From: Judith Unsicker
To: Diane Beaulaurier
Date: 6/13/02 3:20PM
Subject: Re: Fwd: Dave Herbst's Squaw Creek Paper

Please add the report enclosed in the attached email to the reference material for Region 6's 2002 Section 303(d) list update. The report, by Dr. David Herbst of the University of California Sierra Nevada Aquatic Research Laboratory, is a final contract product from our Squaw Creek TMDL project. It shows significant differences in biological communities at stations impacted by sedimentation in comparison with reference stations. The indices are expected to be used in development of TMDL targets, and might also be useful in development of listing factors as part of the State Board's forthcoming policy for use in future Section 303(d) list updates.

The report is contained in seven separate files, including text, graphics, and tables. The Regional Board's Squaw Creek TMDL coordinator is Cadie Olsen of South Lake Tahoe office staff. Her telephone number is (530) 542-5418.

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Development of Biological Water Quality Targets for Assessment of Total Maximum Daily Load (TMDL) of Sediment in the Squaw Creek Watershed (Placer County, California) Final Report April 16, 2002
Contract #9-118-160-0

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Introduction

One approach to defining a TMDL is as an expression of how much pollutant load a waterbody can accommodate without harm or degradation to the integrity of resident stream life. Among the water quality indicators that may be used in developing sediment TMDLs, measures of aquatic invertebrate communities provide direct information on sediment effects to aquatic life uses and a means of evaluating the restoration of biological integrity of stream habitats (USEPA 1999a). Use of quantitative data on the structure of biological communities in evaluating stream habitat quality is known as bioassessment (USEPA 1999b). Bioassessment surveys of baseline conditions can provide an evaluation of the existing status of target watersheds in contrast to reference watersheds that have been selected to reflect the natural spatial and temporal variability expected for similar stream types in minimally disturbed habitats. Differences between reference and target conditions on Squaw Creek (Placer County, California) were used here to evaluate the extent of sediment effects on biological integrity and provide a baseline and goal for monitoring ecological restoration.

Biological structure and integrity of stream environments can be ascertained from a quantitative description of the inhabitant organisms. Aquatic insects and other invertebrates are central to the function of stream ecosystems, consuming organic matter (wood and leaf debris) and algae, and providing food to higher trophic levels (fish and riparian birds). These native organisms also have varying degrees of pollution tolerance and so may be used as indicators of water quality and habitat conditions. Collections of the zoobenthos (bottom-dwelling fauna) may be used to evaluate the relative abundance of different taxa, feeding guilds, pollution indicators, and diversity, in order to develop a quantitative basis for measuring ecological attributes of the stream. Monitoring relative to reference sites (having little or no impact but similar physical setting), and/or over time within subject sites, then permits impact problems or recovery to be quantified (Rosenberg and Resh 1993, Davis and Simon 1995, Karr and Chu 1999). The use of bioassessment data can contribute to developing TMDLs by providing indicators of ecological health of stream habitat as altered by sediment, and setting target values for attaining a restored ecological condition.

Sediment TMDLs are often difficult to assess because transport and deposition of sediment is a natural process of streams. Sedimentation is a natural part of the landscape of watersheds and contributes to the dynamic process of building, shaping, and renewal of stream channels. Sediment can be important to the ecological function of streams in providing habitat and cover for certain kinds of organisms, and as a food resource (organic particles and microbial/algal growth occurring on particle surfaces). It is excessive sediment that can create impairment in the ecological function of streams. The challenge of the TMDL process is to determine at what point excessive sedimentation impairs water quality, and identify indicators that can be used to define and quantify the impairment.

Sediment as a pollutant is particularly harmful to aquatic life uses of stream bottom habitats because fine particles (clay, silt) and sand cause physical disturbance during both transport and deposition. Sediment movement (suspended and bedload) during high flow events scours stream channels and can leave much of the streambed barren of life. During sediment deposition, substrates become covered, embedded, or buried by sediment and life can literally be choked out. Deposition may leave a lasting legacy of lost habitat in streams that may only be recovered slowly by so-called flushing flows (Stalnaker et al. 1994; discharge sufficient to remove fines and sands from the interstices of larger stream bottom substrates). Because of these effects of sediment, benthic organisms such as aquatic invertebrates are a good choice as sensitive indicators for monitoring impairment in stream ecosystems (Waters 1995).

Field Monitoring Study Design and Sampling Strategy

Approach

The monitoring plan was designed to accomplish the following objectives:

1. Describe the existing condition of biological health in Squaw Creek
2. Compare conditions in Squaw Creek to reference watershed streams
3. Examine the relationship between sediment load and biological integrity

The invertebrate communities of reference streams were used here to reflect the potential range of ecological conditions found in stream habitats matched to the Squaw Creek watershed but with minimal or reduced sediment impacts related to land use. Some streams external to the Squaw Creek watershed with moderate to high levels of sediment loading were also sampled to help place sediment effects in a broader context and develop a dose-response relation. Sampling was conducted to frame the natural background spatial and temporal variability of streams nearby and within the Squaw Creek watershed. This was accomplished by sampling a varied size range of reference streams over a 2-year period. In the first year (2000) surveys were conducted during late-season low flows (late August), and in the second year during mid-season moderate flows (early July 2001). This approach allowed the greatest extent of natural differences in stream invertebrate communities to be defined for watersheds that were exposed to minimal land use slope erosion problems compared to the target Squaw Creek watershed, and provided an unbiased standard for evaluating the conditions in Squaw Creek. Quantitative description of biological communities at sites over a range of sediment loading exposures permitted development of a dose-response linkage between sediment stress and biological signals.

