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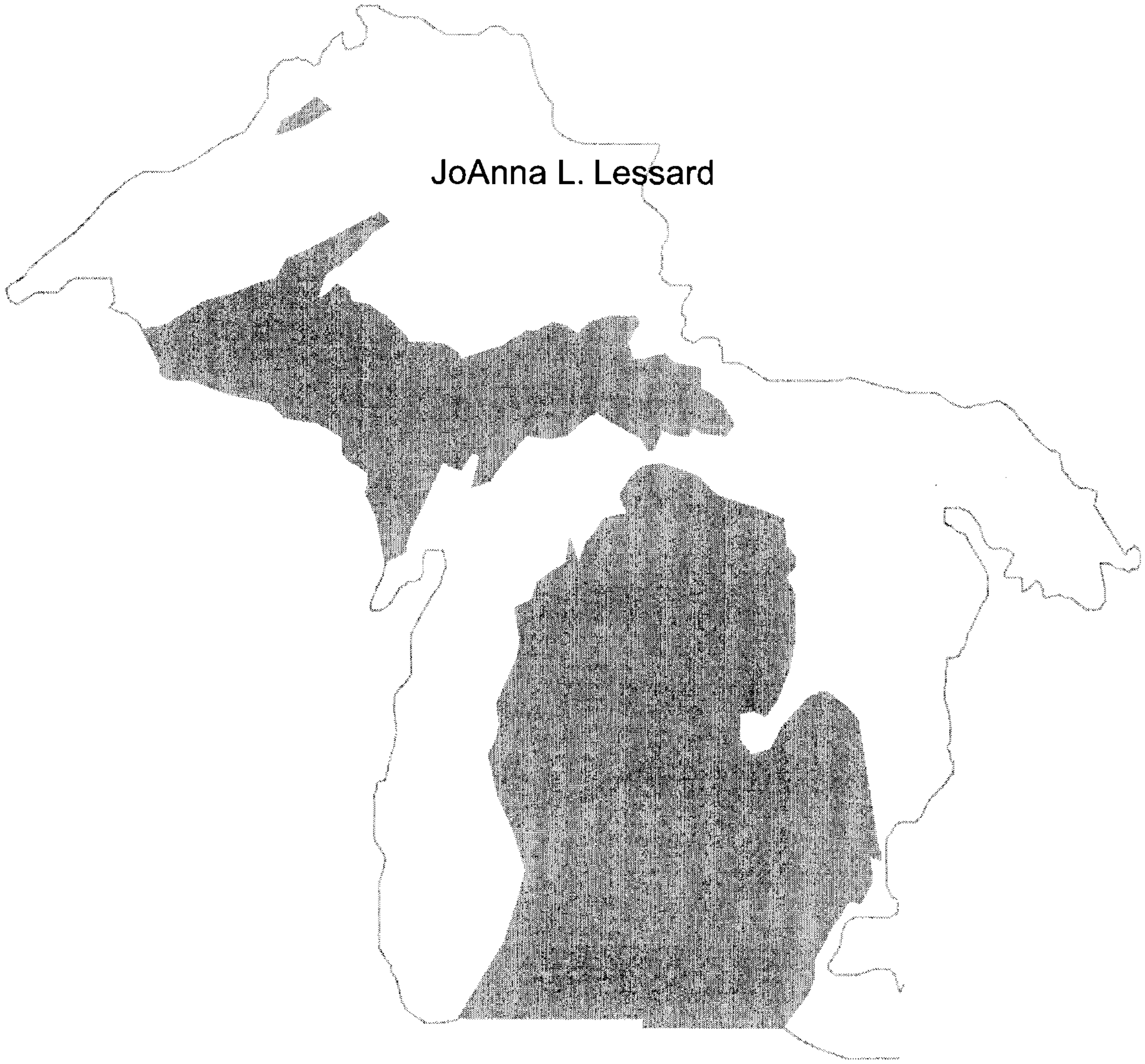
Temperature Effects of Dams on Coldwater Fish and Macroinvertebrate Communities in Michigan

JoAnna L. Lessard



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TEMPERATURE EFFECTS OF DAMS ON COLDWATER FISH AND MACROINVERTEBRATE COMMUNITIES IN MICHIGAN

By

JoAnna Lynn Lessard

A THESIS

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ABSTRACT

TEMPERATURE EFFECTS OF DAMS ON COLDWATER FISH AND MACROINVERTEBRATE COMMUNITIES IN MICHIGAN

By

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In Michigan streams, the impact of small dams on downstream thermal regimes is a major habitat concern. The objective of this study was to examine the effects of temperature increases due to impoundment on downstream fish and macroinvertebrate communities. We sampled fish, macroinvertebrates and habitat upstream and downstream of dams on ten rivers in Michigan during the summers of 1998 and 1999. Fish were collected from block netted sites with an electroshocking unit. A modified Hess sampler was used to collect macroinvertebrate samples. Habitat was assessed using several water quality and habitat parameters. Our results show that small dams can increase downstream temperatures by more than 5 C. Increasing temperatures below impoundments resulted in lower densities of coldwater fish species, specifically brown trout, brook trout, and slimy sculpin, while fish species richness generally increased downstream. Brown trout growth was not related to temperature in these streams. Macroinvertebrates responded to warming with shifts in community composition below dams that significantly increase summer temperature. This study will provide information useful for determining the extent of impact of dams on Michigan's streams, and potentially suggesting modifications in management practices to benefit these resources.

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Introduction

There are over 2,000 dams on Michigan's rivers and streams (Michigan Department of Natural Resources, unpublished data). While these dams return many benefits to society (e.g., hydroelectric power, flood control, water level regulation), they frequently have negative impacts on populations of aquatic organisms, particularly fish. Reductions in fish populations due to impoundments are well documented, and occur through a variety of mechanisms. The best-known mechanism is the reduction in upstream migration that occurs where dams do not have adequate fish passage facilities (Holden 1979; Vogel et al 1988). Effects on fish migration are most obvious for anadromous fishes, and can severely reduce or even extirpate local populations (Brooker 1981, Ward and Stanford 1987). Another mechanism of impact is the change in water quality and habitat conditions that occur below a dam. Creation of a reservoir affects river habitat in many ways, potentially impacting stream-resident fishes as well as migratory species.

It has been widely observed that physical/chemical factors such as stream substrate, dissolved oxygen and water temperature can be affected by impoundment (Cummins 1979; Stanford and Ward 1979; Ward and Stanford 1979; Waters 1995; Hayes 1998). Dams affect substrate by acting as a "sediment sink", holding back finer sediments that normally would be transported downstream (Ward and Stanford 1983a; Ward and Stanford 1987; Waters 1995). This is of particular importance for the region of stream adjacent and immediately upstream of the impoundment, often referred to as the "impacted zone"(Klomp 1998; Mistak 1999). In this area, coarse sediments such as cobble can become covered with sand and silt potentially creating conditions unsuitable for the biota normally occurring in those stream sections (Waters 1995). Downstream reaches are often starved of these finer substrates and therefore become dominated by larger, more stable substrates.

Dissolved oxygen decreases occur when deep release dams draw water from the anoxic hypolimnion (Ward and Stanford 1987). It has been assumed that dissolved oxygen decreases are not as much of a problem for smaller facilities, like those that are common in Michigan streams. This is primarily due to the fact that the size of impoundment often precludes anoxia, recovery of oxygen levels is fairly rapid downstream, and releases are often from the epilimnion (Brooker 1981; Ward and Stanford 1987).

The characteristics common to dams that make dissolved oxygen changes unimportant, often cause an increase in downstream temperatures, making temperature the primary habitat concern. Reservoirs act as sinks for heat as well as chemicals and sediment (Brooker 1981; Ward and Stanford 1987). Surface waters spilling downstream from shallow or deep, stratified reservoirs, are often several degrees warmer than upstream reaches (Fraley 1979; Ward and Stanford 1987). Temperature increases downstream of surface release dams are a major habitat concern in Michigan.

Increases in stream temperature may shift the temperature out of the range that a given species is genetically adapted to. Coldwater stenotherms are genetically adapted to interact, feed, grow and survive better in colder temperatures (Carlander 1969; Allan 1995). They have also evolved to exploit the food sources provided by these colder habitats which are typically cold adapted macroinvertebrates and other fish (Allan 1981; Hubert et al. 1993; Rader 1997). Thus, increases in temperature can not only affect their internal physiology but also their food source.

Many of the dams in Michigan are relatively small facilities that once generated hydro-electrical power for local needs. A majority of these facilities were deregulated in the 1950's and so remain as local landmarks, creating lentic fishing opportunities and stabilizing water levels for lake front property. Without regulation, these dams are primarily surface release facilities which have the greatest potential to increase downstream temperatures.

Due to concerns over the effects of dams on stream habitat conditions, hydro-power facilities are governed to operate under water temperature and dissolved oxygen standards similar to those for point-sources of pollution. The policy of the Michigan Department of Environmental Quality (MDEQ) stipulates that, for coldwater streams, such facilities cannot raise water temperature by more than 2° C or cause dissolved oxygen to drop below 7 mg/l. The basis for these regulatory standards is not well established and the effects of violations or chronic changes within the standards on fish populations is unknown.

This research investigated how changes in stream temperature, downstream of hydro-power facilities impacted coldwater fish and macroinvertebrate communities. My goal was to characterize how temperature increases due to impoundment affected the growth and population density of selected coldwater fish species including: rainbow trout (*Oncorhyncus mykiss*), brown trout (*Salmo trutta*), brook trout (*Salvelinus fontinalis*) slimy sculpin (*Cottus cognatus*) and mottled sculpin (*Cottus bairdi*). I also wanted to characterize how the downstream macroinvertebrate community changed with increasing temperatures. I initially hypothesized that population densities of coldwater fish and macroinvertebrates would be lower downstream of dams that increased downstream temperatures, and that this decrease would be magnified as temperatures rose above their physiological optimum. Additionally, coldwater trout species that remained in waters above their optimum would exhibit poorer growth than individuals of the same species that occurred in cooler, upstream waters. Changes in the macroinvertebrate community have the potential to interact with the direct effects of temperature by altering the primary food source for trout and sculpin. Besides the indirect impact that macroinvertebrate changes may have on the selected target fish species, they also offer another way to examine how temperature increases impact coldwater stream communities.

The specific objectives for this study were to:

(1) Determine and compare population densities of brook, brown and rainbow trout and slimy and mottled sculpin above and below dams in coldwater streams in Michigan.

(2) Determine and compare growth rates of brook, brown and rainbow trout above below dams in coldwater streams in Michigan.

(3) Examine and compare water quality and habitat differences above and below impoundments.

(4) Examine the effects of water temperature differences above and below impoundments on fish growth and population densities of selected species.

(5) Characterize and compare macroinvertebrate community composition above and below impoundments.

(6) Examine the effects of water temperature differences above and below impoundments on the macroinvertebrate community and its impact on selected fish species.

Study Area

Three streams were sampled during the summer of 1998 and seven streams were sampled during the summer of 1999 (Table 1). The requirements for streams being considered in the study were: 1) they had a dam; 2) they had a resident trout population; 3) there was no operational fish ladder (to allow independent comparisons of upstream and downstream populations); and 4) were wadeable (due to equipment constraints). Streams were chosen with the advice of Michigan Department of Natural Resources personnel and were based on the above criteria and a broad geographic distribution in lower Michigan. Due to the natural distribution of coldwater streams, the arrangement of study streams tended to have a high representation in the northwest portion of the lower peninsula (Figure 1).

The majority of streams were sampled at six sites, three upstream and three downstream of the dam (Figure 2). Two streams (Cedar Creek and White River) only had two downstream sites because water depths prevented an additional site. In three of the streams (Cedar Creek, Dowagiac Creek, and Manton Creek), there were two dams in relatively close proximity so an additional site was placed between the two dams. In these streams, upstream refers to sites above both dams and downstream refers to sites below both dams. Site selection was based on access (usually by a road crossing) and distance from the dam. When possible, sites were separated from each other by distances between 1 and 2 river miles. We were careful to select sites upstream that would represent the unregulated river, without going too far upstream to no longer be representative of the river along the normal continuum. We were also careful to not

Table 1. Streams sampled in 1998 (designated by *) and 1999. Number of dams are noted in superscript (temperature, width and depth values are means from all six sites).

Name	**County**	$\bar{x}$ **Temp. (C)**	$\bar{x}$ **Width (m)**	$\bar{x}$ **Depth (cm)**
Boardman River^1	Grand Traverse	16.7	16.9	42.4
Cedar Creek^2	Antrim	15.3	14.3	44.7
*Dowagiac Creek^2	Cass	21.1	11.1	33.5
*Fish Creek^1	Ionia	23.9	18.0	41.3
*Manton Creek^2	Wexford	15.8	6.9	17.3
Maple River^1	Emmet	16.3	11.1	38.2
Middle Branch River^1	Osceola	16.3	10.5	28.7
Prairie Creek^1	Ionia	18.9	12.2	28.7
Sugar River^1	Gladwin	19.4	6.8	23.6
White River^1	Newaygo	17.0	10.2	32.0

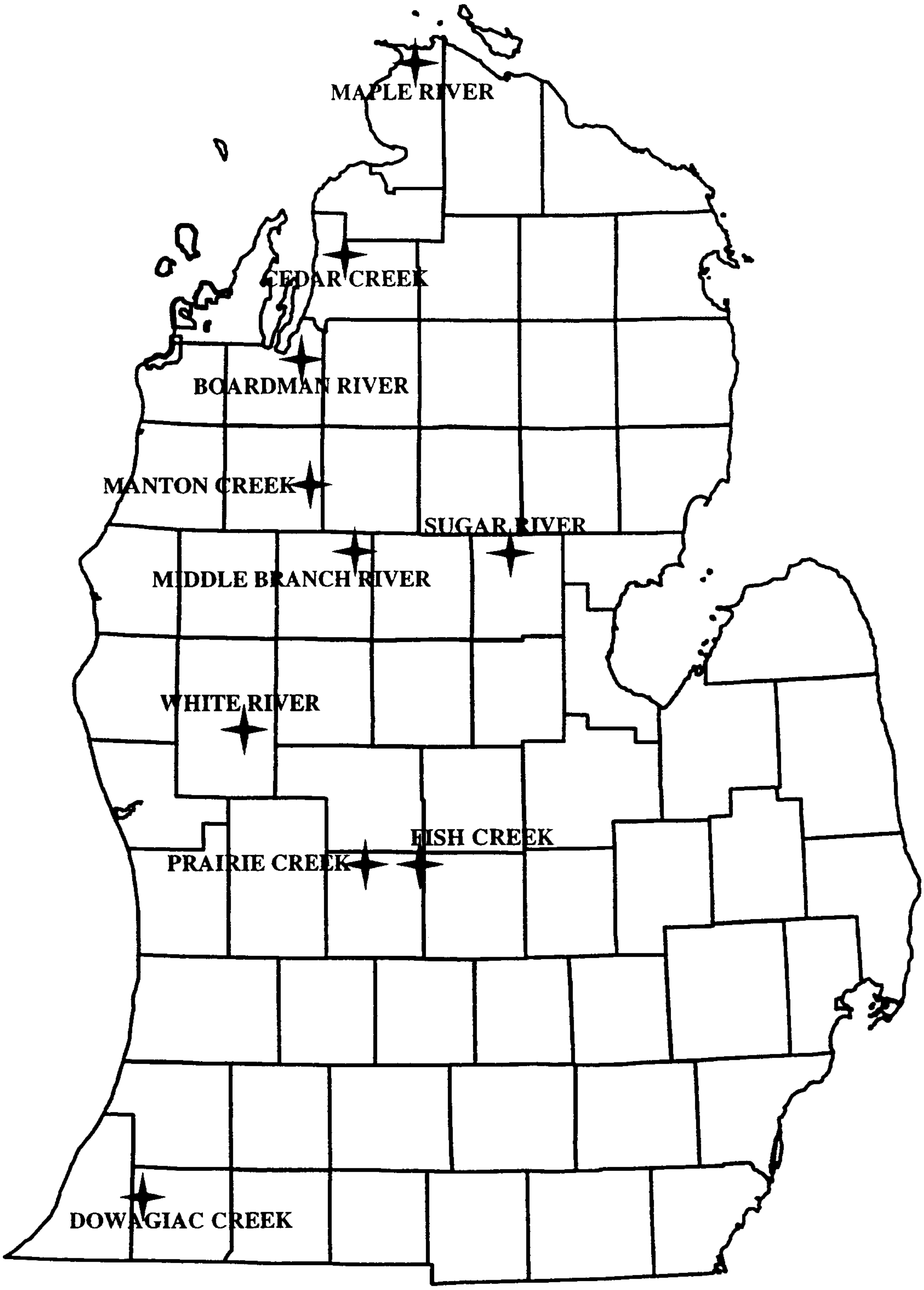


Figure 1. Location of streams sampled in Michigan's Lower Peninsula

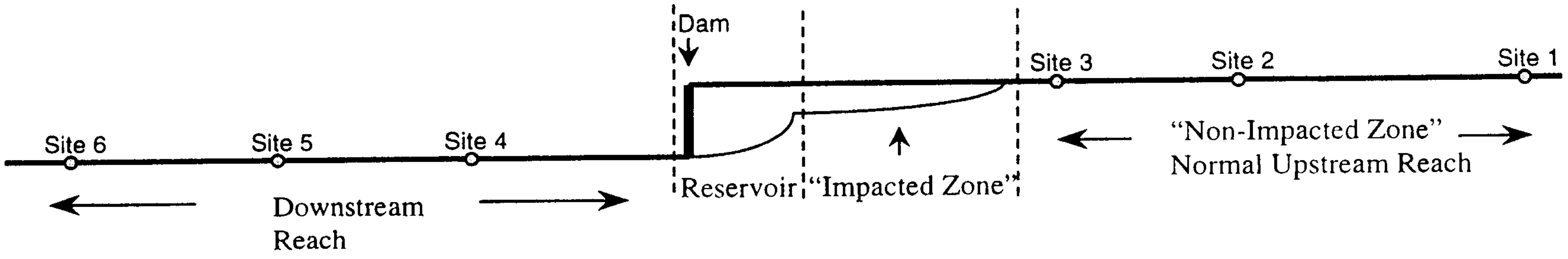


Figure 2. Cross sectional diagram of a study stream shows the areas sampled.

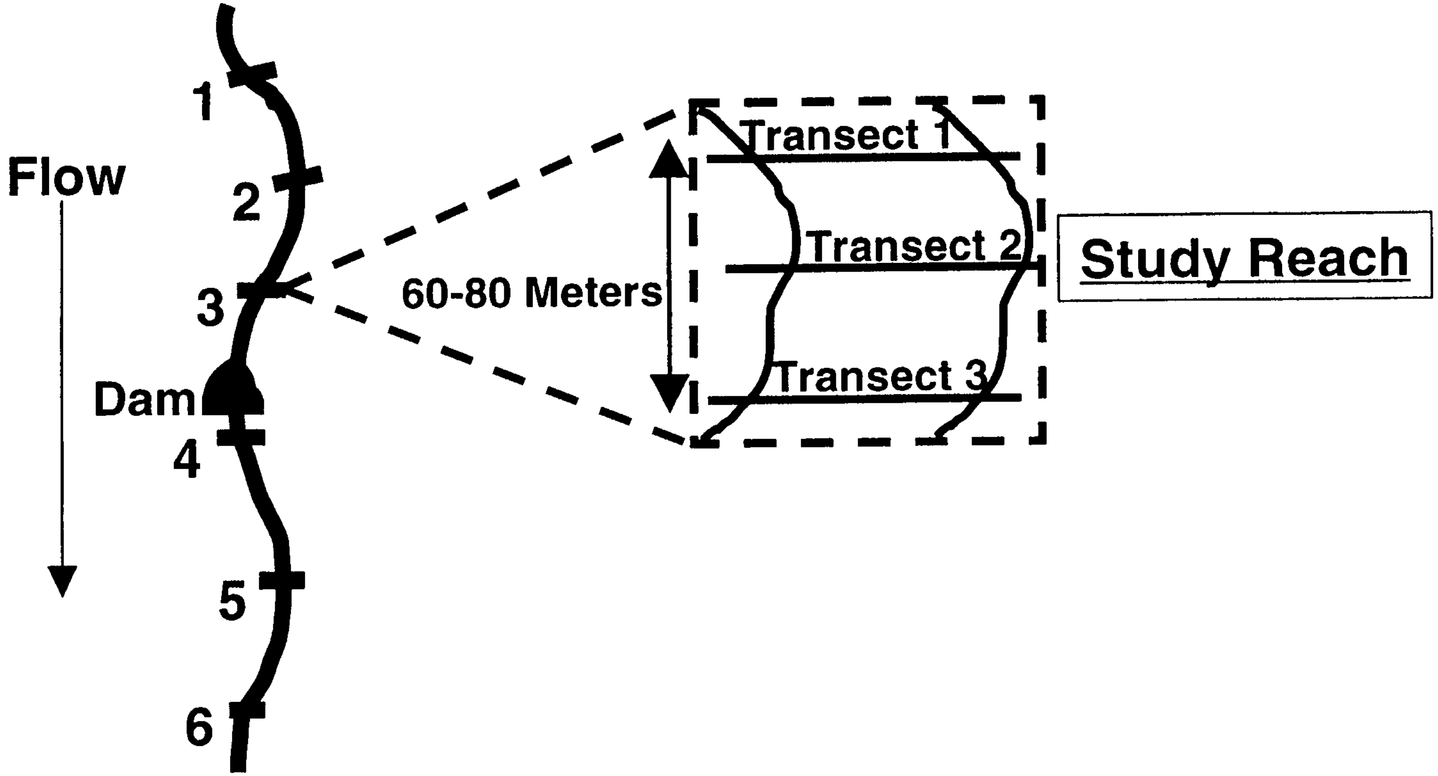
sample too near the reservoir or in the “influence zone”(Figure 2). Below the dams, one site was generally placed approximately 100 m downstream of the plunge pool. The two additional downstream sites were separated by 1 to 2 river miles when possible, in order to determine longitudinal impacts.

Methods

Field Methods

Study reaches (sites) were 60 to 80 m long, generally ending at the boundary between two mesoscale habitat units (e.g. riffle/run, pool/riffle). Each site contained an upstream, downstream and middle transect (Figure 3). The middle transect was located exactly halfway along the length of the site. At each transect, wetted width, depth and substrate were measured. Width was taken to the nearest 0.1 m by extending a tape measure across the stream channel. A depth profile was recorded for each transect by recording the depth in centimeters at one meter intervals across the entire channel. If the stream width was less than 10 m wide, then depth was recorded at shorter intervals so that at least ten depth readings could be recorded. Substrate size composition was determined using the pebble count method (Kondolf and Li 1992). This involved one observer slowly crossing the transect and determining the particle size of the substrate below his/her feet at each step. This process was repeated coming back across the transect. At the middle site above and below each dam (i.e. site 2 and 5) a stream flow profile was recorded at the middle transect using a flow meter at 1 m intervals.

Along with the physical habitat measurements described above we measured several water quality parameters at each site. Stream temperature, dissolved oxygen and conductivity readings were recorded on the day of sampling using a Yellow Springs Instruments (YSI) meter. In 1999, Onset® temperature loggers were placed at the upstream and downstream sites proximal to the dam in each river. For those rivers with two impoundments, an additional logger was placed between the two impoundments, in



Sampling Design for Study Streams

Figure 3. Location of sample sites with an enlarged view of site 3 indicating where the three transects are located.

order for each dam's individual impact to be documented. Hourly temperature readings were recorded from May to October for all ten streams. Loggers were calibrated in the laboratory before being deployed in the field and were recalibrated in the laboratory after data retrieval. Water samples were taken at each site for laboratory analysis of total phosphorous concentrations. Samples were put on ice in the field, and were frozen until analysis. In 1998, we also collected 1 liter of water from the center of the stream at each site for analysis of seston levels. These samples were kept refrigerated until processing.

In addition to habitat conditions, fish and macroinvertebrate populations were sampled at each site. Fish were sampled at sites where the water was deep enough using a barge electroshocking unit and a backpack unit was used at the shallower sites. Block nets were placed at the upstream and downstream transects to keep fish from moving in or out of our site while sampling. A multi-pass removal method was used to estimate the population size of our target species with a minimum of three passes (Van Deventer and Platts 1983). Each pass consisted of shocking the entire stream channel, concentrating on areas with more cover (e.g. weed beds, log jams, overhanging banks etc.), from the downstream transect moving upstream to the top of the site. When the backpack shocker was used it was necessary to shock in a zig-zag motion from one bank to the other along the site. The barge shocker had two anode poles allowing simultaneous shocking of each half of the stream channel. All fish caught in the first pass were measured for total length (up to 25 individuals for each non-target species) and identified to species. Only target species and species not previously caught were identified and measured in the following two passes, and all other species were counted. If there was not enough of a removal of trout in three passes then more passes were done until a good removal was attained. A good removal was defined as a catch of <1/2 as many trout as were caught in the previous pass. Fish capture data was recorded after each pass and fish were returned downstream of the block nets to reduce mortality. Any unknown fish were fixed in 10% formalin and brought back to the lab for identification. Scales were taken from all trout >100 mm in length, above the lateral line between the posterior end of the dorsal fin and the anterior end of the adipose fin (Minard and Dye 1997). Fish <100 mm were recorded as age 0 or young of the year (YOY) trout (Hinz and Wiley, 1998).

Three macroinvertebrate samples were collected at each site using a modified Hess sampler (Merritt et al. 1996). When possible, sites were selected that contained at least a portion of stream shallow enough so that the water did not flow over the top of the sampler. The sampler was placed over a cobble or gravel area so that the sampler laid firmly on the stream bottom. The bottom was disturbed for two minutes and surfaces of the rocks were scraped so that macroinvertebrates on and in the substrates would be swept into the 250μm mesh bag attached to the downstream side of the sampler. If no cobble or gravel was available, or if the cobble was too large for the sampler to fit over it, then a representative stable substrate (e.g. logs, roots, boulders) or soft substrate was sampled. The debris and insects caught in the modified hess sampler were washed into a labeled jar and fixed with ethanol. Samples were transported to the laboratory for processing.

Laboratory Methods

Total, organic and inorganic seston concentrations were determined for the 1998 streams according to the method described by Wallace and Grubaugh (1996). Total phosphorous concentrations were analyzed using persulfate digestion (Valderrama 1981).

Unknown fish specimens were keyed out to species (Eddy 1969; Smith 1989). Trout scales were mounted between two glass slides so that they could be read. Scales were aged and annuli were measured (mm) using an Optimus imaging system. Macroinvertebrates were dyed with rose bengal and divided into two size categories (greater than 1mm, and less than 1 mm but greater than 45μm) using 1mm and 45 μm sieves. This helped to reduce the amount of fine substrates in the portion of the sample with the most insects. The greater than 1mm samples were picked, sorted and identified to the family level unless generic level identification was required for functional feeding group classification (Cummins and Wilzbach 1985; Cummins and Merritt 1996). The less than 1mm but greater than 45μm samples were further divided into 0.5 or 0.25 subsamples depending on the amount of particulates that were in the sample. These samples were then picked, sorted and identified to the family level. Preliminary analysis indicated that a negligible amount and diversity of insects (e.g. minute midge larvae and zooplankton) were lost from the greater than 1mm samples. Because of this, and indications in the literature that prey <1 mm are not very important components of trout diet (Rader 1997), we restricted our analysis to the large category samples.

Data Analysis

The impact of each dam on mean summer temperature was determined by first computing the mean summer temperature from the sites with the data loggers (sites 3 and 4 in each stream)from June 1 to August 31. The logger data gave values for how temperature changed from directly above the dam to directly below the dam. Mean summer temperature at the sites without data loggers, was computed by calculating the deviation of each point measurement at that site from the logger site in that stream section. This was done by taking the difference between the temperature read by the logger in that section, on the day and time of sampling, and the temperature recorded by the YSI meter. For example, site 2 in the Boardman River was 16 °C (± 0.1 °C) at 12:00 p.m on July 27, 1999 (the day we sampled) and the temperature read by the logger at site 3 at the same time and day was 15 °C (± 0.25 °C), so the deviation of site 2 from site 3 was 1 °C (± 0.35 °C). Assuming this deviation remains constant throughout the summer, the best available estimate for mean summer temperature at site 2 was 1 °C (± 0.35 °C) warmer than site 3. The three additional sites placed between the two impoundments in Dowagiac, Cedar and Manton Creeks were not used to adjust temperatures for any other site, but were used to calculate the impacts of each individual dam. Mean summer temperature above each dam was calculated as the mean of all three upstream sites once they were adjusted for time and date of sampling. The same was done for the downstream sites. The overall impact of each dam was calculated as the difference in adjusted mean summer temperature above the dam minus the adjusted mean summer temperature below the dam.

For the purposes of analyzing the streams based on the level of temperature impact, streams were divided into two categories "Low" and "High". Streams where downstream summer temperatures increased by 2.15 °C or less were considered low impact, and high impact streams were streams where downstream summer temperatures increased by more than 2.15 °C. This value was selected as the cut off point because a division of stream impacts was evident, and because this approximates a change of 2 °C which is the maximum temperature increase allowed for Michigan coldwater streams.

Sites were categorized as above or below the dam so that analysis could be done on the differences that occurred between stream sections. Sites between dams were dropped from these analyses due to their sporadic occurrence in this study. So stream sections that were compared for those rivers with two dams were the sections above both dams and below both dams, the between section was ignored. An α value (Type I error) of 0.05 was used for all statistical tests. When possible, all stepwise regression analyses were done using the Maximum R^2 improvement technique (MaxR) and bounds were put on the significance level required for a parameter to enter and remain in the model (α=0.1 and 0.05 respectively). MaxR is considered superior to the normal stepwise technique because it does not settle on a single model but gives a series of models starting with the best one parameter model, offering the largest R^2, and then the best two parameter model and so on. MaxR is also preferable to normal stepwise regression in that it evaluates all switches for their impact on R^2 before any switches are made (Hocking 1976).

Habitat (width, depth, substrate, flow, seston, dissolved oxygen, conductivity, phosphorous and temperature) differences above and below the dams for each stream were estimated using a general linear model analysis. Analysis of habitat differences over all study streams as well as differences in macroinvertebrate family richness, Plecopteran family richness, Ephemeroptera, Plecoptera, Trichoptera (E.P.T.) richness, macroinvertebrate functional feeding group composition (Rosenberg and Resh, 1996) and fish species richness, comparing above and below sections, was done using a mixed model analysis of variance (ANOVA), where stream was treated as a random effect and position was treated as a fixed effect.

$$y=\alpha + \beta x + b + e , \qquad [1]$$

where y is the parameter of interest, α is the intercept and overall mean, β is the slope and effect of position, x is an indicator variable for position, *b* is the effect of stream, *e* is the error term, and *b* and *e* are independent random effects (Littel et al. 1996).

To analyze the relationships between the macroinvertebrate and fish communities above and below the dams and habitat changes, a correlation analysis was run to investigate how each individual habitat parameter was related to macroinvertebrate , Plecopteran, E.P.T. and fish species richness. Next, stepwise regression (MaxR) was used to determine which habitat parameter(s) was the best predictor of each macroinvertebrate and fish community parameter. If significant differences were found in the macroinvertebrate or fish community between upstream and downstream sections, then another stepwise regression (MaxR) was run on these community differences and changes in select habitat parameters (temperature, conductivity, dissolved oxygen, phosphorous, depth and substrate). This was done to see which habitat parameter(s) could be driving changes in community richness.

In order to estimate the impact of the dams on fish community size (i.e. average total length of all individuals of all species combined), an analysis similar to that described for changes in richness was performed. Stepwise regression (MaxR) was used to see which habitat parameter changes were most related to changes in fish community size.

To investigate the impact of the dams on fish and macroinvertebrate community composition, Sϕrensen's similarity index values (Sϕrensen 1948) were computed comparing stream sections using the following equation:

$$QS=2C/(A+B), \qquad [2]$$

where QS is the index of community similarity, A is the number of taxa upstream, B is the number of taxa downstream, and C is the number of taxa common in both stream sections. Species level of taxonomic resolution was used for the fish community and family level was used for macroinvertebrates. Stepwise regression (MaxR) was used to see which habitat changes, if any, were related to fish and macroinvertebrate community similarity.

