

# Managing Our Water Retention Systems

29th Annual USSD Conference Nashville, Tennessee, April 20-24, 2009

> Hosted by Corps of Engineers

# **On the Cover**

Wolf Creek Dam is on the Cumberland River in South Central Kentucky near Jamestown, Kentucky. It provides flood control, hydropower, recreation, water supply, and water quality benefits for the Cumberland River system. Construction began in 1941 and was interrupted by WWII from 1943 to 1946. The reservoir was impounded in December 1950. The 5,736 foot-long dam is a combination earthfill and concrete gravity section. U.S. Highway 127 crosses the top of the dam.

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- Fostering dam technology for socially, environmentally and financially sustainable water resources systems;
- Providing public awareness of the role of dams in the management of the nation's water resources;
- Enhancing practices to meet current and future challenges on dams; and
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#### BIOLOGICAL INDICATORS OF CONDITIONS BELOW DAMS IN THE WESTERN UNITED STATES

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#### ABSTRACT

Aquatic macroinvertebrates were collected below 43 Reclamation dams in the western United States. Multivariate analysis indicated that dam height was an important variable in structuring macroinvertebrate communities. Dam height represents a multitude of impacts which includes those related to temperature and thermal regime modification, sediment transport, hydraulic residence time, and water quality. There appeared to be limited association of hydrologic flow metrics with biological assemblages, perhaps because of the generally homogenous flow characteristics of dammed rivers. Declines in biological values were associated with increased dam height and reservoir surface area. An ecological index, the Biotic Dam Index was designed to classify biological impacts associated with dams. Metrics used in the index included taxa and EPT (Ephemeroptera, Plecoptera, Trichoptera) richness, proportion of non-insects in the community, and periphyton biomass.

### **INTRODUCTION**

Dams are designed to retain water during high flows or store water during periods of low demand and then deliver it for anthropogenic purposes. It is estimated that, throughout the world, over 45,000 large dams were built (World Commission on Dams, 2000) by the end of the  $20^{\text{th}}$  century, resulting in about two-third of the freshwater flowing to the oceans controlled by dams (Naiman et al., 1993). At least 100 of these dams are at heights >150 m (McAllister et al., 2001).

Characterization of dams is often based on an engineering perspective and includes dam size and the surface area of water retained behind the structure, operational purpose such as hydroelectric generation or irrigation, and construction material. Dams traditionally have been used for water supply, hydropower, and flood control. Environmentally sensitive management of these regulated rivers requires that these traditional uses be reconciled with ecosystem management. Site specific ecology of dams has been extensively studied, with case-by-case studies used to identify regulation impacts (Jackson et al., 2007). Despite the abundance of data collected on dam ecosystems, Poff and Hart (2002) suggest that little information is presently available to allow meaningful generalizations of how dams modify river ecosystems. Poff and Hart (2002) also suggest that an ecological classification is needed to describe modifications used to mitigate environmental damage from dams. Gómez-Balandra et al. (2008) suggest that, because of the increasing complexity of systems containing series of dams, environmental assessment to protect ecosystems is of high priority.

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Proponents of this ecological approach present it as a new paradigm that reforms the way humans interact with nature. Norton (1992) suggests a moral obligation to act sustainably in order to protect the natural processes that form the context of human life and culture. In essence, ecosystems, within which humans have evolved, must continue if humans are to thrive. The ecosystem approach then involves defining the basis upon which ecosystems support, over the long-term, human activities. Services provided by free-flowing rivers might include water purification, flood mitigation by decreasing water velocities and allowing water to infiltrate into the floodplain, nutrient cycling, trapping of sediment by vegetation in the riparian area, and high biodiversity. A river below a reservoir would be ecologically sustainable if there was no diminishment of organisms below the dam. Biological indicators linked to dam impacts could be useful metrics for tracking ecosystem improvement through the adaptive management process. Baseline information could also be used to study impacts of whole system changes that might be associated with climate change or invasions of exotic species. The near absence of biological monitoring data below dams is recognized as a serious information gap (McAllister et al., 2001).

