generalized random tessellation stratified (GRTS) design, which uses a reverse hierarchical ordering scheme to generate a relatively even distribution of sites throughout the study area (Stevens and Olsen 2004).

EMAP ~ There was no stratification in the EMAP West design, but site selection weights were adjusted so that Strahler stream order categories (1^{st}, 2^{nd}, 3^{rd}, and 4^{th}+) were sampled in approximately equal proportions throughout the state. We combined 4 separate survey designs for this analysis (see Ode and Rehn 2005 for more detail). Three of these were modifications of the main WEMAP sample frame: 1) the California statewide sites that were part of the larger WEMAP design, 2) the southern coastal California special interest sites, and 3) the northern coastal California special interest sites. A separate GRTS survey was created in 2003 to increase the representation of sites in the central coast region. In each of the designs, the EPA's RF3 hydrology layer was

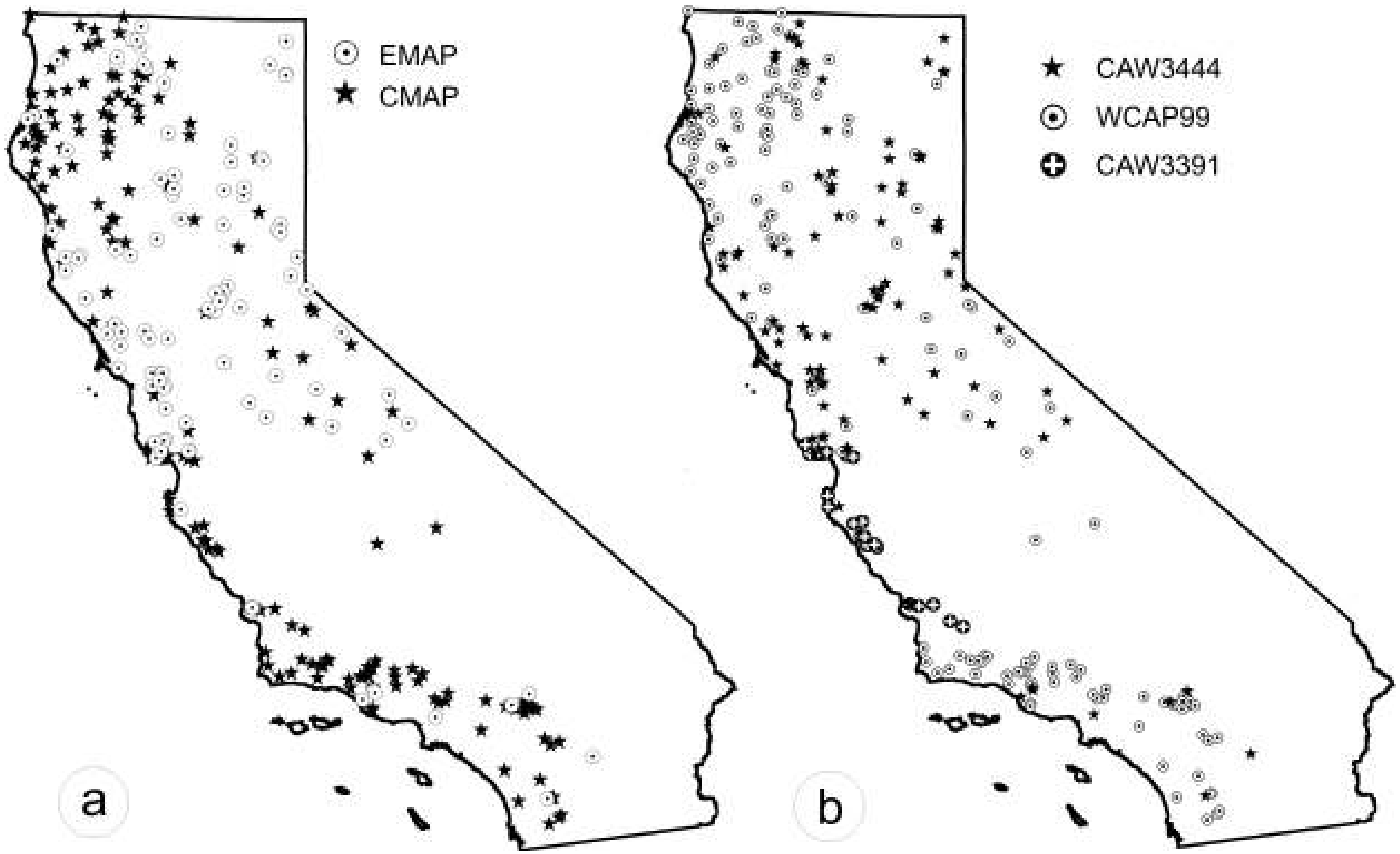

Figure 1. Maps showing the distribution of sites sampled under the EMAP and CMAP programs between 2000 and 2006 coded a) by sampling program and b) by the three major sampling designs.

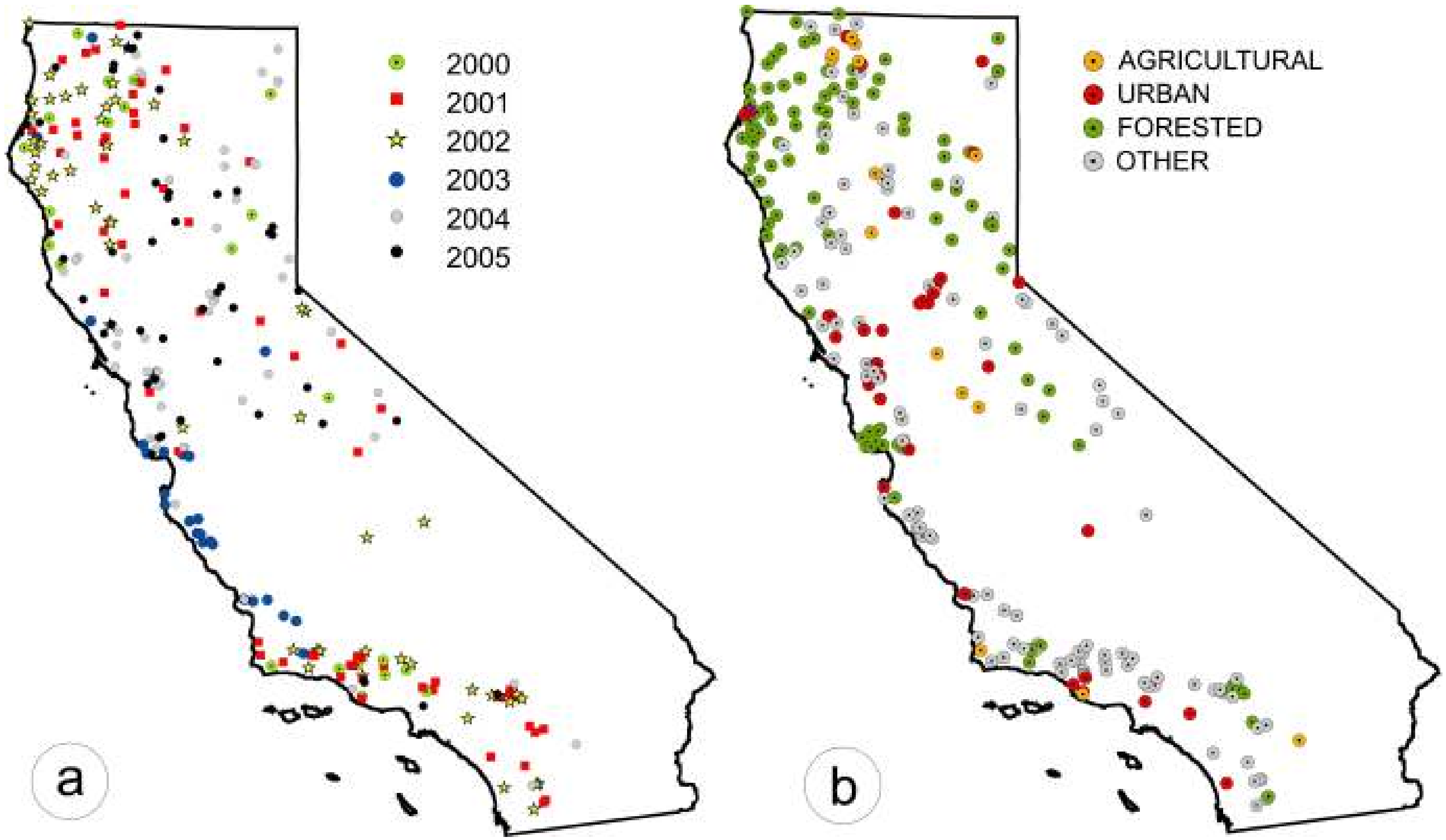

Figure 2. Distribution of sites used in condition assessments coded by a) year sampled or b) landuse designation of the site.

Table 1. Number of sites sampled in each year under the EMAP and CMAP programs.

Project	Sampling Year	Number of Sites	Comments
EMAP	2000	24	First year of project - only had ~half the sampling effort of a normal years
EMAP	2001	61	Normal sampling year
EMAP	2002	55	Normal sampling year
EMAP	2003	26	• All but 3 sites in Central Coast supplemental project area • Plus ~25 targeted reference sites also sampled for EMAP, but these don't contribute to condition assessments
CMAP	2004	51	Normal sampling year
CMAP	2005	51	Normal sampling year

used as the sample frame, excluding modified channels and canals when these classes were coded in the RF3. A list of potential sampling locations was generated randomly from the RF3 hydrology layer as described by Stevens and Olsen (2004). For analyses, each potential sampling site was assigned a weighting factor proportional to the number of stream kilometers it represented.

CMAP ~ The CMAP design was based closely on the original EMAP design, but was modified to enable stream condition assessments based on landuse categories (agricultural, urban, forested and other). The EPA's RF3 hydrology layer was again used as the sample frame, excluding modified channels and canals when this information was coded in the RF3. Note that, in years 2006 and 2007 of the CMAP program, we added a supplemental set of sites using a sample frame consisting of modified channels eliminated in the previous surveys (these results will be reported in subsequent reports).

We used the California Department of Forestry and Fire Protection's (CDF) composite dataset of California landcover to make a preliminary assignment of landuse class to each site in the sample draw (Multi-source Land Cover Data (MMLCD v02_1), http://frap.cdf.ca.gov/data/frapgisdata/select.asp). Based on a mosaic of different landuse datasets primarily from the late 1990s and early 2000s, the CDF data provided the most current and complete coverage of landcover in California at the time of the CMAP sample draw. We chose to not use a popular alternative landcover dataset (the USGS/EPA's National Landcover Data (NLCD 1992)) out of concern that the data were outdated (based on 1992 Landsat imagery). We assigned all the MMLCD categories to one of four landcover classes (agricultural lands, urban lands, forested lands or other) and then reprocessed the original 100 m resolution CDF grids to a lower resolution (300 m) grid needed for the sample draw. Analysts at the EPA's ORD used this grid to assign a preliminary landcover class to each of the sites in the sample draw (based simply on the value of the landcover pixel at the site coordinates) and delivered the list of potential sampling sites to the ABL field crew.

<u>Site evaluation</u>

Once the list of potential sampling coordinates was generated for each region, we conducted a multi-phase process to screen sites meeting the definition of the target population (perennial, natural channels). We first conducted an initial screen of the site list to eliminate sites that were obviously not part of the targeted population (channelized streams, non-perennial streams, etc). Field crews then split up the remaining sites by county and visited county Tax Assessor's offices to identify land owners for each sites. For sites that fell on public lands, we contacted officials to obtain permission to sample and obtain sampling permits where necessary. For sites on private land we contacted owners by letter requesting permission to visit the site. When access permission was granted, field crews performed on-site reconnaissance to identify sites that were part of the target population.

There are many reasons why potential sites were rejected during the reconnaissance phase. In the arid southwest, many lines that are coded as perennial streams on USGS quadrant maps (and the 1:100,000 RF3 stream layer digitized from them) are, in fact, not perennial. Earlier analyses indicated that approximately 50% of stream length indicated as perennial in the southern coastal region was actually non-perennial (Rehn and Ode 2004). Underground pipelines, canals and aqueducts frequently can not be distinguished from streams on the RF3 stream layer, and these also were rejected as non-target during reconnaissance. Also, some perennial sites were inaccessible due to physical barriers (e.g., access was too dangerous or required excessive backpacking). Private ownership further confounded site selection. When landowners denied access to a site, it was impossible to determine its target status, and it was categorized as "status unknown".

EMAP ~ Sites meeting the target criteria were selected for sampling in the order they appeared on the original list to assure random site selection. Site reconnaissance continued until a pool of approximately 60 target sites each was identified and sampled from the northern coast, the southern coast and statewide and 30 sites were sampled from the central coast region. During the reconnaissance process, we evaluated 1140 sites, keeping careful records of each site's target status, and if applicable, reasons why sites were eliminated from the target pool for use in later analyses. We sampled over 200 study reaches throughout California between April and September of 2000 through 2003, sampling southern sites at the beginning of the sampling season and progressing north later in the year.

CMAP ~ Site reconnaissance was identical to that for the EMAP study, except that we added an additional step to help balance the number of sampled sites in the four landcover classes. Sites were selected in the order they appeared on the original random list, but once the goal was reached for each class (e.g., 13 sites of each landcover class per year) all subsequent sites belonging to that class were skipped.

Field Methods

Once target sites were identified and sampling permission obtained, we sampled sites according to standard EMAP West field methods (Peck et al. 2004). A sampling reach was defined as 40 times the average stream width at the center of the reach, with a minimum reach length of 150m. We collected two BMI samples from each reach: 1) a reachwide composite sample (RWB) consisting of 11 one ft^2 samples taken from equally spaced locations throughout the reach and 2) a targeted riffle sample (TRB) consisting of 8 one ft^2 samples taken from fast water habitat units within the reach (Hawkins et al. 2001). Fish and algae samples were collected according to Peck and others (2004) but are not reported here. Water chemistry samples were collected from the mid-point of each reach and analyzed using WEMAP protocols (Klemm and Lazorchak 1994). Field crews recorded physical habitat data using EPA qualitative methods (Barbour et al. 1999) and quantitative methods (Kaufmann et al. 1999).

