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EFFECTS OF DAM REMOVAL ON FLUVIAL
GEOMORPHOLOGY AND FISH

presented by

BRYAN ALAN BURROUGHS

has been accepted towards fulfillment
of the requirements for the

Doctoral degree in Fisheries & Wildlife Science

David Hyndman

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EFFECTS OF DAM REMOVAL ON FLUVIAL GEOMORPHOLOGY AND FISH

By

Bryan Alan Burroughs

A DISSERTATION

Submitted to
Michigan State University
in partial fulfillment of the requirements
for the degree of

DOCTOR OF PHILOSOPHY

Department of Fisheries and Wildlife

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ABSTRACT

EFFECTS OF DAM REMOVAL ON FLUVIAL GEOMORPHOLOGY AND FISH

By

Bryan Alan Burroughs

In this study, two approaches were used to gather information on the effects of dam removals on fluvial geomorphology of rivers and their fish communities. First, a case study of the Stronach Dam removal was performed starting in 1995, encompassing two years prior to the dam removal, six years during the gradual removal, and three years after the removal was complete. Permanent sampling sites were established and monitored annually throughout the period. Secondly, surveys of current conditions were conducted at past dam removal sites of 2, 7, 14, 32 and 39 years since removal occurred. Fluvial geomorphology and fish community characteristics were documented in formerly impounded stream reaches as well as reaches downstream from past dam removals, and were compared to conditions in upstream reference reaches that had not been impacted by the presence of the dams or the dam removals. Immediately following a dam removal, slope of rivers are increased, leading to erosion of the reservoir sediment fill, and its transport and deposition through the stream reach downstream of the dam removal. Immediately this leads a narrowing of the river channel in the reservoir and a widening of the river channel downstream of the dam removal. Water velocity is increased, and any water temperature effects from the impoundment are reversed. Changes in channel morphology (such as width, depth, and sinuosity) continue for approximately the next decade as the

river adjusts to its new slope. Coarsening of substrate, and the development of bedform diversity begins immediately following dam removal, but requires more infrequent high flows to be restored to levels similar to the reference reaches, possibly requiring 20 - 30 years in Michigan. Because dams serve as barriers to fish movements, species compositions upstream and downstream of dams can be quite different prior to dam removal. Following removal, fish quickly distribute throughout the river and species compositions in stream reaches become similar, with higher species richness and diversity. Lotic species, such as trout, increased in abundance as previously impounded river was converted to lotic habitat. Total abundance of fish of all species, did not appear to increase to reference levels until substrate coarseness and bedform diversity returned, decades after dam removal.

ACKNOWLEDGEMENTS

Funding for this research was provided by the Michigan Department of Natural Resources. Recognition is deserved for the individuals from that agency and others that were involved in the decision to initiate and fund a long-term study of the effects of dam removal, long before dam removal research was widely identified as urgently needed.

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EFFECTS OF THE STRONACH DAM REMOVAL ON FLUVIAL GEOMORPHOLOGY IN THE PINE RIVER, MANISTEE COUNTY, MICHIGAN

INTRODUCTION

The intensive construction of dams in North America began in the 1800's (Petts 1980), reaching its peak from 1950 to 1970 (Heinz Center 2002). This resulted in approximately 2.5 million dams being built in the United States (National Resource Council 1992), on nearly every major and minor river system in the lower 48 states (Heinz Center 2002). These dams were built for a variety of purposes including (in order of prevalence); recreation, fire and farm ponds, flood control, municipal water supply, irrigation, tailings and waste containment, mechanical and hydroelectric energy generation, navigation, and wildlife management.

