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AN EVALUATION OF LARGE WOODY DEBRIS RESTORATIONS ON THE MANISTEE AND AU SABLE RIVERS, MICHIGAN

By

Matthew M Klungle

A THESIS

Submitted to
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ABSTRACT

AN EVALUATION OF LARGE WOODY DEBRIS RESTORATIONS ON THE MANISTEE AND AU SABLE RIVERS, MICHIGAN

By

Matthew M Klungle

The addition of large woody debris (LWD) offers a practical management technique for managers enhancing habitat in midwestern forest streams. Although it is well established that LWD provides critically important habitat for salmonids in conifer-dominated watersheds, little is known about the relationship between LWD and fish habitat in northern hardwood forests. I evaluated the effects of LWD restoration efforts on the Manistee and Au Sable rivers. I evaluated fish response at different spatial scales, whole site, individual banks, and individual rivers. There were no significant differences detected in catch per unit effort (CPUE) between treatment and reference reaches for any of the commonly occurring gamefish species at the whole site level. Brown trout and rainbow trout showed significant effects in CPUE between individual banks within treatment and reference reaches while rock bass and smallmouth bass showed no significant differences. Gamefish densities differed significantly between the individual rivers. The presence of LWD structures had no significant effect on the abundance of drifting aquatic invertebrates. Channel morphology and substrate in the vicinity of LWD responded with a trend of aggradation and decreased substrate size occurring at most sites.

This thesis is dedicated in memory of Ida May Klungle, an inspiration who instilled in me an appreciation of nature.

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INTRODUCTION

Large woody debris (LWD) is a natural component of streams and rivers and plays a complex role in hydrological, chemical, and biological processes (Gurnell et al. 1995). Fish use LWD for foraging sites and as protection against current, predators and competitors (Angermeiger and Karr 1984). Large woody debris provides a stable substrate for aquatic organisms such as bacteria, fungi, and invertebrates, all of which are involved in the decomposition of wood and represent major components of the trophic pathways in lotic systems (Angermeiger and Karr 1984; Lemly and Hilderbrand 2000). Consequently, LWD placement has become a common technique for stream restoration and for improving fish habitat to compensate for reductions in LWD due to various anthropogenic activities (Kauffman et al. 1997)

During the late 1800's and early 1900's, timber companies used the Manistee and Au Sable Rivers for their log drives. Historical accounts indicate that before the log drives, these rivers were so full of woody debris they were non-navigable (Miller 1963; Peterson 1972). To facilitate these drives, timber companies removed all natural deadfall trees that once provided habitat for many native fishes. Continued large-scale land clearance and development has also changed the landscape of the river corridors. Completion of Mio dam in 1916, Alcona dam in 1924 and Hodenpyl dam in 1925 also hindered natural recruitment as any LWD that travels through the reservoir and to the face of the dam is blocked from downstream passage or removed. Consequently, natural recruitment of LWD into many northern Michigan rivers has been reduced or even eliminated for the last 100 years.

Stream habitat work by public management agencies in the U.S. was pioneered by the Michigan Conservation Department (predecessor to the Department of Natural Resources). Brush piles and cover structures were added to Michigan trout creeks in 1927; prior to this, almost all habitat manipulations were done privately by anglers (Hubbs et al. 1932). These manipulations were initiated under the assumption that the amount and quality of physical habitat in the stream channel is limiting.

Managers continue to recognize that habitat plays a critical role in the dynamics of fluvial fish stocks. Habitat restoration continues to be resource managers' preferred solution to increasing the number or size of fish in a population as artificial propagation tends to be cost prohibitive and protective regulations do not directly produce fish. The enhancement and restoration of stream habitat for game fishes, particularly salmonids, through the instream placement of structures and LWD has increased dramatically in the Pacific Northwest (Roni and Quinn 2001). In the last 20 years, many studies have emphasized the critical role that LWD plays in creating and maintaining fish habitat in streams. LWD is widely recognized as a critical component of natural streams and rivers and its role is ecologically complex.

Most LWD evaluations have considered small and medium-sized streams but rarely larger rivers. Since the early 1980s, the interest in habitat restoration projects has focused on the Pacific Northwest in an effort to mitigate degradation and loss of habitat due to human disturbance and to stop or reverse significant salmonid population declines (National Research Council 1996). Steeper, high-energy small streams common in the Pacific Coast rain-forest mountains are more vulnerable to logging practices and more sensitive to manipulations than larger lower gradient ones. Large woody debris complexes create essential pools and gravel deposits in steep, cool-climate streams of the Pacific Coast. Evaluations of these restorations have produced highly variable results. There have been numerous reports indicating increased fish abundance in response to restoration efforts (e.g., Angermeier and Karr 1984; Fausch and Nortcote 1992; Gowan and Fausch 1996; Flebbe 1999; Rosenfeld et al. 2000; Roni and Quinn 2001a) along with a number of reports with no significant fish response and even decreases in fish abundance in response to restoration efforts (e.g., Angermeier and Karr 1984; Thevenet and Statzner 1999; Roni and Quinn 2001a).

In lower gradient streams and rivers of the Great Lakes region, LWD can potentially slow flow. Due to the concern that scarce spawning gravels may become buried in sand and silt, and water temperatures may become too warm for trout in the summer there has been little work on lower gradient midwestern streams and rivers. This concern seems to have superceded many of the potential benefits of LWD and consequently, the effects of LWD on larger rivers in the midwest have yet to be investigated. Based on the underlying ethos that LWD is a critical component of a natural river system and that restoration is a viable option, restoration efforts began in the fall of 1998 by the U.S. Forest Service in conjunction with the local conservation districts (Conservation

Resource Alliance and Huron Pines Resource Conservation District) and the Michigan Department of Natural Resources. Whole tree LWD structures were placed with a helicopter in the Mio reach of the Au Sable river as a demonstration project to determine the feasibility and economics. They continued this effort with LWD placements in the Alcona reach of the Au Sable river and the Hodenpyl reach of the Manistee river during the fall of 2000 with additional placements in the Mio and Hodenpyl reaches in the fall of 2002.

The purpose of this study was to evaluate the effectiveness of these large woody debris restoration efforts implemented in the Manistee and Au Sable rivers. These data will provide a base line for possible future studies on established LWD, and will aid in the design of future evaluations of LWD implementation projects.

The objectives of this evaluation were to determine whether the addition of LWD structures produced a significant change in physical habitat, gamefish abundance, and invertebrate abundance in two northern Michigan rivers. Specifically we tested the null hypothesis that paired treatment and reference reaches would not differ in (1) gamefish abundances, (2) survival and retention of gamefish, and (3) invertebrate drift. In addition we documented channel morphology changes in proximity to LWD being added.

METHODS

Study Reaches

Three study sites were selected in the Manistee River from Hodenpyl Dam to Red Bridge, Manistee County, four sites on the Au Sable River from Mio Dam to Cummins Flat pullout, Oscoda County, and three sites on the Au Sable River from Alcona Dam to Thompson's Landing, Alcona County (Figure 1). Hodenpyl and Mio dams are the first barriers encountered heading down stream on these rivers. All of these dams are operated on a run-of-river flow scheme. Gradient in these reaches is somewhat higher, with water quality and biological communities typical of large, cold-cool rivers.

Fish Response to LWD structures

A split-plot design with paired treatment and reference reaches was used to determine the response of gamefish to LWD treatments. Treatment and reference sites 100 m long were selected in each reach and broken into three sections (left bank, middle, right bank; Figure 2). Treatment is defined as the artificial placement of LWD within the active stream channel. This design involves comparison between paired treatment and reference reaches at the 10 study sites. Sites were chosen by proximity to tree harvest sites to minimize turnaround time and maximize the number of trees placed. At all sites, paired treatment and reference sites were also chosen so that pairs would be as similar to each other as possible and so that treatment sites would represent a range of amounts of LWD (Table 1). Whenever possible, I paired treatment and reference sites adjacent to one another to maintain site similarity. When this was not possible, treatment and reference sites were within 200 m of one another. I also tried to alternate the upstream site to avoid confounding from natural down stream migrations (Table 1).

