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Development of a Benthic Index of Biotic Integrity (B-IBI) for Wadeable Streams in Northern Coastal California and its Application to Regional 305(b) Assessment

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Introduction

Bioassessment, or the use of biological assemblages to evaluate ecological condition, has become a widely used technique in water quality monitoring programs around the world. Benthic macroinvertebrates (BMIs) are the most commonly used organisms in bioassessment; in the United States, they are used in over 90% of bioassessment programs (Diamond *et al*. 1996). The index of biotic integrity (IBI) was first introduced by Karr (1981) as a measure of stream condition based on fish assemblages, but IBIs also have been developed for BMI (e.g. Kerans & Karr 1994) and periphyton (e.g. Pan *et al*. 1996) assemblages. An IBI is typically composed of a set of metrics that together represent different attributes of assemblage composition, structure and function such as species richness, tolerance guilds and trophic guilds. Metrics are selected for inclusion in an IBI based on their responsiveness to anthropogenic stressor gradients and/or their ability to discriminate between minimally disturbed reference sites and test sites that are known or suspected to have been exposed to stressors of interest. Adoption of a consistent and quantifiable method for defining reference condition is fundamental to any bioassessment program (Hughes 1995, Bailey *et al*. 2004).

In California, several state and federal agencies have become increasingly involved in developing analytical tools that can be used to assess the biological and physical condition of streams and rivers. For example, the US Environmental Protection Agency (EPA), the California EPA, the US Forest Service (USFS) and California's state and regional Water Quality Control Boards (WQCBs) have collected fish, periphyton and BMIs from California streams and rivers as a critical component of regional water quality assessment and management programs. Together, these agencies have sampled thousands of sites in California, but until recently, no analysis of BMI assemblage data sets based on comprehensively defined regional reference conditions has been undertaken (Ode *et al*. 2005). The purpose of this paper is to develop a benthic IBI (B-IBI) for the northern coastal region of California based on BMI assemblage data and apply the B-IBI to an assessment of all mapped wadeable streams in the region based on the probability stream survey recently completed by EPAs Environmental Monitoring and Assessment Program Western Pilot (WEMAP).

Methods

Study Area

The northern coastal California B-IBI (NorCal B-IBI) was developed for the region that drains directly west to the Pacific Ocean from Marin County in the south to the Oregon border in the north (Fig. 1). This area contains 3 Level III ecoregions (Omernik 1987) and receives the highest annual rainfall totals in California with areas near the Oregon border receiving nearly 200 inches and areas in the south receiving >50 inches over mountain ranges. High rainfall totals combined with rapid uplift due to tectonic subduction and compression, unstable soil types, and land use practices such as logging and grazing that promote erosion result in the state's highest total sediment yields (Mount 1995). The estimated human population in the region, although relatively low compared to other regions in California, exceeded 1,040,000 in 2004 and is concentrated in Marin and Sonoma counties (California Department of Finance, Demographic Research Unit, www.dof.ca.gov). The region currently supplies two-thirds of the state's total timber production and during the 19th century suffered large-scale deforestation and degradation of entire watersheds as timber harvest and mercury mining operations expanded to meet the demands of hydraulic gold mining in the Sierra Nevada.

Field Protocols and Combining Datasets

The NorCal B-IBI is based on BMI and physical habitat data collected from 257 sites (Fig. 1) using the three protocols described below. Sites were sampled during base flow periods between April and early October of 2000-2003.

California Stream Bioassessment Protocol (CSBP, 150 sites) - Both regional WQCBs in northern coastal California have implemented biomonitoring programs in their respective jurisdictions and have collected BMIs according to the CSBP (Harrington 1999). At CSBP sites, three riffles within a 100m reach were randomly selected for sampling. At each riffle, a transect was established perpendicular to the flow, from which three separate areas of 0.18 m^2 each were sampled upstream of a 0.3m wide D-frame net and composited by transect. A total of 1.82m^2 of substrate was sampled per reach and 900 organisms were subsampled from this material (300 organisms were processed separately from each of three transects). Water chemistry data was collected in accordance with the protocols of the different regional WQCBs (Puckett 2002) and qualitative physical habitat characteristics were measured according to Barbour *et al*. (1999) and Harrington (1999).

USFS (32 sites)- The USFS sampled streams on national forest lands in northern California in 2000 and 2001 using the Hawkins *et al*. (2001) targeted riffle protocol. All study reaches were selected non-randomly as part of a program to develop an interpretive (reference) framework for the results of stream biomonitoring studies on national forests in California. BMIs were sampled at study reaches (containing at least 4 fast water habitat units) by disturbing 2 separate 0.09m^2 areas of substrate upstream of a 0.3m wide D-frame net in each of four separate fast water units; a total of 0.72m^2 was disturbed and all sample material from a reach was composited. Field crews used a combination of qualitative and quantitative measures to collect physical habitat and water chemistry data (Hawkins *et al*. 2001). A 500 organism subsample was processed from the composite sample and identified following methods described by Vinson and Hawkins (1996).

WEMAP (75 sites) - The EPA sampled study reaches in northern coastal California from 2000 through 2003 as part of its WEMAP pilot project. 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 and maximum length of 500m. A BMI sample was collected at each site using the USFS methodology described above (Hawkins *et al*. 2001) in addition to a standard WEMAP BMI sample (not used in this analysis). A 500 organism subsample was processed in the laboratory according to WEMAP standard taxonomic effort levels (Klemm *et al*. 1990). 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).

Because USFS style riffle samples were collected at all WEMAP sites, only two field methods were combined in this study. All 900 count CSBP samples were standardized to 500 individuals per reach using a randomized rarefaction technique based on previous analyses that demonstrated no difference between the two methods once counts are adjusted (Ode *et al*. 2005). All WEMAP and CSBP samples were collected and processed by the California Department of Fish and Game's Aquatic Bioassessment Laboratory (ABL) and all USFS samples were processed by the US Bureau of Land Management's Bug Lab in Logan, Utah. Taxonomic data from both labs were combined in an MS Access© database that standardized BMI taxonomic effort levels and metric calculations allowing us to minimize any differences between the two labs that processed samples. Taxonomic effort followed standards defined by the California Aquatic Macroinvertebrate Laboratory Network (2002; www.dfg.ca.gov/cabw/camlnetste.pdf). Sites with fewer than 450 organisms sampled were omitted from the analyses.

Screening Reference Sites

We followed an objective and quantitative reference site selection procedure in which potential reference sites were first screened with quantitative GIS land use analysis at several spatial scales, and then were screened with local condition assessments (in-stream and riparian) to quantify stressors acting within study reaches. We calculated the proportions of different land cover classes and other measures of human activity upstream of each site at four spatial scales that give unique information about potential stressors acting on each site: 1) within polygons delimiting the entire watershed upstream of each sampling site; 2) within polygons representing local regions (defined as the intersection of a 1km radius circle around each site and the primary watershed polygon); 3) within a 100m riparian zone on each side of all streams within each watershed; 4) within a 100m riparian zone in the local region. We used the ArcView® (ESRI 1999) extension ATtILA (Ebert and Wade 2002) to calculate the percentage of various landcover classes (urban, agriculture, natural, etc.) and other measures of human activity (population density, road density, etc.) in each of the four spatial areas defined for each site. Landcover analyses were based on the California Department of Forestry and Fire Protection (CDF) Multisource California Land Cover Mapping and Monitoring Program (LCMMP, http://frap.cdf.ca.gov/data/frapgisdata/select.asp). Where available, these data were supplemented with development footprint layers derived from 2000 census data (CDF) and the California Department of Conservation's 2002 Farmland Mapping data (DOC- FMMP, http://www.consrv.ca.gov/dlrp/fmmp). Population data were derived from the 2000 migrated TIGER dataset (CDF). Stream layers were obtained from the USGS National Hydrography Dataset (NHD). The road network was obtained from the USFS Remote Sensing Lab (http://fsweb/gis/gis_data/calcovs/fs/nwctran03_2.html). The USGS National Elevation Dataset (NED) was used for elevation data. Frequency histograms of land use percentages for all sites were used to establish subjective thresholds for eliminating sites from the potential reference pool (Table 1). Sites were further screened from the reference pool on the basis of reach scale conditions (obvious bank stability or erosion/ sedimentation problems, evidence of mining, dams, grazing, recent fire, recent logging). Once the pool of reference sites was defined, we randomly divided the full set of sites into a development set that was used to screen metrics and establish scoring ranges for component B-IBI metrics and a validation set that was used for independent evaluation of B-IBI performance.

Screening Metrics and Assembling the B-IBI

Seventy-seven metrics were evaluated for possible use in the NorCal B-IBI (Table 2). A multi-step screening process was used to evaluate each metric for: 1) sufficient range to be used in scoring; 2) responsiveness to watershed scale and reach scale disturbance variables; 3) discrimination between reference and test sites; 4) lack of correlation with other responsive metrics.

Pearson correlations between all watershed scale and reach scale disturbance gradients were used to define the smallest suite of independent (non-redundant) disturbance variables against which to test biological metric response. Disturbance variables with correlation coefficients $|r| \geq 0.7$ were considered redundant. Responsiveness was assessed using visual inspection of biotic metric vs. disturbance gradient scatter plots and linear regression coefficients. Metrics were selected as responsive if they showed either a linear or a wedge-shaped relationship with disturbance gradients. Biological metrics often show a wedge-shaped relationship with single disturbance gradients where the upper boundary represents a threshold of biological response. Multiple limiting factors may result in lower metric values than expected if response were to the single gradient alone. For wedge-shaped relationships, we used a method similar to Blackburn *et al.* (1992) and Rankin & Yoder (1999) to characterize response thresholds. The x-axis was divided into 10 equal categories and the three largest metric values were selected from each category (the three smallest values were selected for negative metrics). Ordinary least-squares regressions were calculated for the subsets of data to estimate the upper bound slopes of wedge-shaped polygons. Metrics that passed the range and responsiveness tests were tested for redundancy. Pairs of metrics with Product-Moment correlation coefficients $|r| \geq 0.7$ were considered redundant and the least responsive metric of the pair was eliminated.

Metrics that were significantly correlated ($p<0.05$) with watershed area were normalized to the mean reference watershed area of $35km^2$ following the method described by Urquart (1982). We calculated the regression equation of the metrics with $\log_{10}$ watershed area in km^2 for reference sites. We then applied that reference regression equation to all sites and calculated their residuals. The predicted metric value for a reference site with the mean watershed area of $35km^2$ was determined and this constant was added to all residuals. The sum of the residual plus the constant at each site resulted in a corrected metric value that was unrelated to watershed area with some sites having negative values.

