Alabama Mill Dam Inventory

Final Report

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Prepared by

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Project Synopsis

This study began in 2006 as a cooperative study between Auburn University and the Alabama Department of Conservation and Natural Resources. This study, the Alabama Mill Dam Inventory (AMDI) had three major goals: 1) To produce an online, geo-referenced data base of mill and low head (< 10 m) dams in Alabama, 2) To examine the effect of dams on local physical habitat conditions and biotic assemblages across a physiographic gradient, and 3) To provide resource managers with a tool for identifying, evaluating, and prioritizing potential habitat restoration projects in Alabama.

Our study had a very broad geographic scope (e.g., Alabama) and therefore a high potential to benefit conservation of nearly all sensitive (i.e., Greatest Conservation Need or GCN) aquatic taxa in the state. Small dams are found in nearly every stream in Alabama. Our surveys focused on streams and smaller rivers (i.e., $\leq$ 6th order) because many larger systems have been inexorably altered by larger dams.

We quantified mollusk, crayfish, and fish assemblages at 22 small Alabama dams. Dams were categorized status as intact (N = 12), breached (N = 5), or relict (N = 5). Our data revealed that breached dams may have a more substantial negative effect on a wide array of stream biota. Most surprisingly, we found very few mussels in streams with breached dams. Streams with intact or relict mill dams were much more likely to support abundant and diverse mussel assemblages than were streams with breached dams. Fish assemblage data mirrored the mussel data. Streams with breached dams tended to support widely divergent fish assemblages across sites. Crayfish were less abundant in streams with breached dams than in

streams with intact or relic dams. Our data demonstrate that stream biota may exhibit a broad range of responses to small dams.

Degraded or poorly constructed dams are a hazard not only to recreational activities but they may threaten survival of sensitive aquatic taxa. As such they make excellent subjects for restoration projects. In contrast, removal of more benign structures may pose significant risks to streambed stability and sensitive species persistence. When designing restorations in systems supporting at-risk freshwater biota, extensive case-by-case evaluations will be needed to weigh benefits and costs of dam breaching or removal projects.

Chapter 1: Geospatial attributes of mill and lowhead dam occurrence and persistence in Alabama

Contributing Authors: Michael Gangloff and Kevin White

Abstract

Mill dams and other and low-head impoundments were historically widespread in small rivers and streams across Alabama and the southeastern U.S. We compiled an extensive electronic database of all known dam sites in Alabama. Field surveys (ground-truthing) indicated that relatively few historic dams remain, most notably on the Gulf Coastal Plain. Few previous studies have addressed linkages between landscape-scale factors and dam persistence. We examined differences in land-use/land-cover at 30 small dams across Alabama. We condensed land-use/land-cover attributes to 6 primary categories. Surprisingly we did not find any differences in upstream landuse categories among streams with breached, intact, or relict dams. Future studies should focus on more discrete physiographic area to better understand how changes in land use and landforms can influence stream geomorphic processes.

Introduction

Dams are numerous and widespread in many southeastern U.S. streams, with >10,000 dams in Alabama alone (AL Office of Water Resources http://www.adeca.alabama.gov/) and up to 44% of the mainstem Alabama, Coosa, and Tallapoosa Rivers being impounded (Irwin *et* al. 2007). Physical,

chemical, and biological impacts of dams can be dramatic, and may include altering flow and sediment regimes and channel geomorphology, and blocking upstream migration by fishes and other mobile organisms (Baxter 1977; Blalock and Sickel 1996; Watters 1996; Dean *et al.* 2002; Poff and Hart 2002; McLaughlin *et al.* 2006).

The effects of large dams on aquatic organisms and their habitats in large rivers have been well documented, whereas little research has been done to assess effects of small, surface-release, or low-head dams on low-order streams (Watters 1996; Dean *et al.* 2002; Lessard and Hayes 2003). Low-head dams are dams with a hydraulic height of $\leq$10 m and are typically overflow or spill-way structures (IFC Consulting Report 2005). According to census records, >65,000 low-head dams existed in the eastern US by 1840, most of which were built for water-powered milling (Walter and Merritts 2008). The few studies designed to examine effects of small dams have reported similar types of alterations, but are smaller in magnitude than those of large dams (Graf 2006).

Methods

Geospatial analyses

We compiled dam locality records from the Geographic Names Information System (GNIS) online database (http://geonames.usgs.gov/), USGS topographic maps, and Alabama Geological Survey publications. We selected 30 focal sites for more detailed land-use analysis. We measured primary stream physical macrohabitat parameters (i.e., physiographic province, elevation, gradient, stream link and rank magnitude) using

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topographic maps and Arc-GIS software. Sites were classified as intact, breached or relict based on the proportion of the dam remaining in the stream channel. Breached sites were dams where >30% of the channel is still restricted by a dam or its foundations whereas relict sites had <30% of the channel restricted. Intact sites had functional spillways and over-dam flow for much of the year.

Land cover detection was performed with the utilization of unsupervised classification methods and an ISODATA algorithm (ERDAS Imagine Atlanta, GA). 2001 Landsat ETM+ imagery was acquired for each mill dam site from Earth Explorer (USGS). Each image was taken between mid-October and early November to avoid seasonal bias. The raw band data were then merged with the use of MultiSpec (Purdue University, West Lafayette, IN). The panchromatic (Band 7) and the thermal band (Band 6) were not merged to save processing time. Next, a subset covering each mill dam site was created within ERDAS Imagine. Each subset was then geo-rectified, using image to map rectification. Once the images were rectified ArcGIS was utilized to delineate each mill dam site. Each mill dam was the pour point and delineations upstream of every mill were created with ArcHydro tools. The delineations were then used as a mask to clip the Landsat images to watershed scale.

After image processing was complete, an unsupervised classification method was used with the common ISODATA algorithm. Fifty iterations and 200 classes were created. Each 30m x 30m pixel was then referenced with the use of aerial photography from 2002. Aerial photos were taken 6-8 months after each Landsat image. This accuracy assessment resulted in an

overall accuracy of 88.91%. Furthermore, the classification results were compared to the National Land Cover Data Set from USGS (NLCD 2001). Subsets of the NLCD 2001 from Earth Explorer (USGS) were masked and clipped to each watershed. The NLCD, which also utilized unsupervised classification, proved to be more accurate than the previously calculated land cover. Consequently, the clipped data from the NLCD were utilized and spatial statistics were calculated for each delineation. This resulted in 15 classes. Each class was represented in acres.

Because some land use classes represented a relatively small proportion of upstream landuse, we grouped classes into 6 major landuse categories: % Forest, % Pasture and shrub land, % Row Crop Agriculture, % Urban, and % Wetlands and open water. We used arcsine (square root (X)) to transform proportional data prior to analyses. We used ANOVA to examine differences in upstream watershed land use across streams with breached, relict and intact dams.

Results

Dam and stream habitat/biota databases

All physical, chemical, and biological data collected on this project accompany this report in Appendeces A (Physico-chemical and invertebrate data) and B (fish data). Additionally, one of the sheets in Appendix A includes upstream landuse data for each focal dam site. We compiled records for dam records in Alabama into an ARC-GIS database. Dam data and shapefiles will follow on an accompanying CD (Appendix C).

Landuse and dam persistence

The status of the 28 small dams in our analyses was not related to upstream landuse characteristics. We did not observe any statistically significant differences between broad-scale landuse categories and dam persistence. Subsequent analyses may need to refine landuse categories or consider historical conditions.

Discussion

It is likely that the sample sizes used in this analysis were not large enough to encompass meaningful variation in land-use/land-cover attributes. More focused studies within a single or a subset of Alabama's diverse ecoregions may be better able to elucidate how interactions between landuse, geomorphic change and hydrology may affect dam failure.

This aspect of the study was instigated by a graduate student that wound up switching to the geography program at ASU. Right now, further development of the landuse dataset is on hold pending recruitment of additional GIS expertise.

Literature Cited

Baxter, R.M. 1977. Environmental effects of dams and impoundments. *Annual Review of Ecology and Systematics* 8:255-283.

Blalock, H.N. and J.B. Sickel. 1996. Changes in mussel (Bivalvia: Unionidae) fauna within the Kentucky portion of Lake Barkley since impoundments. *American Malacological Bulletin* 13:111-116.

Dean, J., D. Edds, D. Gillette, J. Howard, S. Sherraden, and J. Tiemann.

2002. Effects of lowhead dams on freshwater mussels in the Neosho River, Kansas. *Transactions of the Kansas Academy of Science* 105:323-240.

Graf, W.L. 2006. Downstream hydrologic and geomorphic effects of large dams on American rivers. *Geomorphology* 79:336-360.

Irwin, E., G. Turner, K. Mickett, and T. Piper. 2007. ACT Aquatic Gap and water quality monitory: Final Report. ADCNR.

Lessard, J.L. and D.B. Hayes. 2003. Effects of elevated water temperature on fish and macroinvertebrate communities below small dams. *River Research and Applications* 19:721-732.

McLaughlin, R.L., L. Porto, D.L.G. Noakes, J.R. Baylis, L.M. Carl, H.R. Dodd, J.D.Goldstein, D.B. Hayes, and R.G. Randall. 2006. Effects of low-head barriers on stream fishes: taxonomic affiliations and morphological correlates of sensitive species. *Canadian Journal of Fisheries and Aquatic Sciences* 63:766-779.

Poff, N.L., and D.D. Hart. 2002. How dams vary and why it matters for the emerging science of dam removal. *BioScience* 52:659-668.

Watters, G.T. 1996. Small dams as barriers to freshwater mussels (Molusca: Pelecypoda: Unionidae) and their hosts. *Biological Conservation* 75:79-85.

Walter, R.C., and D.J. Merritts. 2008. Natural streams and the legacy of water powered mills. *Science* 319:299-304.

Chapter 2: Effects of small dams on stream physicochemical habitat parameters and mollusk assemblages

Contributing Authors: Michael Gangloff, Emily Hartfield, David Werneke, Kevin White, Jack Feminella

Abstract

We examined associations between intact, breached, and small dams and stream physicochemical habitat parameters at 3 sites in each of 22 low-to mid-order Alabama streams. We measured physical habitat parameters (stream depth, flow velocity, channel width, and sediment size and composition) along 15 or 16 cross-channel transects at each site. Water chemistry parameters were measured at summer base flow conditions. Streams with intact mill dams tended to have habitat and water chemistry conditions (velocity, total N) that were similar to streams with relic dams. In contrast, habitat conditions in streams with breached dams were often very different from streams with intact or relic dams. We used principle components analysis (PCA) to examine variation in physicochemical conditions across all 66 sites. PCA reveled that 6 principle components account for only ~70% of the variation in habitat conditions at these 66 sites. Two PC scores (corresponding to substrate size and nitrogen enrichment) were strong predictors of mussel abundance, richness and density. However, mussel abundance data do not appear to be linearly distributed. Instead the data reveal a constraint-envelope pattern.

Introduction

Impoundments are widely recognized as having dramatic negative impacts on freshwater habitats. Dams transform upstream reaches to lentic habitats, restrict downstream sediment movement, and dramatically alter other physicochemical characteristics of downstream reaches. In Alabama, many of these impacts are associated with large, hydroelectric dams. However, low-head dams (i.e., those <10 m height) greatly exceed hydroelectric dams in number across the state and thus affect a much broader range of stream sizes. Recent surveys suggest that many high-conservation priority fish and mollusk taxa are now restricted to tributary streams and small rivers, so an understanding how low-head dams affect stream biota is critical to conservation and recovery of these imperiled taxa.

Perhaps the most obvious and important effect of large dams on streams is the resulting change in the hydrologic regime, which can alter channel geomorphology, water temperature and chemistry, and availability of biotic habitat. Depending on how water levels are managed, dams tend to alter both the magnitude and number of minimum and maximum flow events (Poff *et al.* 1997). Dams managed for hydroelectric energy production often dramatically alter both the frequency of high-flow events and rate of change in flow (flashiness) increases. Additionally, timing of seasonal high and low flow events may be altered, resulting in more predictable and less variable flow regimes (Magilligan and Nislow 2005; Graf 2006). In contrast, small dams, including mill dams and low-head dams, tend to be operated as run-of the river structures (i.e., most of the time water is allowed to pass through or over

the dam). Water releases from mill dams are usually only altered for short durations when power is needed to operate machinery.

Large dams have dramatic effects on riverine food webs and nutrient cycling. In systems where high-flow events are rare, connections with and nutrient uptake by flood plain may reduce flood plain–stream nutrient exchange (Welcomme 1975; Baxter 1977; Kingsford 2000; Junk and Wantzen 2004). In contrast, much less is known about how small dams affect stream food webs or energy cycling.

Both large and small impoundments tend to entrain sediments and many downstream reaches become sediment-starved. Reaches downstream of dams typically exhibit increased channel-bed scour, streambed lowering, and bed coarsening (Baxter 1977; Chien 1985; Graf 2005). Moreover, reduced magnitude of high-flow events and decreased deposition causes tailwaters to become less geomorphically complex, with fewer bars and islands and reduced shallow-water habitat (Poff *et al.* 1997; Graf 2006). When dams are breached, sediments are released and may dramatically alter downstream reaches.

Dams also alter natural riverine thermal regimes (Baxter 1977). Reduced current velocity and increased solar inputs within reservoirs can increase surface water temperatures and lead to depth stratification, resulting in a colder but oxygen-poor hypolimnetic zone relative to surface waters (Baxter 1977, Hart *et al.* 2002). Tailwaters downstream of large, hypolimnetic-release dams often have depauperate fish and invertebrate communities (Benz & Collins 1997). In contrast, most mill and low-head dams

are overflow dams and their tailwaters are typically warmer than up-or downstream reaches (Lessard and Hayes 2003).

Increased surface water temperature in overflow dam tailwaters can result in higher densities of primary producers, which may cause increased respiration rates and anoxia or hypoxia in impounded waters (Baxter 1977, Carmago *et al.* 2005). Low D.O. levels can in turn cause fish kills and release of macronutrients normally bound to bottom sediments (Correll 1998). Additionally, the physical force of water flowing over dams may cause oxygen or nitrogen supersaturation in downstream sections. Increases in N_2 concentrations in downstream reaches of >20% have been documented following dam construction and lead to gas bubble disease in fish (Beiningen & Ebel 1968, Morris *et al.* 1968, Rucker 1972, Baxter 1977).

Additionally, both large and small reservoirs act as nutrient sources and exacerbate downstream eutrophication (Wright 1967). Nitrification increases in impounded reaches where decreased current velocity and increased temperature accelerates transformation to NO_3-N (Polak 2004; Straus *et al.* 2004). In contrast, in deeper anoxic sections, accumulation of ammonium denitrification and increases in N concentrations can occur (Allen 1995).

One of the first objectives of this study was to quantify physicochemical habitat differences associated with intact, breached, and relict mill and low-head dams in Alabama streams. We hypothesized that dam condition would have a strong influence on habitat conditions. First, we predicted that reaches below intact small dams would have coarser substrates relative to breached and relict dams. Second, we predicted that temperature and nutrient level

differences would be greater in intact streams than in breached or relict streams.

Methods

At each dam, we established three 150 m study reaches. One reach extended from the dam to 150 m downstream (Mill reach), a second was located >500 m downsteam from the dam, and a third reach was located in a free-flow reach upstream of the impoundment. Within each study reach, we established 16 transects at 10 m intervals. We measured current velocity and depth at 5 evenly-spaced points along each transect. Depth and velocity measurements correspond to points that were later excavated for quadrats (Chapter 4). We measured channel width and substrate composition (20 particles per transect, N = 160 per site) along transects. We computed both the mean and median particle size as well as the proportion of the streambed that was comprised of unmeasured particles (bedrock, organic matter, woody debris, sand and silt).

We continuously measured water temperature (at 3-h intervals) using iButton data loggers deployed at the upstream, downstream, and mill dam sites. A 4th data logger was deployed in the impoundment. We collected grab samples to measure water chemistry during low water conditions in July and August 2007 from each study reach and also from the impoundment. We measured NO_3-N (mass spectrometry, APHA 1998), PO_4 (persulfate digestion and UV analysis, APHA 1998). Additionally, we measured conductivity (C66 Sharp meter), pH (Sharp pH52 meter), and dissolved oxygen (YSI 55 meter) during mid-summer.

Mollusk Surveys

At each dam, we established three 150 m study reaches. One reach extended from the dam to 150 m downstream (Mill reach), a second was located >500 m downsteam from the dam, and a third reach was located in a free-flow reach upstream of the impoundment. Within each study reach, we established 16 transects at 10 m intervals. At each transect we excavated five 0.25 m^2 quadrats spaced equidistant across the channel to a depth of ~10 cm (N = 75-80 quadrats per site, area ~20m^2). All excavated material was passed through a sieve (mesh size = 10 mm) which was successful at retaining unionids down to 6.5 mm total length. All mollusks encountered in quadrats were enumerated and identified to the lowest practical taxonomic level (species for unionids, viviparids, and some pleurocerids, genus for all other taxa). Vouchers of all material, including snails and fingernail clams were retained and deposited in the Auburn University Museum.

We also conducted timed searches in between transects because mussel densities at some of the focal sites were below quadrat detection limits (the minimum density population that could be detected using our protocol is 0.05 mussels/m^2). Visual-tactile searches were conducted by experienced searchers and proceeded in an upstream direction. All mussels found during timed searches were enumerated, measured, and returned to the section of the study reach from which they were removed.

Statistical Analyses

We compared thin-stream physicochemical parameters across streams with intact, breached and relict mill dams. Physicochemical and biotic data

were pooled at the site scale. We used principle components analysis (PCA) to examine physicochemical habitat variability among focal sites. Data were rotated (varimax) and normalized (Kaiser) to maximize explained variance. We examined relationships between habitat PCs and mollusk assemblage metrics using non-parametric correlation coefficients to account for a lack of normality in the data.

Results and Discussion

Principle Components Analysis (PCA) revealed a total of six principle components with Eigenvalues >1.0 (Table 1). Together all six PCs accounted for only ~70% of the variation in among-site physicochemical habitat conditions. PC1 accounted for nearly 20% of the variation in habitat conditions at all sites and is a proxy for substrate size. Non-parametric correlations revealed a significant negative relationship between PC1 and mussel abundance, taxa richness, and density (Fig. 1). PC5 (Total N and NH3 had high factor loadings) explained a greater proportion of the variation in mussel abundance, richness, and density but the relationship was highly non-linear. Examination of nitrate and NH3 profiles for breach and intact dams suggests that small impoundments and associated autotrophs may act as N sinks. Subsequent work will investigate the role of nutrient subsidization by these small impoundments and its potential role in anomalously high growth rates for bivalves living near impoundments.

Literature Cited

Allen, J.D., ed. 1995. *Stream Ecology: Structure and Function of Running Waters*. New York: Chapman and Hall. 388 pp.

Baxter, R.M. 1977. Environmental effects of dams and impoundments. *Annual Review of Ecology and Systematics* 8:255-283.

Beiningen, K.T., W.J. Ebel. 1968. Effect of John Day Dam on dissolved nitrogen concentrations and salmon in the Columbia River, 1968. *Transactions of the American Fisheries Society* 99:664-671.

Carmago, J.A., A. Alonso, M. de la Puente. 2005. Eutrophication downstream from small reservoirs in mountain rivers of Central Spain. *Water Research* 39:3376-3384.

Chien, N. 1985. Changes in river regime after the construction of upstream reservoirs. *Earth Surface Processes and Landforms* 10:143-159.

Correll, D.L. 1998. The role of phosphorus in eutrophication of receiving waters: a review. *Journal of Environmental Quality* 27:261-266.

Graf, W.L. 2005. Geomorphology and American dams: the scientific, social, and economic context. *Geomorphology* 71:3-26.

Graf, W.L. 2006. Downstream hydrologic and geomorphic effects of large dams on American rivers. *Geomorphology* 79:336-360.

Hart, D.D., T.E. Johnson, K.L. Bushaw-Newton, R.J. Horwitz, A.T. Bednarek, D.F. Charles, D.A. Kreeger, and K.J. Velinsky. 2002. Dam removal: challenges and opportunities for ecological research and river restoration. *BioScience* 52:669-682.

Junk, W.J. and K.M. Wantzen. 2004. The flood pulse concept: new aspects, approaches, and applications – an update. Pages 117-140 *in* R.L. Welcomme and T. Petr, editors. *Proceedings of the Second*

International Symposium on the Management of Large Rivers for Fisheries Volume II, FAO, RAP Publication.

Kingsford, R.T. 2000. Ecological impacts of dams, water diversions and river management on floodplain wetlands in Austrailia. *Austral Ecology* 25:109-127.

Lessard, J.L. and D.B. Hayes. 2003. Effects of elevated water temperature on fish and macroinvertebrate communities below small dams. *River Research and Applications* 19:721-732.

Magilligan, F.J., and K.H. Nislow. 2005. Changes in hydrologic regime by dams. *Geomorphology* 71:61-78.

Morris, L.A., R.N. Langemeier, T.R. Russell, and A. Witt, Jr. 1968. Effects of mainstem impoundments and channelization upon limnology of the Missouri River, Nebraska. *Transactions of the American Fisheries Society* 97:380-388.

Poff, N.L., J.D. Allan, M.B. Bain, J.R. Karr, K.L. Prestegaard, B.D. Richter, R.E. Sparks, J.C. Stromberg. 1997. The natural flow regime. *Bioscience* 47:769-784.