All field sampling occurred during the summers of 2002 and 2003. A Smith-Root Cataraft electrofishing boat was used for all fish sampling efforts on the Mio and Hodenpyl reaches. The electrofishing boat was set at pulsed DC (40% cycle duty) on low range (50 – 500) volts at 4 – 6 amps. Fish were sampled at the three sites along the Hodenpyl reach, once per month (May to August), in 2002 and 2003. The four sites in the Mio reach were sampled once per month (June and July), in 2003. Sites on the Mio reach were shocked at night, due to much greater recreational use during the day than the Hodenpyl or Alcona reaches .

Gamefish populations were sampled using single pass electrofishing. All captured gamefish were measured for total length to the nearest 1 mm. In order to assess site fidelity trout under 150 mm were marked with a bank specific fin clip, and those over 150 mm were marked with Visual Implant alpha-numeric tags. Other gamefish were marked with Floy T-bar tags.

Fish communities on the Alcona reach of the Au Sable river were sampled once per month (June to August) in 2002 and 2003 by visual observations, as electrofishing was not possible due to the remoteness and inaccessibility of the reach. Gamefish populations were sampled using single pass snorkel surveys. Swimmers with a mask and snorkel sampled individual banks within the paired treatment and reference sites noting species encountered in each sampling pass.

Channel response

Localized changes in river channel morphology and substrate were monitored by fixed longitudinal survey transects located around the LWD structures. Benchmarks were established by driving an aluminum nail into the base of a healthy tree with an assumed elevation of 30.5 m. One meter lengths of 1.6 cm rebar were used to delineate endpoints of the 100 m long study sites. Channel morphology was measured at all 10 treatment sites. Standard longitudinal profile surveying techniques were used in 2002 and in 2003 (Harrelson et al. 1994). Morphology was measured every 0.6 m along the longitudinal profile using a stadia rod and surveyor's level to map channel elevation in proximity to trees being added. Substrate was evaluated using a pebble count code classification method (Harrelson et al. 1994). One hundred random streambed particles were classified along each transect.

Invertebrate response

Drifting aquatic invertebrates were collected at monthly intervals (Table 3), as river conditions allowed, from May through August during early evening hours to determine behavioral drift response to LWD placements. Drift samples were collected for 1 hour in 2001 and 15 minutes in 2002 after sunset, above and below LWD structures. Invertebrates were captured with 200 µm mesh drift nets 14.5-cm wide placed at an average depth of 65 cm, and then strained through a 500 µm sieve. Samples were stored in appropriately sized nalgene® bottles or in zip closure freezer bags (doubled bagged) and preserved in 90% ethyl alcohol. I sampled a minimum of six sites per month in this evaluation (Table 3).

To identify other potential limiting factors, water temperature was recorded every 1 hour with Onset Stowaway temperature loggers at each study site in 2002 and 2003.

Statistical analyses

I used data collected from the Hodenpyl reach in 2002 and 2003 and data from the Mio reach in 2003 to test differences in mean catch per unit effort (CPUE) between treatment and reference reaches. All data sets were analyzed using SAS version 8.2. Data analysis focused on the four most common gamefish species encountered, brown trout *Salmo trutta*, rainbow trout *Oncorhynchus mykiss*, rock bass *Ambliopites rupestris*, and smallmouth bass *Micropterus dolomieu*. Differences in gamefish CPUE between treatment and reference reaches were compared using a generalized linear model ANOVA (GENMOD) with user definable distributions to meet necessary assumptions. An alpha level of 0.05 was used to determine statistical significance for each analysis. A negative binomial distribution and a Poisson distribution were applied to the fish data to meet the basic assumptions of GENMOD and to account for a large number of zero's in the data sets. Akaike's information criteria (AIC) was used to determine the most appropriate model for each data set.

Fish can respond to LWD additions at different spatial scales. Thus, I evaluated their response at the scale of whole site (i.e., entire 100 m treatment or reference sites) as well as the scale of individual banks (i.e., river banks within treatment sections that did not receive LWD treatments were considered reference banks). I also evaluated whether the response to LWD varied between the Mio reach of the Au Sable River and the Hodenpyl reach of the Manistee River.

Channel analyses

I used data collected in 2002 and 2003 to test differences in substrate classifications and longitudinal channel transects in treatment reaches. Channel morphology and substrate change between years were determined using a paired *t*-test.

Invertebrate analyses

I used data collected in 2002 and 2003 to test differences in behavioral drift densities above and below LWD structures. Data analysis focused on the following taxonomic groups; Crustacea, Ephemeroptera, Tricoptera, and Diptera, which I broke down to family Chironomidea and other Diptera. Differences in behavioral drift above and below LWD structures were compared using a generalized linear model ANOVA with definable distributions to meet necessary assumptions. An alpha level of 0.05 was used to determine statistical significance for each analysis. A negative binomial distribution was applied to the overall and individual taxonomic group invertebrate data to meet the basic assumptions of GENMOD to account for a large number of zero's in the individual taxon.

RESULTS

Fish Response

Brown and rainbow trout consistently had the highest mean CPUE of gamefish encountered in the Mio and Hodenpyl reaches. This is not surprising as these reaches are heavily stocked yearly by the MDNR. Length frequency distributions of these species (Figure 3) show a lack of fish less than 180mm, indicating few if any young of year or 1+ fish are produced in these river reaches.

There were no significant differences detected in catch per unit effort (CPUE) between treatment and reference reaches for any of the commonly occurring gamefish species as indicated by the GENMOD analysis (Table 5). Point estimates of mean CPUE for brown trout, rainbow trout, and rock bass were higher in treatment reaches (27.7%, 10.4%, and 40.0% respectively) than in reference reaches but were highly variable resulting in no statistical significance between the treatment and reference reaches (Figure 4). Smallmouth bass showed a slightly higher mean CPUE point estimate for the reference reach (4.2%), but the difference was not significant. CPUE of other species was too low to permit statistical analysis.

Brown trout and rainbow trout showed significant difference in CPUE between individual banks within treatment and reference reaches (river banks within treatment sections that did not receive LWD treatments were considered reference banks) while rock bass and smallmouth bass showed no significant differences (Table 9). Point estimates of mean CPUE were consistently higher in the treatment reaches (30.0% brown trout, 41.9% rainbow trout, 58.5% rock bass, and 16.5% smallmouth bass) than reference reaches (Figure 6).

Mean CPUE for the different species differed between the Manistee and Au Sable rivers (Table 6). Overall mean CPUE was consistently higher (ave. 80%) in the Mio reach for each game fish species (Figure 5). In the Mio reach, gamefish mean CPUE was consistently higher in treatment sections with the exception of smallmouth bass (Figure 5). In the Hodenpyl reach, mean CPUE was higher in the treatment sections for brown trout and smallmouth bass and higher in the reference sections for rainbow trout and rock bass (Figure 5). Brown trout, rainbow trout, and smallmouth bass showed highly significant differences between the rivers, with rock bass only marginally significant (Table 6). Within the river reaches, rainbow trout were the only species to have a significant interaction of river*treatment (Table 7). Thus rainbow trout responded differently to the LWD additions between rivers.

In the Alcona reach, visual observation was used to assess fish response to LWD additions. Within treatment sites, the following species were observed walleye (*Stizostedion vitreum*), smallmouth bass, bluegill sunfish (*Lepomis macrochirus*), common shiner (*Luxilus cornutus*), logperch (*Percina caprodes*), blackside darter (*Percina maculata*), silver redhorse (*Moxostoma anisurum*), crappie (*Pomoxis nigromaculatus*), northern hog sucker (*Hypentelium nigricans*), rock bass, white sucker (*Catostomus commersonii*), and johnny darter (*Etheostoma nigrum*). Within the treatment reaches, fish were primarily observed in banks receiving treatments, often with no fish observed in non-treatment banks. Species observed in reference sites were smallmouth bass, logperch (*Percina caprodes*), blackside darter (*Percina maculata*), and bluegill sunfish (*Lepomis macrochirus*). Fish observed in treatment banks were often found in high densities compared to non-treatment banks where often single individuals were found. Redhorse suckers and walleye were occasional observed in the deepest areas of the river. All fish moved substantially (possibly due to snorkeler presence) making it very difficult to count. Overall counts were generally very low, thus precluding statistical analysis

The results of fish tagging were consistent with a low sampling efficiency, as few tagged fish were recaptured. Fish recapture rates for the 2002 Mio reach sampling were 4.9% in the treatment site and 11.1% in the reference site (Table 8). In the Hodenpyl reach, fish recapture rates were 2.1% in the treatment site and 0% in the reference site (Table 8) from combined 2002 and 2003 sampling. All of these recapture rates were considered too low for statistical analysis.