It appears that responses of biota to regulated river systems are often complex and variable, and some biota may be more appropriate as ecological indicators of dam impacts than others. Camargo and de Jalon (1990) for example found that aquatic macroinvertebrates were better than fish communities for detecting environmental changes and for reflecting recovery from alterations caused by dams. Macroinvertebrate assemblages are sensitive to altered habitat, changes in sediment input, water quality, thermal regimes, flow patterns (Ward, 1976; Armitage, 1984; Armitage et al., 1987), and biological or chemical contaminants associated with reservoirs. Kremen et al. (1993) have suggested that arthropods may be especially appropriate as ecological indicators because of their rapid response to environmental changes. Aquatic invertebrates also play a role in transfer of energy to higher trophic levels. Benthic invertebrates are a major part of the food resource for fishes, and changes in invertebrate communities may result in changes in condition of fish communities (e.g., Waters, 1982; Bowlby and Roff, 1986; Wilzbach et al., 1986). Further, many aquatic invertebrates have non-aquatic phases, leading to their importance to other predators, including birds (Paetzold et al. 2005, Sanzone et al. 2003, Skagen et al. 2005). Benthic macroinvertebrates are often collected to help evaluate water quality and/or habitat quality, and many different metrics have been proposed over the years (Cairns and Pratt, 1993). Macroinvertebrate indicators may provide an accurate, low-cost method for environmental assessment and evaluation that is directly related to important resources.

A multidisciplinary approach using physical, chemical, and biological tools was used to provide multiple lines of evidence with which to evaluate the status of benthic communities in tailwaters below Reclamation reservoirs in the Western United States. Data collection was designed to assess Reclamation's tailwater biology and to develop an understanding of the factors that affect the observed conditions. Data obtained were used to develop a biocriteria to assess aquatic system health below dams. Theoretically, environmental assessments that use multiple measures of biological condition result in robust and representative measures of condition. In this study statistical analyses of collected data were used to aid in the design of biological metrics used to rank tailwater sites.

Potential uses of collected data include remediation of specific problems, aid in operational decisions associated with endangered species, and research on factors that affect water quality. Reclamation needs to know whether certain types of biotic responses are isolated or ubiquitous, whether they are associated with certain reservoir operation styles, and whether there are significant differences in conditions among regions. Given a homogenous data set from below a large number of reservoirs it may be possible to associate certain faunal communities with particular below-reservoir sites defined by discharge regime, substrate type, habitat, and water chemistry.

# **METHODS**

Benthic macroinvertebrate community data along with environmental variables were collected below 43 dams (average collection point 278 m downstream of dam) from Arizona, California, Colorado, New Mexico, Washington, Wyoming and Utah (Table 1). Codes used for dam names are also presented in this table. Data were all collected at about the same time of the year in August/September during the years 1999 to 2006.

# Chemical, Physical, and Biological Methods

A 3-minute kick method with a D-frame net (700-800 um mesh) was used for sampling benthic invertebrates along a ca. 25-m wadeable portion of the streams. The net was placed on the stream bottom and upstream substrate disturbed by vigorous kicking. As substrate was disturbed, the operator and net moved upstream for the required time. Samples collected from the net were preserved in 70% alcohol. In the laboratory, samples were washed in a 600-micron mesh sieve to remove alcohol, invertebrates were then picked from the substrate with the aid of an illuminated 10X magnifier, and the sample was enumerated and identified to lowest practical taxon under a binocular dissecting scope. In most cases the entire sample was processed, however, in cases where invertebrates were too numerous to process in a reasonable time, smaller and more abundant organisms were subsampled using an 84-3 X 3 cm square grid. Grid squares were processed. All organisms within a square were processed even if the total number exceeded 100. Final organism counts from the gridded tray were extrapolated to account for the entire sample.

Dissolved oxygen (DO) (mg/L), conductivity ( $\mu$ S/cm), pH, and temperature (°C) were measured with a portable meter. Water samples for alkalinity and hardness (mg/L) were analyzed with titration methods (Hach test kit).

Periphyton samples were collected from rocks or other solid, flat surfaces with a sampling device made from a modified 30-ml syringe with an inside diameter of 2.06 cm (Porter et al., 1993). Three different substrates were sampled from the area where invertebrates were to be collected and composited into a single sample. The sample was

then filtered onto glass-fiber filters. Ash-free-dry-mass was determined using standard methods (Eaton et al., 1995). Filters were dried for 48 hr at  $105^{\circ}$ C, dry weight determined on an analytical balance, filters ashed at  $500^{\circ}$ C for 1 hr, and the mass of the residue (ash weight) determined. Ash-free-dry-mass (g/m<sup>2</sup>) was calculated by subtracting the ash weight from the dry weight of the sample and dividing by the periphyton sample area (9.99 cm<sup>2</sup>).