Population abundances of the five target species (brown, brook and rainbow trout and slimy and mottled sculpin) were estimated using Microfish (Van Deventer and Platts 1985), a software program for removal methods. Microfish calculates maximum likelihood population estimates based on the pattern of removal for an individual fish species that was obtained during sampling. Population densities (fish/hectare) were calculated for each site by dividing the Microfish abundance estimates of each species caught by the area sampled (m^2), and then converting this estimate to number per hectare. The density data were averaged for each position, to get an idea of how the overall population abundance changed below the dams. Because preliminary analyses indicated that the mean density data were not linear and the variance was not homogeneous a $\log_e$ transformation on the density estimates (Ln(density+1)) was used to correct these problems. Next, a correlation analysis was run and Pearson r values were calculated for the transformed data and each habitat parameter to see which correlations appeared strongest. Using this analysis as a guide, stepwise regression analysis (MaxR) was run to determine which habitat parameter(s) were most related to the mean Ln(density+1) of each species (referred to as Lnbrook, Lnbrown, Lnrainbow, Lnslimy, and Lnmottled).

A predictive model was developed for each species' density and the most related habitat parameter(s) using a mixed modeling analysis of covariance (ANCOVA), where stream was treated as a random effect and position and habitat covariates were treated as fixed effects. This was done to see if the habitat parameter most correlated with density had an impact that differed according to position when variability across streams was considered.

Target species were often caught only in one section of the stream (only above or only below). Therefore, logistic regression was used to model how habitat conditions were related to the presence or absence of our target species. Stepwise logistic regression analysis was used to determine which habitat parameter(s) was most related to the presence of our target species and to develop predictive models.

Ageing data were examined to determine the age structure for each species of trout in each study stream. Growth analysis was done using the Hile method (modified from the Fraser-Lee method) of linear regression to calculate the length of the fish when scales were first formed (R_{zero}) and backcalculations of the length at age using the following equation (Francis 1990).

$$B_{age}=((L-R_{zero})*(R_{age}/TR))+R_{zero}, \qquad [3]$$

where B_{age} is the backcalculated length at age, L is the total length of the fish, R_{zero} is the length of the fish at first scale formation, R_{age} is the radius (mm) of the scale at the annuli corresponding to the age of interest, and TR is the total scale radius (mm). These backcalculations were then used to calculate the length at age for the previous year and incremental growth (i.e. growth for each age step). An incremental growth model was selected to model the previous year's growth so that growth rate would not be biased with month of capture (Weisberg 1993; Weisberg and Frie 1987). Since salmonids often show different growth patterns in their first year than in subsequent years due to variability in habitat requirements (Deegan et al. 1999), the analysis was done separately for age 1 fish and all fish ages>1. Age 0 fish were excluded since they only show a partial years growth.

Only brown trout were caught in sufficient numbers above and below the dams to characterize and analyze growth robustly. For brown trout ages>1, regression analysis of incremental growth on previous length at age and stream position was done for each stream sampled (Weisberg and Frie 1987). Mixed modeling ANCOVA was used to determine differences in growth between dam impacts and stream positions, across all streams, by testing the slopes of the regression lines for homogeneity. Previous length at age, impact and position were treated as fixed effects and stream was again a random effect. Stepwise regression (MaxR) was used to see which factor(s) was most related to incremental growth of brown trout age>1. The factor(s) identified by stepwise regression as important was then put into a mixed model to determine if the factor(s) was still significant once stream variance was considered and also to test for differences between stream positions. For age 1 brown trout, a similar mixed modeling ANCOVA was run on incremental growth, to identify what was most related with their growth in the previous year (i.e. age zero growth) across all streams.

Results

Habitat Characteristics

Physical habitat characteristics varied widely among study streams, but several longitudinal trends were apparent (Table 2). The study streams progressively became wider, deeper and had greater discharge below dams compared to sections above the dams (Figure 4). Stream width ranged between 4 and 22 m ($\bar{x}$=10.7 m) upstream of the dams and between 8 and 24 m ($\bar{x}$=13.2 m) downstream. Depth ranged between 8 and 53 cm ($\bar{x}$=28.1 cm) upstream and between 16 and 69 cm ($\bar{x}$=38.4 cm) downstream. Discharge ranged between 0.40 and 20.21 m^3/s ($\bar{x}$=5.6 m^3/s) upstream and between 1.5 and 21.8 m^3/s ($\bar{x}$=7.7 m^3/s) downstream. Stream substrate on average became more coarse downstream of the dams (Figure 5). Most of the study streams were dominated by sand and gravel (3.53) upstream and sand, gravel, cobble (3.84) composites downstream. Seston concentrations for the 1998 streams were not significantly different between above and below sections and so seston was dropped as a parameter of interest in 1999 (Figure 5).

The chemical environment also varied substantially among study streams and among sites within a stream (Table 2). Phosphorous concentrations were highly variable and not significantly different between upstream and downstream sections on average (Figure 6). Analysis of variance (ANOVA) revealed that only the White River contained significantly different total phosphorous concentrations between upstream and downstream sections (p=0.0081). Conductivity ranged between 259 to 540 μs/cm across all streams. Average conductivity was slightly higher below the dams (Figure 6)

Table 2. Habitat characteristics for stream positions in each study stream.
(Substrate Particle sizes: 1=clay 2=silt 3=sand 4=gravel 5=cobble 6=boulder 7=bedrock)

						Seston (mg/l)		
Stream	**Position**	**Temp (C)**	**D.O.(mg/L)**	**Cond (us/cm)**	**Phos (ug/L)**	**Total**	**Inorganic**	**Organic**
Boardman	above	14.7	10.7	260	15.3	.	.	.
Boardman	below	18.7	9.8	289	18.0	.	.	.
Cedar	above	13.8	11.9	263	10.8	.	.	.
Cedar	below	17.6	9.5	303	10.3	.	.	.
Dowagiac	above	20.2	7.6	388	52.8	0.013	0.008	0.005
Dowagiac	below	22.1	7.8	459	37.5	0.003	0.002	0.001
Fish	above	23.0	10.5	533	38.3	0.009	0.003	0.007
Fish	below	25.0	9.6	541	61.7	0.010	0.003	0.006
Manton	above	13.0	11.4	252	10.2	0.005	0.003	0.002
Manton	below	18.5	9.5	324	12.8	0.006	0.002	0.004
Maple	above	16.8	9.7	268	20.1	.	.	.
Maple	below	15.8	9.8	269	9.4	.	.	.
Middle Branch	above	14.3	9.1	365	.	0.007	0.003	0.004
Middle Branch	below	18.2	10.2	332	.	0.010	0.004	0.006
Prairie	above	18.7	10.7	473	16.9	.	.	.
Prairie	below	19.1	10.6	476	26.0	.	.	.
Sugar	above	17.9	10.1	320	18.7	.	.	.
Sugar	below	21.0	9.9	321	19.4	.	.	.
White	above	15.6	9.7	294	15.3	.	.	.
White	below	19.0	9.9	320	27.8	.	.	.
Mean	**above**	**16.8**	**10.2**	**341.7**	**22.0**	**0.009**	**0.004**	**0.004**
Stderr	**above**	0.59	0.29	17.44	2.97	0.0014	0.0009	0.0007
# streams (#sites)	**above**	10(30)	10(30)	10(30)	9(27)	4(12)	4(12)	4(12)
Mean	**below**	**19.5**	**9.7**	**363.4**	**24.8**	**0.007**	**0.003**	**0.004**
Stderr	**below**	0.55	0.23	17.74	3.29	0.0012377	0.0005	0.0009
# streams (#sites)	**below**	10(28)	10(27)	10(28)	9(24)	4(11)	4(11)	4(11)

Table 2 (cont'd)

Stream	**Position**	**Substrate**		**Width (m)**	**Depth (cm)**	**Velocity(m/s)**	**Discharge (cu.M)**
		Mean	**Mode**				
Boardman	above	3.2	3	15.2	39.1	.	.
Boardman	below	3.6	4	18.5	45.7	.	.
Cedar	above	2.9	3	10.3	30.9	.	.
Cedar	below	4.2	4	20.3	65.3	.	.
Dowagiac	above	3.2	4	11.6	22.0	0.16	4.9
Dowagiac	below	3.0	3	10.6	45.1	0.38	14.1
Fish	above	4.3	4	18.1	35.4	0.31	20.2
Fish	below	4.5	5	17.8	47.1	0.29	21.8
Manton	above	3.4	3	5.3	11.7	0.20	1.1
Manton	below	3.2	3	8.5	22.8	0.24	3.3
Maple	above	3.4	3	9.1	34.0	0.27	4.9
Maple	below	4.5	4	13.2	42.4	0.23	10.1
Middle Branch	above	3.3	3	9.9	28.0	0.20	3.2
Middle Branch	below	3.8	4	11.1	29.3	0.27	4.9
Prairie	above	3.9	4	12.7	27.5	0.28	3.5
Prairie	below	4.1	4	11.6	29.9	0.34	4.0
Sugar	above	3.7	4	4.9	16.0	0.13	0.4
Sugar	below	4.1	5	8.7	31.2	0.11	1.6
White	above	4.0	4	9.4	36.6	0.43	6.5
White	below	3.6	3	11.4	25.2	0.22	1.5
Mean	**above**	**3.5**	**4**	**10.7**	**28.1**	**0.25**	**5.6**
Stderr	**above**	0.10	0.14	0.79	1.90	0.02	1.87
# streams (#sites)	**above**	10(30)	10(30)	10(30)	10(30)	8(16)	8(16)
Mean	**below**	**3.8**	**4**	**13.2**	**38.4**	**0.26**	**7.7**
Stderr	**below**	0.13	0.19	0.85	2.53	0.04	1.95
# streams (#sites)	**below**	10(28)	10(28)	10(28)	10(28)	8(16)	8(16)

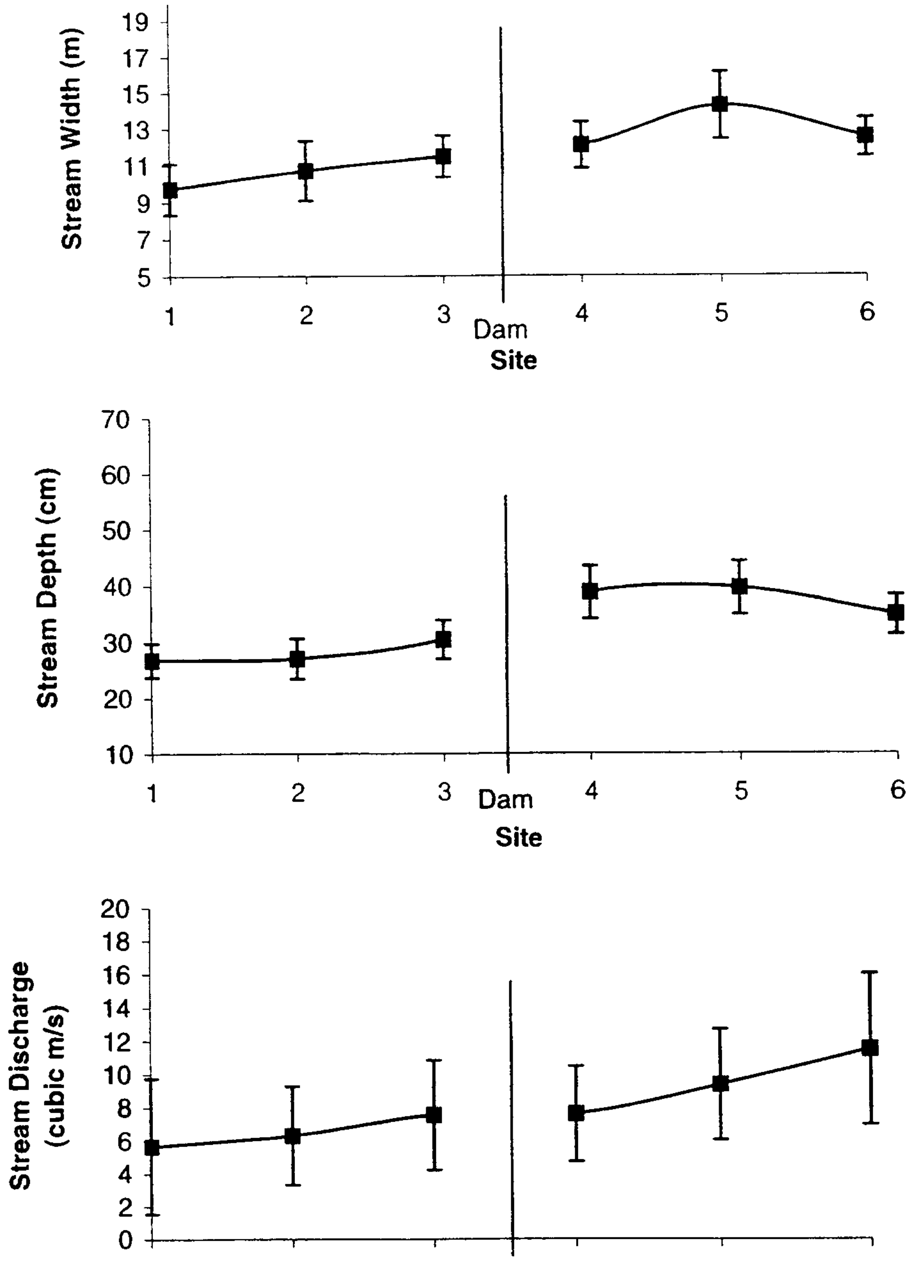
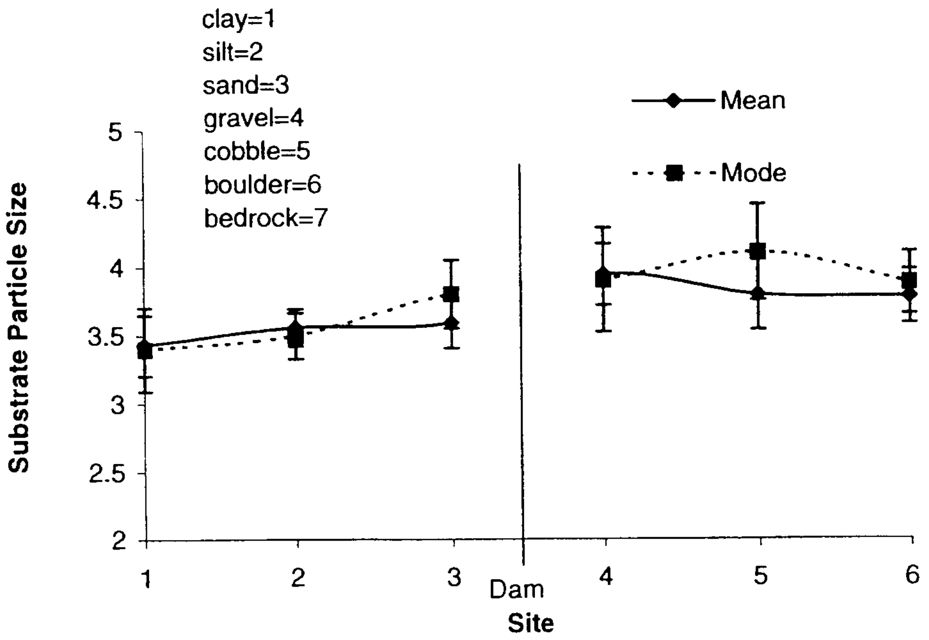


Figure 4. Average trends in width (top), depth (middle) and discharge (Bottom) (+- one standard error) above and below the dams.



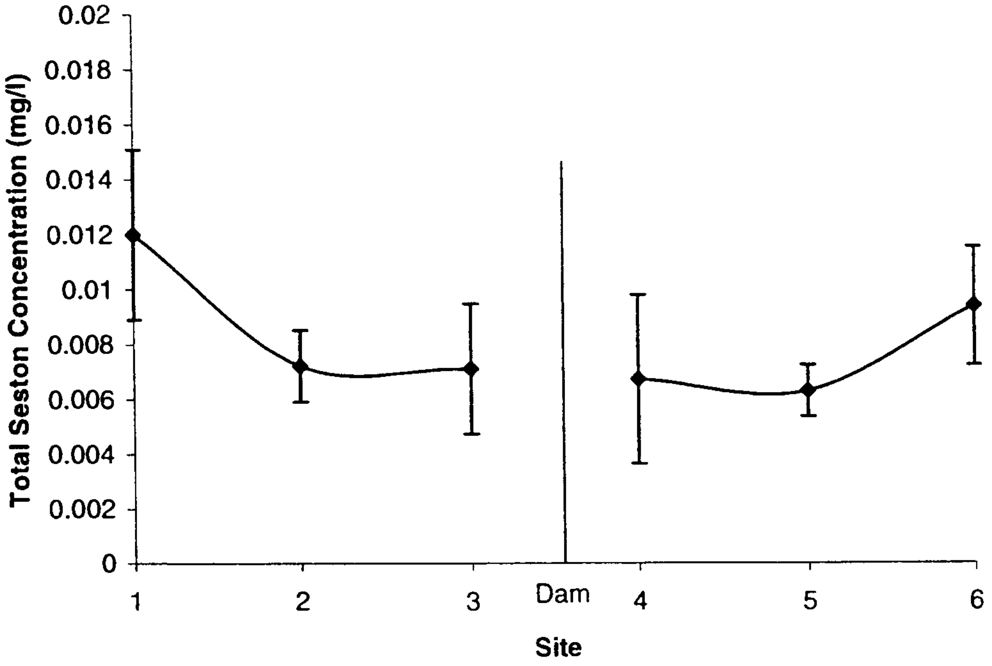


Figure 5. Average trends in substrate particle size (top) and total seston (bottom) (+- on standard error) above and below the dams.

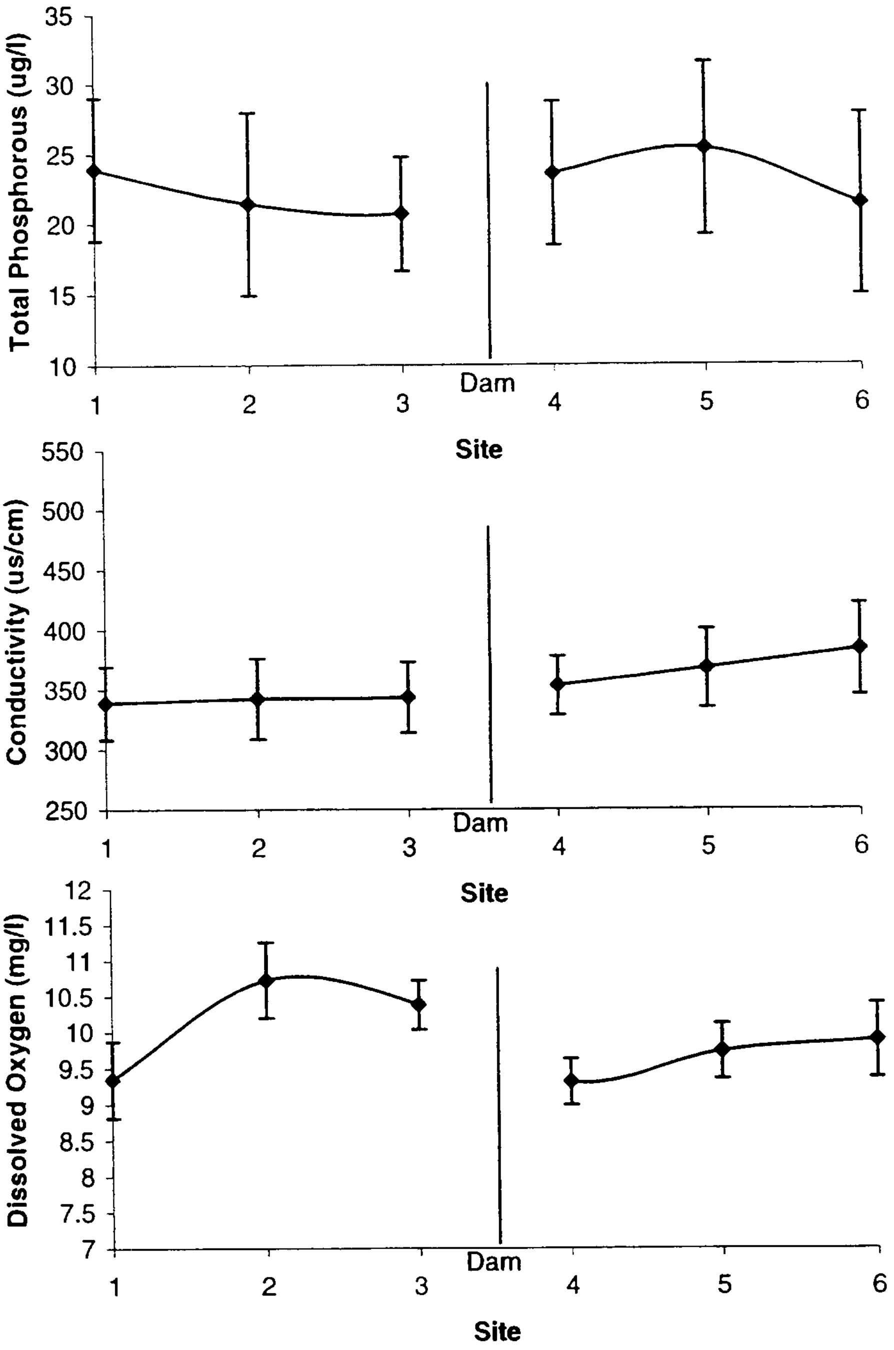


Figure 6. Average trends in total phosphorous (top), conductivity (middle) and dissolved oxygen (bottom) (+- on standard error) above and below the dams.

increasing from 342 to 363 μs/cm (Table 2). ANOVA on each individual stream showed that three streams (Cedar, Dowagiac, and Manton) had significant (α=0.05) increases in conductivity downstream and one stream (Middle Branch) had a significant decrease. Dissolved oxygen (D.O.) concentrations were almost always >7.0 mg/l in our study streams (Dowagiac Creek excepted) and were not significantly different below the dams on average. Only Manton Creek had significantly (α=0.05) lower D.O. downstream (p=0.025).

Stream summer temperatures ranged from 13.0°C to 23°C in upstream sections and 15.8°C to 25°C in downstream sections (Table 2). Averaging across all streams, downstream temperatures increased significantly from 16.8°C to 19.5°C. On a site by site basis, mean summer temperature increased below the dams and did not show a return upstream temperatures, even at the furthest downstream site (Figure. 7). The estimated impact of each dam on mean summer temperature is shown in Table 3. Individual dam impacts ranged from a 0.98°C cooling downstream in the Maple River to a 5.46°C warming downstream in Manton Creek. Four streams (Dowagiac, Fish, Maple, and Prairie) fell into the low impact category (<2.15°C warming) while six streams (Boardman, Cedar, Manton, Middle, Sugar and White) were considered high impact (>2.15°C). Average temperature in the high impact streams increased from 15°C to 19°C downstream, while low impact streams increased from 19.7°C to 20.5°C (Table 4).

Macroinvertebrate Community

Macroinvertebrate family richness was similar in above and below sections (Table 5). Richness ranged from 16 to 31 taxa ($\bar{x}$=22.6 taxa) upstream and 17 to 26 taxa

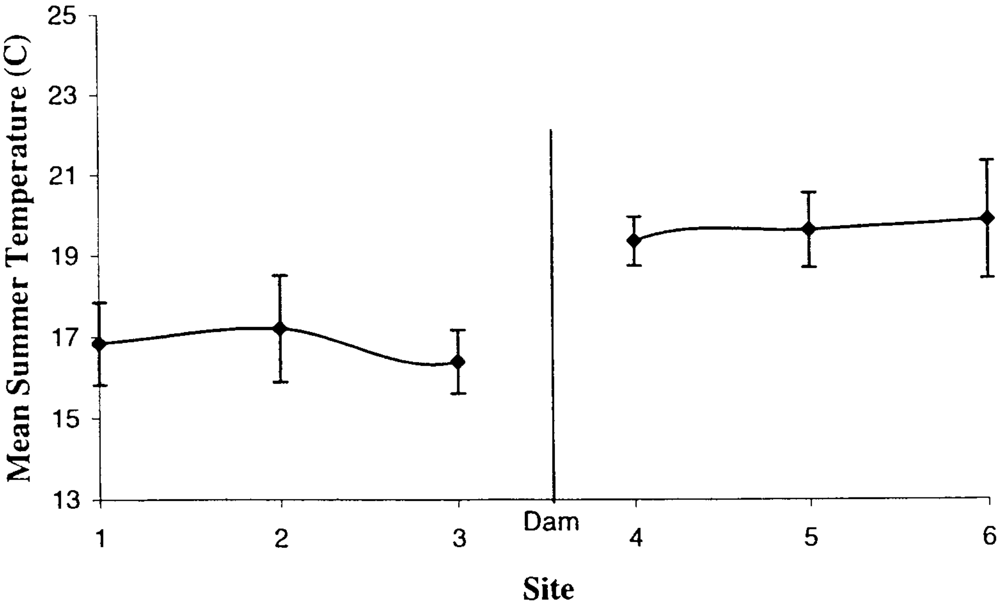


Figure 7. Trends in mean summer temperature (C) (+- one standard error) above and below the dams.

Table 3. The estimated impact of each dam on mean summer temperature (above-below).

Stream	Temp. Change (C)	Impact
Manton	-5.46	High
Boardman	-3.98	High
Middle	-3.90	High
Cedar	-3.84	High
White	-3.29	High
Sugar	-3.14	High
Dowagiac	-2.15	Low
Fish	-1.99	Low
Prairie	-0.36	Low
Maple	0.98	Low
Mean	-2.71	

Table 4. Selected community and habitat attributes (±standard error) averaged across streams in each level of temperature impact.

	Level of Temperature Impact	
Community or Habitat Attribute	**Low**	**High**
Summer Temperature (Upstream)	19.7°C (±0.81)	15°C (±0.4)
Summer Temperature (Downstream)	20.5°C (±1.2)	19°C (±0.3)
Conductivity Change (Above-Below)	-20.5 μs/cm (±16.7)	-22.5 μs/cm (±14.4)
Depth Change (Above-Below)	-11.4 cm (±4.3)	-9.5 cm (±6.3)
Macroinvertebrate Similarity	0.76 (±0.06)	0.58 (±0.032)
Fish Similarity	0.66 (±0.08)	0.47 (±0.11)
Fish Species Richness (Upstream)	19 (±2)	9 (±2)
Fish Species Richness (Downstream)	16 (±3)	16(±2)

Table 5. Macroinvertebrate community changes above and below the dams and Sorenson's similarity index values for macroinvertebrates in each stream.

	Macroinvertebrate Richness			**Plecopteran Family Richness**			**E.P.T Family Richness**			**Sorenson's**
Stream	**Above**	**Below**	**Diff.**	**Above**	**Below**	**Diff.**	**Above**	**Below**	**Diff.**	**Similarity**
Boardman	22	20	2	3	0	3	12	11	1	0.71
Cedar	20	17	3	3	0	3	10	7	3	0.54
Dowagiac	21	23	-2	1	0	1	12	8	4	0.59
Fish	26	19	7	2	1	1	13	10	3	0.76
Manton	21	22	-1	3	0	3	12	10	2	0.51
Maple	31	26	5	3	2	1	12	11	1	0.84
Middle	26	20	6	2	1	1	12	8	4	0.65
Prairie	19	21	-2	1	1	0	11	11	0	0.85
Sugar	24	20	4	2	1	1	10	9	1	0.55
White	16	17	-1	0	1	-1	7	8	-1	0.55
Mean	**22.6**	**20.5**	**2.1**	**2.0**	**0.7**	**1.3**	**11.1**	**9.3**	**1.8**	**0.65**

($\bar{x}$=20.5 taxa) downstream. Correlation analysis, on a site by site basis, indicated that none of the individual habitat parameters measured were significantly (α=0.05) related to macroinvertebrate richness (Table 6). Stepwise regression was unable to form a model with any significant relationships.

Plecopteran family richness was significantly lower below the dams and decreased from 2 to 0.70 families on average (Table 5). Plecopteran family richness tended to be lower below dams in high impact streams and was less affected in low impact streams (Figure 8). Stepwise regression indicated that plecopteran family richness was most related to mean summer temperature (p=0.009, R^2=0.16). While this relationship was significant it does not explain much of the variability in Plecopteran richness, and even the full 9-parameter model only improved the R^2 to 0.32. E.P.T. richness showed a significant (p=0.034) decline downstream. E.P.T. richness upstream and downstream had similar ranges, but different means (7 to 13, $\bar{x}$=11.1 and 7 to 11,$\bar{x}$=9.3 respectively) (Table 5). Stepwise regression indicated that stream size (width and depth) was most related to E.P.T., and that wider, deeper, areas had fewer families in these orders. Again, variability in E.P.T. data reduced the predictive power of this relationship (R^2=0.15).

Macroinvertebrate family similarity, when comparing upstream and downstream sections, was 0.65 on average (Table 5). Similarity ranged from 0.54 (Cedar Creek) to 0.84 (Maple River). Macroinvertebrate similarity was significantly related to mean summer temperature change (R^2 =0.64, p=0.0057) (Figure 9). High impact streams were less similar in terms of the macroinvertebrate community composition than low impact streams. Mean abundance and mean number of taxa calculated for each functional

Table 6. Pearson's r correlation coefficeints and corresponding p-values for macroinvertbrate/fish richness and ten habitat parameters.