The goal of the project is to define biological criteria based on the reference stream sampling that can be used to establish whether and how much the Squaw Creek streams are impaired, and designate a water quality target for attaining recovery of biological integrity. Examination of the biological response over a dose range of sediment may further be used to identify a load level (threshold) at which impairment occurs. This level may be used as a practical guide to identifying a specific TMDL (or in this case annualized or event-related measure of load reduction) needed to attain the reference condition for biological health.

Site Selection

A variety of physical habitat features of streams can affect benthic invertebrate communities (Resh and Rosenberg 1984). In addition to natural erosion and sedimentation, the size, gradient and elevation may contribute to shaping communities as may land use impacts other than the suspected problem source. Site selection for bioassessment was thus guided by the need to account and control for varied environmental background influences.

Six sites were sampled in the target Squaw Creek watershed from the upper to lower portions of the drainage basin. These sites were divided into three stream types based on location and geomorphology: (1) upper watershed tributaries (South and North tributaries at near 6800 ft, representing higher gradient 1^{st}-2^{nd} order streams); (2) low gradient mid-watershed streams (3 sites in the meadows, representing <2% slope 2^{nd}-4^{th} order channel types); and (3) lower watershed streams located near the bottom of drainages (below the terminal valley moraine, just above the Truckee River). Selection of reference watershed streams for each Squaw Creek stream type was based on similarity with regard to:

- stream order (±1)
- channel width (±100-300 cm)
- size/length of upstream watershed (some similar size, others ± 0.25-3X length)
- elevation (mostly within 6,000 – 7,000 ft zone)
- gradient (±2% in most cases)
- aspect (eastern orientation)
- geographic proximity (within 20 mile radius, and tributary to Truckee River)
- geologic and geomorphic setting (metamorphic and granitic rock/soils)

Most of the reference sites were selected to represent the low gradient meadow stream type so that a large sample size was available for analysis of conditions in this longest segment of the Squaw Creek drainage. Twenty-eight surveys were conducted over the 2000-2001 period at 22 separate locations (4 Squaw Creek sites and 2 reference sites were sampled in both years to examine temporal variation).

Reference watershed study reaches were also selected based on the sediment load regime predicted from maps generated by the Annual Agricultural NonPoint Source Model (AnnAGNPS, USDA 2000) developed by the Desert Research Institute of the University of Nevada at Reno (DRI 2001). The AnnAGNPS model generates sediment load predictions for different positions within watersheds based on the effects of a high run-off year on the upstream landscape (dependent on slopes, soils, vegetation cover, erodibility, land use, etc). Streams conforming to the general selection criteria above were selected from these maps to form reference streams, and a range of potential sediment exposures.

Listing of stream survey locations and types:

Watershed location / stream type	Squaw Creek Sites	Reference/ or Exposure Sites
Late Season Low-Flow Regime (late August 2000)		
Upper watershed reach	Squaw Ck -South tributary Squaw Ck -North tributary	Pole Creek
Mid-watershed low gradient reach	Squaw Ck meadows –lower Squaw Ck meadows –middle Squaw Ck meadows –upper	Little Truckee R –Perazzo Cold Creek Sagehen Creek Prosser Creek
Lower watershed reach	Squaw Creek –below moraine	Bear Creek General Creek
Mid-Season Moderate-Flow Regime (early July 2001)		
Upper watershed reach	Squaw Ck -South tributary Squaw Ck -North tributary	Lacey Creek Juniper Creek
Mid-watershed low gradient reach	Squaw Ck meadows –lower Squaw Ck meadows –middle	Little Truckee R. –Coldstream Sagehen Creek Perazzo Creek Independence Creek Martis Creek N. Prosser Creek Alder Creek (load exposure) Trout Creek (load exposure)
Lower watershed reach	Not repeated	Bear Creek