Lab Methods

All BMI samples were processed at DFG's Aquatic Bioassessment Laboratory in Chico, CA. A 500 organism random subsample was taken from each BMI sample and identified according to WEMAP standard taxonomic effort levels (CSBP II, www.dfg.ca.gov/cabw/camlnetste.pdf). All taxonomic data were entered into an MS Access database (CalEDAS) that allowed us to produce standardized taxa lists at different standard effort levels. Five percent of taxa were re-identified for quality assurance and archived vials of all samples are housed at the Chico facility.

Calculating Biological Condition Scores

We calculated biological condition scores for all sites using recently developed predictive models based on the River Invertebrate Prediction and Classification System (RIVPACS, Wright 1984). Like multimetric approaches (Kerans and Karr 1994, Ode *et al.* 2005, Rehn and Ode 2005), predictive modeling techniques establish thresholds of ecological impairment based on a characterization of the biotic assemblages expected to occur under minimal human disturbance (Wright *et al.* 1984, 1989, 2000). However, predictive models compare assemblages at test sites to an expected taxonomic composition rather than expected metric values. Taxon-based models have seen widespread use since the first BMI models were created in Great Britain in the late 1970s (Norris and Georges 1993, Hawkins *et al.* 2000, Van Sickle *et al.* 2005) and have been promoted in the US (Hawkins *et al.* 2000, Hawkins and Carlisle 2001) as an alternative to the multimetric approach initially endorsed by the EPA (Barbour *et al.* 1999). For this analysis, we employed newly developed California RIVPACS models (C. Hawkins unpublished) that can be used to score sites throughout the state.

The goal of RIVPACS is to compare the list of taxa observed at a site (O) to the list of taxa predicted to occur at a given site in the absence of human disturbance (E). The approach has four components: 1) reference sites are classified according to degree of taxonomic similarity, 2) environmental variables associated with each class are identified, 3) discriminant functions analysis (DFA) is used to predict class membership of new test sites based on the values of their environmental predictor variables, 4) the observed list of taxa is compared to the expected list to calculate the O/E ratio.

The most recently derived RIVPACS models for California streams were completed in June 2005 (Hawkins unpublished presentation). Preliminary attempts to create one model for California resulted in relatively imprecise models, but an initial classification step using precipitation and temperature variables produced 3 separate sub-models with better performance. To apply the new RIVPACS models to our WEMAP data, we prepared separate files of taxa and predictor variables for each of the 3 sub-models.

Benthic Invertebrate Taxonomic Data ~ Taxonomic lists generated from CalEDAS were modified for compatibility with the formats used in the RIVPACS models by: 1) eliminating ambiguous taxa, 2) using a rarefaction subroutine to subsample 300 organism counts from the original 500 count samples, 3) converting the final taxonomic names to the operational taxonomic names (OTUs) used in the models (converting chironomid midges to subfamily), and 4) cross-tabulating the taxonomic list into a taxon by site matrix. Steps 2 and 4 were performed with software developed by Dave Roberts ("subsample.exe" and "matrify.exe" available through the Western Center for Monitoring and Assessment of Freshwater Ecosystems).

Habitat Variables ~We determined the values of six map-based predictor variables for each site: 1 and 2) geographic coordinates (latitude and longitude) were obtained from the original study design file, 3) watershed area was calculated by delineating upstream watershed boundaries for each site in using automated GIS scripts and manual delineation where necessary, 4) log mean "normal" precipitation was estimated by overlaying sites on a GIS grid of mean monthly precipitation (1961-1990) obtained from the Oregon Climate Center (OCC, www.ocs.orst.edu/prism), 5) mean "normal" temperature was estimated from mean monthly temperature grids (1961-1990) also obtained from the OCC, 6) percent sedimentary geology was estimated from an unpublished GIS geology classification of the western United States derived by John Olson, (Utah State University) from a generalized geologic map of the coterminous US (Reed and Bush, pubs.usgs.gov/atlas/geologic/).

Once predictor variables were determined for each site, we used precipitation and temperature data to assign each site to one of the three classes based on the following criteria. Sites with mean monthly temperatures (Tmean) less than 9.3ºC were assigned to Class 3, sites with temperatures greater than 9.3ºC were assigned to Class 2 if they had log mean monthly precipitation values (logPPT) less than 2.952, and to Class 1 if logPPT was greater than 2.952. The three sub-models required different sets of predictor variables: Class 1 used latitude, log watershed area, and mean temperature; Class 2 used longitude, percent sedimentary geology and mean precipitation; Class 3 used log watershed area and mean temperature.

The three sets of site files were uploaded to the web interface containing the California models at the Western Center for Monitoring and Assessment of Freshwater Ecosystems (http://129.123.10.240/WMCPortal/DesktopDefault.aspx?tabindex=2&tabid=27). The model output included the probability matrix, O/E scores, and taxon sensitivity scores.

We calculated O/E scores for all sites (and both TRB and RWB samples, where available) using versions of the RIVPACS models in which chironomid midges (Diptera: Chironomidae) were reported at the subfamily level (= OTU2). Unless otherwise specified, we report O/E ratios are based on the O/E comparisons that include only common taxa ($p > 0.5$) since these tend to be more sTable (Hawkins, personal communication) than ones that include all taxa ($p >0.0$).

We based our analyses on scores calculated from the targeted riffle samples (since the models were based on TRB data), but we used RWB samples in the handful of cases where TRB data were unavailable or had low counts (<275 organisms after subsampling and elimination of ambiguous taxa).

Recalculation of Landuse Assignments

The original assignment of landuse categories to CMAP sites during the initial sample draw was used as a quick way to screen potential sites. However, since this preliminary assignment of sites was relatively coarse (based only on the landuse class present in a 300 m pixel overlapping the site), we went through a more intensive GIS process to assign sites to landuse classes based on landuse percentages in upstream drainages of each site.

We used the newly released national landcover dataset (NLCD 2001) for site assignments, converting the NLCD landuse codes (Table 2) to one of the four landuse categories according to the values in Table 3, applying these re-assignments to both EMAP and CMAP data. We calculated landuse percentages for the four categories at each of three spatial scales (Figure 3): 1) the entire upstream drainage, 2) a portion of the upstream watershed within 5km of the site (5k_buffer) and 3) a portion of the upstream watershed within 1 km of the site (1k_buffer). The upstream watershed boundaries were delineated for EMAP sites by manually clipping them from existing CalWater V2.2 shapefile boundaries. Boundaries for CMAP sites were delineated using automated scripts developed by the CSU Chico Geographic Information Center. These scripts used 30m DEM data to infer watershed boundaries and generally produced very accurate delineations. However, the automated scripts had trouble delineating watersheds for a handful of very low gradient sites and these were finished manually as for EMAP sites. Creation of the local watershed clips was performed with automated scripts developed by Will Patterson (DFG's BDB).

Table 2. Coding scheme for the 2001 NLCD dataset. Landcover assignments are listed in the column "LC Class".

NLCD 2001 Landcover Coding Scheme			
Code	**Definition**	**LC Class**	**ATtILA Custom Codes**
11	Open Water	Not assigned	Water/ No Data
12	Perennial Ice/ Snow	Not assigned	Water/ No Data
21	Developed, Open Space (e.g., lawns, parks, roadside vegetation)	Code 21 (assigned conditionally)	Urban/ Recreational Grasses
22	Developed, Low Intensity	URB	Low Density Residential
23	Developed, Medium Intensity	URB	High Density Residential
24	Developed, High Intensity	URB	Commercial/ Industrial
31	Barren Land	OTHER	Natural Barren
41	Deciduous Forest	FOR	Forest
42	Evergreen Forest	FOR	Forest
43	Mixed Forest	FOR	Forest
52	Shrub/ Scrub	OTHER	Shrublands
71	Grasslands/ Herbaceous	OTHER	Natural Grasslands
81	Pasture/ Hay	AG_P	Pasture
82	Cultivated Crops	AG_C	Row Crops
90	Woody Wetlands	OTHER	Wetlands
95	Emergent Herbaceous Wetlands	OTHER	Wetlands
No Data	No Data	Not Assigned	

Once watersheds and local clip files were created, we used the ArcView 3.x extension ATtILA (Ebert and Wade 2004) to calculate landuse percentages for each of the four landuse/ landcover categories at each of the three spatial scales. All sites were then assigned to one of the landuse categories using the following decision criteria: 1) if a site had greater than 25% urban landuse at any of the three spatial scales it was assigned to the "urban" landuse class, 2) if a site had greater than 50% agricultural landuse at any of the three spatial scales it was assigned to the "agriculture" landuse class, 3) if a site had greater than 75% forested landcover at any of the three spatial scales it was assigned to the "forested" landcover class, 4) sites that did not meet any of these criteria were assigned to the "other" category. In the few cases where sites met more than one of the criteria, sites were assigned to multiple categories (Table 4). ***Note: The landuse/landcover thresholds used to assign sites to the different categories were intentionally set using fairly high values to ensure that sites assigned to these categories were associated with these landuses/landcover classes. However, these data can be easily re-evaluated with different thresholds. In future reports, we will explore the relationship between these thresholds and condition estimates.***

Table 3. Stressor thresholds used for calculating stressor extent and relative risk estimates. Thresholds in bold were those used for the EPA's western EMAP condition assessments, while thresholds in italics were assigned by the author as described in text.

Stressor Class	Stressor Name	Xeric Ecoregions		Mountain Ecoregions	
		Most Disturbed	Least Disturbed	Most Disturbed	Least Disturbed
Water Chemistry	*Chloride (CL)*	*> 245 μeq/L*	*< 100 μeq/L*	*> 245 μeq/L*	*< 100 μeq/L*
	Total Nitrogen (TN)	**> 600 μg/L**	**≤ 200 μg/L**	**> 200 μg/L**	**≤ 125 μg/L**
	Total Phosphorus (TP)	**> 175 μg/L**	**≤ 40 μg/L**	**> 40 μg/L**	**≤ 10 μg/L**
	Specific Conductance (COND)	**>1000 μS/cm**	**≤ 500 μS/cm**	**> 1000 μS/cm**	**≤ 500 μS/cm**
	Total Suspended Solids (TSS)	*> 50 mg/L*	*≤ 15 mg/L*	*> 50 mg/L*	*≤ 15 mg/L*
	Turbidity (TURB)	*> 20 NTU*	*≤ 5 NTU*	*> 20 NTU*	*≤ 5 NTU*
Landuse	*Unnatural Index (U_INDEX = AG+URB ws, 5k, 1k)*	*> 40%*	*< 10%*	*> 40%*	*< 10%*
	Percent Urban (URB ws, 5k,1k)	*> 25%*	*< 5%*	*> 25%*	*< 5%*
	Percent Agricultural (AG ws, 5k, 1k)	*> 50%*	*< 10%*	*> 50%*	*< 10%*
Biological Indicator	*RIVPACS O/E Score (NoChiros, P05)*	*< 0.55*	*> 0.77*	*<0.55*	*> 0.77*
Physical Habitat	Streambed Stability (LRBS_BW5)	**< -1.7 or > 0.3**	**≥ -0.9 and ≤ -0.1**	**< -1.3 or > 0.6**	**≥ -0.7 and ≤ 0.1**
	Percent Sand and Fines (PCT_SAFN)	*> 25%*	*<10%*	*> 25%*	*<10%*
	Riparian Disturbance (W1_HALL)	**> 0.9**	**≤ 0.7**	**> 0.95**	**≤ 0.35**
	Riparian Vegetation (XCMGW)	**< 0.132**	**≥ 0.270**	**< 0.23**	**≥ 0.67**
	Habitat Complexity (XFC_NAT)	**< 0.32**	**≥ 0.60**	**< 0.14**	**≥ 0.33**

Table 4. Breakdown of site assignments to four landcover classes (urban, forested, agricultural, and other) for both EMAP and CMAP projects based on NLCD 2001 data. NLCD landcover code # 21 (developed, open space) was assigned to either the urban or forested landcover class based on watershed scale landcover thresholds (i.e., if forested > 60 %, if urban>20 %). Table includes values for all 278 scored sites (including 13 sites sampled only with the RWB collection method that were not included in 2005 report); site totals are greater than 278 in case where sites meet classification criteria for more than one landcover class.

Watershed Scale	U_ws	F_ws	A_ws	O_ws	Total				
Total	10	74	3	191	278				
EMAP	4	51	1	120	176				
CMAP	6	23	2	71	102				
Local 5k Buffer	U_5k	F_5k	A_5k	O_5k	Total				
Total	15	78	12	174	279				
EMAP	5	57	4	111	177				
CMAP	10	21	8	63	102				
Local 1k Buffer	U_1k	F_1k	A_1k	O_1k	Total				
Total	29	73	15	161	278				
EMAP	8	57	4	107	176				
CMAP	21	16	11	54	102				
Any Scale (using 25/50/75 cutoffs)	U_all	F_all	A_all	O_all	Urban + Forested	Urban + Agricultural	Forested + Agricultural	Urban + Forested + Agricultural	Total
Total	34	96	17	134	1	1	1	0	283
EMAP	12	71	4	90	0	1	0	0	178
CMAP	22	25	13	44	1	0	1	0	105
Any Scale (using 20/40/70 cutoffs)	U_all	F_all	A_all	O_all	Urban + Forested	Urban + Agricultural	Forested + Agricultural	Urban + Forested + Agricultural	Total
Total	41	108	17	116	1	2	1	0	285
EMAP	14	77	4	83	0	2	0	0	180
CMAP	27	31	13	33	1	0	1	0	105

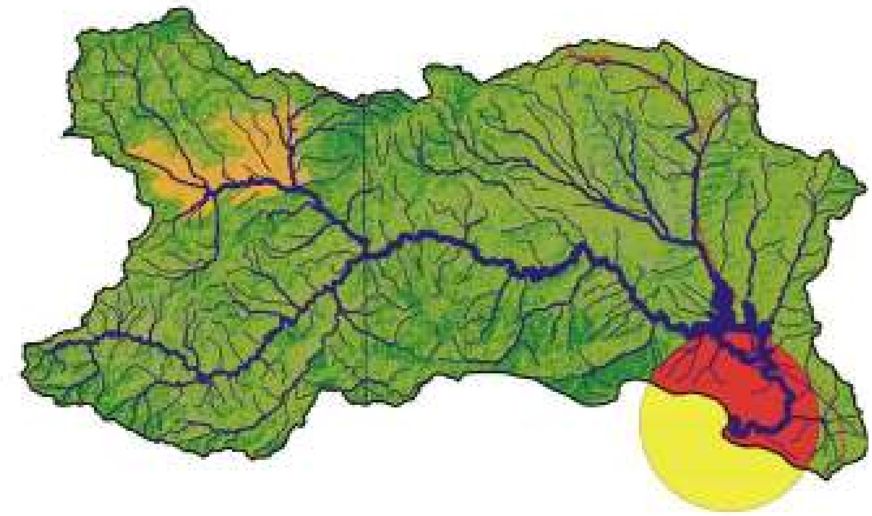

Figure 3. Example watershed showing areas used for assigning landuse categories. The red area indicates the intersection of the watershed polygon and a circular buffer around the sampling location (i.e., the region summarized by local clips).