Size composition of the substrate was visually estimated at each site in the area where macroinvertebrates were collected. Categories were expressed as percent bedrock, boulders, cobble, coarse gravel, fine gravel, and sand/fines. Percentage categories were converted to a single substrate index (SI) value (e.g., Jowett and Richardson, 1990) using the formula S.I.=0.08\* bedrock + 0.07\* boulder + 0.06\* cobble +0.05\* gravel +0.04\* fine gravel + 0.03\* sand and fines. Wet width of the stream was measured with a measuring tape or a range finder.

Water velocity at 10 cm above the substrate was measured at three discrete points in the invertebrate collection area. The average of these three measurements was used in analysis.

Habitat disturbance was estimated with Pfankuch's Index (Pfankuch, 1975). This subjective, composite index involves scoring 15 stream channel variables along the upper bank, lower bank, and stream bottom. High scores represent unstable channels at the reach scale. This index has been found to accurately describe disturbance in streams in independent studies (Townsend et al., 1997).

Dam characteristics such as dam height, reservoir surface area, dam crest elevation, and age were obtained from literature sources including Water and Power Resources Service (1981).

# **Hydrologic Indices**

For each site, a record of daily averaged discharge values was sought for the period of time 4 years prior to the sampling date. It was assumed that the behavior of the flow regime greater than 4 years before sampling would not significantly influence the population at the sample time. The primary source for flow data was the USGS National Water Information System website (waterdata.usgs.gov/nwis). Twelve sites had USGS flow records for the 4 year period of interest. The USBR Hydromet on-line database (www.usbr.gov) served as a secondary source and was searched for additional flow data. An additional 10 sites were found with valid periods of interest. The remaining 21 sites were subsequently disqualified because the available flow records either did not cover the relevant period of interest or contained significant gaps in the record. An input file was created and the full complement of metrics for each site was generated using the batch processing feature in GeoTools. Hydrologic indices from these data were calculated by Brian Bledsoe and Mike Brown at Colorado State University. Because of the redundancy associated with the numerous indices which have been promulgated, a subset of 4 types was utilized in the analysis based on recommendations of Olden and Poff

(2003).  $M_A 26$  is related to variability in monthly flows measured as the coefficient of variation of monthly flow values,  $M_L 17$  is a baseflow index defined as the seven-day minimum flow divided by mean annual daily flows across the years,  $M_H 20$  is the mean annual maximum flow divided by catchment area, and  $F_H 1$  is the average number of flow events with flows above a threshold equal to the 75th percentile value for the flow record of interest, in this case 4 years.

						Hydrologic	Comments
	Dam/		Year	Dam	Hydro-	indices	
CODE	Reservoir	State	sampled	type	Electric?	calculated	
							Maintains bead for the
							Casper-
ALCO	Alcova	WY	2005	Earthfill	У	У	Alcova canal.
BONN	Bonny	CO	2001	Earthfill	n	n	0
							Some concrete, but
				<b>a</b>			mostly
BRAN	Brantley	NM	2000	Concrete	n	n	earthfill.
BOWP	Bumping	VVA	2002	Earthfill	n	У	
CABA	Caballo	NM	2000	Earthfill	n	У	
CLEA	Clear Creek	CO	2000	Earthfill	n	n	
CLEE	Cle Elum	WA	2002	Earthfill	n	У	
CLEL	Clear Lake	WA	2002	Concrete	n	n	Demonstration
							Reregulating afterbay for
DAVI	Davis	AZ	2001	Earthfill	у	У	Hoover Dam.
EAST	Easton	WA	2002	Diversion	n	У	
	Elephant			_			
ELE	Butte	NM	2000	Concrete	у	У	
	Flaming		1000	<b>O</b> a manta			
	Gorge		1999	Concrete	у	У	
	Fontanelle	VVY	2000	Earthfill	n	У	
GLEN	Giendo	VV Y	2001	Earthfill	у	У	
GRAN	Granby	00	2000	Earthfill	n	n	Reregulating
							reservoir for
GRAY	Gray Reef	WY	2006	Earthfill	n	n	Alcova.
GUER	Guernsey	WY	2001	Earthfill	n	n	
	Headgate	<u>م ح</u>	2004				
HEAD	ROCK Horeotooth	AZ	2001	Diversion	у	n	
	(Soldier						
HORS	Canvon)	CO	2000	Earthfill	n	n	
IMPE	Imperial	AZ	2001	Diversion	n	V	
KACH	Kaches	WA	2002	Earthfill	n	v	
KEE	Keechelus	WA	2001	Earthfill	n	V	
						5	Reregulating
KEGW	Konwick	C ^	1000	Concrete	N/	N/	afterbay for
NL3W	IVE2MICK	UA.	1999	CONCIECE	у	У	Reregulatino
KORT	Kortes	WY	2005	Concrete	у	n	reservoir.
OLY	Olympus	CO	2006	Earthfill	у	n	
PARK	Parker	AZ	2001	Concrete	у	У	
PATH	Pathfinder	WY	2005	Concrete	n	n	

Table 1. Reservoir tailwaters which were sampled during study years 1999-2006.