Sampling Methods

The data gathered consisted of physical habitat surveys and biological sampling of benthic macroinvertebrates, algae and organic matter. Each site was defined as a 150-meter length study reach, located by GPS-UTM coordinates and elevation (near lower end of each site). The longitudinal distribution and length of riffle and pool habitats were first defined then used to determine random locations for sampling of benthic macroinvertebrates from riffle habitat. Slope over the reach was measured with a survey transit and stadia rod, and sinuosity was estimated from straight-line distance over the 150 m channel, or maps of 500-1000 meters of stream length centered on the study reach. Physical habitat was measured over the length of each reach using 15 transects spaced at 10 meter intervals. Water depth, substrate type and current velocity were measured at five equidistant points on each transect along with stream width, bank structure (cover/substrate type and stability rating), riparian canopy cover, and bank angle. Bank structure between water level and bankfull channel level was rated as open, vegetated, or armored (rock or log), and as stable or eroded (evidence of collapse or scour scars). Bank angles were scored as shallow, moderate, or undercut (<30°, 30-90°, and >90°, respectively), and riparian cover was estimated from vegetation reflected on a grid in a concave mirror densiometer (sum of grid points for measurements taken at each stream edge and at mid-stream facing up- and downstream). The type and amount of riparian vegetation along the reach was also estimated by qualitative visual evaluation. The embeddedness of cobble size substrate was estimated as the volume of the rock buried by silt or fine sand for 25 cobbles (encountered during transect surveys or supplemented with random selected cobbles). Discharge was calculated from each transect as the sum of one-fifth the width times depth and current velocity at each of the five transect points, and averaged. Basic water chemistry and related measures consisted of dissolved oxygen, conductivity, alkalinity, pH, temperature, nitrogen, phosphorus, silica, hardness, sulfate, and turbidity. Documentation also included photographs taken at mid-stream looking upstream at 0, 50, and 100 meters, and downstream at 150 meters. Biological sampling consisted of 5 replicate benthic samples taken in riffle zones with a 30-cm wide D-frame kick-net. Each replicate was comprised of a composite of 3 30x30 cm sample areas taken across the riffle transect or over riffle areas of varied depth, substrate and current. This composite of microhabitats provides a more representative sampling and reduces the variability among replicate samples. Samples were processed in the field by washing and removing large organic and rock debris in sample buckets followed by repeated elutriation of the sample to remove invertebrates from remnant sand and gravel debris. Remaining debris was inspected in a shallow white pan to remove any remaining cased caddisflies (e.g., Glossosomatidae), snails or other molluscs. Elutriated and inspected sample fractions were then preserved in ethanol, and a small volume of rose bengal stain added to aid in lab processing. Invertebrate field samples were subsampled in the laboratory using a rotating drum splitter, sorted from subsamples under a magnifying visor and microscope, and identified to the lowest practical taxonomic level possible (usually genus; species when possible based on the availability of taxonomic keys, except for oligochaetes and ostracods). A minimum count of 250 organisms was removed from each replicate for identification (in practice averaging about 300-500). Data analysis yielded information on taxonomic composition by density and relative abundance. Metrics of community structure were calculated to express biological health in terms of diversity, composite community tolerance, number of sensitive taxa (mayfly-stonefly-caddisfly), dominance, and other measures of composition. All stages of sample processing and identification were checked using quality control procedures to assure uniformity, standardization and validation (QAPP; Herbst 2001).

The benthic food resources of stream invertebrates were also quantified in sampling of organic matter and algae. Particulate organic matter was sampled using a 250-micron mesh D-frame net, sampling stream bottom riffles as above for invertebrates (3 replicate riffle samples). These samples were poured through a 1-mm screen, with the retained wood and leaf particle debris then weighed as a wet biomass measure of coarse particulate organic matter (CPOM). The fine fraction passing through the screen (particle range 250 microns to 1000 microns) was collected in a 100-micron mesh aquarium net, placed in a sample vial, preserved in formalin, and then dried and ashed in a muffle furnace at the laboratory to quantify ash-free dry mass of fine particulate organic matter (FPOM). Algal periphyton was quantified by scrubbing attached algae off rock surfaces using a wire brush, homogenizing the algae removed using a large syringe, and subsampling the homogenate for (a) chlorophyll-a by filtration through 1-micron pore-size glass fiber filters, and (b) archival of algae for cell counts and taxonomic identifications (preserved in formalin and Lugol's stain). This was performed on three replicate cobble-size rocks from mid-stream riffle habitats. The area of each rock was estimated from measures of length, width, height and circumference, and the chlorophyll-a per area determined by extraction of stored frozen filters in ethanol and reading light absorbance of the extract in a fluorometer relative to a standard curve.

Data Analysis (dose and response variables)

A recent National Research Council review of the scientific basis for use of TMDLs (NRC 2001) recognized that biological criteria or aquatic life uses of streams should be integrated into water quality targets because "biocriteria are a better indicator of designated uses than are chemical criteria." The design developed for the Squaw Creek TMDL anticipated the recommendations of this review in that biological criteria and an empirical dose-response model of the stressor (sediment) were planned from the outset of this study. Appendix I excerpts this review as further justification for the approach used.

The biological response variables used were based on measures that have been commonly applied in bioassessment analyses and have an expected (and documented) response to stress. After correlation analysis with environmental variables, selected metrics were combined into a standardized biological condition score to reduce the measures into a single index of biological integrity (the multimetric approach; Karr and Chu 1999).

Stream habitats with minimal human-related disturbance, heterogeneity in stream bed substrates and food resources, stable banks, mixed riparian cover, and unaltered flow regime typically contain a diverse array of sensitive taxa inhabiting varied microhabitats, using different food resources, and having varied life cycles. Stressors compromise the quality and variety in stream habitats, resulting in the loss of structural and functional diversity, and of organisms intolerant of stress (diversity is lost, composition changes).

List of selected invertebrate community structure metrics and expected response to stress: (based on mean values from replicate samples)

Biological Metric	Metric Definition	Expected Response to Stress
Taxa Diversity (mean of samples)	Total number or richness of taxa found in a sample (reflecting resource variety)	Decrease
EPT Diversity Index (ephemeroptera, plecoptera, and trichoptera)	Number of taxa belonging to mayfly, stonefly, and caddisfly orders, usually regarded as intolerant of pollution	Decrease
%EPT	Percent of the organisms present belonging to one of the EPT orders	Decrease
Biotic Index	Composite measure of community tolerance to pollution (based on tolerance values and relative abundance)	Increase
No. of Sensitive Taxa (0-2)	Number of taxa with tolerance values of 0, 1, or 2 (scale of 10; least to most tolerant)	Decrease
% Tolerant Taxa (7-10)	Percent of organisms with tolerance values of 7-10 (scale of 10)	Increase
%Dominance	Percent of organisms comprising the most abundant taxon (resource imbalance)	Increase
R-50 Dominance (pooled samples) [=diversity at 50% total count, and decreases as dominance increases]	Number of taxa required to reach 50% (half) of the ranked abundance of all organisms – an inverse dominance measure	Decrease

Variables to express the exposure to, or dose of sediment loading were derived both from model predictions (the AnnAGNPS model for the Truckee River watershed), and from empirical on-site measures of sediment-related physical features of the stream environment at each study reach. This complementary approach could also be used to verify whether observed habitat features matched the model predictions.