Chemistry Data

All chemical analyses for the EMAP dataset, except those measurements collected *in situ* were performed by laboratories managed by the EMAP program. Field samples were shipped to EMAP directly. All chemical analyses for CMAP were performed by the DFG Water Pollution Control Laboratory in Rancho Cordova (WPCL) following the same methods used by EMAP. Where necessary, we converted analytical units used in CMAP to those used in EMAP for all combined assessments.

Methods Comparison

Since we collected two BMI samples from most sites (TRC and RWB), we were able to compare the performance of the two methods side by side. Comparisons from the EMAP dataset (2000 through 2003) are reported elsewhere (Rehn et al. 2007), and these comparisons based on 102 sites sampled under the CMAP program (2004 and 2005) generally follow the approach used in that paper. As in the EMAP comparisons, we graphed the difference between site scores generated by the two samples as a function of various local and watershed scale variables (both natural and anthropogenic) to evaluate whether there was any evidence of bias between the methods with respect to these gradients.

Probability Survey Assessments

Because all the sites sampled in these studies were selected probabilistically, we know the proportion of total stream length that each site represents and the amount of error in that estimated length. This relationship serves as the basis for a set of products generated by this kind of probability survey: 1) population estimates based on the reconnaissance data, 2) condition estimates (and their underlying cumulative distribution frequency plots) of the target population, 3) stressor extent estimates of the percent stream length with stressor values greater than set thresholds, and 4) relative risk estimates of the increased risk of biological impairment associated with stressor levels in exceedence of the thresholds used in the stressor extent estimates. We produced these products for several different temporal ranges: 1) annual estimates, 2) four-year rolling averages (2000-2003, 2001-2004, 2002-2005), and 3) six-year combined estimates. In some cases, we made slight adjustments to these as the data necessitated; these adjustments are noted in the appropriate sections. Where data were sufficient to permit it, we also produced combined estimates for the major products for each of the four landcover/landuse classes defined in the CMAP program.

All probabilistic survey analyses were derived using the script "psurvey.analysis" developed in the R programming language (Version 2.4.1, www.r-project.org) by the EPA's Office of Research and Development in Corvallis, Oregon (see EPA's ORD website, http://www.epa.gov/nheerl/arm/analysispages/monitanalysisinfo.htm, for more detailed discussion). The analysis package was used to combine the five design models and adjust site weights to reflect their percent contribution to the target population. The presence of the CMAP landuse stratification element required us to assign all EMAP sites (including those in the reconnaissance set) to one of the four CMAP landuse classes, greatly complicating the weight adjustment step.

Population Estimates

We used the probability relationships to estimate the total stream length in the following categories (these population estimates are based on the reconnaissance data): the sampled target population (all perennial wadeable streams, TS + TNS), the non-target population (NT), stream length not sampled due to denial of landowner access (LD), stream length not sampled due to the presence of physical barriers to sampling (PB). Since several CMAP sites were evaluated over the course of two years, we were unable to generate annual population estimates for all years, but instead report annual estimates for the first four years and a combined estimate for 2004 and 2005, two four-year rolling averages (2000-2003, 2002-2005) and the six-year averages.

Condition Assessments

Adjusted sites weights were used in conjunction with the RIVPACS scores calculated for each sampled target site to estimate the percentage of stream miles in three ecological condition categories: "Non-Impaired", "Impaired" and "Very Impaired". We used thresholds of 1.5 and 3 standard deviations below an O/E score of 1.0 (the score expected under no impairment) to set the boundaries between Non-Impaired and Impaired (O/E

<0.77), and Impaired and Very Impaired (O/E <0.55), respectively. Although we could have used separate thresholds for each of the three models based on their respective standard deviations, we used the average standard deviation for the three sub-models (0.15) because they were nearly identical (ranging between 0.14 and 0.16). ***Note: The term "impaired" is used throughout this report to refer to biological conditions outside those expected under reference conditions and has no regulatory meaning.***

Assignment of Stressor Thresholds for Stressor Extent and Relative Risk Estimates
Both stressor extent and relative risk estimates depend on the assignment of thresholds for each stressor that represent high levels of the variable. The stressor thresholds used in this report are identical those used in the EMAP West analyses (Stoddard et al. 2005) where the stressors overlap. Landuse thresholds were the same as those used for landuse assignments and the remaining stressors (Cl^-, TSS, TURB and PSAFN) were assigned by the author based on the distribution of stressor values in the combined dataset. Note that several of the EMAP thresholds vary with in the three major ecoregion groupings used in the analyses. All stressors and their thresholds are presented in Table 2. ***Note: Since chemical concentrations vary diurnally and seasonally, the stressor and extent estimates for these analytes should be interpreted with caution. Sampling was performed during index periods that were chosen to represent periods of relatively*** ***s***Table ***flows. Chemical concentrations are therefore likely to have higher peaks at some point in a year than we measured. Thus, our stressor extent estimates are likely underestimates.***

Stressor Extent
For our stressor extent and relative risk estimates, we evaluated 14 local and watershed scale attributes that had the potential to affect biological condition of the sampling sites. The attributes fell into three categories: 1) ambient water chemistry, 2) landuse, 3) local physical habitat (instream and riparian). Landuse variables were based on the three spatial scales used to assign sites to landuse/ landcover classes (watershed, 5k buffer, 1k buffer). We considered the influence of three landcover measures (% agricultural, % urban, % un-natural (AG + URB). Most of the 14 stressor variables can be directly or indirectly altered as a result of human activity and have been known to have harmful effects on stream biota (Stoddard et al. 2005). Physical habitat variables were selected to reflect a range of instream and riparian impacts likely to affect benthic macroinvertebrate condition (Kaufmann et al. 1999).

Relative Risk
Relative risk estimates were generated using the relative risk function provided in the psurvey.analysis R scripts (Van Sickle et al. 2006). The function calculates relative risk as the ratio of two ratios:

$$\frac{\dfrac{\text{Biologically Impaired Length ASSOCIATED with Stressor X}}{\text{Total Stream Length Impaired by Stressor X}}}{\dfrac{\text{Biologically Impaired Length NOT ASSOCIATED with Stressor X}}{\text{Total Stream Length Not Impaired by Stressor X}}}$$

RESULTS and DISCUSSION

The protection of the ecological condition of flowing waters is one of the highest priorities under the Clean Water Act and this objective is increasingly adopted as a primary foundation for monitoring programs at both state and federal levels. This refocused attention on the condition of aquatic life has been coupled with major advances in the science of landscape ecology (Allan and Johnson 1997, Allan 2004, Hansen et al. 2005, Burcher et al. 2007), which provides insight into the relationship between anthropogenic activities in watersheds and the condition of aquatic resources in those landscapes.

This recent surge of interest in applied stream ecology/ landscape ecology has produced a large body of studies that have summarized the landscape factors that control aquatic life use (ALU) condition (Roy et al. 2003a, Allan 2004, Brown and Veras 2005, Burcher and Benfield 2006, Booth et al. 2007), mechanisms by which they affect ALU (Townsend and Hildrew 1994, Roy et al. 2003b, Burcher et al. 2007) and spatial scales at which these variables act (Townsend et al. 2003, Feld and Herring 2007). A recent synthesis by Burcher and others (2007) argues that since natural and anthropogenic influences (e.g., agricultural or urban development, wildfires) occurring in the watershed do not directly affect biota but rather influence biota through a series of intermediate factors (e.g., changes in discharge, eutrophication, fine sediment deposition), protection of ecological condition requires an understanding of these intermediate pathways. This Landcover Cascade (LCC) provides a conceptual framework for organizing the relationships among the multitude of landscape factors affecting ALU in streams.

Probability surveys provide a powerful tool for monitoring programs committed to protecting ALU because they provide an objective means of identifying the relative strength of pathway elements in the LCC. Coupled with frameworks like the LCC, probability surveys provide an efficient mechanism for organizing monitoring data into information that can be used to prioritize protection and remediation efforts.

Probability Survey Assessments (Statewide)

The products of the probability surveys are reported in several different time groupings depending on the nature of the data: 1) annual estimates, 2) 4 year rolling averages (2000-2003, 2001-2004, 2002-2005), and 3) 6 year combined estimates. Results are available for all combinations, but are not always presented because low sample size or other factors prevented meaningful analyses of those groupings.

Population Estimates~ Results

Estimates of the percentage of stream length represented by different reconnaissance fates are presented in Table 5 and summarized for the 6-year averages in Figure 4. In all year groupings, the majority of stream length was non-target (NT). The total sample frame included approximately 405,000 km of streams, approximately 65% of which was

Table 5. Extent estimates indicating percent of total stream length and stream length estimates in the sample fram based on results of reconnaissance data for different combinations of survey years. Category codes: LD= landowner denial, NS= not sampled, target status unknown, NT= non target, PB= physical barrier, TNS= target, not sampled, TS= target sampled.

Subpopulation	Category	n	% Stream Length	Standard Error	Lower 95% Confidence Limit	Upper 95% Confidence Limit	Kilometers	Standard Error (km)	Lower 95% Confidence Limit (km)	Upper 95% Confidence Limit (km)
All_Years	LD	175	12.81	2.05	8.79	16.82	51870	7914	36359	67381
All_Years	NS	4	0.05	0.02	0.01	0.09	214	78	62	366
All_Years	NT	594	65.33	3.40	58.66	71.99	264566	33787	198346	330787
All_Years	PB	71	3.73	0.82	2.12	5.33	15086	3122	8967	21206
All_Years	TNS	89	6.28	1.10	4.11	8.44	25414	4053	17470	33357
All_Years	TS	294	11.81	1.64	8.60	15.03	47834	5672	36717	58952
All_Years	**Total**	**1227**	**100.00**	**0.00**	**100.00**	**100.00**	**404984**	**34918**	**336546**	**473423**
2000-2003	LD	97	9.51	2.01	5.56	13.46	38510	703	18719	63703
2000-2003	NS	4	0.15	0.06	0.04	0.27	626	20	143	1262
2000-2003	NT	348	62.93	3.27	56.52	69.34	254866	1142	190223	328284
2000-2003	PB	57	5.08	1.02	3.09	7.07	20568	355	10386	33479
2000-2003	TNS	30	7.43	2.08	3.36	11.50	30105	725	11316	54466
2000-2003	TS	192	14.89	2.41	10.16	19.62	60310	843	34189	92908
2000-2003	**Total**	**728**	**100.00**	**0.00**	**100.00**	**100.00**	**404984**	**0**	**336546**	**473423**
2002-2005	LD	129	13.13	2.45	8.32	17.94	53179	857	28008	84933
2002-2005	NS	4	0.06	0.02	0.02	0.11	262	9	57	532
2002-2005	NT	440	65.71	3.92	58.03	73.39	266118	1368	195308	347437
2002-2005	PB	48	3.63	0.93	1.82	5.44	14697	323	6109	25768
2002-2005	TNS	84	7.59	1.41	4.82	10.35	30722	492	16231	48996
2002-2005	TS	181	9.88	1.70	6.54	13.22	40005	595	22009	62572
2002-2005	**Total**	**886**	**100.00**	**0.00**	**100.00**	**100.00**	**404984**	**0**	**336546**	**473423**
2004-2005	LD	78	14.52	3.09	8.47	20.57	58795	1078	28502	97368
2004-2005	NT	246	66.57	4.92	56.92	76.22	269593	1719	191567	360825
2004-2005	PB	14	3.02	1.10	0.88	5.17	12245	383	2950	24479
2004-2005	TNS	59	5.67	1.27	3.19	8.16	22982	442	10750	38610
2004-2005	TS	102	10.21	2.24	5.82	14.61	41369	784	19573	69185
2004-2005	**Total**	**499**	**100.00**	**0.00**	**100.00**	**100.00**	**404984**	**0**	**336546**	**473423**
2000	LD	25	19.39	5.75	8.11	30.67	78520	2009	27299	145176
2000	NT	76	67.61	5.87	56.10	79.12	273801	2051	188795	374561
2000	PB	13	6.29	2.88	0.63	11.94	25455	1007	2132	56514
2000	TNS	2	0.31	0.25	0.00	0.79	1252	86	0	3753
2000	TS	40	6.41	1.30	3.87	8.95	25956	452	13028	42359
2000	**Total**	**156**	**100.00**	**0.00**	**100.00**	**100.00**	**404984**	**0**	**336546**	**473423**
2001	LD	21	4.74	1.32	2.15	7.32	19177	461	7224	34673
2001	NT	78	60.34	6.29	48.01	72.67	244362	2197	161563	344041
2001	PB	10	2.38	1.15	0.12	4.64	9628	403	398	21951
2001	TNS	3	0.63	0.30	0.04	1.23	2570	106	139	5813
2001	TS	73	31.91	6.23	19.70	44.13	129247	2176	66292	208923
2001	**Total**	**185**	**100.00**	**0.00**	**100.00**	**100.00**	**404984**	**0**	**336546**	**473423**
2002	LD	17	9.59	3.94	1.87	17.32	38855	1377	6279	82009
2002	NT	52	66.88	5.87	55.38	78.38	270853	2049	186367	371085
2002	PB	27	8.94	2.49	4.06	13.82	36213	869	13679	65422
2002	TNS	4	1.84	1.32	0.00	4.42	7436	461	0	20930
2002	TS	57	12.75	2.81	7.24	18.25	51627	981	24374	86417
2002	**Total**	**157**	**100.00**	**0.00**	**100.00**	**100.00**	**404984**	**0**	**336546**	**473423**
2003	LD	34	4.56	1.23	2.14	6.97	18464	430	7213	33021
2003	NS	4	0.74	0.28	0.19	1.29	2997	99	627	6126
2003	NT	142	56.38	6.82	43.01	69.75	228340	2382	144758	330221
2003	PB	7	2.86	1.58	0.00	5.95	11569	552	0	28181
2003	TNS	21	32.13	7.10	18.21	46.05	130128	2479	61300	218004
2003	TS	22	3.33	1.12	1.14	5.52	13487	390	3845	26124
2003	**Total**	**230**	**100.00**	**0.00**	**100.00**	**100.00**	**404984**	**0**	**336546**	**473423**