Scoring ranges were defined for each metric using techniques described in Hughes *et al.* (1998), McCormick *et al.* (2001) and Klemm *et al.* (2003). Metrics were scored on a 0-10 scale using statistical properties of the raw metric values from both reference and non-reference sites to define upper and lower thresholds. For positive metrics (those that increase as disturbance decreases), any site with a metric value equal to or greater than the 80th percentile of reference sites received a score of 10; any site with a metric value equal to or less than the 5th percentile of the non-reference sites received a score of 0; these thresholds were reversed for negative metrics (20th percentile of reference and 95th percentile of non-reference). In both cases, the remaining range of intermediate metric values was divided equally and assigned scores of 1 through 9. Before assembling the B-IBI, we used Kruskal-Wallace tests to determine whether any of the final metrics were significantly different between Klamath, coastal and chaparral reference sites in the northern California coastal region, in which case they would require separate scoring ranges in the B-IBI. Finally, an overall B-IBI score was calculated for each site by summing the constituent metric scores and adjusting the B-IBI to a 100 point scale.

Validation of B-IBI and Measurement of Performance Characteristics

To test whether the distribution of B-IBI scores in reference and test sites might have resulted from chance, we compared score distributions in the development set to those in the validation set. We also investigated a separate performance issue that ambient bioassessment studies often neglect: spatial variation at the reach scale. Although our use of a validation dataset tested whether the B-IBI scoring range is repeatable (Fore *et al*. 1996, McCormick *et al*. 2001), we designed a separate experiment to explicitly measure within-site index precision. In September 2004, we estimated within-site variance in B-IBI scores due to sampling error by taking 3 nested, replicate samples from a single 150m reach at 15 different streams following the USFS protocol. B-IBI scores were then calculated for each replicate. The mean squared error from an ANOVA with site as the independent variable was used as the variance among replicates to calculate the minimum detectable difference (MDD) between two B-IBI scores based on a two-sample *t*-test model (Zar 1999). The index range was divided by the MDD to estimate the number of stream condition categories detectable by the B-IBI (Doberstein *et al*. 2000, Fore *et al*. 2001).

Regional Assessment of Stream Condition

We used the B-IBI in conjunction with WEMAP's probabilistic sampling design and weighted frequency distribution of streams (Herlihy *et. al*. 2000) to estimate the total length of streams in the region achieving a particular condition. We calculated 95% confidence bounds for these measures over the entire study area. A stratified random sampling design was developed wherein each stream segment in EPA's 1:100,000 scale "River Reach File Version 3" (RF3) was given a probability of selection that was roughly inverse to its percent contribution to the total estimated resource population. First order streams were assigned a relatively low probability of selection, whereas larger order streams (fourth order and higher) were assigned a relatively high probability of selection to ensure that the final stratified random sample would contain sample reaches across all stream orders. Each potential sampling site was assigned an associated weight equal to the number of stream kilometers represented by that sample reach.

Results

Reference Sites

Ninety-one sites passed all the land use and local condition screens and were selected as reference sites, leaving 164 sites in the test group (Fig. 1). The development set comprised 190 sites (66 reference/124 test) and the validation set comprised 67 sites (24 reference/43 test).

Selected metrics

Seven non-redundant stressor gradients were selected for metric screening: percent watershed unnatural, percent watershed in agriculture, road density in local watershed, qualitative channel alteration score, percent sand and fine substrates, conductivity and total phosphorous. Thirteen biological metrics failed the range test, 14 metrics were unresponsive to stressor gradients, 22 metrics were redundant with other more responsive metrics ($|r| \geq 0.7$), 5 metrics showed poor discrimination between reference and test sites, and 15 metrics were rejected because they were biologically redundant, but not statistically redundant, with selected metrics (Table 2). A final set of 8 minimally correlated metrics was selected for the B-IBI: EPT richness, Coleoptera richness, Diptera Richness, percent intolerant individuals, percent non-gastropod scraper individuals, percent predator individuals, percent shredder taxa, and percent non-insect taxa (Table 3). All metrics rejected as statistically redundant were derived from taxa similar to those of selected metrics, but had weaker relationships with stressor gradients. Regression coefficients were significant between all 8 selected metrics and at least two stressor gradients: road density in local watershed and percent sand and fine sediment ($p \leq 0.0008$ when a Bonferroni correction for multiple tests is applied; Table 3). The final eight metrics included several metric types: richness, composition, tolerance measures and functional feeding groups.

Six of the final 8 metrics were significantly different between reference sites in the Klamath, northern coast range and chaparral ecoregions (Kruskal-Wallis $p < 0.05$; Fig. 2). We adjusted for these differences by creating separate scoring scales for the six metrics in the three ecoregions (Table 4). The metric percent intolerant individuals was significantly correlated with watershed area ($|r| = -0.371$; $p < 0.0001$) and was adjusted by the following stepwise procedure:

1. The predicted metric at each site (y) = -0.089($\log_{10}$ watershed area) + 0.433
2. The difference (residual) between the observed metric value and the predicted metric value was calculated.
3. The constant 0.296 (the predicted proportion of intolerant individuals at the mean watershed area of 35km^2) was added to each site's residual and multiplied by 100 to convert to percent.

After adjustment, the metric percent intolerant individuals was unrelated to watershed area. Each site's final B-IBI score was multiplied by 1.25 to adjust the scoring range to a 100 point scale.

Validation of B-IBI and Measurement of Performance Characteristics

The distribution of B-IBI scores at reference and non-reference sites was nearly identical between the development and validation data sets (Figure 3), indicating that our characterization of reference conditions and subsequent B-IBI scoring was repeatable and not likely due to chance. Although IBI scores were significantly different between reference and test sites in both the development and validation sets (Mann-Whitney U tests: $p < 0.0001$ and $p = 0.001$, respectively), there was overlap between the reference and test quartiles in both data sets. We speculated that this overlap might be due to the large number of test sites in the Klamath and northern coast ranges that were omitted from the reference pool but were relatively unaffected by human land use. No overlap in quartiles was observed when we compared the reference distributions to sites with ≥ 25% upstream watershed unnatural, indicating that the B-IBI provides good discrimination between reference sites and highly degraded sites (Fig. 3).

Based on a two-sample *t*-test model, setting $\alpha = 0.05$ and $\beta = 0.10$, the MDD for the NorCal IBI is 19.7. Thus, we have a 90% chance of detecting a 19.7 point difference between sites at the $p = 0.05$ level. Dividing the 100-point B-IBI scoring range by the MDD indicates that the NorCal B-IBI can detect approximately 5 biological condition categories, a result similar to other recent estimates of B-IBI precision (Doberstein *et al*. 2000, Fore *et al*. 2001, Ode *et al*. 2005). The B-IBI scoring range can simply be divided into 5 equal categories as follows: 0-20 = “very poor”, 21-40 = “poor”, 41-60 = “fair”, 61-80 = “good” and 81-100 = “very good” (Figure 4). By contrast, a threshold of biological impairment can be established at 2 SDs below the mean reference site score (B-IBI score= 52) with the consequence that some “fair” sites would be considered impaired and others would be considered unimpaired (Fig. 4).

We ran a Principle Components analyses (PCA) on the environmental stressor values used for testing metric responsiveness plus several additional variables that quantified stressor and natural gradients in study watersheds. The PCA was restricted to a subset of 97 sites from which we had data for 13 variables that together defined a multi-factorial axis of watershed condition. Only the first PCA axis was significant, having eigenvalues larger than those predicted from the broken stick model (McCune and Grace 2002). The first PCA axis accounted for 43% of the variance in the environmental data and was highly correlated with B-IBI score ($r = -0.774$, $p < 0.0001$), which decreased with increasing human disturbance (Fig. 4). The axis clearly reflects a water quality, land cover and habitat gradient; percent watershed unnatural and nutrient concentrations had the highest positive loadings, and percent forest in local watershed and qualitative epifaunal substrate score had the highest negative loadings.

Finally, we tested whether our scoring adjustments removed relationships between B-IBI scores and several natural gradients (Fig. 5). We found no significant relationship between reference site B-IBI scores and ecoregion (Kruskal-Wallace test, $p = 0.09$), $\log_{10}$ watershed area ($r^2 = 0.02$, $p = 0.09$), or elevation ($r^2 = 0.007$, $p = 0.53$). There was a significant relationship between B-IBI and Julian date ($r^2 = 0.06$, $p = 0.009$). However, this relationship was driven by two low scoring chaparral reference sites that were sampled early in the year, and was not observed when these two sites were removed from the test ($r^2 = 0.03$, $p = 0.06$). Moreover, the low value of r^2 indicates a weak relationship that is significance only because the regression slope $\neq 0$ (Fig. 5), thus does not indicate that B-IBI score is affected by sampling date.

Regional Assessment of Stream Condition

The total target sampleable, wadeable stream length mapped at a 1:100,000 scale in northern coastal California was estimated to be 7317 km. A total of 7451 km was not assessed because of land-owner denial and physical inaccessibility, but if assumed to be perennial brings the estimated target stream length to 14,768 km (Table 5). Over 50% of the total target stream length was estimated to be in Good condition based on our B-IBI, with 95% confidence intervals ranging between 47% and 72% (Table 6). The second most common condition was Very Good (between 12% and 31%). Between 6% and 27% of the total target stream length was estimated to be in Fair condition, and between 0% and 5% was estimated to be in Poor condition. None of the probability sites were in Very Poor condition.

Discussion

The NorCal B-IBI is the first quantitative index that allows assessment of biological condition of streams in northern coastal California in relation to multiple anthropogenic stressors. Our B-IBI can be a valuable tool for resource management when used to compare biological condition among sites and has enough precision to distinguish 5 categories of biological condition. However, when making regional assessments, it is also necessary to define what IBI scores constitute "acceptable" versus "impaired" conditions. Section 303(d) of the Clean Water Act requires states to list impaired waters, establish total maximum daily loads (TMDLs) for pollutants causing the impairment, and establish plans to rehabilitate those waters. Various options for determining impaired waters have been proposed (e.g. Hughes *et al*. 1998; McCormick *et al*. 2001; Ode *et al*. 2005), but the process is inherently subjective. Because of its widespread statistical acceptance, we set our threshold at 2 SDs below the mean reference site, or a B-IBI score of 52. Using this threshold, only 6% of the mapped, wadeable, sampleable stream length in northern coastal California is impaired (Fig. 6a). Herlihy *et al*. (2005) presented similar results based on a benthic IBI for headwater streams in western Oregon, a region that shares 2 of the 3 ecoregions included in the present study. According to those authors, only 6% of sites in western Oregon were impaired and 62% of sites had no impairment.

We also used a vertebrate (amphibians + fish) IBI recently developed for coldwater streams of western Oregon and Washington (Hughes *et al*. 2004) to score vertebrate assemblages collected at north coast probability sites. Use of this IBI is appropriate for northern coastal California since two of the ecoregions (Klamath Mtns. and Northern Coastal Ranges) in our study area also occur in western Oregon, and since sampling protocols used at northern California sites were similar or identical to those used in

Oregon and Washington. Using the same threshold of impairment for the vertebrate IBI (2 SDs below mean of reference), 7.5% of the mapped, wadeable, sampleable stream length in northern coastal California is impaired (Fig. 6b). By contrast, Hughes *et al*. (2004) found that 45% of stream kilometers in western Oregon and Washington were impaired based on the vertebrate IBI.