PERC	Percha	NM	2000	Diversion	n	n	Primarily
							earthfill, with
PUEB	Pueblo	СО	1999	Concrete	n	v	a concrete mid-section.
RED	Red Bluff	CA	1999	Diversion	n	v	
ROZA	Roza	WA	2002	Diversion	у	n	
RUED	Ruedi	CO	2001	Earthfill	n	n	
SEMI	Seminoe	WY	2005	Concrete	у	у	
SHAD	Shadow Mtn	CO	2000	Earthfill	n	n	
SHAS	Shasta	CA	1999	Concrete	у	У	
STAR	Starvation	UT	2001	Earthfill	n	n	
SUGR	Sugarloaf	CO	1999	Earthfill	n	n	
SUMN	Sumner	NM	2000	Earthfill	n	У	
SUNN	Sunnyside	WA	2004	Diversion	n	n	
TEIT	Tieton	WA	2002	Concrete	n	У	
TWIN	Twin Lakes	CO	2000	Earthfill	n	n	
WHIS	Whiskeytown	CA	2000	Earthfill	у	У	
WILL	Willow Crk	CO	2000	Earthfill	n	n	

## Data Analysis

Constrained ordination techniques (CANOCO 4.5) were used to examine gradients in benthic data and to identify environmental variables most closely associated with invertebrate distributions. It has been recognized that no single method of ordination is best for describing multivariate data sets. Recently, Ruokolainen and Salo (2006), in a comparison of ordination techniques (including Non-metric Multidimensional Scaling (NMS)) using field data, found little evidence that one method was better than another. Initial analyses of data using detrended correspondence analysis (DCA) revealed that the data set had a relatively long gradient length (greater than 3), suggesting that analysis using unimodal models was appropriate. Infrequent taxa (<3 individuals) were deleted and faunal data transformed (square root) before analysis.

Environmental variables were normalized, if needed, with ln (X+1) transformations. If environmental variables were strongly correlated (Pearson correlation, r>0.6), only a single variable was selected for use in CCA to avoid problems with multicollinearity. Forward selection of environmental variables and Monte Carlo permutations (1000 permutations) were used to determine whether variables exerted a significant ( $p \le 0.05$ ) effect on invertebrate distributions. In the ordination diagram, taxa and sites are represented by geometric symbols and environmental variables by arrows. Arrows orient in the direction of maximum variation in value of the given variable.

Because data were absent from most sites, hydrologic indices were examined on their own with the macroinvertebrate data set, then significant indices were presented as supplementary data in the final analysis that incorporated all of the significant variables.

One-way ANOVA followed by Tukey's post-hoc test was used to compare mean richness metrics and abundance measures for different dam heights. Pearson correlation was used

to test for relationships between some variables. Variance is presented as standard error (SE).

Standard metrics that made use of community response (taxa richness) along with the indicator group of EPT (members of the orders Ephemeroptera, Plecoptera, and Trichoptera) were initially used for metric development. A biological metric related to disturbance from dams (Biotic Dam Index, BDI) was developed from these benthic metrics along with biological data that were significant in CCA and correlation analyses. A 1-3-5 scoring system was used with a score of 1 being a low biotic value and 5 being high. Multiple metrics were then added together for final scoring. Data were defined for categories through the use of percentiles, with  $\leq 25^{th}$  percentile, 26-74<sup>th</sup> percentiles, and  $\geq 75^{th}$  percentile defining three different classifications. The BDI is based on redundancy, with subsets of taxa organized into groups associated with quality of the below dam ecosystem. For example, some low quality sites could have high taxa richness, but low numbers of EPT taxa. High taxa richness represents the bias towards biodiversity that is important for protecting ecosystems in the long-term, while EPT richness represents a group of taxa that are considered sensitive to anthropogenic impacts, including dams (e.g., Rehn et al., 2008).