Invertebrate response

No effects of LWD structures on overall mean invertebrate behavioral drift density were detected (Table 10). The presence of LWD structures had no significant effect on invertebrate behavioral drift density or on the density of any particular taxon captured in the drift nets. There were no observed differences in mean density of the taxon Crustacea, Ephemeroptera, Tricoptera, and Diptera between above and below LWD structures (Table 10). CPUE of Lepidoptera and Plecoptera was too low to permit statistical analysis. Invertebrate drift numbers were dominated by Diptera (84.1%, 77.6% Chironomidae and 6.6% other Diptera) followed by Crustacea (6.8%), Ephemeroptera (5.1%), Tricoptera (3.3%). Coleoptera (0.33%), Plecoptera (0.24%), and Lepidoptera (0.05%) combined to make up less than 1% of the drift net composition (Figure 7).

Channel response

Channel morphology changed in all of the Mio sites (Table 11) and substrate classification differed statistically for three of the four sites (Table 12). In the Hodenpyl reach, sites one and two showed significant channel changes (Table 11) and sites two and three showed significant substrate differences

(Table 12). In the Alcona reach, only site three showed significant morphology differences (Table 10) while sites one and three showed significant substrate differences (Table 11). It is important to note here that paired t-tests are sensitive statistical analyses, and it may be more insightful to compare yearly morphology figures. Mio site one showed channel aggradation (Figure 8) of finer sediments (Figure 9). Mio site two showed consistent channel morphology (Figure 8) and stable substrates (Figure 9) between the sample years 2002 and 2003. This is expected because this site only received one treatment tree in 1998, in the bank surveyed, and no treatment in 2002. Mio site three showed substantial aggradation (Figure 8) while substrate seemed to stay similar from year to year (Figure 9). Mio site four showed some channel incision with a shift towards finer substrates. The Hodenpyl channel morphology showed minimal change from year to year (Figure 10), the only changes appear to be some substrate redistribution at site 1 and a shift to finer substrate at site 2 (Figure 11). The three Alcona sites also appear to stay relatively unchanged (Figure 12) with a slight redistribution of substrates that appear to shift to finer substrates (Figure 13).

Temperature

Daily mean water temperatures in 2002 and 2003 for the Mio reach reached critically high temperatures in mid June and stay there until mid August. There was a dip in mid June in 2002, which was probably due to a large rain event. In the Hodenpyl reach, water temperatures for 2002 went into the critical zone in mid May and stayed there at least until August. In 2003, the water temperature did not reach the critical zone until early July and stayed there until at least

August. In the Alcona reach, water temperatures were in the critical zone the entire temperature recording duration.

Discussion

The benefits of LWD as a habitat restoration technique have been well established and well documented; LWD provides current breaks, cover from predators, and forage sites for fish (Angermeiger and Karr 1984; Gowan and Fausch 1996; Flebbe 1999; Fausch and Nortcote 1992; Lemly and Hilderbrand 2000; Rosenfeld et al. 2000; Roni and Quinn 2001a). Most recent studies on habitat restoration are from the western United States and Canada and have reported significant physical responses to restoration; but, fish response to habitat restoration, specifically LWD placements, and the mechanisms responsible have been less well researched and produced more variable results. Also, these recent studies are often on debris dams spanning the stream channel; few evaluations of gamefish in streams where LWD complexes (60+ trees fully in the stream channel) only encompass 30% of the stream channel are published. Because the actual response of fish to LWD additions is less well understood and studies from the Midwest are lacking, I conducted this study to determine response and help fill in gaps in understanding of LWD restorations on larger midwestern streams.

LWD structures were placed in the Manistee and Au Sable rivers to improve habitat for valued gamefishes and help restore the natural riverine ecosystem. These LWD structures were installed under the premise that physical habitat may be a limiting factor due to historic land use practices. With the apparent direct influence of LWD on the quantity and quality of habitat available to fish, it was expected that the LWD additions would increase gamefish density in these reaches.

At the whole site level, treatment sites generally had higher densities, but the highly variable data yielded no significant results. This was also true when the response was evaluated for individual rivers. However, a test of fish density at the individual bank scale showed a significant increase in mean CPUE for brown trout and rainbow trout in treatment banks, indicating that habitat may be a limiting factor for these species in these reaches. The impact of LWD structures in these reaches, however, appears to produce a very localized effect.

Because natural LWD was lacking in much of these river reaches, I expected a larger response from game fishes. Angermeier and Karr (1984), Gowan and Fausch (1996), and Roni and Quinn (2001a) all found higher fish densities in areas where LWD was present or added. There are several potential reasons why a greater response was not observed. First, the Hodenpyl reach has holes >6 m deep (Tonello 2003), which combined with the high summer water temperatures, may prove to be preferred refugia over the shallower, warmer areas with LWD structures. These deep holes may concentrate fish and explain the lower mean CPUE for game fish in this reach, as the holes are too deep to sample with electrofishing gear. Although sampling efficiency was low based on mark-recapture analysis, the relative comparison between treatment and reference sites is still valid. Another potential problem is that hatchery trout are known to have difficulties efficiently adapting to natural habitats when stocked (Bettinger and Bettoli 2002). From the trout length frequency charts (Figure 3) it is obvious that hatchery fish dominate these populations, with little yearly hold over. This limited hold over may be due to the poor adaptation to natural environments, the high summer water temperatures (Figure 14) or a combination of these factors.

Non-salmonid response was puzzling given the cool summer water temperature regime and the physical benefits provided by the LWD structures. Studies have indicated that warmwater species are more abundant in the presence of LWD than its absence (Angermeier and Karr 1984; DuFour 1989). I expected a response from species like smallmouth bass and rock bass. Although these were some of the most abundant gamefish species in these reaches, there was no response to LWD structures at any scale.

Large woody debris also provides substrate for aquatic organisms such as bacteria, fungi, and invertebrates (Bilby and Likens 1980; Lemly and Hilderbrand 2000), all major components of riverine ecosystems. By increasing substrate for invertebrates colonize and food for invertebrates by increasing bacteria, fungi, and detritus inputs, LWD complexes can increase invertebrate production (Angermeier and Karr 1984; Lemly and Hilderbrand 2000). Thus, I expected to see an increase in invertebrate drift density below LWD structures. This expectation was not supported by the data, however. There are several potential reasons for this observation. First, drift may not be representative of density nor production in these reaches. Further, only some taxa drift. Thus, the density and production of inverts as a whole could vary without showing substantial changes in drift. Also, variability in the data may obscure any real patterns. Finally, the coarser substrates of these river reaches may provide invertebrates with refugia from currents and predators keeping invertebrate densities naturally high in reference reaches.

In the six sites in which substrate differed between 2002 and 2003, substrate size decreased (Figures 9,11, and 13). This suggests that the LWD structures are collecting finer sediments. Overall though, the substrate of these river reaches consists largely of cobble and gravel, which are relatively resistant to redistribution. This may be due to past dam management (i.e., daily peaking flows until the early 80's) that may have armored the substratum making it difficult for any scouring effects to be observed. So, it is not surprising that observed morphometry changes are generally aggradations and substrate changes tended towards finer substrates being deposited in the current breaks provided by the LWD structures. It is important to note that in Mio site 3 the survey data suggest channel aggregation of three to four feet. It is possible this could be due to sampling error, a shift in one of the fixed points the readings are based on, or extensive point bar development.