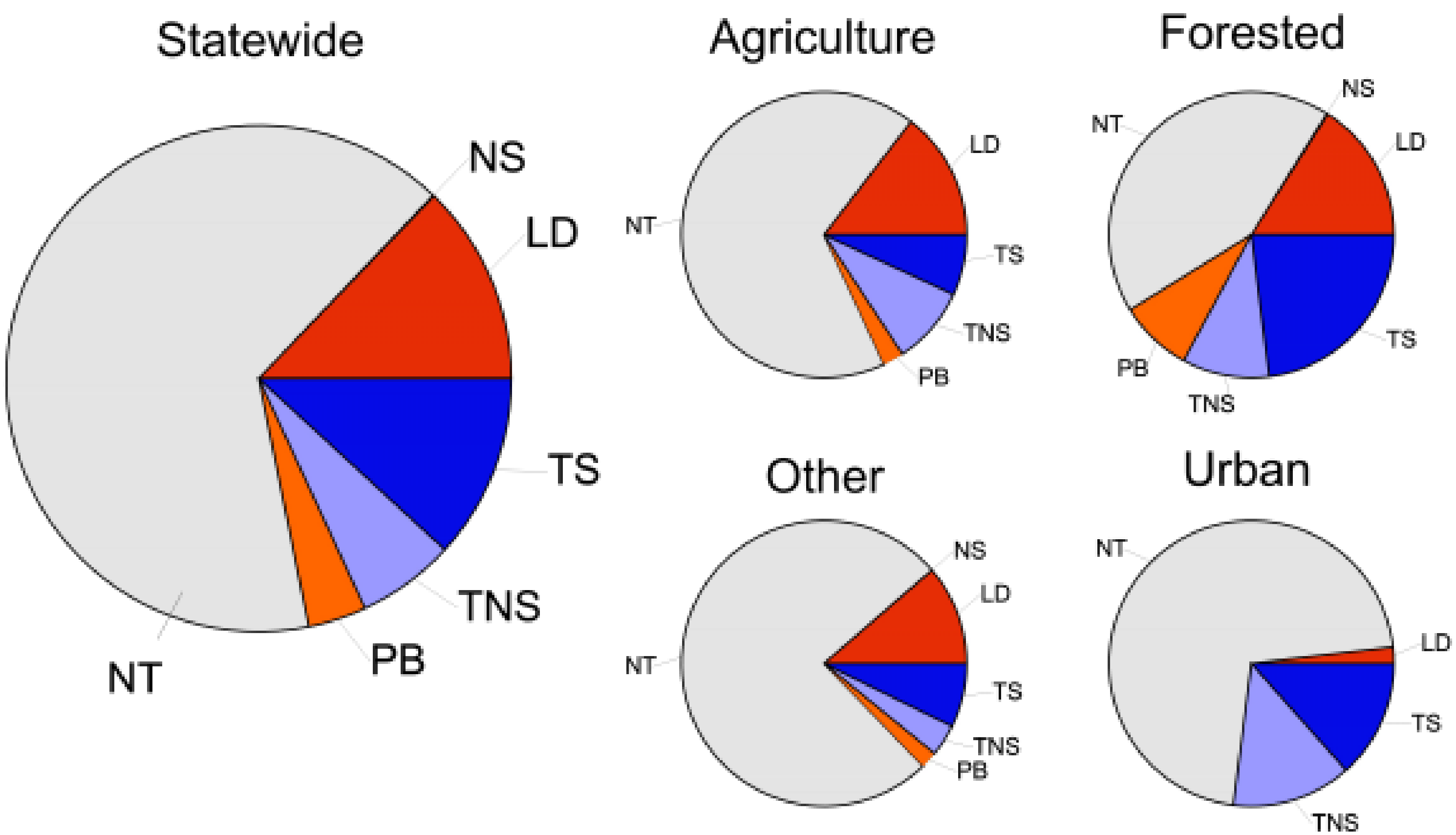

Figure 4. Estimates of the percentage of total stream length falling into one of six reconnaissance fate classes: LD= landowner denial, NS= not sampled, target status unknown, NT= non target, PB= physical barrier, TNS= target, not sampled, TS= target sampled. Results are shown for all sites and a separate estimates for each of the four landcover/ landuse classes. Percentage and total stream length estimates are listed in Table 6.

non-target. Most of this non-target stream length was comprised of dry channels or otherwise non-perennial streams, but some was comprised of pipelines or constructed channels that were erroneously indicated as natural stream channels in the NHD+ hydrology. The target population was estimated at approximately 18% of total stream length (~73,000 km). Two common fates of prospective sites in the reconnaissance effort (landowner denial, LD and permanent barriers, PB) represented approximately 17% of total stream length (approximately 67,000 km) that could not be assessed for target status.

Population Estimates~ Implications
As we found in the EMAP survey, the amount of non-target stream length in the NHD+ sample frame was approximately 65% of the total sample frame. This added significant labor costs to the reconnaissance efforts and contributes to the overall error in our estimates. The significant proportion of stream length that we were unable to assign to either the target or non-target populations (due primarily to landowner denials and physical barriers) further reduced the proportion of the resource that we were able to assess. It is likely that much of this unassessed population is likely to be dominated by non-target stream channels (i.e., mostly dry, non-perennial streams), but this uncertainty also contributes to the overall variability in our assessments. Both of these factors illustrate the need for a sample frame that better reflects the location of perennial and non-perennial channels in the state.

Condition Assessments~ Results
The condition estimates for the 6-year average and 4-year rolling averages are presented in Figure 5 and Figure , and all estimate data are summarized in Table 6. The cumulative distribution functions underlying these figures are shown in Figures .

Since the target status of ~17% of total stream length could not be assessed, we have presented the overall condition assessment in two alternative forms. Figure 5a displays the proportion of unassessed stream length (reconnaissance fate codes LD, PB and NS) extrapolated from the sampled target site data (TS + TNS) while Figure 5b displays the unassessed stream length left as a distinct category (represented by a question mark). The former presentation requires the assumption that the unassessed stream length has the same proportion of stream condition as the assessed stream length, while the latter makes no assumptions about this portion of the stream population.

The overall proportion of stream length in the three condition classes for the 6-year dataset (2000-2005) was similar to that reported for the first four years in 2005 (Ode and Rehn 2005). The percent of stream length in unimpaired condition (~57%) was slightly lower than we reported in 2005 (57% vs. 65%, Figure 5a), but this percentage was consistent across the three 4-year rolling average assessments (Figure 6), ranging between 53% and 62% (Table 6).

Table 6. Condition estimates for wadeable perennial streams in California for three groupings: 1) annual estimates, 2) 4-year rolling averages and 3) 6-year average.

Subpopulation	Category	N	% Stream Length	Standard Error	Lower 95% Confidence Limit	Upper 95% Confidence Limit
2000	**NI**	18	59.08	11.79	36.0	82.2
2000	**I_3sd**	6	32.42	11.52	9.8	55.0
2000	**VI**	2	8.51	4.45	0.0	17.2
2000	**Total**	**26**	**100**	**0**	**100.0**	**100.0**
2001	**NI**	43	63.84	12.14	40.1	87.6
2001	**I_3sd**	16	32.32	12.26	8.3	56.4
2001	**VI**	5	3.84	1.81	0.3	7.4
2001	**Total**	**64**	**100**	**0**	**100.0**	**100.0**
2002	**NI**	34	36.36	8.2	20.3	52.4
2002	**I_3sd**	18	37.83	11.75	14.8	60.9
2002	**VI**	9	25.82	12.41	1.5	50.1
2002	**Total**	**61**	**100**	**0**	**100.0**	**100.0**
2003	**NI**	10	34.14	11.6	11.4	56.9
2003	**I_3sd**	8	18.89	6.11	6.9	30.9
2003	**VI**	10	46.97	12.06	23.3	70.6
2003	**Total**	**28**	**100**	**0**	**100.0**	**100.0**
2004	**NI**	16	75.09	8.93	57.6	92.6
2004	**I_3sd**	12	13.68	5.82	2.3	25.1
2004	**VI**	23	11.23	5.38	0.7	21.8
2004	**Total**	**51**	**100**	**0**	**100.0**	**100.0**
2005	**NI**	12	42.07	9.63	23.2	61.0
2005	**I_3sd**	13	25	8.18	9.0	41.1
2005	**VI**	26	32.93	8.57	16.1	49.7
2005	**Total**	**51**	**100**	**0**	**100.0**	**100.0**
2000-2003	**NI**	105	52.88	7.48	38.2	67.5
2000-2003	**I_3sd**	48	33.14	7.92	17.6	48.7
2000-2003	**VI**	26	13.98	4.62	4.9	23.0
2000-2003	**Total**	**179**	**100**	**0**	**100.0**	**100.0**
2001-2004	**NI**	103	61.69	6.55	48.9	74.5
2001-2004	**I_3sd**	54	25.21	6.14	13.2	37.2
2001-2004	**VI**	47	13.1	3.76	5.7	20.5
2001-2004	**Total**	**204**	**100**	**0**	**100.0**	**100.0**
2002-2005	**NI**	72	54.91	6.25	42.7	67.2
2002-2005	**I_3sd**	51	22.5	4.74	13.2	31.8
2002-2005	**VI**	68	22.59	4.79	13.2	32.0
2002-2005	**Total**	**191**	**100**	**0**	**100.0**	**100.0**
All_Years	**NI**	133	57.39	5.34	46.9	67.9
All_Years	**I_3sd**	73	25.40	4.91	15.8	35.0
All_Years	**VI**	75	17.20	3.60	10.1	24.3
All_Years	**Total**	**281**	**100**	**0**	**100.0**	**100.0**

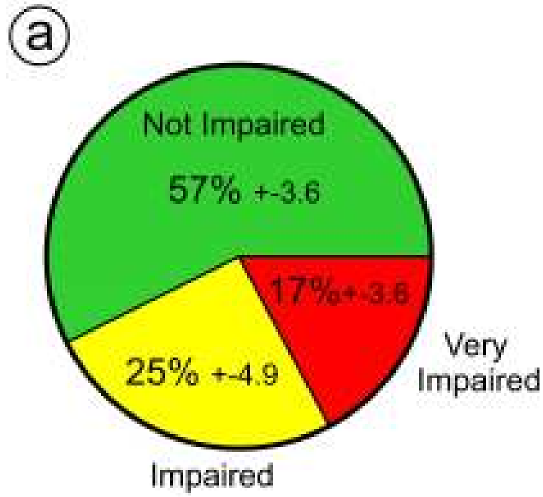

Figure 5. The percent of total stream length in different biological condition classes with two presentation alternatives: a) with the proportion of unassessed stream length (reconnaissance fate codes LD, PB and NS) extrapolated from the sampled target site data (TS + TNS), b) with unassessed stream length left as a distinct category (represented here by a question mark).

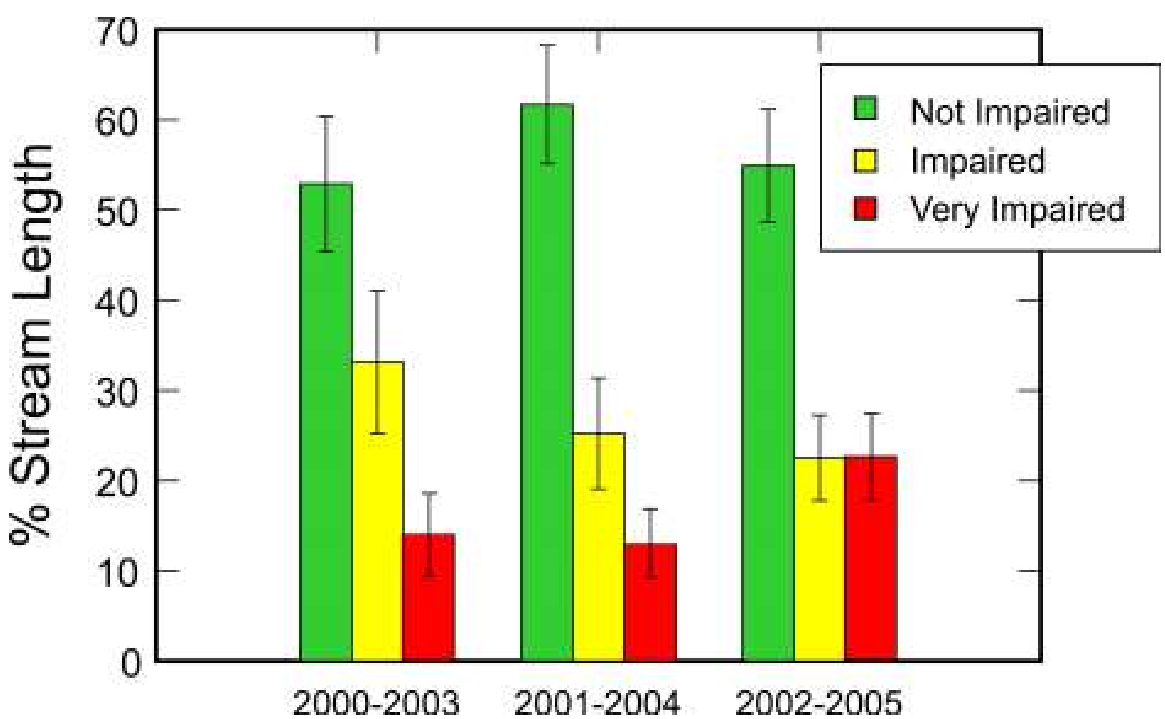

Figure 6. Percent of stream length (± 1se) in each of three condition categories for perennial wadeable streams in California. Bar sets represent four year rolling averages.
Condition Assessments~ Implications