### RESULTS

A total of 104 taxa were identified from the 43 sites from which data were collected (Appendix A). This appendix also contains the code names for the various taxa that are found in succeeding analyses. Dam height ranged from 2.4 to 183 m, age from 11 to 97 years, and reservoir surface area up to 17,005 ha (Appendix B). Water quality as indicated by DO appeared to be relatively high with only two sites having fairly low values of 2.6 mg/L at Bonny and 3.2 mg/L at Elephant Butte. Other values were higher, with an overall average DO of 8.3 mg/L in Reclamation tailwaters.

Dam height was positively correlated (r>0.6, p<0.05) with whether hydroelectricity was generated at the dam, type of dam construction material, and surface area of the reservoir; while conductivity was positively correlated (r>0.6, p<0.05) with temperature, alkalinity, and hardness. Substrate index was highly negatively correlated with % sand. An example of how dam height and dam type are related is shown in Figure 1. After removal of correlated variables, the initial set of variables in the CCA model included: DO, SI, velocity, dam age, elevation, conductivity, dam height, Pfankuch Index, stream width, and periphyton biomass. CCA analysis suggested differences between aquatic invertebrate communities could be explained by species–environment relationships with conductivity, velocity, stream width, elevation and dam height significant (p<0.05) in the model (Figure 2). The statistical test for axes was significant (F-ratio=1.533, p=0.001) for all canonical axes. The first two axes accounted for 10.3% of taxa data variation and 21.5% of species-environment relationships. Geographic variation in taxa from this wide ranging data set is likely responsible for part of the relatively low % of variation explained. Eigenvalues for Axis 1 were 0.57 and for Axis II, 0.477.



Figure 1. Association between dam type and dam height. Data are presented as the mean  $\pm 1$  SE.

The CCA analysis indicated that water quality was the most important driver at sites below dams, with high conductivity sites separated from the others. Some of this was related to longitudinal difference as represented by increased stream width associated with higher order rivers. It appeared that low conductivity sites also had higher velocities (perhaps because they were mostly found at high gradient sites associated with snow melt runoff). Elevation (which was not correlated with dam height) suggests that low and high conductivity sites both experienced changes in invertebrate assemblages as dam height increased. The relationships that were detected appeared to be largely related to specific attributes of dams rather than dam sites (i.e., elevation).



Figure 2. CCA triplot showing relationships between sites (filled circles), taxa (triangles), and environmental variables (arrows). The supplementary variable FH1 is also shown.

Only a single hydrologic metric,  $F_H1$ , was significant in the CCA and is presented as supplementary data in Figure 2. Just the first axis was canonical with 3.9% of the taxa data variation explained. The eigenvalue for the first axis was 0.247.  $F_H1$  is the number of high flood pulses and can be interpreted as a measure of the disruptive stress caused by flooding. Highest  $F_H1$  values were found below dams on the lower Colorado River at Imperial, Parker, and Davis and the significance in the model may be a result of these irrigation associated facilities that are subjected to large changes in flows (see Appendix B).

Velocity was significant in the CCA model along Axis I (Figure 2). High velocity taxa such as the mayflies *Serratella* and *Drunella* were found along the positive portion of Axis I, while those associated with more lentic conditions such as *Tricorythodes* were in the negative portion of the diagram (Figure 3). Non-insects like *Hydra* and *Gammarus* 

also appeared in this portion of the diagram. Velocity was not significantly correlated with invertebrate taxa richness (p=0.3804) or abundance (p=0.1203) but was negatively correlated with periphyton (r=-0.3301, p=0.0306). Periphyton was not identified as a significant variable in the CCA model but was positively correlated with reservoir surface area (r=0.4554, p=0.0021), negatively correlated with taxa richness (r=-0.3108, p=0.0425) (Figure 4), and positively correlated with the proportion of non-insect abundance (r=0.4057, p=0.0069). Non-significance of periphyton in the CCA model was likely caused by the association of periphyton with other significant variables, such as velocity that were in the model. Periphyton was not significantly correlated with measures of substrate such as S.I. or percent sand (p≥ 0.1935).



Figure 3. Velocity (m/S) contours from CCA analysis. Contours are shown in relation to taxa.



Figure 4. Relationship between periphyton biomass and taxa richness.

While periphyton increased with decreased velocities it also increased with dam height (r=0.3011, p=0.0498). The CCA association of periphyton biomass with the taxa found below dams is shown in figure 5.



Figure 5. Periphyton biomass  $(g/m^2)$  contours from CCA analysis.

Dam height was significant along Axis II and the contours associated with this analysis are presented in Figure 6.



Figure 6. Dam height (m) contours from CCA analysis.