The most striking outcome of this study is the summer water temperatures in these river reaches. For brown trout, the critical thermal limit for growth and feeding is 19° to 20° C and the critical thermal lethal limit is approximately 25° C (Elliot 1994). As shown by Figure 14, mean daily water temperatures in these river reaches is consistently above critical thermal limits during the summer. The consistently high water temperature regimes, minimal yearly holdover, and lack of young of year suggest that high summer temperatures may be the primary factor limiting trout populations in these reaches. Consequently, a put, grow, and take fish stocking practice may be the only way to maintain a trout fishery in these river reaches unless water temperature can also be managed.

APPENDIX

Tables and Figures

Table 1. Study site details within each river reach including sampling design, fish sampling method for each river reach, the year of LWD placement, number of placed trees per treatment site, number of natural trees per treatment site, and location of treatment site in relation to reference site with A representing upstream and B representing downstream.

Reach	Hodenpyl			Mio				Alcona		
Design	After placement			After placement				After placement		
Sampling Method	Electrofishing[1]			Electrofishing[2]				Snorkel		
Year of sampling	2002 and 2003			2003				2002 and 2003		
Year of placement	2001 and 2002			1998 and 2002				2001		
Treatment sites	1	2	3	1	2	3	4	1	2	3
No. placed trees	12	35	35	21	62	28	31	4	6	3
No. natural trees	6	0	8	0	0	0	0	0	0	0
Treatment location	A	B	B	A	A	B	B	A	A	B

[1]Electrofishing conducted during the day
[2]Electrofishing conducted at night

Table 2. Size classes and codes used to classify substrate particles.

Size Code	Size Class (mm)	Particle
0		Trash
1		Organic
2	0.00024 – 0.004	Clay
3	0.04 – 0.062	Silt
4	0.062 – 2	Sand
5	2 - 4	Very fine gravel
6	4 - 8	Fine gravel
7	8 - 16	Medium gravel
8	16 – 32	Coarse Gravel
9	32 – 64	Very coarse gravel
10	64 – 128	Small Cobble
11	128 – 256	Large Cobble
12	256 – 512	Small Boulder
13	>512	Medium Boulder

Table 3. Monthly summer invertebrate sampling schedule for each river reach and site within the reaches sampled.

Reach	Hodenpyl	Mio	Alcona
May	1,2	1,2	1,2
June	2,3	3,4	2,3
July	1,3	1,3	1,3
August	1,2,3	2,4	1,2,3

Table 4. Best-fit distribution as determined by AIC patterns used for brown trout (BNT), rainbow trout (RBT), rock bass (RB), and smallmouth bass (SMB) at each scale of GENMOD analysis.

Species	Overall	Individual Rivers	Individual Banks
BNT	Negative Binomial	Negative Binomial	Negative Binomial
RBT	Negative Binomial	Negative Binomial	Poisson
RB	Poisson	Poisson	Negative Binomial
SMB	Poisson	Poisson	Poisson

Table 5. *P*-values from a GENMOD analysis of variance modeling the effects of LWD on overall mean CPUE for brown trout (BNT), rainbow trout (RBT), rock bass (RB), and smallmouth bass (SMB). Mean and SE of CPUE reported are back transformed least square means from GENMOD.

		Treatment		Reference			
Spp.	df	Mean	SE	Mean	SE	X^2	*P*-values
BNT	55	7.94	1.17	5.65	1.17	2.49	0.115
RBT	55	2.11	1.23	1.89	1.26	0.12	0.731
RB	55	0.19	1.58	0.11	1.79	0.49	0.484
SMB	55	0.84	1.24	0.87	1.234	0.02	0.886

Table 6. *P*-values from a GENMOD analysis of variance modeling the effects of LWD on individual river mean CPUE for brown trout (BNT), rainbow trout (RBT), rock bass (RB), and smallmouth bass (SMB). Mean and SE of CPUE reported are back transformed least square means from GENMOD.

		Hodenpyl		Mio			
Spp.	df	Mean	SE	Mean	SE	X^2	*P*-values
BNT	54	3.61	1.14	12.43	1.20	14.37	<0.001
RBT	54	0.64	1.25	5.65	1.21	44.38	<0.001
RB	54	0.07	1.84	0.25	1.75	4.42	0.036
SMB	54	0.33	1.31	2.18	1.18	14.67	<0.001

Table 7. *P*-values from a GENMOD analysis of variance modeling the effects of the river treatment interaction for CPUE of brown trout (BNT), rainbow trout (RBT), rock bass (RB), and smallmouth bass (SMB). Mean and SE of CPUE reported are back transformed least square means from GENMOD.

		Hodenpyl				Mio					
		Treatment		Reference		Treatment		Reference			
Spp.	df	Mean	SE	Mean	SE	Mean	SE	Mean	SE	X^2	*P*-values
BNT	54	4.33	1.20	3.00	1.22	14.38	1.30	10.75	1.30	0.03	0.866
RBT	54	0.48	1.41	0.86	1.31	8.25	1.29	3.87	1.33	5.32	0.021
RB	54	0.05	2.72	0.10	2.03	0.50	1.65	0.13	2.72	1.57	0.21
SMB	54	0.38	1.42	0.29	1.50	1.99	1.26	2.38	1.28	0.52	0.471

Table 8. Numbers of game fish tagged and recaptured during sampling events in the Mio and Hodenpyl River reaches. Mio results from fish sampled in 2003 and Hodenpyl results from fish sampled in 2002 and 2003.

	Mio		Hodenpyl	
	Treatment	Reference	Treatment	Reference
Tagged	163	108	96	85
Recaptured	8	12	2	0

Table 9. *P*-values from a GENMOD analysis of variance modeling the effects of LWD on individual bank mean CPUE brown trout (BNT), rainbow trout (RBT), rock bass (RB), and smallmouth bass (SMB). Mean and SE of CPUE reported are back transformed least square means from GENMOD.

		Treatment		Reference			
Spp.	df	Mean	SE	Mean	SE	X^2	*P*-values
BNT	112	4.09	1.16	2.74	1.14	4.09	0.043
RBT	114	1.29	1.15	0.75	1.16	8.63	0.003
RB	114	0.11	1.63	0.05	1.82	1.2	0.273
SMB	114	0.46	1.26	0.40	1.22	0.21	0.646

Table 10. *P*-values from a GENMOD analysis of variance modeling the effects of LWD structures on invertebrate behavioral drift density for overall, Crustecea, Ephemeroptera, Tricoptera, Chironomidae, and other Diptera.

Family	df	Above mean	SE	Below mean	SE	X^2	*P*-values
Overall	40	46.34	1.24	53.67	1.25	0.22	0.638
Crustacea	40	4.99	1.60	1.81	1.63	2.23	0.135
Ephemeroptera	40	2.33	1.41	2.81	1.41	0.15	0.702
Tricoptera	40	1.71	1.36	1.57	1.36	0.04	0.841
Chironomidae	40	34.38	1.26	43.19	1.26	0.48	0.488
other Diptera	40	2.43	1.40	4.14	1.38	1.28	0.257

Table 11. *P*-values from a paired *t*-test for longitudinal channel morphology in each study site for sampling years 2002 and 2003 with mean change in channel depth and SE. The letter represents each reach (Mio, Hodenpyl, and Alcona) and the number designating which site within the reach.

Site	df	mean	SE	*t*-value	*p*-value
M1	105	-1.00	0.0546	-18.38	<.0001
M2	99	-0.13	0.0274	-4.70	<.0001
M3	105	0.11	0.0340	3.31	0.0013
M4	106	-0.09	0.0119	-7.70	<.0001
H1	105	0.38	0.0495	7.64	<.0001
H2	117	-0.17	0.0413	-4.04	<.0001
H3	108	-0.05	0.0850	-0.62	0.5370
A1	105	0.01	0.0297	0.42	0.6764
A2	92	0.05	0.0552	0.87	0.3855
A3	72	-0.42	0.0452	-9.24	<.0001

Table 12. *P*-values from a paired *t*-test for substrate classification in each study site for sampling years 2002 and 2003 with mean change in classification and SE. The letter represents each reach (Mio, Hodenpyl, and Alcona) and the number designating which site within the reach. No statistical analysis for Hodenpyl site 3 due to lost 2002 data.