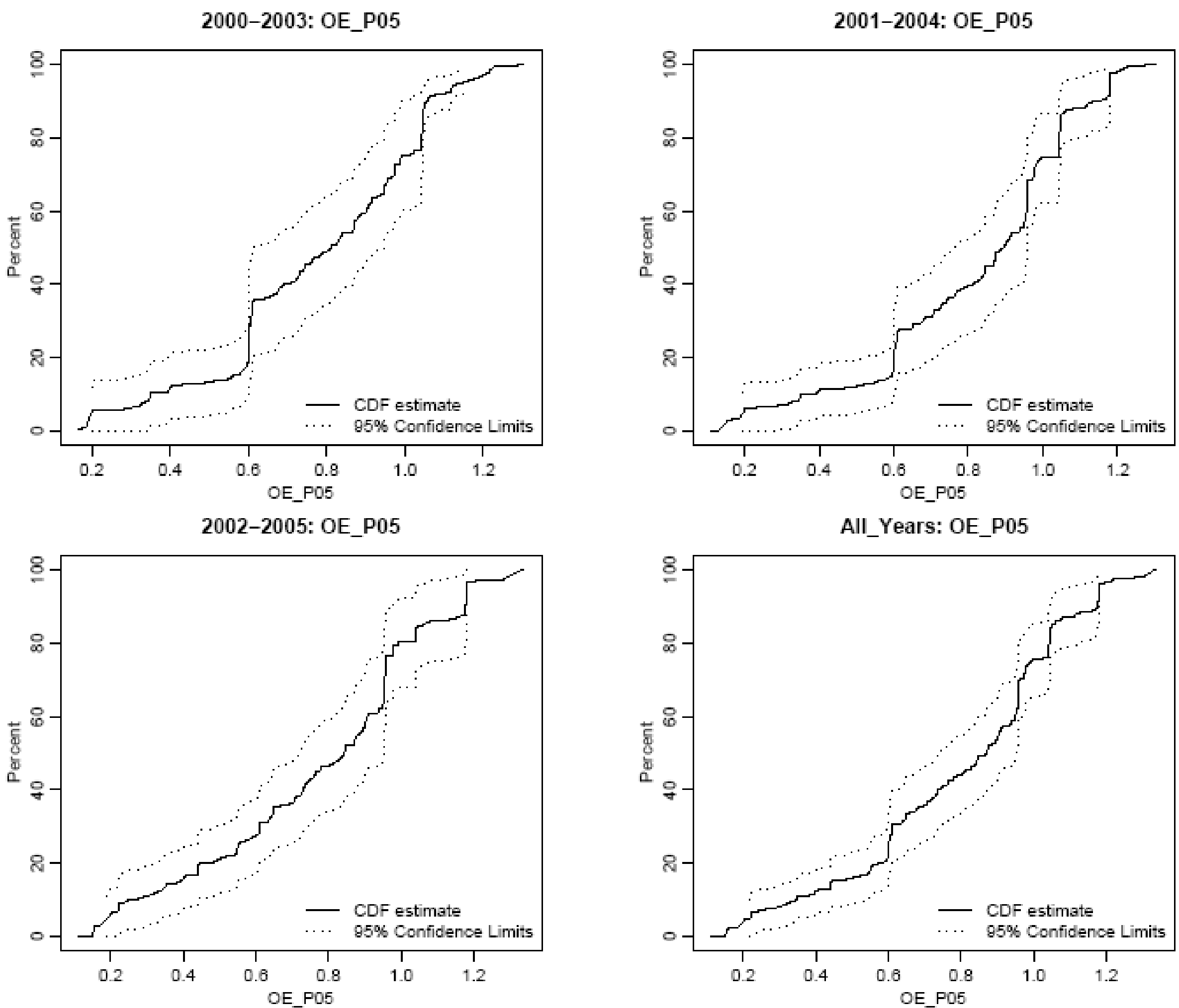

Figure 7. Cumulative distribution frequency graphs of biological condition scores for each of three 4-year rolling averages and for the 6-year average.

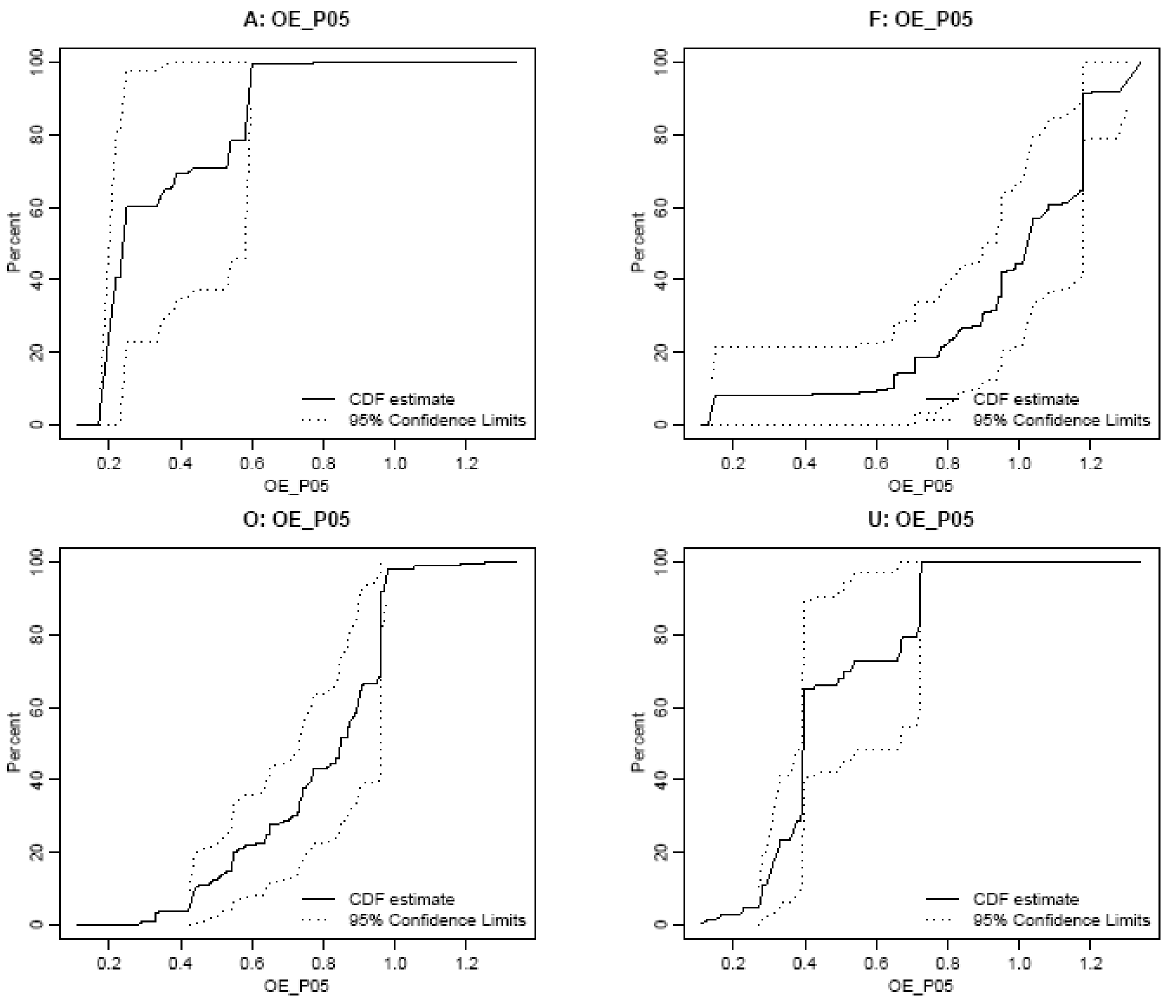

Figure 8. Cumulative distribution frequency graphs of biological condition scores for the 2002 – 2005 samples. Individual CDFs are presented for each of the four land cover classes used in the study (A= agricultural, n=13; F= forested, n=55; O= other, n=89, U= urban, n=30).

The overall condition assessments presented here were generally similar to those reported in 2005 (Ode and Rehn 2005), and the condition estimates were consistent over the first three years of rolling averages. However, the condition estimates of perennial wadeable streams in California indicate a slightly lower overall stream condition than we reported in 2005. The most likely explanation for this difference is the fact that we added several sites that were not included in the original assessments (sites that were only scored with RWB data). These RWB sites tended to be of poorer quality, driving the average condition down.

Stressor Extent~ Results

Correlations between biological response scores (O/E) and all the various chemical, habitat and landscape stressors were significant at the $p<0.0001$ level (Figure 9, Figure 10). Most of the landuse variables had strong negative associations with biological condition scores; only % forested landcover had a positive relationship with biological condition. These patterns were consistent across all three spatial scales (Figure 10). There was surprisingly little correlation among the various stressor variables that were measured, indicating a considerable amount of independence in these measures (Table 7). The majority of correlations with values > 0.5 were for relationships among various landuse measures and most of these examples were autocorrelation among single landuse classes measured at different spatial scales.

Six-year totals of the percent of total stream length with stressor levels greater than the thresholds listed in Table 2 are presented in Figure 11. All three sets of landuse stressors (urban, agricultural and urban+agricultural) displayed a similar pattern with respect to the spatial distribution of these landcover classes. The extent of urban and agricultural lands greater than the analytical thresholds (25% and 50% respectively) was much greater at local scales (5k and 1k buffers) than at the watershed scale for each of these variables. Agricultural landcover percentages greater than 50% were more common than urban landcover percentages greater than 25%, but even taken together (i.e., U_INDEX greater than 40%), stream length affected by high levels of these landcover classes represented a relatively small proportion of the total target stream population (approximately 10%). High nutrient levels and chloride levels were present at a relatively large percentage of stream length statewide (PTL ~13%, NTL ~35%, CL ~27%), while conductance, total suspended sediments and turbidity were higher than threshold levels in less than 5% of stream length statewide. Approximately 30% of stream length had low scores for instream habitat complexity (XFC_NAT), while riparian vegetative complexity (XCMGW) and riparian disturbance (W1_HALL) scores were low at approximately 10% and 25% of total stream length, respectively. Approximately 10% of stream length had fine sediment levels > 25% (P_SAFN), while ~35 of total stream length had low streambed stability scores (LRBS).

Although sample sizes were relatively low for agricultural and urban sites, we calculated stressor extent estimates for chemical and physical habitat stressors for each of the four main NPS classes (Figure 12). Both chemical and physical habitat stressors were much more prevalent in the agricultural and urban populations than in either the forested or “other” populations, frequently reaching greater than 70% of stream length.

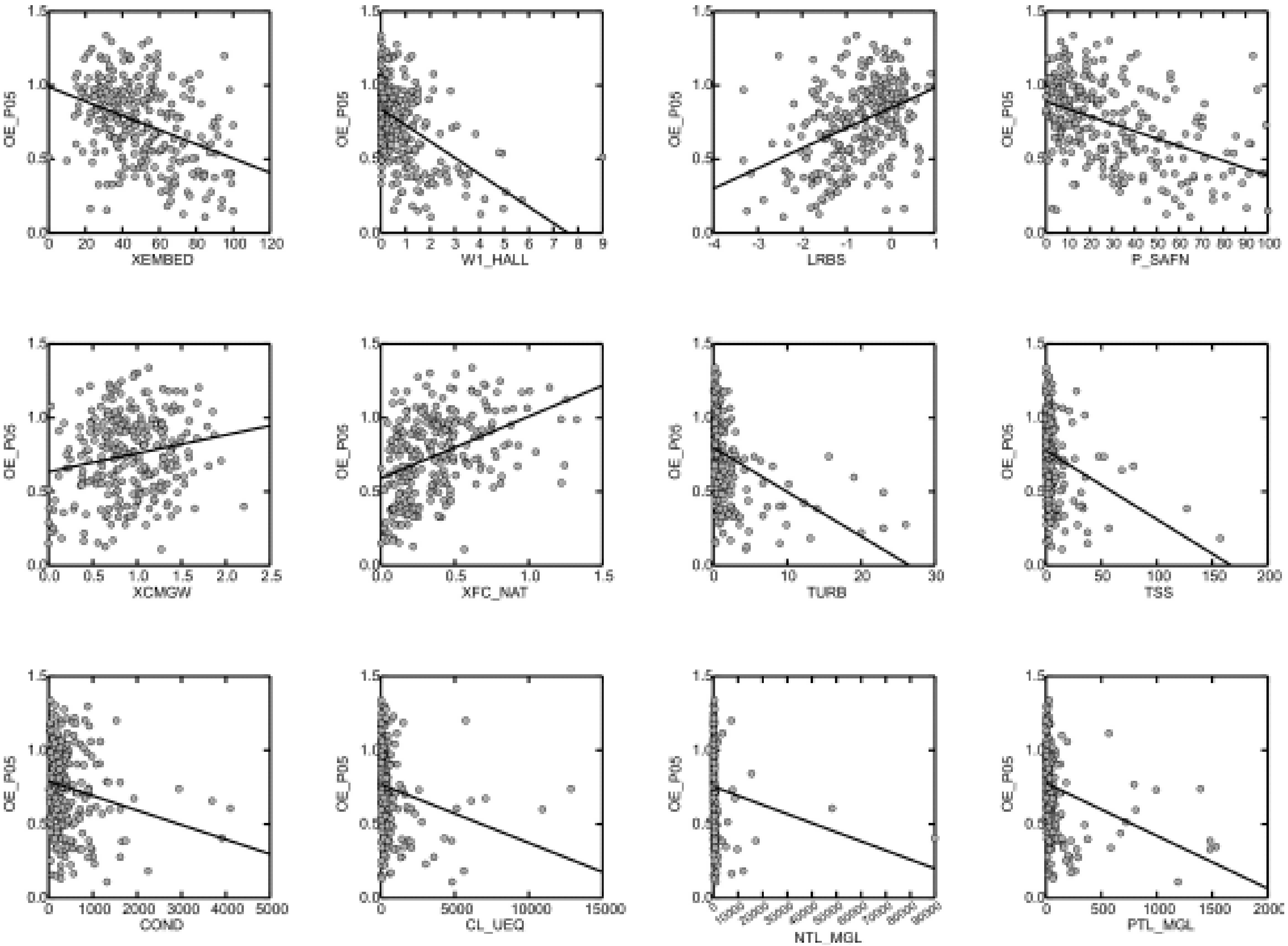

Figure 9. Scatterplots of relationships between biological condition scores (O/E) and various physical and chemical stressor gradients. Note that three outlier sites with very high TURB, TSS or XFC_NAT values (113WE0619, 403CE0156, 403CE0188) have been removed to clarify the patterns in these variables, but all sites were included in analyses. All relationships were significant at the p <0.0001 or less.