Polak, J. 2004. Nitrification in the surface water of the Wloclawek Dam Reservoir: The process contribution to biochemical oxygen demand (N-BOD). *Polish Journal of Environmental Studies* 13:415-424.

Rucker, R.R. 1972. Gas bubble disease of salmonids: a critical review. *U.S. Bureau of Sport Fisheries and Wildlife Technical Paper Number 58*. 11pp.

Strauss, E.A., W.B. Richardson, L.A. Bartsch, J.C. Cavanaugh, D.A.

Bruesewitz, H.Imker, J.A. Heinz, and D.M. Soballe. 2004. Nitrification

in the Upper Mississippi River: patterns, controls, and contribution to

the NO_3^- budget.

Welcomme, R.L. 1975. The fisheries ecology of African floodplains. *CIFA*

Technical Paper Number 3. Rome: FAO.

Table 1. Principle components extracted from the physicochemical data set with eigenvalues >1.0 and component factor loadings. Data were subjected to Varimax rotation with Kaiser normalization. Missing data were replaced by means.

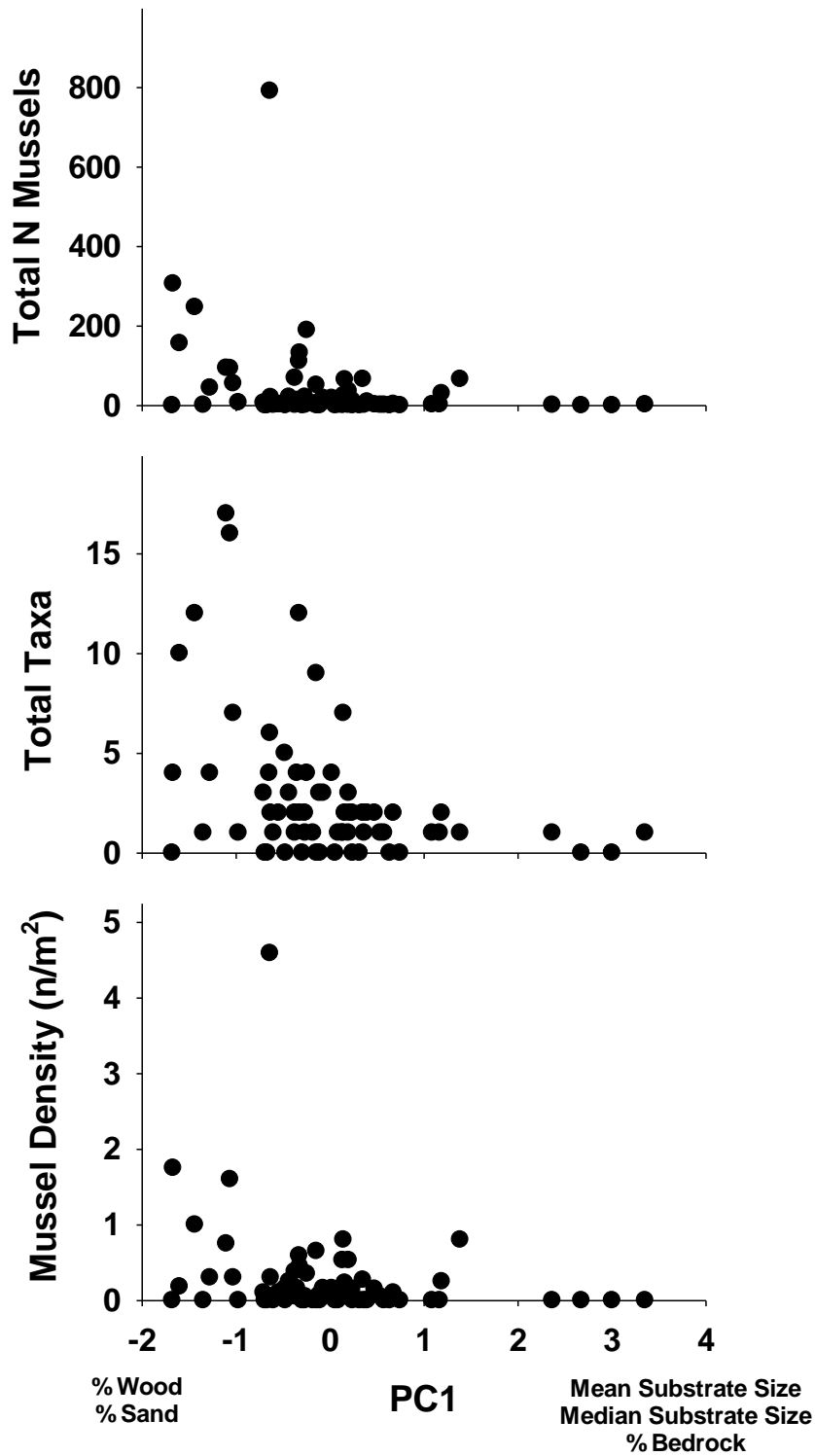| Variable	PC1	PC2	PC3	PC4	PC5	PC6
Width						0.678
Depth						0.821
Velocity			0.862			
Mean Substrate	0.905					
Median Substrate	0.860					
% Bedrock	0.823					
% Wood	-0.540					
% Organic				0.733		
% Sand	-0.515					
% Silt			-0.633			
Conductivity				0.536		
pH		0.864				
Temperature		-0.844				
D.O.			0.645			
Total N					0.772	
Total P				0.730		
NH3					0.825	
% Variance	18.53	13.14	11.17	10.79	10	8.28
Cumulative	18.53	31.67	42.84	53.62	63.62	71.9

Figure 1. Relationship between PC1 (Substrate size) and mussel abundance (upper), taxa richness (middle), and density (lower) at 66 sites near the locations of extant or former small (<10 m) dams in Alabama, 2006-2008.
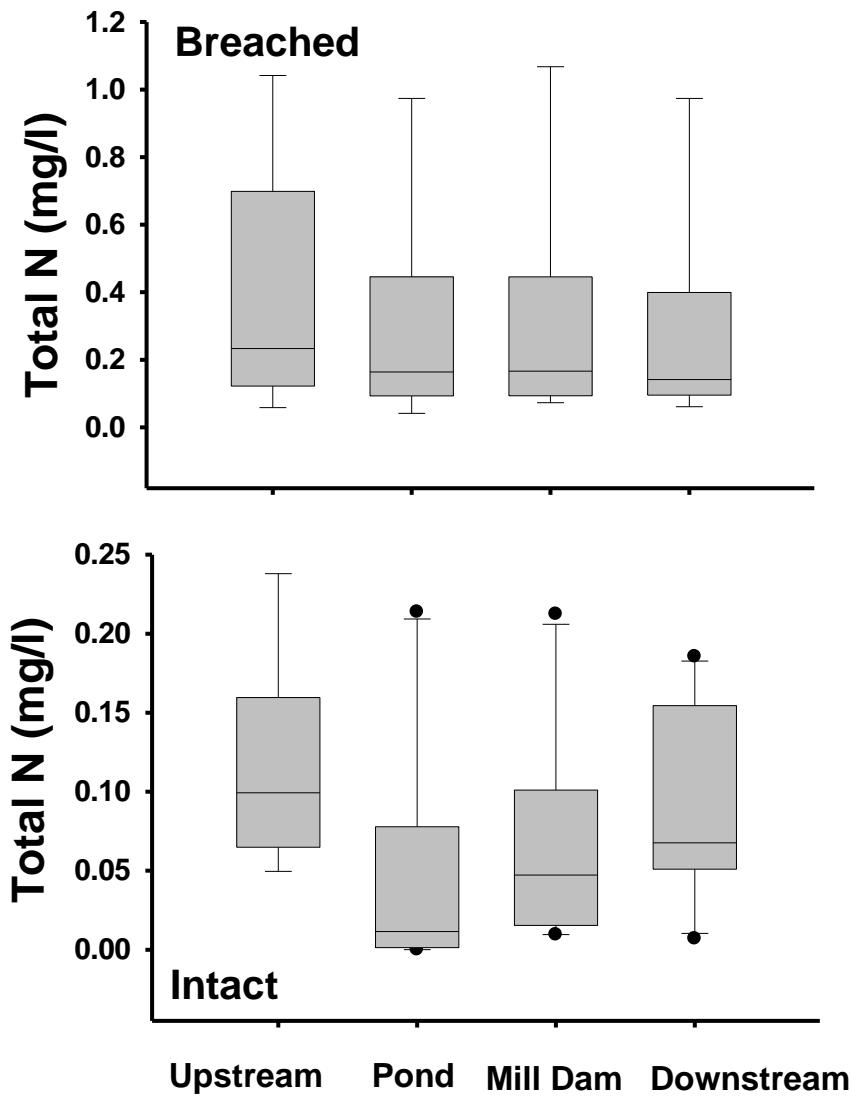
Figure 2. Longitudinal relationship between total Nitrogen (as Nitrate) concentration at in breached (above) and intact (below) stream systems.

Chapter 3: Influence of small dams on freshwater mollusk assemblages in Alabama streams

Contributing Authors: Michael Gangloff, Erin Singer, Emily Hartfield, and Jack Feminella

Abstract

Although the impacts of large dams to fluvial ecosystems are frequently dramatic and have been well documented, impacts of smaller structures remain under-studied. A comprehensive understanding of the impacts of low-head dams (i.e., those <10 m height) is critical to imperiled species management because 1) low-heads greatly outnumber larger dams and 2) dam removal is increasingly part of stream restoration projects. During 2006 and 2007 we quantified responses of invertebrate and fish assemblages at 18 small Alabama dams. We categorized dam status as intact, breached, or relict. Preliminary data indicate that responses ranged widely between taxa and appeared related to dam status. For example, crayfish were less abundant downstream from 16 of the 18 dams but differences were most dramatic immediately downstream of intact dams. In contrast, we measured much greater mussel densities immediately downstream of some intact dams and we found that mussel extirpations were greatest near breached or relict dams. These data suggest that the response of invertebrate populations to small dams and changes in physicochemical conditions varies widely between taxa with different life histories. Further, they suggest that extensive case-by-

case studies are needed to weigh consequences of dam removal for at-risk

freshwater taxa

Introduction

Dams are numerous and widespread in many southeastern U.S.

streams, with >10,000 dams in Alabama alone (AL Office of Water Resources

http://www.adeca.alabama.gov/) and up to 44% of the mainstem Alabama,

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chemical, and biological impacts of dams can be dramatic, and may include

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Blalock and Sickel 1996; Watters 1996; Dean *et al.* 2002; Poff and Hart 2002;

McLaughlin *et al.* 2006).

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large rivers have been well documented, whereas little research has been

done to assess effects of small, surface-release, or low-head dams on low-

order streams (Watters 1996; Dean *et* al. 2002; Lessard and Hayes 2003).

Low-head dams are dams with a hydraulic height of $\leq$8m and are typically

overflow or spill-way structures (IFC Consulting Report 2005). According to

census records, >65,000 low-head dams existed in the eastern US by 1840,

most of which were built for water-powered milling (Walter and Merritts 2008).

The few studies designed to examine effects of small dams have report

similar types of alterations, but are smaller in magnitude than those of large

dams (Graf 2006).

Alabama supports 60% of North America's native mussel species, 38% of native fishes, and 24% of native crayfishes, many of which are endemic to the southeastern US (Lydeard and Mayden 1995; Crandall et al. 2000; Schuster and Taylor 2004). Impoundments have the potential to cause loss of habitat critical for many imperiled species (Travnicheck et al 1995; Jager et al 2001; Dean et al 2002; Lessard and Hayes 2003; Irwin et al. 2007). In the southeastern US, including Alabama, the high prevalence of dams along streams and rivers has the potential to affect many imperiled aquatic species.

Unnatural flow regimes from impoundments are a major source of habitat degradation that may alter stream animal assemblages (Fraser 1972; Cushman 1985; Irvine 1985; Travinchek et al. 1995; Gerhke et al. 2002; McLaughlin et al. 2006) and even riparian vegetation (Janson et al. 2000). Coarsening of the stream bed by erosion of sediment-poor tailwaters reduces habitat availability for benthic species by decreasing habitat heterogeneity, which, in turn, may reduce diversity and richness (Armitage and Blackburn 1990; Hauer et al. 1989; Poff et al. 1997). Alterations in temperature regimes from impoundments also may alter organism distribution and behavior. Increased temperatures downstream of overflow dams can eliminate thermal cues vital to some invertebrate life cycles (Lehmkuhl 1974; Ward and Stanford 1982; Irvine 1985). In addition, increased water temperature can increase metabolic rates for fish and invertebrates, which, in turn, increases demands for food to maintain growth and survival (Gibbons 1976; Wotton 1995; Perry et al. 1987; Vinson 2001; Lessard and Hayes 2003). Within reservoirs, deep, cold, anoxic water often is a fish- and mollusk-free zone (Headrick and Carline 1993, Dean et al. 2002).

Dams may restrict freshwater mussel distributions by impeding migration and distribution of their host fish through impounded sections (Watters 1996; Kelner and Sietman 2000). Studies of mollusk populations in Midwestern streams suggest that low-head dams frequently have negative consequences for mussel assemblages (Watters 1996). In many parts of North America, natural resource agencies have begun aggressively removing mill dams and other low-head structures to restore stream connectivity, and the American Fisheries Society has begun developing protocols for these projects. However, removal of these structures is often politically controversial and some removals have been shown to have negative consequences for mollusk populations in downstream reaches (Sethi et al. 2004).

Methods

Mollusk surveys

At each dam, we established three 150 m study reaches. One reach extended from the dam to 150 m downstream (Mill reach), a second was located >500 m downsteam from the dam, and a third reach was located in a free-flow reach upstream of the impoundment. Within each study reach, we established 16 transects at 10 m intervals. At each transect we excavated five 0.25 m^2 quadrats spaced equidistant across the channel to a depth of ~10 cm (N = 75-80 quadrats per site, area ~20m^2). All excavated material was passed through a sieve (mesh size = 8 mm) which was successful at retaining unionids down to 6.5 mm total length. All mollusks encountered in quadrats were enumerated and identified to the lowest practical taxonomic level

(species for unionids, viviparids, and some pleurocerids, genus for all other taxa). Vouchers of all material, including snails and fingernail clams were retained and deposited in the Auburn University Museum.

We also conducted timed searches in between transects because mussel densities at some of the focal sites were below quadrat detection limits (the minimum density population that could be detected using our protocol is 0.05 mussels/m^2). Visual-tactile searches were conducted by experienced searchers and proceeded in an upstream direction. All mussels found during timed searches were enumerated, measured, and returned to the section of the study reach from which they were removed.

Statistical Analyses

We computed site scale means for all mollusk data. We used total number of mussels as a qualitative measure of mussel abundance but all other metrics were quantitative (mussel density, taxa richness) or semi-quantitative (e.g., mussel CPUE). We examined differences in mollusk assemblage metrics across streams with intact, breached, or relict small dams using ANOVA and LSD post-hoc tests.

Additionally, we examined differences in mussel shell lengths between populations living in the mill reach and those living up-or downstream of the intact dams in 3 east-central Alabama streams. We used 3 *Elliptio* species found in the Tallapoosa and Chattahoochee drainages, *E. arca*, *E. arctata*, and *E. fumata* to determine if mussels living near mill dams attain greater sizes than populations living nearby in the same stream.

To ensure that shell length differences accurately reflect growth rate differences between populations, we thin-sectioned shells from the Sandy Creek focal sites and examined internal growth rings using a light microscope. We examined differences in growth rates between populations by first regressing mussel shell length on age and determining the residual score for each point. We then examined differences in residual scores for each population to determine which had the fastest relative growth rate using ANOVA. Mussels with a positive residual score grew at a rate that was faster than the overall population.

Results

Streams with breached small dams had far fewer mussels than streams with either intact or relict dams. Mussel abundance, mussel taxa richness, and mussel density did not differ significantly between streams with relict and intact dams. However, mussel abundance, richness, and density were all significantly lower in streams with breached dams (Figs. 1-2). In contrast, other native mollusks did not exhibit significant differences among streams (Fig 3).

Mussel shells from 2 of the 3 mill reach populations were significantly larger than shells from up-or downstream populations (Fig. 4). Shell lengths of *Elliptio fumata* in Halawakee Creek did not differ significantly between mill reach and downstream populations. However, *Elliptio arctata* in Loblockee Creek and *E. arca* in Sandy Creek exhibited marked differences in shell sizes across sites (Fig. 4).

Residual scores revealed considerable variation in growth rates both among and across Sandy Creek *E. arca* population (Fig. 5). Standardized residual scores were significantly higher for mussel populations inhabiting the mill reach compared to both up-and downstream reaches suggesting that these individuals grew more rapidly than did individuals from the other study populations (Fig. 6).

Discussion

Our data demonstrate that small dams are not always detrimental to lentic biota. We found that mussels are more abundant in streams with intact dams than in streams with breached dams. Mussel abundance, richness, and density were similar between streams with intact and relict dams but were typically much higher than in streams with breached dams.

Within the three Piedmont streams with intact, historic mill dams, we observed that mussel density and shell length were generally greater in the mill reach than in up or downstream reaches. Halawakee Creek did not exhibit differences in mussel size or density between the mill reach and downstream sites, probably because the downstream reach is still benefited by the mill dam. Further downstream, the bed of Halawakee Creek fills with sand and mussels become very difficult to find (M. Gangloff, pers. obs.). Surveys conducted further downstream from Beans Mill will likely find fewer, smaller-bodied mussels in lower Halawakee Creek.

Taken together, the findings of this study suggest that small dams may have benefits for freshwater mussels although the mechanisms responsible for these benefits remain unclear. Therefore, breaching or removing relatively

benign smaller dams may ultimately prove detrimental to sensitive mussel taxa. Streams that still have intact, older dams also typically supported large mussel populations, likely because the channel is stable across broad temporal and spatial scales. Removing small dams from stable channels may re-invigorate channel down-cutting and translate geomorphic disturbances over broad spatial scales.

This line of inquiry comprises a substantial portion of the research currently being carried out by my graduate assistant, Ms. Erin Singer. Erin will be examining growth rings and monitoring water temperature and food availability to determine why some small dams appear to benefit mussels.

Literature Cited

Armitage, P.D., and J.H. Blackburn. 1990. Environmental stability and communities of Chironomidae (Diptera) in a regulated river. *Regulated Rivers: Research and Management* 5:319-328

Baxter, R.M. 1977. Environmental effects of dams and impoundments. *Annual Review of Ecology and Systematics* 8:255-283.

Blalock, H.N. and J.B. Sickel. 1996. Changes in mussel (Bivalvia: Unionidae) fauna within the Kentucky portion of Lake Barkley since impoundments. *American Malacological Bulletin* 13:111-116.

Crandall, K.A, D.J. Harris, J.W. Fetzner Jr. 2000. The monophyletic origin of freshwater crayfish estimated from nuclear and mitochondrial DNA sequences. *Proclamations of the Royal Society of London* 267:1679-1686.

Cushman, R.M. 1985. Review of ecological effects of rapidly varying flows

downstream of hydroelectric facilities. *North American Journal of Fisheries Management* 5:330-339.

Dean, J., D. Edds, D. Gillette, J. Howard, S. Sherraden, and J. Tiemann. 2002. Effects of lowhead dams on freshwater mussels in the Neosho River, Kansas. *Transactions of the Kansas Academy of Science* 105:323-240.

Fraser, J.C. 1972. Regulated discharge and the stream environment. Pages 263-286 *in* R. Olgesby, C.A. Carlson, and J. McCann, editors. *River Ecology and Management.* Academic Press, New York.

Gehrke, P.C., D.M. Gilligan, and M. Barwick. 2002. Changes in fish communities of the Shoalhaven River 20 years after construction of Tallowa Dam, Australia. *River Research and Applications* 18:265-286.

Graf, W.L. 2006. Downstream hydrologic and geomorphic effects of large dams on American rivers. *Geomorphology* 79:336-360.

Gibbons, J.W. 1976. Thermal alteration and the enhancement of species populations. *In* G.W. Esch, R.W. McFarlane, editors. *Thermal Ecology II*, ERDA Symposium Series.

Hauer, F.R., J.A. Stanford, J.V. Ward. 1989. Serial discontinuities in a Rocky Mountain river. II. Distribution and Abundance of Trichoptera. *Regulated Rivers: Research and Management* 3:177-182.

Headrick, M.R. and R.F. Carline. 1993. Restricted summer habitat and growth of northern pike in two southern Ohio impoundments. *Transactions of the American Fisheries Society* 122:228-236.

Irvine, J.R. 1985. Effects of successive flow perturbations on stream

invertebrates. *Canadian Journal of Fisheries and Aquatic Sciences*
42:1922-1927.

Irwin, E., G. Turner, K. Mickett, and T. Piper. 2007. ACT Aquatic Gap and
water quality monitory: Final Report. ADCNR.

Jager, H.I., J.A. Chandler, K.B. Lepla, and W.V. Winkle. 2001. A theoretical
study of river fragmentation by dams and its effects on white sturgeon
populations. *Environmental Biology of Fishes* 60:347-361.

Jansson, R., C. Nilsson, and B. Renofalt. 2000. Fragmentation of riparian
floras in rivers with multiple dams. *Ecology* 81:899-903.