	Macroinvertebrate Taxa Richness			**Fish Species Richness**		
Habitat Variable	**Pearson's r**	**P-value**	**Sample Size(n)**	**Pearson's r**	**P-value**	**Sample Size(n)**
Mean Summer Temperature	-0.15	0.2563	58	0.68	0.0001	56
Conductivity	-0.12	0.3736	58	0.58	0.0001	56
Total Phosphorous	-0.21	0.1371	51	0.44	0.0015	49
Stream Discharge	-0.15	0.4124	32	0.44	0.0160	30
Substrate Mode	-0.14	0.2921	58	0.31	0.0192	56
Substrate Mean	-0.09	0.5195	58	0.31	0.0221	56
Stream Width	-0.03	0.8309	58	0.27	0.0478	56
Water Depth	-0.15	0.2550	58	0.03	0.8242	56
Velocity	0.15	0.4239	32	-0.19	0.3163	30
Dissolved Oxygen	0.23	0.0846	57	-0.25	0.0690	55

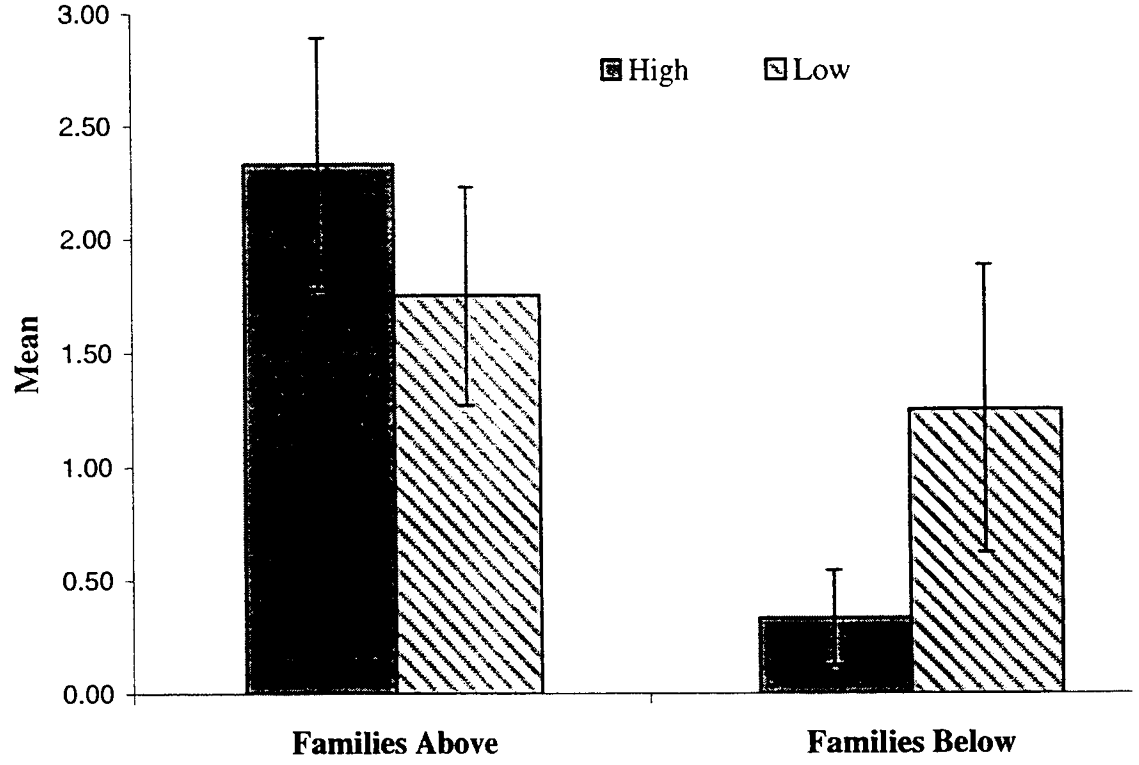


Figure 8. Average plecopteran family richness (+- one standard error) above and below the dams in high and low impact streams

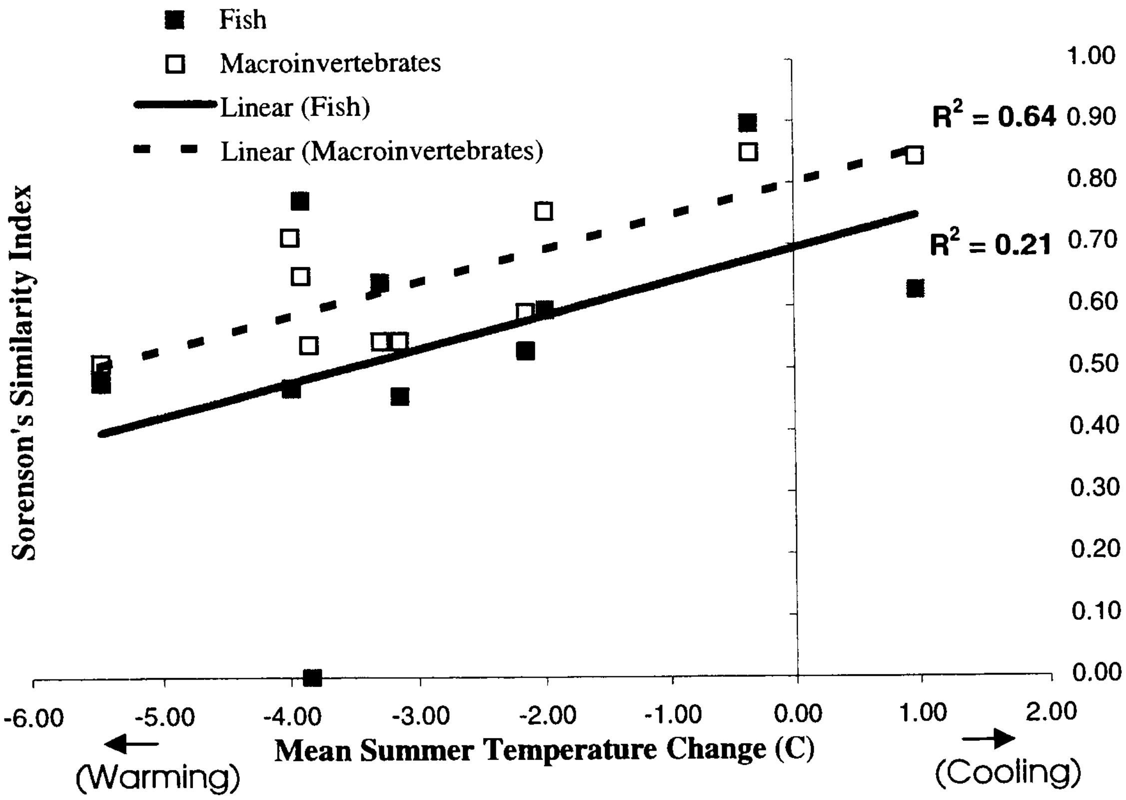


Figure 9. Influence of mean summer temperature change (i.e dam impact) on the similarity of macroinvertebrates and fish, above and below the dams. The data, linear functions and corresponding R-square values are shown.

feeding group in each stream section (above and below) are shown in tables 7 and 8 respectively. Analysis of changes in functional feeding groups below the dams showed no significant changes in predators, scrapers or shredders. Gathering collectors were similar at upstream sites and declined with increasing distance below the dams. Filtering collectors showed an increase just below the dams and then also decreased further downstream, but these data were highly variable among different streams and sites (Figure 10).

Fish Community Analysis

Fish species richness across all streams was higher downstream of the dams than upstream (Table 9). Richness ranged between 4 and 22 species upstream ($\bar{x}$=13.1 species) and between 10 and 25 species downstream ($\bar{x}$=15.9 species). Across all sites, fish species richness was most correlated with mean summer temperature, conductivity and total phosphorous (Pearson's r=0.68, 0.58 and 0.44; p=0.0001, 0.0001, and 0.0015 respectively) (Table 6). Stepwise regression indicated that temperature was the best single predictor of fish species richness on a site by site basis (p=0.0001, R^2=0.49). Further analysis on the relationship between changes in fish species richness and changes in selected habitat parameters (temperature, conductivity, dissolved oxygen, phosphorous, and depth) below the dams, using stepwise regression, showed that changes in fish richness were most related to changes in mean summer temperature (p=0.003, R^2=0.79) (Figure 11). When the streams were separated based on level of impact, high impact streams (>2.15°C increase downstream) had significantly more species downstream, and low impact streams showed no significant difference between sections (<2.15°C increase

Table 7. Average abundance (#/square meter) and standard error estimates of functional feeding groups for macroinvertebrates above and below the dams.

	Total		Filtering/Collecters		Gathering/Collecters		Predators		Scrapers		Shredders	
Stream	**Above**	**Below**	**Above**	**Below**	**Above**	**Below**	**Above**	**Below**	**Above**	**Below**	**Above**	**Below**
Boardman	**10102.9**	**9104.9**	**1121.4**	**2294.2**	**7654.3**	**4475.3**	**432.1**	**308.6**	**812.8**	**1944.4**	**82.3**	**82.3**
stderr	3769.4	1732.9	536.2	1184.3	2494.9	1937.7	246.9	145.9	537.6	468.4	27.2	37.1
Cedar	**10298.4**	**7222.2**	**4043.2**	**3071.0**	**4856.0**	**2206.8**	**92.6**	**262.4**	**864.2**	**138.9**	**442.4**	**1543.2**
stderr	2522.3	3086.4	1416.7	3009.3	525.4	1651.2	47.1	262.3	492.2	46.3	142.9	1265.4
Dowagiac	**8446.5**	**1687.2**	**2211.9**	**154.3**	**4578.2**	**555.6**	**51.4**	**72.0**	**1327.2**	**833.3**	**277.8**	**72.0**
stderr	2653.0	558.5	1119.1	17.8	2728.8	377.2	10.3	44.8	312.2	266.1	158.4	10.3
Fish	**18909.5**	**6018.5**	**6070.0**	**442.4**	**9722.2**	**4845.7**	**1111.1**	**174.9**	**1831.3**	**524.7**	**174.9**	**30.9**
stderr	9579.4	1871.3	4347.6	235.3	3840.9	1700.7	687.4	145.1	832.3	257.0	67.5	17.8
Manton	**10936.2**	**8549.4**	**4012.4**	**771.6**	**5514.4**	**5946.5**	**401.2**	**51.4**	**565.8**	**1749.0**	**442.4**	**30.9**
stderr	4352.0	5043.5	3919.8	556.7	1245.5	4936.2	263.7	37.1	520.2	523.9	273.9	30.9
Maple	**13323.1**	**9104.9**	**6707.8**	**2407.4**	**5637.9**	**4639.9**	**288.1**	**432.1**	**514.4**	**524.7**	**174.9**	**1100.8**
stderr	5667.6	3785.9	3321.1	1101.2	2919.0	2352.9	98.1	355.1	353.9	355.1	131.4	905.5
Middle	**9084.4**	**4567.9**	**1131.7**	**1790.1**	**7263.4**	**1831.3**	**92.6**	**216.1**	**288.1**	**699.6**	**308.6**	**30.9**
stderr	1323.5	1794.1	594.8	1340.5	1782.3	57.3	17.8	47.1	20.6	422.6	200.8	17.8
Prairie	**6471.2**	**9022.6**	**2242.8**	**5607.0**	**3189.3**	**2458.9**	**113.2**	**102.9**	**730.5**	**812.8**	**195.5**	**41.2**
stderr	2105.4	5240.6	2104.4	4108.4	225.6	784.3	10.3	37.1	185.5	370.5	91.4	27.2
Sugar	**5967.1**	**8374.5**	**792.2**	**3096.7**	**3631.7**	**4269.6**	**277.8**	**133.7**	**1059.7**	**740.7**	**205.8**	**133.7**
stderr	1209.4	2255.2	344.8	439.9	813.5	1940.8	170.0	51.4	314.9	357.7	131.4	133.7
White	**9876.5**	**7361.1**	**1810.7**	**2191.4**	**7139.9**	**4475.3**	**51.4**	**154.3**	**586.4**	**416.7**	**288.1**	**123.5**
stderr	2702.4	4089.5	965.3	2160.5	2261.9	1604.9	51.4	123.5	338.6	169.8	161.7	30.9
Mean	**10341.6**	**7101.3**	**3014.4**	**2182.6**	**5918.7**	**3570.5**	**291.2**	**190.8**	**858.0**	**838.5**	**259.3**	**318.9**

Table 8. Average number of families (standard error) in each functional feeding group for macroinvertebrates above and below the dams.

	Filtering/Collecters		**Gathering/Collecters**		**Predators**		**Scrapers**		**Shredders**	
Stream	**Above**	**Below**	**Above**	**Below**	**Above**	**Below**	**Above**	**Below**	**Above**	**Below**
Boardman	2.3(0.9)	3(0.6)	5.3(0.3)	6(0)	3(1)	1.3(0.3)	1.7(0.3)	2(1)	1.3(0.3)	1(0)
Cedar	3.3(0.7)	2.5(0.5)	5.3(0.3)	3(1)	1.7(0.3)	2.5(2.5)	1.3(0.3)	1(0)	3(1)	2(0)
Dowagiac	2(0.6)	2.7(0.3)	5(1)	1.3(0.3)	1(0)	1(0.6)	3.3(0.7)	4.3(0.9)	1.7(0.3)	1.3(0.3)
Fish	2.3(0.9)	2.3(1.2)	5.3(0.3)	4(1.2)	3.7(0.9)	1(0.6)	2.7(0.3)	1.3(0.3)	2(0.6)	1(0.6)
Manton	2.7(0.9)	2.7(1.2)	5(1.2)	3.3(0.3)	2(0.6)	1.7(1.2)	0.7(0.3)	2.7(0.3)	2(0)	0.7(0.7)
Maple	4(1)	3.7(0.3)	4.3(1.2)	4.3(0.9)	3.3(1.2)	2(0)	1.7(0.3)	2.7(0.3)	1.7(0.9)	2.3(0.7)
Middle	3(0.6)	1.3(0.7)	5.3(0.3)	3.3(0.7)	2.3(0.9)	2.7(0.3)	2.3(0.7)	3.3(1.2)	2(0)	0.7(0.3)
Prairie	2(0.6)	2(0.6)	3.3(0.3)	5(1)	1.7(0.3)	2.7(0.9)	2.7(0.9)	2.7(0.3)	1(0)	0.7(0.3)
Sugar	3(0.6)	2.7(0.3)	3.7(0.3)	3.4(0.9)	2.7(0.3)	1.7(0.7)	2.3(0.7)	4(0.6)	1.7(0.7)	0.3(0.3)
White	2.3(0.7)	1.5(0.5)	4(0.6)	3.5(0.5)	0.7(0.7)	2(1)	2.7(1.3)	2.5(0.5)	1(0)	2(0)
Mean	2.7	2.4	4.6	3.7	2.2	1.9	2.1	2.7	1.7	1.2
Stderr	0.2	0.2	0.2	0.4	0.3	0.2	0.2	0.3	0.2	0.2

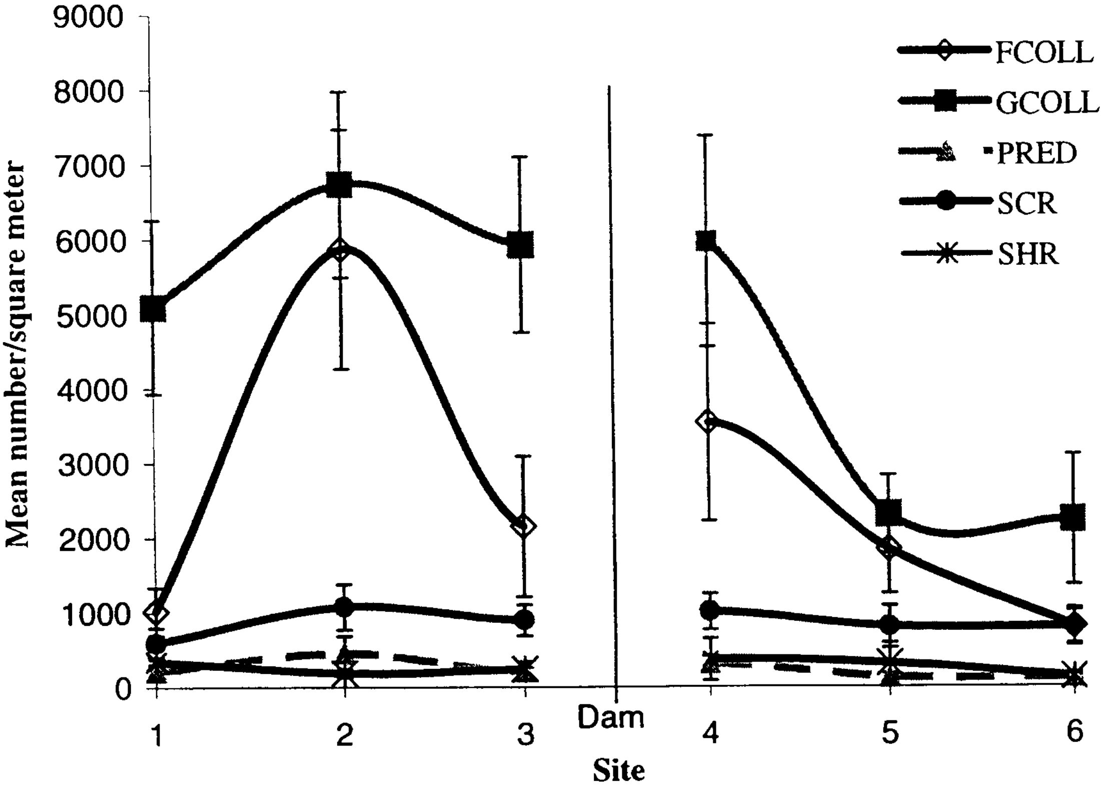


Figure 10. Average trends in functional feeding group abundances above and below the dams (FCOLL=Filtering/Collectors GCOLL=Gathering/Collectors, PRED=Predators, SCR=Scrpaers SHR=Shredders).

Table 9. Fish community changes above and below the dams and Sorenson's similarity index values for fish in each stream.

Stream	**Fish Species Richness**			**Mean Fish Length (mm)**			**Sorenson's**
	Above	**Below**	**Diff.**	**Above**	**Below**	**Diff.**	**Similarity**
Boardman	4	13	-9	125.3	155.1	-29.8	0.47
Cedar	5	10	-5	106.2	90.9	15.3	0.00
Dowagiac	20	14	6	88.0	174.7	-86.7	0.53
Fish	22	25	-3	72.4	70.7	1.7	0.60
Manton	7	18	-11	94.6	87.9	6.6	0.48
Maple	21	11	10	111.3	91.4	20.0	0.63
Middle	15	16	-1	83.3	82.8	0.5	0.77
Prairie	14	15	-1	80.5	76.6	4.0	0.90
Sugar	13	22	-9	85.5	111.8	-26.2	0.46
White	10	15	-5	143.4	87.5	55.9	0.64
Mean	**13.1**	**15.9**	**-2.8**	**99.1**	**102.9**	**-3.9**	**0.55**

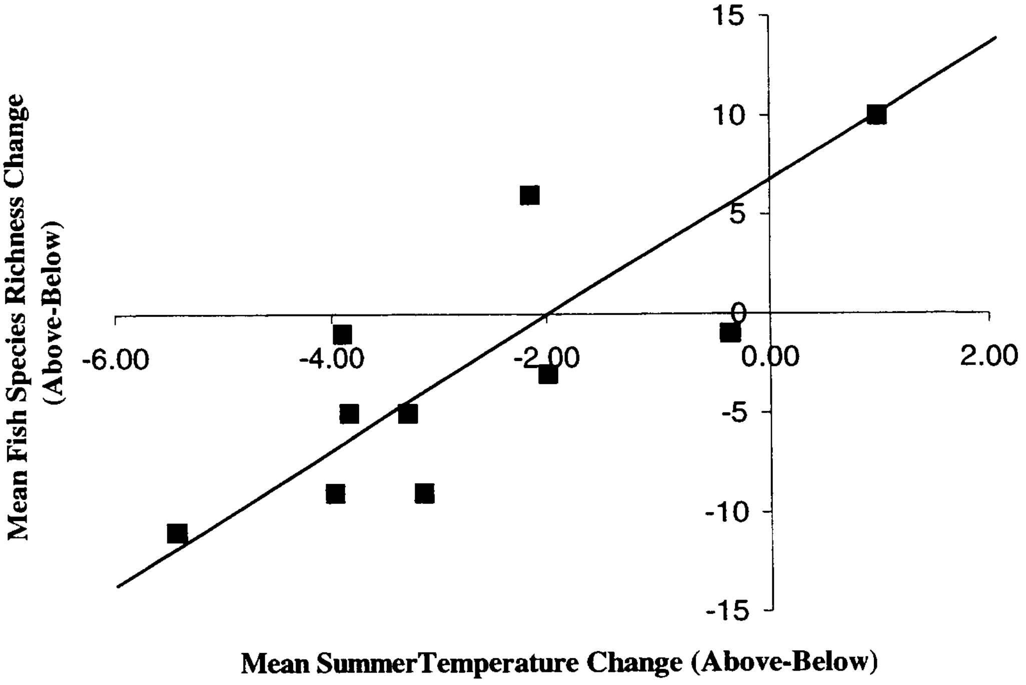


Figure 11. Influence of changes in mean summer temperature (dam impact) on changes in fish species richness (R-square=0.78).

downstream) (Figure 12). Changes in conductivity alone were not significantly related to changes in fish species richness below the dams ($p=0.62$, Adj.$R^2=-0.09$), but when added to the model with temperature changes it was a significant parameter and improved the model fit substantially ($p=0.0005$, $R^2=0.98$).

Fish community similarity, comparing above and below sections, was 0.55 on average and ranged from 0 (Cedar Creek) to 0.90 (Prairie Creek) (Table 9). Fish similarity was not significantly related to changes in mean summer temperature ($p=0.28$, Adj.$R^2=0.089$) (Figure 9) or conductivity ($p=0.16$, Adj.$R^2=0.134$). Stepwise regression indicated that changes in mean depth were most related to fish community similarity ($p=0.027$, Adj.$R^2=0.59$) (Figure 13). Streams that were significantly deeper downstream had lower similarity values.

Mean length of the fish community increased from 99 to 103 mm on average, but this was not a significant increase (Table 9). The average length of the fish community ranged from 72.4 to 143.4 mm upstream and 70.7 to 174.7 mm downstream. Changes in mean community length were not related to any of the habitat characteristics used in the regression analysis.

Density of Target Fish Species

Across all streams, total trout density averaged 921 fish/ha upstream and 280 fish/ha downstream, a 70% decrease (Table 10). Trout and slimy sculpin, when present, were always found upstream of the dams and were found at varying densities downstream. Brook trout were present in six of the ten study streams and were not found below the dams in three of those streams. Brook trout density decreased below the dams

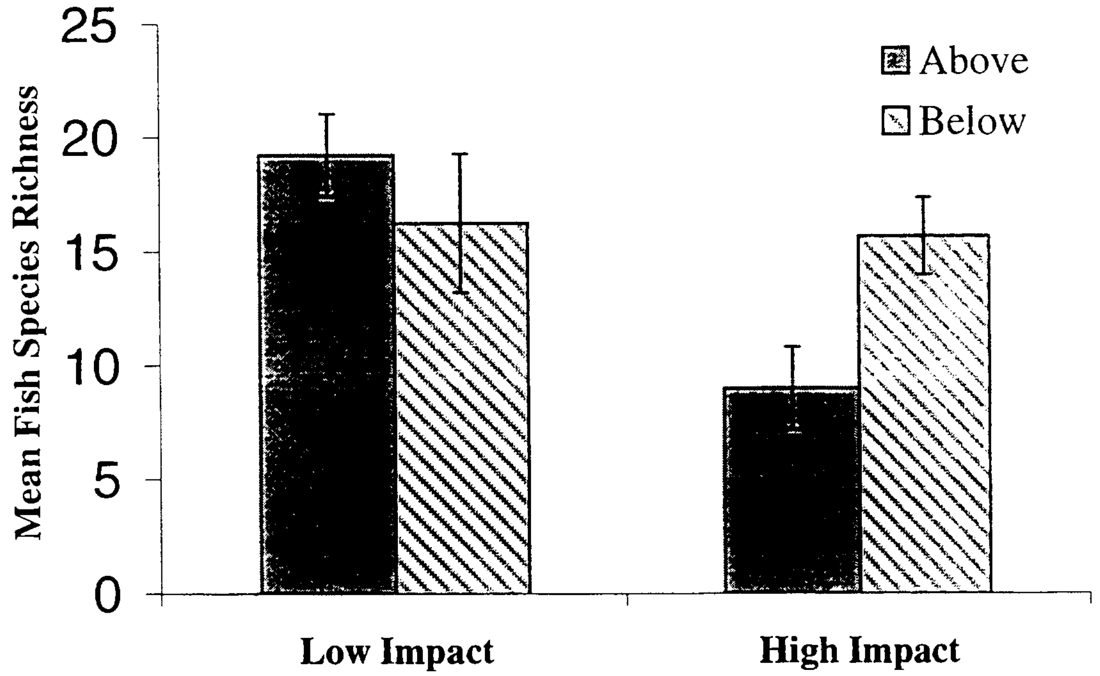


Figure 12. Mean fish species richness above and below the dams for low (<2.15 C increase in temperature) and high (>2.15 C increase) impact streams (+/- one standard error).

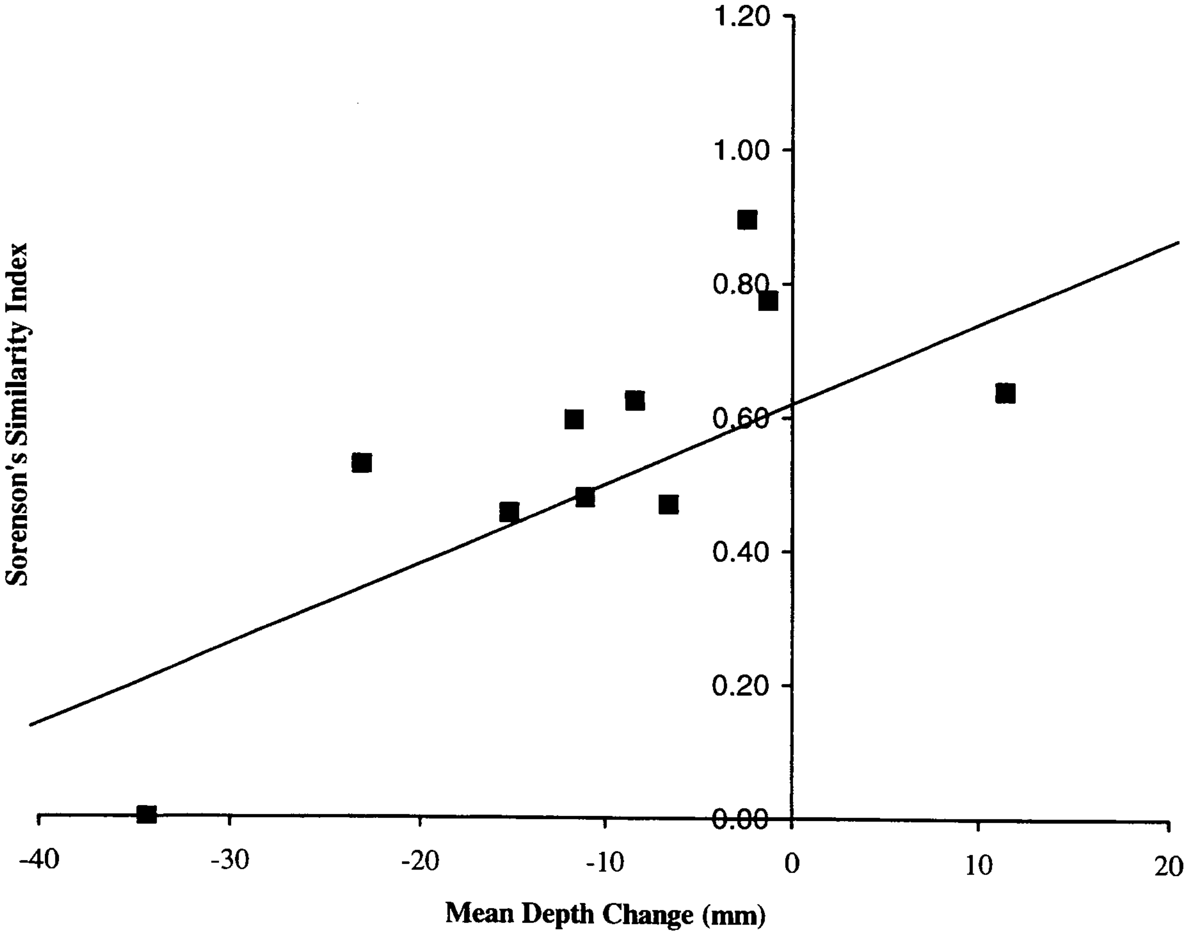


Figure 13. Influence of changes in mean depth on Sorenson's similarity index values for fish above and below the dams (R-square=0.59).

Table 10. Density (fish/ha) and standard error estimates for the five target fish species, above and below the dams, in the ten study streams.

	BROOK TROUT		BROWN TROUT		RAINBOW TROUT		SLIMY SCULPIN		MOTTLED SCULPIN		TOTAL TROUT	
STREAM	Above	Below	Above	Below	Above	Below	Above	Below	Above	Below	Above	Below
Boardman	**234**	**18**	**585**	**342**	**0**	**0**	**1124**	**277**	**0**	**0**	**819**	**361**
stderr	44	13	125	111	0	0	432	144	0	0	86	108
Cedar	**191**	**0**	**872**	**0**	**0**	**0**	**2264**	**0**	**0**	**0**	**1064**	**0**
stderr	76	0	185	0	0	0	981	0	0	0	133	0
Dowagiac	**0**	**0**	**73**	**0**	**0**	**0**	**0**	**0**	**1079**	**0**	**73**	**0**
stderr	0	0	67	0	0	0	0	0	540	0	67	0
Fish	**0**	**0**	**23**	**0**	**0**	**0**	**0**	**0**	**52**	**0**	**23**	**0**
stderr	0	0	23	0	0	0	0	0	47	0	23	0
Manton	**2416**	**19**	**2878**	**598**	**0**	**0**	**1475**	**0**	**11**	**22**	**5294**	**617**
stderr	1113	19	466	162	0	0	307	0	11	22	1561	178
Maple	**497**	**84**	**147**	**749**	**23**	**65**	**824**	**990**	**0**	**4**	**668**	**897**
stderr	408	9	147	615	23	29	337	399	0	4	377	648
Middle	**4**	**0**	**214**	**37**	**0**	**0**	**0**	**0**	**556**	**29**	**217**	**37**
stderr	4	0	62	37	0	0	0	0	164	15	65	37
Prairie	**0**	**0**	**76**	**55**	**136**	**47**	**0**	**0**	**482**	**347**	**213**	**103**
stderr	0	0	38	29	71	39	0	0	277	248	107	68
Sugar	**0**	**0**	**558**	**0**	**0**	**0**	**0**	**0**	**60**	**0**	**558**	**0**
stderr	0	0	306	0	0	0	0	0	34	0	306	0
White	**115**	**0**	**170**	**782**	**0**	**0**	**0**	**0**	**1049**	**338**	**284**	**782**
stderr	68	0	59	372	0	0	0	0	186	21	127	372
Mean	346	12	560	256	16	11	569	127	329	74	921	280
stderr	236	8	272	105	14	8	258	100	139	45	497	113

from 346 to 12 fish/ha. Brown trout were present in all ten study streams and were not found below three of the dams. Brown trout density decreased from 560 to 256 fish/ha. Rainbow trout were only present in two of the study streams precluding useful analysis. Slimy sculpin were present in four of the study streams and were not found below the dams in two of those streams. Slimy sculpin density decreased from 569 to 127 fish/ha. Mottled sculpin were caught in eight of the study streams and were not found below the dams in three of those streams. Mottled sculpin density decreased from 329 to 74 fish/ha downstream. One study stream (Maple River) had mottled sculpin only below the dam.

Temperature was the most highly correlated habitat variable for both Lnbrook and Lnbrown trout density, across all streams (Pearson's $r= -0.66$ and -0.64; $p=0.0001$ and 0.0001 respectively) (Table 11). Based on the results of stepwise regression analysis, the best single parameter for predicting brook (Lnbrook) and brown (Lnbrown) trout density was also mean summer temperature ($p=0.0001$, $R^2=0.57$; $p=0.0001$ $R^2=0.43$ respectively). For both brook and brown trout, no additional habitat parameters were significantly related to density and adding them to the model did not improved the model fit substantially (Appendix A). To look at the relationship between mean summer temperature and trout abundance further, I used a mixed modeling analysis of covariance (ANCOVA) procedure where stream was treated as a random effect. Each target species was analyzed separately to develop a predictive model for density given a mean summer temperature. Analyses were done on average densities within each position (above or below the dam).

An unequal slopes model and a full model relating Lnbrook with stream position

Table 11. Pearson's correlation coefficients and corresponding p-values and N (numer of samples) that relates the Ln(density +1) estimates of the four most abundant target fish species with ten habitat parameters and two measurements of community richness.

	LnBrook			LnBrown			LnSlimy			LnMottled		
Habitat Variable	**Pearson's r**	**P-value**	**N**	**Pearson's r**	**P-value**	**N**	**Pearson's r**	**P-value**	**N**	**Pearson's r**	**P-value**	**N**
Mean Summer Temperature	-0.66	0.0001	56	-0.64	0.0001	56	-0.56	0.0001	56	-0.065	0.63	56
Conductivity	-0.61	0.0001	56	-0.44	0.0007	56	-0.62	0.0001	56	0.266	0.05	56
Fish Species Richness	-0.57	0.0001	56	-0.30	0.0262	56	-0.56	0.0001	56	0.297	0.03	56
Total Phosphorous	-0.49	0.0004	49	-0.48	0.0004	49	-0.42	0.0025	49	0.187	0.20	49
Stream Discharge	-0.29	0.1176	30	-0.55	0.0018	30	-0.23	0.2319	30	-0.292	0.12	30
Substrate Mode	-0.20	0.1339	56	-0.28	0.0377	56	-0.23	0.0874	56	-0.032	0.82	56
Stream Width	-0.20	0.1483	56	-0.24	0.0766	56	-0.02	0.8554	56	-0.156	0.25	56
Water Depth	-0.16	0.2499	56	-0.34	0.0104	56	0.03	0.8291	56	-0.315	0.02	56
Substrate Mean	-0.15	0.2606	56	-0.21	0.1203	56	-0.18	0.1850	56	-0.024	0.86	56
Velocity	-0.03	0.8629	30	-0.26	0.1670	30	-0.10	0.5929	30	-0.071	0.71	30
Macroinvertebrate Richness	0.26	0.0562	56	0.19	0.1684	56	0.26	0.0520	56	-0.017	0.90	56
Dissolved Oxygen	0.35	0.0086	55	0.28	0.0353	55	0.35	0.0083	55	-0.155	0.26	55

(above or below the dams), mean summer temperature and an interaction term were run on the data from the six brook trout streams. The slopes were not equal to zero but they were equal to each other, meaning that the interaction term was not significant (i.e. above and below sections had equal slopes). Position was also found to be non-significant and so a reduced model was selected. The final mixed model relating Lnbrook to mean summer temperature is:

$$\text{Lnbrook}=17.9-0.915(T), \qquad [4]$$

where Lnbrook is the $\log_e$ transformed mean brook trout density and T is the mean summer temperature. No brook trout were found in sites with mean summer temperatures >19 °C (Figure 14).