Site	df	mean	SE	*t*-value	*p*-value
M1	99	0.76	0.3699	2.05	0.0425
M2	99	-0.33	0.2349	-1.41	0.1631
M3	99	0.58	0.2610	2.22	0.0285
M4	99	1.62	0.2485	6.52	<.0001
H1	99	-0.78	0.4282	-1.82	0.0715
H2	99	0.71	0.2687	2.64	0.0096
H3					
A1	99	1.94	0.3187	6.09	<.0001
A2	99	-0.03	0.3647	-0.08	0.9346
A3	99	1.88	0.3295	5.71	<.0001

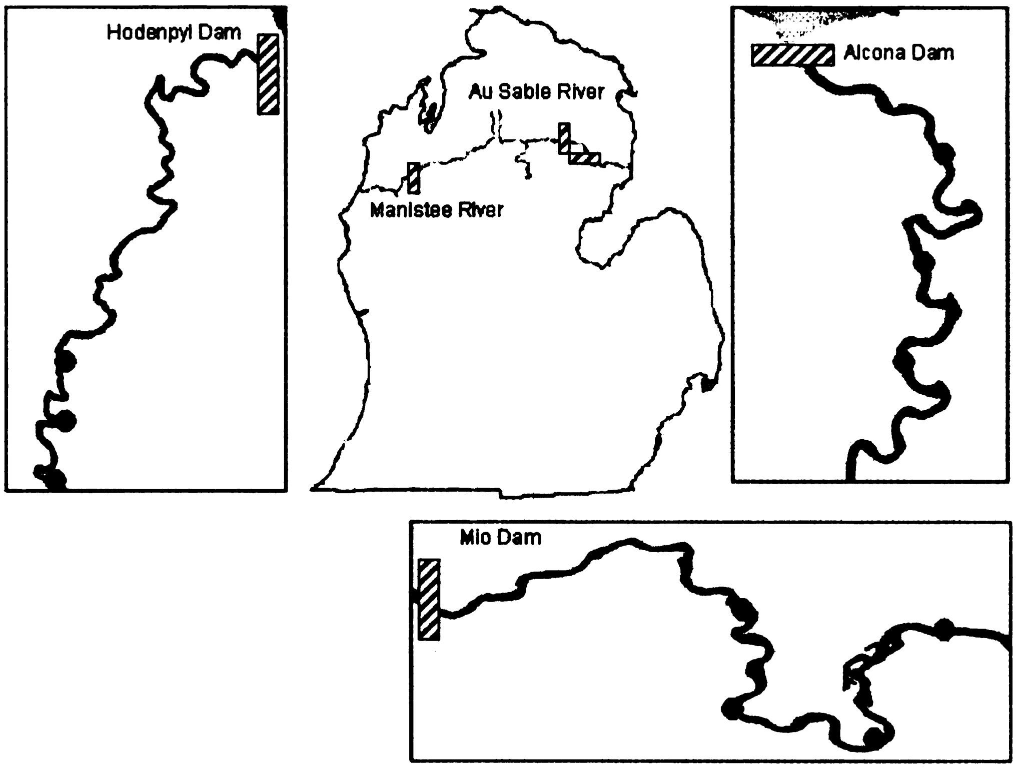

Figure 1. River reach and study site locations on the Manistee and Au Sable rivers.

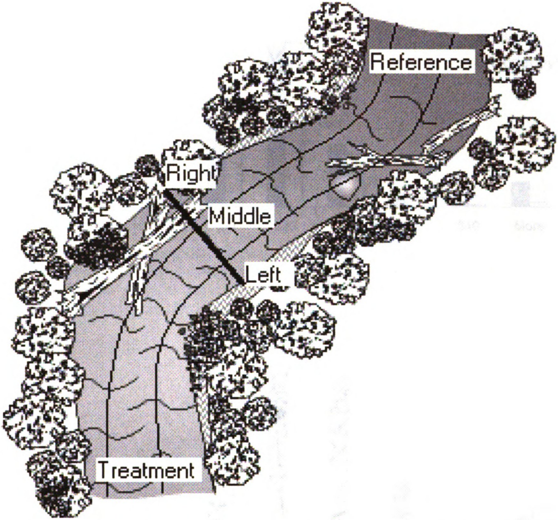

Figure 2. Split-plot design with paired treatment and reference reaches.

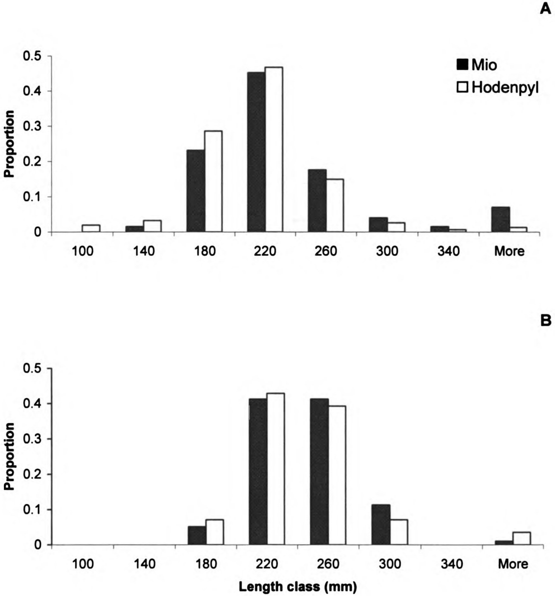

Figure 3. Length class (in millimeters) frequency distributions for brown trout (A) and rainbow trout (B) in the Hodenpyl reach and Mio reach.

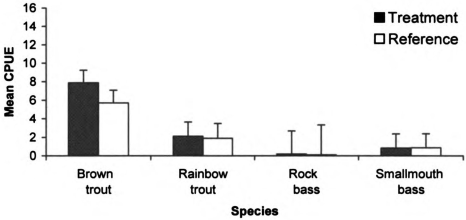

Figure 4. Overall estimates of mean (± 2SE) back-transformed catch per unit effort of the four most commonly encountered gamefish in the treatment and reference sections of the combined Hodenpyl and Mio reaches of the Manistee and Au Sable rivers as determined by electrofishing.