Kelner, D.E. and B.E. Sietman. 2000. Relic populations of the Ebony Shell,
Fusconaia ebena (Bivalvia: Uniondiae), in the Upper Mississippi River
Drainage. *Journal of Freshwater Ecology* 15:371-378.

Lessard, J.L. and D.B. Hayes. 2003. Effects of elevated water temperature
on fish and macroinvertebrate communities below small dams. *River
Research and Applications* 19:721-732.

Lehmkuhl, D.M. 1974. Thermal regime alterations and vital environmental
physiological signals in aquatic systems. Pages 216-222 *in* J.W.
Gibbons, and R.R. Sharitz, editors. *Thermal Ecology*, AEC
Symposium Series.

Lydeard, C. and R.L. Mayden. 1995. A diverse and endangered aquatic
ecosystem of the Southeast United States. *Conservation Biology*
9:800-805

McLaughlin, R.L., L. Porto, D.L.G. Noakes, J.R. Baylis, L.M. Carl, H.R. Dodd,
J.D.Goldstein, D.B. Hayes, and R.G. Randall. 2006. Effects of low-
head barriers on stream fishes: taxonomic affiliations and

morphological correlates of sensitive species. *Canadian Journal of Fisheries and Aquatic Sciences* 63:766-779.

Perry, S.A., W.B. Perry, J.A. Stanford. 1987. Effects of thermal regime on size, growth rates and emergence of two species of stoneflies (Plecoptera: Taeniopterygidae, Pteronarcyidae) in the Flathead River, Montana. *The American Midland Naturalist* 117:83-93.

Poff, N.L., J.D. Allan, M.B. Bain, J.R. Karr, K.L. Prestegaard, B.D. Richter, R.E. Sparks, J.C. Stromberg. 1997. The natural flow regime. *Bioscience* 47:769-784.

Poff, N.L., and D.D. Hart. 2002. How dams vary and why it matters for the emerging science of dam removal. *BioScience* 52:659-668.

Schuster, G.A., and C.A. Taylor. 2004. Report of the crayfishes of Alabama: literature and museum database review, species list with abbreviated annotations and proposed conservation statuses. Illinois Natural History Survey, Center for Biodiversity Technical Report.

Travnicheck, V.H., M.B. Bain, M.J. Maceina. 1995. Recovery of a warmwater fish assemblage after the initiation of minimum-flow release downstream from a hydroelectric dam. *Transactions of the American Fisheries Society* 124:836-844.

Vinson, M.R. 2001. Long-term dynamics of an invertebrate assemblage downstream from a large dam. *Ecological Applications* 11:711-730.

Ward, J.V. and J.A. Stanford. 1982. Thermal resonses in the evolutionary ecology of aquatic insects. *Annual Review of Entomology* 27:97-117.

Walter, R.C., and D.J. Merritts. Natural streams and the legacy of water powered mills. *Science* 319:299-304.

Watters, G.T. 1996. Small dams as barriers to freshwater mussels (Molusca: Pelecypoda: Unionidae) and their hosts. *Biological Conservation* 75:79-85

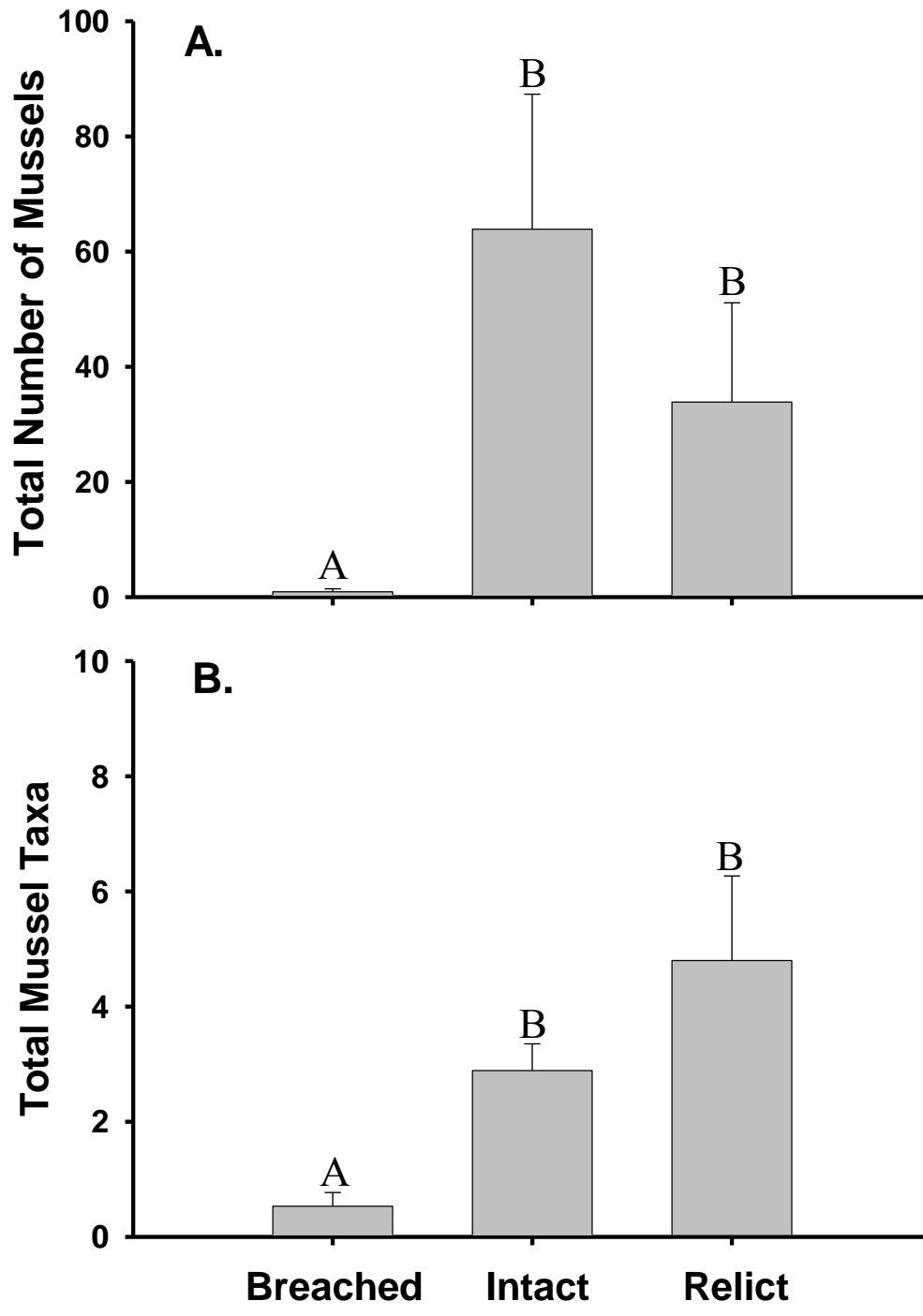Wooton, R.S. 1995. Temperature and lake-outlet communities. *Journal of Thermal Biology* 20:121-125.

Figure 1. Mean total mussel abundance (A) and taxa richness (B) in streams with breached, intact, or relict small dams in Alabama. Bars sharing the same letter are not significantly different from one another.
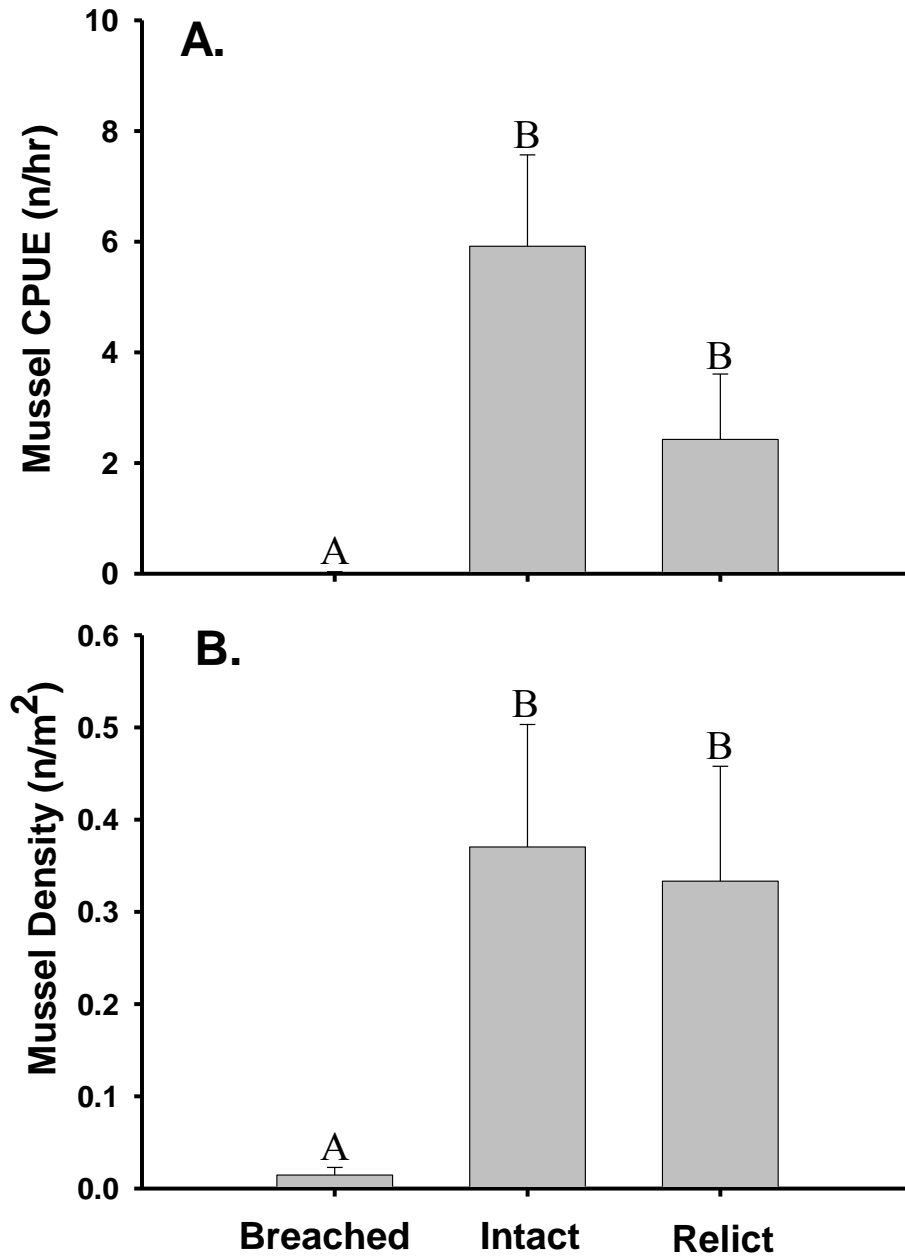
Figure 2. Mean total mussel catch per unit effort (CPUE, A) and density (B) in streams with breached, intact, or relict small dams in Alabama. Bars sharing the same letter are not significantly different from one another
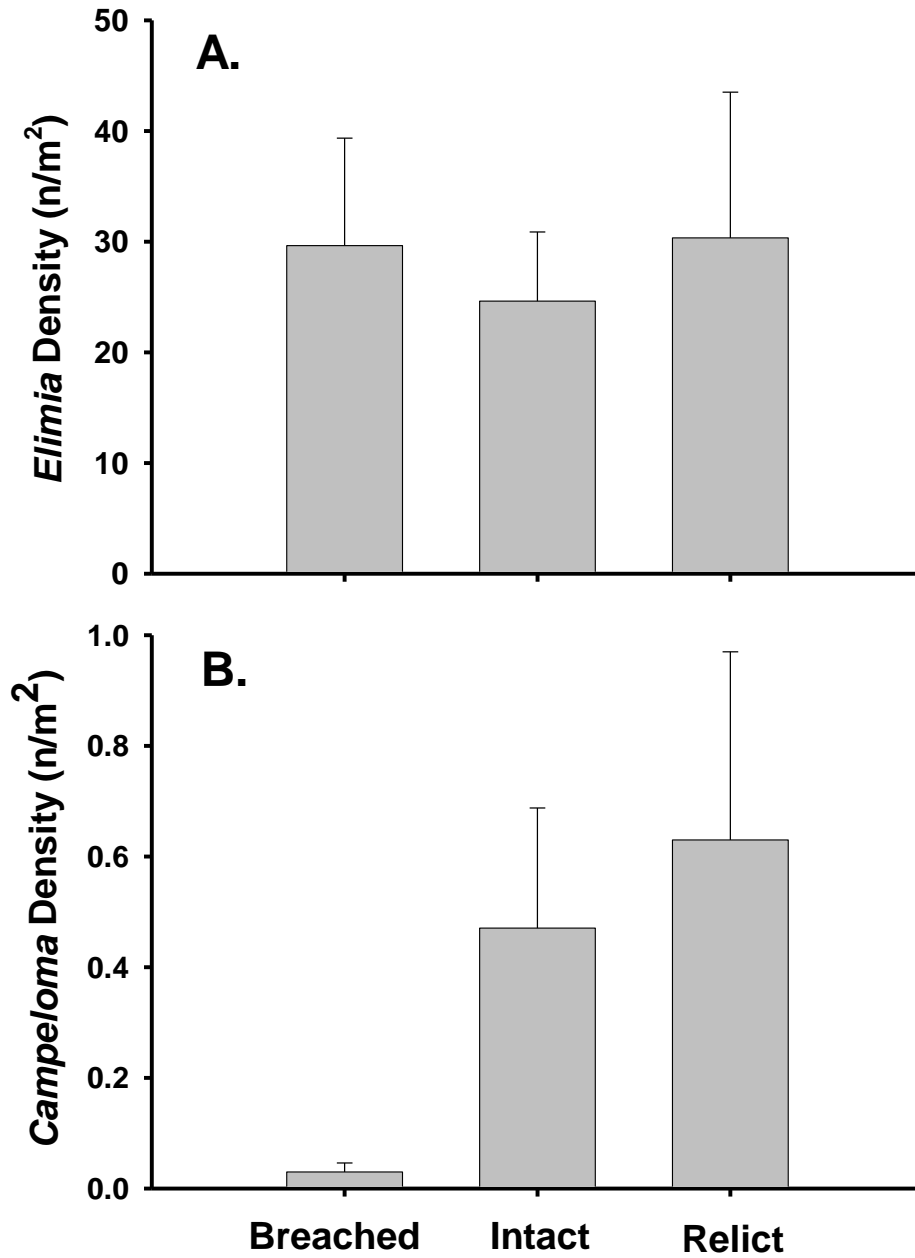
Figure 3. Mean total *Elimia* (A) and *Campeloma* (B) density in streams with breached, intact, or relict small dams in Alabama. Means are not significantly different from one another.
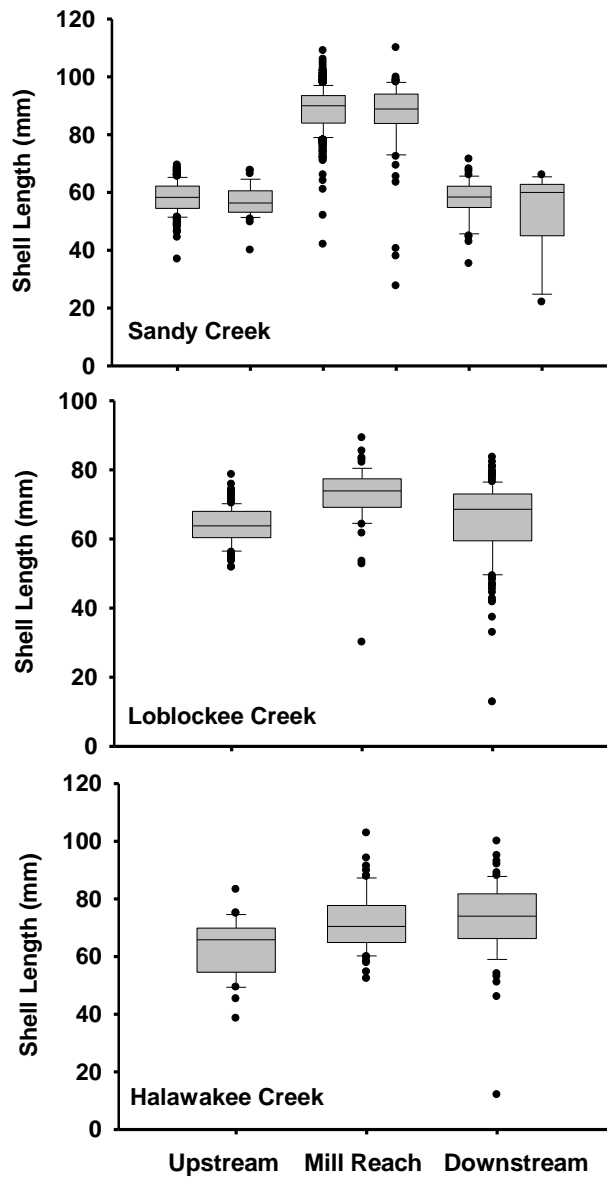
Figure 4. Boxplot showing median, 95% confidence interval, and standard error for mussel shell lengths from *Elliptio* spp. populations in 3 east central Alabama streams. Data were obtained from sites located upstream the impounded zone formed by the mill dam (Upstream), immediately downstream of the mill dam (Mill Reach) and >500 m downstream from the mill dam (Downstream). Mussel shell lengths from Mill Reach populations were significantly greater than up-or downstream conspecifics in all streams except Halawakee Creek. Halawakee Creek's downstream study reach is

closer to the dam (~500 m) than the other downstream sites. Upper panel

shows both the timed search (left box) and quadrat-derived densities for the
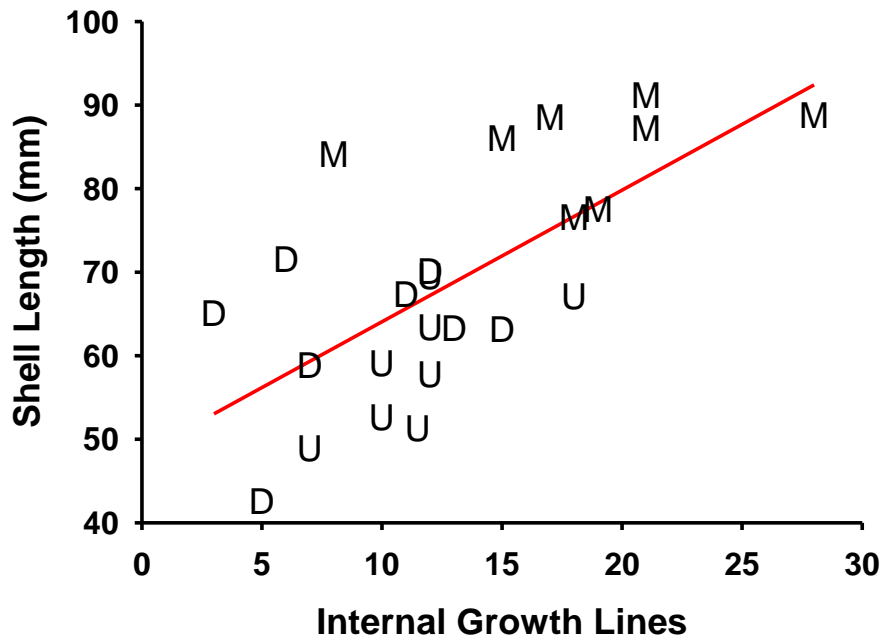
Sandy Creek populations.

Figure 5. Length at age plot for Sandy Creek *Elliptio arca* shells collected
from the downstream (D), mill reach (M), and upstream (U) study sites.
Residual scores for each shell were calculated by determining the distance
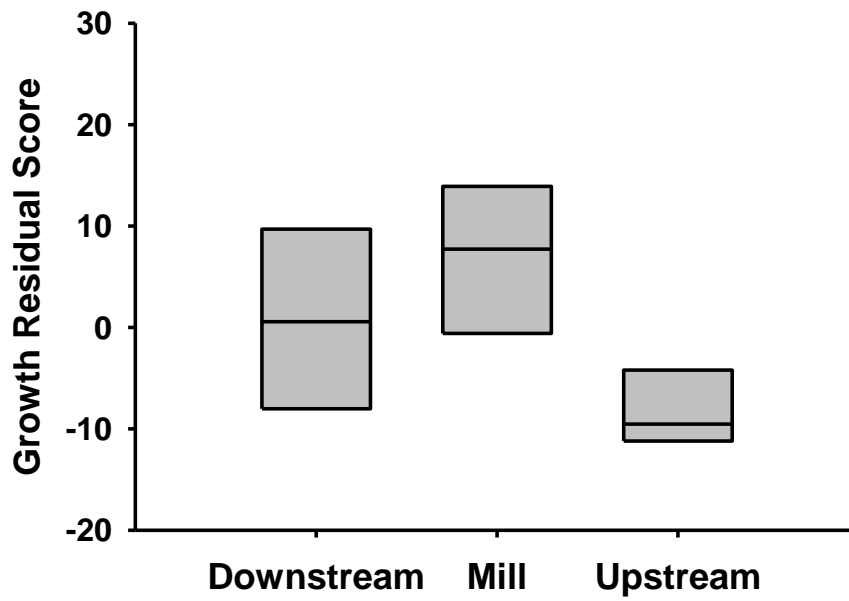from each point to the regression line.