Macroinvertebrate taxa richness was negatively correlated with dam height (r=-0.3863, p=0.0105) while the proportion of non-insect (e.g., *Hyalella* and Cladocera) abundance was positively correlated with dam height (r=0.4161, p=0.0055). When dam height was grouped into 25<sup>th</sup> percentiles, the corresponding taxa richness values for the groups were significantly different (ANOVA, p=0.0295) from low dams to high dams (Figure 7). There were no significant differences in abundance (p=0.3417) between dam height groups. This was likely because of replacement of insects with non-insects as dam height increased.



Figure 7. Mean macroinvertebrate taxa richness  $\pm 1$  SE below dams of different heights. Bars associated with a richness level with the same letter indicate no significant difference between mean values, while those with different letters are significantly different.

EPT richness also differed between dam height groups but discriminated significantly (ANOVA, p=0.0143) between dam heights in both the 43-74 m and >74m groups (Figure 8) when compared to the 0-21 m group.



Figure 8. Mean EPT richness  $\pm$  1 SE found below reservoirs with different dam heights. Height groups with similar letters were not statistically different.

Using information from this study, a biological metric, the Biotic Dam Index (BDI), was developed with taxa and EPT richness, proportion of non-insect abundance, and periphyton biomass. The metrics and derivation of index values are presented in Table 2. The ultimate metric BDI could range from 4 (LOW) to 20 (HIGH) as a final score. BDI values for individual dams are presented in Appendix B.

Metric	Response to		Scoring Boundaries	5
	degradation	1	3	5
Taxa richness	Decrease	<u>&lt;</u> 7	8-14	<u>&gt; 15</u>
EPT richness	Decrease	<u>&lt;</u> 1	2-4	<u>&gt;</u> 5
Proportion non-	Increase	0.33-1	0.07-0.34	<u>&lt; 0.06</u>
insect				
Periphyton	Increase	<u>&gt; 51</u>	7.0-50	<u>&lt;</u> 6.9
biomass (g/m <sup>2</sup> )				
BDI	Decrease	<u>&lt;</u> 10 (LOW)	11-13 (MOD)	≥ 14 (HIGH)

Table 2.	Metrics	used to	derive	the	Biotic	Dam	Index	(BDI).
1 40010 20	1.1.0.1.0.0				210010			()

ANOVA comparing variables at the three different scoring levels (LOW, MOD, and HIGH) for BDI indicated that there were no significant differences ( $p \le 0.05$ ) between the categories for DO, FH<sub>1</sub>, pH, temperature, dam age, velocity, stream width, or elevation. Conductivity, however, differed significantly between categories (ANOVA, p=0.0107) with LOW-BDI sites having higher conductivity values than HIGH-BDI sites. While it might be expected that low elevation sites would tend to be associated with higher temperatures, finer substrates, and inherently lower biotic values, and that high elevation sites (sometimes associated with incised canyons suitable for higher dams) with higher gradients and coarser substrates might have higher values of BDI, this sort of pattern was not seen with BDI, suggesting that the observed results were related to effects from dams. Once again it should be noted that there were no significant relationships between BDI and temperature (p=0.1014) or elevation (p=0.9725), underlying geographic factors that might be expected to influence biological metrics on a large scale.

There were significant differences between BDI categories when important dam characteristics were examined. Dam height differed significantly (ANOVA, p=0.0231), with LOW-BDI and HIGH-BDI categories different from each other while the MOD-BDI category did not differ from either of the other categories. Surface area also differed significantly (ANOVA, p=0.0001) with LOW-BDI differing from the other two categories which did not differ from each other. The BDI was significantly correlated with dam height (r=-0.3921, p=0.0093) but was more highly correlated with reservoir surface area (r= -0.5768, p= 0.0001) (Figure 9). Examination of Figure 9 suggests that there are 3 outliers for dams that create reservoirs that are less than 2000 ha in size. All three of these reservoirs, which include Bonny, Headgate Rock, and Ruedi, have BDI values of 6. Bonny has a very low DO concentration that could affect BDI. BDI may be low at Ruedi because while it has a small surface area (403 ha) the dam is high (87 m). It is unclear why the BDI at Headgate Rock is so low, but as previously noted this site has a large number of high flood pulses.

It appears then, that the BDI is sensitive to dam impacts across a range of water quality and geographic-based variables. The value of this index is that it allows for an ecological based classification that can be used to compare Reclamation tailwaters and perhaps to determine whether adaptive management schemes used at individual dams are successful in altering biota. There may also be value in utilizing this metric to predict impacts of new dams on biological attributes.



Figure 9. Relationship between reservoir surface area and BDI.