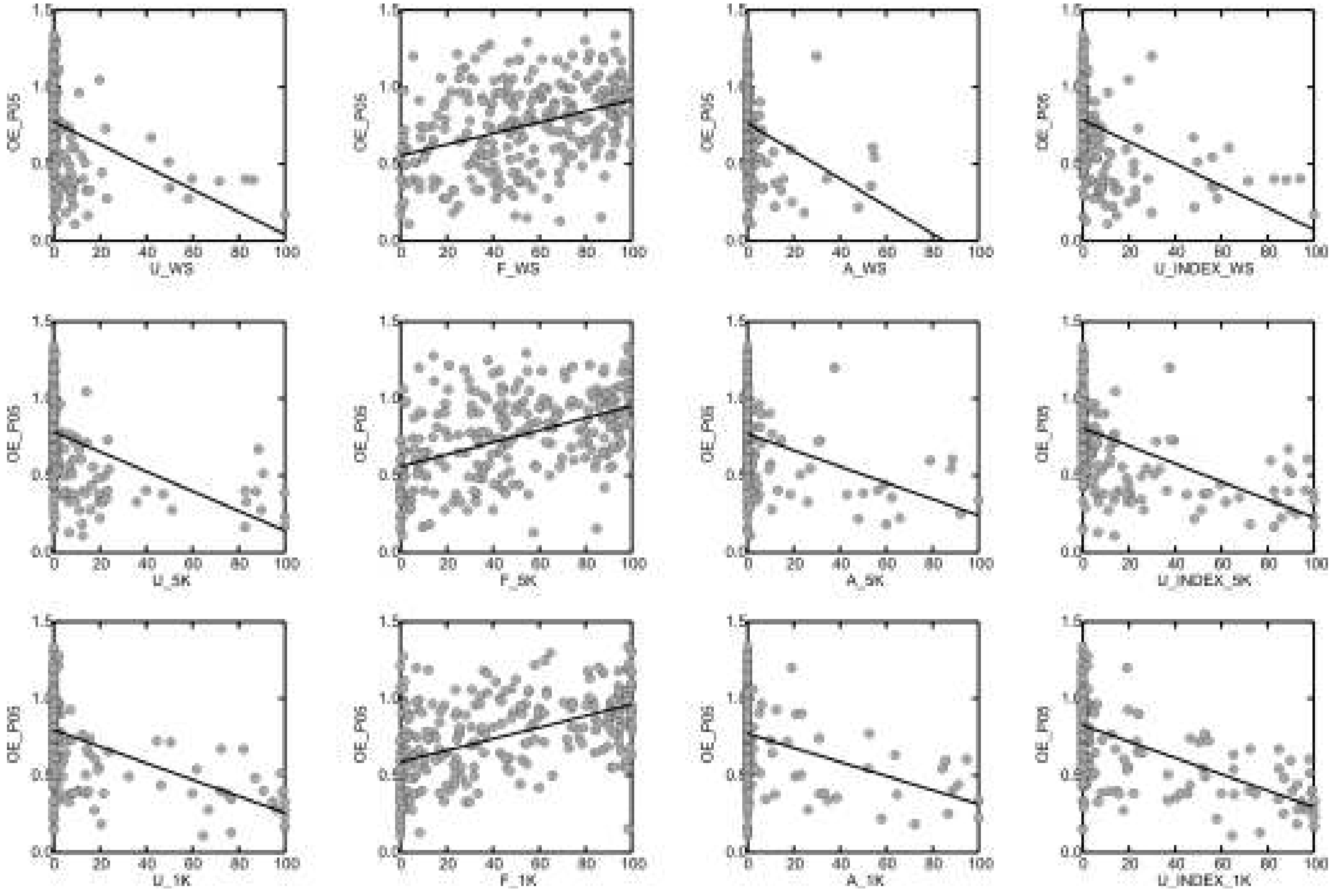

Figure 10. Scatterplots of the relationship between biological condition scores (O/E) and 12 landuse/ landcover gradients. All relationships were significant at the p <0.0001 or less. (See Table 2 for landuse definitions.

Table 7. Stressor correlation matrix (Pearson product moment correlations). Correlations >0.5 are highlighted in yellow.

	W1_HALL	LRBS	P_SAFN	XCMGW	XFC_NAT	NH4_UEQ	TURB	TSS	COND	CL_UEQ	NTL_MGL	PTL_MGL	U_1K	U_5K	U_WS	F_WS	A_1K	A_5K	A_WS	O_WS	F_5K	O_5K	F_1K	O_1K	U_INDEX_1K	U_INDEX_5K
W1_HALL	1.00																									
LRBS	-0.39	1.00																								
P_SAFN	0.45	**-0.84**	1.00																							
XCMGW	-0.28	0.20	-0.19	1.00																						
XFC_NAT	-0.22	0.17	-0.28	0.32	1.00																					
NH4_UEQ	0.21	-0.18	0.20	-0.05	-0.09	1.00																				
TURB	0.29	-0.14	0.15	-0.22	-0.14	0.15	1.00																			
TSS	0.25	-0.17	0.16	-0.21	-0.09	0.37	**0.92**	1.00																		
COND	0.22	-0.31	0.43	-0.20	-0.20	0.26	0.12	0.18	1.00																	
CL_UEQ	0.22	-0.28	0.36	-0.11	-0.12	0.29	0.09	0.14	**0.76**	1.00																
NTL_MGL	0.23	-0.27	0.30	-0.13	-0.10	0.21	0.03	0.11	**0.60**	0.34	1.00															
PTL_MGL	0.30	-0.21	0.24	0.00	-0.10	0.37	0.24	0.28	0.26	0.41	0.13	1.00														
U_1K	**0.56**	-0.26	0.35	-0.13	-0.22	0.07	0.24	0.20	0.15	0.14	0.05	0.20	1.00													
U_5K	**0.53**	-0.21	0.35	-0.15	-0.17	0.08	0.08	0.04	0.22	0.24	0.19	0.16	**0.71**	1.00												
U_WS	0.36	-0.21	0.39	-0.13	-0.14	0.07	0.06	0.03	0.24	0.25	0.31	0.26	0.50	**0.79**	1.00											
F_WS	-0.30	0.33	-0.42	0.27	0.18	-0.16	-0.16	-0.16	-0.47	-0.39	-0.22	-0.29	-0.30	-0.35	-0.38	1.00										
A_1K	0.36	-0.33	0.35	-0.26	-0.21	0.28	0.20	0.24	0.30	0.16	0.45	0.11	0.00	0.07	0.08	-0.25	1.00									
A_5K	0.40	-0.35	0.41	-0.28	-0.19	0.34	0.24	0.28	0.31	0.20	0.44	0.15	0.05	0.06	0.07	-0.31	**0.89**	1.00								
A_WS	0.24	-0.32	0.35	-0.27	-0.16	0.19	0.08	0.12	0.43	0.27	**0.53**	0.06	0.07	0.09	0.13	-0.36	**0.62**	**0.74**	1.00							
O_WS	0.09	-0.19	0.19	-0.17	-0.10	0.10	0.13	0.13	0.30	0.24	-0.05	0.18	0.07	-0.03	-0.10	**-0.85**	0.08	0.12	0.08	1.00						
F_5K	-0.41	0.38	-0.48	0.32	0.27	-0.17	-0.22	-0.21	-0.44	-0.37	-0.20	-0.30	-0.39	-0.39	-0.35	**0.91**	-0.34	-0.38	-0.33	**-0.77**	1.00					
O_5K	-0.06	-0.13	0.12	-0.14	-0.10	-0.03	0.07	0.08	0.22	0.18	-0.11	0.16	-0.02	-0.20	-0.13	**-0.68**	-0.10	-0.11	-0.05	**0.85**	**-0.72**	1.00				
F_1K	-0.43	0.44	**-0.53**	0.30	0.32	-0.16	-0.19	-0.19	-0.39	-0.31	-0.16	-0.26	-0.40	-0.34	-0.28	**0.76**	-0.33	-0.32	-0.28	**-0.65**	**0.87**	**-0.63**	1.00			
O_1K	-0.12	-0.12	0.14	-0.11	-0.08	-0.03	-0.07	-0.06	0.16	0.15	-0.10	0.09	-0.29	-0.18	-0.09	-0.49	-0.15	-0.15	-0.07	**0.62**	**-0.50**	**0.77**	**-0.64**	1.00		
U_INDEX_1K	**0.66**	-0.40	0.49	-0.25	-0.30	0.22	0.31	0.30	0.30	0.22	0.31	0.22	**0.81**	**0.62**	0.45	-0.39	**0.58**	**0.56**	0.41	0.11	**-0.51**	-0.07	**-0.52**	-0.32	1.00	
U_INDEX_5K	**0.64**	-0.37	**0.52**	-0.28	-0.25	0.27	0.21	0.20	0.36	0.31	0.41	0.22	**0.57**	**0.79**	**0.64**	-0.45	**0.59**	**0.66**	**0.52**	0.05	**-0.53**	-0.21	-0.46	-0.22	**0.81**	1.00
U_INDEX_WS	0.40	-0.31	0.49	-0.23	-0.19	0.14	0.08	0.08	0.40	0.34	**0.50**	0.24	0.45	**0.71**	**0.89**	-0.48	0.34	0.39	**0.55**	-0.05	-0.44	-0.13	-0.36	-0.11	**0.57**	**0.78**

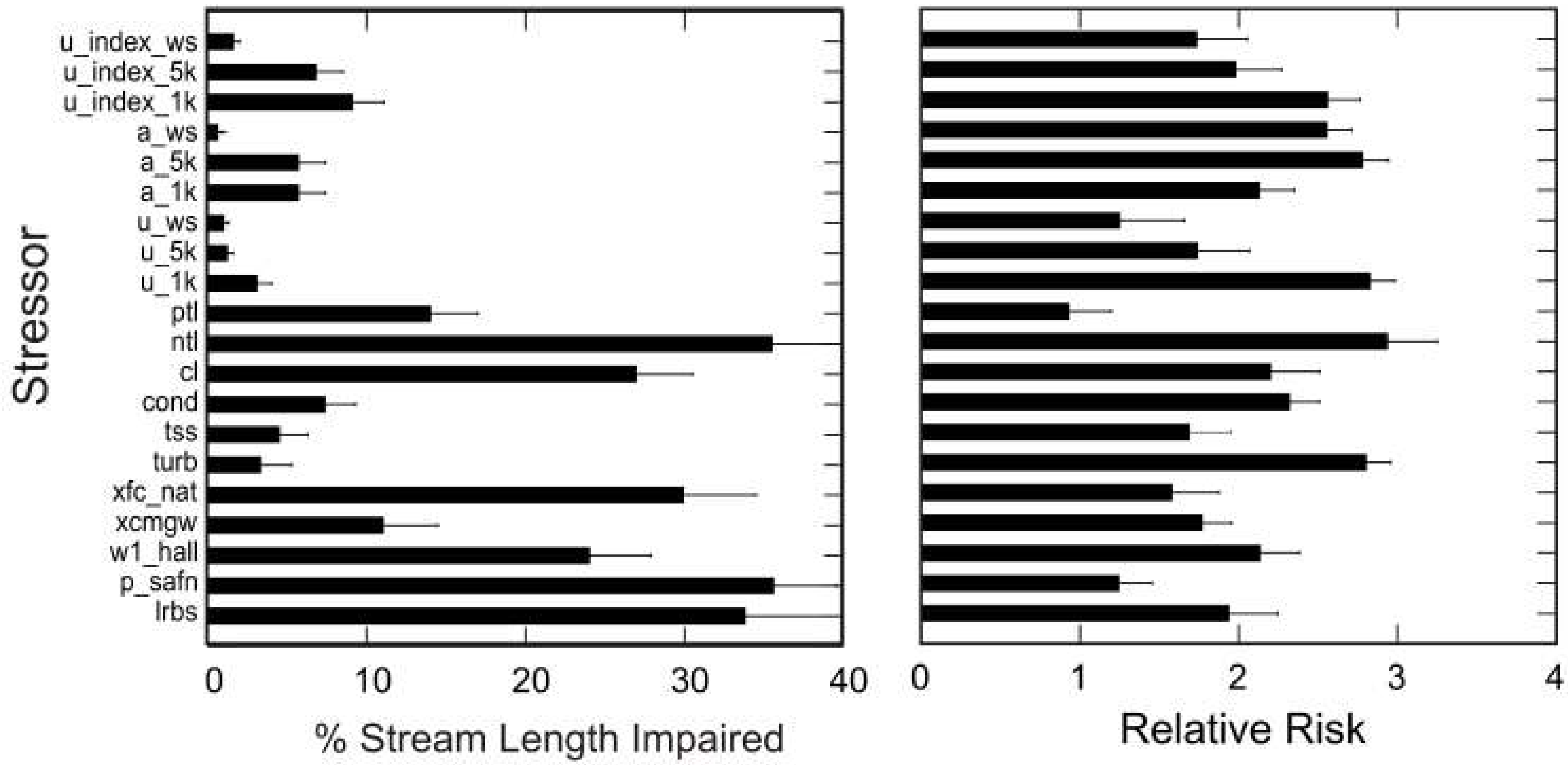

Figure 11. Stressor extent and relative risk estimates for wadeable perennial streams sampled between 2000 and 2006: a) percent of total stream length impaired by each of 20 potential measures of anthropogenic stress and b) relative risk of biotic impairment associated with the presence of high stressor values. (See Table 2 for definitions of stressor abbreviation.