Figure 6. Box plot of standardized residual scores for Sandy Creek *Elliptio arca* shells. ANOVA revealed that mussels in the mill reach exhibited significantly higher growth rates compared to both up-and downstream populations. Positive residual scores reveal growth rates that are high relative to the total population and negative residual scores indicate the converse.

Chapter 4: Analysis of the effects of mill and lowhead dams on fish assemblages in Alabama streams

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Abstract

From 2006 to 2008, we sampled fish assemblages in 22 streams containing mill dams of various conditions (intact, breached, or relict) across the state of Alabama. Three 100m reaches were sampled in each stream: 500-1000 m downstream of the dam, 0-100m downstream of the dam, and upstream of the impoundment using a backpack electroshocker. There was no difference in total catch, species diversity or richness between the 3 dam conditions or between instream reaches. However, there were feeding guild differences, with a larger proportion of piscivores and smaller proportion of invertivores in the breached sites than the other sites. There were also fewer narrow endemic species at the breached sites than the intact sites. NMDS ordinations revealed that the Euclidean distance from the downstream to the mill dam to the upstream reaches on average was larger in the breached than the relict dams, suggesting greater assemblage dissimilarity from downstream to upstream in these streams. Taken together, breached dams appear to exert a stronger influence on contemporary stream fish assemblage structure than relict or intact dams across the state by causing shifts toward general piscivorous species, reductions in invertivore proportions, and increased longitudinal differences in fish assemblages.

Introduction

Physical, chemical, and biological impacts of dams can be dramatic, and may include altering hydrologic regime, sediment composition, and channel geomorphology. Additionally, dams impede or prevent migration by fishes and other stream biota, thereby fragmenting and exacerbating extinction risks of imperiled aquatic populations (Baxter 1977, Blalock & Sickel 1996, Watters 1996, Dean *et al.* 2002, Poff & Hart 2002, McLaughlin *et al.* 2006).

According to census records, >65,000 low-head dams existed in the eastern US by 1840, most of which were built for water-powered milling (Walter and Merritts 2008). Dams are also numerous and widespread in many southeastern U.S. streams, with >10,000 dams in Alabama alone (AL Office of Water Resources http://www.adeca.alabama.gov/, Chapter 1). Although numerous large, hydroelectric dams have radically altered Alabama's large rivers, low-head dams are much more abundant (Chapter 1, CD Appendix B). Low-head dams have a hydraulic height of <10 m and are typically overflow or spill-way structures (IFC Consulting Report 2005).

Alabama supports 38% of North America's native freshwater fish species, many endemic to the southeastern U.S. (Lydeard & Mayden 1995; Crandall et al. 2000; Schuster & Taylor 2004). Impoundments have been linked to imperilment of many fish taxa (Travnicheck et al 1995; Jager et al 2001; Dean et al. 2002; Lessard & Hayes 2003; Irwin et al. 2007). In Alabama the construction of large, hydroelectric dams in many large rivers in the Mobile and Tennessee basins, has been linked to the extinction and imperilment of many fish taxa (Benz 1997).

Unnatural flow regimes from impoundments are a major source of habitat degradation that may alter stream communities (Fraser 1972; Cushman 1985; Irvine 1985; Travinchek *et al.* 1995; Gerhke *et al.* 2002; McLaughlin *et al.* 2006) and even riparian vegetation (Janson *et al.* 2000). Coarsening of the stream bed downstream of many dams reduces habitat availability for benthic species by decreasing habitat heterogeneity, which, in turn, may reduce diversity and richness (Armitage and Blackburn 1990; Hauer *et al.* 1989; Poff *et al.* 1997).

Alterations in temperature regimes from impoundments also may alter fish distribution and behavior. Increased temperatures downstream of overflow dams can eliminate thermal cues vital to some invertebrate life cycles (Lehmkuhl 1974; Ward and Stanford 1982; Irvine 1985). In addition, increased water temperature can increase metabolic rates for fish and invertebrates, which, in turn, increases demands for food to maintain growth and survival (Gibbons 1976; Wotton 1995; Perry *et al.* 1987; Vinson 2001; Lessard and Hayes 2003). Within reservoirs, deep, cold, anoxic water often is a fish-free zone and many tailwaters of hypolimnetic-release dams often support depauperate fish communities (Headrick and Carline 1993, Benz et al. 1997, Dean *et al.* 2002).

Dams also impede longitudinal movements of stream organisms (Baxter 1977; Watters 1996; Dean *et al.* 2002). Upstream movement is frequently halted by dams, and this may prevent individuals from reaching feeding and/or spawning habitat, and cause population declines (Raymond 1979; Larinier 2001). Among fish, decreased longitudinal connectivity across streams may cause population fragmentation and isolation (Neraas and

Spruell *et al.* 2001; Olden *et al.* 2001). Moreover, one-way (downstream) migration may reduce genetic diversity and population size, especially in upstream sections (Jager *et al.* 2001; Morita and Yamamoto 2002; Yamamoto *et al.* 2004).

Some states (mostly in the upper Midwest and New England) have begun aggressively removing low-head dams and restoring stream habitats, and the American Fisheries Society has begun developing protocols for these projects. However, removal of these structures is often costly and removal often can have negative consequences on downstream biota (Sethi et al. 2004). A quantitative, statistically rigorous approach is therefore needed to better understand how low-head structures affect stream biota.

Although the effects of large dams on aquatic organisms in large rivers are well documented, few studies have assessed effects of mill and low-head dams on smaller stream biota (Watters 1996, Dean *et al.* 2002, Lessard & Hayes 2003, Graf 2006). The objective of this portion of the study was to assess the impact of small dams on fish assemblages across the 22 focal streams.

Methods

Fish Sampling

Mill dams were sampled for fishes at 22 sites within the state of Alabama. Each mill dam was sampled at three localities: 500–1000 meters downstream of the dam, 0–100 meters below the dam, and upstream of the impoundment effects of the dam (or putative impoundment effects in the case of historical dam sites). At each locality ten representative microhabitats were sampled

which were composed of 3 riffle-run-pool sequences (3 riffles, 3 runs, and 3 pools) and one stream-bank sample.

Fishes were sampled using a Smith-Root LR-24 electrofisher. A team of three people, with one person using the electrofisher and two people collecting stunned fishes with dip nets, made collections. Upon capture fish were placed into a six-gallon screw–top pail until sampling was completed. Once sampling was finished at a site, fish were anesthetized in tricane methanesulfonate (MS-222), and fixed in a 10% formalin solution once all fish were dead. After seven days, fish were transferred from formalin solution to water for three days then transferred into 70% ethanol for permanent storage. Fishes collected were identified in the laboratory and deposited in the Auburn University Museum Fish Collection.

Fish data from each mill dam were coded for dam state and locality relative to dam. Dam states are defined as: relict (R), breached (B), or intact (I). Locality relative to dam is defined as: downstream (D), immediately below the mill dam (M), or upstream of impoundment effects (U). Multiple metrics were then calculated for each dam state and each locality.

Assemblage metrics

We calculated species diversity (Shannon's H'), richness and total catch at each locality. Richness and H' are commonly used metrics for comparing fish assemblages; however, human disturbance may cause only nominal changes in H' or species richness but major changes in composition (Scott and Helfman, 2001; Walters et al., 2005). Therefore we also used non-metric multidimensional scaling (NMDS) to describe overall variation in fish

assemblages among localities. We used relative species abundance data (arcsin-square root transformed), which resulted in an 66 x 114 site–species matrix on which we based ordinations using a Sorenson distance measure (McCune and Grace, 2002). From the 2 most explanatory axes of the NMDS ordination (which are not necessarily the first 2 axes in an NMDS) we calculated the total distance between the sequential scores for each locality (i.e., downstream to mill dam, mill dam to upstream, downstream to upstream) to obtain a measure of site-specific spatial variation, with higher distance values indicating larger spatial differences in assemblage structure. The underlying NMDS is based on the Sorenson distance measure, but the resulting ordination is viewed in a Euclidean manner (McCune and Grace 2002); thus, Euclidean distance was used to measure distance between sequential site scores.

Feeding and breeding guild metrics

We assigned species to feeding and breeding guilds (Muncy et al., 1979; Berkeman and Rabeni, 1987, Boschung and Mayden 2004) as potential indicators of disturbance associated with mill dams. For feeding guilds, fish were classified as piscivores, insectivores, herbivores, omnivores, or filter feeders. For breeding guilds, we based classification on benthic substrate requirements for spawning. Generalist spawners were those that either showed a wide preference in benthic substrate conditions (e.g. *Lepomis gulosus*) or actively maintained a nest to suit its preference (e.g. *Nocomis leptocephalus*). Lithophilic spawners were those that do not actively maintain their nest sites and require clean, gravel or other rock substrate while crevice

spawners are those species that deposit their eggs in crevices of logs, stumps, gravel and bedrock.   We calculated the proportion of catch for each breeding and feeding guild for each locality.

Range metrics

Different fish species can show considerable variation in their geographical range, thus we coded each fish species in accordance to the spatial extent of its range. Narrow endemics were those species constrained to one or a few sub-basins (e.g. upper Tallapoosa, upper Coosa).   Many of these fish are of special conservation concern. Broad endemics are those species endemic to a large basin (e.g. Mobile basin) while widespread species are those found in multiple large basins. We calculated the proportion of catch for each range type for each locality.

Data analysis

All continuous variables were log-transformed and proportion variables were arcsine-squareroot transformed as needed to meet assumptions of normality (Zar, 1998).   We used General Linear Models (GLMs) with the full dataset to determine any differences between dam state, locality, and state / locality interactions for all metrics followed by Tukey multiple comparisons. To account for the possibility of differential effects of dam state on fish assemblages, we grouped sites by dam state and examined the effects of locality in relation to the dam using 1-way ANOVA for each measured metric. Finally, since streams were sampled throughout the state of AL, there may be ecoregional differences in assemblages obscuring the impacts of mill dams.

Therefore, for all streams, we calculated the percent change in all metrics and the Euclidean distance (determined from the NMDS) from 1) downstream to mill dam, 2) mill dam to upstream, and 3) downstream to upstream. We used 1-way ANOVA to determine any differences in percent change for all metrics.

Results

Sampling yielded 1439 lots of fish from 70 localities, of which 1096 lots representing 66 localities were informative. Collections deemed uninformative were those in which the entire locality could not be sampled in a single day due to weather or stream conditions. At total of 114 species of fishes from 13 families were represented in the samples.

Axes 2 and 3 from the NMDS ordination (final stress = 17.777, final instability = 0.00001, iterations = 341) explained 24.1 and 29.3% of the total variation, respectively (Figure 1a). These two most explanatory axes revealed strong groupings by site and short vector lengths, particularly for the relict dam sites (Figure 1b), suggesting minimal longitudinal variation in fish assemblages at these sites. There were few noticeable groupings and longer vector lengths from sites with breached dams, suggesting greater longitudinal variation in assemblages at these sites (Figure 1b).

From the GLMs, differences in dam condition on streams were most strongly manifested in the fish feeding guilds (Table 1). The proportion of herbivores was significantly higher in the breached dam sites than the relict dam sites (Table 1, Figure 2) while the proportion of predators was significantly higher in the breached dam sites than the intact dam sites (Table 1, Figure 3). Conversely, the proportion of insectivores was significantly

lowest in the breached sites (Table1, Figures 4).   Also, the proportion of

crevice spawners was marginally reduced in the breached sites as compared

to the relict sites (Table 1, Figure 5) while the proportion of narrow endemics

was marginally increased in the intact dam sites as compared to the breached

sites (Table 1, Figure 6). There were no significant differences in respect to

locality or significant interactions between dam condition and locality for any

of the measured variables (Table 1).

The grouping of sites by dam condition largely reflected the results from

the GLM. In general, there were no significant differences between measured

fish variables at downstream, mill dam, and upstream localities for relict

(Table 2), breached (Table 3), or intact (Table 4) dams.   However, for the

intact sites, there were marginal trends for reduced diversity and proportion of

herbivores at the upstream locality as compared to the downstream and dam

localities (Table 4).

The comparison of proportional change and overall similarity in

assemblages from downstream to upstream revealed few differences (Table

5). However, the proportional change in the number of fish collected from the

mill dam to upstream was significantly lower at the relict sites (Figure 6).

Also, the Euclidean distance, based on the NMDS ordination, from the mill

dam to upstream localities and from downstream to upstream localities was

shorter at sites with relict dams than those with breached dams (Table 5,

Figure 1b), suggesting more longitudinal variation in assemblages at

breached dam sites.

Discussion

In Alabama, large dams have likely impacted fish populations in streams by blocking migrations and altering physicochemical habitat parameters (Lessard and Hayes 2003, Fraley 1979). However, our data suggest small dams may have more dramatic effects when they breach and are not repaired or removed. Breached dams allow previously entrained sediments to be displaced downstream and unconsolidated material may persist for years in the stream (Stanley et al. 2002, Doyle et al. 2003). In addition to smothering habitats downstream, sediment released from breached mill dams can contain heavy metals, polychlorinated biphenyls (PCBs), and other contaminants that can have adverse effects on fishes, other aquatic biota, and humans (Shuman 1995, Gray and Ward 1982). Stream channel scouring effects downstream of a breach are equally destructive because they can destroy natural stream habitats.

NMDS analyses suggest that streams with breached dams support fish communities characteristic of degraded habitats. Analyses revealed long vector lengths suggesting greater disparity in fish populations associated with breached mill dams. Similarly, feeding guild shifts associated with breached mill dams, compared to intact and relict sites, suggest impaired fish communities. The high relative abundance of herbivores at breached sites is most likely due to increased flow velocity and streambed scouring associated with the remaining structure. Many streambeds immediately downstream of breached mills were scoured down to bedrock. These reaches typically supported luxuriant algal growth and harbored vast numbers of algivorous minnows (e.g. *Campostoma* spp.).

Increased piscivore abundance in streams with breached dams may

reflect out-migration from semi-impounded upstream reaches. Alternatively, these predators may be capitalizing on microhabitat conditions frequently associated with breached dams (i.e., high current velocity chutes, deep scour holes, rubble and boulders). Dam breaches have both acute (post-breach) and chronic impacts to streams. In the short term, large amounts of debris and unconsolidated fine sediments are mobilized and inundate gravel interstices, and smothering benthic organisms. However, over longer intervals, increased flow velocity near breached dams can lead to bed scouring and ultimately reduce available habitat heterogeneity.

At the site scale, we did not observe any statistically significant effects of intact small dams on fish community metrics. However, we did observe a trend of decreased herbivore abundance and fish diversity in upstream reaches. Additionally, endemic fishes were more abundant immediately downstream of small dams, suggesting that dams may either aggregate endemic fish or provide habitat conditions that are more suitable to these taxa. Ongoing analyses are attempting to disentangle effects of stream size and species pools on dam-fish community interactions.

Literature Cited

Armitage, P.D., and J.H. Blackburn. 1990. Environmental stability and communities of Chironomidae (Diptera) in a regulated river. *Regulated Rivers: Research and Management* 5:319-328

Baxter, R.M. 1977. Environmental effects of dams and impoundments. *Annual Review of Ecology and Systematics* 8:255-283.

Benz, G.W. & D.E. Collins (eds). 1997. Aquatic fauna in peril: the southeastern perspective. Special publication 1,Southeast Aquatic Research Institute, Lenz Design and Communications, Decateur, GA.

Berkman, H. E. and Rabeni, C. F. 1987. Effect of Siltation on Stream Fish Communities. *Experimental Biology of Fishes* **18**: 285-294.

Blalock, H.N. and J.B. Sickel. 1996. Changes in mussel (Bivalvia: Unionidae) fauna within the Kentucky portion of Lake Barkley since impoundments. *American Malacological Bulletin* 13:111-116.

Boschung, H.T., Jr., and Mayden, R.L. 2004 Fishes of Alabama. Smithsonian Books, Washington, D.C., USA.

Crandall, K.A, D.J. Harris, J.W. Fetzner Jr. 2000. The monophyletic origin of freshwater crayfish estimated from nuclear and mitochondrial DNA sequences. *Proclamations of the Royal Society of London* 267:1679-1686.

Cushman, R.M. 1985. Review of ecological effects of rapidly varying flows downstream of hydroelectric facilities. *North American Journal of Fisheries Management* 5:330-339.

Dean, J., D. Edds, D. Gillette, J. Howard, S. Sherraden, and J. Tiemann. 2002. Effects of lowhead dams on freshwater mussels in the Neosho River, Kansas. *Transactions of the Kansas Academy of Science* 105:323-240

Doyle, M.C., E.H. Stanley, and J.M. Harbor. 2003. Channel adjustments following two dam removals in Wisconsin. Water Resources Research 39:1-15.

Fraser, J.C. 1972. Regulated discharge and the stream environment. Pages

263-286 *in* R. Olgesby, C.A. Carlson, and J. McCann, editors. *River Ecology and Management.* Academic Press, New York.

Gehrke, P.C., D.M. Gilligan, and M. Barwick. 2002. Changes in fish communities of the Shoalhaven River 20 years after construction of Tallowa Dam, Australia. *River Research and Applications* 18:265-286.

Gibbons, J.W. 1976. Thermal alteration and the enhancement of species populations. *In* G.W. Esch, R.W. McFarlane, editors. *Thermal Ecology II*, ERDA Symposium Series.

Graf, W.L. 2006. Downstream hydrologic and geomorphic effects of large dams on American rivers. *Geomorphology* 79:336-360.

Hauer, F.R., J.A. Stanford, J.V. Ward. 1989. Serial discontinuities in a Rocky Mountain river. II. Distribution and Abundance of Trichoptera. *Regulated Rivers: Research and Management* 3:177-182.

Headrick, M.R. and R.F. Carline. 1993. Restricted summer habitat and growth of northern pike in two southern Ohio impoundments. *Transactions of the American Fisheries Society* 122:228-236.

IFC Consulting. 2005. A summary of existing research on low-head dam removal projects, Technical Report, AASHTO.

Irvine, J.R. 1985. Effects of successive flow perturbations on stream invertebrates. *Canadian Journal of Fisheries and Aquatic Sciences* 42:1922-1927.

Irwin, E., G. Turner, K. Mickett, and T. Piper. 2007. ACT Aquatic Gap and water quality monitory: Final Report. ADCNR.

Jager, H.I., J.A. Chandler, K.B. Lepla, and W.V. Winkle. 2001. A theoretical

study of river fragmentation by dams and its effects on white sturgeon populations. *Environmental Biology of Fishes* 60:347-361.

Jansson, R., C. Nilsson, and B. Renofalt. 2000. Fragmentation of riparian floras in rivers with multiple dams. *Ecology* 81:899-903.

Larinier, M. 2001. Environmental issues, dams and fish migration. FAO technical paper 419:45-89.

Lehmkuhl, D.M. 1974. Thermal regime alterations and vital environmental physiological signals in aquatic systems. Pages 216-222 *in* J.W. Gibbons, and R.R. Sharitz, editors. *Thermal Ecology*, AEC Symposium Series.

Lessard, J.L. and D.B. Hayes. 2003. Effects of elevated water temperature on fish and macroinvertebrate communities below small dams. *River Research and Applications* 19:721-732.

Lydeard, C. and R.L. Mayden. 1995. A diverse and endangered aquatic ecosystem of the Southeast United States. *Conservation Biology* 9:800-805.

McCune, B. and Grace, J.B. 2002. Analysis of Ecological Communities. MjM Software Design, Glenedon Beach, Oregon, USA.

McLaughlin, R.L., L. Porto, D.L.G. Noakes, J.R. Baylis, L.M. Carl, H.R. Dodd, J.D.Goldstein, D.B. Hayes, and R.G. Randall. 2006. Effects of low-head barriers on stream fishes: taxonomic affiliations and morphological correlates of sensitive species. *Canadian Journal of Fisheries and Aquatic Sciences* 63:766-779.

Morita, K. and S. Yamamoto. 2002. Effects of habitat fragmentation by

damming on the persistence of stream-dwelling charr populations. *Conservation Biology* 16:1318-1323.

Muncy, R.J., Atchinson, G.J., Bulkley, R.V. Menzel, B.W., Perry, L.G., and Summerfelt, R.C. 1979. Effects of Suspended Solids and Sediment on Reproduction and Early Life of Warmwater Fishes: A Review. EPA-600/3-79-89

Neraas, L.P. and P. Spruell. 2001. Fragmentation of riverine systems: the genetic effects of dams on bull trout (*Salvelinus confluentus*) in the Clark Fork River system. *Molecular Ecology* 10:1153-1164.

Olden, J.D., D.A. Jackson, and P.R. Peres-Neto. 2001. Spatial isolation and fish communities in drainage lakes. *Oecologia* 127:572-585.

Perry, S.A., W.B. Perry, J.A. Stanford. 1987. Effects of thermal regime on size, growth rates and emergence of two species of stoneflies (Plecoptera: Taeniopterygidae, Pteronarcyidae) in the Flathead River, Montana. *The American Midland Naturalist* 117:83-93.

Poff, N.L., J.D. Allan, M.B. Bain, J.R. Karr, K.L. Prestegaard, B.D. Richter, R.E. Sparks, J.C. Stromberg. 1997. The natural flow regime. *Bioscience* 47:769-784.

Poff, N.L., and D.D. Hart. 2002. How dams vary and why it matters for the emerging science of dam removal. *BioScience* 52:659-668.