Predicted sediment loads (tons) were obtained from GIS analysis of the AnnAGNPS model using the UTM coordinates of each study reach as geospatial reference points for calculating the sum of upstream sediment that could reach that point in the watershed. The step-wise procedure used is documented in Appendix II (A. Sutherland, LRWQCB; personal communication).

Reasoning that sediment is transported and deposited from upstream sources over and along stream courses, the model-predicted sediment load was distributed both relative to the upstream channel length (both perennial and intermittent), and the study reach stream width (i.e., tons divided by sum of upstream miles, divided by mean stream width). This "distributed model" (tons/upstream mile/m width) was used to express the potential exposure to sediment loading at each site. In making these calculations, it was further assumed that lakes along the catchment basins serve as sediment traps, so any stream miles above lakes were excluded from the measure of upstream length. For streams surveyed in both years, widths were calculated as the mean of all transects combined. No model estimate of load was available for General Creek, so an approximation was made by using the load for Independence Creek (a similar forested watershed about 50% larger), and reducing this amount by about 10%.

Several measures taken during physical habitat surveys were also used to express the exposure or dose of sediment received at each study reach. Sediment remaining in a stream represents the legacy of past transport and the amount of load deposition onto the habitat of benthic invertebrates. Substrate type measures made along survey transects were used to calculate percent fines, percent fines + sand, and D-50 particle size (particle size at which cumulative distribution reaches 50%; calculated as fraction of size class range attaining the 0.5 proportion). In addition, percent cobble embeddedness is a measure of the extent to which substrate in this size class is buried by fines or sand. Turbidity was also examined as an indicator of sediment transport (though since transport is a transient process, point-sampling of turbidity is unlikely to detect sediment flux).

Once both sets of biological response metrics and sediment dose measures were summarized, a correlation analysis was performed to establish (1) the relation of the distributed sediment load model predictions to in-stream measures of sediment deposition, and (2) the relation of sediment to invertebrate community structure and composition. Each of the biological variables displaying correlations of $R>0.5$ (negative or positive) with some measure of stream sediment were then combined (after being converted to standard scores) to produce a single biological condition score for each stream. The full range of this score was then divided into to produce a scale for rating impairment thresholds.

Results and TMDL Development

The physical and chemical features of all stream study reaches are summarized in Tables 1a and 1b (low gradient reaches), Table 2 (upper watershed), and Table 3 (lower watershed). Contrast of the Squaw Creek sites with reference sites within each stream type shows that reference conditions frame the target sites with respect to most features except that discharge was lower on Squaw Creek. This was especially true in 2000 when flows were discontinuous over parts of the watershed (subsurface flows over portions of some study reaches). Such spatially intermittent channels come about during low flow periods and often form in reaches with permeable deposits of sediments and gravel (Stanley et al. 1997). Sediment deposition within the channel of Squaw Creek has produced a deep bed of alluvium within which surface water may infiltrate, promoting the occurrence of intermittent flows, especially in the low gradient meadow reaches that form the longest portion of the stream. Sediment deposition and flow variability are interconnected attributes of the Squaw Creek stream channel.

Management of sedimentation requires that there is a reasonable basis for understanding the sources of erosion that need to be controlled to improve water quality. The AGNPS modeling approach explicitly identifies landscape features that contribute to erosion. Examining the relationship between sediment load predictions and the size of watersheds, and in-stream measures of deposition can test the validity of the model. First, load is expected to scale with channel length or discharge (Leopold 1994) in reference watersheds, and Squaw Creek load should be above that expected for its size. Second, increased sediment transport loads should leave behind deposition of smaller particles. These expectations were verified, with Squaw Creek sites showing loads well above the regression-line among all sites surveyed outside the Squaw watershed (Figure 1), and decreased particle size with higher distributed load in low gradient streams (smaller D-50 particle size and greater percent of fines + sand; Figure 2). The clustering of sites along the gradient of distributed sediment loads (Figure 3) also provides a basis for identifying the streams that define the reference condition for each stream type. Low gradient, upper watershed, and lower watershed stream types each have reference sites that possess reduced loadings relative to Squaw Creek. The low gradient stream sites, with the most survey data, show that loads below the bin range of 300-400 tons/mile/m width define the reference stream load level (reference sites listed on upper panel, Figure 3).

Correlations between sediment-related physical variables and metrics of invertebrate community structure are shown as a matrix in Table 4. Data were derived from surveys of 28 streams, 140 benthic samples, and over 80,000 organisms counted. Of the physical variables examined, the distributed sediment load model, along with D-50 particle size and percent fines + sand, showed the best correlations with biological metrics. Turbidity, embeddedness, and %fines alone showed low correlation with metrics, and also did not correspond to the other sediment measures. Invertebrate community metrics that showed the highest correlations with the load, particle size and fines + sand measures of sediment included the biotic index, total taxa diversity, EPT taxa diversity, %EPT, number of sensitive taxa, percent tolerant taxa, and the R-50 measure of dominance and diversity. Selected examples of these dose-response relations are shown in Figures 4 through 6 (for low gradient stream type), Figure 7 (upper watershed stream type), and Figure 8 (lower watershed stream type). This set of physical and biological measures provide the most useful indicators for setting water quality targets and as future monitoring tools for tracking the progress of erosion control measures in habitat restoration.