Due to the large number of sites where no brook trout were caught, we employed logistic regression to model the relationship between mean summer temperature and brook trout presence, using all sites. The logistic curve declines sharply around 17 °C, indicating that brook trout are less likely to be found in streams with mean summer temperatures above 17 °C (p=0.0001, odds ratio=0.4, 95% CI =0.25(L) and 0.64(U)) (Figure 15). The odds ratio can be interpreted as indicating how quickly the probability of brook trout presence in a stream changes with increasing temperature. An odds ratio of 0.4 indicates that for every increase in temperature of 1 °C the odds of having brook trout in a stream are 0.4 of what they were at the cooler temperature. The fact that the 95% CI on the odds ratio does not include one, is further evidence of the significance of this relationship, because an odds ratio of one would indicate no change in the probability of brook trout presence with increasing temperature. The equation for the relationship

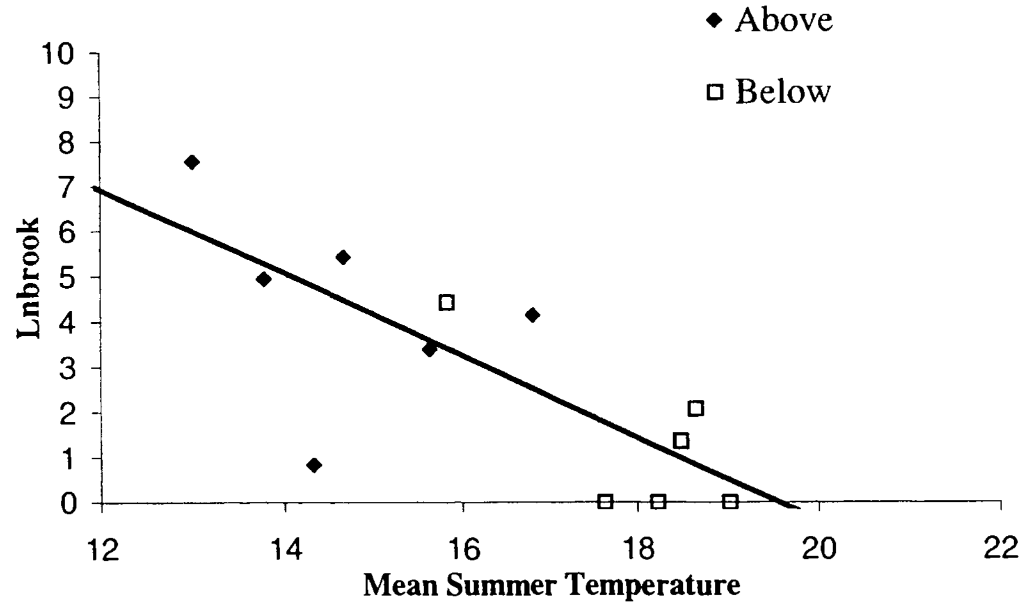


Figure 14. Influence of mean summer temperature (C) on mean LnBrook density in the six streams containing brook trout.

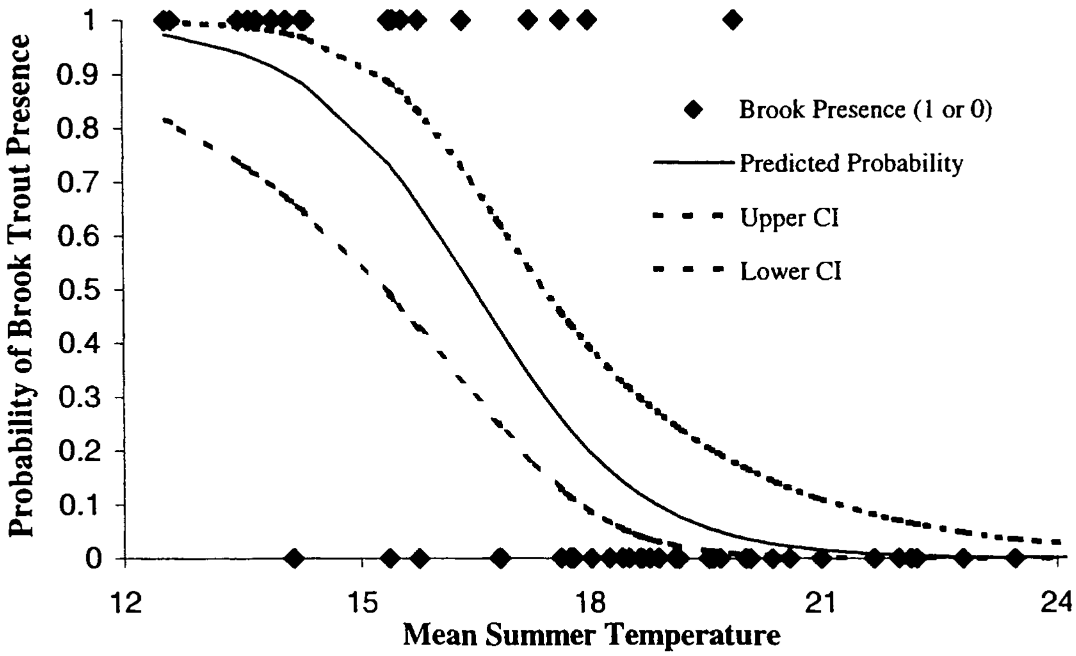


Figure 15. Logistic curve showing how the probability of brook trout presence declines with increasing mean summer temperatures in all ten study streams.

between brook trout presence and mean summer temperature is:

$$P(Brook)= \exp(15.08-0.91(T))/[1+\exp(15.08-0.91(T))], \qquad [5]$$

where P(Brook) is the probability of brook trout being present and T is the mean summer temperature. Stepwise logistic regression revealed that conductivity ($p=0.0067$, odds ratio=0.95, CI=0.92(L) and 0.99(U)) and mean summer temperature ($p=0.0066$, odds ratio=0.41, CI=0.21(L) and 0.79(U)) together were better predictors of brook trout presence in a. stream. The probability of brook trout presence declines with increasing summer temperatures at a faster rate when conductivity values are higher (Figure 16). The equation for this relationship is:

$$P(Brook)= \exp(29.7-0.89(T)-0.049(C))/[1+\exp(29.7-0.89(T)-0.049(C))], \qquad [6]$$

where P(Brook) is the probability of brook trout being present, T is the mean summer temperature, and C is the mean conductivity. Small changes in temperature have more of an influence on brook trout presence than small changes in conductivity, as indicated by the odds ratio values for each habitat parameter. The interaction of the two parameters, however, gives a more specific picture of what type of habitats brook trout prefer. In general, brook trout prefer habitats with mean summer temperatures below 20°C and conductivity values below 300μs/cm (Figure 17).

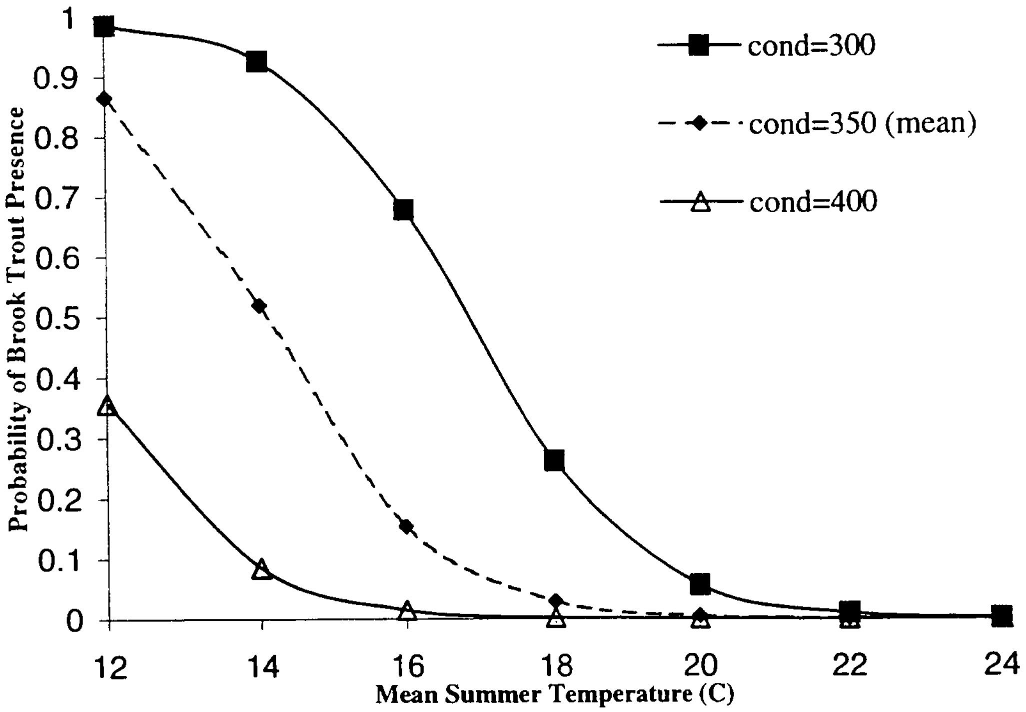


Figure 16. Logistic curve showing how the probability of brook trout presence declines with increasing mean summer temperature at three conductivity values.

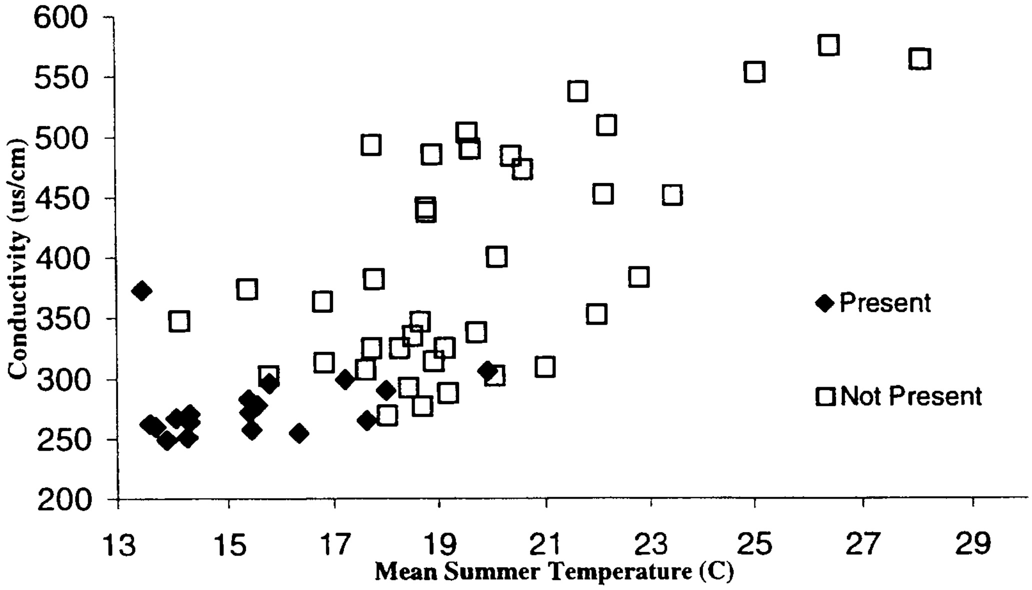


Figure 17. Pattern of presence and absence of brook trout in study streams with increasing mean summer temperature and conductivity.

A similar approach was used to model Lnbrown trout density. As with brook trout, the slopes relating Lnbrown to temperature were not equal to zero, but were equal to each other. Position was also not a significant parameter for brown trout density and so was dropped from the model. The final mixed model which relates Lnbrown and mean summer temperature is:

$$\text{Lnbrown}=14.73-0.602(T), \qquad [7]$$

where Lnbrown is the $\log_e$ transformed mean brown trout density and T is the mean summer temperature.

Brown trout tended to persist below the dams more often than brook trout, but were present at very low densities when mean summer temperatures were >20°C (Figure 18). Stepwise logistic regression revealed that mean summer temperature was the most important parameter for predicting brown trout presence in a stream ($p=0.0007$, odds ratio=0.58, 95% CI=0.42(L) and 0.8(U)). The equation relating brown trout presence and temperature is:

$$P(\text{Brown})= \exp(10.9-0.55(T))/[1+\exp(10.9-0.55(T))], \qquad [8]$$

where P(Brown) is the probability of brown trout being present and T is the mean summer temperature. Brown trout are less likely to be present in a stream with mean summer temperatures >20°C (Figure 19). Increases in mean summer temperature affect brown trout and brook trout similarly, but brown trout do not decline as sharply and persist at higher temperatures. For one degree increase in summer temperature, the odds of brown trout presence will be 0.58 of what it was at the lower temperature. Conductivity was not an important factor for modeling brown trout presence.

Using stepwise regression, the best parameter for predicting natural log of slimy

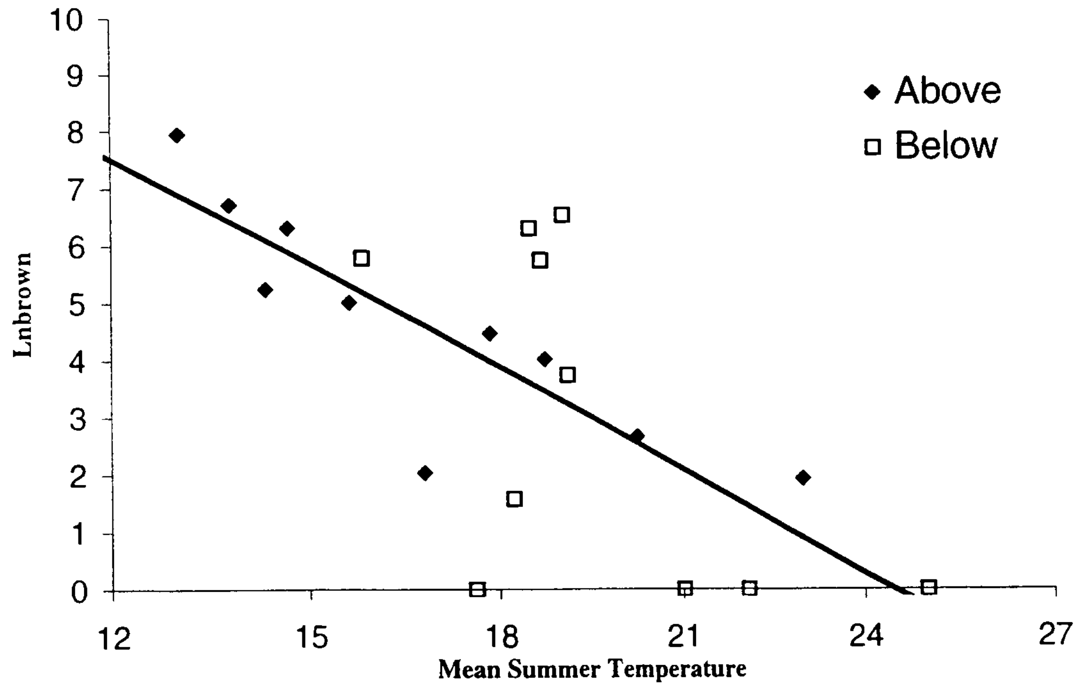


Figure 18. Influence of mean summer temperature (C) on LnBrown density in all ten study streams.

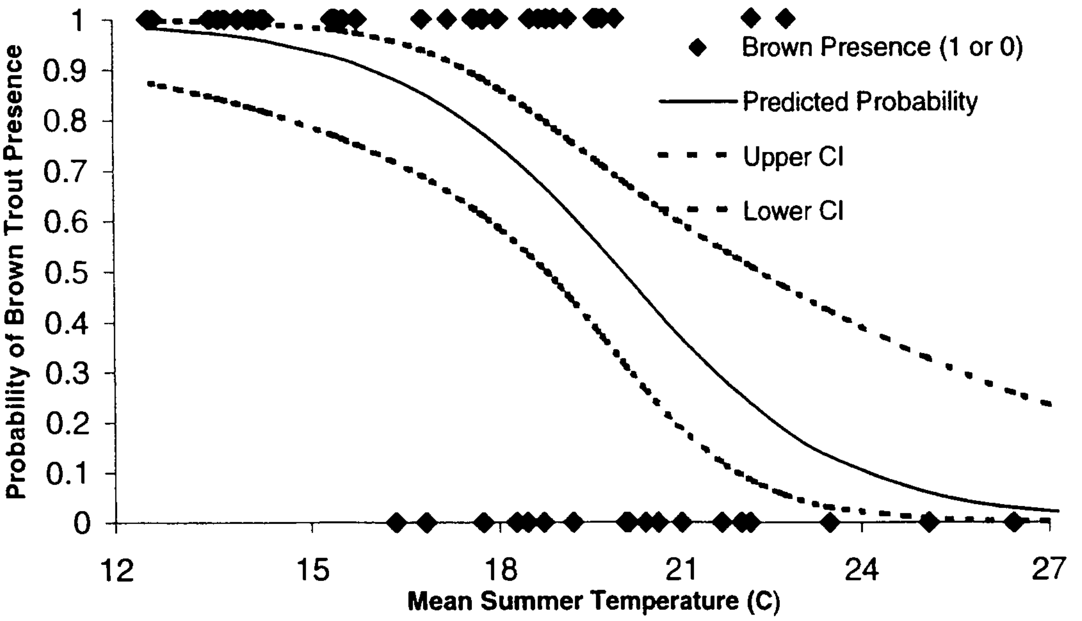


Figure 19. Logistic curve showing how the probability of brown trout presence declines with increasing mean summer temperatures in all study streams.

sculpin abundance (Lnslimy) was also mean summer temperature ($p=0.0001$, $R^2=0.45$). Including conductivity in the model with mean summer temperature only improved the model slightly ($p=0.068$ and 0.011 respectively, $R^2=0.48$). The mixed model analysis on the log transformed density data indicated the slopes were not equal to zero and no interaction terms were significant. Position above or below the dams was also not significant and therefore was dropped from the model. The final reduced mixed model relating Lnslimy to mean summer temperature is:

$$\text{Lnslimy}=13.89-0.655(T), \qquad [9]$$

where Lnslimy is the $\log_e$ transformed mean slimy sculpin density and T is the mean summer temperature. Figure 20, shows how this line fits the slimy sculpin density data in the study streams. It appears that other factors are likely to also play a role in determining slimy sculpin abundance.

These data appear particularly suited to logistic regression analysis, since slimy sculpin often showed very spotty distributions, but were abundant wherever they occurred. Stepwise logistic regression showed that conductivity was the single parameter most related to slimy sculpin presence ($p=0.0048$, odds ratio=0.86, 95% CI=0.77(L) and 0.95(U)). Similar to the brook trout, the probability of slimy sculpin being present in a stream declines quickly above conductivity values of $300\mu s/cm$ (Figure 21). The equation for this relationship is:

$$P(\text{Slimy})= \exp(44.6-0.15(C))/[1+\exp(44.6-0.15(C))], \qquad [10]$$

where P(Slimy) is the probability of slimy sculpin being present and C is the mean conductivity.

The natural log of mottled sculpin abundance was most related to fish species

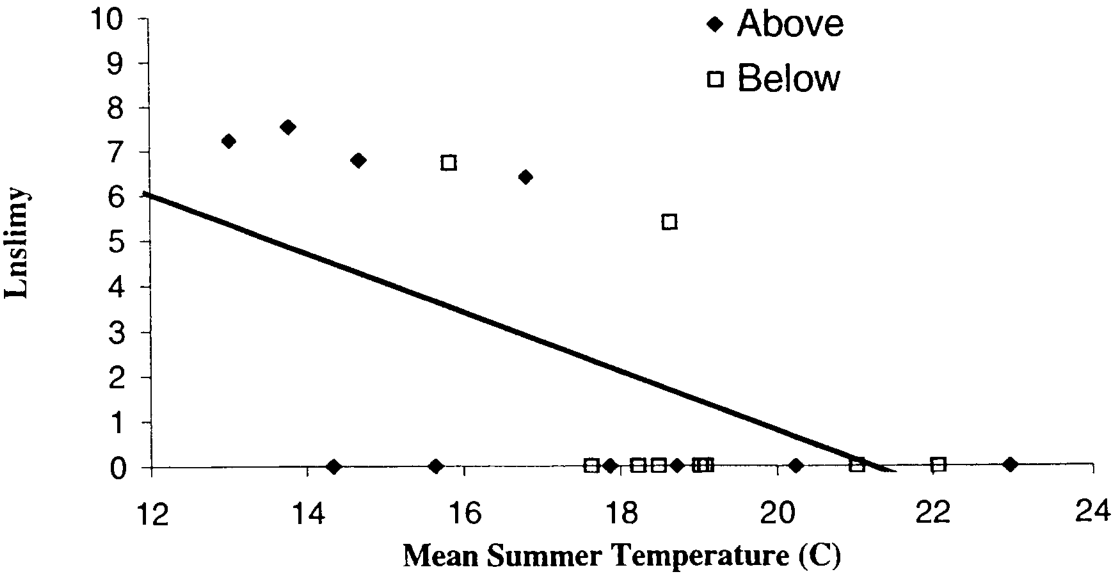


Figure 20. Influence of mean summer temperature (C) on mean LnSlimy density in all ten study streams.

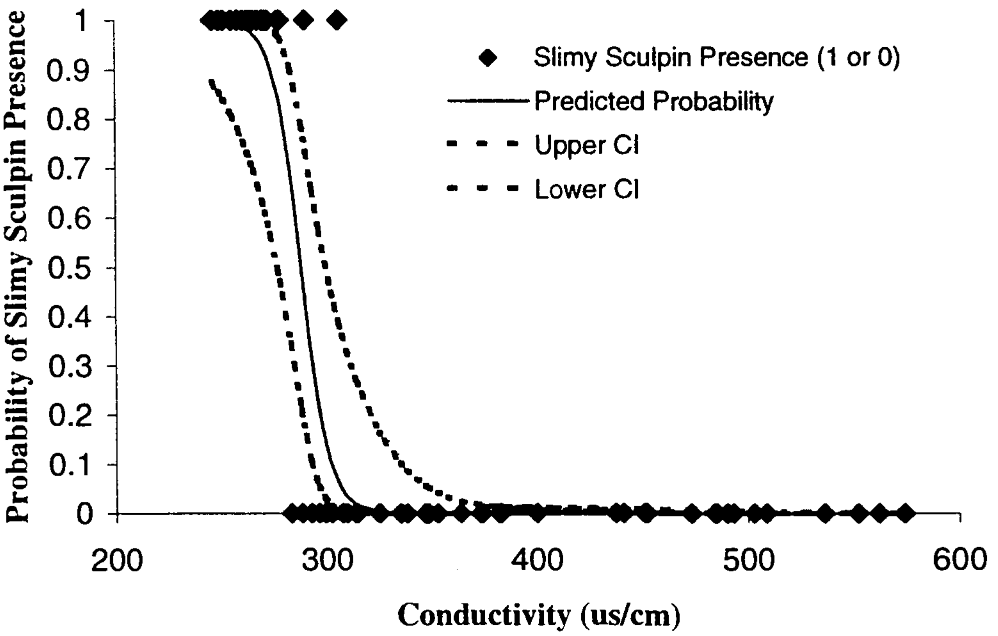


Figure 21. Logistic curve showing how the probability of slimy sculpin presence declines with increasing conductivity in all ten study streams.

richness ($p=0.015$, $R^2=0.12$), however this relationship was quite weak. Depth and Fish species richness together made the best two parameter model ($p=0.035$ and 0.013, $R^2=0.21$ respectively). Using the mixed modeling approach to develop a predictive model for mottled sculpin was a bit more difficult. Since mottled sculpin were not strongly related to any of the parameters we measured, as indicated by the correlation analysis (Table 11), I attempted to find the best model using the data available. Based on the stepwise regression results the best two parameter model should include fish species richness and depth. Using a similar approach as in the previous analyses, I first looked to see if the slopes were equal to zero and equal to each other. Depth was not a significant parameter in this analysis and so was dropped from the model. The slopes of the model with only the fish species richness data were not equal to zero but were equal to each other. Position in this case was significant and so there are two models relating mottled sculpin abundance and fish species richness.

$$\mathrm{Lnmottled}_{\mathrm{above}}=1.18+0.245(F), \qquad [11]$$

$$\mathrm{Lnmottled}_{\mathrm{below}}=-1.03+0.245(F), \qquad [12]$$

where $\mathrm{Lnmottled}_{\mathrm{above}}$ and $\mathrm{Lnmottled}_{\mathrm{below}}$ are the $\log_e$ transformed mean mottled sculpin densities for each position and F is fish species richness. Although these relationships were relatively weak, mottled sculpin densities tended to be higher above the dams and, in both positions, were higher in streams with higher overall fish diversities (Figure 22).

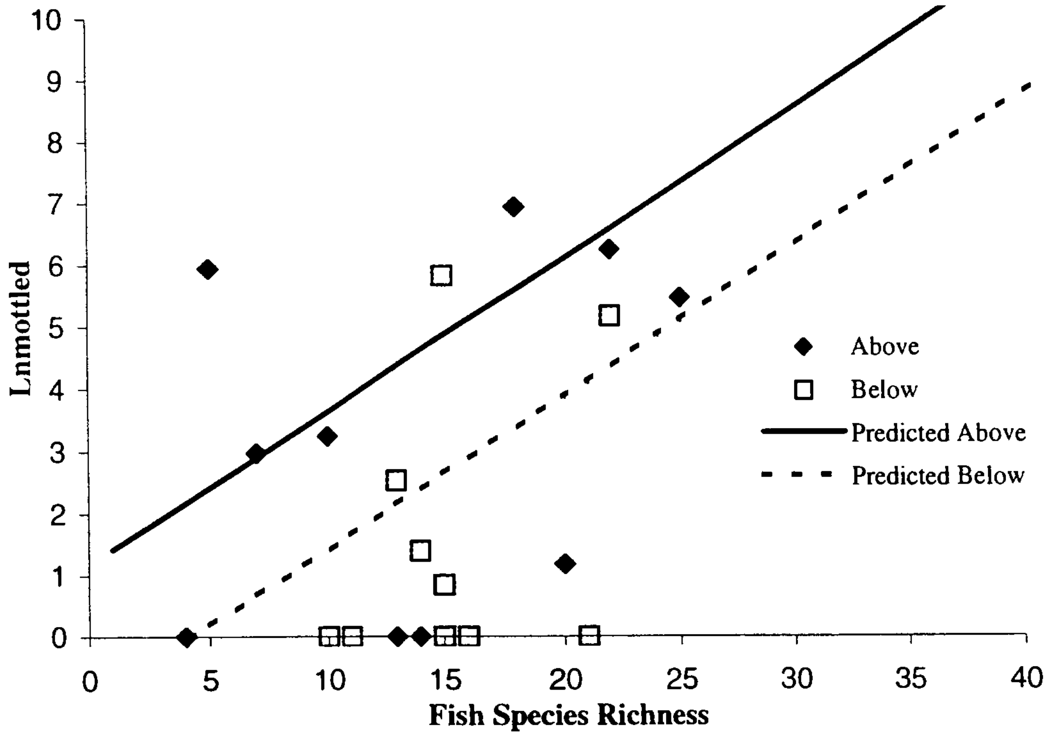


Figure 22. Influence of fish species richness on LnMottled density in above and below sections in all ten study streams.

Stepwise logistic regression showed that conductivity and depth were the two parameters most related to mottled sculpin presence (p=0.02, odds ratio=1.008, 95% CI=1.002 (L) and 1.014(U); p=0.004, odds ratio=0.92, CI=0.87 (L) and 0.98 (U) respectively). The equation for this relationship is:

$$P(\text{Mottled})=\exp(-0.44-0.083(D)+0.008(C))/[1+\exp(-0.44-0.083(D)+0.008(C))], \qquad [13]$$

where P(Mottled) is the probability of mottled sculpin being present, D is the depth and C is the conductivity. Conductivity, while significant, did not have a strong effect on mottled sculpin presence. In general, the probability of mottled sculpin presence decreased with increasing depth more slowly at higher conductivities (Figure 23). Mottled sculpin were never present in waters >50 cm deep and tended to be found at conductivity levels >250 μs/cm (Figure 24).

Trout Age and Growth Analysis

Brook trout ages ranged from zero to three over all six streams and age zero (young of the year) was the most abundant age class captured (Table 12). Brown trout ages ranged from zero to seven over all ten streams and age one was the most frequently caught age class (Table 13). Mean age was similar between positions for both species of trout.

A regression analysis was used to determine the length of brown trout at first scale formation (R_{zero}) (Figure 25). Due to the curvature in the relationship between length and scale radius exhibited by brown trout in the study streams, the R_{zero} value calculated from the regression line was negative, and so a value of zero was substituted. Regression analysis of incremental growth (fish age>1) on previous length at age and stream position, revealed that only two streams (Boardman and White) had significant differences

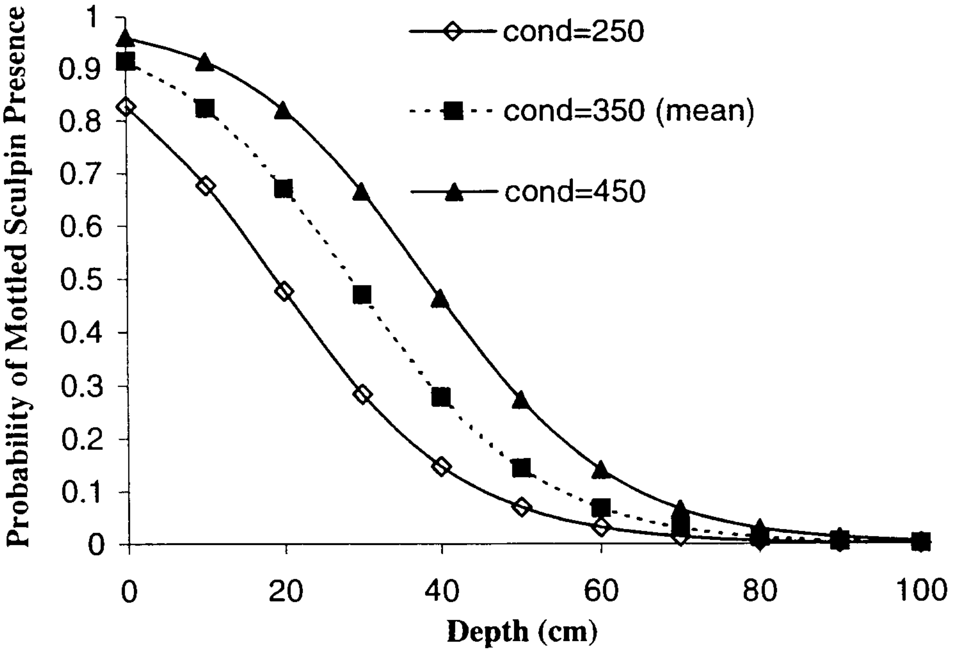


Figure 23. Logistic curve showing how the probability of mottled sculpin presence declines with increasing depth at three conductivity values.

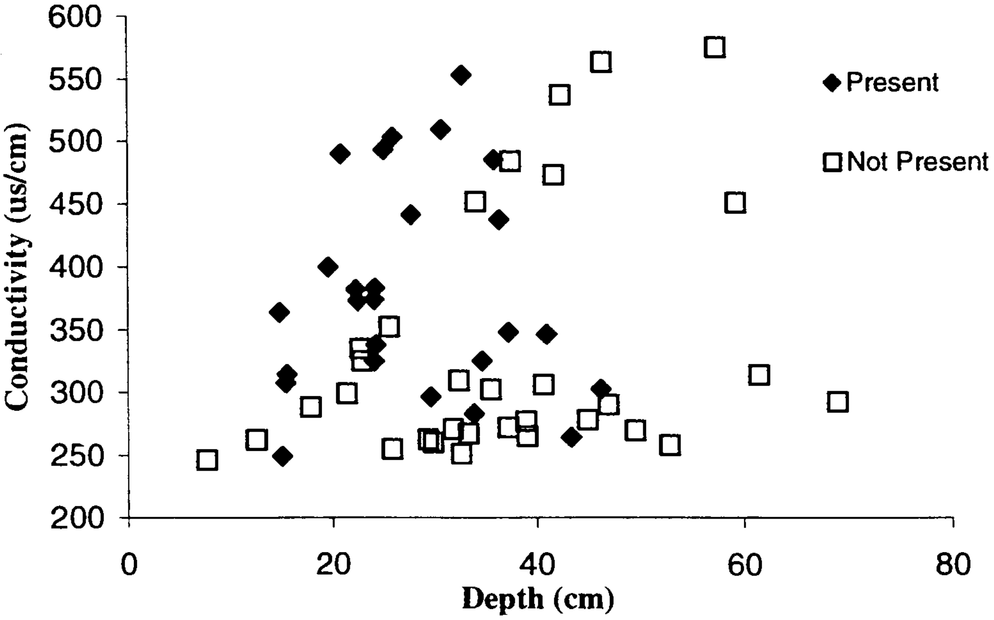


Figure 24. Pattern of presence and absence of mottled sculpin in study streams with increasing depth and conductivity.

Table 12. Number at age and mean age of brook trout, above (Top) and below (Bottom) the dams, in each stream.