Raymond, H.L. 1979. Effects of dams and impoundments on migrations of juvenile Chinook salmon and steelhead from the Snake River, 1966 to 1975. *Transactions of the American Fisheries Society* 108:505-529.

Scott, M.C. and Helfman, G.S. 2001. Native Invasions, Homogenization, and the Mismeasure of Integrity of Fish Assemblages. *Fisheries.* **26**: 6 –

15.

Schuster, G.A., and C.A. Taylor. 2004. Report of the crayfishes of Alabama: literature and museum database review, species list with abbreviated annotations and proposed conservation statuses. Illinois Natural History Survey, Center for Biodiversity Technical Report.

Sethi, S.A., A.R. Selle, M.W. Doyle, E.H. Stanley, & H.E. Kitchel. 2004. Response of Unionid Mussels to Dam Removal in Koshkonong Creek, Wisconsin (USA). Hydrobiologia 525:157-165.

Stanley, E.H., M.A. Luebke, M.W. Doyle, D.W. Marshall. 2002. Short-term changes in channel form and macroinvertebrate communities following low-head dam removal. Journal of the North American Benthological Society 21:172-187.

Summerfelt, R.C. 1979. Effects of Suspended Solids and Sediment on Reproduction and Early Life of Warmwater Fishes: A Review. EPA-600/3-79-0, Covallis Environmental Research Laboratory, Corvallis, 100 pp.

Travnicheck, V.H., M.B. Bain, M.J. Maceina. 1995. Recovery of a warmwater fish assemblage after the initiation of minimum-flow release downstream from hydroelectric dam. *Transactions of the American Fisheries Society* 124:836-847.

Vinson, M.R. 2001. Long-term dynamics of an invertebrate assemblage downstream from a large dam. *Ecological Applications* 11:711-730.

Walter, R.C., and D.J. Merritts. 2008. Natural streams and the legacy of water powered mills. *Science* 319:299-304.

Walters, D.M., Freeman, M.C., Leigh, D.S., Freeman, B.J., and Pringle, C.M. 2005. Urbanization Effects on Fishes and Habitat Quality in a Southern Piedmont River Basin. *In* Effects of Urbanization on Stream Ecosystems. *Edited by* L.R. Brown, R.H. Gray, R.H. Hughes, and M.R. Meador. American Fisheries Society, Symposium 47, Bethesda, Maryland. pp. 69-86

Watters, G.T. 1996. Small dams as barriers to freshwater mussels (Molusca: Pelecypoda: Unionidae) and their hosts. *Biological Conservation* 75:79-85.

Ward, J.V. and J.A. Stanford. 1982. Thermal resonses in the evolutionary ecology of aquatic insects. *Annual Review of Entomology* 27:97-117.

Wooton, R.S. 1995. Temperature and lake-outlet communities. *Journal of Thermal Biology* 20:121-125.

Yamamoto, S. K. Morita, I. Koizumi, and K. Maekawa. 2004. Genetic differentiation of white-spotted charr (*Salvelinus leucomaenis*) populations after habitat fragmentation: spatial-temporal changes in gene frequencies. *Conservation Genetics* 5:529-538.

Zar, J. 1999 Biostatistical Analysis. 4th edition. Prentice-Hall, Inc. Upper Saddle River, New Jersey, USA

Table 1. Descriptive statistics and GLM results for fish metrics calculated at each of 3 localities in each stream sampled. **Dam Condition**, **Locality**, and **Interaction** are p-values for the GLM. Bolded values are significant at α = 0.05. See text for explanations of the various metrics.

Metric	Average	Range	Dam Condition	Locality	Interaction
Assemblage					
Number Collected	257.7	4 - 1172	0.426	0.479	0.880
Diversity	1.98	0.59 - 2.76	0.919	0.378	0.631
Richness	16.6	4 - 33	0.702	0.726	0.802
Feeding guild proportions					
Herbivore	0.086	0 - 0.406	**0.022**	0.666	0.585
Insectivore	0.0819	0.197 - 1.000	**0.018**	0.821	0.875
Omnivore	0.031	0 - 0.250	0.952	0.606	0.972
Predator	0.064	0 - 0.541	**0.027**	0.925	0.956
Breeding guild proportions					
Generalist	0.462	0.112 - 0.951	0.579	0.259	0.334
Crevice	0.18	0 - 0.647	**0.083**	0.339	0.969
Lithophil	0.35	0.018 - 0.722	0.732	0.465	0.584
Range proportions					
Narrow Endemic	0.159	0 - 0.692	**0.084**	0.955	0.973
Broad Endemic	0.177	0 - 0.880	0.340	0.909	0.663
Widespread	0.663	0.067 - 1.000	0.513	0.963	0.871

Table 2. Descriptive statistics and ANOVA results for fish metrics calculated at each of 3 stream localities in the *Relict dam* streams (N = 5). **Downstream, Mill Dam**, and **Upstream** are averages with standard deviations in parentheses, and **p** are values from associated 1-way ANOVA. See text for explanations of the various metrics.

Metric	Average	Range	Stream Location			p
			Downstream	**Mill Dam**	**Upstream**	
Assemblage						
Number Collected	195.1	51 - 416	183.0 (82.8)	175.8 (90.8)	226.4 (145.3)	0.739
Diversity	2.03	1.18 - 2.76	2.13 (0.593)	1.98 (0.565)	1.99 (0.457)	0.895
Richness	17.7	10 - 30	17.6 (7.1)	17.4 (7.7)	18.2 (6.6)	0.983
Feeding guild proportions						
Herbivore	0.044	0 - 0.164	0.059 (0.077)	0.028 (0.043)	0.045 (0.071)	0.885
Insectivore	0.883	0.759 - 0.974	0.871 (0.073)	0.897 (0.061)	0.881 (0.069)	0.814
Omnivore	0.028	0 - 0.151	0.026 (0.022)	0.038 (0.063)	0.020 (0.017)	0.933
Predator	0.044	0.009 - 0.137	0.040 (0.053)	0.037 (0.022)	0.053 (0.028)	0.698
Breeding guild proportions						
Generalist	0.414	0.227 - 0.767	0.418 (0.123)	0.309 (0.119)	0.514 (0.210)	0.150
Crevice	0.255	0.047 - 0.645	0.244 (0.178)	0.327 (0.231)	0.195 (0.142)	0.594
Lithophil	0.319	0.046 - 0.543	0.327 (0.189)	0.347 (0.162)	0.281 (0.127)	0.849
Range proportions						
Narrow Endemic	0.144	0 - 0.552	0.111 (0.145)	0.196 (0.256)	0.126 (0.166)	0.943
Broad Endemic	0.129	0 - 0.732	0.126 (0.222)	0.100 (0.123)	0.160 (0.320)	0.994
Widespread	0.727	0.259 - 1.00	0.763 (0.245)	0.704 (0.264)	0.714 (0.306)	0.965

Table 3. Descriptive statistics and ANOVA results for fish metrics calculated at each of 3 stream localities in the *Breached dam* streams (N = 6). **Downstream, Mill Dam**, and **Upstream** are averages with standard deviations in parentheses, and **p** are values from associated 1-way ANOVA. See text for explanations of the various metrics.

Metric	Average	Range	Stream Location			p
Assemblage			Downstream	Mill Dam	Upstream	
Number Collected	277.9	4 - 639	224.8 (208.9)	365.5 (233.2)	243.3 (165.9)	0.452
Diversity	1.96	1.39 - 2.68	2.05 (0.33)	1.90 (0.54)	1.93 (0.36)	0.825
Richness	16.2	4 - 24	16.0 (5.8)	16.7 (7.8)	15.8 (3.8)	0.968
Feeding guild proportions						
Herbivore	0.141	0 - 0.406	0.119 (0.124)	0.148 (0.141)	0.157 (0.179)	0.941
Insectivore	0.710	0.197 - 0.989	0.730 (0.290)	0.701 (0.260)	0.699 (0.298)	0.969
Omnivore	0.030	0 - 0.250	0.017 (0.028)	0.053 (0.097)	0.019 (0.024)	0.653
Predator	0.119	0 - 0.541	0.134 (0.206)	0.098 (0.148)	0.126 (0.147)	0.867
Breeding guild proportions						
Generalist	0.486	0.181 - 0.951	0.418 (0.202)	0.571 (0.192)	0.468 (0.204)	0.413
Crevice	0.141	0 - 0.406	0.165 (0.144)	0.130 (0.103)	0.127 (0.160)	0.679
Lithophilic	0.366	0.018 - 0.681	0.411 (0.206)	0.289 (0.155)	0.396 (0.182)	0.474
Range proportions						
Narrow Endemic	0.082	0 - 0.524	0.108 (0.176)	0.041 (0.097)	0.098 (0.210)	0.743
Broad Endemic	0.235	0 - 0.686	0.263 (0.252)	0.318 (0.289)	0.123 (0.155)	0.146
Widespread	0.683	0.312 - 1.000	0.629 (0.241)	0.612 (0.268)	0.780 (0.221)	0.534

Table 4. Descriptive statistics and ANOVA results for fish metrics calculated at each of 3 stream localities in the *Intact dam* streams (N = 11). **Downstream, Mill Dam**, and **Upstream** are averages with standard deviations in parentheses, and **p** are values from associated 1-way ANOVA. See text for explanations of the various metrics.

Metric	Average	Range	Stream Location			p
Assemblage			Downstream	Mill Dam	Upstream	
Number Collected	275.2	11 - 1172	227.7 (185.7)	337.6 (195.3)	260.3 (318.3)	0.553
Diversity	1.96	0.59 - 2.60	2.08 (0.21)	2.07 (0.40)	1.74 (0.51)	**0.085**
Richness	16.3	6 - 33	16.4 (5.6)	18.6 (6.7)	14.0 (3.7)	0.165
Feeding guild proportions						
Herbivore	0.074	0 - 0.328	0.099 (0.092)	0.091 (0.083)	0.033 (0.044)	**0.055**
Insectivore	0.849	0.500 - 1.000	0.832 (0.111)	0.811 (0.155)	0.903 (0.078)	0.225
Omnivore	0.034	0 - 0.249	0.025 (0.045)	0.045 (0.076)	0.031 (0.049)	0.696
Predator	0.044	0 - 0.205	0.045 (0.057)	0.052 (0.067)	0.033 (0.053)	0.742
Breeding guild proportions						
Generalist	0.471	0.112 - 0.933	0.381 (0.198)	0.462 (0.209)	0.570 (0.268)	0.152
Crevice	0.166	0 - 0.647	0.175 (0.163)	0.203 (0.222)	0.121 (0.185)	0.464
Lithophil	0.355	0.067 - 0.722	0.428 (0.132)	0.329 (0.159)	0.308 (0.184)	0.175
Range proportions						
Narrow Endemic	0.209	0 - 0.691	0.219 (0.239)	0.207 (0.244)	0.200 (0.277)	0.971
Broad Endemic	0.168	0 - 0.880	0.147 (0.190)	0.139 (0.167)	0.219 (0.295)	0.753
Widespread	0.623	0.067 - 1.000	0.634 (0.299)	0.655 (0.325)	0.581 (0.342)	0.878

Table 5. ANOVA results for proportional change in select metrics and the Euclidean distance between different localities (**Downstream to Mill Dam**, **Mill Dam to Upstream**, and **Downstream to Upstream**). Values in **Dam Condition** are averages with standard deviations in parentheses for each metric or Euclidean distance for each dam condition and values in **p** are p-values for associated ANOVA. Bold values are significant at $\alpha = 0.10$..

Metric	Dam Condition			p
	Intact (11)	Breached (6)	Relict (5)	
Downstream to Mill Dam				
Number Collected	0.504 (0.323)	0.933 (0.678)	0.268 (0.134)	0.160
Diversity	0.143 (0.142)	0.163 (0.138)	0.106 (0.033)	0.863
Richness	0.161 (0.124)	0.286 (0.324)	0.125 (0.090)	0.385
Euclidean distance	0.266 (0.192)	0.287 (0.165)	0.220 (0.065)	0.806
Mill Dam to Upstream				
Number Collected	0.825 (0.451)	1.034 (0.524)	0.270 (0.188)	**0.008**
Diversity	0.504 (0.294)	0.484 (0.197)	0.425 (0.180)	0.645
Richness	0.389 (0.331)	0.404 (0.457)	0.128 (0.200)	0.177
Euclidean distance	0.322 (0.186)	0.565 (0.338)	0.256 (0.095)	**0.087**
Downstream to Upstream				
Number Collected	0.689 (0.612)	0.872 (0.392)	0.436 (0.228)	0.426
Diversity	0.302 (0.296)	0.184 (0.114)	0.160 (0.126)	0.519
Richness	0.349 (0.260)	0.189 (0.210)	0.239 (0.160)	0.221
Euclidean distance	0.288 (0.157)	0.538 (0.462)	0.129 (0.039)	**0.019**

Figure Legends

Figure 1.   Non-metric multidimensional scaling ordination (NMDS) of sites in ordination space. Axes are scaled proportionate to the longest axis (% of max). Symbols are the 66 localities (3 per stream) coded by dam state as described in text. Axes 2 and 3 explained 24.1 and 29.3% of the total variation, respectively. Figure 1a depicts site groupings and Figure 1b depicts vectors connecting localities within a stream. Vectors originate downstream and terminate upstream.

Figure 2. The average proportion of the entire catch as herbivores for streams of each dam state (a) and each locality (b).   Letters reflect significant differences at $\alpha = 0.10$.

Figure 3. The average proportion of the entire catch as predators for streams of each dam state (a) and each locality (b).   Letters reflect significant differences at $\alpha = 0.10$.

Figure 4. The average proportion of the entire catch as insectivores for streams of each dam state (a) and each locality (b).   Letters reflect significant differences at $\alpha = 0.10$.

Figure 5. The average proportion of the entire catch as narrow endemics for streams of each dam state (a) and each locality (b).   Letters reflect significant differences at $\alpha = 0.10$.

Figure 6.   The average total catch (a) and proportional change in total catch between localities (b, c, d) for each dam state. Letters reflect significant differences at $\alpha = 0.10$
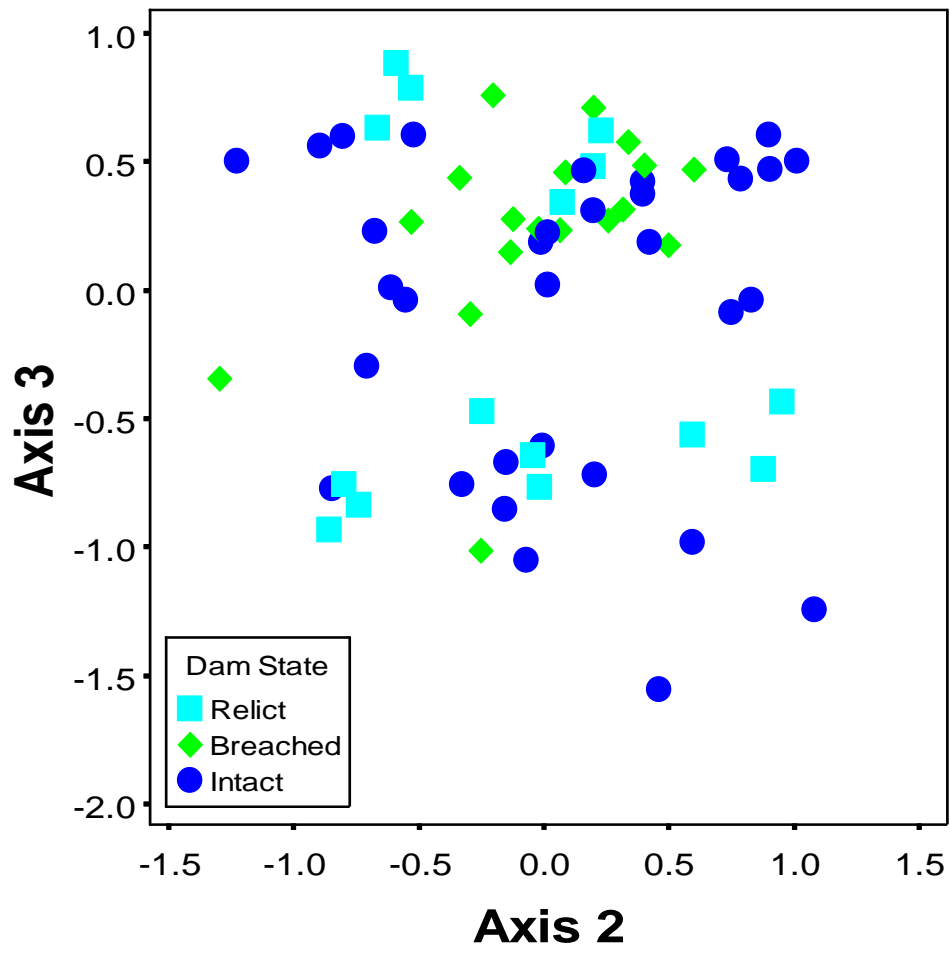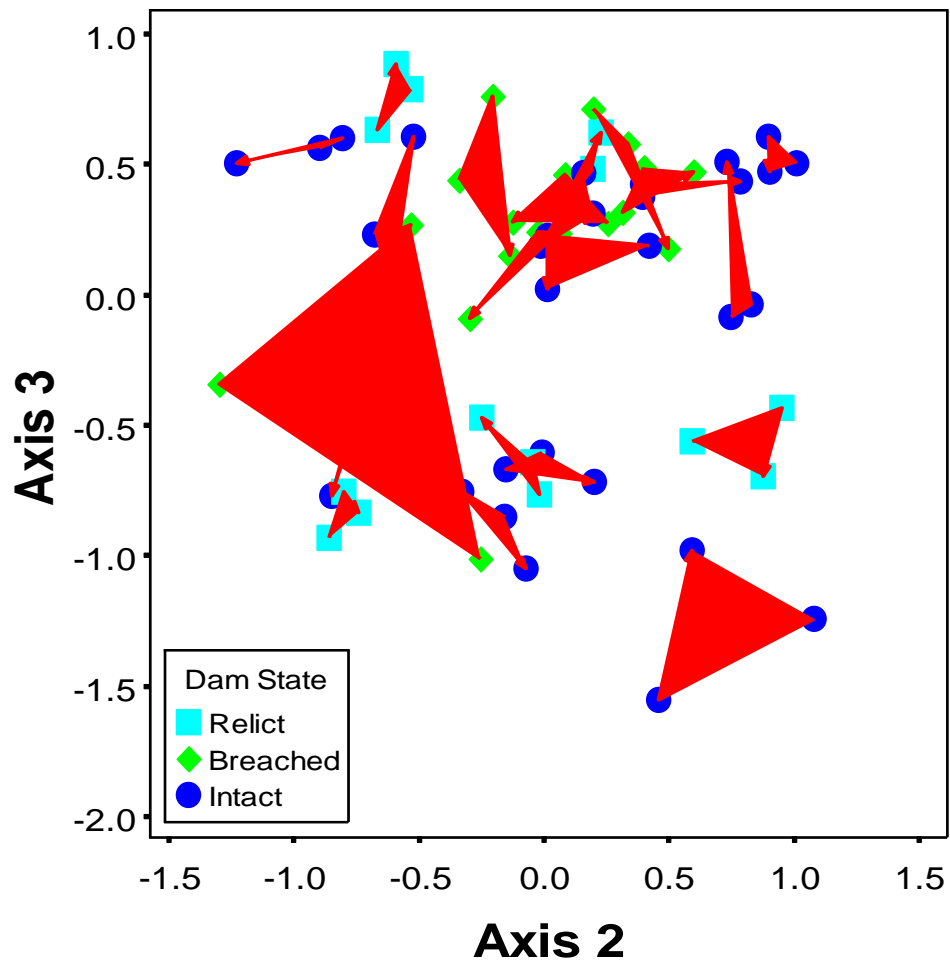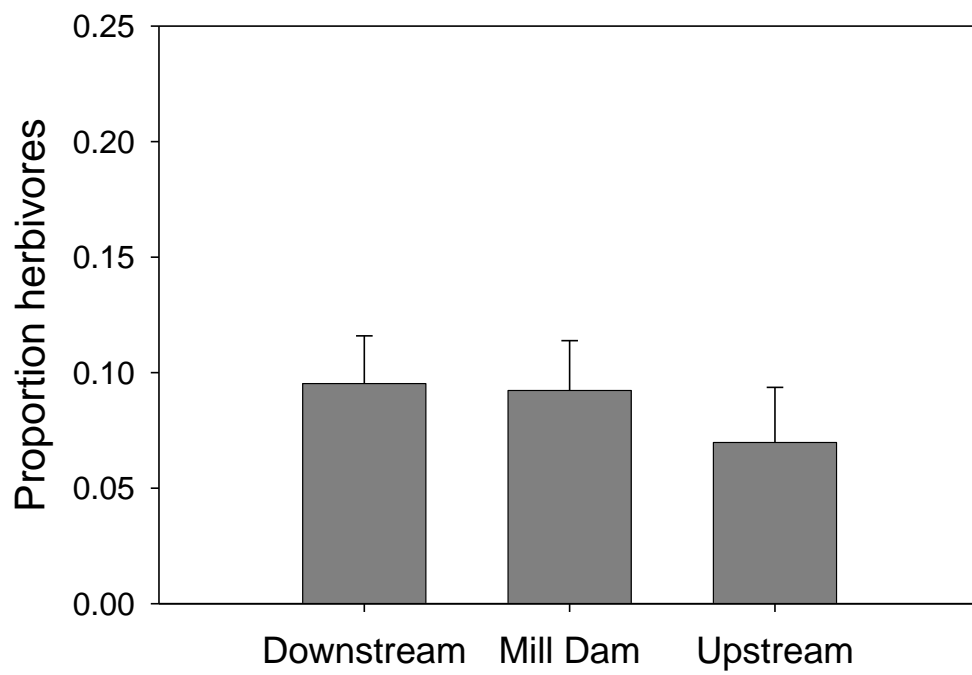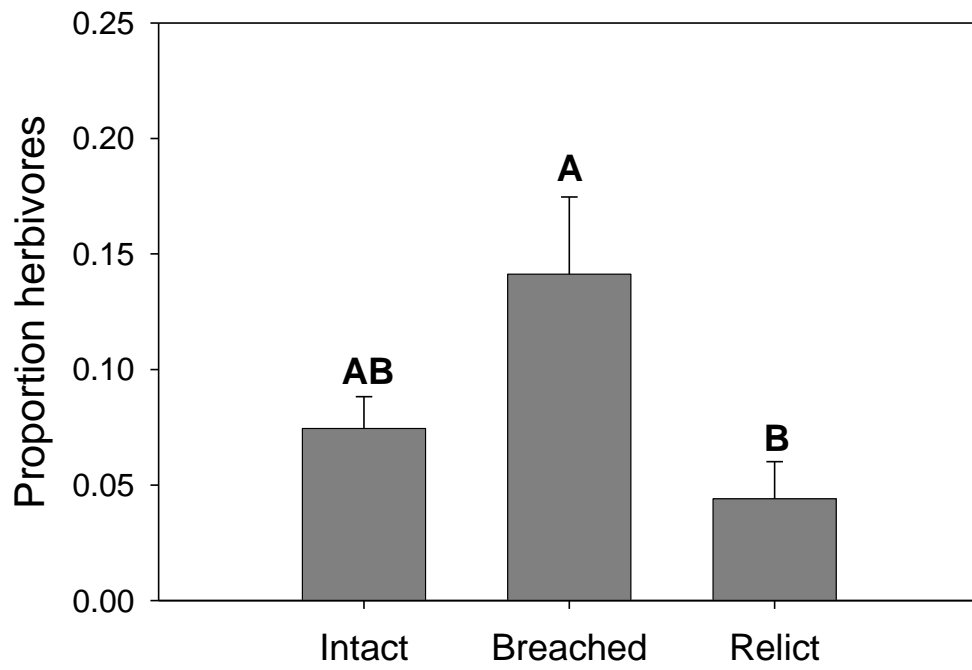
Fig. 1a