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Table 1. List of minimum or maximum land use thresholds used for rejecting potential reference sites.

Stressor	Threshold
Percentage of unnatural land use at the local scale	> 5%
Percental of urban land use at the local scale	> 3%
Percentage of total agriculture at the local scale	> 5%
Road density at the local scale	> 1.5 km/km^2
Population density (2000 census) at the local scale	> 25 ind./ km^2
Percentage of unnatural land use at the watershed scale	> 5%
Percentage of urban land use at the watershed scale	> 3%
Percentage of total agriculture at the watershed scale	> 5 %
Road density at the watershed scale	> 2.0 km/km^2
Population density (2000 census) at the watershed scale	> 50 ind./ km^2

Table 2. Seventy-seven metrics screened for used in the NorCal B-IBI. Metrics that failed the range test are marked with an asterisk (*); metrics that failed the responsiveness test ($r^2 < 0.2$, $p > 0.05$) are in italics; metrics that were adopted for use in the B-IBI are in bold. Poor discrimination between reference and test sites, statistical redundancy and biological redundancy with selected metrics are indicated in the second column.

Metric	Redundancy/Discrimination
CF + CG Richness	poor
Coleoptera Richness	
Collector Filterer Richness*	
Collector Gatherer Richness	poor
Diptera Richness	
Elmidae Richness*	
Ephemerellidae Richness*	
Ephemeroptera Richness	EPT Richness
EPT Richness	
Hydropsychidae Richness*	
Intolerant EPT Richness	
Intolerant Richness	EPT Richness
Mollusca Richness*	
Non-insect Richness	
Crustacea + Mollusca Richness*	
Plecoptera Richness	EPT Richness
Predator Richness	EPT Richness
Scraper Richness	EPT Richness
Shredder Richness	EPT Richness

Table 2. Continued.

Metric	**Redundancy/Discrimination**
Taxonomic Richness	EPT Richness
Trichoptera Richness	EPT Richness
% Baetidae Individuals	
% CF + CG Individuals	% Intolerant Individuals
% CF + CG Taxa	poor
% CF Taxa	
% CG Taxa	% Intolerant Individuals
% Chironomidae Individuals	
% Collector-Filterer Individuals	poor
% Collectors Gatherer Individuals	% Non-Gastropod Scraper Individuals
% Corbicula Individuals *	
% Crustacea Individuals	
% Diptera Individuals	
% Diptera Taxa	
% Dominant Taxon	poor
% Elmidae Individuals	biologically redundant
% Ephemeroptera Individuals	biologically redundant
% Ephemeroptera Taxa	biologically redundant
% EPT Individuals	% Intolerant Individuals
% EPT Taxa	EPT Richness
% Gastropoda Individuals	
% Glossosomatidae Individuals *	
% Hydropsychidae Individuals	biologically redundant
% Hydroptilidae Individuals *	
% Intolerant Individuals	
% Intolerant Diptera Individuals *	
% Intolerant Ephemeroptera Individuals	biologically redundant
% Intolerant Scraper Individuals	biologically redundant
% Intolerant Taxa	EPT Richness
% Intolerant Trichoptera Individuals	% Intolerant Individuals
% Mollusca Individuals	
% Non Baetis Fallceon Ephemeroptera Individuals	biologically redundant
% Non Hydro Cheumato Trichoptera Individuals	biologically redundant
% Non-Gastropoda Scraper Individuals	
% Non-Hydropsyche Hydropsychidae Individuals *	
% Non-Insecta Taxa	
% of Ephemeroptera Individuals that are Intolerant	biologically redundant
% of Trichoptera Individuals that are Intolerant	biologically redundant
% Oligochaeta Individuals	
% Perlodidae Individuals *	
% Philopotamidae Individuals *	

Table 2. Continued.

Metric	**Redundancy/Discrimination**
% Plecoptera Individuals	biologically redundant
% Plecoptera Taxa	biologically redundant
% Predator Taxa	
% Predator Individuals	
% Rhyacophildae Individuals	biologically redundant
% Scraper Taxa	
% Scraper Individuals	% Non-Gastropod Scraper Individuals
% Sensitive EPT Individuals	% Intolerant Individuals
% Shredder Taxa	
% Shredder Individuals	biologically redundant
% Simuliidae Individuals	
% Tolerant Individuals	
% Tolerant Taxa	% Non-Insect Taxa
% Trichoptera Individuals	biologically redundant
% Trichoptera Taxa	% Non-Insect Taxa
Shannon Diversity	EPT Richness
Tolerance Value	EPT Richness

Table 3. Values of r^2 from mean and upper bound regressions between metrics adopted for use in the NorCal B-IBI and stressor gradients used in metric screening. Significant values ($p < 0.0008$ after Bonferroni correction) are indicated in bold.

Stressor	**EPT Richness**	**EPT Richness upper bound**	**Coleoptera Richness**	**Coleoptera Richness upper bound**	**Diptera Richness**	**Diptera Richness upper bound**	**% Intolerant Individuals**	**% Intolerant Individuals upper bound**	**% Non-Gastropod Scraper Individuals**	**% Non-Gastropod Scraper Individuals upper bound**	**% Predator Individuals**	**% Predator Individuals upper bound**	**% Shredder Taxa**	**% Shredder Taxa upper bound**	**% Non-Insect Taxa**	**% Non-Insect Taxa upper bound**
% of watershed unnatural	**-0.366**	**-0.702**	**-0.17**	-0.235	**-0.172**	**-0.499**	**-0.231**	**-0.618**	**-0.157**	**-0.417**	**-0.087**	-0.191	**-0.121**	**-0.468**	**0.473**	**0.826**
% of watershed in agriculture	**-0.137**	-0.328	**-0.078**	-0.238	**-0.064**	-0.053	**-0.062**	-0.296	**-0.105**	-0.291	**-0.045**	**-0.476**	-0.021	-0.019	**0.097**	0.197
road density in local watershed	**-0.266**	**-0.677**	**-0.119**	**-0.689**	**-0.171**	**-0.697**	**-0.19**	**-0.722**	**-0.116**	**-0.656**	**-0.106**	**-0.593**	**-0.112**	**-0.597**	**0.297**	**0.51**
% sand and fine substrates	**-0.341**	**-0.793**	**-0.189**	**-0.665**	**-0.169**	**-0.658**	**-0.163**	**-0.744**	**-0.108**	**-0.653**	**-0.082**	**-0.629**	**-0.09**	**-0.513**	**0.452**	**0.707**
conductivity	**-0.263**	**-0.467**	0.005	-0.128	-0.028	-0.152	**-0.189**	-0.239	**-0.093**	**-0.457**	**-0.17**	**-0.517**	**-0.108**	-0.355	**0.274**	0.249
total phosphorous	**-0.332**	**-0.629**	**-0.159**	-0.518	-0.114	-0.343	**-0.173**	-0.488	-0.114	-0.438	**-0.203**	-0.413	-0.108	-0.603	0.107	0.472
qualitative channel alteration	**0.133**	**0.807**	0.053	**0.750**	**0.09**	**0.803**	0.071	**0.644**	**0.079**	**0.633**	0.053	**0.647**	0.063	**0.7**	**-0.171**	**-0.629**

Table 4. Scoring ranges for 8 component metrics in the NorCal B-IBI. Six metrics have separate scoring ranges for the three Omernik Level III ecoregions in northern coastal California region (1= Coast Ranges, 6=Chaparral and Oak Woodlands, 78=Klamath Mountains).

Metric Score	EPT Richness		Coleoptera Richness		Diptera Richness	% Intolerant Individuals		% Non-Gastropoda Scraper Individuals		% Predator Individuals		% Shredder Taxa		% Non-Insect Taxa
	1 & 78	**6**	**1 & 78**	**6**	**All Sites**	**1 & 78**	**6**	**1**	**6 & 78**	**78**	**1 & 6**	**1**	**6& 78**	**All Sites**
10	>25	>20	≥6	≥8	≥10	≥41	≥28	≥41	≥18	≥22	≥16	≥20	≥16	0-7
9	23-25	19-20	5	7	9	36-40	24-27	37-40	17	19-21	14-15	18-19	14-15	8-13
8	21-22	17-18		6	8	31-35	21-23	33-36	15-16	17-18	12-13	16-17	12-13	14-18
7	18-20	15-16	4		7	26-30	17-20	29-32	13-14	15-16	11	14-15	11	19-24
6	16-17	13-14		5	6	21-25	14-16	25-28	11-12	13-14	9-10	12-13	9-10	25-29
5	13-15	11-12	3	4	5	16-20	10-13	21-24	9-10	10-12	8	10-11	8	30-35
4	11-12	9-10		3	4	11-15	7-9	17-20	7-8	8-9	6-7	8-9	6-7	36-40
3	8-10	7-8	2		3	6-10	3-6	13-16	5-6	6-7	5	6-7	5	41-46
2	6-7	5-6		2	2	1-5	0-2	9-12	3-4	4-5	3-4	4-5	3-4	47-51
1	3-5	3-4	1	1	1	-4 to 0	-4 to -1	5-8	1-2	2-3	2	2-3	2	52-56
0	0-2	0-2	0	0	0	≤-5	≤-5	0-4	0	0-1	0-1	0-1	0-1	≥57

Table 5. Estimated percentage and length of stream kilometers in each evaluation category.

Status	n	Estimated % of stream km (95% confidence interval)	Length of stream km
Landowner Denied	33	20.0 ± 5.4	4529
Non-Target	40	29.5 ± 6.4	6668
Physical Barrier	21	12.9 ± 4.4	2922
Target Sampled	59	32.4 ± 6.3	7317

Table 6. Estimated percentage and length of stream kilometers in each condition category based on B-IBI.

Condition Category	n	Estimated % of stream km (95% confidence interval)	Length of stream km
Very Poor	0		0
Poor	4	2.1 ± 2.6	154
Fair	10	16.8 ± 10.5	1227
Good	31	59.7 ± 12.4	4369
Very Good	14	21.4 ± 9.8	1569

Figure 1. Map of study area and Omernik Level III ecoregions.