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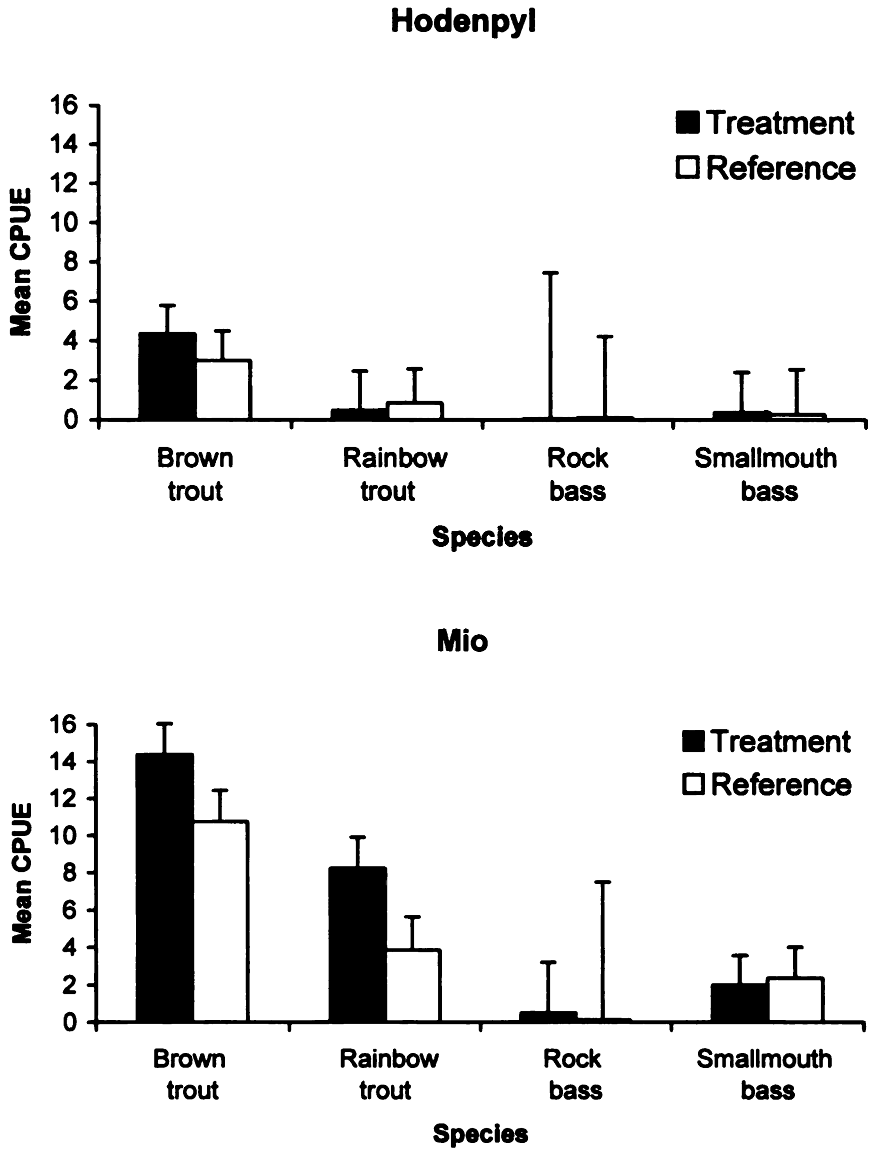

Figure 5. Overall estimates of mean (± 2SE) back-transformed catch per unit effort of the four most commonly encountered gamefish in the treatment and reference sections on each the Hodenpyl reach of the Manistee river and the Mio reach of the Au Sable river as determined by electrofishing.

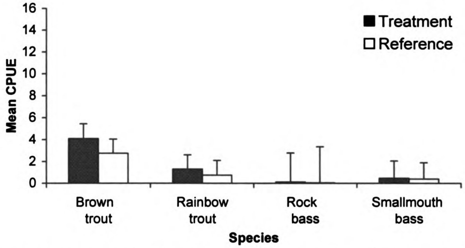

Figure 6. Individual bank estimates of mean (± 2SE) back-transformed catch per unit effort of the four most commonly encountered gamefish in the treatment and reference sections of the combined Hodenpyl and Mio reaches of the Manistee and Au Sable rivers as determined by electrofishing.

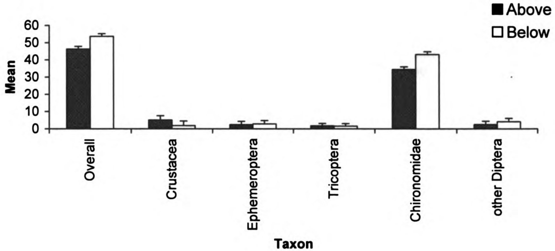

Figure 7. Estimates of mean (± 2SE) back-transformed catch per unit effort of the invertebrate taxa occurring above and below LWD structures of the combined Hodenpyl and Mio reaches of the Manistee and Au Sable rivers as determined in monthly behavioral drift samples.

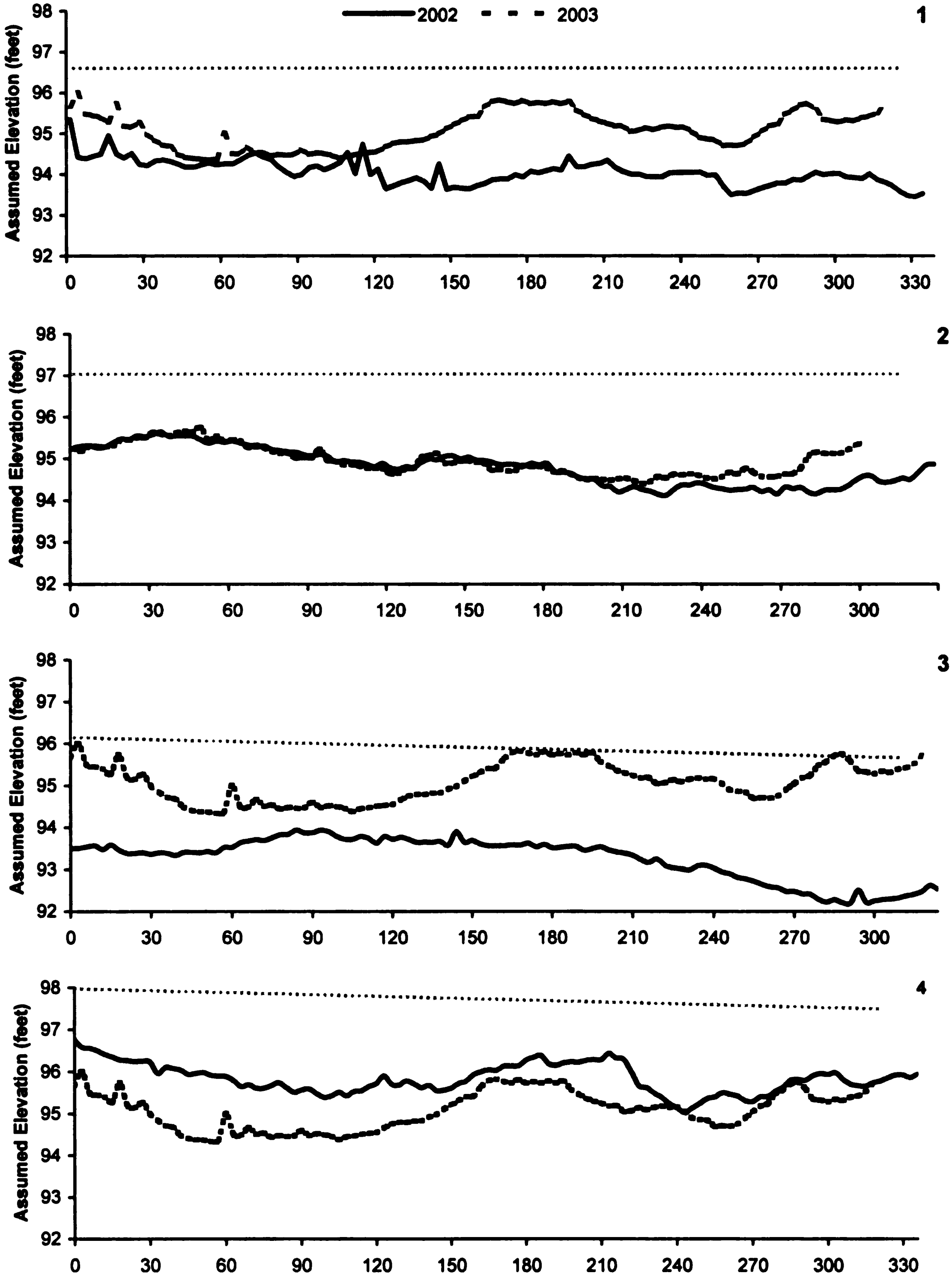

Figure 8. Survey transects from each of the Mio study sites, showing the longitudinal river channel for 2002 and 2003. Water surface (2003) is represented by the dotted top line. LWD structures were present along the entire transect in 2003.