Fig. 1b

Fig. 2
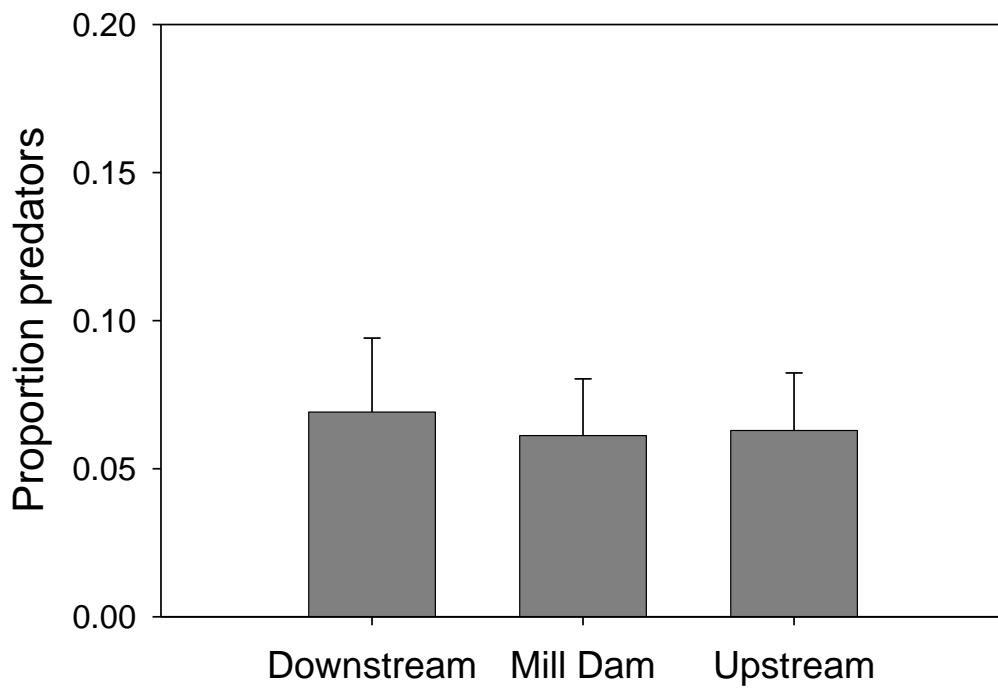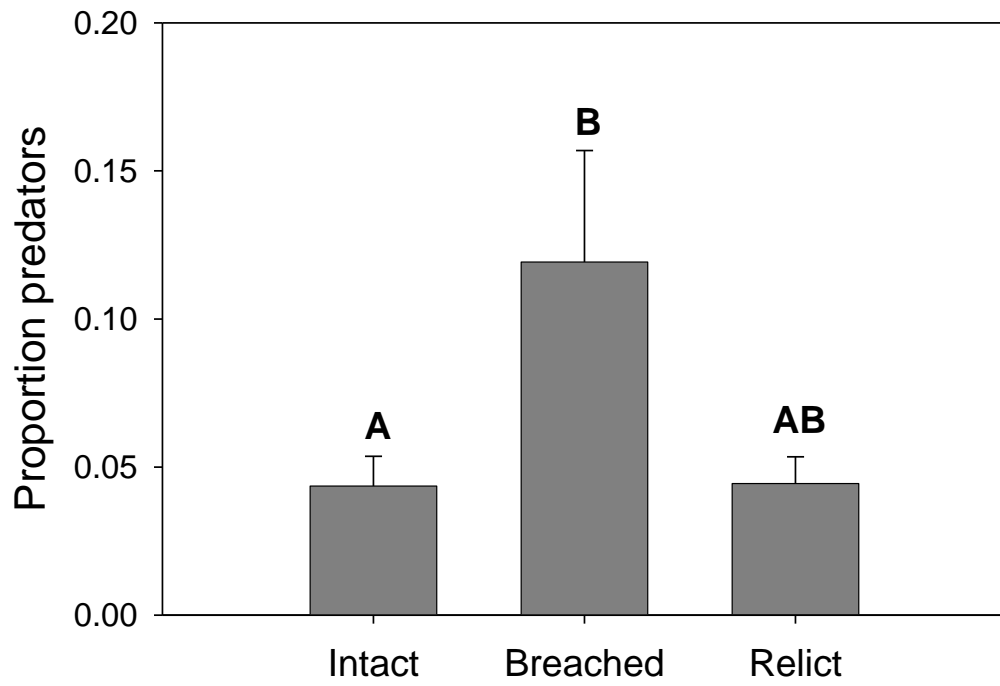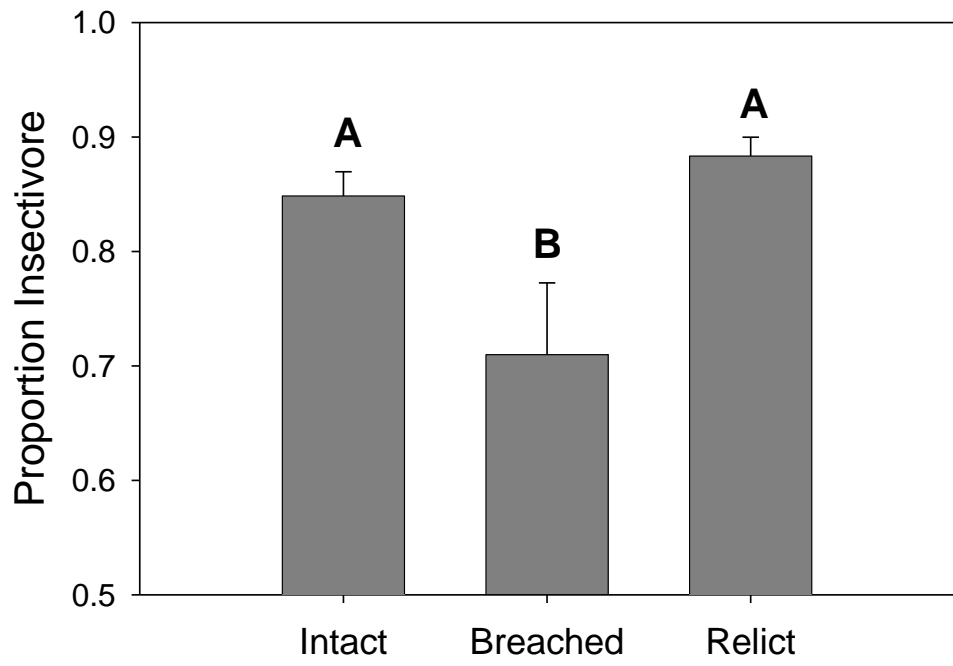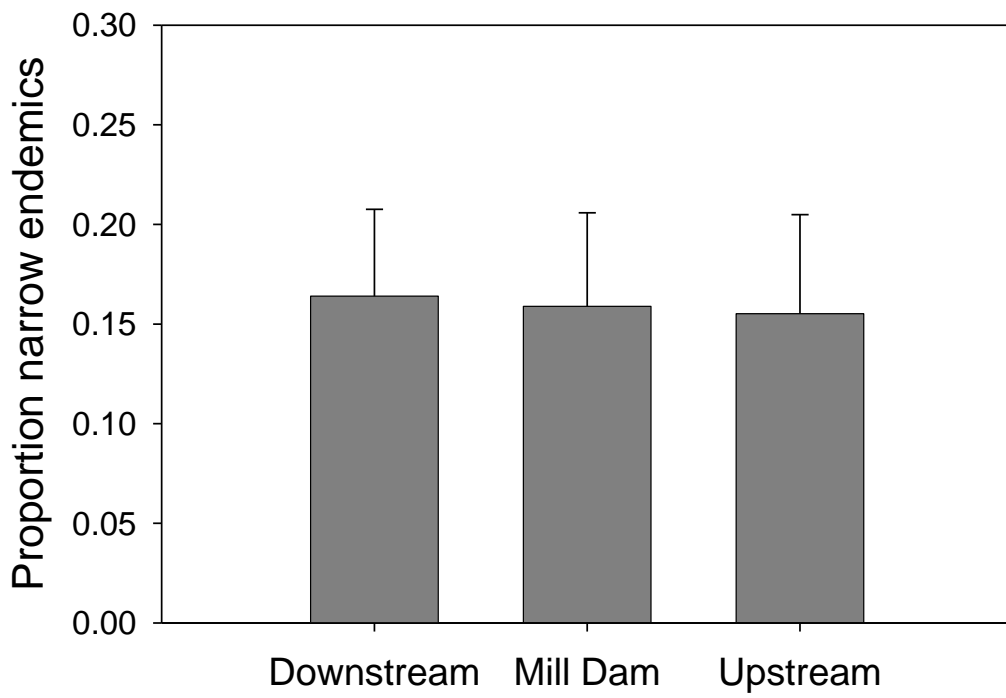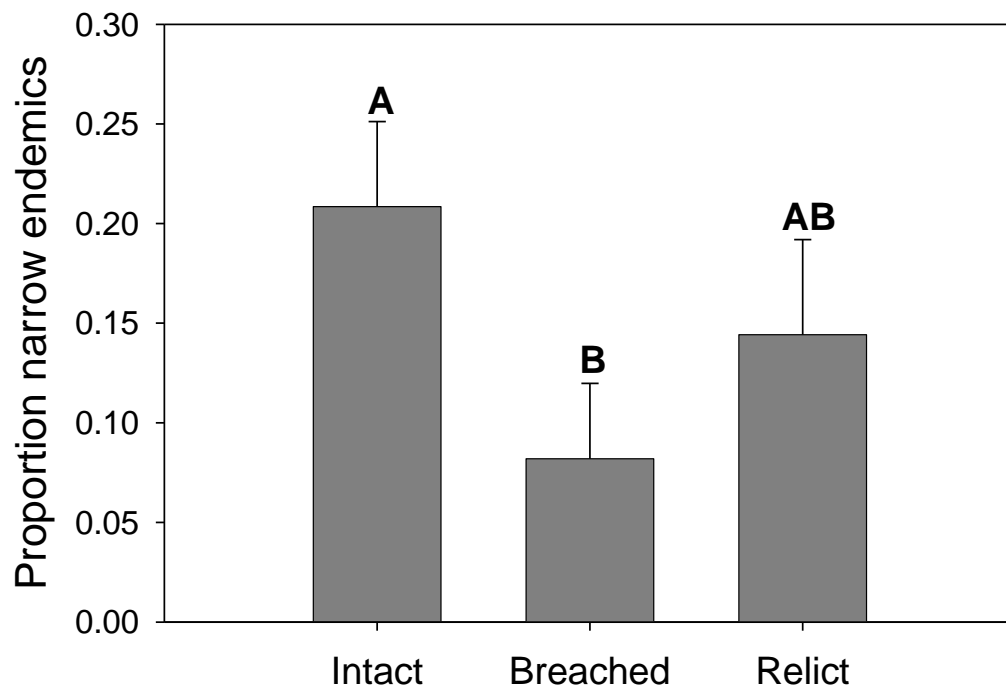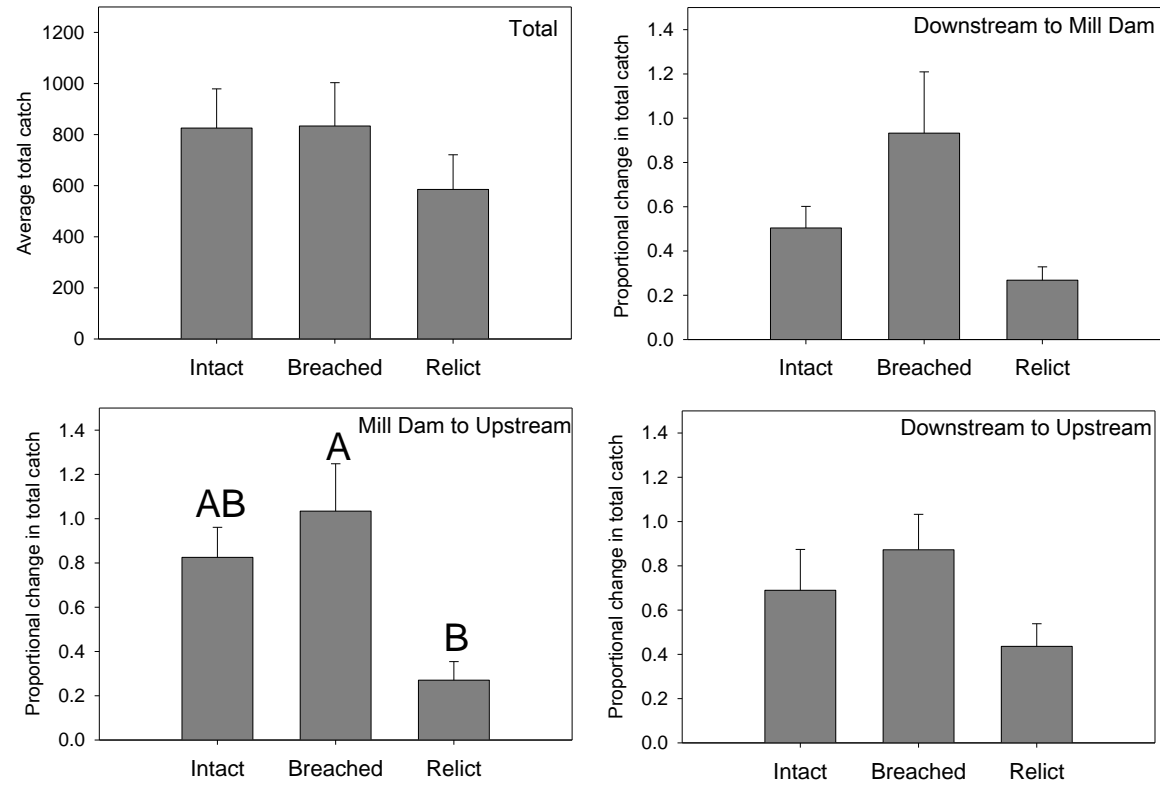
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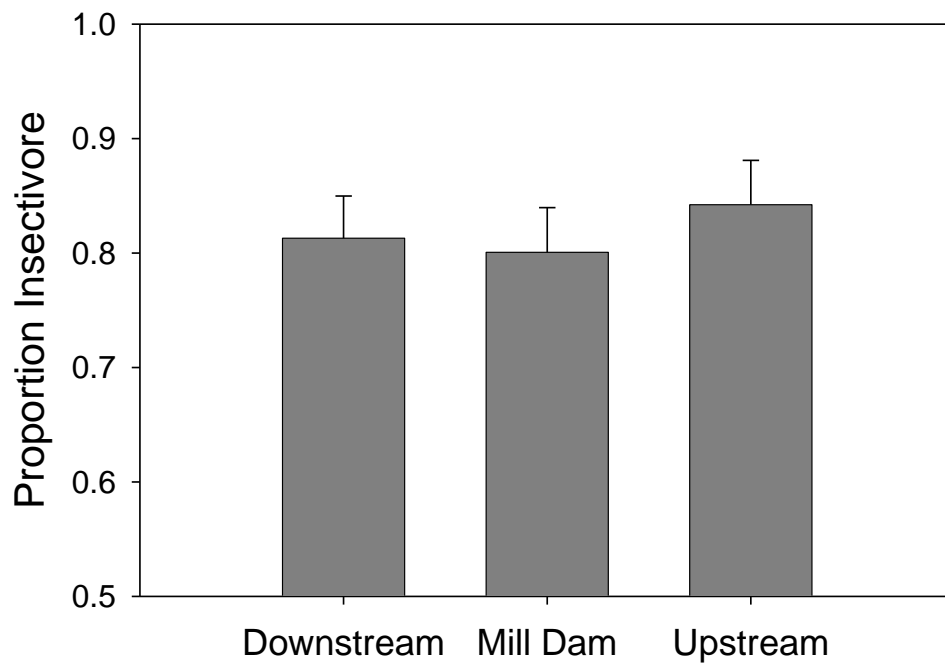
Fig. 4

Fig. 5

Fig. 6

Chapter 5: Analysis of the effects of mill and lowhead dams on crayfish assemblages

Contributing Authors: Emily Hartfield, Michael Gangloff, and Jack Feminella

Abstract

Dams are numerous and widespread in many southeastern U.S. streams, with >10,000 dams in Alabama alone. Physical, chemical, and biological impacts of dams can be dramatic, and may include altering flow and sediment regimes and channel geomorphology, and blocking migration by fishes and other mobile organisms. These physical barriers to migration can create habitat fragmentation and population isolation, which can decrease genetic diversity, making populations more vulnerable to extinction. In 2006 and 2007 we quantified crayfish and their habitats at reaches located upstream, immediately downstream, and $\geq$ 500m downstream of 22 low-head mill dams in 9 river drainages in Alabama. Of these dams, 10 are intact, 7 are partially breached, and 6 are relics. Data indicate that crayfish abundance at mill sites with intact dams is significantly lower than those sites upstream or further downstream; crayfish abundance upstream of breached dams is significantly higher than at mill or downstream sites; and crayfish abundance is similar among all sites on streams with relic dams.

Introduction

Dams are numerous and widespread in many southeastern U.S. streams, with >10,000 dams in Alabama alone (AL Office of Water Resources

) and up to 44% of the mainstem Alabama, Coosa, and Tallapoosa Rivers being impounded (Irwin *et al.* 2007). Physical, chemical, and biological impacts of dams can be dramatic, and may include altering flow and sediment regimes and channel geomorphology, and blocking upstream migration by fishes and other mobile organisms (Baxter 1977; Blalock and Sickel 1996; Watters 1996; Dean *et al.* 2002; Poff and Hart 2002; McLaughlin *et al.* 2006).

The effects of large dams on aquatic organisms and their habitats in large rivers have been well documented, whereas little research has been done to assess effects of small, surface-release, or low-head dams on low-order streams (Watters 1996; Dean *et al.* 2002; Lessard and Hayes 2003). Low-head dams are dams with a hydraulic height of $\leq 8m$ and are typically overflow or spill-way structures (IFC Consulting Report 2005). According to census records, >65,000 low-head dams existed in the eastern US by 1840, most of which were built for water-powered milling (Walter and Merritts 2008). The few studies designed to examine effects of small dams have report similar types of alterations, but are smaller in magnitude than those of large dams (Graf 2006).

Alabama supports 60% of North America's native mussel species, 38% of its native fishes, and 24% of its native crayfishes, including many endemic to the southeastern US (Lydeard and Mayden 1995; Crandall et al. 2000; Schuster and Taylor 2004). Impoundments have the potential to cause loss of habitat critical for many imperiled species (Travnicheck et al. 1995; Jager et al 2001; Dean et al. 2002; Lessard and Hayes 2003; Irwin et al. 2007). In the southeastern US, including Alabama, the high prevalence of dams along

streams and rivers has the potential to affect many imperiled aquatic species.