#### DISCUSSION

The geographical location of a dam in the environment appeared to play a large and not unexpected role in the presence of specific taxa found in macroinvertebrate assemblages, with velocity, conductivity, and stream width of primary importance and elevation of secondary importance in the multivariate analysis. This likely represents the overriding template of longitudinal variation that affects all invertebrate communities. Dam height was the most important variable directly related to river regulation, and was also highly correlated with surface area, construction materials, and whether the dam was used in production of hydroelectricity. Height was also positively correlated with periphyton biomass which may be associated with increased nutrient concentrations that are sometimes found below deep release dams (e.g., Camargo et al., 2005). There appeared to be changes in macroinvertebrate assemblages that corresponded with increases in noninsect abundance and decreased taxa and EPT richness as dam height increased. A recent study of aquatic macroinvertebrates below hydroelectric projects in California also found metrics such as EPT richness to be helpful in discrimination of impacts from dams (Rehn et al., 2008). The biological metric, BDI was used to incorporate a variety of biotic values and could be used as an ecologically based classification of dam impacts. The relationships that were detected appeared to be largely related to dams (height and surface area) rather than dam sites and it appeared that the metric may be useful in comparing dams from a variety of geographic settings. It is expected that BDI would

respond to changes in operations at individual dams, however, this remains untested at this time.

Hydrologic metrics played a small role in describing the macroinvertebrate distributions with only  $F_{\rm H}1$  of significance in the multivariate analysis. Hydrologic metrics below dams are profoundly altered in similar ways when compared to "natural" flows (e.g., Magilligan et al., 2003) and this may explain the lack of importance of these metrics in the analysis. Other investigators such as Poff et al. (2007) also indicate that flow variables are homogenized below dams. Dam height represents a multitude of impacts which include temperature and thermal regime modification, sediment transport, hydraulic residence time (likely related to reservoir surface area and volume), and water quality (e.g., Poff et al., 2007). As such, dam height may be a suitable general metric for describing impacts of dams in the environment. It is unfortunate that most of the literature describing impacts of dams to aquatic invertebrates does not report this information. Surface area, however, which was also correlated with BDI is more often reported.

It seems that, because of the multivariate nature of impacts which increase with dam height, small changes in operations will not result in improvements to aquatic invertebrates below high dams. Changes to a single attribute, like flow, are unlikely to positively alter temperature or, in many cases, sediment availability or transport. Reregulation of rivers below dams has been hypothesized as a way to sustain the natural attributes of rivers (Stanford et al., 1996). However, actual tests of flow restoration and altering temperatures by selective withdrawal have not demonstrated any unambiguous successes. Vinson (2001) found that installation of a multi-level outlet which increased water temperatures (but did not restore the thermal regime) did not improve taxa richness below Flaming Gorge Dam. Little change in benthos was observed below Tennessee Valley Authority dams with changes in flow, until water quality was also improved (Bednarek and Hart, 2005). Multiple experimental floods were used to alter macroinvertebrate communities below the Livigno Reservoir in the River Spöl in Switzerland. Despite the 15 floods between 2000 and 2006, it was believed that macroinvertebrate assemblages still had not achieved the level of pre-regulated conditions (Mannes et al., 2008). Rader et al. (2008) had similar experience with implementation of floods below a storage reservoir on the Colorado River. They found that occasional floods did not restore macroinvertebrate biodiversity in the system. Moyle and Mount (2007 direct attention to the false perception that flow regime alteration can result in large environmental benefits, and that this benefit can be achieved at a low monetary cost. Part of the problem may be that when natural flow regimes are mimicked, much lower volumes of water are used (e.g., Moyle and Mount, 2007) compared to flows prior to damming.

Difficulties in ecological management of high dams are demonstrated in the results of Jackson et al. (2007) who compared regulated and unregulated rivers at sites in Scotland. They suggested that both temperature and discharge metrics were equally important in explaining differences in macroinvertebrate communities. Krause et al. (2005) found modifications that would result in desired flows often resulted in undesirable