(Above)	**Age**					
Stream	**0**	**1**	**2**	**3**	**Total #**	**Mean Age**
Boardman	1	11	5	1	18	1.33
Cedar	0	13	3	0	16	1.19
Manton	61	16	0	0	77	0.21
Maple	1	13	17	1	32	1.56
Middle Branch	0	1	0	0	1	1.00
White	0	2	4	1	7	1.86
Mean	10.5	9.3	4.8	0.5	25.2	1.2
Total	63	56	29	3		

(Below)	**Age**					
Stream	**0**	**1**	**2**	**3**	**Total #**	**Mean Age**
Boardman	0	2	1	0	3	1.33
Cedar	0	0	0	0	0	.
Manton	0	1	0	0	1	1.00
Maple	0	3	4	0	7	1.57
Middle Branch	0	0	0	0	0	.
White	0	0	0	0	0	.
Mean	0	1	0.83	0	1.83	1.30
Total	0	6	5	0		

Table 13. Number at age and mean age of brown trout, above (Top) and below (Bottom) the dams, in each stream.

(Above)	Age									
Stream	**0**	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**Total #**	**Mean Age**
Boardman	1	31	35	25	4	0	0	0	28	6.86
Cedar	0	27	18	14	2	0	0	1	96	1.25
Dowagiac	1	8	0	0	0	0	0	0	62	0.13
Fish	0	5	0	0	0	0	0	0	9	0.56
Manton	48	32	28	5	0	0	0	0	5	20.60
Maple	0	3	7	3	1	0	0	0	113	0.27
Middle Branch	0	0	12	7	1	1	0	0	14	3.86
Prairie	0	0	8	6	0	0	0	0	21	1.62
Sugar	0	12	12	2	0	0	0	0	14	3.00
White	0	3	7	5	2	2	0	0	26	1.92
Mean	5	12.1	12.7	6.7	1	0.3	0	0.1	38.8	4.0
Total	50	121	127	67	10	3	0	1		

(Below)	Age									
Stream	**0**	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**Total #**	**Mean age**
Boardman	15	17	38	24	6	0	1	0	101	1.93
Cedar	0	0	0	0	0	0	0	0	0	.
Dowagiac	0	0	0	0	0	0	0	0	0	.
Fish	0	0	0	0	0	0	0	0	0	.
Manton	16	22	9	1	0	0	0	0	48	0.90
Maple	3	10	16	4	2	0	0	0	35	1.77
Middle Branch	0	0	0	0	1	0	0	0	1	4.00
Prairie	0	0	5	6	0	0	0	0	11	2.55
Sugar	0	0	0	0	0	0	0	0	0	.
White	0	1	20	2	0	0	0	0	23	2.04
Mean	3.4	5	8.8	3.7	0.9	0	0.1	0	21.9	2.2
Total	34	50	88	37	9	0	1	0		

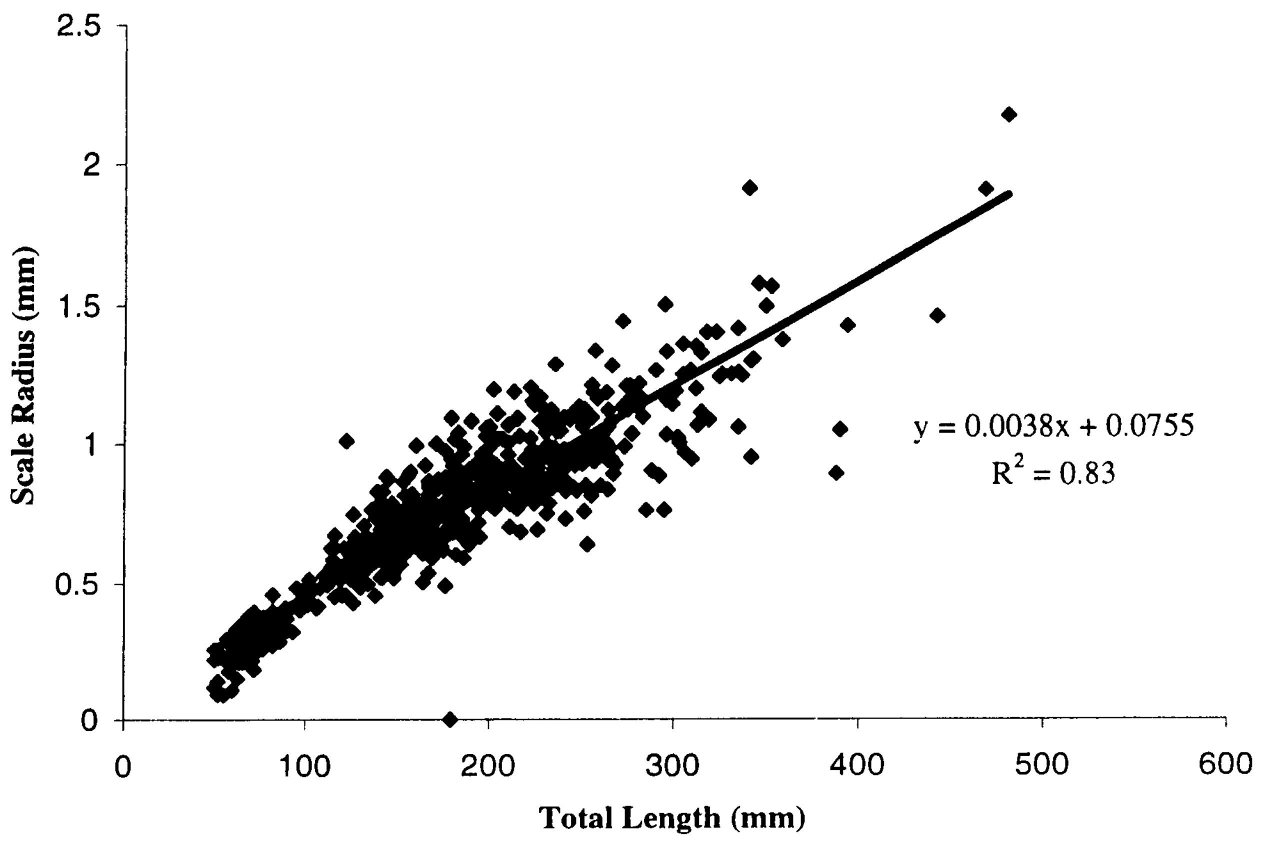


Figure 25. Regression of brown trout total length on scale radius for back-calculations of length at age.

($p=0.004$, and 0.003 respectively) in growth between above and below sections (Table 14). The Boardman River had greater brown trout growth below the dam, while the White River showed poorer growth downstream (Figure 26). Mixed modeling ANCOVA of incremental growth on previous length at age and position, where stream was treated as random and previous length and position were treated as fixed, showed no significant difference in brown trout growth between stream sections ($p=0.107$). Once position was dropped from the model, the final mixed model relating incremental growth and previous length at age ($p<0.001$) for brown trout across all streams was:

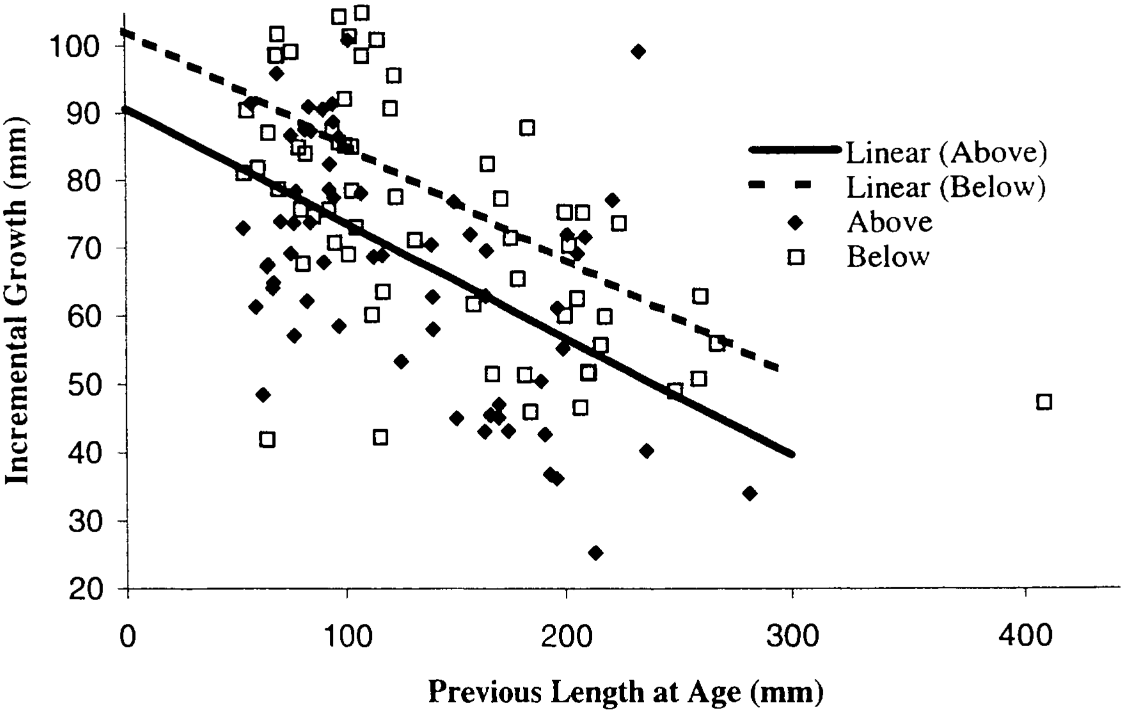
$$I=87.8-0.146(P\_len_{age}), \qquad [14]$$

where I is the incremental growth for the age step of interest and P_len_{age} is the length of the fish at the previous age. The data showed a large number of observations below 200 mm, due to the lack of older fish caught in the study streams (Figure 27). Also, variability among study streams, in terms of habitat, food availability etc., creates substantial variability in incremental growth at the same previous length.

Since incremental growth decreases as fish become older and larger (Figure 27), I ran stepwise regression analysis on three fish lengths representing the range of lengths caught in the study streams. These length intervals and their corresponding incremental growth labels are 100 mm (incre100), 200 mm (incre200) and 300 mm (incre300). Stepwise regression (MaxR) revealed that incremental growth of brown trout (age>1) was most related to stream conductivity for all three length bins ($p=0.017$, 0.005, and 0.007; $R^2=0.53$, 0.64 and 0.67 respectively). Mixed modeling ANCOVA, where stream was treated as random and position was treated as a fixed effect, revealed that incremental growth for all three length intervals was significantly related to stream conductivity

Table 14. Growth model parameters, number of fish (N) and length range for brown trout above and below the dams in each stream. Predicted incremental growth for fish in three length bins (100, 200 and 300 mm) are also shown. Streams with significant differences between position are noted with an *.

Stream	**Position**	**Intercept**	**Slope**	**N**	**Length Range (mm)**	**Incre100**	**Incre200**	**Incre300**
Boardman*	Above	90.67	-0.17	64	126-388	73.67	56.67	39.67
	Below	102.275	-0.17	69	180-481	85.275	68.275	51.275
Cedar	Above	87.5	-0.156	35	157-545	71.9	56.3	40.7
	Below	n/a	n/a	n/a	n/a	n/a	n/a	n/a
Dowagiac	Above	n/a	n/a	n/a	n/a	n/a	n/a	n/a
	Below	n/a	n/a	n/a	n/a	n/a	n/a	n/a
Fish	Above	n/a	n/a	n/a	n/a	n/a	n/a	n/a
	Below	n/a	n/a	n/a	n/a	n/a	n/a	n/a
Manton	Above	94.283	-0.173	33	156-276	76.983	59.683	42.383
	Below	106.87	-0.173	10	196-318	89.57	72.27	54.97
Maple	Above	109.91	-0.189	11	176-359	91.01	72.11	53.21
	Below	112.14	-0.189	22	195-469	93.24	74.34	55.44
Middle	Above	78.31	-0.066	21	134-194	71.71	65.11	58.51
	Below	77.55	-0.066	1	305	70.95	64.35	57.75
Prairie	Above	86.13	-0.431	14	122-222	43.03	-0.07	-43.17
	Below	87.96	-0.431	11	129-180	44.86	1.76	-41.34
Sugar	Above	103.54	-0.219	14	178-285	81.64	59.74	37.84
	Below	n/a	n/a	n/a	n/a	n/a	n/a	n/a
White*	Above	100.6	-0.202	16	132-209	80.4	60.2	40
	Below	69.04	-0.202	22	132-209	48.84	28.64	8.44



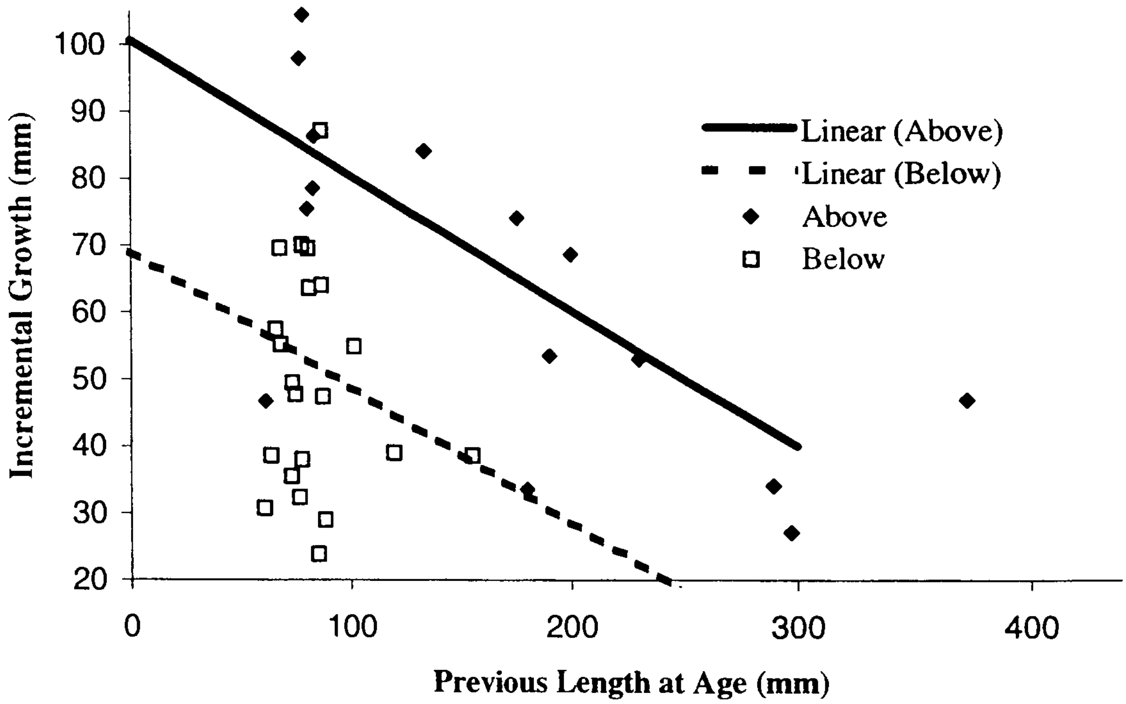


Figure 26. Growth of brown trout above and below the dams for the Boardman (Top) and White (Bottom) rivers.

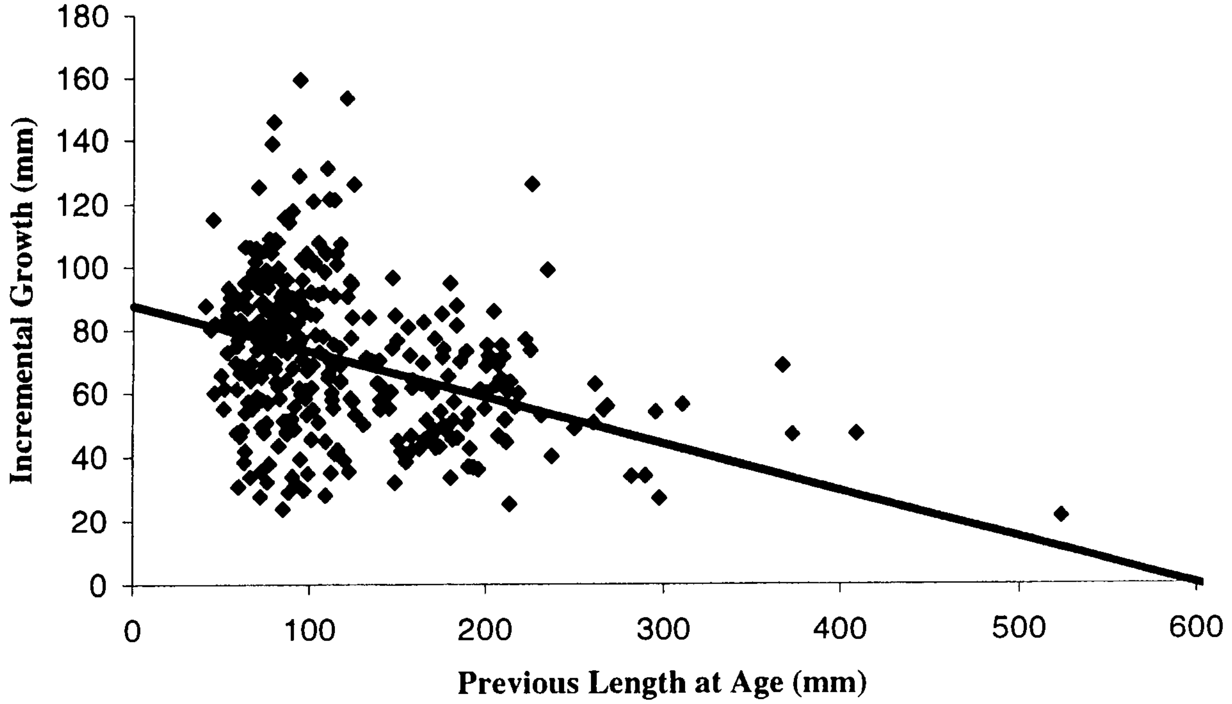


Figure 27. Growth of brown trout over all study streams.

(p=0.055, 0.045, and 0.048 respectively), although marginally for incre1, and that there was no significant difference between stream sections. For example, the model relating incre200 and conductivity is:

$$\text{Incre200}=130.47-0.25(C), \qquad [15]$$

where Incre200 is the incremental growth for a 200 mm brown trout and C is the stream conductivity. Again, the low numbers of large fish makes this relationship more difficult to describe with confidence, but in general incremental growth of brown trout (age>1) decreases with increasing conductivity (Figure 28). Mean summer temperature was not a significant covariate for any of the length bins (p=0.19, 0.26, and 0.46 respectively).

Stepwise regression of age zero growth, revealed that mean summer temperature was the most related single parameter (p=0.0001, R^2=0.26). Mixed modeling ANCOVA of age zero growth on temperature and position, where stream was treated as random and position was treated as a fixed effect, indicated that there was a significant interaction between temperature and position. The interaction term was ignored and dropped from the analysis because with very few below sites having age one fish, one anomalous site was driving an interaction that should not be interpreted with this data. Once the interaction term was dropped, position was also found to be non-significant and therefore was dropped as well. The final mixed model relating age zero growth and mean summer temperature was:

$$I=32.6+3.18(T), \qquad [16]$$

where I is age zero brown trout growth and T is the mean summer temperature (°C). Age zero brown trout growth increased with increasing temperature (Figure 29).

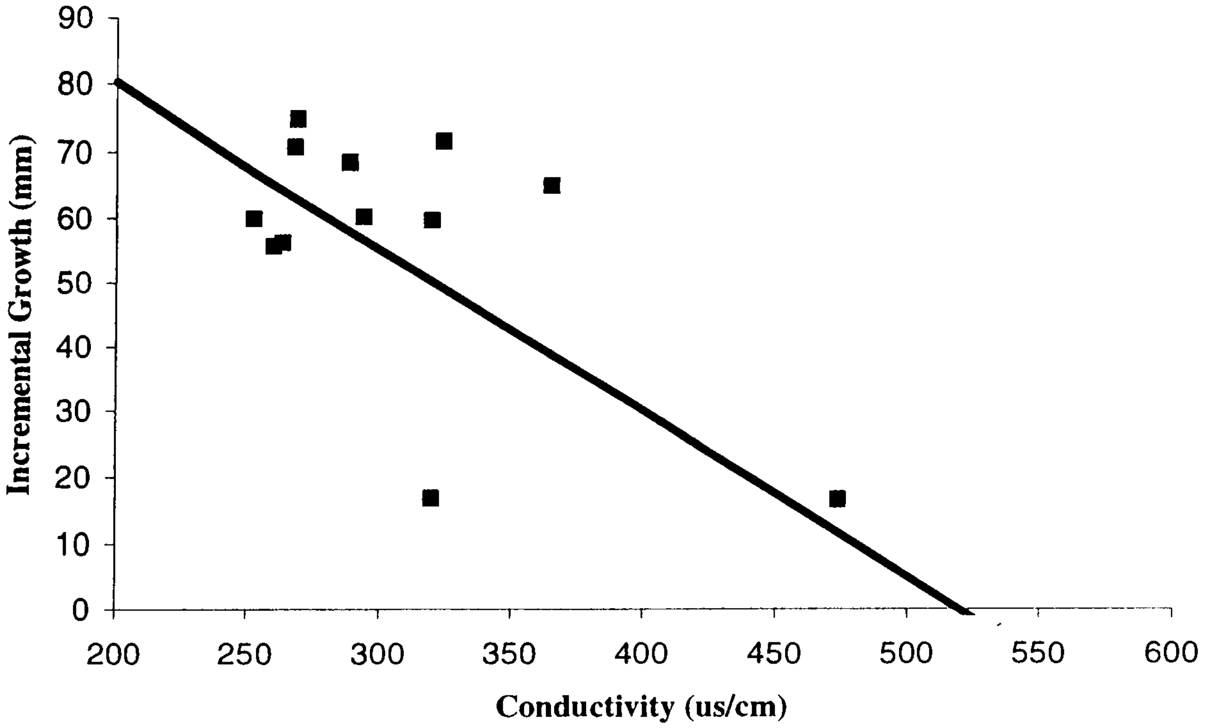


Figure 28. Influence of conductivity on Incremental growth of brown trout with previous length of 200 mm.

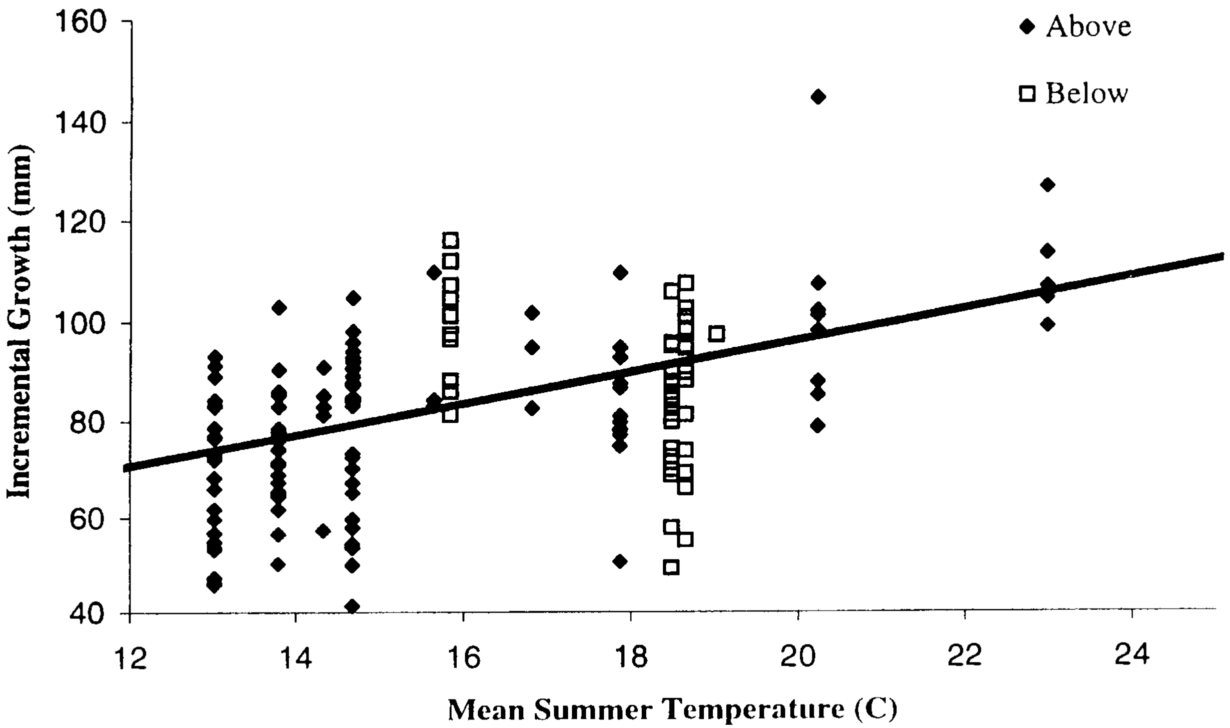


Figure 29. Influence of temperature on age zero incremental growth of brown trout in study streams.

Discussion

General Habitat

Stream habitat is expected to change predictably in a downstream direction (Vannote et al. 1980). Temperate headwater streams are generally shallower, narrower, and are dominated by coarser substrate than higher order reaches downstream (Cummins 1977; Vannote et al. 1980). With the widening and deepening of the channel, stream discharge increases as well as temperature fluctuation and daily maximum temperature (Cummins 1977; Vannote et al. 1980). Stream regulation (i.e. damming) alters the natural longitudinal flow of energy and materials, thereby altering the physical and chemical characteristics of downstream reaches and the biological community (Vannote et al. 1980; Ward and Stanford 1983a; Ward and Stanford 1983b). The serial discontinuity theory (Ward and Stanford 1983a) offers a hypothesis for the impact that a dam will have on downstream habitat and biota. This theory, however, was developed with the assumption that dams operate via hypolimnetic draw, and so many of the conclusions, especially those involving predictions about water quality and biotic changes, run contrary to what we expected to find in our study streams (Table 15).

The study streams showed the expected longitudinal increase in width, depth and discharge as predicted by the River Continuum Concept, and did not appear to be significantly altered by the dams (Vannote et al. 1980)(Figure 4). There was an increase in substrate particle size downstream, which is consistent with what is expected for dammed middle order streams (Ward and Stanford 1983a)(Figure 5). Seston quality (i.e. organic content of seston) is expected to peak below surface release reservoirs in the same way as has been documented for lake outlets, while total seston values often remain

Table 15. Selected stream community and habitat changes below dams as predicted by the serial discontinuity theory for low to middle order streams (Ward and Stanford 1983a), predictions for the streams in this study, and observed changes.

Community or Habitat Attribute	**Predicted Change Downstream of Dams (Serial Discontinuity)**	**Predicted Change Downstream of Dams in This Study**	**Observed Change Downstream of Dams in This Study**
Substrate Size	Increase	Increase	Slight Increase
Summer Temperature	Decrease	Increase	Significant Increase
Fish Diversity	Decrease	Increase	Increase
Nutrient Levels	Decrease	Decrease	No Change

unchanged or increase downstream (Herlong and Mallin 1985; Richardson and Mackay 1991). The streams sampled in 1998 showed no significant difference between upstream and downstream sections in both seston quality (measured by total organic seston) and total seston concentration (Table 2).

Total phosphorous concentration, is often used as an indication of nutrient limitation and overall stream productivity (Wetzel 1975; Allan 1995). Total phosphorous values were highly variable between streams, and generally were at or below that of normal rainwater (Wetzel 1975; Allan 1995). Phosphorous levels in individual streams were not significantly different between upstream and downstream sections, with the exception of the White River. Averaging across all streams, phosphorous showed no longitudinal trends. Conductivity (i.e. total ionic content), which is related to total dissolved solids (TDS) and also is used as a surrogate for productivity (Ryder et al. 1974; Ryder 1982; Allan 1995), was slightly higher in downstream sections when compared to upstream sections. Conductivity increased significantly downstream in three streams (Cedar, Dowagiac and Manton). It was difficult to determine what caused this increase in conductivity, but it could have been due to increased urban and/or agricultural runoff carrying excess salts and other nutrients into downstream reaches. In lakes, total dissolved solid (TDS) concentrations often vary differently from total phosphorous concentrations, especially if phosphorous is limiting (Ryder et al. 1974). Cycling of limiting nutrients does not consume the ion, so fluctuations in phosphorous would not necessarily affect TDS or conductivity (Ryder et al. 1974). Because of this, conductivity (as a correlate to TDS) is sometimes a better indicator of productivity (Ryder et al. 1974). The serial discontinuity theory predicts a decrease in nutrients below dams in middle reaches, because the reservoir often acts as a nutrient sink (Ward and Stanford 1983a). My data did not show any difference between sites 3 (last upstream site) and 4 (first downstream site) in terms of total phosphorous concentrations or stream conductivity, but instead a downstream trend of increasing conductivity and variable phosphorous levels. Because of the relatively small changes in average conductivity and lack of significant differences in total phosphorous concentrations, it appeared that upstream and downstream reaches were similar in terms of productivity.

Temperature

Ward and Stanford (1979) summarized the potential temperature alterations caused by dams into six categories: 1) increased diurnal constancy, 2) increased seasonal constancy, 3) summer depression, 4) summer elevation, 5) winter elevation, and 6) thermal pattern changes. Historically, large hydroelectric operations were of greatest concern, because their mode of operation often caused catastrophic changes downstream (Brooker 1981). Hydroelectric facilities often draw water from the hypolimnion, and therefore summer decreases and winter increases have been the primary focus for research on dam induced temperature changes (Brooker 1981; Crisp 1987; Jensen 1987; Ward and Stanford 1987).

Temperature increases have been given little attention, and primarily have focused on impacts on the macroinvertebrate community (Ward and Stanford 1979; Fraley 1979). It has been suggested that impacts on temperature are restricted to the area of stream directly below the impoundment and that temperatures quickly equilibrate with the air (Brooker 1981). Downstream reaches with increased summer temperatures, however, will not shed heat during hot summer days, but will continue to warm according to normal River Continuum theory (Boon and Shiers 1976; Vannote et al 1980). This heat loading has the potential to increase downstream temperatures so that they remain elevated, above the range that cold-stenotherms can survive in. Webb and Walling (1993) were able to detect 0.5 °C mean summer increase 5 km below an impoundment and Fraley (1979) found significant summer increases that never returned to upstream temperatures 56 km downstream of a dam.

In this study, I focused on increases in mean summer temperature below dams, and how it related to alterations in downstream communities. While dams can alter temperature in other ways (e.g. reduced diel fluctuation) and in other seasons, it is during the summer that there is maximum potential for physiological heat stress to occur in adult cold stenotherms. Additionally, bioenergetic models indicate that mean summer temperature is especially important for growth because of the relationship between metabolism and temperature as well as feeding and temperature (Gibbons 1976). Another reason for the focus on mean summer temperature changes versus daily flux was that without continuous loggers at all sites it would be impossible to determine how daily flux changed with distance from the dam. Fraley (1979) found that while mean summer temperature changes downstream of a surface release dam were consistently higher than upstream reaches, average diurnal fluctuation was lower directly below the dam and then returned to upstream levels with increasing distance downstream.

Impact of the dams on mean summer temperatures was variable, but on average streams were 2.7 °C warmer downstream than upstream. Individual impacts among the study streams ranged from a slight cooling to more than 5 °C increase downstream, and were able to be separated into two general categories (high and low impact). Factors influencing temperatures below dams include the size of impoundment (specifically depth and surface area), residence time, whether or not the impoundment stratifies, and the release depth (Ward and Standford 1979; Brooker 1981). All dams in this study were surface release facilities with the greatest potential to increase downstream temperatures. Variability in impact, therefore, most likely was due to differences in surface area, depth and residence time. Another important factor that can contribute to variability is the amount of groundwater coming in downstream of the dam. Groundwater increases the stability of flow and also cools stream summer temperatures (Allan 1995; Giller and Malmqvist 1998). In the Maple River, for example, it appeared that there was sufficient groundwater recharging the system downstream to more than compensate for the impact of the reservoir, resulting in slightly colder downstream temperatures than upstream. The dam on Prairie Creek created a very small impoundment with almost no residence time, therefore there was virtually no difference in mean summer temperature between stream sections. Dowagiac and Fish Creeks both had relatively high temperatures upstream and lower overall temperature impacts, but even a small increase in temperature for a marginal coldwater stream could have dire consequences for cold-stenotherms downstream. This is especially true if temperatures are raised above 20°C (Allan 1995; Giller and Malmqvist 1998; Taniguchi et al. 1998). For those streams that did have increased summer temperatures downstream, this impact remained with increasing distance from the dam and in most cases increased further.