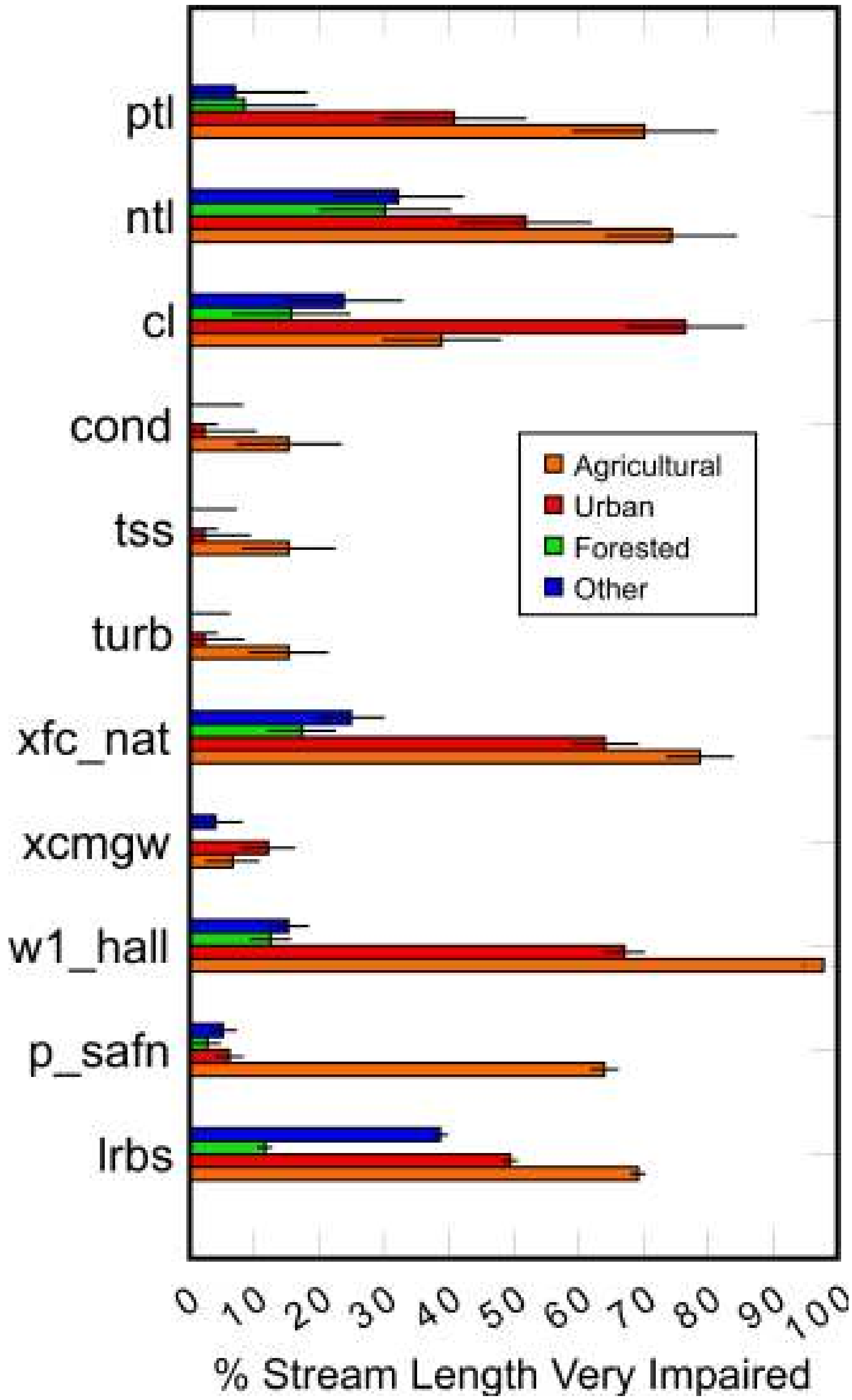

Figure 12. Stressor extent estimates for 11 physical and chemical attributes. Stressor threshold and abbreviations are defined in Table 2.

Relative Risk~ Results
Six-year estimates of the relative risk of biotic impairment associated with high stressor levels are presented in Figure 11. The relative risk of biotic impairment was generally greater from urbanization when it was present at the local scale than when it was present in the watershed. Presence of urbanization at the local levels was associated with up to 3x greater likelihood of biotic impairment than when it wasn't present. This effect was also observed for the combined U_INDEX. However, this pattern was not observed for agricultural landuse, where presence of high amounts of agricultural land cover at local scales was not associated with higher risk than for landuse distributed throughout the watershed.

Combining Stressor Extent and Relative Risk
The combination of information from the stressor extent and relative risk estimates can provide valuable insight into the magnitude of the effects various stressors have on aquatic life use condition. For example, while large concentrations of urban and agricultural landuses were not associated with a large percentage of the state's wadeable perennial streams, when they are present they have a strong negative affect on biological condition, especially when present locally. Likewise, whereas high levels of conductance, total suspended solids and turbidity are only present at less than 5% of total stream length, when they were present they had a strong negative effect on biological condition. High total nitrogen levels were present in ~35% of the total stream length and when present were associated with a three-fold increase in risk of biological impairment. In contrast, total phosphorus concentration (PTL) had low values at ~15% of the stream population, but where it was present, it was not associated with an increased risk of biological impairment. ***Note: It is important to remember that the stressor extent and relative risk estimates are very sensitive to the thresholds used to define high levels of each stressor. We are working on methods to evaluate the effects of varying stressor thresholds on stressor extent and relative risk results. These will be presented in later reports.***

<u>Probability Survey Assessments (NPS Classes)</u>

Although we only have two years of CMAP data, we assessed enough sites in the main non-point source landuse/landcover classes (NPS) to summarize the distribution of condition scores for each class (Figure 13) and present a preliminary condition assessment of streams affected by these NPS classes (Figure 14). These topics will be treated in greater detail when the full results of the CMAP NPS project are presented in December 2008.

In this initial assessment, sites with significant upstream agricultural and urban landcover had dramatically different distributions of condition scores than the forested and "other" classes. Nearly all the stream length in these categories had impaired biological condition to some degree, and the vast majority of the stream length (~80%) in both classes had very impaired biological condition (i.e., had many fewer of the species predicted to occur at these sites under the reference conditions defined by the O/E models used to score sites).

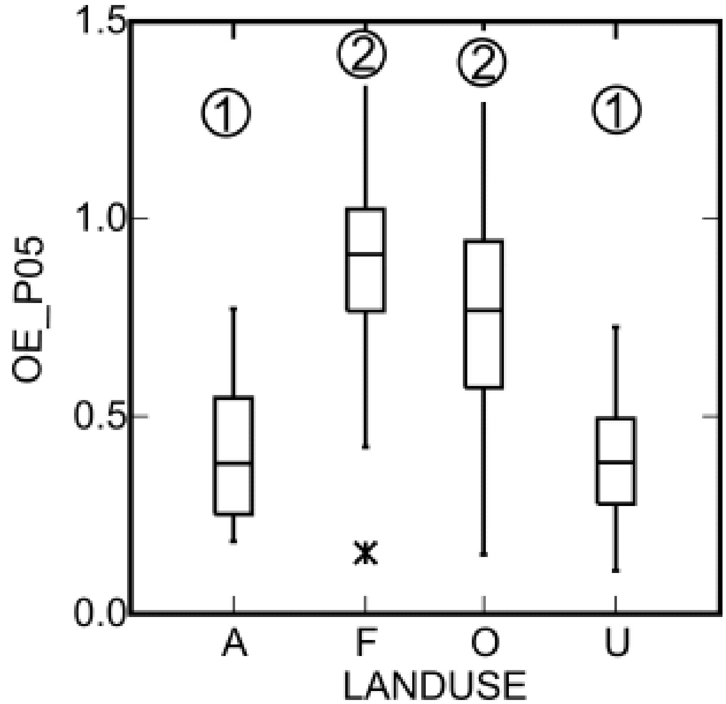

Figure 13. Boxplots of distribution of biological condition scores (OE_P05) by landuse assignment (A= agricultural, F= forested, O= other, U= urban). Score distributions differ significantly for classes with different number codes (ANOVA F = 47.73, p< 0.00001).

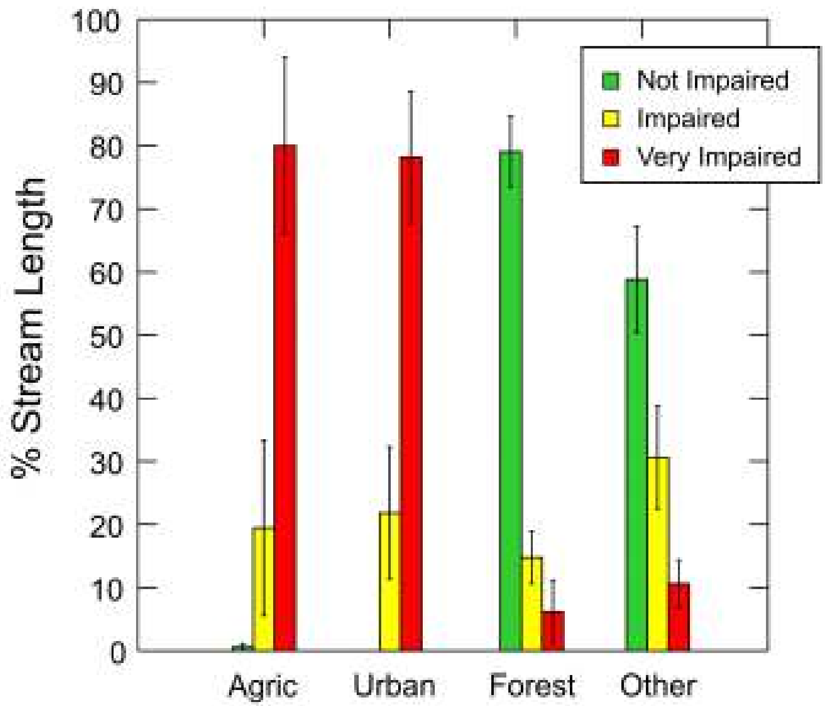

Figure 14 Percent of wadeable perennial stream length in California in three condition classes in each of the four landuse/ landcover classes (2000-2006). Sample sizes: Agriculture = 13, Urban = 32, Forested = 96 and Other = 133.

Based on our extent estimates, only about 10% of stream length statewide met the definitions of agriculture-dominated or urban-dominated streams that we used for these analyses. However, nearly all the stream length in these categories had biological condition that was impaired to some degree, and the vast majority of the stream length in both classes had many fewer of the species predicted to occur at these sites under the reference conditions defined by the O/E models used to score sites. Biological condition was more negatively associated with local urbanization than for urbanization spread throughout the watershed; this pattern was not seen for agricultural landuse.

While these preliminary results offer useful perspective, they should be interpreted with caution for two reasons: 1) low sample sizes (especially in agricultural streams) may result in a biased sample of the population of agricultural streams and 2) the RIVPACS models that were used to score biological condition scores may overestimate the degree of impairment in low elevation regions for which the models had few reference sites. Since these regions tend to be dominated by agricultural and urban landuses, the impairment thresholds based on higher elevation reference sites likely over-estimate impairment.

Note: Relative risk estimates are not presented for NPS categories here because sample sizes were too low to generate meaningful estimates.

Methods Comparisons

The comparison of RWB and TRC field collection methods are presented in Figures 15-18. RIVPACS O/E scores generated both by models that included ($p>0.0$) and excluded ($p<0.5$) rare taxa were highly correlated between the two methods, although the two methods were more highly correlated when scored against models that excluded rare taxa (Figure 4). Models that excluded rare taxa were more highly correlated than ones that included rare taxa. Samples collected with the TRC method consistently scored slightly higher than RWB method by approximately 0.05 units on the O/E scale. Figures 16-18 present scatterplots showing the relationship between the score differential for the methods and key environmental gradients (physical and chemical variables, instream and riparian condition and landuse variables, respectively). None of the 38 gradients that we evaluated showed any discernable relationship with the score differential.

The results of the methods comparison (levels of correlation between methods, lack of evidence of systematic bias with respect to key environmental gradients) are similar to those observed in EMAP datasets (Gerth and Herlihy 2006, Rehn et al. 2007). As in the early study, we again saw consistently higher TRC scores (~0.05 higher than for RWB), but this bias was not influenced by any of the environmental gradients (natural or anthropogenic) we evaluated. These results support the general conclusion from the EMAP dataset that TRC and RWB data can be interchanged as long as a correction factor is applied to account for the higher TRC scores.

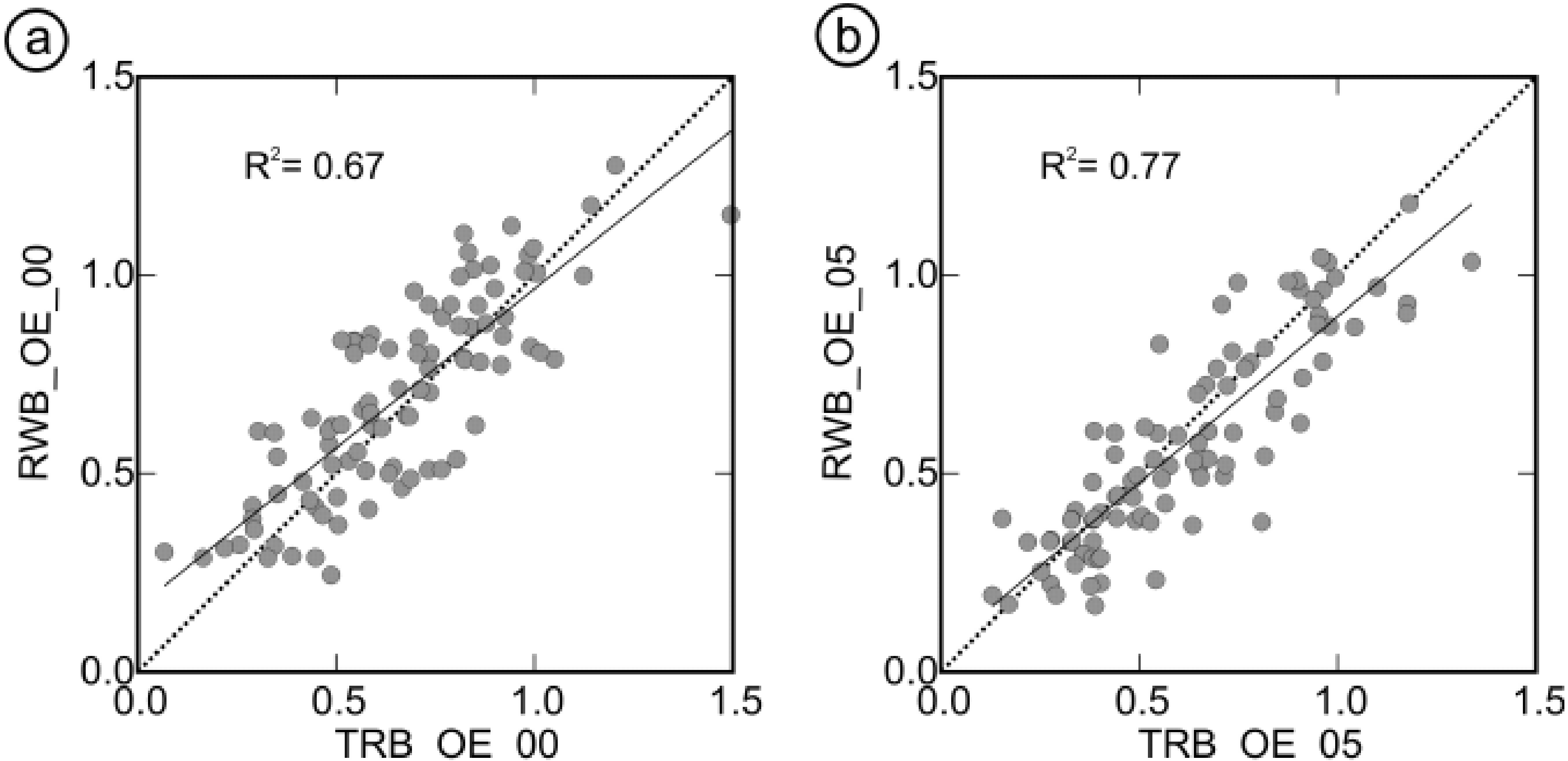

Figure 15. Scatterplots of relationships between O/E scores for paired targeted riffle (TRB) vs. reachwide benthos (RWB) samples collected at 102 sites under the CMAP program in 2004 and 2005 using output from models that either: a) include (00) or b) exclude (05) less common taxa. Dotted lines represent a 1:1 relationship, while the solid lines represent the best fit linear regression line.
Note: Mean difference between TRB and RWB is 0.045 (TRB> RWB). This difference is significant at p=0.001