temperatures. Magilligan et al. (2003) presents data that indicates there is a critical threshold that corresponds to the periodicity of the pre-dam 5 year flood. The greater flows deviate from this criterion, the greater the impact to riverine ecology. It is unclear whether there are adaptive management schemes that could implement these sorts of ecologically important flows. Much of the present literature seems to suggest that there is an inability to control flows to the degree necessary for biological improvement. At least in some cases, large flows are constrained by power plant capacity (e.g., Kearsley et al., 1994) in most years. It is also likely that limited changes in flows or temperatures will have little effect on periphyton biomass which may continue to have large impacts on macroinvertebrate assemblages. Environmental flows from Bendora Dam in Australia were not successful in recreating periphyton conditions similar to unregulated streams in the region (Chester and Norris, 2006). Scrapers such as heptageniid mayflies, which feed on periphyton, are often decreased below reservoirs because, although periphyton biomass is increased, epilithic diatoms (their food source) are often diminished. Decreased disturbance from flow events was believed to have resulted in altered periphyton communities below dams along the Cotter River in Australia. The macroinvertebrate community was also changed because of dietary requirements, with diatom feeders such a Leptophlebiidae, Glossosomatidae, and Elmidae negatively impacted below dams where filamentous algae was abundant (Chester and Norris, 2006). There may also be cases where, despite the return of parameters to their original values, the benthic community may not return to its original state because of hysteresis (e.g., Beisner et al., 2003); instead residing in an alternative stable state that may be difficult to change.

Large dams have the greatest impact on the environment, and recovery below dams such as Glen Canyon (65,000 ha surface area; no recovery at a point 387 km below the dam; Stevens et al., 1997), Flaming Gorge (17,000 ha surface area; recovery at between 69-125 km; Vinson, 2001), and Barren River Lake (4,000 ha surface area; incomplete recovery at 21.1 km; Novotny, 1985) demonstrate the extent of the impacts.

Conversely, downstream macroinvertebrates associated with smaller reservoirs (110-353 ha surface area) may recover relatively quickly from the effects of dams, with recovery distances reported from 2.6 to about 8.5 km downstream of the dam (Ward, 1974; Petts et al., 1993; Imbert and Stanford, 1996). Differences in recovery rates demonstrate the importance of height and surface area as metrics for describing impacts from dams. Rehn et al. (2008) found that, in most cases, macroinvertebrates from sites below diversion dams did not differ from those at reference sites unassociated with dams. The rapid recovery and relatively high taxa richness found below lower height dams suggests that modification of these structures may not have a very high benefit/cost ratio, at least in the case of aquatic macroinvertebrates. Recent literature related to dam removal (e.g., Maloney et al., 2008) suggests that this is indeed the case, with findings of no changes in macroinvertebrate assemblages after low-dam removal. Low dams (<5m in height), however, are often the focus of dam removal. This is probably not related to removing dams with the greatest impact to macroinvertebrate communities but is instead associated with ease of removal (economic factors) and because many of these dams have been abandoned by their previous owners (e.g., Poff and Hart, 2002). Impacts from removal

of high dams, such as sediment loading, may make the public less than eager to embrace their removal.

In the absence of improvements in biodiversity below large dams through management of small changes in flows or temperatures and the relatively moderate effects caused by low dams, it may be that avoidance of environmental damage should be through careful placement of new dams and improvement of existing dam projects. Recovery distance can be mitigated by placing dams just above where an unregulated tributary flows into the impacted river (Munn and Brusven, 1987) allowing for recovery of flow and temperature characteristics. McAllister et al. (2001) suggest that one way in which to avoid additional environmental damage is to upgrade and thus boost the performance (e.g., better turbines) of existing projects. They suggest that this increased performance would come at little environmental impact and possibly avoid construction of new environmentally damaging projects.

Whatever options there are to restore ecologically sustainable conditions in rivers below dams, data collected during this study indicates that aquatic macroinvertebrates could play a role in any monitoring activities. The linkage between increasing dam height/surface area and biological metrics demonstrates the value of this monitoring agent.

## ACKNOWLEDGEMENTS

I thank the S&T program for providing funding in a variety of forms over the years. David Raff was instrumental in obtaining a portion of the funding used to sample below dam environments. Special thanks to Brian Bledsoe for analyzing and providing information on hydrologic indicators. Thanks to Doug Andersen and Jim Yahnke for reviewing an early draft of the manuscript.

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	10					0	01	8	1785	8908	8	59.9	10	5.4	14	8	0.69	135	105	16.5	375	7.46	8.27	PATH ,
	12	4.25	0.52018341	0.39969713	0.25059335	-	3	67	1679	666	81	24.3	20	4.6	64	ß	0	148	117	18.4	419	7.92	8.19	100 0
	14					0	3	45	1425	73.6	10.8	52.7	10	6.3	37	46	0.32	139	108	18.7	376	8.65	8.53	3RAY C
	18					_	3	57	2280	75	21	6.4	10	5.3	12	49	0.48	18	23	13.74	58	7.35	7.14	JLY