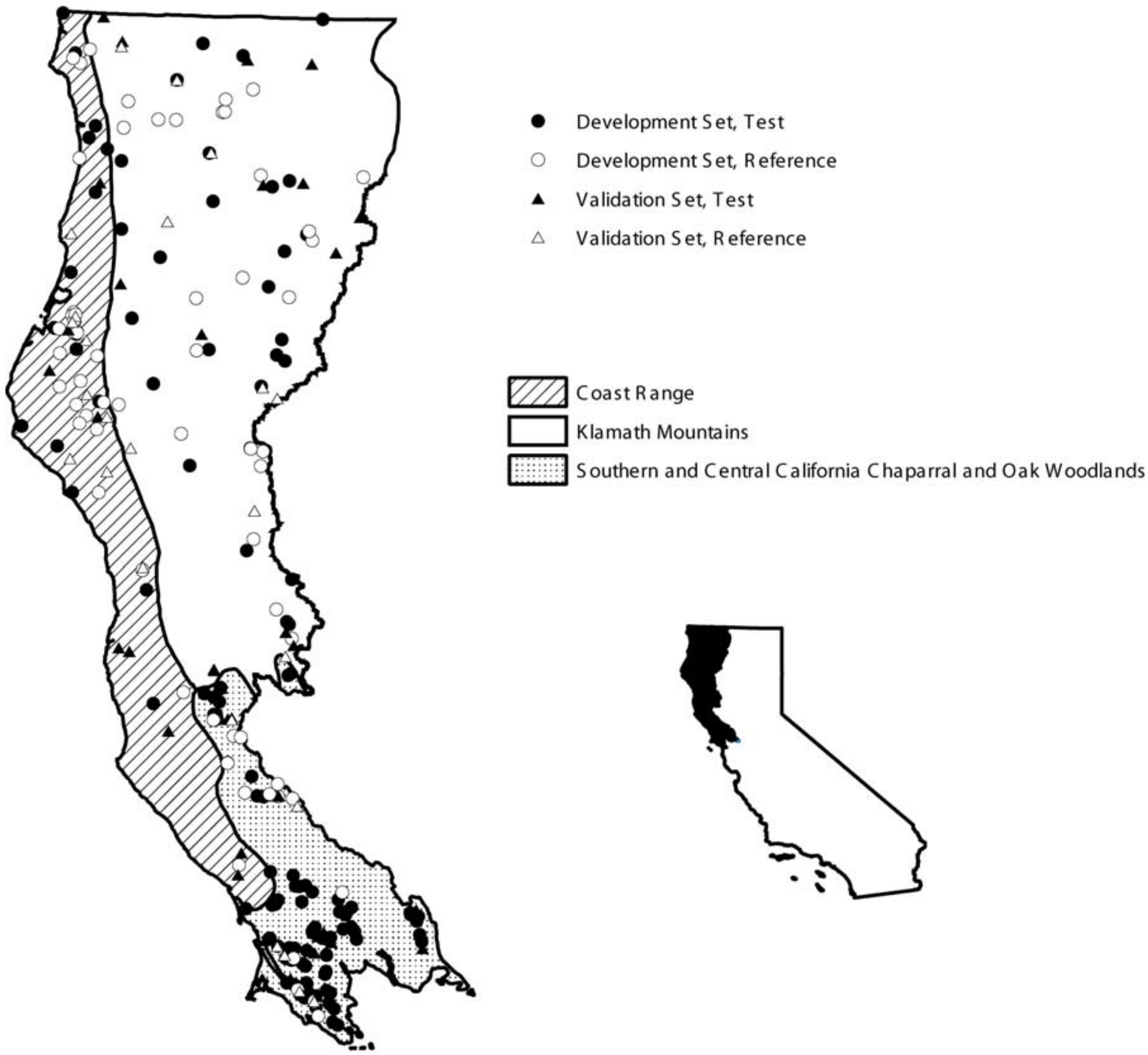


Figure 2. Box plots of metric distributions in reference sites in each Omernik Level III ecoregion in northern coastal California. Separate scoring scales were developed for metrics that differed significantly (Kruskal-Wallis p <0.05) between ecoregions.

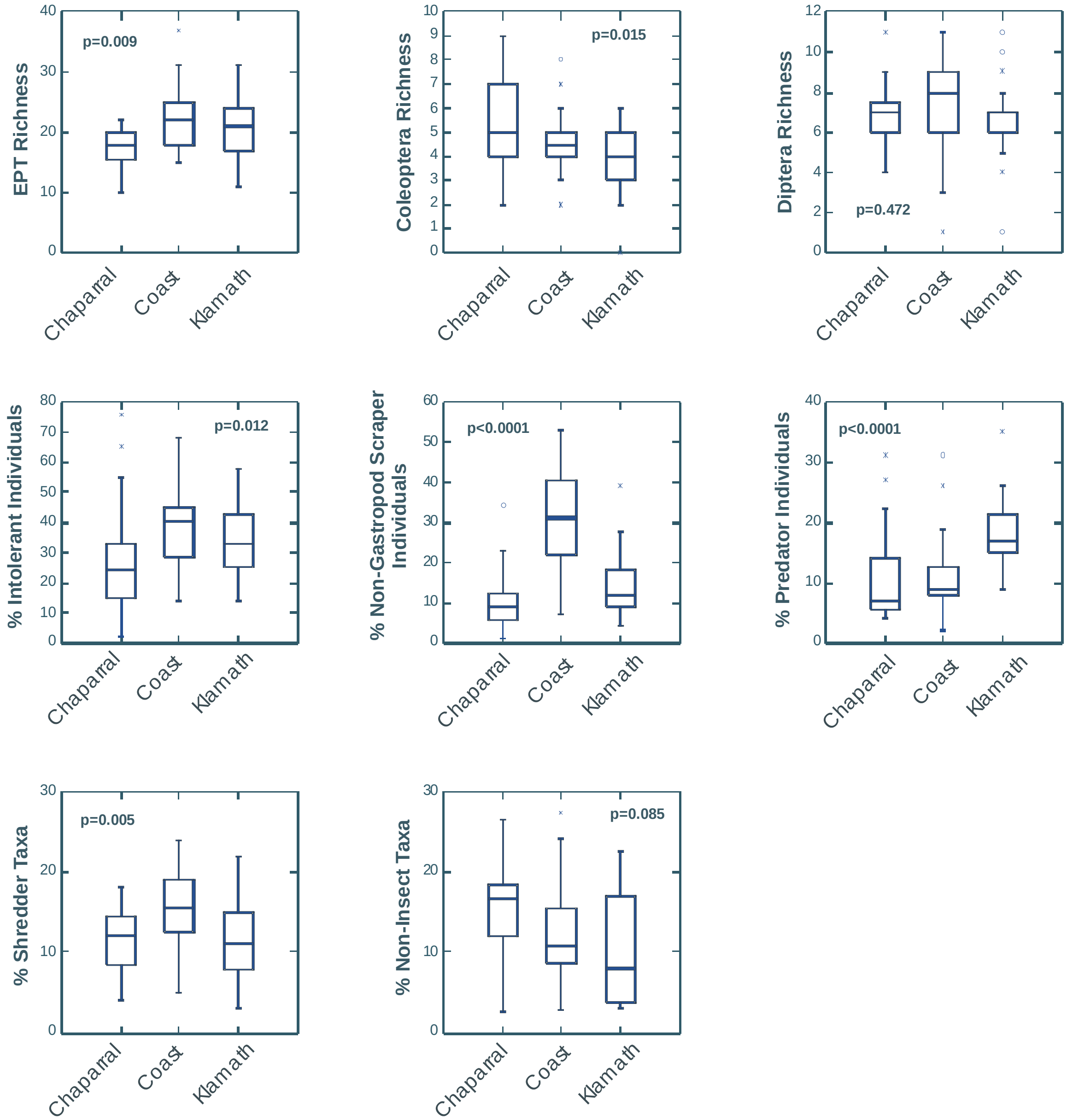


Figure 3. Box plots of B-IBI scores for reference (R), test (T) and "worst" (W) groups in development and validation data sets. "Worst" sites have ≥ 25% upstream watershed in unnatural land use (agriculture and urban). Dotted lines indicate condition category boundaries.

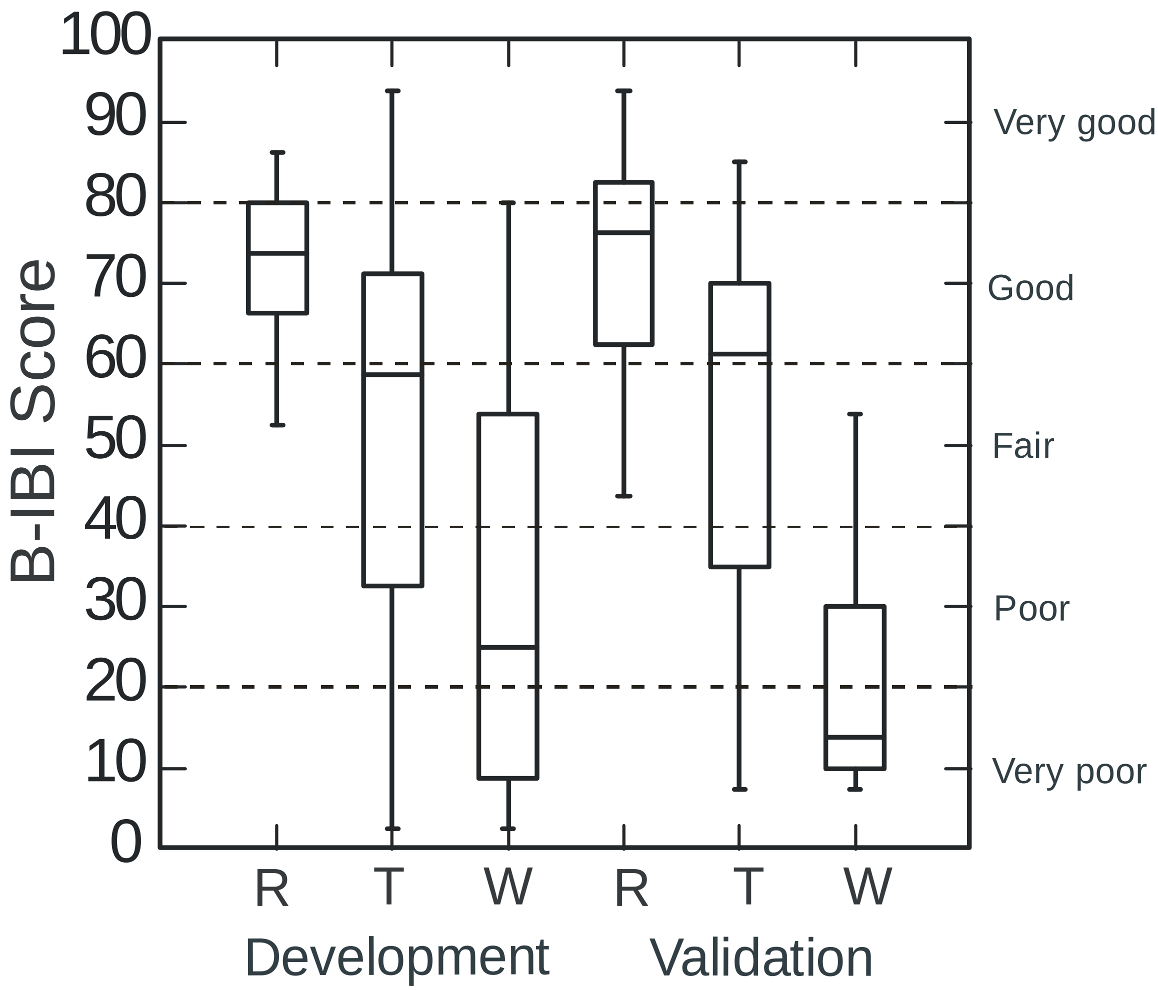
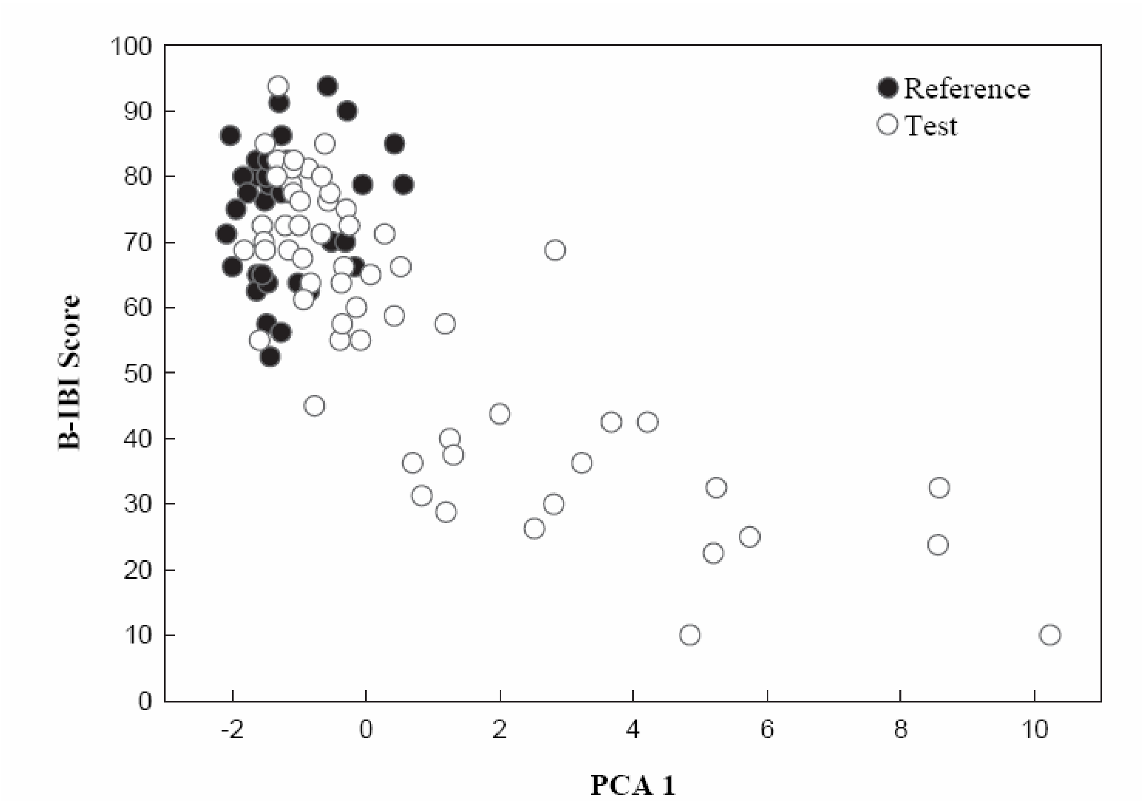