Inspection of the dose-response graphs for the low gradient stream types suggest the following sediment targets may be associated with improved biological integrity:

- Figure 4: below a distributed sediment load of 400 tons/mile/m stream width
- Figure 5: above a geometric mean D-50 particle size of 40 mm
- Figure 6: below 25% fines + sand cover of the stream bottom

It is apparent that other factors may also ameliorate the negative effects of these levels of sedimentation indicators (since some reference sites also exceed these levels). Flow velocity, the availability of larger substrates, and turbulence (mostly related to gradient and bed roughness) may for example contribute to improved habitat, but the strong response of enhanced measures of the quality of stream life with low sedimentation argues for use of these measures as guidance in the load reductions needed to alleviate sediment stress. Of the low gradient Squaw Creek meadow sites, the lower meadow has the greatest distributed load value at nearly 800 tons/mile/m, suggesting that a load reduction of at least 50% will be required to improve habitat to below the exposure level of 400 tons/mile/m. With reference sites in the load range of 100-300, even greater reduction may be needed to attain this level of habitat quality. Since this load exposure is based on a long-term high-flow year (1996-97 water year), it is the in-stream measures of particle size and fines/sand cover that may be the best short-term indicators of the success of erosion control. If slope erosion is minimized, natural flushing flows may serve to gradually transport sediment out of the channel of Squaw Creek, and improve substrate conditions. A detailed analysis of the annual sediment input-output budgets would be needed to evaluate the conditions that would promote streambed cleansing.

In order to reduce the complexity of information contained in the various metrics of invertebrate community structure, standard scores were assigned to each metric for each stream, based on the distribution of values for each metric (USEPA 1999b), and summed to produce a single biological condition index. The scores assigned to the actual value for each metric comprising the index were as follows:

	Biological Condition Scores Assigned to Metric Value Ranges		
Metric	**5**	**3**	**1**
Biotic Index	< 3.5	3.5 – 4.5	>4.5
Taxa Richness	>50.0	40.0 – 50.0	<40.0
EPT Diversity Index	>20.0	15.0 – 20.0	<15.0
%EPT of Total	>50%	35 – 50%	<35%
No. Sensitive Taxa	>18.0	12.0 – 18.0	<12.0
% Tolerant Taxa	<5%	5-10%	>10%
R-50 Index	>5.0	3.0 – 5.0	<3.0
Biological Condition Score Sum: Rating the loss of biological integrity / water quality			
Reference Score	20-30% impaired	35-50% impaired	>50% impaired
25-35	20-25	15-20	<15

Note that the reference sites, defined *a priori* according to the distributed sediment load model (Figure 3), conform to the threshold set for the biological reference condition (i.e. they score index values of 25 or greater, with the exception of Martis Creek). The other thresholds were set to express different levels of impairment relative to the mid-range of the reference condition (a value of 30).

Biological condition scores for low gradient stream reach types show that impairment of Squaw Creek meadow sites was severe in 2000 when flows were discontinuous, but improved somewhat in 2001 when flows were continuous (Figure 9). Instability in community structure between years in the Squaw meadows stream reaches is another sign of habitat disturbance (community composition measures changed substantially). As a criterion for recovery, the biological condition score should reach a reference value of 25, but recognizing inter-annual variability, this target level should be attained consistently (as a 5-year mean for example) to demonstrate stability in biological health.

Significant impacts to upper and lower watershed Squaw Creek reaches appear to be absent except on the South tributary in 2000 (biological condition scores of Table 5). This may be attributable to load movement through the system in the higher gradient upper watersheds, and upstream sediment capture in low gradient reaches (above the lower watershed Squaw site, below moraine). The South tributary has the highest distributed sediment load (about 2,700 tons/mile/m) and low flow conditions in 2000 may not have been sufficient to transport sediment and maintain high biological quality.

The approach used in this study provides useful guidance for the sediment TMDL because it combined (1) reference site sampling to establish a biological water quality target, (2) dose-response evaluation of impairment thresholds, and (3) determination of sediment exposure both from modeling data and in-stream field measures. With so many potential sources of confounding variation present in field data, the strong relation found between sediment and impaired biological quality attests to the reliability of the results.

Conclusions

Water quality targets can be defined for Squaw Creek using the reference biological data (25^{th} to 75^{th} percentile of observations), and associated sediment effect levels as follows:

Biotic Index	Taxa Diversity	EPT Taxa	%EPT Taxa	Sensitive Taxa	Tolerant Taxa	R-50 Index	Biological Condition Index
3.09 - 4.22	47.2 - 52.6	20.8 - 24.9	36 - 46%	16.8 - 19.9	0.4 - 1.7%	2.6 - 5.9	≥ 25

Distributed Load (tons/mile/m)	D-50 Size (mm)	%F+S Cover
< 400	> 40	< 25

Low gradient meadow reaches of Squaw Creek should be the focus of further monitoring of recovery indicators because these reaches represent cumulative effects, and are the most impaired stream habitats. Additional monitoring of reference watersheds under other flow conditions will also make target values more robust and applicable to a wider range of conditions.