Dams alter the flow of water and fundamentally change the functioning of river ecosystems. From the earliest times of dam construction, people have recognized some of the inherent consequences of building dams. The first laws restricting dams on rivers, with the acknowledgement that they restrict crucial migrations of fish, were in the Magna Carta (Neilsen 1999). However it wasn't until after the peak period of dam construction ended in the 1970's that strong scientific proof for the multitude of ways dams impact river systems began to emerge. The effects that dams have on river ecosystems are now more fully understood (e.g., Hammad 1972, Petts 1980, Williams and Wolman 1984, Cushman 1985, Bain et al. 1988, Ward and Stanford 1989, Benke 1990, Ligon et al. 1995, Collier et al. 1996, Lessard and Hayes 2003, Shields et al. 2000), including interruption to the flow of water, sediment, energy, nutrients, biota, and associated indirect changes.

Many dams are still functioning and fulfilling their intended purpose, providing social and economic benefits. However, as dams age they require maintenance to prolong their function and safety. There are now, a large and growing number of dams that no longer fulfill their intended purpose and may not sustain sufficient benefits as to outweigh the negative ecological impacts they cause.

Of the estimated 2.5 million dams in the U.S., 76,000 are six feet or greater in height; a minimum size for dam safety regulatory concerns (Federal Emergency Management Agency (FEMA) and U.S. Army Corps of Engineers (USACE) 1996). Of these 76,000 dams, 80% or 60,000 are expected to be 50 years of age or older by the year 2020 (FEMA and USACE 1996). Given that the average design life expectancy of dams is approximately 50 years, this implies that a large number of dams in the U.S. will be in need of maintenance or considered for removal (River Alliance of Wisconsin and Trout Unlimited 2000). Over the last several decades, the rate at which dams have been removed in the U.S. has risen from approximately one per year during the 1960's to approximately 20 per year during the 1990's (Pohl 2003). Given all of these estimates, the practice of removing dams is likely to become increasingly common in the future.

At least 400 dams have been removed to date in the U.S. (Pohl 2003), but the scientific literature on the effects of dam removal is still sparse. Much of the existing dam removal literature focuses on the administrative, legal, and socioeconomic aspects of executing dam removals (Born et al. 1998, River Alliance of Wisconsin and Trout Unlimited 2000, Smith et al. 2000, Graber et al. 2001, Trout Unlimited 2001, Bowman 2002, Johnson and Graber 2002). Using analogies from various disciplines, other researchers have developed several hypotheses of river ecosystem responses to dam removals (Pizzuto 2002, Stanley and Doyle 2002, Shafroth et al. 2002, Gregory et al. 2002, Whitelaw and MacMullan 2002, Doyle et al. 2002), while others have hypothesized the outcomes of specific proposed dam removals (Shuman 1995, Freeman et al. 2002, Heinz Center 2002). Despite this emerging conceptual basis for the effects of dam removals, the field continues to lack the empirical information that is needed to verify these hypotheses, calibrate pre-existing models for use with dam removal, and generate novel insights into the effects of dam removal (Doyle et al. 2002, Bushaw-Newton et al. 2002, Graf 2003, Hart et al. 2003, Doyle et al. 2005). Qualitative observations on the effects of dam removal exist for several dam removal case studies (American Rivers et al. 1999, Smith et al. 2000), and several quantitative studies exist on the effects of dam removal on fluvial geomorphology (Evans et al. 2000, Wohl and Cenderelli 2000, Bushaw-Newton et al. 2002, Stanley et al. 2002, Chaplin 2003), aquatic insects (Bushaw-Newton et al. 2002, Stanley et al. 2002) and fish (Kanehl et al. 1997, Hill et al. 1994, Bushaw-Newton et al. 2002). While these studies provide unique insights into the outcomes of dam removals, many were relatively short in time duration (i.e., 1-2 years post dam removal), and the empirical information on the effects of dam removal is still considered very limited (e.g., Graf 2003, Doyle et al. 2005).

The goal of this study was to document the effects of dam removal on fluvial geomorphology. In particular, this study was designed to address questions such as; what types of changes occur in rivers following dam removal, what are the magnitudes of these changes, what are the spatial extents of the changes, and how long do these changes take to occur? Answers to questions such as these should lead to more informed decision making processes regarding dam removals, improved predictions on the outcomes of dam removal, and could lead to improvements in how dams are removed in the future.