Figure 4. B-IBI score as a function of a multivariate watershed condition axis (PCA 1). Pearson correlation = -0.774, $p < 0.0001$.



Axis 1 loadings

% watershed unnatural	0.368
% watershed in agriculture	0.344
% local watershed forested	-0.336
% local watershed in agriculture	0.229
road density in local watershed	0.224
population density in local watershed	0.210
mid-channel canopy density	-0.117
% sand and fine sediment	0.252
conductivity	0.281
total phosporous	0.336
chloride	0.352

Figure 5. Relationship between B-IBI scores at 91 reference sites and (a) Omernik Level III ecoregion, (b) Julian date (day of year), (c) $\log_{10}$ watershed area, and (d) elevation.

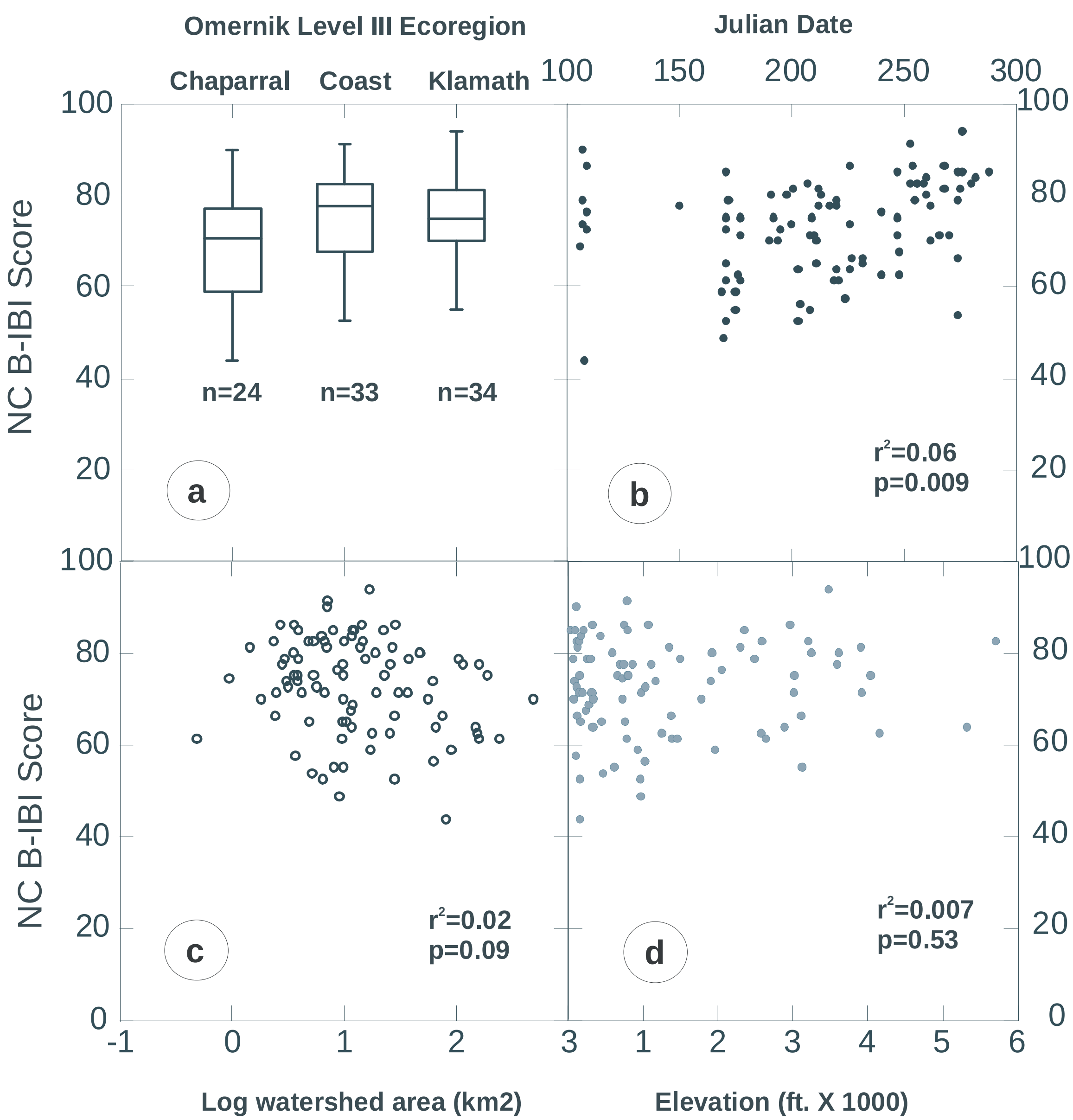
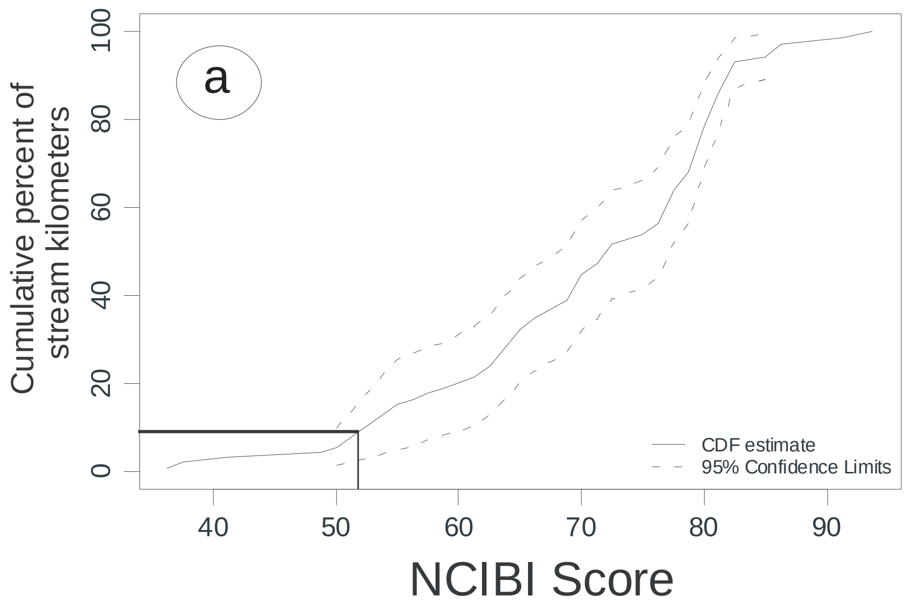
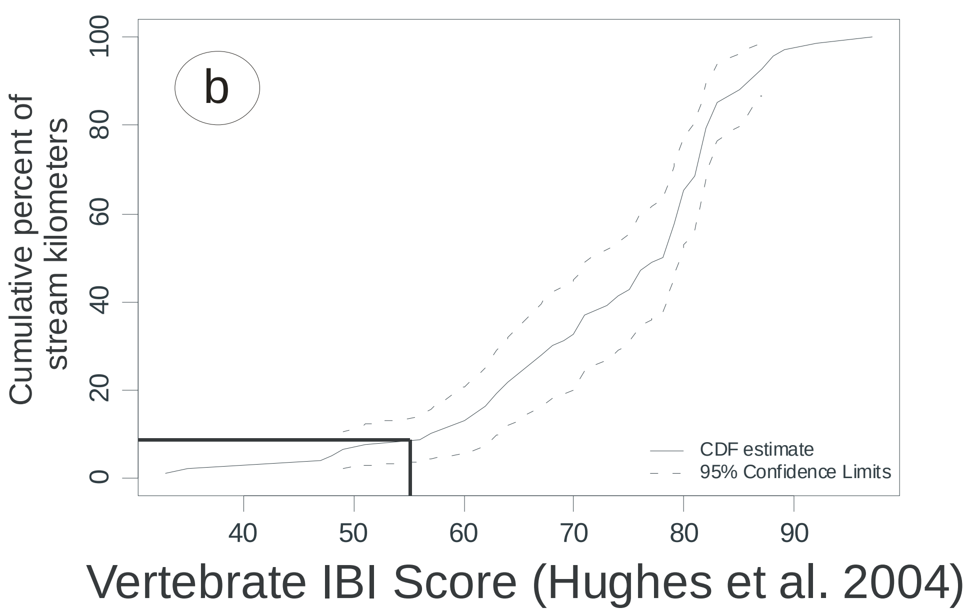


Figure 6. Cumulative distribution function of B-IBI scores (a) and vertebrate IBI scores (b) estimated from 59 probability sites in northern coastal California. Impairment thresholds 2 SDs below the mean reference score (B-IBI =52; vertebrate IBI=55) are indicated.