Little is known about how dams affect habitat conditions of freshwater

crustaceans, particularly crayfish, although impounded waters likely produce

impacts on these populations similar to those of other stream animals (Miya

and Hamano 1988).

Methods

Physicochemical habitat parameters

At each dam, we established three 150 m study reaches. One reach

extended from the dam to 150 m downstream (Mill reach), a second was

located >500 m downstream from the dam, and a third reach was located in a

free-flow reach upstream of the impoundment. Within each study reach, we

established 16 transects at 10 m intervals. We measured current velocity and

depth at 5 evenly-spaced points along each transect. Depth and velocity

measurements correspond to points that were later excavated for quadrats.

We measured channel width and substrate composition (20 particles per

transect, N = 160 per site) along transects.

We continuously measured water temperature (at 3-h intervals) using

iButton data loggers deployed at the upstream, downstream, and mill dam

sites. A 4th data logger was deployed in the impoundment. We collected grab

samples to measure water chemistry during low water conditions in July and

August 2007 from each study reach and also from the impoundment. We

measured NO_3-N (mass spectrometry, APHA 1998), PO_4 (persulfate digestion

and UV analysis, APHA 1998). Additionally, we measured conductivity (C66

Sharp meter), pH (Sharp pH52 meter), and dissolved oxygen (YSI 55 meter) during mid-summer.

Crayfish sampling

We quantified crayfish abundance using a combination of trapping, seining, and electrofishing, thus minimizing sampling bias of any single method (Rabeni et al.1997, Ratcliffe & DeVries 2004). At each focal site, we deployed 8 crayfish traps, baited with canned cat food, and left them over night once during the summer or early fall. Additionally, we used seines to isolate stream mesohabitats and then used an electrofishing unit to push crayfish and fish into the seine. We used seines and the electrofishing unit to collect between 9 and 15 samples within each study reach. Samples were collected from all mesohabitats in proportion to their availability in the study reach and a minimum of 3 replicate samples was collected from each mesohabitat.

Crayfish were frozen and later preserved in 95% EtOH for use in molecular analyses (Chapter 6). All crayfish were identified to species or the lowest practical taxonomic level using keys in Hobbs (1981, 1989) and Taylor and Schuster (2004). Problematic specimens were sent to Dr. Geunter Schuster (Eastern Kentucky University) for confirmation. All crayfish were deposited in the Auburn University Museum of Natural History.

Total richness (number of species) and abundance of all crayfish (total number of individuals collected by all methods) were determined for each reach. Trapping success (crayfish per trap) and electrofishing success (number crayfish per shocking/kick-seining minute) also were used as indices

of crayfish abundance in order to compare the separate methods and determine if one is a better indicator for abundance or if all three are necessary to accurately estimate population size.

The Kruskal-Wallace test was used to test the null hypothesis that crayfish abundance and richness do not differ between dams in different conditions (intact, breached, and relic). This is the non-parametric equivalent of an ANOVA and uses ranks in place of values to avoid the assumption of normality. Friedman's Test was used to test the null hypothesis that crayfish abundance and richness do no differ between sites (Upstream, Mill, and Downstream) at streams with different dam conditions. Friedman's test is a non-parametric equivalent to a 2-way ANOVA, and was used to control for variation between streams. Multiple comparisons were done by using the Friedman's Test on each pair of treatments and making a Bonferroni correction.

Results

We collected 19 crayfish taxa from 22 third-sixth order Alabama streams. Crayfish total abundance and richness did not differ significantly across streams with different types of dams (H=0.0331, 2 df, P=0.9836; and H=0.2437, 2 df, P=0.8853, respectively). Mean CPUE differed significantly across streams with dams in different conditions (H=7.9233, 2 df, P=0.0190). Overall trap success was significantly lower for sites with breached dams (1.28 crayfish/trap) than for sites with relic dams (2.77 crayfish/trap), but not for sites with intact dams (2.32 crayfish/trap; Fig. 1.).

Mean Trap Success
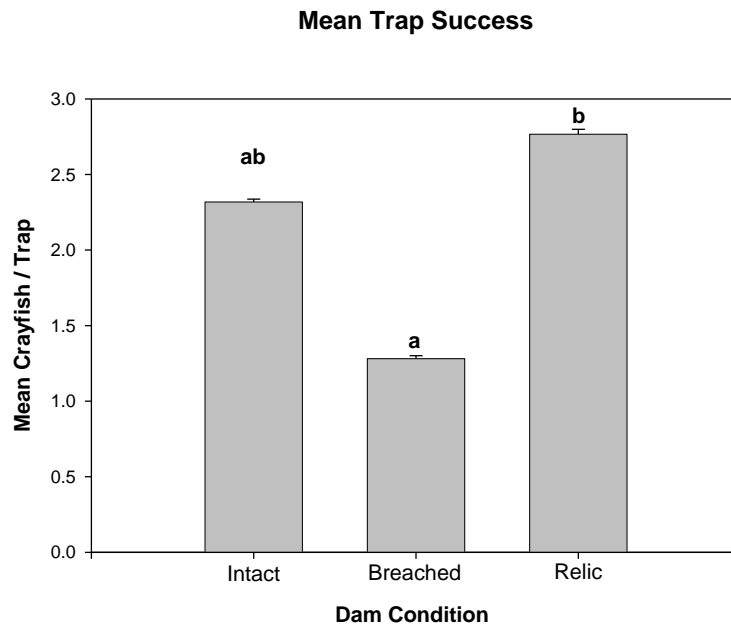
Figure 1. Mean crayfish per trap (CPUE) in streams with intact, breached, or relic dams. Bars sharing letters are not significantly different from one another.

On streams with breached dams, we observed that crayfish abundance was significantly lower at mill sites than at upstream sites (X_r^2=7.8947, 2 df, P=0.0193), crayfish abundance at downstream sites did not differ significantly from mill sites. Abundance did not differ significantly between streams with intact or relic dams (Fig. 1).

Total Abundance
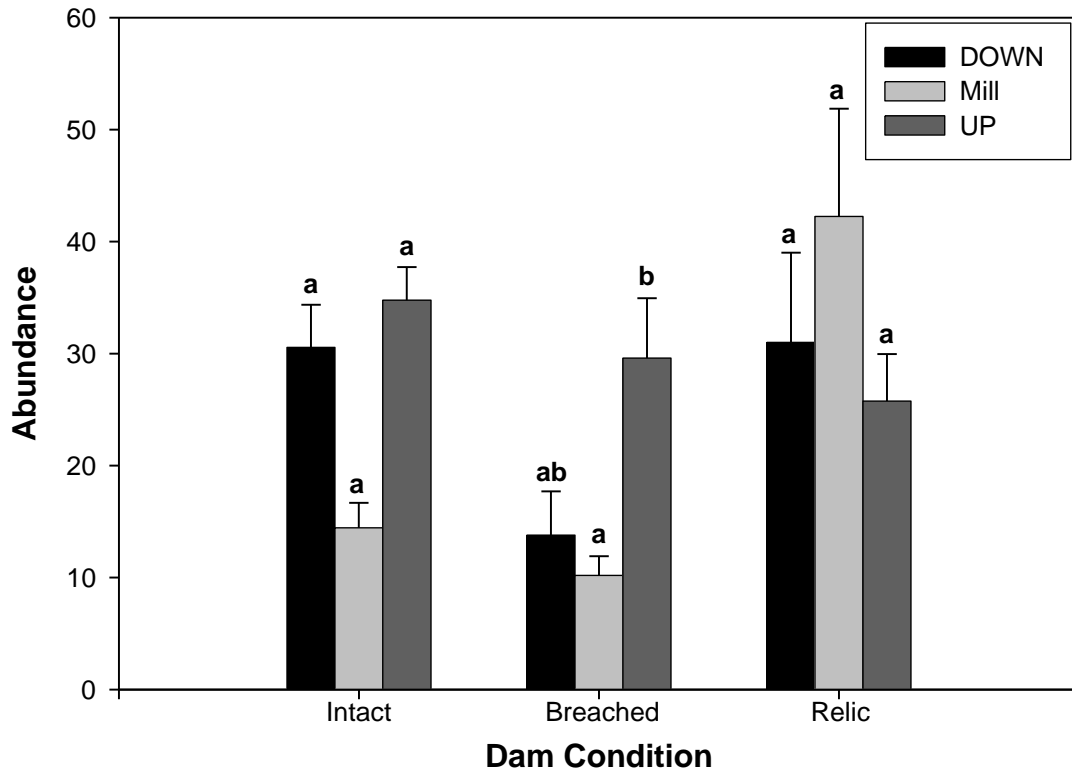
Figure 2. Crayfish total abundance (number captured using all techniques) at upstream, mill dam, and downstream sites in 22 Alabama streams. Bars sharing the same letter are not significantly different from one another.

Crayfish trap success at mill sites with intact dams is significantly lower than those sites upstream or further downstream (X_r^2=21.88, P<0.0001); crayfish abundance upstream of breached dams is significantly higher than at mill or downstream sites (X_r^2=10.83, P=0.0045); and crayfish abundance is similar among all sites on streams with relic dams (X_r^2=1.90, P=0.3858; Fig. X).

Mean Trap Success
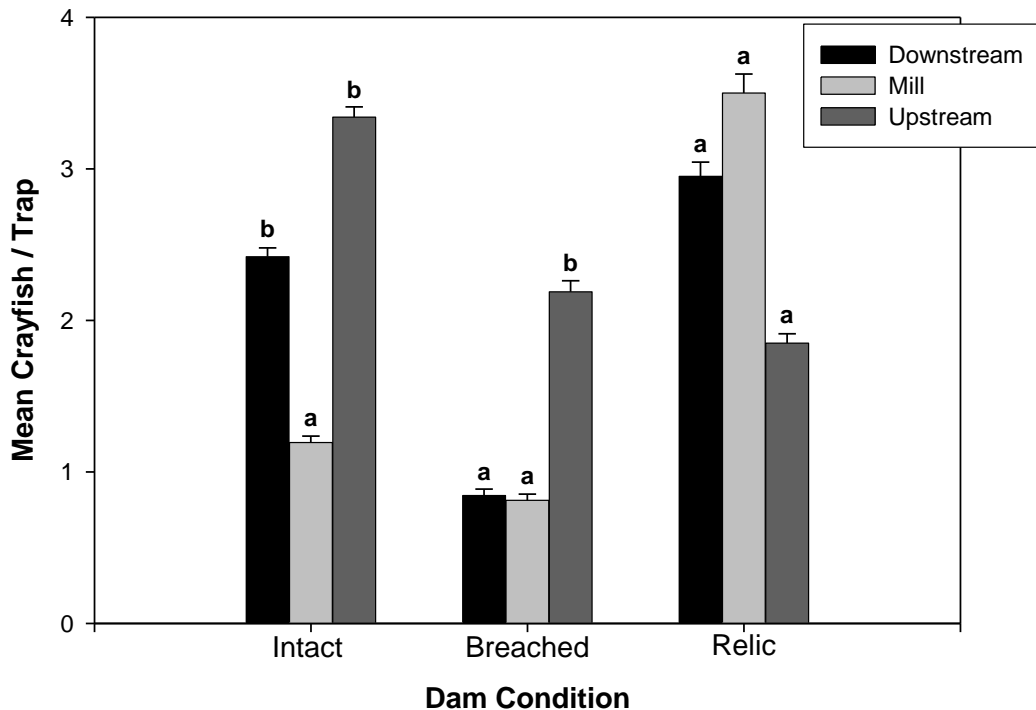
Figure 3. Differences in crayfish abundance (CPUE) across up- mill dam, and downstream sites associated with intact, breached or relic small dams.

Discussion

We found that crayfish abundance was frequently lower immediately downstream from intact and breached dams. It is possible that reductions in crayfish abundance near dams may be due to fish aggregations. We found that fish assemblages downstream of dams often are dominated centrarchids and other predatory fishes (Chapter 4). Curiously, many downstream (i.e. >500m downstream from a dam) sites appear to also support very few crayfish. It is also possible that episodic releases of sediments from behind small dams may reduce the availability of crevices, interstitial space, and other refugia. Episodic or chronic sediment pulses may reduce crayfish

habitat quality for large distances downstream of the disturbance. Large pulses of fine sediments may reduce stream crayfish habitat heterogeneity or stability.

Small dams appear to affect crayfish more acutely than they do mollusks or fishes. Breached dams appear to have a greater negative effect on crayfish, compared to intact or relic dams. This may be because breached dams both degrade habitat and aggregate predaceous fish.

Literature Cited

Baxter, R.M. 1977. Environmental effects of dams and impoundments. *Annual Review of Ecology and Systematics* 8:255-283.

Blalock, H.N. and J.B. Sickel. 1996. Changes in mussel (Bivalvia: Unionidae) fauna within the Kentucky portion of Lake Barkley since impoundments. *American Malacological Bulletin* 13:111-116.

Crandall, K.A, D.J. Harris, J.W. Fetzner Jr. 2000. The monophyletic origin of freshwater crayfish estimated from nuclear and mitochondrial DNA sequences. *Proclamations of the Royal Society of London* 267:1679-1686.

Dean, J., D. Edds, D. Gillette, J. Howard, S. Sherraden, and J. Tiemann. 2002. Effects of lowhead dams on freshwater mussels in the Neosho River, Kansas. *Transactions of the Kansas Academy of Science* 105:323-240.

Graf, W.L. 2006. Downstream hydrologic and geomorphic effects of large dams on American rivers. *Geomorphology* 79:336-360.

Hobbs, H.H., Jr. 1981. The crayfishes of Georgia. *Smithsonian*

Contributions to Zoology 318:1-549.

Hobbs, H.H., Jr. 1989. An illustrated checklist of the American crayfishes (Decapoda: Astacidae, Cambaridae, and Parastacidae). *Smithsonian Contributions to Zoology* 480:1-236.

IFC Consulting. 2005. A summary of existing research on low-head dam removal projects, Technical Report, AASHTO.

Irwin, E., G. Turner, K. Mickett, and T. Piper. 2007. ACT Aquatic Gap and water-quality monitoring: Final Report. ADCNR.

Jager, H.I., J.A. Chandler, K.B. Lepla, and W.V. Winkle. 2001. A theoretical study of river fragmentation by dams and its effects on white sturgeon populations. *Environmental Biology of Fishes* 60:347-361.

Lessard, J.L. and D.B. Hayes. 2003. Effects of elevated water temperature on fish and macroinvertebrate communities below small dams. *River Research and Applications* 19:721-732.

Lydeard, C. and R.L. Mayden. 1995. A diverse and endangered aquatic ecosystem of the Southeast United States. *Conservation Biology* 9:800-805.

McLaughlin, R.L., L. Porto, D.L.G. Noakes, J.R. Baylis, L.M. Carl, H.R. Dodd, J.D.Goldstein, D.B. Hayes, and R.G. Randall. 2006. Effects of low-head barriers on stream fishes: taxonomic affiliations and morphological correlates of sensitive species. *Canadian Journal of Fisheries and Aquatic Sciences* 63:766-779.

Miya, Y. and T. Hamano. 1988. The influence of a dam having no fish-way on the distribution of decapod crustaceans in the Yukinoura River, Nagasaki, Japan. *Nippon Suisan Gakkaishi* 54:429-435.

Poff, N.L., and D.D. Hart. 2002. How dams vary and why it matters for the emerging science of dam removal. *BioScience* 52:659-668.

Rabeni, C.F., K.J. Collier, and S.M. Parkyn. 1997. Evaluating techniques for sampling stream crayfish (*Paranephrops planiforms*). *New Zealand Journal of Marine and Freshwater Research* 31:693-700.

Ratcliffe, J.A. and D.R. DeVries. 2004. The crayfishes (Crustacea: Decapoda) of the Tallapoosa River Drainage, Alabama. *Southeastern Naturalist* 3:417-430.

Schuster, G.A., and C.A. Taylor. 2004. Report of the crayfishes of Alabama: literature and museum database review, species list with abbreviated annotations and proposed conservation statuses. Illinois Natural History Survey, Center for Biodiversity Technical Report.

Taylor, C.A. and G.A. Schuster. 2004. *The crayfishes of Kentucky*. Illinois Natural History Survey. Special Publication No. 28:1-219.

Travnicheck, V.H., M.B. Bain, M.J. Maceina. 1995. Recovery of a warmwater fish assemblage after the initiation of minimum-flow release downstream from a hydroelectric dam. *Transactions of the American Fisheries Society* 124:836-844.

Walter, T.C., and D.J. Merritts. 2008. Natural streams and the legacy of water-powered mills. *Science* 319:299-304.

Watters, G.T. 1996. Small dams as barriers to freshwater mussels (Molusca: Pelecypoda: Unionidae) and their hosts. *Biological Conservation* 75:79-85.

Table 1. List of crayfish species collected at dam focal sites 2006-2008.

Drainage	Stream	Dam	Status	Latitude	Longitude	Crayfish Species
Alabama	Big Flat	Rikard's Mill	Intact	31.7821	-87.2229	*Orconectes holti*
						Procambarus spiculife
Cahaba	Cahaba	Grant's Mill	Relict	33.5089	-86.6436	*Orconectes virilis*
						Cambarus latimanus
Cahaba	Little Cahaba	n/a	Intact	33.4512	-86.6935	*Orconectes virilis*
						Cambarus latimanus
						Cambarus striatus
Black Warrior	Lost	Boshell's Mill	Intact	33.8545	-87.4143	*Orconectes perfectus*
Black Warrior	Brushy	Brushy Lake Dam	Intact	34.2919	-87.2733	*Cambarus striatus*
						Cambarus obstipus
						Orconectes validus
Black Warrior	Blue Springs	Chamblee's Mill	Relict	34.0600	-86.6617	*Cambarus striatus*
						Orconectes erichsonianus
Chattahoochee	Halawakee	Bean's Mill	Intact	32.6969	-85.2668	*Cambarus howardi*
						Cambarus latimanus
						Procambarus spiculifer
Chattahoochee	Osanippa	Ferguson's	Relict	32.7778	-85.1928	*Cambarus howardi*

Drainage	Stream	Dam	Status	Latitude	Longitude	Crayfish Species
		Mill				*Procambarus spiculifer*
Chattahoochee	Little Uchee	Meadow's Mill	Intact	32.5283	-85.2531	*Cambarus striatus*
						Procambarus spiculifer
Choctawhatchee	Pea	Shellgrove Mill	Relict	31.5214	-85.8685	*Procambarus versutus*
Coosa	Big Canoe	Goodwin's Mill	Breached	33.8192	-86.3841	*Cambarus coosae*
						Orconectes latimanus
						Orconectes erichsonianus
Coosa	Yellow Leaf	Shannon's Mill	Intact	32.9355	-86.6114	*Cambarus coosae*
						Procambarus spiculifer
Coosa	Hatchett	Old AL Power Mill	Intact	33.0684	-86.0960	*Cambarus coosae*
						Cambarus latimanus
Tallapoosa	Choctafaula	Vaughn's Mill	Breached	32.512	-85.5783	*Cambarus halli*
						Procambarus versutus
						Procambarus verrucosus
Tallapoosa	Little Hillabee	Carr Mill	Relic	33.2047	-85.9446	*Cambarus striatus*
						Cambarus halli
						Procambarus spiculifer
Tallapoosa	Loblockee	Macon's Mill	Intact	33.6533	-85.5842	*Cambarus halli*
						Procambarus versutus

Drainage	Stream	Dam	Status	Latitude	Longitude	Crayfish Species
Tallapoosa	Sandy	Jone's Mill	Intact	32.7508	-85.5596	*Cambarus striatus*
						Cambarus englishi
						Procambarus verrucosus
Tennessee	Paint Rock	Butler's Mill	Relic	34.5794	-86.3011	*Orconectes erichsonianus*
						Cambarus striatus
Tennessee	Clear Fork	Masterson Mill	Intact	34.5385	-87.2832	*Orconectes putnami*
						Orconectes validus
						Cambarus striatus
Tombigbee	Buttahatchee	n/a	Intact	34.1261	-87.8369	*Orconectes validus*
Tombigbee	New	Kelly's Mill	Intact	33.9297	-87.6800	*Orconectes perfectus*
						Cambarus striatus
Tombigbee	Pearce's Mill	Pearce's Mill	Breached	34.1220	-87.8364	*Orconectes validus*
						Cambarus striatus
						Cambarus obstipus

Analysis of the effects of mill and lowhead dams on crayfish population genetics

Emily Hartfield, Michael Gangloff, Scott Santos, Jack Feminella

Abstract

Dams are numerous and widespread in many southeastern U.S. streams, with >10,000 dams in Alabama alone. Physical, chemical, and biological impacts of dams can be dramatic, and may include altering flow and sediment regimes and channel geomorphology, and blocking migration by fishes and other mobile organisms. These physical barriers to migration can create habitat fragmentation and population isolation, which can decrease genetic diversity, making populations more vulnerable to extinction. We examined genetic diversity in widespread (*Cambarus striatus*) and narrowly endemic (*Cambarus coosae*) crayfish populations in 2 impounded Alabama Piedmont streams (Sandy and Hatchet creeks). We collected 30 crayfish from both up and downstream of each dam. We extracted whole genomic DNA and amplified a fragment of the CO1 gene. To date, 10 individuals from both (upstream and downstream) populations in 2 streams have been sequenced and analyzed. Hapoltype networks suggest extensive genetic diversity within the widely-distributed *C. striatus* whereas *C. coosae* exhibited very low genetic diversity. Moreover, haplotype data indicate suggest divergent lineages within Sandy Creek *C. striatus* populations. These differences may be attributable to the

greater age of Jones Mill (150 y) relative to the Hatchet Creek dam or they may indicate cryptic diversity within this poorly-known group of freshwater invertebrates.