To answer these questions, we monitored several aspects of fluvial geomorphology from 1995 through 2006, before, during and after the gradual removal of Stronach Dam, on the Pine River, Manistee County, Michigan. The specific objectives of this study were to: (1) document the spatial and temporal extent of sediment erosion, transport and deposition that occurred due to the dam removal; and (2) document the changes in river morphology attributes (i.e., slope, width, depth, water velocity, substrate composition, and bedform (riffle-pool) diversity) that occurred due to the sediment erosion, transport and deposition. The 12 years of detailed quantitative monitoring of the outcomes of this dam removal provide a unique dataset useful for validating and refining existing hypotheses about the effects of dam removals, generating novel hypotheses, leading to improvements in the ongoing practice of removing dams.

Site Description

Stronach Dam was located on the Pine River, a tributary to the Manistee River, in the northwestern Lower Peninsula of Michigan (Figure 1). The Pine River is 77 km long, a fourth order stream, and drains a 68,635 ha watershed dominated by sandy glacial outwash plains, recessional moraines, and areas of consolidated clay (Hansen 1971, Rozich 1998). Mean daily discharge recorded at two U.S. Geological Survey gaging stations on the Pine River (8 and 13.7 km upstream from Stronach Dam) averaged 8.10 m^3/s during 34 years of record, with a minimum discharge of 4.56 m^3/s, a maximum of 69.09 m^3/s, and an average annual ratio of low to high mean monthly flows of 2.02, indicating "stable to very stable" flows (Rozich 1998). It carries a large bedload of sand due to the local geology and extensive logging operations of the late 1800's, which created unstable banks along the river. Hansen (1971) estimated a mean annual sediment discharge of 50,000 tons at Stronach Dam from 1967 to 1970, 70 - 75% of which was sand.

Stronach Dam was constructed from 1911 to 1912, 5.6 km upstream from the confluence of the Pine River and the Manistee River (Figure 1). Stronach Dam was an earth embankment dam with a concrete corewall; a 4.57 m fixed-concrete spillway section with 0.91 m of flashboards on top of the spillway; a concrete and brick powerhouse with two turbine bays; and an upstream fish ladder (Consumers Power Company 1994). Stronach Dam, with 5.49 m of head height possible, was operated mostly around 5.18 m of head. This created a