ADDENDUM TO THE NORTH COAST IBI
December 11, 2008
Andrew C. Rehn, Peter R. Ode and Jason T. May

In March 2006, after reviewing a draft technical report entitled *Development of a Benthic Index of Biotic Integrity (B-IBI) for Wadeable Streams in Northern Coastal California and its Application to Regional 305(b) Assessment* (Rehn et al. 2005), the North Coast Water Resources Control Board (NCWRCB) identified two potential limitations to the applicability of the index in regional monitoring programs:

1) Timber harvest was not included as an explicit stressor gradient in IBI development, but is a predominant land use in north coast watersheds. The NCWRCB expressed the need to evaluate the sensitivity of the IBI to timber harvest, not just urbanization and agriculture as had been done in the original draft.

2) Reference conditions may have been defined inappropriately because road density thresholds were based on incomplete GIS layers. Specifically, the road layers that were used in the first report did not include small dirt roads associated with timber harvest practices. The NCWRCB was concerned that reference criteria in the original report therefore set the bar too low for regional streams, and that the north coast stream condition assessment underestimated the extent of impaired stream miles.

The purpose of this addendum is to address the concerns expressed by the NCWRCB with supplemental data analyses and thereby finalize the North Coast IBI.

Timber harvest

To address the first concern, we used remote sensing GIS layers of vegetative change developed by the California Land Cover Mapping and Monitoring Program (LCMMP) for north coast watersheds (Levien et al. 2003). Vegetative change was quantified over two time periods: 1993-1998 and 1998-2003. The NCIBI boundary spanned two different LCMMP analysis areas ("North Coast" and "North East"), so we combined across remote sensing cycles to match remote sensing dates appropriately (e.g., we combined North Coast Cycle 2 with North East Cycle 1 for the 1993-1998 time period in our analysis). The cause of vegetative change was attributed on a pixel-by-pixel basis at 30 m resolution to five potential sources (wildfire, timber harvest, pest infestation, development and 'other') using ancillary data sets such as fire perimeter and severity maps, timber harvest plans, aerial photo interpretation, ground truthing, etc. Canopy cover decrease (or increase) was divided into 4 change classes in the original vegetative change layers: 71-100% decrease, 41-70% decrease, 16-40% decrease, and little or no change (15% increase to 15% decrease). Readers should refer to Levien et al. (2003) for complete information on remote sensing vegetation layers, change labeling and cause verification.

We used the ArcView 3.x extension ATtILA (Ebert and Wade 2004) to quantify the extent of canopy decrease due to timber harvest at the same spatial scales used for land use evaluations in the original report: within polygons delimiting the entire watershed upstream of each sampling site and within polygons representing local regions defined as the intersection of a 1km radius circle around each site and the primary watershed polygon. Total harvest within each spatial scale was calculated as the weighted sum of decrease in each change class using change class midpoints as weighting values. For example, if a watershed had 10% decrease in the 71-100% class, 10% decrease in the 41-71% class, and 10% decrease in the 16-40% class, total harvest for the watershed was: (10*0.85) + (10*0.55) + (10*0.28) = 16.8%. The "little or no change" class was not used in calculating total watershed harvest. Change (decrease) due to harvest was summed for the two 5-year time periods.

We then evaluated the response of individual BMI metrics and the North Coast IBI to regional timber harvest gradients. In general, individual metrics were weakly correlated with timber harvest in north coast watersheds (Table 1): 'positive metrics' (such as EPT richness which increases as human disturbance decreases) were positively correlated with timber harvest, and 'negative metrics' (such as % collector individuals which decreases as human disturbance decreases) were negatively correlated with timber harvest. IBI scores were positively correlated with timber harvest, but the relationship is not linear (Fig. 1), and the timber harvest gradient was incompletely captured by this data set, i.e., most watersheds had 0% to 10% total harvest. If anything, a small amount of timber harvest in a watershed has a positive effect on BMI metrics and IBI scores.

Roads

To improve assessment of relationships between road densities and BMI responses, we created a custom road layer for all the watersheds in our analyses. Using the original road data as a base layer, we added additional linear features from two sources: 1) road layers reported in North Coast timber harvest plans (THP, available from California Department of Fire) and 2) road layers reported for US Forest Service lands available from the USFS GIS Clearinghouse. The USFS road layers covered all USFS lands in the North Coast region, while the THP roads were just included for the areas defined by the watersheds in our analyses. Both of these data sources include logging roads and other small access roads that weren't available in our original analysis.

Reference conditions

To address the second concern, we used an updated land cover data set (National Land Cover Data Set, NLCD 2001) and the custom road layer (see "Roads" section above) to re-screen north coast reference sites. The same land use thresholds outlined in the original report (Rehn et al. 2005) were used to re-screen sites. Additional riparian disturbance and water chemistry thresholds defined by Stoddard et al. (2005) for the western EMAP pilot also were used whenever appropriate data were available. The net result was a decrease in the total north coast reference pool from 91 to 61 sites; the Coast Range ecoregion (Omernik 1987) suffered the largest decrease in reference sites (from 33 to 9). Most sites were dropped from the reference pool because they no longer passed road density thresholds.

To evaluate whether this shift in reference sites would affect IBI scoring, we compared IBI scores and raw values of the 8 component metrics between the original and new reference pools for each of the 3 ecoregions covered in the original report. In general, IBI scores and raw metric values did not differ between original and new reference pools for each ecoregion, i.e., the quartiles of the original and new distributions showed considerable overlap (Fig. 2). Notable exceptions were: 1) percent shredder taxa in the Coast Range ecoregion, which was actually higher in the original reference distribution; 2) percent non-insect taxa in the Chaparral ecoregion, which is higher in the new reference distribution. We concluded from these observations that the original scoring for percent shredder taxa in the Coast Ranges sets the bar higher for this metric than if we were to re-score based on the new reference distribution, and that an adjusted scoring for percent non-insect taxa in the Chaparral would have a negligible overall effect on the IBI.

Revised Tables and Figures

Table 3 from the original report (presented here as Table 2) has been revised to include timber harvest as a stressor gradient and to show regression coefficients between the final 8 metrics and updated road densities. Figure 4 from the original report (presented here as Figure 3) has been revised to include timber harvest as part of a multivariate watershed condition axis (defined by Principle Components Analysis).

Conclusions

The North Coast IBI and its component metrics had a slight positive correlation with timber harvest in regional watersheds, but most watersheds evaluated had little total harvest (between 0% and 10%). Full characterization of stream benthos response to timber harvest will require targeted sampling of complete timber harvest gradients so that streams with recent and intensive clearcutting, or with > 50% total harvest in the watershed, are included.

The regional reference pool decreased substantially (from 91 to 61 sites) when updated land use and road density GIS layers were used in reference screening, mostly because fewer sites passed road density screens. However, the decreased reference pool had little effect on BMI metric or IBI scoring distributions in any of the three ecoregions. We conclude that the current IBI can therefore be utilized as a general indicator of ecological condition in north coast streams.

Acknowledgements

We wish to thank staff at the USFS and CDF for assistance with obtaining the GIS layers used in these analyses and staff at TetraTech and the CSU Chico Geographic Information Center for help in compiling the final datasets. Lisa Fischer and Carlos Ramirez provided the LCMMP data, Ralph Warbington provided access to the USFS road layers, Suzanne Lang provided THP road data, Martin Hurd and Jeff Sturmann helped merge the THP road records and Erik Fintel and Jason Schwenkler helped prepare the LCMMP layers for analysis.

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Table 1. Pearson correlations between percent timber harvest at watershed and local (1k) scales, north coast IBI scores, and BMI metrics. Metrics used in the IBI and correlation coefficients significant at $p < 0.05$ are shown in bold. $N = 257$.

	Watershed timber harvest	Local (1k) timber harvest
Watershed timber harvest	1.00	
Local (1k) timber harvest	0.73	1.00
IBI Score	0.22	0.19
Number EPT Taxa	**0.31**	**0.29**
Number Collector Filterer + Collector Gatherer Taxa	0.03	0.02
Number Coleoptera Taxa	0.17	0.08
Number Collector Filterer Taxa	-0.06	-0.10
Number Collector Gatherer Taxa	0.06	0.06
Number Diptera Taxa	0.02	0.07
Number Elmidae Taxa	0.05	-0.05
Number Ephemerellidae Taxa	0.10	0.02
Number Ephemeroptera Taxa	0.26	0.23
Number Hydropsychidae Taxa	-0.07	-0.05
Number Intolerant EPT Taxa	**0.33**	**0.32**
Number Intolerant Taxa	**0.30**	0.27
Number Mollusca Taxa	-0.02	-0.08
Number Non-insect Taxa	0.01	0.00
Number of Crustacea + Mollusca Taxa	-0.04	-0.03
Number Plecoptera Taxa	**0.31**	0.26
Number Predator Taxa	0.18	0.17
Number Scraper Taxa	0.20	0.17
Number Shredder Taxa	**0.43**	**0.41**
Number Trichoptera Taxa	0.20	0.23
Percent Baetidae Individuals	-0.20	-0.20
Percent Collector Filterer + Collector Gatherer Individuals	**-0.31**	-0.24
Percent Collector Filterer + Collector Gatherer Taxa	-0.24	-0.22
Percent Collector Filterer Taxa	-0.18	-0.19
Percent Collector Gatherer Taxa	-0.20	-0.17
Percent Chironomidae Individuals	-0.08	-0.03
Percent Collector-Filterer Individuals	-0.16	-0.16
Percent Collectors Gatherer Individuals	-0.25	-0.18
Percent Crustacea Individuals	0.02	0.07
Percent Diptera Individuals	-0.11	-0.06
Percent Diptera Taxa	-0.19	-0.14
Percent Dominant Taxon	-0.11	-0.11
Percent Elmidae Individuals	0.16	0.15
Percent Ephemeroptera Individuals	0.02	-0.05
Percent Ephemeroptera Taxa	0.09	0.07
Percent EPT Individuals	0.12	0.07
Percent EPT Taxa	0.23	0.21
Percent Gastropoda Individuals	0.00	-0.04

Table 1 continued.