Macroinvertebrate Community

Studies comparing rivers across the globe have shown macroinvertebrate family richness to be linearly related with stream temperature, with diversity increasing with increasing temperature (Jacobsen et al. 1997). In this study, macroinvertebrate family richness was not significantly different between stream sections and was not related to any of the habitat parameters we measured. Macroinvertebrates often show shifts in habitat preferences along environmental gradients within a family, and so identification to the species level may have shown some different relationships (Fraley 1979; Hauer et al. 2000). Fraley (1979) found a decrease in macroinvertebrate diversity directly below a surface release dam, but then similar levels of diversity between upstream sites and sites further downstream.

Ward and Stanford (1979) summarized the results of several studies on dam impacts and showed that regardless of how temperature was altered (increased, decreased, daily flux etc.) plecopterans and ephemeropterans tended to decrease below dams and trichopterans showed variable results. Plecopterans, on the whole, are considered cold-stenotherms, although certain families (e.g. Perlidae) are found in warmer habitats as well (Ward 1992). In my study streams, plecopteran richness showed a significant, negative correlation with mean summer temperature. When streams were separated according to level of impact, only high impact streams showed significant decreases in plecopteran richness downstream (Figure 8). In high impact streams, plecopterans were often eliminated below the dams and when plecopterans were found, they were primarily Perlidae. From these data, it appears that the type of temperature alteration has an effect on how plecopteran richness changes below dams. E.P.T. richness values also were lower downstream, but had higher variability and were not strongly related to any habitat parameters, although they tended to be higher in shallower, narrower reaches.

Functional feeding group abundances tended to be similar between upstream and downstream sections with the exception of an increase in filtering collectors directly below the dam. This response by filtering collectors below dams has been noted elsewhere and is most likely due to an increase in suspended food particles coming out of the reservoir (Fraley 1979; Herlong and Mallin 1985; Richardson and Mackay 1991). Although our data did not detect a difference in seston in the four study streams where seston was examined, it's possible that water samples for seston analysis were taken too far downstream to detect a difference. Studies have shown that below reservoirs and lake outlets there is often rapid use of high quality seston by high densities of filtering/collector macroinvertebrates, reducing the concentration of these suspended particulates longitudinally downstream (Wallace and Merritt 1980; Herlong and Mallin 1985; Richardson and Mackay 1991). This may have been a factor in this study, because of the observed peak in filtering collectors at the first site downstream when compared to the last site upstream, across all streams. Following this peak there was a general decline in filterers moving downstream, indicating some orientation with respect to the dam (Figure 10).

Macroinvertebrate community similarity was significantly related to differences in mean summer temperature, and level of impact was a good gauge of how similar stream sections above and below dams are likely to be. In general, when a dam increases downstream temperatures 3-5 °C, then similarity would be expected to be around 0.56 (∓0.032), and if a dam increases downstream temperatures 2 °C or less, similarity would be higher, around 0.76 (∓0.06) (Table 4). The fact that macroinvertebrate richness was not significantly different between stream sections indicates that it is not a change in diversity downstream causing this relationship, but rather a replacement of taxa. The relationship between similarity and level of impact points to temperature as the reason for this replacement.

Fish Community

Species diversity is expected to increase as you move from headwater areas to middle reaches and then decrease again toward the mouth (Vannote et al. 1980). Increased environmental heterogeneity through normal stream processes or moderate levels of human disturbance is predicted to increase habitat availability and allow for more species to co-exist (Gorman and Karr 1978; Vannote et al. 1980; Ward and Stanford 1983b). Ward and Stanford (1983a) postulated in the serial discontinuity theory that biotic diversity below a dam would decrease either from reduced detrital transport in headwater streams or from reduced temperature fluctuation in middle reaches.

In my study streams, fish species richness was almost always highest directly below the dam, particularly in high impact streams. High impact streams had significantly lower fish species richness upstream when compared to downstream sections of high impact streams and upstream sections of low impact streams (Table 4). High and low impact streams had similar levels of fish species richness downstream, most likely due to their similar average temperatures downstream (Table 4). Coldwater habitats tend to have fewer species due to their harsh nature as well as more constant temperatures (Allan 1995). High impact dams increased summer temperatures from a mean of 15 °C to a more moderate mean of 19 °C, which was accompanied by a significant increase in fish species richness from 9 to 16 (Table 4). If we think of increased temperature as a disturbance, as it is defined in the intermediate disturbance hypothesis (Ward and Stanford 1983b), then it should be considered a "moderate disturbance", since temperatures did not rise into the lethal range for most cool- and warm-water fish species. As such, there would be an expected increase in diversity downstream, even if diel fluctuation was reduced directly below the dams. In the low impact streams, where temperature on average only increased from 19.7°C to 20.5°C, fish species richness was not significantly different (Table 4).

Lyons (1996) found cold, headwater streams to be dominated by a few species of salmonids and sculpins, and noted an increase in species diversity as well as a shift in dominance to more eurythermal, warm-water species in streams with higher summer temperatures. Wehrly et al. (1998) showed that warmer streams consistently contained more species than cold streams across and within three different levels of July temperature fluctuation. Other studies of differences in fish species richness below dams also indicate that temperature change is important, and show decreasing richness with decreasing temperatures (Brooker 1981). A study of low-head barrier effects on fish species richness noted a peak in richness just below dams and then a decline in richness with increasing distance downstream, toward what was observed for uninterrupted streams (Dodd 1999). These barriers did not alter temperature and so the peak below dams was attributed to a pooling of fish species at the barrier (Dodd 1999). Aggregation of fish species attempting to move upstream may have played a role in my study streams as well, but the fact that fish species richness remained elevated with increasing distance downstream in both high and low impact streams indicates that temperature was probably more important. Sheldon (1968) found stream depth to be the most important factor in determining diversity within a stream. In this study, both high and low impact streams showed similar longitudinal increases in depth moving downstream. When all the habitat parameters were considered, the best predictor of fish species richness, as well as changes in richness with respect to the dams, was temperature. It seems reasonable, therefore, to conclude that changes in mean summer temperature below these types of dams are primary in determining how diverse the fish community downstream will be. It is also likely that the reduction in fish species richness seen below deep-release dams, is probably more related to the reduction in temperature below the physiological optimal for most warm-water fish than previous studies have indicated.

Fish community similarity was not related to differences in mean summer temperature, but instead was negatively related with stream depth changes. This result was surprising since low and high impact streams had similar mean differences in depth between stream sections, but different mean similarities. Studies on the relationship between depth and fish community richness showed that increased habitat complexity from expanded vertical habitat allowed for more species to co-exist in deeper waters (Sheldon 1968; Gorman and Karr 1978). In these streams, depth was not related to fish species richness, and therefore it seems unlikely that depth changes would cause a replacement of species downstream since fish are not usually excluded from habitat based on depth alone. High variability in stream fish similarities made it difficult to attribute any strong relationships between habitat and similarity in these streams. Low impact streams, however, were more similar on average than high impact streams (Table 4). The average similarity of the low impact streams (0.66) was almost equal to what was calculated by Dodd (1999), comparing upper and lower reaches of unregulated streams in Michigan (0.68). This indicates that the presence of a dam alone does not necessarily impact the fish community in terms of similarity, but requires some habitat alteration as well. At this time it is difficult to point to one habitat parameter as the reason for the level of fish community similarity that was demonstrated in these streams.

Density of Target Species

Brook and brown trout are coldwater stenotherms that have positive growth between 4 and 20 °C, with upper lethal limits around 25 °C (Elliott 1994; Allan 1995; Marod 1995). Brook trout catch and movement patterns have been related to the number of days temperature exceeded 16 °C (Marod 1995), and in my study streams brook trout presence declined sharply above 17 °C. Brook trout were more often eliminated below the dams than brown trout. The former showed a 96% downstream decrease across all streams, while the latter were 54% lower downstream. For both of these species, mean summer temperature was the most important predictor of presence and density.

Both slimy and mottled sculpin are typically found in cold, shallow stream habitats and are usually associated with brook trout, although slimy sculpin are more consistently considered a headwater species (Scott and Crossman 1973). Stream substrate and temperature together are considered the primary determinants for sculpin habitat suitability (Scott and Crossman 1973). Both sculpin species were 78% lower below the dams on average, but slimy sculpin were in fewer streams. There were only a few instances where both species were found together and in general slimy sculpin were in the coldest streams with lower conductivities and mottled sculpin were in the warmer streams with higher conductivities (Figure 30). This may have been a coincidence since the natural distribution of slimy sculpin only dips into the northern portion of Michigan's lower peninsula and mottled sculpin distribution is said to be more spotty in the Great Lakes basin (Scott and Crossman 1973). The coldest streams in this study tended to be

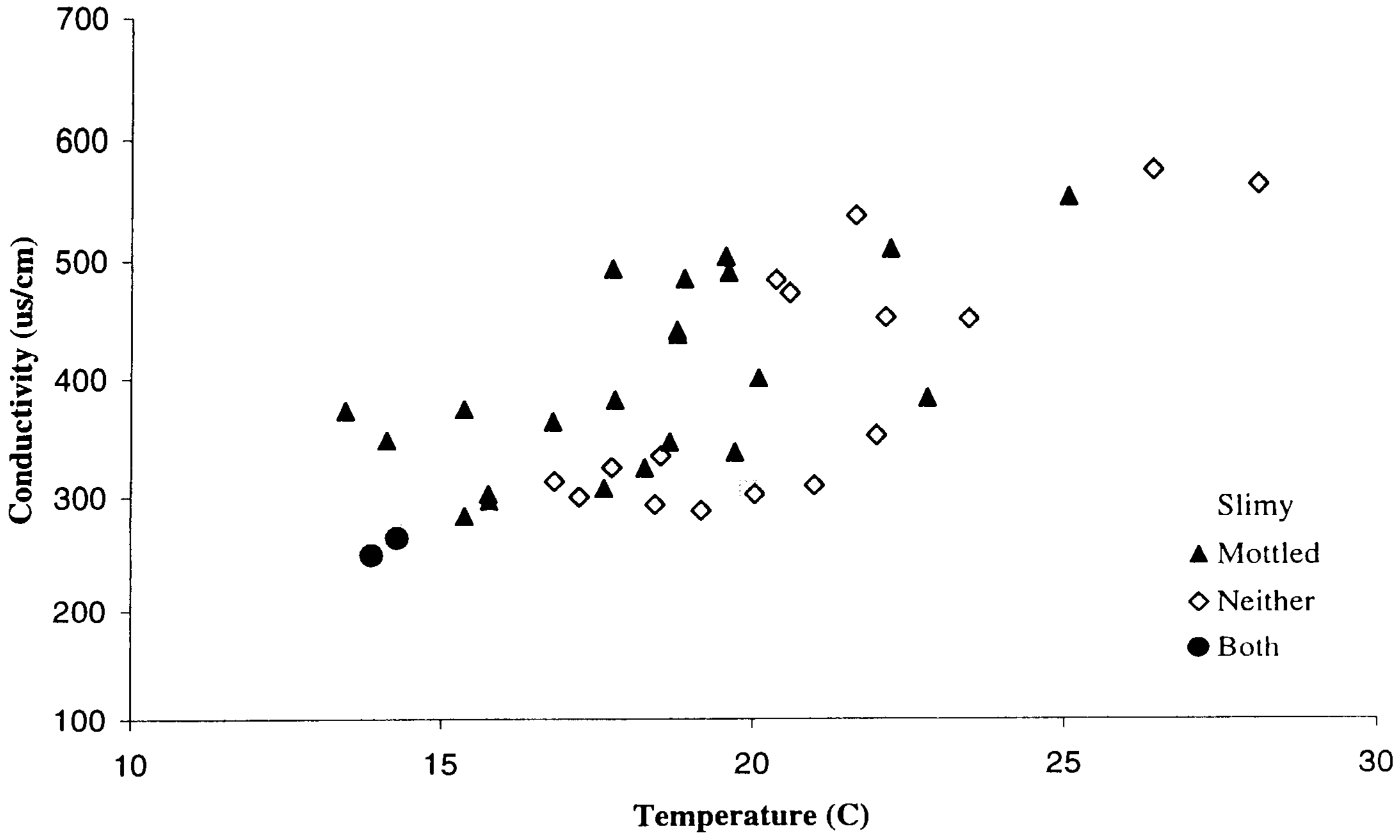


Figure 30. Relationship between conductivity and temperature for sites where mottled sculpin were present, slimy sculpin were present, neither sculpin species were present, or both sculpin species were present.

more northern, due to the distribution of trout streams in this state. However, other research of sculpin distribution in northern Michigan streams found both species present, but slimy sculpin showed a more narrow distribution with respect to temperature and mottled sculpin were found in both cold and warmer habitats (Wehrly et al.1998). In my streams, slimy sculpin density was most related to mean summer temperature and mottled sculpin density was not strongly related to any habitat parameters. Other research relating mottled sculpin abundance and habitat in Michigan was also unable to detect any significant relationships (Newman 1999).

Position with respect to the dam was not an important factor in determining trout or slimy sculpin abundance, indicating that the presence of a dam alone does not negatively impact these species. Instead, it is the alteration of downstream habitat away from what is preferred, that is detrimental. Hayes et al. (1998) have indicated the value of cold, headwater streams for providing thermal refugia for brook trout during the summer. Brook trout under suboptimal thermal conditions will move into colder, headwaters or tributaries and then return to middle reaches in the fall and winter (Scott and Crossman 1973; Marod 1995). Dams that prevent this migration and increase downstream summer temperatures above 20°C limit the ability of brook trout to escape or survive until fall. Brown trout also show behavioral movements in response to temperature (Elliott 1994), and are probably impacted by increased temperatures below dams in a similar way as brook trout. Brown trout, however, are able to remain competitive at warmer temperatures than brook trout (Taniguchi et al. 1998), and therefore persist in downstream reaches more often. Considering my results and those from other studies, it appears that a dam which increases summer temperatures above 17°C can be expected to hinder brook trout survival, and if temperatures are increased above 20°C all three species (brook trout, brown trout and slimy sculpin) are likely to be reduced.

Conductivity was related to the probability that brook trout, slimy sculpin and mottled sculpin would be present in the study streams. In general, brook trout and slimy sculpin were more likely to be present in stream sections with conductivity values below 300μs/cm and mottled sculpin were more likely to be present at higher conductivities. At present it is difficult to determine what could have caused these relationships. The conductivity values documented for these streams were well within the normal ranges for streams in Michigan (Michigan Department of Environmental Quality, unpublished data). There appears to be a geographical shift in conductivities in Michigan due to differences in soils and geology, with northern streams usually having lower conductivities than southern streams (Michigan Department of Environmental Quality, unpublished data). This may have been a factor for the two sculpin species which seem to show a geographic separation.

Modde et al. (1986) found brook trout to be tolerant of moderate watershed alterations including changes in pH, turbidity and nutrient loading at temperatures between 9°C and 12°C. In other studies, brook trout densities and growth have been shown to be positively correlated with stream conductivities ranging between 27μs/cm and 869μs/cm (Cooper and Sherer 1967; Scarneccia and Bergersen 1987;Kwak and Waters 1997). I can only speculate on why brook trout presence showed a negative relationship with conductivity in these streams. This could be a spurious correlation, or possibly something that we did not measure is also correlated with conductivity and is impacting the brook trout, but in any case it should receive further research.

Depth was related to mottled sculpin presence and density, although during mixed modeling analysis it was dropped from the density model. Mottled sculpin tended to be found in depths below 50cm. This type of specialization is surprising since mottled sculpin occupy lake habitat as well as streams (Scott and Crossman 1973). Van Snik and Stauffer (1999) found both slimy and mottled sculpin to be generalists when it comes to depth selection. Experience in these rivers has shown catchability of sculpin to vary with flow velocity, substrate and depth. This could have been a factor in the deeper, downstream areas of the rivers where mottled sculpin were more difficult to catch.

Trout Growth

Growth of fishes is influenced by many environmental factors, but among the most important are food quantity and quality, and temperature (Elliott 1994). How these relate to growth of individual species depends on the endogenous physiology of each species and body size (Elliott 1994; Diana 1995). Metabolic maintenance increases with increasing temperature, and so if food is limited, fish may seek colder temperatures to maximize growth between food availability and metabolic costs (Diana 1995). Fish size also is of importance, because larger trout are less susceptible to changes in water temperature than small trout (Elliott 1994).

Brook and brown trout growth is generally positive between 4°C and 20°C (Baldwin 1956; Elliott 1994; Marod 1995). Thermal stress in most trout is expected to begin at temperatures above 20°C, which means growth would stop and body size could decrease if metabolic costs exceed what the fish can consume (Elliott 1994). Jensen (1987), predicted growth of brown trout below a dam that decreased summer temperature ranges from 12-15°C down to 9.5-12.5°C, to be 19.7 mm shorter than trout in unregulated sections. In a study conducted on the relationship between brook trout growth and feeding at three temperatures, the highest rate of feeding and weight gain was at 13 °C while lower rates were found at both 9 °C and 17 °C (Baldwin 1956).

The age distribution of brook trout and brown trout in these streams was similar to what other investigators have found in Michigan trout streams (Gowing and Alexander 1980; Mistak 1999). Brook trout densities below the dams were too small to do a meaningful comparison of growth rates between stream sections. In general, brown trout (age>1) growth was not significantly different between stream sections. The Boardman and White Rivers were exceptions, however, and showed opposite trends, with higher growth downstream in the Boardman and lower growth downstream in the White. Given the differences in temperature between streams and sections within a stream and the strong relationship between brown trout density and temperature, it was surprising that conductivity was most related to incremental growth of brown trout (age>1) and that temperature was not a significant factor. Another surprising result was that the relationship between conductivity and growth of brown trout was negative. Significant variation among the streams in terms of productivity (using conductivity as a surrogate) would be expected to lead to variation in growth, with more productive streams having higher growth as long as the temperature was within the limits for trout. But in these streams, conductivity was not substantially different and there was a negative trend between productivity and growth. I suspect that the uneven distribution of length and age data for brown trout in these streams could be indicating a false correlation, and that if we had caught more fish of larger sizes, the relationships shown here would change. Further indication of this is that Prairie Creek, which had relatively high conductivity and appeared to provide good trout habitat, showed poor brown trout growth. Prairie Creek is one of the few rivers in the southern portion of Michigan's lower peninsula that maintains cool temperatures and is known for relatively good trout fishing. Because of this, and its proximity to urban areas, this river appears to get high fishing pressure. Michigan's catch regulations only allow fish to be kept if their length is at least 203 mm. For a heavily fished population, this means that the older fish that remain in the system are often the slow growers (i.e. fish of the same age but smaller than 203 mm). Prairie Creek is really driving the regression line relating brown trout growth and conductivity, and without it this relationship would not be significant (Figure 31). Obviously, what is determining growth of brown trout (age>1) in these streams needs to be looked at further.

Age zero brown trout growth was significantly related to mean summer temperature. Age zero brown trout incremental growth increased with increasing temperature. This reinforces the notion that smaller fish are more sensitive to temperature than older, larger fish. These data are also more robust since fishing pressure does not directly alter natural variability in growth of young of the year (YOY) fish.

As with density, growth of trout below dams would not be expected to be different from upstream reaches unless the dam altered the downstream habitat in a way that was significant to growth. Significant changes would include alterations of the downstream thermal regime and changes in the macroinvertebrate community. Another study of trout growth above and below a dam that did not alter temperature found no differences in growth of brook, brown or rainbow trout (Klomp 1998). Temperature plays a dual role in that it directly impacts growth, as I have described earlier, and indirectly impacts growth by directly influencing primary and secondary production.

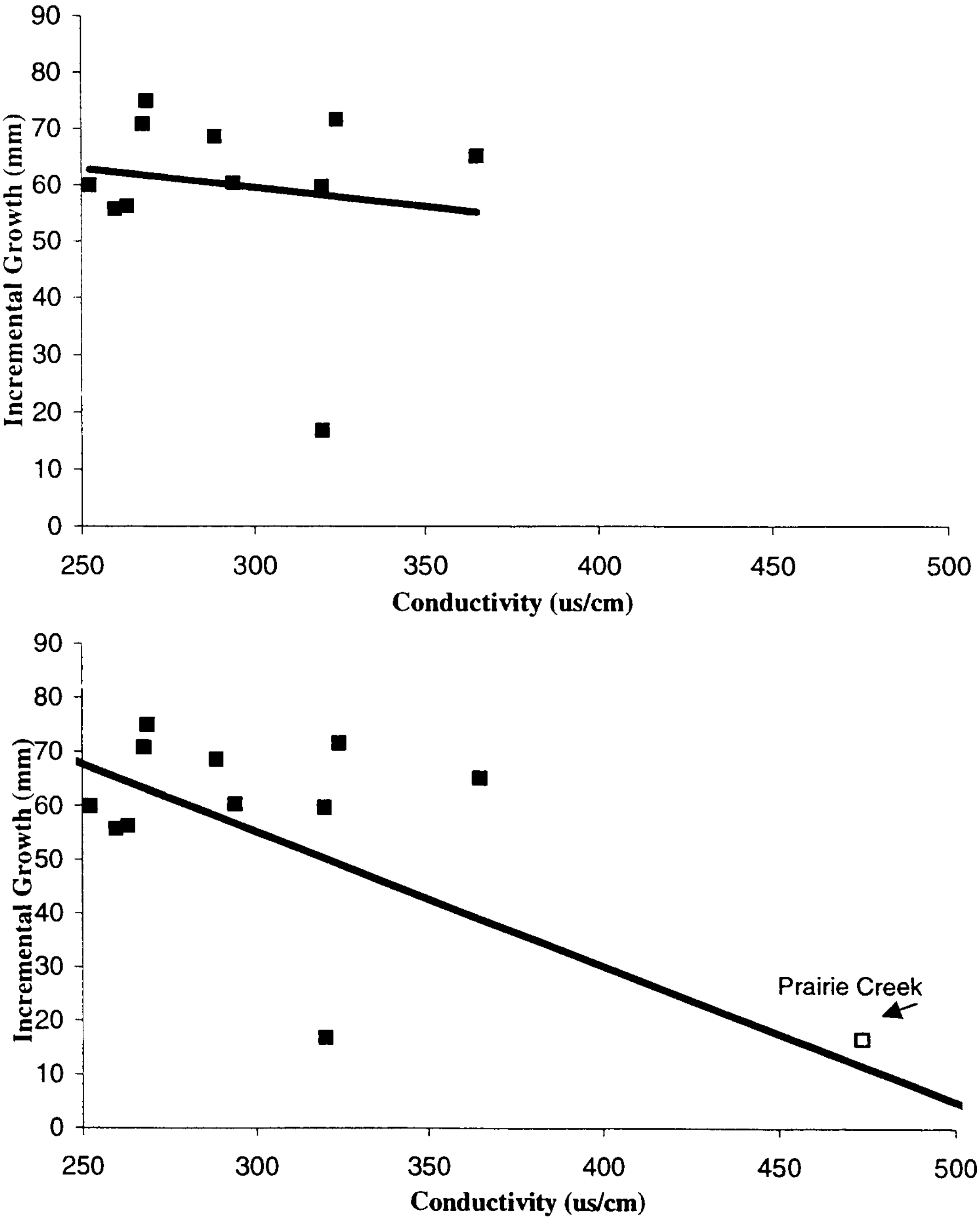


Figure 31. The relationship between incremental growth of brown trout age>1 and conductivity without Prairie Creek's data (Top) and with Prairie Creek included (Bottom).

Stream macroinvertebrates, which make up a large portion of the secondary production in streams, are a primary food source for trout and can be strongly altered by temperature (Fraley 1979; Ward and Stanford 1979; Hawkins et al. 1983; Rader 1997). Models predicting growth of brown trout fed on maximum rations were not appreciably higher than the growth rates of brown trout measured in field studies, suggesting that food supply in nature is often adequate and therefore temperature may be the primary determinant of growth rates (Edwards et al. 1979). In this study, the macroinvertebrate community showed shifts in certain taxa according to temperature increases below the dams, but there were no significant decreases in overall abundance of macroinvertebrates (Dowagiac Creek excepted). Therefore, food for trout in these streams does not appear to have been a major factor in determining their density or growth rates. Temperature was still most likely the primary factor determining growth rates of both YOY and older brown trout.

A dam can improve growth by increasing summer temperatures to a point, but once the physiological optimum of trout is exceeded growth would be expected to decline. In these streams, the trout populations tended to respond with density changes rather than large growth differences. It may be that a temperature stressed fish will choose to escape to cooler downstream reaches, if they are available, but if they can find no refugia they may die before showing poor growth. Temperature alterations from these dams in other seasons may also impact egg, larval and juvenile mortality rates which would eventually alter overall density estimates and not necessarily growth.

Conclusions

Stream habitats change from headwater reaches to the mouth, and dams are known to alter certain habitat parameters so that the normal continuum is disrupted. In this study I compared several habitat parameters in stream sections above and below ten dams in Michigan. The dams on these rivers did not appear to significantly alter the physical attributes (e.g. depth, width, substrate size) of these streams beyond the natural progression that would be expected for downstream reaches when compared to upstream reaches of unregulated streams. Likewise, chemical attributes of these streams (e.g. dissolved oxygen, nutrient levels) also were not significantly altered downstream, with the exception of mean summer temperature. In general, mean summer temperature was substantially increased downstream by these small, surface release facilities.

Downstream macroinvertebrate and fish communities responded differently to increased mean summer temperatures below dams. The macroinvertebrate community showed no significant change below dams in terms of family richness, but did respond with decreases in certain taxa that are sensitive to increased temperatures, particularly plecopterans. Dams that significantly increased summer temperatures (i.e. high impact streams) showed relatively large changes in community composition. This resulted in a strong relationship between macroinvertebrate community similarity and temperature change below the dams. It appeared, at the family level, that macroinvertebrates may have been responding to temperature increases by replacing taxa based on temperature preferences rather than changes in richness.

Fish community composition also showed a general shift from more cold-stenothermic species upstream to more cool-water, eurythermic species below high impact dams. However, the relationship between temperature increases (below dams) and fish community similarity was not as strong as for macroinvertebrates. Instead, the fish community below high impact dams showed increases in species richness rather than a consistent pattern of species replacement. Although the macroinvertebrate and fish communities changed differently below high impact dams, the discontinuity in both communities below these dams indicates that dams that increase temperature may have an important impact on downstream communities.

Investigations of dam impacts on the presence and density of coldwater fish species including brook trout, brown trout, slimy sculpin and mottled sculpin revealed differing relationships with temperature. Both trout species showed significant, negative relationships with increasing summer temperature, resulting in lower population densities below dams where downstream temperature exceeded 20°C . Probability of trout occurring in the study streams also declined with increasing temperature. Brook trout were more sensitive to warming, which resulted in their elimination below many of the dams in this study. Slimy sculpin presence and density were also negatively related with increasing summer temperature, but not as strongly as the two trout species. Mottled sculpin presence and density were not significantly related to temperature or any of the habitat parameters measured in this study. The relationship that the two trout species have with temperature as well as the dramatic reductions noted below high impact dams, leads me to conclude that these types of dams are having important, detrimental effects on these coldwater species.

Growth of brown trout age1 and older was not related to temperature, but instead showed a significant, negative relationship with stream conductivity. These results were surprising and are most likely due to the uneven distribution of fish caught above age one. Determinants of brown trout growth for older fish in these streams needs further investigation. Age zero brown trout growth (i.e. growth of age 1 fish in the previous year) was significantly, positively related to mean summer temperature. This relationship was not surprising since temperatures, where brown trout were present, were within the tolerance range for this species.

It appears that trout populations respond to temperatures above their optimum with decreases in abundance rather than reductions in growth rates. At this time I can only speculate as to the mechanism leading to these reductions in trout density below high impact dams, but it may be that these dams also affect these species at other life stages. Increased mortality at the egg, larval or juvenile stage below high impact dams would lead to decreases in population densities downstream without necessarily leading to reductions in growth. It may be that individuals that do live to be adults feed and grow well downstream of dams, especially with reduced intra-specific competition. Once temperatures become too warm in the summer, trout species most likely migrate downstream in search of thermal refugia, and perish if none is available.

My results indicate that small, surface release dams often significantly increase downstream summer temperatures. These increases in temperature were maintained at least 2-3 miles below the dams. When this occurs, downstream communities often respond to warming below dams with reductions in plecopterans, shifts in the macroinvertebrate community, increased fish species richness, reductions in brown trout, brook trout and slimy sculpin population densities and increased growth rates in age1 brown trout. If summer temperatures below a dam exceeds 20°C, it is very likely that brook trout will be eliminated entirely from downstream waters.

In order to retain viable coldwater fisheries downstream of these types of dams, alteration of the dam operation appears necessary. Releasing water from a deeper, colder level, in stratified reservoirs, is one option. Many of these dams, however, are small facilities, with reservoirs that donot stratify and no longer produce power. Another option, which may be viable for certain communities, is to construct a channel to bypass the dam which would allow communities to keep their reservoirs and potentially also a downstream coldwater fishery. One complication though is that dams require frequent upkeep to retain their physical integrity and public safety. For this reason, the removal of dams should be seriously considered. In conclusion, communities need to consider their mutual interests and do a cost/benefit analysis when making decisions about dam maintenance, fishery management, and dam removal.

APPENDICES

APPENDIX A. Maximum R-square improvement technique (MaxR) results for the first three models for each stepwise regression analysis that used MaxR (Logistic regression did not allow for MaxR using SAS software).

Dependent Variable		**1-parameter**	**2-parameter**		**3-parameter**		
Macroinvertebrate Richness	Covariate(s)	phosphorous	phosphorous	mode substrate	conductivity	dissolved oxygen	mode substrate
	p-value	0.126	0.144	0.27	0.225	0.263	0.339
	R-square	0.05	0.075	0.075	0.096	0.096	0.096
Plecopteran Family Richness	Covariate(s)	temperature	temperature	dissolved oxygen	temperature	fish richness	dissolved oxygen
	p-value	0.009	0.071	0.072	0.023	0.15	0.097
	R-square	0.16	0.23	0.23	0.27	0.27	0.27
E.P.T. Index Values	Covariate(s)	depth	depth	width	depth	phosphorous	width
	p-value	0.047	0.005	0.049	0.003	0.18	0.02
	R-square	0.079	0.15	0.15	0.19	0.19	0.19
Macroinvertebrate Similarity	Covariate(s)	temp_change	temp_change	phos_change	temp_change	phos_change	d.o._change
	p-value	0.026	0.028	0.4	0.058	0.47	0.57
	R-square	0.59	0.65	0.65	0.68	0.68	0.68
Fish Species Richness	Covariate(s)	temperature	temperature	depth	temperature	depth	mean substrate
	p-value	0.0001	0.0001	0.08	0.0001	0.023	0.038
	R-square	0.49	0.52	0.52	0.57	0.57	0.57
Fish Species Richness Change (above-below	Covariate(s)	temp_change	temp_change	cond_change	temp_change	cond_change	depth_change
	p-value	0.003	0.0001	0.0005	0.0001	0.002	0.66
	R-square	0.79	0.98	0.98	0.99	0.99	0.99
Fish Similarity	Covariate(s)	depth_change	depth_change	d.o._change	fish_change	depth_change	d.o._change
	p-value	0.027	0.09	0.11	0.67	0.13	0.41
	R-square	0.59	0.76	0.76	0.78	0.78	0.78
Ln(brook +1)	Covariate(s)	temperature	temperature	insect richness	temperature	conductivity	dissolved oxygen
	p-value	0.0001	0.0001	0.15	0.002	0.11	0.1
	R-square	0.57	0.59	0.59	0.61	0.61	0.61

APPENDIX A. (cont'd)

Dependent Variable		**1-parameter**	**2-parameter**		**3-parameter**		
Ln(brown +1)	Covariate(s)	temperature	temperature	fish richness	temperature	fish richness	depth
	p-value	0.0001	0.0001	0.02	0.0001	0.062	0.064
	R-square	0.43	0.5	0.5	0.53	0.53	0.53
Ln(slimy +1)	Covariate(s)	temperature	temperature	insect richness	temperature	conductivity	width
	p-value	0.0001	0.0001	0.057	0.005	0.032	0.06
	R-square	0.45	0.49	0.49	0.53	0.53	0.53
Ln(mottled +1)	Covariate(s)	fish richness	fish richness	depth	temperature	fish richness	conductivity
	p-value	0.015	0.013	0.035	0.002	0.003	0.005
	R-square	0.12	0.21	0.21	0.32	0.32	0.32
Incre100	Covariate(s)	conductivity	mean substrate	mode substrate	conductivity	mean substrate	mode substrate
	p-value	0.02	0.003	0.006	0.04	0.02	0.007
	R-square	0.53	0.74	0.74	0.88	0.88	0.88
Incre200	Covariate(s)	conductivity	conductivity	phosphorous	temperature	conductivity	phosphorous
	p-value	0.006	0.003	0.07	0.004	0.0004	0.0009
	R-square	0.65	0.79	0.79	0.95	0.95	0.95
Incre300	Covariate(s)	conductivity	conductivity	phosphorous	temperature	conductivity	phosphorous
	p-value	0.004	0.02	0.04	0.002	0.0001	0.0003
	R-square	0.67	0.82	0.82	0.97	0.97	0.97
Age zero incremental growth	Covariate(s)	temperature	temperature	depth	trout density	phosphorous	mean substrate
	p-value	0.0001	0.0001	0.0001	0.0001	0.001	0.0001
	R-square	0.26	0.32	0.32	0.39	0.39	0.39

APPENDIX B: Total catch of each fish species (all passes combinded) for all sites on all ten study streams.