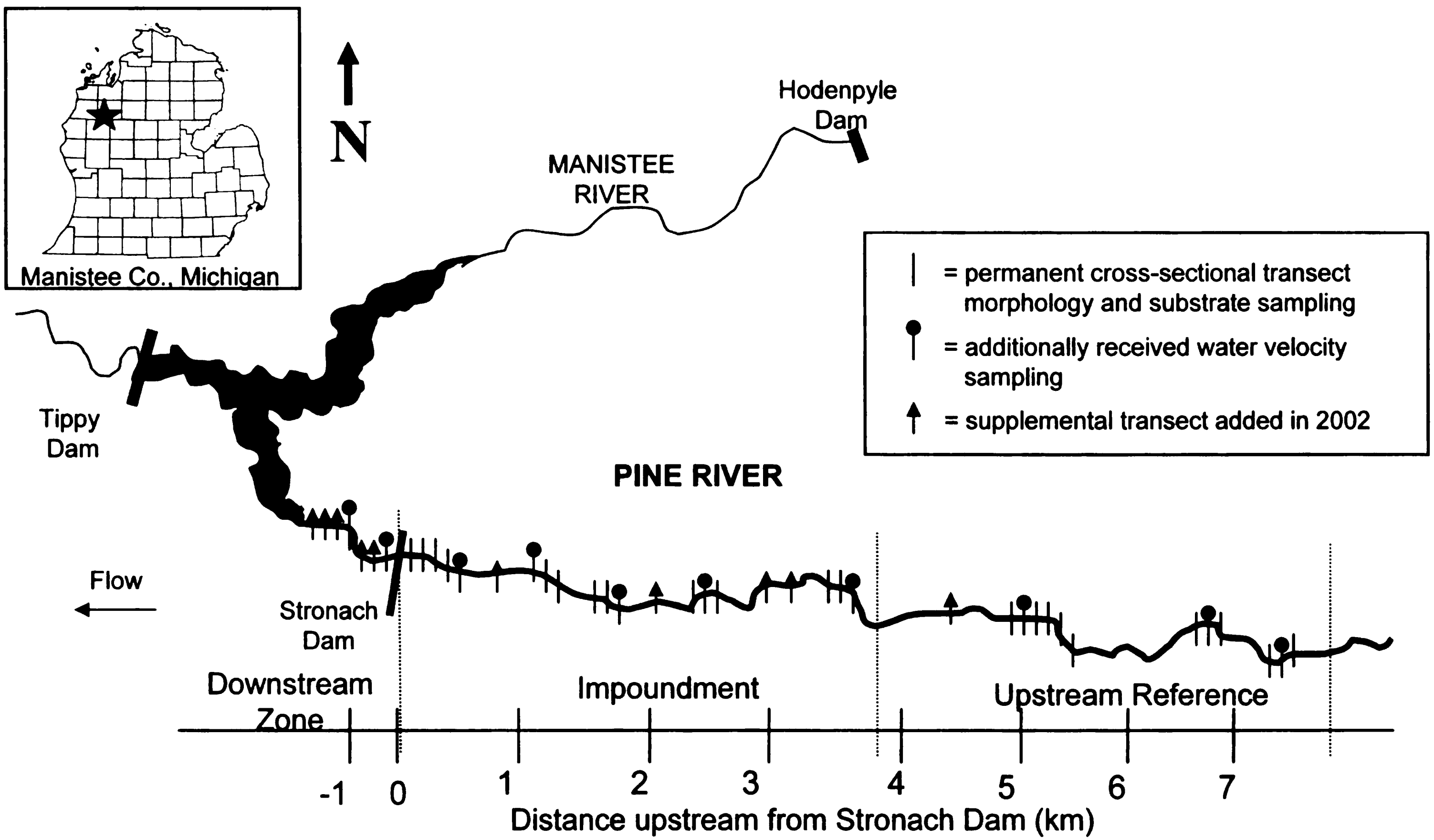

Figure 1. Locations of Stronach Dam and the Pine River in relation to the State of Michigan; and the location of permanent cross-sectional surveying transects within the study area of the Pine River.

26.7 ha reservoir with a 789,428 m^3 volume capacity (Consumers Power Company 1994, Hansen 1971). Tippy Dam (17.07 m head height) was constructed in 1918 immediately downstream of the confluence of the Pine and Manistee Rivers (Rozich 1998) (Figure 1). This created a 428 ha, 48,722,530 m^3, impoundment over the high gradient confluence area of the two rivers, blocked all upstream fish migration from Lake Michigan, and impounded water upstream to Stronach Dam.

Due to the Pine River's large sediment load, the Stronach reservoir quickly filled with sediment and problems arose with the operation of the dam's turbines. Attempts were made in the 1930's to remove the accumulation of sediment behind the dam. These efforts were only marginally successful and dredging eventually became uneconomical (Consumers Power Company 1994). In 1953, 41 years after the dam's construction, Stronach Dam was decommissioned by the owner, Consumers Power Company. The generator rooms were demolished, the fish ladder was removed, and the river flow was directed over the spillway. The spillway flashboards were removed gradually over the following years; the last was removed in 1983 (Consumers Power Company 1994).