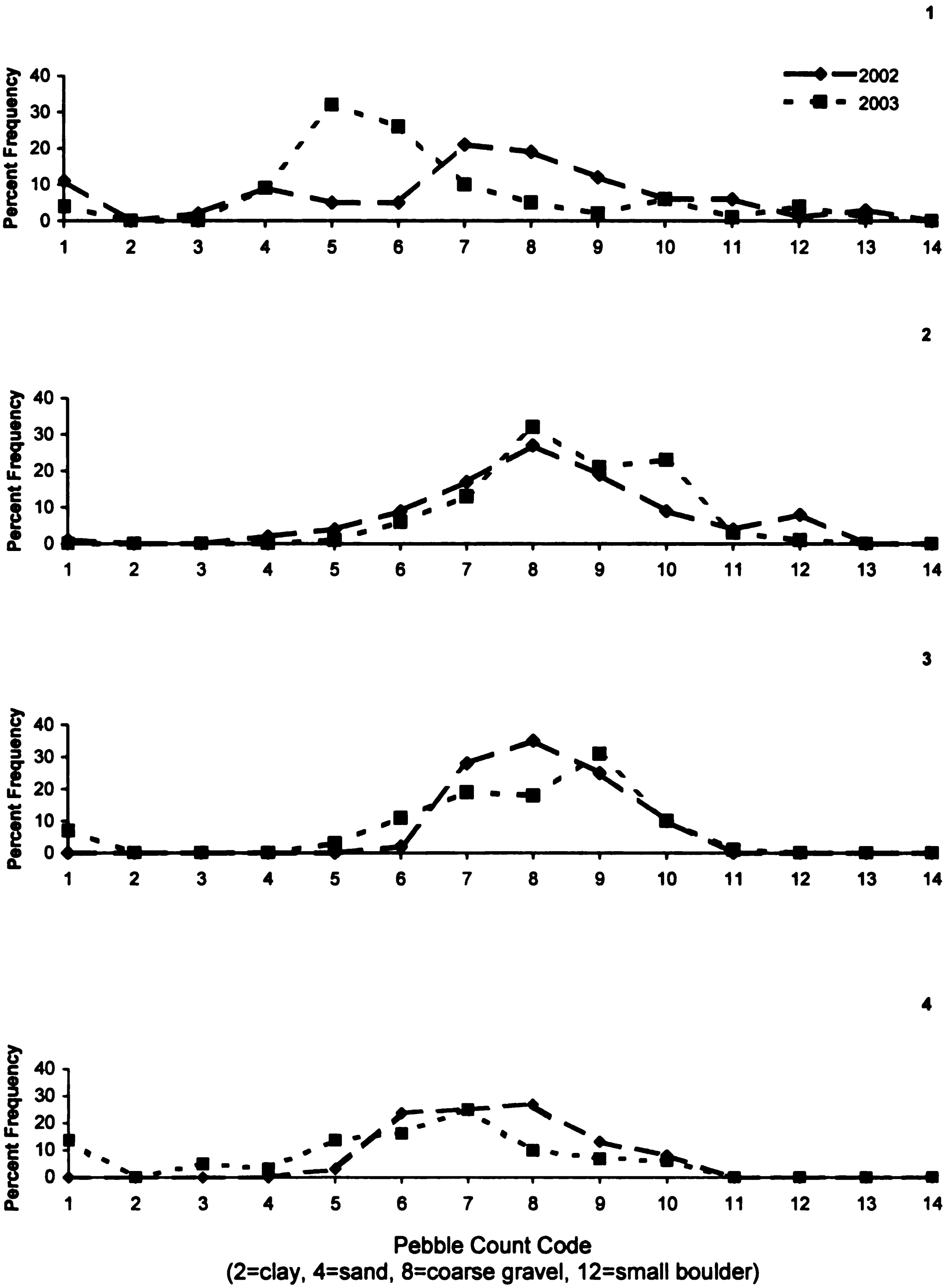

Figure 9. Substrate size percent frequency distribution for each Mio site in 2002 and 2003.

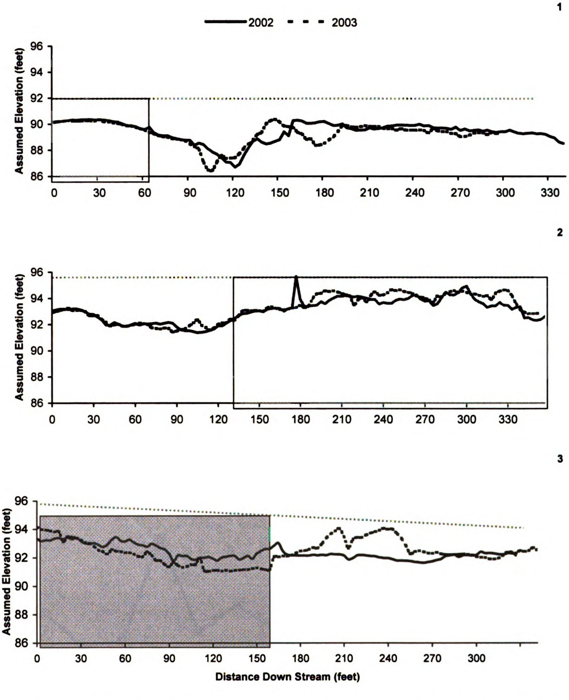

Figure 10. Survey transects from each of the Hodenpyl study sites, showing the longitudinal river channel profile for 2002 and 2003. Water surface (2003) is represented by the dotted top line with the shaded area representing location of LWD structures in 2002. In 2003 LWD structures were present along the entire transect.

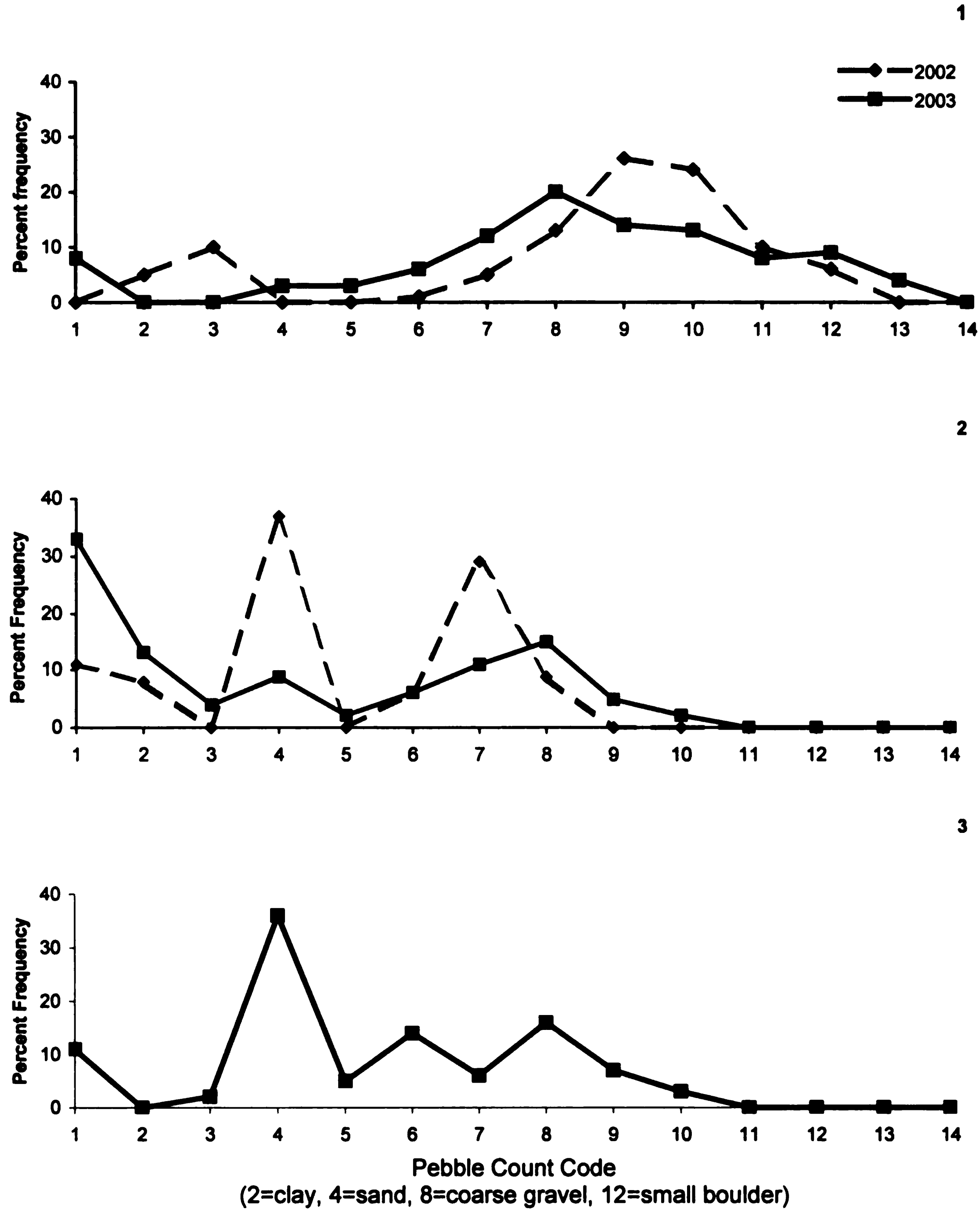

Figure 11. Substrate size percent frequency distribution for Hodenpyl sites one and two in 2002 and 2003. Hodenpyl site three missing 2002 data.