Stream	Site	Species	Total
BOARDMAN	1	Brook Trout	17
BOARDMAN	1	Brown Trout	62
BOARDMAN	1	Slimy Sculpin	28
BOARDMAN	2	Brook Trout	20
BOARDMAN	2	Brown Trout	27
BOARDMAN	2	Slimy Sculpin	93
BOARDMAN	2	White Sucker	3
BOARDMAN	3	Brook Trout	16
BOARDMAN	3	Brown Trout	56
BOARDMAN	3	Slimy Sculpin	58
BOARDMAN	4	Blacknose Dace	3
BOARDMAN	4	Brown Trout	38
BOARDMAN	4	Common Shiner	18
BOARDMAN	4	Creek Chub	2
BOARDMAN	4	Lamprey	2
BOARDMAN	4	Largemouth Bass	3
BOARDMAN	4	Rock Bass	8
BOARDMAN	4	Slimy Sculpin	14
BOARDMAN	4	Smallmouth Bass	1
BOARDMAN	4	Warmouth	2
BOARDMAN	4	White Sucker	51
BOARDMAN	5	Blacknose Dace	8
BOARDMAN	5	Brook Trout	1
BOARDMAN	5	Brown Trout	53
BOARDMAN	5	Common Shiner	2
BOARDMAN	5	Green Sunfish	1
BOARDMAN	5	Slimy Sculpin	13
BOARDMAN	5	Warmouth	1
BOARDMAN	5	White Sucker	2
BOARDMAN	6	Blacknose Dace	30
BOARDMAN	6	Brook Trout	5
BOARDMAN	6	Brown Trout	26
BOARDMAN	6	Common Shiner	4
BOARDMAN	6	Slimy Sculpin	36
BOARDMAN	6	White Sucker	4
CEDAR	1	Brook Trout	9
CEDAR	1	Brown Trout	18
CEDAR	1	Central Stoneroller	9
CEDAR	1	Slimy Sculpin	95
CEDAR	2	Brook Trout	2
CEDAR	2	Brown Trout	54
CEDAR	2	Slimy Sculpin	51
CEDAR	3	Brook Trout	19
CEDAR	3	Brown Trout	60
CEDAR	3	Slimy Sculpin	71
CEDAR	3	White Sucker	1
CEDAR	4	Brook Trout	2
CEDAR	4	Brown Trout	10

APPENDIX B cont'd

Stream	Site	Species	Total
CEDAR	4	Central Stoneroller	1
CEDAR	4	Largemouth Bass	3
CEDAR	4	Rock Bass	1
CEDAR	4	Slimy Sculpin	13
CEDAR	4	White Sucker	4
CEDAR	5	Bluegill	3
CEDAR	5	Common Shiner	1
CEDAR	5	Green Sunfish	3
CEDAR	5	Iowa Darter	2
CEDAR	5	Largemouth Bass	2
CEDAR	5	Logperch	2
CEDAR	5	Rock Bass	13
CEDAR	6	Common Shiner	1
CEDAR	6	Grass Pickeral	1
CEDAR	6	Green Sunfish	4
CEDAR	6	Largemouth Bass	4
CEDAR	6	Logperch	4
CEDAR	6	Rock Bass	3
CEDAR	6	Smallmouth Bass	3
CEDAR	6	Yellow Perch	12
DOWAGIAC	1	Blacknose Dace	5
DOWAGIAC	1	Bluntnose Minnow	16
DOWAGIAC	1	Common Shiner	49
DOWAGIAC	1	Creek Chub	13
DOWAGIAC	1	Green Sunfish	11
DOWAGIAC	1	Johnny Darter	14
DOWAGIAC	1	Largemouth Bass	3
DOWAGIAC	1	Mottled Sculpin	1
DOWAGIAC	1	Rainbow Darter	5
DOWAGIAC	1	River Chub	10
DOWAGIAC	1	Rock Bass	3
DOWAGIAC	1	Yellow Bullhead	1
DOWAGIAC	2	Blacknose Dace	96
DOWAGIAC	2	Bluegill	4
DOWAGIAC	2	Bluntnose Minnow	11
DOWAGIAC	2	Brown Trout	8
DOWAGIAC	2	Central Stoneroller	9
DOWAGIAC	2	Common Shiner	24
DOWAGIAC	2	Creek Chub	44
DOWAGIAC	2	Grass Pickeral	1
DOWAGIAC	2	Green Sunfish	15
DOWAGIAC	2	Johnny Darter	4
DOWAGIAC	2	Lamprey	11
DOWAGIAC	2	Largemouth Bass	1
DOWAGIAC	2	Mottled Sculpin	76
DOWAGIAC	2	Pumpkinseed	1
DOWAGIAC	2	Rock Bass	2
DOWAGIAC	2	White Sucker	48
DOWAGIAC	2	Yellow Bullhead	2

APPENDIX B cont'd

Stream	Site	Species	Total
DOWAGIAC	3	Blacknose Dace	47
DOWAGIAC	3	Bluegill	6
DOWAGIAC	3	Brown Trout	1
DOWAGIAC	3	Central Stoneroller	2
DOWAGIAC	3	Green Sunfish	1
DOWAGIAC	3	Johnny Darter	1
DOWAGIAC	3	Lamprey	5
DOWAGIAC	3	Mottled Sculpin	108
DOWAGIAC	4	Bowfin	1
DOWAGIAC	4	Common Shiner	1
DOWAGIAC	5	Bowfin	1
DOWAGIAC	5	Grass Pickeral	1
DOWAGIAC	5	Rainbow Darter	1
DOWAGIAC	5	Rock Bass	1
DOWAGIAC	5	Smallmouth Bass	2
DOWAGIAC	5	Stonecat	1
DOWAGIAC	5	Yellow Bullhead	3
DOWAGIAC	6	Common Shiner	1
DOWAGIAC	6	Creek Chub	5
DOWAGIAC	6	Green Sunfish	1
DOWAGIAC	6	Northern Hogsucker	3
DOWAGIAC	6	Rock Bass	1
DOWAGIAC	6	Stonecat	2
DOWAGIAC	7	Carp	1
DOWAGIAC	7	Johnny Darter	1
FISH	1	Blacknose Dace	51
FISH	1	Blacksided Darter	2
FISH	1	Bluegill	2
FISH	1	Bluntnose Minnow	1
FISH	1	Brook Stickleback	5
FISH	1	Brown Trout	6
FISH	1	Central Stoneroller	3
FISH	1	Common Shiner	86
FISH	1	Creek Chub	102
FISH	1	Hornyhead Chub	6
FISH	1	Johnny Darter	23
FISH	1	Lamprey	4
FISH	1	Largemouth Bass	4
FISH	1	Mottled Sculpin	13
FISH	1	Northern Hogsucker	2
FISH	1	Rainbow Darter	94
FISH	1	River Chub	9
FISH	1	Rock Bass	3
FISH	1	Rosyface Shiner	1
FISH	1	White Sucker	24
FISH	2	Blacknose Dace	49
FISH	2	Blacksided Darter	4
FISH	2	Bluegill	15
FISH	2	Brook Stickleback	13

APPENDIX B cont'd

Stream	Site	Species	Total
FISH	2	Common Shiner	106
FISH	2	Creek Chub	29
FISH	2	Hornyhead Chub	9
FISH	2	Johnny Darter	12
FISH	2	Lamprey	2
FISH	2	Mottled Sculpin	1
FISH	2	Rainbow Darter	49
FISH	2	Rosyface Shiner	4
FISH	2	White Sucker	1
FISH	5	Blacknose Dace	78
FISH	5	Blacksided Darter	27
FISH	5	Bluegill	3
FISH	5	Carp	2
FISH	5	Central Stoneroller	1
FISH	5	Common Shiner	149
FISH	5	Creek Chub	97
FISH	5	Green Sunfish	4
FISH	5	Johnny Darter	106
FISH	5	Lamprey	10
FISH	5	Logperch	6
FISH	5	Northern Hogsucker	9
FISH	5	Northern Pike	1
FISH	5	Pumpkinseed	3
FISH	5	Rainbow Darter	13
FISH	5	Rock Bass	13
FISH	5	Shorthead Redhorse	28
FISH	5	Smallmouth Bass	8
FISH	5	White Sucker	115
FISH	6	Blacknose Dace	63
FISH	6	Blacksided Darter	29
FISH	6	Bluegill	1
FISH	6	Bluntnose Minnow	2
FISH	6	Burbot	1
FISH	6	Common Shiner	183
FISH	6	Creek Chub	79
FISH	6	Emerald Shiner	1
FISH	6	Green Sunfish	1
FISH	6	Johnny Darter	33
FISH	6	Lamprey	19
FISH	6	Logperch	2
FISH	6	Northern Hogsucker	1
FISH	6	Pumpkinseed	1
FISH	6	Rock Bass	17
FISH	6	Rosyface Shiner	4
FISH	6	Shorthead Redhorse	17
FISH	6	Smallmouth Bass	17
FISH	6	White Sucker	44
FISH	6	Yellow Bullhead	1

APPENDIX B cont'd

Stream	Site	Species	Total
MANTON	1	Brook Trout	42
MANTON	1	Brown Trout	39
MANTON	1	Central Stoneroller	1
MANTON	1	Slimy Sculpin	19
MANTON	2	Brook Trout	28
MANTON	2	Brown Trout	34
MANTON	2	Central Stoneroller	1
MANTON	2	Slimy Sculpin	20
MANTON	3	Brook Trout	27
MANTON	3	Brown Trout	70
MANTON	3	Lamprey	2
MANTON	3	Mottled Sculpin	1
MANTON	3	Slimy Sculpin	26
MANTON	3	White Sucker	1
MANTON	4	Blacknose Dace	123
MANTON	4	Brook Stickleback	1
MANTON	4	Brown Trout	42
MANTON	4	Central Stoneroller	1
MANTON	4	Creek Chub	25
MANTON	4	Mottled Sculpin	14
MANTON	4	White Sucker	22
MANTON	5	Blacknose Dace	39
MANTON	5	Brown Trout	9
MANTON	5	Central Stoneroller	1
MANTON	5	Common Shiner	48
MANTON	5	Creek Chub	42
MANTON	5	Hornyhead Chub	2
MANTON	5	Lamprey	3
MANTON	5	Mottled Sculpin	2
MANTON	5	Northern Pike	2
MANTON	5	Rock Bass	3
MANTON	5	Rosyface Shiner	1
MANTON	5	Spottail Shiner	5
MANTON	5	White Sucker	15
MANTON	6	Blacknose Dace	55
MANTON	6	Brown Trout	20
MANTON	6	Common Shiner	18
MANTON	6	Creek Chub	54
MANTON	6	Northern Pike	46
MANTON	6	Pumpkinseed	1
MANTON	6	White Sucker	20
MANTON	7	Blacknose Dace	40
MANTON	7	Bluegill	3
MANTON	7	Brook Trout	2
MANTON	7	Brown Trout	19
MANTON	7	Creek Chub	27
MANTON	7	Lamprey	2
MANTON	7	Logperch	2
MANTON	7	White Sucker	4

APPENDIX B cont'd

Stream	Site	Species	Total
MAPLE	1	Emerald Shiner	6
MAPLE	1	Rock Bass	10
MAPLE	1	Slimy Sculpin	9
MAPLE	1	White Sucker	3
MAPLE	2	Brook Stickleback	1
MAPLE	2	Brook Trout	55
MAPLE	2	Central Stoneroller	3
MAPLE	2	Lamprey	1
MAPLE	2	Slimy Sculpin	40
MAPLE	2	White Sucker	1
MAPLE	3	Brook Trout	8
MAPLE	3	Brown Trout	17
MAPLE	3	Creek Chub	1
MAPLE	3	Rainbow Trout	3
MAPLE	3	Slimy Sculpin	30
MAPLE	3	White Sucker	4
MAPLE	4	Banded Killifish	1
MAPLE	4	Blacknose Dace	1
MAPLE	4	Bowfin	1
MAPLE	4	Central Stoneroller	1
MAPLE	4	Creek Chub	39
MAPLE	4	Emerald Shiner	11
MAPLE	4	Golden Shiner	7
MAPLE	4	Johnny Darter	1
MAPLE	4	Largemouth Bass	1
MAPLE	4	Northern Pike	3
MAPLE	4	White Sucker	3
MAPLE	4	Yellow Perch	1
MAPLE	5	Blacknose Dace	1
MAPLE	5	Brook Trout	4
MAPLE	5	Central Stoneroller	4
MAPLE	5	Common Shiner	13
MAPLE	5	Creek Chub	33
MAPLE	5	Lamprey	2
MAPLE	5	Mottled Sculpin	9
MAPLE	5	Northern Pike	1
MAPLE	5	Slimy Sculpin	50
MAPLE	5	White Sucker	27
MAPLE	6	Brook Trout	4
MAPLE	6	Brown Trout	7
MAPLE	6	Creek Chub	4
MAPLE	6	Rainbow Trout	3
MAPLE	6	Rock Bass	6
MAPLE	6	Slimy Sculpin	98
MAPLE	6	Yellow Perch	1
MAPLE	7	Brook Trout	4
MAPLE	7	Brown Trout	28
MAPLE	7	Rainbow Trout	5
MAPLE	7	Slimy Sculpin	36

APPENDIX B cont'd

Stream	**Site**	**Species**	**Total**
MAPLE	7	Spottail Shiner	1
MAPLE	7	White Sucker	3
MAPLE	7	Yellow Perch	1
MAPLE	8	Brook Trout	4
MAPLE	8	Brown Trout	12
MAPLE	8	Largemouth Bass	1
MAPLE	8	Mottled Sculpin	1
MAPLE	8	Rainbow Trout	2
MAPLE	8	Slimy Sculpin	33
MAPLE	8	Yellow Perch	1
MIDDLE	1	Blacknose Dace	8
MIDDLE	1	Brown Trout	4
MIDDLE	1	Central Stoneroller	4
MIDDLE	1	Creek Chub	4
MIDDLE	1	Green Sunfish	1
MIDDLE	1	Johnny Darter	1
MIDDLE	1	Lamprey	1
MIDDLE	1	Longnose Dace	1
MIDDLE	1	Mottled Sculpin	19
MIDDLE	1	Warmouth	4
MIDDLE	2	Black Bullhead	3
MIDDLE	2	Blacknose Dace	7
MIDDLE	2	Bluegill	4
MIDDLE	2	Brook Trout	1
MIDDLE	2	Brown Trout	23
MIDDLE	2	Central Stoneroller	1
MIDDLE	2	Creek Chub	10
MIDDLE	2	Longnose Dace	1
MIDDLE	2	Mottled Sculpin	50
MIDDLE	2	Warmouth	2
MIDDLE	2	White Sucker	14
MIDDLE	3	Black Bullhead	1
MIDDLE	3	Blacknose Dace	5
MIDDLE	3	Blacksided Darter	1
MIDDLE	3	Brown Trout	8
MIDDLE	3	Central Stoneroller	2
MIDDLE	3	Creek Chub	20
MIDDLE	3	Green Sunfish	4
MIDDLE	3	Johnny Darter	1
MIDDLE	3	Mottled Sculpin	24
MIDDLE	3	Warmouth	6
MIDDLE	3	White Sucker	7
MIDDLE	4	Black Bullhead	13
MIDDLE	4	Blacknose Dace	23
MIDDLE	4	Blacksided Darter	1
MIDDLE	4	Bowfin	2
MIDDLE	4	Brown Trout	4
MIDDLE	4	Common Shiner	53
MIDDLE	4	Creek Chub	27

APPENDIX B cont'd

Stream	Site	Species	Total
MIDDLE	4	Johnny Darter	1
MIDDLE	4	Lamprey	8
MIDDLE	4	Mottled Sculpin	3
MIDDLE	4	Northern Hogsucker	1
MIDDLE	4	Northern Pike	1
MIDDLE	4	Warmouth	40
MIDDLE	4	White Sucker	5
MIDDLE	5	Blacknose Dace	3
MIDDLE	5	Blacksided Darter	1
MIDDLE	5	Bowfin	3
MIDDLE	5	Central Stoneroller	5
MIDDLE	5	Common Shiner	1
MIDDLE	5	Creek Chub	23
MIDDLE	5	Johnny Darter	6
MIDDLE	5	Lamprey	1
MIDDLE	5	Mottled Sculpin	4
MIDDLE	5	Warmouth	9
MIDDLE	5	White Sucker	5
MIDDLE	6	Blacknose Dace	22
MIDDLE	6	Blacksided Darter	3
MIDDLE	6	Central Stoneroller	13
MIDDLE	6	Common Shiner	5
MIDDLE	6	Creek Chub	9
MIDDLE	6	Johnny Darter	13
MIDDLE	6	Lamprey	1
MIDDLE	6	Longnose Dace	5
MIDDLE	6	Warmouth	2
MIDDLE	6	White Sucker	2
PRAIRIE	1	Blacknose Dace	55
PRAIRIE	1	Brown Trout	6
PRAIRIE	1	Central Stoneroller	8
PRAIRIE	1	Creek Chub	63
PRAIRIE	1	Green Sunfish	2
PRAIRIE	1	Johnny Darter	18
PRAIRIE	1	Largemouth Bass	1
PRAIRIE	1	Mottled Sculpin	78
PRAIRIE	1	Pumpkinseed	1
PRAIRIE	1	Rainbow Trout	6
PRAIRIE	1	Rainbow Darter	7
PRAIRIE	1	White Sucker	34
PRAIRIE	2	Blacknose Dace	110
PRAIRIE	2	Brown Trout	9
PRAIRIE	2	Central Stoneroller	1
PRAIRIE	2	Creek Chub	50
PRAIRIE	2	Johnny Darter	5
PRAIRIE	2	Lamprey	1
PRAIRIE	2	Mottled Sculpin	2
PRAIRIE	2	Rainbow Trout	15
PRAIRIE	2	Rainbow Darter	1

APPENDIX B cont'd

Stream	Site	Species	Total
PRAIRIE	3	Blacknose Dace	26
PRAIRIE	3	Bluegill	1
PRAIRIE	3	Brown Trout	1
PRAIRIE	3	Creek Chub	31
PRAIRIE	3	Green Sunfish	1
PRAIRIE	3	Johnny Darter	20
PRAIRIE	3	Mottled Sculpin	20
PRAIRIE	3	Rainbow Trout	4
PRAIRIE	3	Rainbow Darter	4
PRAIRIE	3	White Sucker	1
PRAIRIE	4	Blacknose Dace	40
PRAIRIE	4	Brown Trout	10
PRAIRIE	4	Central Stoneroller	2
PRAIRIE	4	Creek Chub	90
PRAIRIE	4	Green Sunfish	16
PRAIRIE	4	Johnny Darter	15
PRAIRIE	4	Lamprey	5
PRAIRIE	4	Largemouth Bass	1
PRAIRIE	4	Mottled Sculpin	49
PRAIRIE	4	Pumpkinseed	1
PRAIRIE	4	Rainbow Trout	11
PRAIRIE	4	Rainbow Darter	32
PRAIRIE	4	Warmouth	4
PRAIRIE	4	White Crappie	2
PRAIRIE	4	White Sucker	11
PRAIRIE	5	Blacknose Dace	42
PRAIRIE	5	Brown Trout	1
PRAIRIE	5	Creek Chub	25
PRAIRIE	5	Johnny Darter	6
PRAIRIE	5	Mottled Sculpin	10
PRAIRIE	5	Rainbow Trout	1
PRAIRIE	5	Rainbow Darter	17
PRAIRIE	5	White Sucker	2
PRAIRIE	6	Blacknose Dace	12
PRAIRIE	6	Brown Trout	2
PRAIRIE	6	Central Stoneroller	1
PRAIRIE	6	Creek Chub	36
PRAIRIE	6	Johnny Darter	10
PRAIRIE	6	Lamprey	1
PRAIRIE	6	Mottled Sculpin	2
SUGAR	1	Blacknose Dace	2
SUGAR	1	Green Sunfish	1
SUGAR	1	Northern Pike	3
SUGAR	1	Warmouth	1
SUGAR	1	White Sucker	1
SUGAR	2	Blacknose Dace	19
SUGAR	2	Bluegill	1
SUGAR	2	Brown Trout	16
SUGAR	2	Central Stoneroller	1

APPENDIX B cont'd

Stream	Site	Species	Total
SUGAR	2	Common Shiner	1
SUGAR	2	Largemouth Bass	3
SUGAR	2	Mottled Sculpin	1
SUGAR	2	Northern Pike	5
SUGAR	2	Pumpkinseed	1
SUGAR	3	Bluegill	10
SUGAR	3	Brown Trout	16
SUGAR	3	Green Sunfish	1
SUGAR	3	Largemouth Bass	3
SUGAR	3	Mottled Sculpin	4
SUGAR	3	Northern Pike	1
SUGAR	3	Rock Bass	1
SUGAR	4	Blacksided Darter	1
SUGAR	4	Bluegill	3
SUGAR	4	Common Shiner	2
SUGAR	4	Creek Chub	6
SUGAR	4	Green Sunfish	5
SUGAR	4	Logperch	1
SUGAR	4	Longnose Dace	1
SUGAR	4	Rainbow Darter	1
SUGAR	4	Rock Bass	2
SUGAR	4	Warmouth	1
SUGAR	4	White Crappie	5
SUGAR	4	White Sucker	7
SUGAR	4	Yellow Bullhead	5
SUGAR	5	Blacknose Dace	1
SUGAR	5	Common Shiner	20
SUGAR	5	Creek Chub	3
SUGAR	5	Green Sunfish	1
SUGAR	5	Hornyhead Chub	6
SUGAR	5	Lamprey	1
SUGAR	5	Logperch	2
SUGAR	5	Northern Hogsucker	1
SUGAR	5	Rainbow Darter	3
SUGAR	5	White Crappie	1
SUGAR	5	White Sucker	22
SUGAR	6	Blacknose Dace	2
SUGAR	6	Blacksided Darter	6
SUGAR	6	Bluegill	1
SUGAR	6	Brown bu	2
SUGAR	6	Central Stoneroller	4
SUGAR	6	Common Shiner	7
SUGAR	6	Creek Chub	1
SUGAR	6	Green Sunfish	1
SUGAR	6	Hornyhead Chub	1
SUGAR	6	Iowa Darter	2
SUGAR	6	Lamprey	2
SUGAR	6	Logperch	3
SUGAR	6	Longnose Dace	2

APPENDIX B cont'd

Stream	Site	Species	Total
SUGAR	6	Rainbow Darter	6
SUGAR	6	Rock Bass	2
SUGAR	6	White Crappie	1
SUGAR	6	Yellow Bullhead	1
WHITE	1	Blacknose Dace	2
WHITE	1	Brook Trout	5
WHITE	1	Brown Trout	7
WHITE	1	Creek Chub	1
WHITE	1	Green Sunfish	1
WHITE	1	Mottled Sculpin	26
WHITE	1	White Sucker	16
WHITE	2	Brook Trout	7
WHITE	2	Brown Trout	13
WHITE	2	Mottled Sculpin	44
WHITE	2	White Sucker	31
WHITE	3	Blacknose Dace	6
WHITE	3	Brown Trout	6
WHITE	3	Central Stoneroller	1
WHITE	3	Creek Chub	5
WHITE	3	Longnose Dace	6
WHITE	3	Mottled Sculpin	29
WHITE	3	Northern Pike	1
WHITE	3	White Sucker	7
WHITE	4	Blacknose Dace	72
WHITE	4	Brown Trout	17
WHITE	4	Common Shiner	20
WHITE	4	Creek Chub	10
WHITE	4	Green Sunfish	1
WHITE	4	Johnny Darter	2
WHITE	4	Lamprey	2
WHITE	4	Longnose Dace	2
WHITE	4	Mottled Sculpin	11
WHITE	4	Rainbow Darter	2
WHITE	4	White Sucker	52
WHITE	5	Blacknose Dace	20
WHITE	5	Brown Trout	10
WHITE	5	Central Stoneroller	5
WHITE	5	Common Shiner	12
WHITE	5	Creek Chub	10
WHITE	5	Hornyhead Chub	1
WHITE	5	Iowa Darter	1
WHITE	5	Johnny Darter	11
WHITE	5	Lamprey	2
WHITE	5	Longnose Dace	1
WHITE	5	Mottled Sculpin	25
WHITE	5	Pumpkinseed	2
WHITE	5	White Sucker	24

APPENDIX C: Total number of macroinvertebrates per sample and per square meter in each site for all ten study streams.

Stream	Site	Taxa	Total	No./Sq.M
BOARDMAN	1	Tipulidae (Antocha)	2	0.065
BOARDMAN	1	Athericidae	11	0.356
BOARDMAN	1	Baetidae	39	1.264
BOARDMAN	1	Brachycentridae	1	0.032
BOARDMAN	1	Chironomidae	320	10.368
BOARDMAN	1	Elmidae	56	1.814
BOARDMAN	1	Empididae	17	0.551
BOARDMAN	1	Ephemeridae	5	0.162
BOARDMAN	1	Glossosomatidae	5	0.162
BOARDMAN	1	Gomphidae	1	0.032
BOARDMAN	1	Hydropsychidae	10	0.324
BOARDMAN	1	Limnephilidae	4	0.130
BOARDMAN	1	Oligocheates	11	0.356
BOARDMAN	1	Perlodidae	1	0.032
BOARDMAN	1	Philipotamidae	1	0.032
BOARDMAN	1	Simulidae	53	1.717
BOARDMAN	1	Tricorythidae	10	0.324
BOARDMAN	2	Baetidae	18	0.583
BOARDMAN	2	Chironomidae	190	6.156
BOARDMAN	2	Elmidae	12	0.389
BOARDMAN	2	Empididae	1	0.032
BOARDMAN	2	Ephemeridae	1	0.032
BOARDMAN	2	Gomphidae	2	0.065
BOARDMAN	2	Hydropsychidae	10	0.324
BOARDMAN	2	Isopoda	1	0.032
BOARDMAN	2	Oligocheates	25	0.810
BOARDMAN	2	Perlodidae	1	0.032
BOARDMAN	2	Pteronarcyidae	2	0.065
BOARDMAN	2	Simulidae	29	0.940
BOARDMAN	2	Tabanidae	2	0.065
BOARDMAN	2	Tricorythidae	16	0.518
BOARDMAN	3	Tipulidae (Antocha)	1	0.032
BOARDMAN	3	Athericidae	6	0.194
BOARDMAN	3	Baetidae	3	0.097
BOARDMAN	3	Chironomidae	92	2.981
BOARDMAN	3	Elmidae	5	0.162
BOARDMAN	3	Ephemeridae	2	0.065
BOARDMAN	3	Heptageniidae	1	0.032
BOARDMAN	3	Nemouridae	1	0.032
BOARDMAN	3	Oligocheates	9	0.292
BOARDMAN	3	Simulidae	5	0.162
BOARDMAN	4	Tipulidae (Antocha)	3	0.097
BOARDMAN	4	Baetidae	10	0.324
BOARDMAN	4	Chironomidae	38	1.231
BOARDMAN	4	Elmidae	52	1.685
BOARDMAN	4	Ephemeridae	5	0.162
BOARDMAN	4	Hydropsychidae	138	4.471
BOARDMAN	4	Hirudinea	19	0.616

APPENDIX C cont'd

Stream	Site	Taxa	Total	No./Sq.M
BOARDMAN	4	Lepidstomatidae	5	0.162
BOARDMAN	4	Oligocheates	60	1.944
BOARDMAN	4	Simulidae	13	0.421
BOARDMAN	4	Tricorythidae	1	0.032
BOARDMAN	5	Tipulidae (Antocha)	10	0.324
BOARDMAN	5	Athericidae	3	0.097
BOARDMAN	5	Baetidae	6	0.194
BOARDMAN	5	Brachycentridae	11	0.356
BOARDMAN	5	Chironomidae	20	0.648
BOARDMAN	5	Elmidae	93	3.013
BOARDMAN	5	Ephemeridae	8	0.259
BOARDMAN	5	Hydropsychidae	21	0.680
BOARDMAN	5	Limnephilidae	1	0.032
BOARDMAN	5	Oligocheates	7	0.227
BOARDMAN	5	Simulidae	1	0.032
BOARDMAN	5	Tricorythidae	2	0.065
BOARDMAN	6	Tipulidae (Antocha)	13	0.421
BOARDMAN	6	Athericidae	4	0.130
BOARDMAN	6	Baetidae	11	0.356
BOARDMAN	6	Brachycentridae	12	0.389
BOARDMAN	6	Chironomidae	150	4.860
BOARDMAN	6	Elmidae	38	1.231
BOARDMAN	6	Ephemeridae	9	0.292
BOARDMAN	6	Glossosomatidae	1	0.032
BOARDMAN	6	Helicopsychidae	1	0.032
BOARDMAN	6	Heptageniidae	4	0.130
BOARDMAN	6	Hydropsychidae	10	0.324
BOARDMAN	6	Hirudinea	4	0.130
BOARDMAN	6	Philipotamidae	12	0.389
BOARDMAN	6	Chironomidae (Rheotanytarsus)	80	2.592
BOARDMAN	6	Rhyacophilidae	2	0.065
BOARDMAN	6	Simulidae	5	0.162
BOARDMAN	6	Tricorythidae	2	0.065
CEDAR	1	Baetidae	25	0.810
CEDAR	1	Brachycentridae	4	0.130
CEDAR	1	Ceratopogonidae	1	0.032
CEDAR	1	Chironomidae	94	3.046
CEDAR	1	Elmidae	6	0.194
CEDAR	1	Ephemeridae	5	0.162
CEDAR	1	Hydropsychidae	43	1.393
CEDAR	1	Nemouridae	5	0.162
CEDAR	1	Perlodidae	1	0.032
CEDAR	1	Philipotamidae	3	0.097
CEDAR	1	Rhyacophilidae	1	0.032
CEDAR	1	Simulidae	2	0.065
CEDAR	2	Amphipoda	12	0.389
CEDAR	2	Tipulidae (Antocha)	1	0.032
CEDAR	2	Baetidae	54	1.750
CEDAR	2	Brachycentridae	1	0.032

APPENDIX C cont'd

Stream	Site	Taxa	Total	No./Sq.M
CEDAR	2	Ceratopogonidae	2	0.065
CEDAR	2	Chironomidae	105	3.402
CEDAR	2	Elmidae	19	0.616
CEDAR	2	Ephemeridae	7	0.227
CEDAR	2	Hydropsychidae	20	0.648
CEDAR	2	Oligocheates	1	0.032
CEDAR	2	Philipotamidae	15	0.486
CEDAR	2	Rhyacophilidae	3	0.097
CEDAR	2	Simulidae	94	3.046
CEDAR	2	Tabanidae	4	0.130
CEDAR	3	Amphipoda	8	0.259
CEDAR	3	Baetidae	102	3.305
CEDAR	3	Chironomidae	73	2.365
CEDAR	3	Elmidae	23	0.745
CEDAR	3	Ephemeridae	2	0.065
CEDAR	3	Glossosomatidae	36	1.166
CEDAR	3	Hydropsychidae	66	2.138
CEDAR	3	Isopoda	2	0.065
CEDAR	3	Nemouridae	7	0.227
CEDAR	3	Oligocheates	3	0.097
CEDAR	3	Perlodidae	1	0.032
CEDAR	3	Pteronarcyidae	2	0.065
CEDAR	3	Rhyacophilidae	3	0.097
CEDAR	3	Simulidae	145	4.698
CEDAR	4	Amphipoda	1	0.032
CEDAR	4	Baetidae	10	0.324
CEDAR	4	Chironomidae	115	3.726
CEDAR	4	Elmidae	3	0.097
CEDAR	4	Hydropsychidae	111	3.596
CEDAR	4	Isopoda	8	0.259
CEDAR	4	Philipotamidae	28	0.907
CEDAR	4	Simulidae	58	1.879
CEDAR	5	Aeshnidae	1	0.032
CEDAR	5	Amphipoda	89	2.884
CEDAR	5	Baetidae	1	0.032
CEDAR	5	Brachycentridae	1	0.032
CEDAR	5	Ceanidae	1	0.032
CEDAR	5	Chironomidae	13	0.421
CEDAR	5	Coenagrionidae	13	0.421
CEDAR	5	Elmidae	6	0.194
CEDAR	5	Ephemeridae	3	0.097
CEDAR	5	Isopoda	2	0.065
CEDAR	5	Mesoveliidae	1	0.032
CEDAR	5	Naucoridae	1	0.032
CEDAR	5	Nepidae	1	0.032
CEDAR	5	Polycentropodidae	1	0.032
DOWAGIAC	1	Amphipoda	18	0.583
DOWAGIAC	1	Baetidae	1	0.032
DOWAGIAC	1	Brachycentridae	1	0.032