	Watershed timber harvest	Local (1k) timber harvest
Percent Glossosomatidae Individuals	0.04	0.10
Percent Hydropsychidae Individuals	-0.09	-0.06
Percent Hydroptilidae Individuals	-0.09	-0.05
Percent Intolerant Individuals	0.23	0.20
Percent Intolerant Diptera Individuals	0.12	0.17
Percent Intolerant Ephemeroptera Individuals	0.07	0.01
Percent Intolerant Scraper Individuals	0.13	0.12
Percent Intolerant Taxa	0.25	0.20
Percent Intolerant Trichoptera Individuals	0.13	0.13
Percent Mollusca Individuals	0.00	-0.04
Percent Non-*Baetis Fallceon* Ephemeroptera Individuals	0.19	0.10
Percent Non-*Hydropsyche Cheumatopsyche* Trichoptera Individuals	0.10	0.10
Percent Non-Gastropoda Scraper Individuals	0.27	0.21
Percent Non-*Hydropsyche* Hydropsychidae Individuals	-0.02	0.01
Percent Non-Insecta Taxa	-0.13	-0.13
Percent of Ephemeroptera that are Intolerant	0.15	0.14
Percent of Trichoptera that are Intolerant	0.23	0.21
Percent Oligochaeta Individuals	-0.12	-0.10
Percent Perlodidae Individuals	0.13	0.10
Percent Philopotamidae Individuals	-0.05	-0.03
Percent Plecoptera Individuals	0.23	0.21
Percent Plecoptera Taxa	0.24	0.18
Percent Predator Taxa	-0.01	0.00
Percent Predator Individuals	0.08	0.05
Percent Rhyacophildae Individuals	0.11	-0.01
Percent Scraper Taxa	0.11	0.07
Percent Scraper Individuals	0.26	0.19
Percent Shredder Taxa	**0.39**	**0.36**
Percent Shredder Individuals	0.21	0.17
Percent Simuliidae Individuals	-0.11	-0.13
Percent Tolerant Individuals	-0.07	-0.07
Percent Tolerant Taxa	-0.15	-0.11
Percent Trichoptera Individuals	0.02	0.05
Percent Trichoptera Taxa	0.11	0.14
Percent Sensitive EPT Individuals	0.23	0.19
Shannon Diversity	0.16	0.14
Taxonomic Richness	0.24	0.23
Tolerance Value	-0.21	-0.17

Table 2. Values of r^2 from mean and upper bound regressions between metrics adopted for use in the NorCal B-IBI and stressor gradients used in metric screening. Significant values ($p < 0.0001$ after Bonferroni correction) are indicated in bold. Compare with Table 3 in original report.

Stressor	EPT Richness	EPT Richness upper bound	Coleoptera Richness	Coleoptera Richness upper bound	Diptera Richness	Diptera Richness upper bound	% Intolerant Individuals	% Intolerant Individuals upper bound	% Non-Gastropod Scraper Individuals	% Non-Gastropod Scraper Individuals upper bound	% Predator Individuals	% Predator Individuals upper bound	% Shredder Taxa	% Shredder Taxa upper bound	% Non-Insect Taxa	% Non-Insect Taxa upper bound
% of watershed unnatural	**-0.366**	**-0.702**	**-0.17**	-0.235	**-0.172**	**-0.499**	**-0.231**	**-0.618**	**-0.157**	**-0.417**	**-0.087**	-0.191	**-0.121**	**-0.468**	**0.473**	**0.826**
% of watershed in agriculture	**-0.137**	-0.328	**-0.078**	-0.238	**-0.064**	-0.053	**-0.062**	-0.296	**-0.105**	-0.291	**-0.045**	**-0.476**	-0.021	-0.019	**0.097**	0.197
road density in local watershed	**-0.112**	**-0.70**	-0.05	**-0.82**	**-0.11**	**-0.80**	**-0.09**	**-0.76**	-0.005	**-0.61**	**-0.08**	**-0.74**	-0.02	**-0.57**	**0.20**	**n/a**
% sand and fine substrates	**-0.341**	**-0.793**	**-0.189**	**-0.665**	**-0.169**	**-0.658**	**-0.163**	**-0.744**	**-0.108**	**-0.653**	**-0.082**	**-0.629**	**-0.09**	**-0.513**	**0.452**	**0.707**
conductivity	**-0.263**	**-0.467**	0.005	-0.128	-0.028	-0.152	**-0.189**	-0.239	**-0.093**	**-0.457**	**-0.17**	**-0.517**	**-0.108**	-0.355	**0.274**	0.249
total phosphorous	**-0.332**	**-0.629**	**-0.159**	-0.518	-0.114	-0.343	**-0.173**	-0.488	-0.114	-0.438	**-0.203**	-0.413	-0.108	-0.603	0.107	0.472
qualitative channel alteration	**0.133**	**0.807**	0.053	**0.750**	**0.09**	**0.803**	0.071	**0.644**	**0.079**	**0.633**	0.053	**0.647**	0.063	**0.7**	**-0.171**	**-0.629**
% timber harvest in full watershed	0.077	--	0.004	--	0.005	--	0.03	--	**0.108**	--	0.024	--	**0.141**	--	0.004	--
% timber harvest in local (1k) watershed	0.073	--	0.005	--	0.005	--	0.038	--	**0.105**	--	0.010	--	**0.108**	--	0.006	--

Figure 1. Scatterplots of North Coast IBI score vs. percent timber harvest at watershed and local (1k) scales.

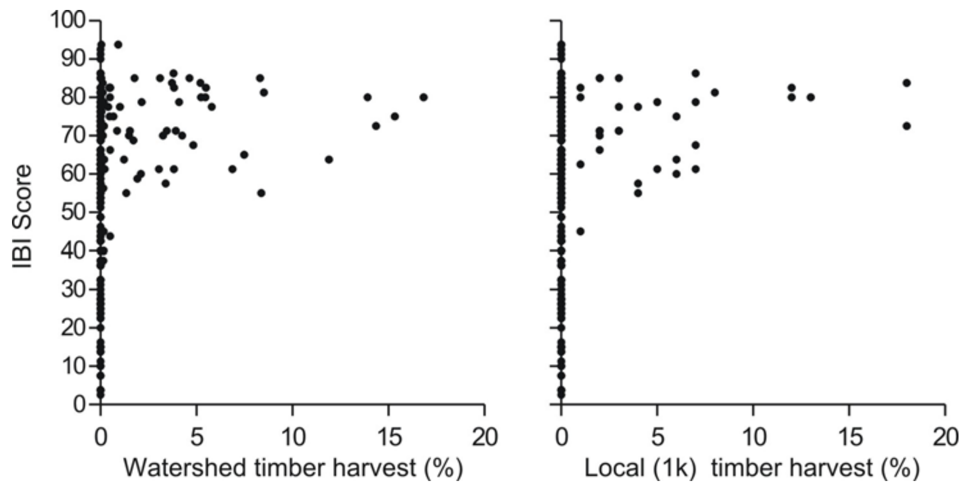


Figure 2. Box plots showing the distributions of IBI scores and the eight component IBI metrics for original and new pools of reference sites in each of the three ecoregions covered by the IBI. Sample sizes are indicated above boxes in the first plot.

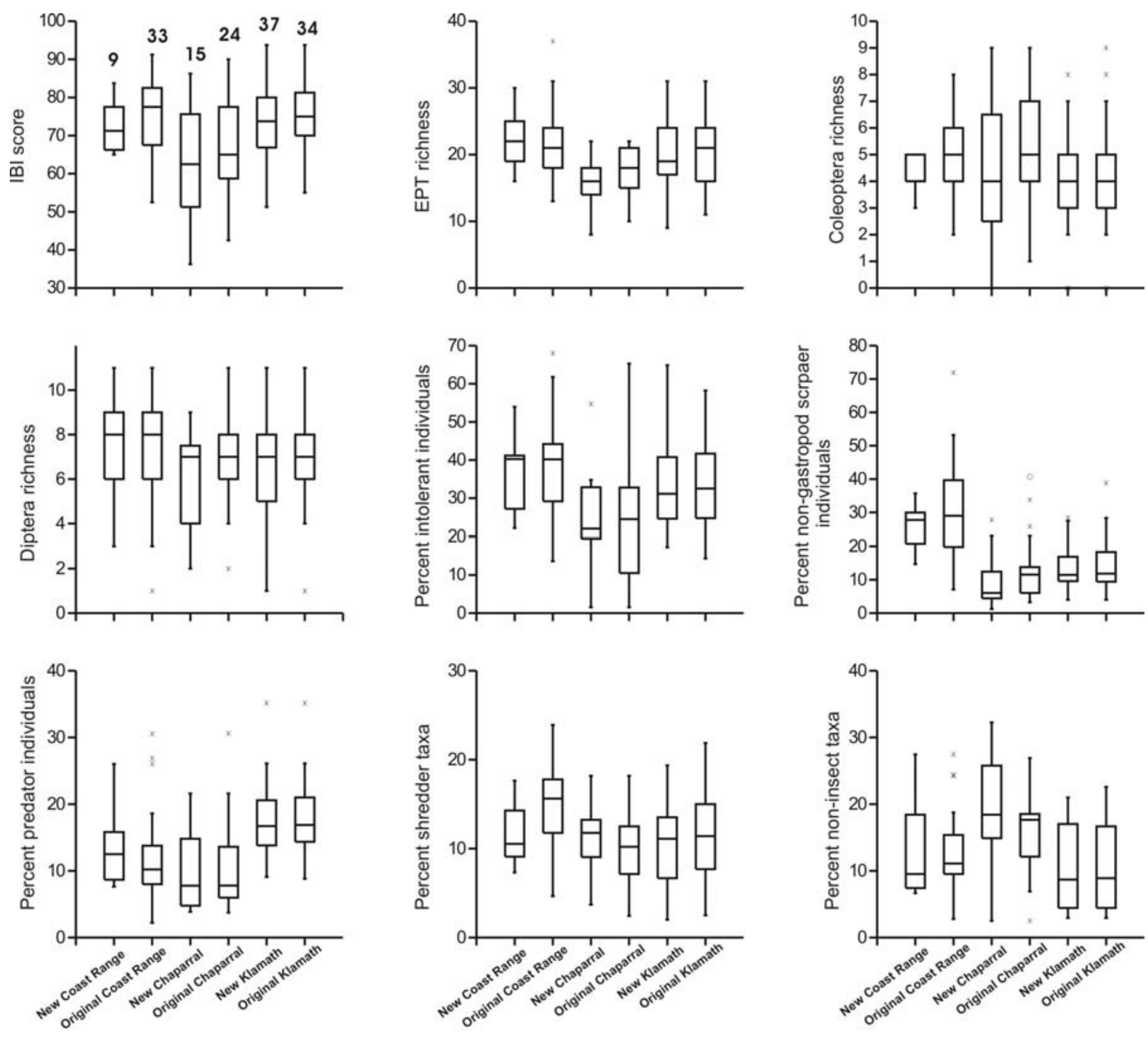
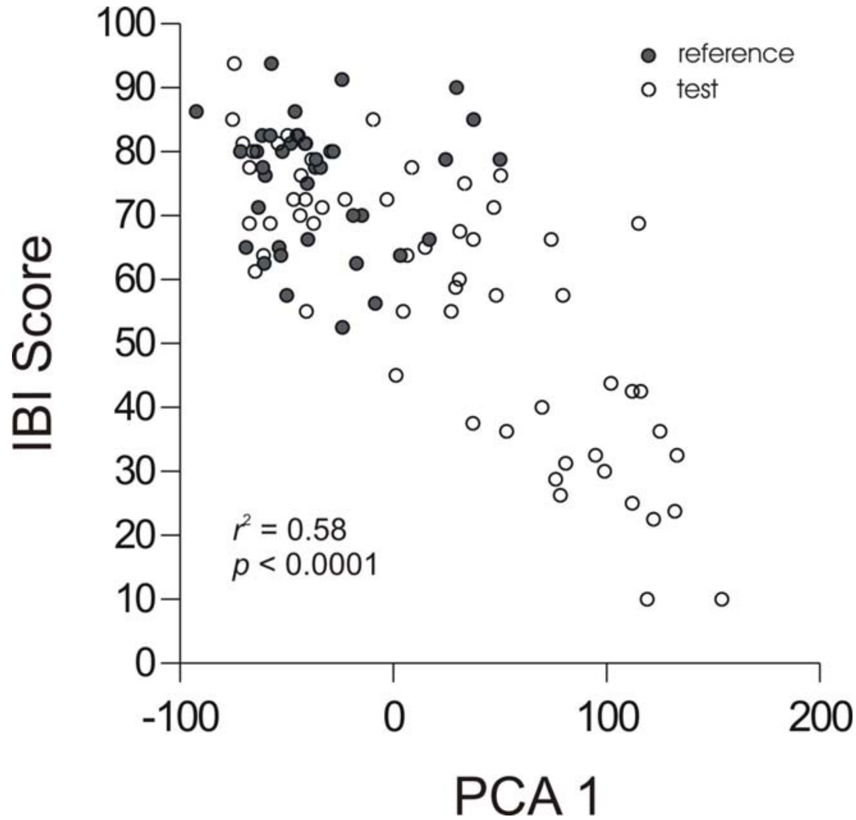