The sediment load reductions necessary to (a) reduce impairment below an apparent threshold at 400 tons/mile/m is about 50%, and (b) achieve target values corresponding to loadings and biological condition of reference sites is about 75%. Inspection of the AnnAGNPS model terms, and the historic flow regime may provide insight to what control strategies could produce load reductions in this range (e.g. vegetation cover), or remove accumulated sediment (flushing flow level, below erosion thresholds).

As a final note, the data showed that Trout Creek at Bennett Flat had among the highest levels of sediment impairment of aquatic life uses. The sources and control of erosion in this small watershed should be considered in future water quality planning.

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Table 4. **Correlation Matrix for Sediment Dose and Biological Response Variables (R-values)**

	Load	*D-50*	*%Embed.*	*Turbidity*	*%F+S*	*%F*
Load	1.000					
D-50	***-0.596***	1.000				
%Embed.	0.190	-0.100	1.000			
Turbidity	0.120	-0.108	-0.202	1.000		
%F+S	***0.675***	***-0.502***	0.304	0.081	1.000	
%F	***0.730***	***-0.509***	0.258	0.116	***0.757***	1.000
Total Richness	***-0.506***	0.428	0.088	-0.144	***-0.650***	-0.428
Biotic Index (mod.HBI)	***0.642***	***-0.608***	-0.353	0.387	***0.586***	0.472
Mean Richness	***-0.545***	0.454	0.071	-0.181	***-0.680***	-0.407
EPT Diversity	***-0.619***	***0.566***	0.317	-0.289	***-0.660***	-0.472
Density (#/m^2)	-0.206	-0.025	0.118	-0.184	-0.339	0.006
%Dominance	0.368	-0.436	0.083	-0.129	0.176	0.369
%Chironomidae	0.066	-0.280	-0.474	0.350	-0.017	-0.115
Chironomidae richness	-0.265	0.185	-0.304	0.215	-0.436	-0.302
EPT/Chironomidae	-0.251	0.352	0.416	-0.325	-0.237	-0.174
%EPT total	***-0.510***	***0.560***	0.359	-0.308	-0.356	-0.431
%EPT (w/o B,H)	-0.307	0.456	0.293	-0.222	-0.210	-0.344
No. Sensitive (0-2)	***-0.597***	***0.514***	0.310	-0.292	***-0.651***	-0.389
% Tolerant (7-10)	***0.632***	-0.422	-0.139	0.373	***0.649***	***0.541***
R-50 Dominance Index	-0.322	***0.541***	-0.094	0.067	-0.289	-0.407

Correlations with a value of greater than 0.5 (negative or positive) are highlighted in bold italics for relationships among sediment variables (above line) and between sediment dose measure and biological response measure (below line).

Load refers to distributed model of predicted sediment load, D-50 is the geometric mean particle size, % embed. is the percent embeddedness of cobble substrates, turbidity is suspended particles, %F+S refers to percent fines and sand cover on the stream bottom.

Note that figures do not show error bars for the means plotted. For an indication of the error term in the metrics, the coefficient of variation (below) can be used. Metrics in left column have some of the best correlations with physical habitat variables and also the lowest values for coefficient of variation.

Coefficient of Variation for Biological Metrics (all 28 stream surveys)

Metric	Mean %CV	Metric	Mean %CV
Biotic Index	9.2	Density	38.0
Taxa Richness	10.8	%Dominance	28.3
EPT Taxa Diversity	12.6	%Chironomidae	29.1
%EPT Taxa	20.0	Chironomid Richness	17.2
No.sensitive taxa (tv 0-2)	15.8	EPT/Chiro. ratio	33.9
		%EPT(w/o *Baetis,Hydropsyche*)	23.0
		%Tolerant taxa (tv 7-10)	76.2

Appendix I: **National Research Council TMDL report excerpts**

Excerpts from: Assessing the TMDL Approach to Water Quality Management
Committee to Assess the Scientific Basis of the Total Maximum Daily Load Approach to Water Pollution Reduction

Water Science and Technology Board
Division on Earth and Life Studies
National Research Council
National Academy Press
Washington, D.C. 2001

from Executive Summary (p. 16):
Biological criteria should be used in conjunction with physical and chemical criteria to determine whether a waterbody is meeting its designated use. In general, biological criteria are more closely related to the designated uses of waterbodies than are physical or chemical measurements. However, guiding management actions to achieve water quality goals based on biological criteria also depends on appropriate modeling efforts.

All chemical criteria and some biological criteria should be defined in terms of magnitude, frequency, and duration. The frequency component should be expressed in terms of a number of allowed excursions in a specified period. Establishing these three dimensions of the criterion is crucial for successfully developing water quality standards and subsequently TMDLs.

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p. 26:

Box 3-2 The Information Value of Monitoring Multiple Criteria

The tendency for misdiagnosis of impairment by relying on only one type of criterion was illustrated in a study of more than 2,500 paired stream and river sampling sites in Ohio (Ohio EPA, 1990; Rankin and Yoder, 1990). In 51.6 percent of the samples, the results from biomonitoring and chemical monitoring agreed—that is, they both detected either impairment or attainment of the water quality standard. This was particularly true for certain classes of chemicals (e.g., toxicants), where an exceedance as measured by the chemical parameter was always associated with a biocriteria impairment. However, in 41.1 percent of the samples, impairment was revealed by exceedance of the biocriteria but not by exceedance of the chemical criteria. These results suggest that impairment may go unreported in areas where only chemical measurements are made. Interestingly, in 6.7 percent of the samples, chemical assessment revealed impairment that was not detected by bioassessment (especially for parameters such as ammonia-N, dissolved oxygen (DO), and occasionally copper). This latter occurrence is likely related to the fact that biocriteria have been stratified to reflect regional or ecotype peculiarities, and the more generically derived chemical criteria have not. Both the under- and overprotective tendencies of a chemical-criteria-only approach to water quality management can be ameliorated by joint use of chemical criteria and biocriteria, each used within their most appropriate indicator roles and within an adequate monitoring and assessment framework.