APPENDIX C cont'd

Stream	Site	Taxa	Total	No./Sq.M
DOWAGIAC	1	Ceanidae	14	0.454
DOWAGIAC	1	Ceratopogonidae	1	0.032
DOWAGIAC	1	Chironomidae	25	0.810
DOWAGIAC	1	Bivalvia	20	0.648
DOWAGIAC	1	Elmidae	13	0.421
DOWAGIAC	1	Ephemeridae	3	0.097
DOWAGIAC	1	Heptageniidae	12	0.389
DOWAGIAC	1	Hydropsychidae	10	0.324
DOWAGIAC	1	Lepidstomatidae	1	0.032
DOWAGIAC	2	Amphipoda	4	0.130
DOWAGIAC	2	Tipulidae (Antocha)	2	0.065
DOWAGIAC	2	Baetidae	4	0.130
DOWAGIAC	2	Brachycentridae	2	0.065
DOWAGIAC	2	Ceanidae	24	0.778
DOWAGIAC	2	Chironomidae	276	8.942
DOWAGIAC	2	Elmidae	16	0.518
DOWAGIAC	2	Ephemeridae	12	0.389
DOWAGIAC	2	Glossosomatidae	2	0.065
DOWAGIAC	2	Helicopsychidae	14	0.454
DOWAGIAC	2	Hydropsychidae	38	1.231
DOWAGIAC	2	Limnephilidae	2	0.065
DOWAGIAC	2	Oligocheates	2	0.065
DOWAGIAC	2	Gastropoda	12	0.389
DOWAGIAC	2	Tabanidae	2	0.065
DOWAGIAC	2	Tricorythidae	4	0.130
DOWAGIAC	3	Amphipoda	2	0.065
DOWAGIAC	3	Baetidae	12	0.389
DOWAGIAC	3	Ceanidae	2	0.065
DOWAGIAC	3	Chironomidae	58	1.879
DOWAGIAC	3	Elmidae	32	1.037
DOWAGIAC	3	Glossosomatidae	20	0.648
DOWAGIAC	3	Helicopsychidae	6	0.194
DOWAGIAC	3	Heptageniidae	2	0.065
DOWAGIAC	3	Hydropsychidae	144	4.666
DOWAGIAC	3	Oligocheates	6	0.194
DOWAGIAC	3	Perlidae	2	0.065
DOWAGIAC	4	Amphipoda	2	0.065
DOWAGIAC	4	Calopterygidae	3	0.097
DOWAGIAC	4	Chironomidae	8	0.259
DOWAGIAC	4	Bivalvia	2	0.065
DOWAGIAC	4	Culicidae	1	0.032
DOWAGIAC	4	Elmidae	27	0.875
DOWAGIAC	4	Helicopsychidae	2	0.065
DOWAGIAC	4	Heptageniidae	3	0.097
DOWAGIAC	4	Hydropsychidae	1	0.032
DOWAGIAC	4	Isopoda	1	0.032
DOWAGIAC	4	Uenoidae (Neophylax)	3	0.097
DOWAGIAC	4	Oligocheates	2	0.065
DOWAGIAC	4	Psphenidae	1	0.032

APPENDIX C cont'd

Stream	Site	Taxa	Total	No./Sq.M
DOWAGIAC	4	Gastropoda	2	0.065
DOWAGIAC	4	Tabanidae	2	0.065
DOWAGIAC	5	Amphipoda	2	0.065
DOWAGIAC	5	Baetidae	2	0.065
DOWAGIAC	5	Elmidae	7	0.227
DOWAGIAC	5	Hydropsychidae	2	0.065
DOWAGIAC	5	Philipotamidae	1	0.032
DOWAGIAC	5	Polycentropodidae	2	0.065
DOWAGIAC	5	Psphenidae	3	0.097
DOWAGIAC	6	Brachycentridae	3	0.097
DOWAGIAC	6	Chironomidae	42	1.361
DOWAGIAC	6	Elmidae	25	0.810
DOWAGIAC	6	Helicopsychidae	1	0.032
DOWAGIAC	6	Hydropsychidae	3	0.097
DOWAGIAC	6	Limnephilidae	2.	0.065
DOWAGIAC	6	Gastropoda	7	0.227
FISH	1	Baetidae	102	3.305
FISH	1	Brachycentridae	8	0.259
FISH	1	Chironomidae	166	5.378
FISH	1	Elmidae	44	1.426
FISH	1	Ephemeridae	40	1.296
FISH	1	Heptageniidae	4	0.130
FISH	1	Hydropsychidae	98	3.175
FISH	1	Limnephilidae	8	0.259
FISH	1	Perlidae	18	0.583
FISH	1	Psphenidae	2	0.065
FISH	1	Pteronarcyidae	2	0.065
FISH	1	Sialidae	2	0.065
FISH	1	Tricorythidae	2	0.065
FISH	2	Tipulidae (Antocha)	1	0.032
FISH	2	Athericidae	52	1.685
FISH	2	Baetidae	38	1.231
FISH	2	Brachycentridae	1	0.032
FISH	2	Ceratopogonidae	5	0.162
FISH	2	Chironomidae	402	13.025
FISH	2	Elmidae	102	3.305
FISH	2	Ephemeridae	58	1.879
FISH	2	Helicopsychidae	4	0.130
FISH	2	Heptageniidae	4	0.130
FISH	2	Hydropsychidae	420	13.608
FISH	2	Isonychidae	1	0.032
FISH	2	Limnephilidae	3	0.097
FISH	2	Perlidae	18	0.583
FISH	2	Simulidae	51	1.652
FISH	2	Tricorythidae	34	1.102
FISH	3	Amphipoda	1	0.032
FISH	3	Athericidae	1	0.032
FISH	3	Ceanidae	30	0.972
FISH	3	Chironomidae	40	1.296

APPENDIX C cont'd

Stream	Site	Taxa	Total	No./Sq.M
FISH	3	Elmidae	8	0.259
FISH	3	Ephemeridae	13	0.421
FISH	3	Heptageniidae	10	0.324
FISH	3	Hydropsychidae	11	0.356
FISH	3	Limnephilidae	2	0.065
FISH	3	Perlidae	5	0.162
FISH	3	Chironomidae (Rheotanytarsus)	10	0.324
FISH	3	Sialidae	1	0.032
FISH	3	Tipulidae	1	0.032
FISH	3	Tricorythidae	9	0.292
FISH	4	Amphipoda	1	0.032
FISH	4	Baetidae	2	0.065
FISH	4	Brachycentridae	1	0.032
FISH	4	Chironomidae	218	7.063
FISH	4	Elmidae	29	0.940
FISH	4	Ephemeridae	10	0.324
FISH	4	Hydropsychidae	16	0.518
FISH	4	Isonychidae	2	0.065
FISH	4	Perlidae	15	0.486
FISH	4	Simulidae	7	0.227
FISH	5	Chironomidae	189	6.124
FISH	5	Elmidae	1	0.032
FISH	5	Tricorythidae	3	0.097
FISH	6	Amphipoda	1	0.032
FISH	6	Tipulidae (Antocha)	2	0.065
FISH	6	Athericidae	1	0.032
FISH	6	Ceanidae	7	0.227
FISH	6	Chironomidae	33	1.069
FISH	6	Bivalvia	1	0.032
FISH	6	Elmidae	11	0.356
FISH	6	Ephemeridae	6	0.194
FISH	6	Heptageniidae	10	0.324
FISH	6	Hydropsychidae	9	0.292
FISH	6	Isonychidae	7	0.227
FISH	6	Isopoda	1	0.032
FISH	6	Sialidae	1	0.032
FISH	6	Tricorythidae	1	0.032
MANTON	1	Amphipoda	3	0.097
MANTON	1	Tipulidae (Antocha)	1	0.032
MANTON	1	Athericidae	1	0.032
MANTON	1	Baetidae	18	0.583
MANTON	1	Brachycentridae	2	0.065
MANTON	1	Chironomidae	237	7.679
MANTON	1	Chloroperlidae	1	0.032
MANTON	1	Nemouridae	4	0.130
MANTON	1	Perlidae	1	0.032
MANTON	2	Amphipoda	2	0.065
MANTON	2	Athericidae	20	0.648
MANTON	2	Baetidae	14	0.454

APPENDIX C cont'd

Stream	Site	Taxa	Total	No./Sq.M
MANTON	2	Brachycentridae	14	0.454
MANTON	2	Ceanidae	8	0.259
MANTON	2	Chironomidae	74	2.398
MANTON	2	Elmidae	52	1.685
MANTON	2	Ephemeridae	16	0.518
MANTON	2	Hydropsychidae	366	11.858
MANTON	2	Leptoceridae	2	0.065
MANTON	2	Perlidae	10	0.324
MANTON	2	Philipotamidae	2	0.065
MANTON	2	Pteronarcyidae	2	0.065
MANTON	2	Chironomidae (Rheotanytarsus)	2	0.065
MANTON	2	Simulidae	2	0.065
MANTON	2	Tricorythidae	44	1.426
MANTON	3	Amphipoda	31	1.004
MANTON	3	Tipulidae (Antocha)	12	0.389
MANTON	3	Baetidae	38	1.231
MANTON	3	Brachycentridae	1	0.032
MANTON	3	Chironomidae	65	2.106
MANTON	3	Elmidae	3	0.097
MANTON	3	Ephemeridae	1	0.032
MANTON	3	Hydropsychidae	2	0.065
MANTON	3	Oligocheates	4	0.130
MANTON	3	Perlidae	6	0.194
MANTON	3	Philipotamidae	1	0.032
MANTON	3	Tipulidae	1	0.032
MANTON	4	Baetidae	8	0.259
MANTON	4	Chironomidae	500	16.200
MANTON	4	Corydalidae	1	0.032
MANTON	4	Elmidae	79	2.560
MANTON	4	Heptageniidae	1	0.032
MANTON	4	Hydropsychidae	9	0.292
MANTON	4	Gastropoda	1	0.032
MANTON	4	Tricorythidae	4	0.130
MANTON	5	Tipulidae (Antocha)	3	0.097
MANTON	5	Baetidae	1	0.032
MANTON	5	Chironomidae	43	1.393
MANTON	5	Bivalvia	2	0.065
MANTON	5	Elmidae	18	0.583
MANTON	5	Ephemeridae	2	0.065
MANTON	5	Glossosomatidae	5	0.162
MANTON	5	Heptageniidae	1	0.032
MANTON	5	Hydropsychidae	50	1.620
MANTON	5	Isonychidae	1	0.032
MANTON	5	Philipotamidae	1	0.032
MANTON	5	Simulidae	7	0.227
MANTON	6	Amphipoda	2	0.065
MANTON	6	Baetidae	6	0.194
MANTON	6	Ceanidae	1	0.032
MANTON	6	Ceratopogonidae	1	0.032

APPENDIX C cont'd

Stream	Site	Taxa	Total	No./Sq.M
MANTON	6	Chironomidae	10	0.324
MANTON	6	Bivalvia	3	0.097
MANTON	6	Tipulidae (Dicranota)	1	0.032
MANTON	6	Elmidae	64	2.074
MANTON	6	Empididae	1	0.032
MANTON	6	Glossosomatidae	1	0.032
MANTON	6	Gomphidae	1	0.032
MANTON	6	Hydropsychidae	2	0.065
MANTON	6	Limnephilidae	1	0.032
MAPLE	1	Amphipoda	11	0.356
MAPLE	1	Baetidae	20	0.648
MAPLE	1	Calopterygidae	4	0.130
MAPLE	1	Chironomidae	17	0.551
MAPLE	1	Hydropsychidae	4	0.130
MAPLE	1	Limnephilidae	2	0.065
MAPLE	1	Rhyacophilidae	1	0.032
MAPLE	1	Simulidae	11	0.356
MAPLE	1	Gastropoda	1	0.032
MAPLE	2	Tipulidae (Antocha)	1	0.032
MAPLE	2	Athericidae	1	0.032
MAPLE	2	Baetidae	14	0.454
MAPLE	2	Brachycentridae	40	1.296
MAPLE	2	Chironomidae	122	3.953
MAPLE	2	Elmidae	6	0.194
MAPLE	2	Ephemeridae	1	0.032
MAPLE	2	Heptageniidae	4	0.130
MAPLE	2	Hydropsychidae	124	4.018
MAPLE	2	Hirudinea	4	0.130
MAPLE	2	Perlidae	1	0.032
MAPLE	2	Perlodidae	3	0.097
MAPLE	2	Philipotamidae	1	0.032
MAPLE	2	Polycentropodidae	7	0.227
MAPLE	2	Simulidae	210	6.804
MAPLE	2	Tricorythidae	13	0.421
MAPLE	3	Tipulidae (Antocha)	3	0.097
MAPLE	3	Baetidae	45	1.458
MAPLE	3	Brachycentridae	111	3.596
MAPLE	3	Calopterygidae	2	0.065
MAPLE	3	Chironomidae	236	7.646
MAPLE	3	Corydalidae	2	0.065
MAPLE	3	Elmidae	31	1.004
MAPLE	3	Empididae	1	0.032
MAPLE	3	Heptageniidae	8	0.259
MAPLE	3	Hydropsychidae	49	1.588
MAPLE	3	Isopoda	2	0.065
MAPLE	3	Oligocheates	1	0.032
MAPLE	3	Perlidae	2	0.065
MAPLE	3	Perlodidae	8	0.259
MAPLE	3	Philipotamidae	9	0.292

APPENDIX C cont'd

Stream	Site	Taxa	Total	No./Sq.M
MAPLE	3	Polycentropodidae	1	0.032
MAPLE	3	Pteronarcyidae	1	0.032
MAPLE	3	Chironomidae (Rheotanytarsus)	35	1.134
MAPLE	3	Simulidae	85	2.754
MAPLE	3	Tricorythidae	40	1.296
MAPLE	4	Amphipoda	75	2.430
MAPLE	4	Brachycentridae	25	0.810
MAPLE	4	Chironomidae	270	8.748
MAPLE	4	Corixidae	1	0.032
MAPLE	4	Elmidae	17	0.551
MAPLE	4	Hydropsychidae	27	0.875
MAPLE	4	Hydroptilidae	1	0.032
MAPLE	4	Isopoda	18	0.583
MAPLE	4	Hirudinea	36	1.166
MAPLE	4	Oligocheates	5	0.162
MAPLE	4	Chironomidae (Rheotanytarsus)	27	0.875
MAPLE	4	Simulidae	3	0.097
MAPLE	4	Gastropoda	22	0.713
MAPLE	4	Tipulidae	1	0.032
MAPLE	5	Tipulidae (Antocha)	1	0.032
MAPLE	5	Baetidae	47	1.523
MAPLE	5	Brachycentridae	8	0.259
MAPLE	5	Chironomidae	35	1.134
MAPLE	5	Elmidae	3	0.097
MAPLE	5	Heptageniidae	2	0.065
MAPLE	5	Hydropsychidae	35	1.134
MAPLE	5	Isopoda	1	0.032
MAPLE	5	Oligocheates	1	0.032
MAPLE	5	Perlidae	1	0.032
MAPLE	5	Perlodidae	1	0.032
MAPLE	5	Polycentropodidae	1	0.032
MAPLE	5	Chironomidae (Rheotanytarsus)	3	0.097
MAPLE	5	Simulidae	104	3.370
MAPLE	5	Gastropoda	1	0.032
MAPLE	5	Tricorythidae	1	0.032
MAPLE	6	Baetidae	5	0.162
MAPLE	6	Brachycentridae	3	0.097
MAPLE	6	Calopterygidae	1	0.032
MAPLE	6	Chironomidae	50	1.620
MAPLE	6	Ephemeridae	4	0.130
MAPLE	6	Heptageniidae	4	0.130
MAPLE	6	Hydropsychidae	17	0.551
MAPLE	6	Isopoda	6	0.194
MAPLE	6	Limnephilidae	3	0.097
MAPLE	6	Perlodidae	2	0.065
MAPLE	6	Polycentropodidae	8	0.259
MAPLE	6	Pteronarcyidae	3	0.097
MAPLE	6	Simulidae	3	0.097
MAPLE	6	Gastropoda	1	0.032

APPENDIX C cont'd

Stream	Site	Taxa	Total	No./Sq.M
MAPLE	6	Tricorythidae	2	0.065
MIDDLE	1	Amphipoda	22	0.713
MIDDLE	1	Tipulidae (Antocha)	3	0.097
MIDDLE	1	Baetidae	8	0.259
MIDDLE	1	Chironomidae	181	5.864
MIDDLE	1	Bivalvia	13	0.421
MIDDLE	1	Elmidae	8	0.259
MIDDLE	1	Hydropsychidae	8	0.259
MIDDLE	1	Psychomyiidae	7	0.227
MIDDLE	1	Tabanidae	2	0.065
MIDDLE	1	Tipulidae	1	0.032
MIDDLE	1	Tricorythidae	2	0.065
MIDDLE	2	Amphipoda	2	0.065
MIDDLE	2	Tipulidae (Antocha)	34	1.102
MIDDLE	2	Athericidae	1	0.032
MIDDLE	2	Baetidae	3	0.097
MIDDLE	2	Chironomidae	106	3.434
MIDDLE	2	Elmidae	7	0.227
MIDDLE	2	Ephemeridae	1	0.032
MIDDLE	2	Glossosomatidae	1	0.032
MIDDLE	2	Heptageniidae	2	0.065
MIDDLE	2	Hydropsychidae	31	1.004
MIDDLE	2	Hirudinea	1	0.032
MIDDLE	2	Perlidae	1	0.032
MIDDLE	2	Philipotamidae	1	0.032
MIDDLE	2	Psychomyiidae	2	0.065
MIDDLE	2	Rhyacophilidae	1	0.032
MIDDLE	2	Simulidae	43	1.393
MIDDLE	2	Stratiomyidae	1	0.032
MIDDLE	2	Tricorythidae	11	0.356
MIDDLE	3	Amphipoda	3	0.097
MIDDLE	3	Tipulidae (Antocha)	36	1.166
MIDDLE	3	Athericidae	3	0.097
MIDDLE	3	Baetidae	1	0.032
MIDDLE	3	Brachycentridae	10	0.324
MIDDLE	3	Chironomidae	275	8.910
MIDDLE	3	Elmidae	3	0.097
MIDDLE	3	Heptageniidae	5	0.162
MIDDLE	3	Hydropsychidae	1	0.032
MIDDLE	3	Hydroptilidae	2	0.065
MIDDLE	3	Isonychidae	2	0.065
MIDDLE	3	Limnephilidae	1	0.032
MIDDLE	3	Perlodidae	1	0.032
MIDDLE	3	Psychomyiidae	7	0.227
MIDDLE	3	Simulidae	1	0.032
MIDDLE	3	Tricorythidae	29	0.940
MIDDLE	4	Amphipoda	2	0.065
MIDDLE	4	Athericidae	1	0.032
MIDDLE	4	Ceanidae	2	0.065

APPENDIX C cont'd

Stream	Site	Taxa	Total	No./Sq.M
MIDDLE	4	Chironomidae	55	1.782
MIDDLE	4	Elmidae	1	0.032
MIDDLE	4	Tabanidae	4	0.130
MIDDLE	5	Amphipoda	1	0.032
MIDDLE	5	Tipulidae (Antocha)	5	0.162
MIDDLE	5	Athericidae	4	0.130
MIDDLE	5	Baetidae	3	0.097
MIDDLE	5	Brachycentridae	1	0.032
MIDDLE	5	Chironomidae	49	1.588
MIDDLE	5	Elmidae	7	0.227
MIDDLE	5	Glossosomatidae	5	0.162
MIDDLE	5	Heptageniidae	3	0.097
MIDDLE	5	Hydropsychidae	142	4.601
MIDDLE	5	Hydroptilidae	32	1.037
MIDDLE	5	Perlidae	4	0.130
MIDDLE	5	Psphenidae	1	0.032
MIDDLE	5	Psychomyiidae	1	0.032
MIDDLE	6	Tipulidae (Antocha)	7	0.227
MIDDLE	6	Baetidae	5	0.162
MIDDLE	6	Ceanidae	2	0.065
MIDDLE	6	Chironomidae	49	1.588
MIDDLE	6	Corydalidae	1	0.032
MIDDLE	6	Elmidae	12	0.389
MIDDLE	6	Heptageniidae	3	0.097
MIDDLE	6	Hydropsychidae	30	0.972
MIDDLE	6	Hydroptilidae	3	0.097
MIDDLE	6	Isonychidae	1	0.032
MIDDLE	6	Psphenidae	1	0.032
MIDDLE	6	Tabanidae	3	0.097
PRAIRIE	1	Baetidae	25	0.810
PRAIRIE	1	Chironomidae	70	2.268
PRAIRIE	1	Corydalidae	3	0.097
PRAIRIE	1	Elmidae	3	0.097
PRAIRIE	1	Ephemeridae	6	0.194
PRAIRIE	1	Helicopsychidae	13	0.421
PRAIRIE	1	Heptageniidae	9	0.292
PRAIRIE	1	Hydropsychidae	1	0.032
PRAIRIE	1	Hydroptilidae	2	0.065
PRAIRIE	1	Limnephilidae	12	0.389
PRAIRIE	1	Perlidae	1	0.032
PRAIRIE	1	Polycentropodidae	1	0.032
PRAIRIE	2	Athericidae	2	0.065
PRAIRIE	2	Baetidae	28	0.907
PRAIRIE	2	Chironomidae	38	1.231
PRAIRIE	2	Elmidae	2	0.065
PRAIRIE	2	Ephemeridae	51	1.652
PRAIRIE	2	Helicopsychidae	1	0.032
PRAIRIE	2	Heptageniidae	9	0.292
PRAIRIE	2	Hydropsychidae	193	6.253

APPENDIX C cont'd

Stream	Site	Taxa	Total	No./Sq.M
PRAIRIE	2	Limnephilidae	5	0.162
PRAIRIE	2	Perlidae	1	0.032
PRAIRIE	2	Philipotamidae	1	0.032
PRAIRIE	2	Simulidae	15	0.486
PRAIRIE	3	Tipulidae (Antocha)	1	0.032
PRAIRIE	3	Ceanidae	7	0.227
PRAIRIE	3	Chironomidae	83	2.689
PRAIRIE	3	Empididae	4	0.130
PRAIRIE	3	Heptageniidae	32	1.037
PRAIRIE	3	Hydropsychidae	7	0.227
PRAIRIE	3	Limnephilidae	2	0.065
PRAIRIE	3	Tricorythidae	1	0.032
PRAIRIE	4	Tipulidae (Antocha)	3	0.097
PRAIRIE	4	Athericidae	1	0.032
PRAIRIE	4	Baetidae	12	0.389
PRAIRIE	4	Chironomidae	82	2.657
PRAIRIE	4	Elmidae	36	1.166
PRAIRIE	4	Empididae	1	0.032
PRAIRIE	4	Ephemeridae	24	0.778
PRAIRIE	4	Heptageniidae	14	0.454
PRAIRIE	4	Hydropsychidae	416	13.478
PRAIRIE	4	Isonychidae	1	0.032
PRAIRIE	4	Perlidae	2	0.065
PRAIRIE	4	Simulidae	26	0.842
PRAIRIE	4	Siphlonouridae	1	0.032
PRAIRIE	4	Tipulidae	1	0.032
PRAIRIE	4	Tricorythidae	1	0.032
PRAIRIE	5	Tipulidae (Antocha)	12	0.389
PRAIRIE	5	Baetidae	8	0.259
PRAIRIE	5	Ceanidae	2	0.065
PRAIRIE	5	Chironomidae	50	1.620
PRAIRIE	5	Corydalidae	1	0.032
PRAIRIE	5	Elmidae	8	0.259
PRAIRIE	5	Empididae	1	0.032
PRAIRIE	5	Ephemeridae	7	0.227
PRAIRIE	5	Gomphidae	1	0.032
PRAIRIE	5	Helicopsychidae	2	0.065
PRAIRIE	5	Heptageniidae	8	0.259
PRAIRIE	5	Hydropsychidae	94	3.046
PRAIRIE	5	Limnephilidae	3	0.097
PRAIRIE	5	Perlidae	2	0.065
PRAIRIE	5	Polycentropodidae	1	0.032
PRAIRIE	5	Siphlonouridae	2	0.065
PRAIRIE	6	Tipulidae (Antocha)	3	0.097
PRAIRIE	6	Baetidae	8	0.259
PRAIRIE	6	Chironomidae	24	0.778
PRAIRIE	6	Elmidae	1	0.032
PRAIRIE	6	Helicopsychidae	2	0.065
PRAIRIE	6	Heptageniidae	8	0.259

APPENDIX C cont'd

Stream	Site	Taxa	Total	No./Sq.M
PRAIRIE	6	Hydropsychidae	7	0.227
PRAIRIE	6	Perlidae	1	0.032
SUGAR	1	Aeshnidae	1	0.032
SUGAR	1	Amphipoda	15	0.486
SUGAR	1	Baetidae	24	0.778
SUGAR	1	Chironomidae	43	1.393
SUGAR	1	Elmidae	14	0.454
SUGAR	1	Empididae	1	0.032
SUGAR	1	Hydropsychidae	4	0.130
SUGAR	1	Hydroptilidae	1	0.032
SUGAR	1	Leptophlebiidae	3	0.097
SUGAR	1	Simulidae	10	0.324
SUGAR	2	Athericidae	1	0.032
SUGAR	2	Brachycentridae	1	0.032
SUGAR	2	Chironomidae	116	3.758
SUGAR	2	Corydalidae	2	0.065
SUGAR	2	Elmidae	27	0.875
SUGAR	2	Glossosomatidae	14	0.454
SUGAR	2	Heptageniidae	2	0.065
SUGAR	2	Hydropsychidae	5	0.162
SUGAR	2	Isopoda	1	0.032
SUGAR	2	Leptophlebiidae	1	0.032
SUGAR	2	Leuctridae	2	0.065
SUGAR	2	Odontoceridae	1	0.032
SUGAR	2	Oligocheates	1	0.032
SUGAR	2	Perlidae	1	0.032
SUGAR	2	Psychomyiidae	4	0.130
SUGAR	2	Simulidae	42	1.361
SUGAR	3	Athericidae	1	0.032
SUGAR	3	Baetidae	26	0.842
SUGAR	3	Brachycentridae	3	0.097
SUGAR	3	Chironomidae	107	3.467
SUGAR	3	Elmidae	39	1.264
SUGAR	3	Ephemeridae	9	0.292
SUGAR	3	Glossosomatidae	1	0.032
SUGAR	3	Heptageniidae	6	0.194
SUGAR	3	Hydropsychidae	2	0.065
SUGAR	3	Hydroptilidae	19	0.616
SUGAR	3	Odontoceridae	1	0.032
SUGAR	3	Polycentropodidae	2	0.065
SUGAR	3	Simulidae	8	0.259
SUGAR	3	Tricorythidae	19	0.616
SUGAR	4	Baetidae	4	0.130
SUGAR	4	Chironomidae	254	8.230
SUGAR	4	Elmidae	12	0.389
SUGAR	4	Empididae	1	0.032
SUGAR	4	Heptageniidae	2	0.065
SUGAR	4	Hydropsychidae	55	1.782
SUGAR	4	Hirudinea	4	0.130

APPENDIX C cont'd

Stream	Site	Taxa	Total	No./Sq.M
SUGAR	4	Perlidae	1	0.032
SUGAR	4	Philipotamidae	5	0.162
SUGAR	4	Pyralidae	1	0.032
SUGAR	4	Simulidae	61	1.976
SUGAR	5	Tipulidae (Antocha)	1	0.032
SUGAR	5	Baetidae	12	0.389
SUGAR	5	Chironomidae	70	2.268
SUGAR	5	Elmidae	15	0.486
SUGAR	5	Empididae	1	0.032
SUGAR	5	Ephemeridae	20	0.648
SUGAR	5	Heptageniidae	29	0.940
SUGAR	5	Hydropsychidae	103	3.337
SUGAR	5	Oligocheates	9	0.292
SUGAR	5	Philipotamidae	1	0.032
SUGAR	5	Psphenidae	1	0.032
SUGAR	5	Simulidae	3	0.097
SUGAR	5	Gastropoda	2	0.065
SUGAR	6	Baetidae	14	0.454
SUGAR	6	Chironomidae	27	0.875
SUGAR	6	Elmidae	1	0.032
SUGAR	6	Ephemeridae	3	0.097
SUGAR	6	Glossosomatidae	1	0.032
SUGAR	6	Helicopsychidae	5	0.162
SUGAR	6	Heptageniidae	2	0.065
SUGAR	6	Hydropsychidae	69	2.236
SUGAR	6	Limnephilidae	13	0.421
SUGAR	6	Perlidae	6	0.194
SUGAR	6	Psphenidae	1	0.032
SUGAR	6	Psychomyiidae	1	0.032
SUGAR	6	Simulidae	4	0.130
WHITE	1	Amphipoda	1	0.032
WHITE	1	Baetidae	32	1.037
WHITE	1	Chironomidae	90	2.916
WHITE	1	Simulidae	23	0.745
WHITE	1	Tricorythidae	1	0.032
WHITE	2	Amphipoda	19	0.616
WHITE	2	Tipulidae (Antocha)	17	0.551
WHITE	2	Baetidae	4	0.130
WHITE	2	Brachycentridae	7	0.227
WHITE	2	Chironomidae	177	5.735
WHITE	2	Elmidae	31	1.004
WHITE	2	Empididae	4	0.130
WHITE	2	Ephemeridae	1	0.032
WHITE	2	Glossosomatidae	4	0.130
WHITE	2	Heptageniidae	1	0.032
WHITE	2	Tipulidae (Hexatoma)	1	0.032
WHITE	2	Hydropsychidae	19	0.616
WHITE	2	Hydroptilidae	2	0.065
WHITE	2	Simulidae	95	3.078

APPENDIX C cont'd

Stream	Site	Taxa	Total	No./Sq.M
WHITE	2	Tricorythidae	1	0.032
WHITE	3	Amphipoda	8	0.259
WHITE	3	Tipulidae (Antocha)	7	0.227
WHITE	3	Baetidae	4	0.130
WHITE	3	Brachycentridae	7	0.227
WHITE	3	Chironomidae	355	11.502
WHITE	3	Elmidae	8	0.259
WHITE	3	Ephemeridae	5	0.162
WHITE	3	Heptageniidae	2	0.065
WHITE	3	Hydropsychidae	24	0.778
WHITE	3	Hydroptilidae	6	0.194
WHITE	3	Simulidae	1	0.032
WHITE	3	Gastropoda	3	0.097
WHITE	4	Amphipoda	1	0.032
WHITE	4	Chironomidae	179	5.800
WHITE	4	Elmidae	11	0.356
WHITE	4	Empididae	2	0.065
WHITE	4	Ephemeridae	13	0.421
WHITE	4	Heptageniidae	7	0.227
WHITE	4	Hydropsychidae	135	4.374
WHITE	4	Hirudinea	6	0.194
WHITE	4	Leptoceridae	4	0.130
WHITE	4	Odontoceridae	4	0.130
WHITE	4	Perlidae	1	0.032
WHITE	4	Philipotamidae	6	0.194
WHITE	4	Psychomyiidae	1	0.032
WHITE	4	Gastropoda	1	0.032
WHITE	5	Ameletidae	5	0.162
WHITE	5	Amphipoda	2	0.065
WHITE	5	Chironomidae	90	2.916
WHITE	5	Ephemeridae	1	0.032
WHITE	5	Heptageniidae	3	0.097
WHITE	5	Hydropsychidae	1	0.032
WHITE	5	Isopoda	1	0.032
WHITE	5	Perlidae	1	0.032
WHITE	5	Tricorythidae	2	0.065

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