Introduction

Dams are numerous and widespread in many southeastern U.S. streams, with >10,000 dams in Alabama alone (AL Office of Water Resources http://www.adeca.alabama.gov/) and up to 44% of the mainstem Alabama, Coosa, and Tallapoosa Rivers being impounded (Irwin *et al.* 2007). Physical, chemical, and biological impacts of dams can be dramatic, and may include altering flow and sediment regimes and channel geomorphology, and blocking upstream migration by fishes and other mobile organisms (Baxter 1977; Blalock and Sickel 1996; Watters 1996; Dean *et al.* 2002; Poff and Hart 2002; McLaughlin *et al.* 2006).

The effects of large dams on aquatic organisms and their habitats in large rivers have been well documented, whereas little research has been done to assess effects of small, surface-release, or low-head dams on low-order streams (Watters 1996; Dean *et al.* 2002; Lessard and Hayes 2003). Low-head dams are dams with a hydraulic height of $\leq$10m and are typically overflow or spill-way structures (IFC Consulting Report 2005). According to census records, >65,000 low-head dams existed in the eastern US by 1840, most of which were built for water-powered milling (Walter and Merritts 2008). The few studies designed to examine effects of small dams have reported similar types of alterations, but are smaller in magnitude than those of large dams (Graf 2006).

Alabama supports 60% of North America's native mussel species, 38% of native fishes, and 24% of native crayfishes, many of which are endemic to the southeastern US (Lydeard and Mayden 1995; Crandall et al. 2000; Schuster and Taylor 2004). Impoundments have the potential to cause loss of habitat critical for many imperiled species (Travnicheck et al. 1995; Jager et al. 2001; Dean et al. 2002; Lessard and Hayes 2003; Irwin et al. 2007). In the southeastern US, including Alabama, the high prevalence of dams along streams and rivers has the potential to affect many imperiled aquatic species.

Ecological theory predicts that isolation from habitat fragmentation can decrease population size and genetic diversity, making populations more vulnerable to extinction (MacArthur and Wilson 1967; Lande 1988; Lande 1999). Isolation may occur from either natural or anthropogenic barriers to migration (Dillon 1988; Santos 2006). Genetic drift may result after such separations because rare alleles become common or fixed in a population while other alleles become less frequent or disappear (Lande 1976). Divergence of separated populations can be quantified by analyzing accumulation of mutations in the genome (Nei 1977). DNA sequences also can be used to estimate the time since separation by quantifying the number of genetic changes becoming fixed between populations since separation (Sarich and Wilson 1973). Natural selection may expedite the divergence between reproductively isolated populations occurring in different habitats (Felsenstein 1976), but isolation has a greater influence on genetic divergence between populations than does habitat variation and selection (Dillon 1984; Finlay *et* al. 2006).

Physical barriers such as dams are capable of impeding longitudinal movements of stream organisms (Baxter 1977; Watters 1996; Dean *et al.* 2002). Upstream migration of fishes and other mobile organisms can be halted by dams, preventing individuals from reaching feeding and/or spawning habitat, which may cause population declines (Raymond 1979; Larinier 2001). Among fish, decreased longitudinal connectivity across streams may cause habitat fragmentation and population isolation (Neraas and Spruell *et al.* 2001; Olden *et al.* 2001). One-way (downstream) migration of fish, commonly observed in impounded systems, may reduce genetic diversity and population size, especially in upstream sections (Jager *et al.* 2001; Morita and Yamamoto 2002; Yamamoto *et al.* 2004). Dams can have similar effects on freshwater mussels by restricting migration and distribution of their host fish through impounded sections (Watters 1996; Kelner and Sietman 2000). Little is known about how dams affect habitat conditions of freshwater crustaceans, particularly crayfish, although impounded waters likely produce impacts on these populations similar to those of other stream animals (Miya and Hamano 1988).

We predicted to find a structured population with fewer haplotypes and lower genetic diversity at upstream sites than at mill or downstream sites, reflecting reduced crayfish movement upstream within intact dams and, thus, genetic isolation of upstream populations from downstream sites

Methods

We selected 2 focal sites with intact dams, Sandy Creek (Tallapoosa Drainage) and Hatchett Creek (Coosa Drainage), and 2 locally common crayfish, *Cambarus striatus* (Sandy Creek) and *Cambarus coosae* (Hatchett Creek) to examine gene flow between crayfish populations across impoundments. *Cambarus striatus* (common name, the ambiguous crayfish) is a widespread, versatile species, capable of exploiting diverse habitat types through out the Southeast and employing different life history strategies. *Cambarus coosae* (Coosa crayfish) is endemic to the Coosa River Drainage and is an obligate stream-dwelling crayfish (Hobbs 1981). A sub-objective of this study was to compare effects of dams on the population genetics of both endemic and widespread crayfish species.

Both crayfish species are abundant at the respective study sites and it was relatively easy to collect large sample sizes (Chapter 5, Table 1). We collected tissue samples from the abdominal muscle of EtOH-preserved specimens from 30 individuals from each stream, including 10 individuals from each of the upstream, mill, and downstream study reaches (Fetzner & Crandall 2003). We quantified gene flow by examining a 590bp fragment of the mitochondrial cytochrome oxidase 1 (COI) gene. We extracted whole genomic DNA using a 2x CTAB extraction protocol (Coffroth *et al.* 1992). We amplified CO1 mitochondrial gene fragments using the primers HCO2198 and LCO1490 (Folmer *et al.* 1994). The PCR protocol for 25 µL reactions was as follows follows: 2.5 µL 10 x buffer (1.5 µM), 0.5 µL dNTPs (10 µM), 0.1 µL Taq Polymerase, 0.5 µL magnesium chloride (25 µM), 1 µL of each primer (10 µM),

and 1 μL DNA with water added to total 25 μL. PCR reactions were performed in

a PTC-100^{TM} thermocycler (MJ Reactions) using the following program: an initial

denaturing step of 96°C for 3 min, followed by 40 cycles of 94°C for 1 min,

annealing at 50°C for 1 min, and 72°C for 1 min, with a final elongation of 72°C

for 5 min.

Amplified sequences were purified using Montage^{TM} PCR Filter Units

(Millipore) according to the supplier's recommendations and sequenced using an

ABI 3100 Genetic Analyzer (Applied Biosystems). We edited sequences by

comparing them to the compliment strand using SEQUENCHER version 4.6

(Gene Codes Corporation). We aligned sequences manually using SEQ-AL

version 2.0a11 (available at http://evolve.zoo.ox.ac.uk/).

To date, we have sequenced gene fragments from 10 *C. striatus* (5 from

upstream and 5 from downstream) and 11 *C. coosae* (6 from upstream and 5

from downstream). Laboratory work will be completed by Fall 2009.

Supplementary funds have been acquired to support this project.

We calculated nucleotide (π) and haplotype (H$_d$) diversity estimates (Nei

1987) using DNASP 4.06 (Rozas *et al.* 2003). We tested genetic differentiation

between populations using the nearest neighbor statistic, S*nn* (Hudson 2000),

and gene flow using pairwise Fst (Hudson et al. 1992) values and Nm(Lynch and

Crease 1990) with >1000 permutations in DNASP 4.06. We constructed

haplotype networks using TCS 1.21 (Clement *et al.* 2000).

Once all samples have been sequenced, we will use nested clade

analysis to test for genetic differentiation among sites (Templeton et al. 1987).

To test the null hypothesis of no geographic association of haplotypes, we will use GeoDis 2.5 to calculate clade distance (D_c) and nested clade distance (D_n) (Posada *et al.* 2000) with 5000 permutations. D_c measures the geographical range of a haplotype at each nested level whereas D_n measures the evolutionary distance between two haplotypes from the center (oldest) haplotype. The GeoDis output is used to answer questions in the NCA inference key and to help explain what evolutionary events (e.g., restricted gene flow) may have led to current genetic diversity and geographic distributions (Templeton 2005).

Results

We detected 4 CO1 haplotypes in Sandy Creek *Cambarus striatus* populations. All four haplotypes were detected at sites downstream of Jones Mill dam, but only two haplotypes were detected upstream from the dam. Both nucleotide diversity (π) and haplotype diversity (H_d) were higher in downstream populations (Table 1). At Hatchet Creek, only 3 haplotypes were detected, with each site having both one shared and one unique haplotype. Haplotype diversity was slightly higher downstream from the dam compared to upstream and overall. However, nucleotide diversity was slightly higher upstream than downstream or overall. Data were also used to create a preliminary haplotype network (Figure 1).

Table 1. Nucleotide and Haplotype Diversity.

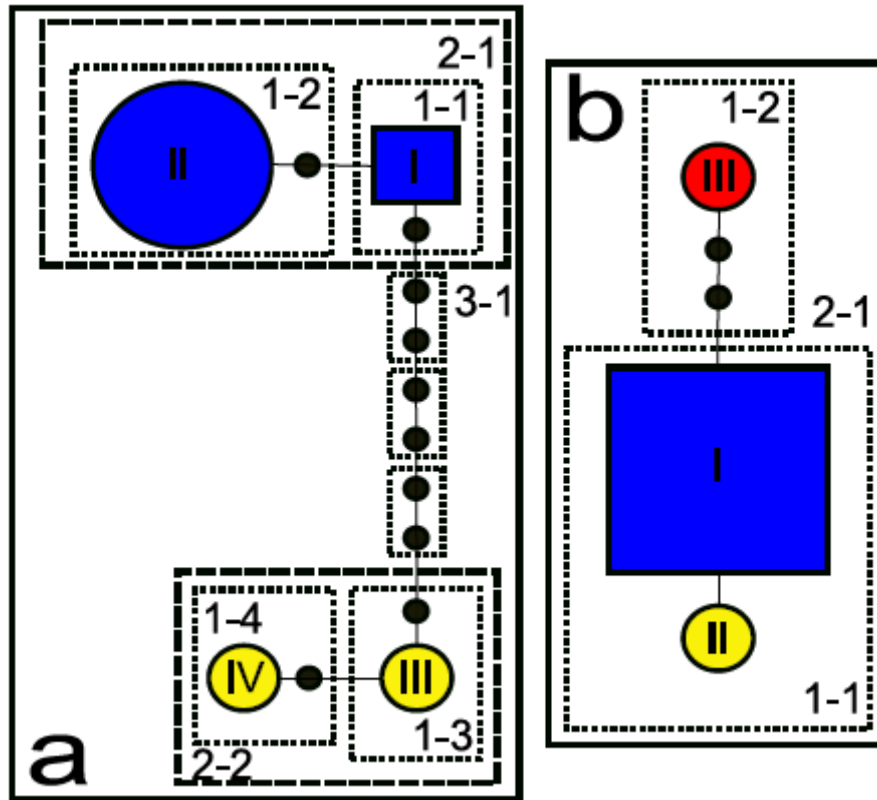| Species	Population	haplotypes	π	H_d	F_{st}	N_m	S_{nn}
C. striatus	Overall	4	.00823	.644	.19922	1.00	.48000
	Downstream	4	.01305	.900			
	Upstream	2	.00141	.400			
C. coosae	Overall	3	.00123	.345	.00000		.45455
	Downstream	2	.00068	.400			
	Upstream	2	.00169	.333			

Figure 1. Haplotype networks with nesting levels for Sandy Creek (a) and Hatchet Creek (b). Numbers represent haplotypes while black dots represent un-sampled or extinct haplotypes. Squares (I) represent ancestral haplotypes. Larger shapes represent higher frequency, but sizes are not proportional. Colors correspond to locations (yellow = downstream; red = upstream; blue = downstream and upstream).

Discussion

Relative genetic diversity measures for *C. striatus* in Sandy Creek are higher downstream than upstream, as predicted. F_{st} and N_m suggest structured populations with limited gene flow. S_{nn} values, however, suggest that migration between populations is not limited. This discrepancy could be due to small

sample size, or a one-way migration of individuals moving downstream, but not upstream.

Genetic diversity for *C. coosae* is much lower than *C. striatus* and populations show little evidence of isolation. Differences in overall diversity may be accounted for by considering range sizes of each species. Typically endemic species with restricted ranges are less genetically diverse than widespread species. *Cambarus coosae* is found only in the Coosa Basin, whereas *C. striatus* is widespread throughout Alabama and the entire Southeast. Another explanation for the lack of population structure seen in *C. coosae* could be the relative newness of the dam on Hatchett Creek, which dates back to the 1920's, whereas Jones Mill on Sandy Creek was built before 1850.

Ongoing work (E. Hartfield M.S. Thesis) is focused on increasing sample size in these systems to make diversity statistics more meaningful. In addition, CO1 fragments of *C. striatus* and *C. coosae* from other stream systems are being sequenced in order to compare range-wide diversity to system-wide diversity.

Literature Cited

Baxter, R.M. 1977. Environmental effects of dams and impoundments. *Annual Review of Ecology and Systematics* 8:255-283.

Blalock, H.N. and J.B. Sickel. 1996. Changes in mussel (Bivalvia: Unionidae) fauna within the Kentucky portion of Lake Barkley since impoundments. *American Malacological Bulletin* 13:111-116.

Clement, M. et al. 2000. TCS: a computer program to estimate gene

genealogies. *Molecular Ecology* 9:1657-1659.

Crandall, K.A, D.J. Harris, J.W. Fetzner Jr. 2000. The monophyletic origin of freshwater crayfish estimated from nuclear and mitochondrial DNA sequences. *Proclamations of the Royal Society of London* 267:1679-1686.

Dean, J., D. Edds, D. Gillette, J. Howard, S. Sherraden, and J. Tiemann. 2002. Effects of lowhead dams on freshwater mussels in the Neosho River, Kansas. *Transactions of the Kansas Academy of Science* 105:323-240.

Dillon, R. T., Jr. 1984. Geographic distance, environmental difference, and divergence between isolated populations. *Systematic Zoology* 33:69-82.

Dillon, R. T., Jr. 1988. The influence of minor human disturbance on biochemical variation in a population of freshwater snails. *Biological Conservation* 43:137-144.

Felsenstein, J. 1976. The theoretical population genetics of variable selection and migration. *Annual Review of Genetics* 10:253-280.

Finlay, J. B., J. E. Buhay, and K. A. Crandall. 2006. Surface to subsurface freshwater connections: phylogeographic and habitat analyses of *Cambarus tenebrosus*, a facultative cave-dwelling crayfish. *Animal Conservation* 9:375-387.

Graf, W.L. 2006. Downstream hydrologic and geomorphic effects of large dams on American rivers. *Geomorphology* 79:336-360.

Hobbs, H.H., Jr. 1981. The crayfishes of Georgia. *Smithsonian Contributions to Zoology* 318:1-549.

Hudson, R.R. 2000. A new statistic for detecting genetic differentiation. *Genetics* 155:2011-2014.

Hudson, R. R., D.D. Boos, and N.L. Kaplan. 1992. A statistical test for detecting geographic subdivision. *Molecular Biology and Evolution* 9:138-151.

IFC Consulting. 2005. A summary of existing research on low-head dam removal projects, Technical Report, AASHTO.

Irwin, E., G. Turner, K. Mickett, and T. Piper. 2007. ACT Aquatic Gap and water quality monitory: Final Report. ADCNR.

Jager, H.I., J.A. Chandler, K.B. Lepla, and W.V. Winkle. 2001. A theoretical study of river fragmentation by dams and its effects on white sturgeon populations. *Environmental Biology of Fishes* 60:347-361.

Kelner, D.E. and B.E. Sietman. 2000. Relic populations of the Ebony Shell, *Fusconaia ebena* (Bivalvia: Uniondiae), in the Upper Mississippi River Drainage. *Journal of Freshwater Ecology* 15:371-378.

Lande, R. 1976. Natural selection and random genetic drift in phenotypic evolution. *Evolution* 10:314-334.

Lande, R. 1988. Genetics and demography in biological conservation. *Science* 241:1455-1460.

Lande, R. 1999. Extinction risks from anthropogenic, ecological, and genetic factors. Pages 1-22 *in* L.F. Landweber and A.P. Dobson, editors. *Genetics and the Extinction of Species*. Princeton University Press, Princeton.

Larinier, M. 2001. Environmental issues, dams and fish migration. FAO

technical paper. 419:45-89.

Lessard, J.L. and D.B. Hayes. 2003. Effects of elevated water temperature on

fish and macroinvertebrate communities below small dams. *River

Research and Applications* 19:721-732.

Lydeard, C. and R.L. Mayden. 1995. A diverse and endangered aquatic

ecosystem of the Southeast United States. *Conservation Biology* 9:800-

805.

Lynch, M. and T.J. Crease. 1990. The analysis of population survey data on

DNA   sequence variation. *Molecular Biology and Evolution* 7:377-394.

MacArthur, O.H. and E.O. Wilson. 1967. *The theory of island biogeography*.

Princeton University Press. Princeton, New Jersey.

McLaughlin, R.L., L. Porto, D.L.G. Noakes, J.R. Baylis, L.M. Carl, H.R. Dodd,

J.D.Goldstein, D.B. Hayes, and R.G. Randall. 2006. Effects of low-head

barriers on stream fishes: taxonomic affiliations and morphological

correlates of sensitive species. *Canadian Journal of Fisheries and

Aquatic Sciences* 63:766-779.

Miya, Y. and T. Hamano. 1988. The influence of a dam having no fish-way on

the distribution of decapod crustaceans in the Yukinoura River, Nagasaki,

Japan. *Nippon Suisan Gakkaishi* 54:429-435.

Morita, K. and S. Yamamoto. 2002. Effects of habitat fragmentation by

damming on the persistence of stream-dwelling charr populations.

Conservation Biology 16:1318-1323.

Nei, M. 1977. F-statistics and analysis of gene diversity in subdivided

populations. *Annals of Human Genetics* 41:225-233.

Nei, M.. 1987. *Molecular Evolutionary Genetics*. Columbia University Press,
New York.

Neraas, L.P. and P. Spruell. 2001. Fragmentation of riverine systems: the
genetic effects of dams on bull trout (*Salvelinus confluentus*) in the Clark
Fork River system. *Molecular Ecology* 10:1153-1164.

Olden, J.D., D.A. Jackson, and P.R. Peres-Neto. 2001. Spatial isolation and fish
communities in drainage lakes. *Oecologia* 127:572-585.

Poff, N.L., and D.D. Hart. 2002. How dams vary and why it matters for the
emerging science of dam removal. *BioScience* 52:659-668.

Posada, D., K.A. Crandall, A.R. Templeton. 2000. GeoDis: a program for the
cladistic nested anaylsis of the geographical distribution of genetic
haplotypes. *Molecular Ecology* 9:487-488.

Raymond, H.L. 1979. Effects of dams and impoundments on migrations of
juvenile Chinook salmon and steelhead from the Snake River, 1966 to
1975. *Transactions of the American Fisheries Society* 108:505-529.

Rozas, J., J.C. Sanchez-DelBarrio, X. Messeguer, and R. Rozas. 2003. DnaSP,
DNA polymorphism analysis by the coalescent and other methods.
Bioinformatics 19(18):2496-2497.

Santos, S. R. 2006. Patterns of genetic connectivity among anchialine habitats:
a casestudy of the endemic Hawaiian shrimp *Halocaridina rubra* on the
island of Hawaii. *Molecular Ecology* 15:2699-2718.

Sarich, V.M. and A.C. Wilson. 1973. Generation time and genomic evolution in

primates. *Science* 179:1144-1147.

Schuster, G.A., and C.A. Taylor. 2004. Report of the crayfishes of Alabama:
literature and museum database review, species list with abbreviated
annotations and proposed conservation statuses. Illinois Natural History
Survey, Center for Biodiversity Technical Report.

Templeton, A. R. 2005. Inference key for the nested haplotype tree analysis of
geographical distances. http://darwin.uvigo.es/software/geodis.html

Templeton, A.R., E. Boerwinkle, and C.F. Sing. 1987. A cladistic analysis of
phenotypic associations with haplotypes from restriction endonuclease
mapping. I. Basic theory and analysis of alcohol dehydrogenase activity in
Drosophala. Genetics 117:343-351.

Travnicheck, V.H., M.B. Bain, M.J. Maceina. 1995. Recovery of a warmwater
fish assemblage after the initiation of minimum-flow release downstream
from a hydroelectric dam. *Transactions of the American Fisheries Society*
124:836-844.

Walter, T.C., and D.J. Merritts. 2008. Natural streams and the legacy of water-
powered mills. *Science* 319:299-304.

Watters, G.T. 1996. Small dams as barriers to freshwater mussels (Molusca:
Pelecypoda: Unionidae) and their hosts. *Biological Conservation* 75:79-
85.

Yamamoto, S. K. Morita, I. Koizumi, and K. Maekawa. 2004. Genetic

differentiation of white-spotted charr (*Salvelinus leucomaenis*) populations

after habitat fragmentation: spatial-temporal changes in gene frequencies.

Conservation Genetics 5:529-53