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p. 35:

Box 3-5 Index Systems for Bioassessment

During the past two decades, biological assessment evaluating human-caused biotic changes apart from those occurring naturally has become a part of water managers' tool kits. Two major approaches to ambient biological monitoring are used—the river invertebrate prediction and classification system (RIVPACS) and the multimetric index of biological integrity (IBI). Although their conceptual and analytical details differ, both RIVPACS and IBI (1) focus on biological endpoints to define waterbody condition, (2) use a concept of a regionally relevant reference condition as a benchmark, (3) organize sites into classes with similar environmental characteristics, (4) assess change and degradation caused by human effects, (5) require standardized sampling, laboratory, and analytical methods, (6) score sites numerically to reflect site condition, (7) define "bands," or condition classes, representing waterbody condition, and (8) furnish needed information for diverse management decisions (Karr and Chu, 2000). RIVPACS was developed in England (Wright et al., 1989, 1997) with clones available for use in Australia (Norris et al., 1995) and Maine (Davies and Tsomides, 1997). IBI was developed in the United States (Karr, 1981; Karr et al., 1986; Karr and Chu, 1999) with clones applied by state and federal agencies (Ohio EPA, 1988; Davis et al., 1996; Barbour et al., 1999) and abroad (Hughes and Oberdorff, 1999). Although applications of RIVPACS are historically limited to invertebrates in rivers, IBI applications have been developed for diverse taxonomic groups and waterbody types. For example, a multimetric index (RFAI, reservoir fish assessment index) has been developed as a component of Tennessee Valley Authority's (TVA) "vital signs" monitoring program to assess fishery management success in reservoirs (Jennings et al., 1995; McDonough and Hickman, 1999). As a general example, consider a minimally disturbed Pacific Northwest stream supporting self-sustaining populations of salmon and associated assemblages of invertebrates. With urban development, salmon decline and cutthroat trout become relatively more abundant, and certain invertebrate taxa (e.g., stoneflies) are reduced or eliminated. Tiered beneficial uses could in this case differentiate between streams supporting salmon vs. cutthroat trout, using an index based on the invertebrate assemblage as the biocriterion. Recent work in these streams suggests that a benthic index of biological integrity (B-IBI) of about 35 is a minimum required to maintain a healthy salmon population (Karr, 1998). If the IBI drops below 20 because of continued development, even the cutthroat trout will eventually disappear.

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p. 43:

Box 3-6 Understanding Sources of Variability in Bioassessment

Sources of error evaluated in one study of biological monitoring data from New England lakes (Karr and Chu, 1999) included three types of variance: interlake variability (differences among lakes); intralake variability (variability associated with sampling different sites within a lake as decided by the field crew), and lab error (error related to subsample work in the lab). The interlake variability was the effect of interest, and the goal was to determine if that source of variability was dominant. Distribution of variance varied as a function of biological metric selected. Those measures with reduced variance except for the context of interest (e.g., interlake variability) were selected for inclusion in IBI to increase the probability of detecting and understanding the pattern of interest. Two other studies involved an examination not of the individual metrics, but of the overall IBI (i.e., after individual metrics were tested and integrated into an IBI). For Puget Sound streams, 9 percent of variation came from differences within streams and 91 percent was variability across streams (reported in Karr and Chu, 1999, Fig. 35). For a study in Grand Teton National Park, streams were grouped in classes reflecting different amounts of human activity in their watersheds. In this case, 89 percent of the variance came from differences among the groups, and 11 percent came from differences among members of the same group (reported in Karr and Chu, 1999). In all these cases, the goal was to find ways of measuring that emphasize differences among watersheds with differing human influences, while keeping other sources of variation small. Success in these examples was based on the development of an earlier understanding of sources of variation and then establishing sampling protocols that avoid other irrelevant sources of variation (such as variation stemming from the differing abilities of personnel to select and use methods). If these sources of variation are controlled for, then the study can emphasize the kind of variation that is of primary interest (e.g., human influence gradients).

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p. 77:

MODELS FOR BIOTIC RESPONSE: A CRITICAL GAP

The development of models that link stressors (such as chemical pollutants, changes in land use, or hydrologic alterations) to biological responses is a significant challenge to the use of biocriteria and for the TMDL program. There are currently no protocols for identifying stressor reductions necessary to achieve certain biocriteria. A December 2000 EPA document (EPA, 2000) on relating stressors to biological condition suggests how to use professional judgment to determine these relationships, but it offers no other approaches. As discussed below, informed judgment can be effectively used in simple TMDL circumstances, but in more complex systems, empirical or mechanistic models may be required. There have been some developments in modeling biological responses as a function of chemical water quality. One approach attempts to describe the aquatic ecosystem as a mechanistic model that includes the full sequence of processes linking biological conditions to pollutant sources; this typically results in a relatively complex model and depends heavily on scientific knowledge of the processes. The alternative is to build a simpler empirical model of a single biological criterion as a function of biological, chemical, and physical stressors. Both approaches have been pursued in research dating back at least 30 years, and there has been some progress on both fronts. One promising recent approach is to combine elements of each of these methods. For example, Box 4-3 describes a probability network model that has both mechanistic and empirical elements with meaningful biological endpoints. Advances in mechanistic modeling of aquatic ecosystems have occurred primarily in the form of greater process (especially trophic) detail and complexity, as well as in dynamic simulation of the system (Chapra, 1996). Still, mechanistic ecosystem models have not advanced to the point of being able to predict community structure or biotic integrity. Moreover, the high level of complexity that has been achieved with this approach has made it difficult to use statistically rigorous calibration methods and to conduct comprehensive error analyses (Di Toro and van Straten, 1983; Beck, 1987). The empirical approach depends on a statistical equation in which the biocriterion is estimated as a function of a stressor variable. Success with this empirical approach has been primarily limited to models of relatively simple biological metrics such as chlorophyll a (Peters, 1991; Reckhow et al., 1992). For reasons that are not entirely clear, empirical models of higher-level biological variables, such as indices of biotic integrity, have not been widely used. Regressions of biotic condition on chemical water quality measures are potentially of great value in TMDL development because of their simplicity and transparent error characteristics. Two accuracy issues, however, need to be considered. First is the obvious question of whether the level of statistical correlation between biotic metrics and pollutant concentrations is strong enough that prediction errors will be acceptable to regulators and stakeholders. A second and more difficult issue is that of gaining assurance of a cause–effect relationship between chemical predictors and biotic metrics. The construction of empirical models of biotic condition would benefit greatly from (1) observational data that show the effects of changes in chemical concentrations over a time period when other factors have remained relatively constant and (2) inclusion of as many factors that are relevant to biotic condition as possible. The latter, of course, increases the requirement for observational data. Despite these limitations, in the near term, empirical models may more easily fill the need for biological response models than would mechanistic models.

Conclusions and Recommendations

1. EPA should promote the development of models that can more effectively link environmental stressors (and control actions) to biological responses. Both mechanistic and empirical models should be explored, although empirical models are more likely to fill short-term needs. Such models are needed to promote the wider use of biocriteria at the state level, which is desirable because biocriteria are a better indicator of designated uses than are chemical criteria.

[Note: references cited in this appendix not given; please refer to the original document]

Appendix II: **Outline of AnnAGNPS estimate of sediment loading**

MAXIMUM SEDIMENT LOADING ESTIMATES TO INVERTEBRATE SAMPLING SITES IN THE TRUCKEE RIVER WATERSHED (GIS analysis of AnnAGNPS model)
From Anne Sutherland, Lahontan Regional Water Quality Control Board

Rationale:

GIS polygons representing watershed subareas could be linked to a model output file containing estimates of the total mass of sediment leaving each watershed subarea. The sum of the total mass values for each subarea upstream of a sampling location could be used to estimate the "maximum" potential sediment load at that sampling point. This maximum is based on model validation with the 1996-97 high-flow water year.

Comments:

- Shapefiles and datafiles were provided by DRI for the Truckee River Watershed GIS Database.
- Refer to the Truckee River Watershed Assessment (TRWA) report (July 2001), for an overview of the AnnAGNPS model used to generate sediment loading estimates for this analysis.
- Analysis was done using ESRI's Arcview GIS software, version 3.2., by Anne Sutherland and Cadie Olsen of Lahontan RWQCB.
- Analysis was performed in the coordinate system and datum of the Truckee River Watershed GIS Database: UTM Zone 10, NAD 27, meters.

Documentation of maximum sediment loading GIS analysis:

1. Convert UTM northings and eastings of sampling sites into dBase format.
2. Import into new Arcview (AV) project.
3. Add to view as an "event theme."
4. Check sampling locations by adding stream, lakes, and watershed themes into view.
5. Adjust locations as needed with input from David Herbst.
6. Convert sampling locations event theme to AV shapefile (.shp).
7. Convert fullmass30801 text file (.txt) to dBase (.dbf) format.
   Notes: Fullmass30801.txt was provided by DRI, generated from AnnAGNPS model. File contains estimates of the total mass of sediment leaving each watershed subarea, not the mass from each subarea that reaches the Truckee river. Fields include: ID, silt, sand, clay, and total mass. Estimates are in tons.

8. Import fullmass.dbf into AV project.
9. Perform "table join" operation on modelbasin.shp and fullmass.dbf using "ID" field common to both attribute tables. This allows for analysis of modelbasin.shp by data in fullmass.dbf.
   Notes: Modelbasin.shp was provided by DRI as part of the TRWA GIS database. It is an Arcview polygon feature of subwatershed basins calculated by the AnnAGNPS model. See Appendix C to the TRWA report for metadata information on the GIS database.

10. Use "select" tool in AV and staff's knowledge of on-the-ground conditions represented by the spatial data to select appropriate watershed polygons upstream of the sampling locations.
    Notes: The entire polygon that contained the sampling site was always included, to give an estimate of worst-case or maximum sediment load. The sum of the total mass values for each subarea upstream of the sampling location was used an estimate of the "maximum" potential sediment load at that point.

11. Perform "convert to shapefile" operation on selected polygons. One shapefile per sampling site was generated, except for certain instances (Squaw Meadows, for example) where one shapefile represented the maximum sediment load for several sites due to their close proximity to each other.
12. Use AV to calculate summary statistics for each shapefile showing sum, mean, min, max, range and standard deviation for each size class, export as Excel (.xls) files.
13. Generate maps depicting selected polygons with total mass sediment load estimates for each site, export as .jpg files.