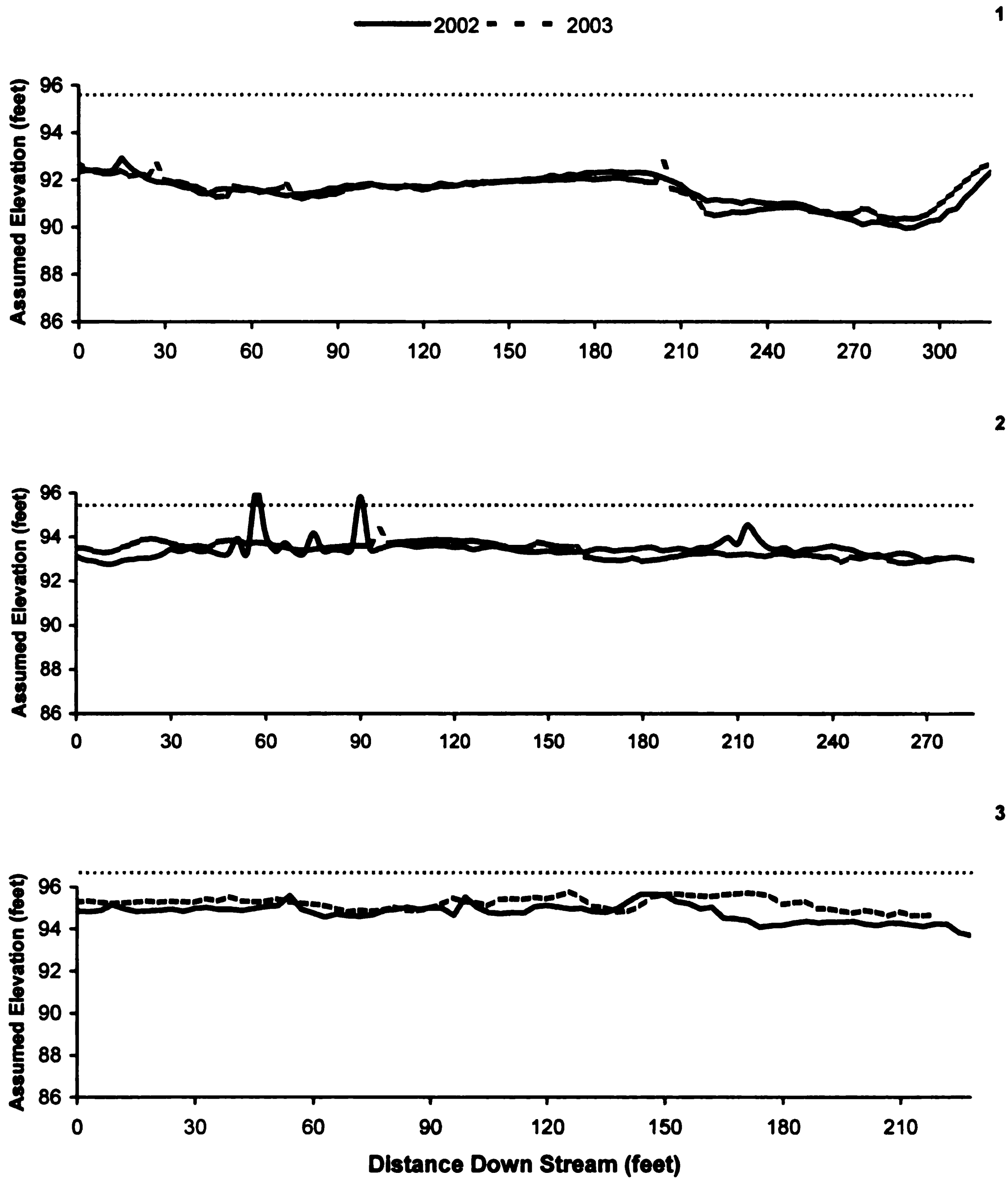

Figure 12. Survey transects from each of the Alcona study sites, showing the longitudinal river channel profile for 2002 and 2003. Water surface (2003) is represented by the dotted top line. Location and size of LWD structures did not change between 2002 and 2003.

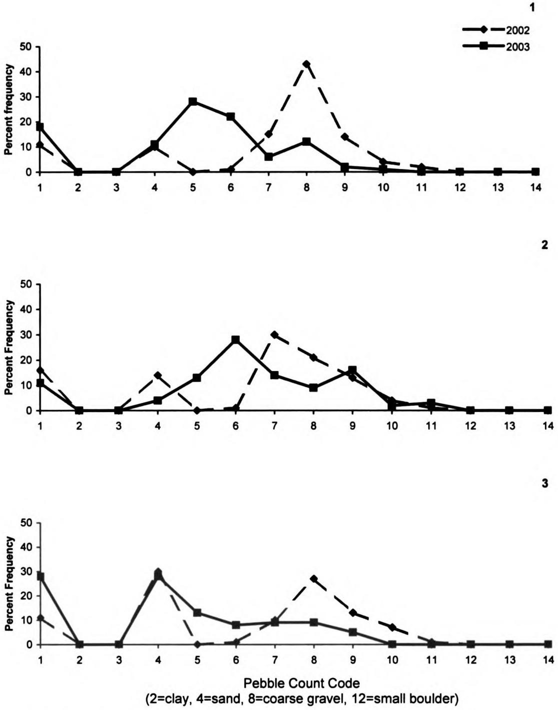

Figure 13. Substrate size percent frequency distribution for each Alcona site in 2002 and 2003.

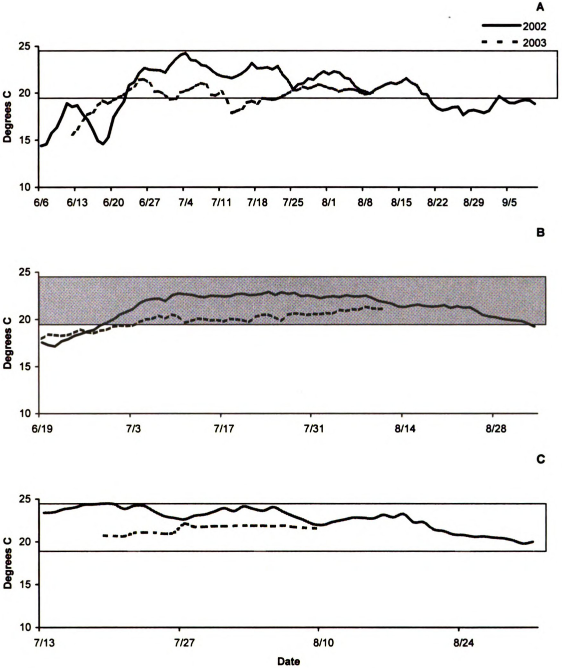

Figure 14. Mean daily water temperatures in °C during the summers of 2002 and 2003 in the Mio (A), Hodenpyl (B), and Alcona (C) river reaches. Shaded area represents critical thermal limits at 19°C (upper thermal limit for growth) and 24.7°C (lethal) for brown trout.

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