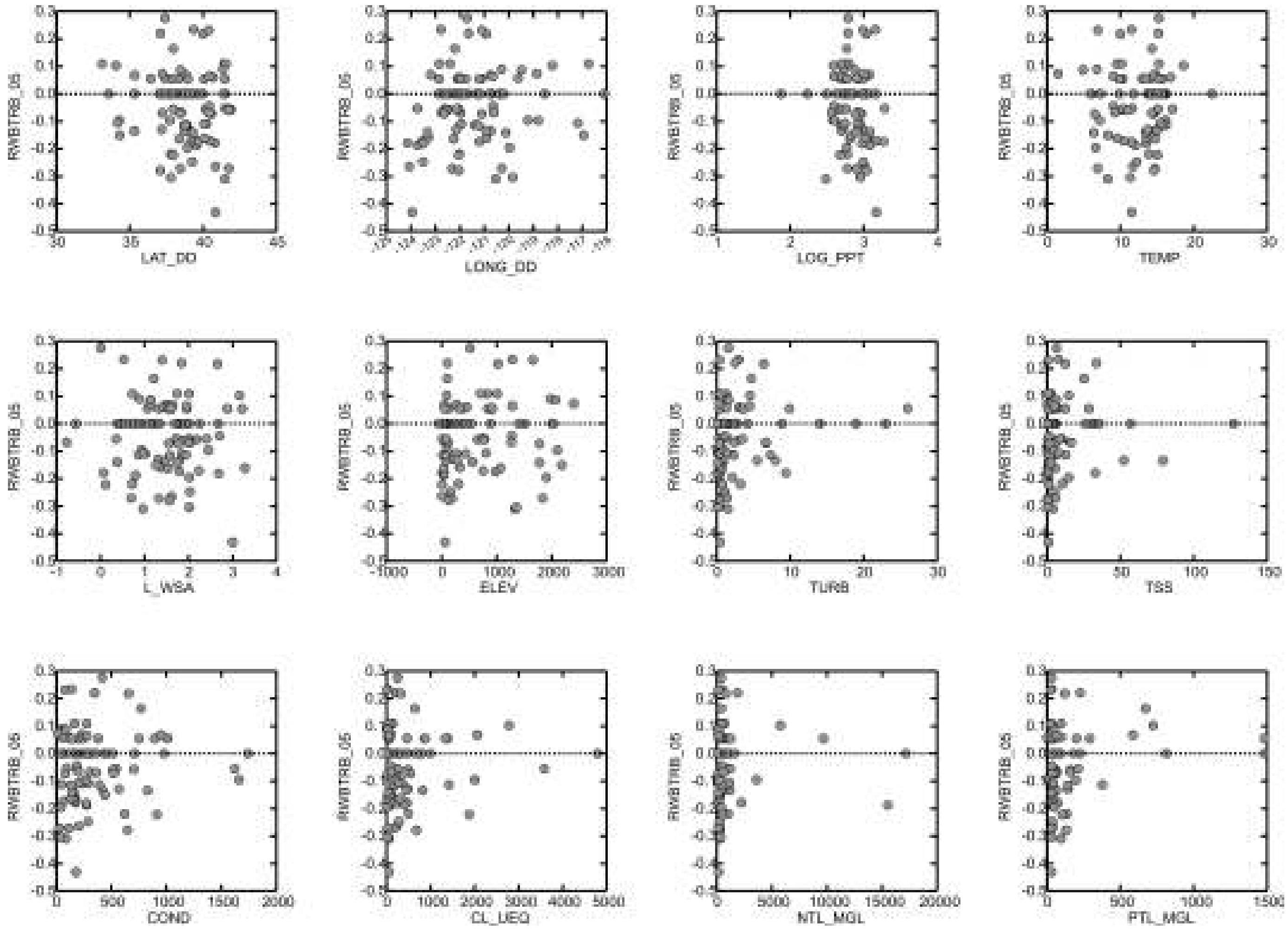

Figure 16. Relationships between the difference between sampling method (RWB minus TRB O/E scores for paired samples using output from models that exclude (p> 0.5) less common taxa) and various physical and chemical gradients using CMAP data from 2004 and 2005. Dotted lines indicate the relationship expected if there is no effect of the gradient on the difference in methods. Note two sites (403CE0156 and 403CE0188) with very high TSS and TURB values were removed from these Figures to clarify the relationship at the lower end of these gradients, but these had no effect on the relationship. (See Table 2 for stressor definitions.)

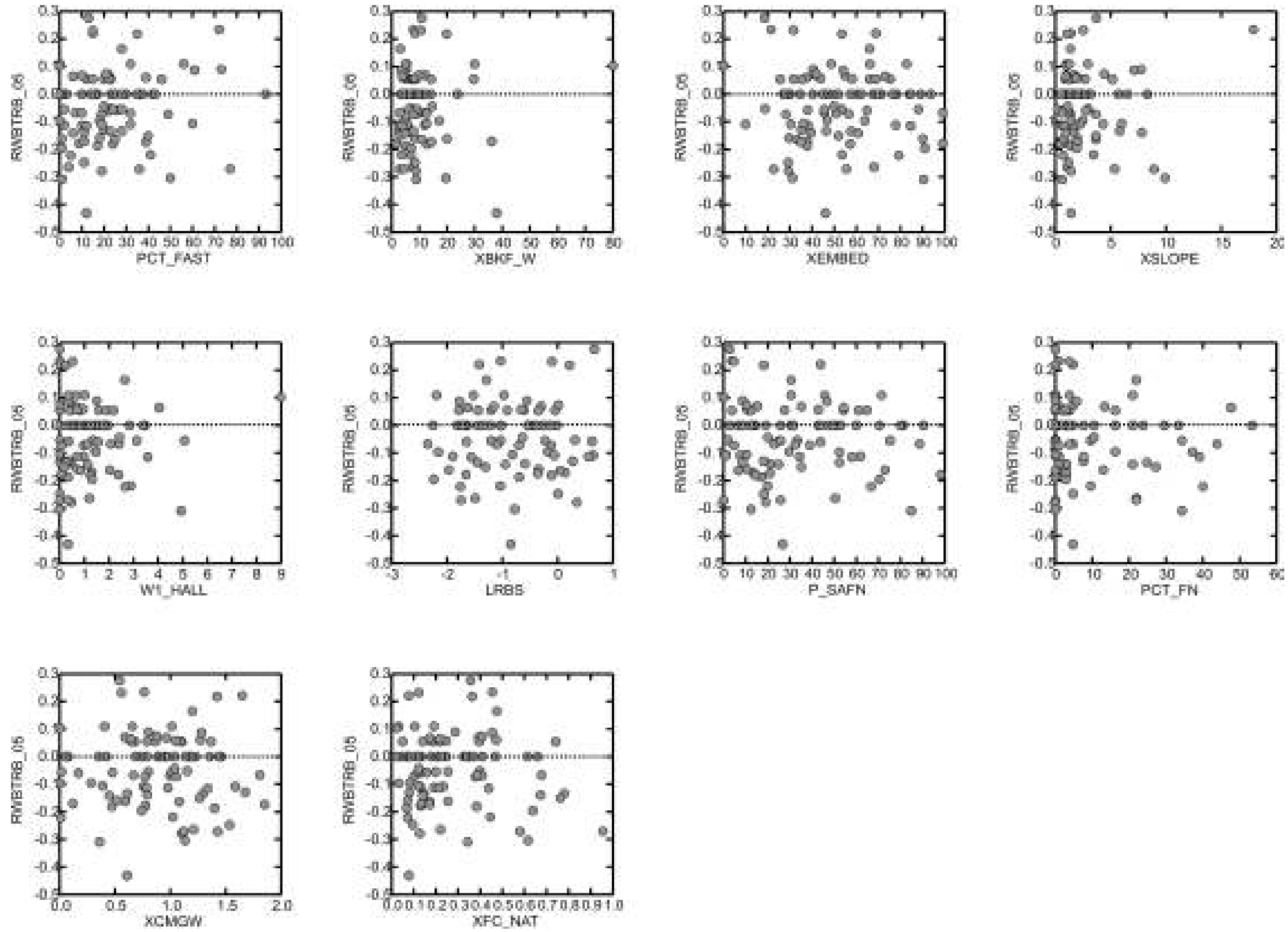

Figure 17. Relationships between the difference between sampling method (RWB minus TRB O/E scores for paired samples using output from models that exclude (p> 0.5) less common taxa) and various physical habitat condition gradients using CMAP data from 2004 and 2005. Dotted lines indicate the relationship expected if there is no effect of the gradient on the difference in methods. (See Table 2 for stressor definitions.)

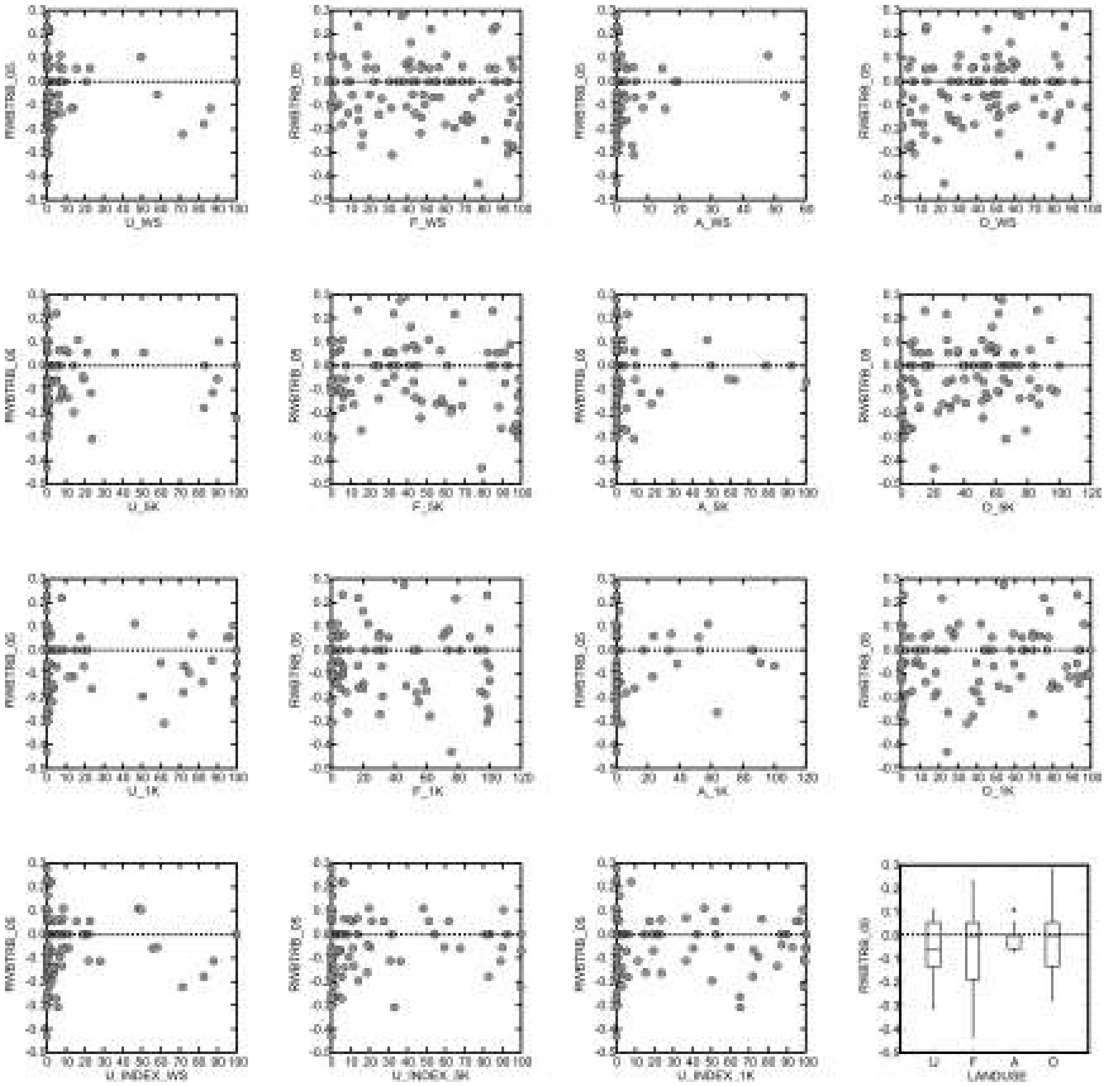

Figure 18. Relationships between the difference between sampling method (RWB minus TRB O/E scores for paired samples using output from models that exclude (p> 0.5) less common taxa) and various landuse/ landcover gradients using CMAP data from 2004 and 2005. Dotted lines indicate the relationship expected if there is no effect of the gradient on the difference in methods. (See Table 2 for stressor definitions.)

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