Figure 3. B-IBI score as a function of a multivariate watershed condition axis (PCA 1; 38.4% of variance explained). Compare with Fig. 4 in original report.



Axis 1 loadings

% watershed unnatural	0.311
% watershed in agriculture	0.21
% local watershed forested	-0.331
% local watershed unnatural	0.244
road density in local watershed	0.102
population density in full watershed	0.352
population density in local watershed	0.334
mid-channel canopy density	-0.176
% sand and fine sediment	0.307
conductivity	0.277
total phosphorous	0.318
chloride	0.357
channel alteration	-0.104
% timber harvest in watershed	-0.052
% timber harvest in local watershed	-0.021

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Secretary for Environmental Protection

California Regional Water Quality Control Board
North Coast Region
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Arnold Schwarzenegger
Governor

Subject: Addendum II to the North Coast Regional Water Quality Control Board comments on Index of Biotic Integrity (IBI) for Wadeable Stream in Northern Coastal California

File: SWAMP North Coast IBI

Executive Summary

The objectives of the Clean Water Act are to restore and maintain the chemical, physical and biological integrity of the nation's waters. While methods for monitoring and assessing the chemical and physical conditions are well established, monitoring the biological integrity of the nation's waters presents a greater challenge.

The North Coast Regional Water Quality Control Board (Water Board) is very supportive of the development and use of biological indicators for assessing stream health. The diversity and vigor of biological communities reflect watershed conditions and can be good indicators of the overall quality of the water and habitat it supports. The Water Board has been and continues to be involved in the development and use of benthic macroinvertebrates (BMI) for stream assessment, including the development of the North Coast Index of Biotic Integrity (NCIBI, Rehn et al. 2005) as a biological indicator. The NCIBI is very useful when assessing stream conditions for urban or agricultural impacts. The NCIBI can also be used to test for potential toxic impacts or to monitor point-source discharges.

However, the application of the NCIBI to monitoring timber harvest activities appears to be more complicated. Considerable disagreement exists between regional stream condition assessments based on the NCIBI and the number of stream reaches currently listed for sediment and/or temperature impairment under Clean Water Act section 303(d). According to probability surveys that used the NCIBI as a biological indicator, 6% of stream length in the North Coast is biologically impaired (Rehn et al. 2005). By contrast, watersheds with 303(d)-listed sediment impaired stream reaches comprise 61% of the region, and 62% of the region is listed for temperature impairment. The original 303(d) listings were based primarily on federal Endangered Species Act (ESA) threatened and endangered species listings. Specifically, sediment and temperature impacts associated with historical and recent timber harvest have impaired salmon and steelhead habitat throughout most of the North Coast Region. The 303(d) listings have undergone quantitative evaluations by the National Marine Fisheries Service (NMFS) (Good et al. 2005), and by the USEPA in Total Maximum Daily Loads (TMDLs) development, which have confirmed the water quality impairment in these watersheds.

When NMFS Biological Review Teams updated the status of ESA-listed evolutionarily significant units of salmon and steelhead, they found that Chinook, coho, and steelhead populations “continue to exhibit depressed populations sizes relative to historical abundances,” and trends continue downward in several areas (Good et al., 2005). These findings are of particular concern for the endangered Central California Coast Coho, whose range overlaps part of the North Coast Region. A number of coho populations in the southern portion of the range appear to be either extinct or nearly so, including those in the Gualala, Garcia, and Russian rivers (Good et al., 2005).

The discrepancies between BMI-based condition assessments and 303(d) listings based primarily on depressed salmonid populations suggests to the Water Board that the NCIBI is not sensitive to sediment and temperature impacts associated with timber harvest. To examine the relationship between NCIBI scores and impacts associated with timber harvesting, an addendum to the NCIBI included timber harvesting as an explicit stressor (Rehn et al. 2008). However, the data sets that were used to evaluate this relationship were limited to a 10-year time period during which little harvesting took place in most of the watersheds. Whereas NCIBI scores were shown to decrease with urban and agriculture activities (Rehn et al. 2005), NCIBI scores did not decrease as timber harvest in a watershed increased. The reasons for the diverging relationships are not clear, especially since individual NCIBI metrics responded strongly and negatively to increasing percent sand and fine substrate measured in NCIBI study reaches (Rehn et al. 2005). However, it is apparent that while a stream may be sediment and temperature impaired from a salmon and steelhead population perspective, the macroinvertebrate community as expressed through the current NCIBI may not be. Consequently, the use of the NCIBI as a sole determinant of impacts from timber harvest is not advised.

The North Coast water quality impairments and their relationships to timber harvesting impacts and macroinvertebrates are further examined below. The discussion is meant to inform the reader, and hopefully won’t detract from the usefulness of the NCIBI for other assessments.

North Coast Region Water Quality Impairments

Twenty sediment and ten temperature quantitative TMDL assessments have been completed in the North Coast Region, all of which generally confirm the water quality impairments in these watersheds, with timber harvesting a large contributor to the impairments.

Other regional studies also indicate the impaired water quality status of North Coast Region. For example, Klein et al. (2008) showed that there are very large differences in turbidity between managed and unmanaged watersheds, and these large differences were correlated to timber harvesting rates. Klein et al. (2008) went on to propose several conceptual models that link the large differences in turbidity to potential effects on salmonids and their populations.

Considering the available information, water quality impairments resulting from timber harvesting are quite apparent in the North Coast Region. However, the first time benthic macroinvertebrates were used to assess the water quality conditions on a regional basis for the North Coast Region, the resulting assessment, particularly the NCIBI came to a different conclusion. The Clean Water Act section 305(b) report (California Water Boards, 2006) used two general types of benthic macroinvertebrate indices for the assessment: the observed/expected index (O/E index) and the NCIBI. The O/E index compares the number of taxa expected to exist at a site (E) to the number that are actually observed (O). The taxa expected at individual sites are based on models developed from data collected at reference sites. The NCIBI is the sum of a number of individual measures of biological condition, such as taxonomic richness and pollution tolerance. In both cases, the ability to recognize ecological degradation relies on understanding conditions expected in the absence of human disturbance. The NCIBI indicates that only 6% of the North Coast is in an impaired condition while the California O/E index indicated 40% impairment. Some of the disagreement between the macroinvertebrate indices can be explained by different impairment thresholds used by Rehn et al. (2005) and California Water Boards (2006), but other differences between the methods (e.g., O/E is more sensitive to loss of specific taxa whereas IBI responds to changes in assemblage structure) and land-use (e.g. timber harvesting) also appear to be playing a role.

Timber harvesting and benthic macroinvertebrates

There are a number of concurrent mechanisms at play during timber harvesting that result in the potential to impact streams. Timber harvesting increases sediment and nutrient discharges, allows more light to reach streams, and can shift riparian species composition from conifer to hardwoods. Each of these effects can influence the benthic macroinvertebrate community in different ways.

The Addendum to the North Coast IBI states that the “IBI and its component metrics had a slight positive correlation with timber harvest in regional watersheds,” which is the opposite direction from the way the IBI responds to other water quality impairments. For example, the number of EPT (Ephemeroptera, Plecoptera, and Trichoptera, also known as mayflies, stoneflies and caddisflies, respectively) taxa generally decrease with water quality impairment. However, this study detected an increase in EPT taxa with timber harvesting. A similar relationship was observed with the percent of shredder taxa. It is possible that, from a macroinvertebrate point of view, positive effects (e.g. changing riparian community) from timber harvesting counter-balance the negative impacts of timber harvesting (e.g. increases in sediment discharges).

Other regional studies concluded that while macroinvertebrate abundance increased with timber harvesting, assemblage metrics generally were not sensitive to sediment impacts associated with timber harvesting. These results indicate that macroinvertebrate metrics respond more strongly to natural environmental gradients than to harvesting (Herlihy et al., 2005 and Williams et al., 2002). However, smaller scale studies do show fine sediment impacts on some macroinvertebrate taxa. In the Klamath River watershed, Cover et al. (2008) found that many metrics commonly used for detecting effects of water pollution were not useful for assessing harvesting impacts in the steep forested streams even though the modeled sediment discharges had dramatically increased. Cover et al. were able to identify several taxa that showed strong responses to the increased fine sediment levels.

Further studies are needed to investigate the relationships between sediment and temperature impacts associated with timber harvesting and macroinvertebrate assemblages. Limited investigation indicates that the NCIBI is not sensitive to sedimentation and increased stream temperatures resulting from timber harvesting activities. Benthic macroinvertebrates may respond to recent harvest, but may not be sensitive to longer-term legacy effects of historical timber harvest, which may explain the disagreement between BMI-based condition assessments and 303(d) listings derived primarily from depressed salmonid populations. Alternatively, relatively small increases in the percentage of fines and sand in a stream may have a more deleterious effect on salmonid egg and embryo survival than on macroinvertebrate assemblages. The important point is that a good NCIBI score in the North Coat Region should not be interpreted to mean that a stream is also healthy from a salmonid perspective. An indicator species approach, using both salmonid population status and specific macroinvertebrate taxa shown to be responsive to low levels of sedimentation, could improve overall assessments of regional stream condition. In any case, it is not advisable to use the NCIBI as a sole determinant of aquatic impacts from timber harvesting.

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