In the early 1990's, removal of Stronach Dam was negotiated as part of a Federal Energy Regulatory Commission (FERC) agreement in the relicensing of Tippy Dam. Removal of Stronach Dam began in the spring of 1997 and was completed in December of 2003. A "staged" or gradual removal was decided upon in order to allow gradual river channel adjustments with the least amount of environmental impact, at the lowest cost, and without impacting the operation of

Tippy Dam (Battige et al. 1997). In 1996, a 3.6 m high "stop-log" structure was installed in the old powerhouse to allow a gradual drawdown of the river. The stop-log structure consisted of hollow metal pipes (15 cm diameter) stacked one on top of another, with a metal grate called a "trash-rack" immediately upstream to protect the stop-logs from debris impingement. The original removal schedule called for one stop-log to be removed every three months, for a total of 0.60 m per year, over the course of six years; with corresponding trash-rack removal. This plan was altered due to recreational safety concerns, feasibility issues, and technical difficulties with removal (Table 1) (Battige personal communication 2002). The actual sequence of the staged dam removal is shown in Table 1.

METHODS

In 1995, two years prior to the commencement of dam removal activities, the Pine River was assessed to document the spatial extent of impoundment effects due to Stronach Dam. This assessment involved the surveying and description of physical characteristics, including categorization of the stream into bedform units of runs, riffles, pools, rapids, or complexes (a designation where more than one category applied), following the criteria developed by Hicks and Watson (1985). This survey method allowed detection of impoundment effects well upstream of the readily noticeable reservoir area. This "impoundment" area of the river extended for 3.89 km upstream of Stronach Dam and was typified as being relatively wide, slow-flowing, sand-bottomed, and generally consisted of only run bedform units. An upstream reference reach was chosen, extending for

Table 1. Schedule of removal events during the staged removal of Stronach Dam on the Pine River, Manistee County, Michigan. Stop-logs are 15.24 cm diameter hollolw metal pipes stacked on top of one another. Trash-rack removal estimates are approximate. Cumulative meters removed are in parentheses. (Dave Battige, Consumers Energy, personal communication 2003).

Date	Number of Stop-logs removed	Meters of Trash-rack removed
March 17, 1997	1 (0.15)	0 (0)
June 5, 1997	1 (0.30)	0 (0)
June 16, 1997	2 (0.61)	0 (0)
June 24, 1997	2 (0.91)	0 (0)
September 15, 1997	1 (1.07)	0 (0)
December 15, 1997	1 (1.22)	0 (0)
March 16, 1998	1 (1.37)	0 (0)
May 7, 1998	0 (1.37)	1.83 (1.83)
May 29, 1998	0 (1.37)	0.30 (2.13)
June 15, 1998	1 (1.52)	0 (2.13)
September 8, 1998	1 (1.68)	0.30 (2.44)
December 14, 1998	1 (1.83)	0.30 (2.74)
March 15, 1999	1 (1.98)	0 (2.74)
May 11, 1999	1 (2.13)	0 (2.74)
September 13, 1999	2 (2.44)	0 (2.74)
September 16, 1999	0 (2.44)	0.61 (3.35)
April 17, 2000	2 (2.74)	0 (3.35)
October 2, 2000	2 (3.05)	0 (3.35)
October 5, 2000	0 (3.05)	0.61 (3.96)
May 8, 2001	2 (3.35)	0 (3.96)
September 8, 2001	2 (3.66)	0 (3.96)
November 11, 2002	0 (3.66)	1.52 (5.49)
December 2003	Remaining spillway and dam superstructure removed	

3.70 km upstream from the upstream boundary of the impoundment. This study zone was chosen as a reference reach because no impoundment effects from the dam were evident. The river was narrower, faster-flowing, had coarser substrates, and showed high bedform heterogeneity. A third study zone was chosen downstream of Stronach Dam, where the river was wide, very slow-flowing, sand-bottomed, and consisted entirely of run bedforms. Prior to the removal of Stronach Dam, water was impounded in this study zone from Tippy Dam Reservoir downstream, and the zone extended for only 0.63km downstream of Stronach Dam. During the dam removal process, slope increased in this zone and the impoundment of water by Tippy Dam was not observed for 2.55 km downstream of Stronach Dam. In 2002, this downstream study zone was lengthened from 0.63 km to 2.55 km downstream of Stronach Dam.

Thirty-one permanent cross-sectional transects were established in 1996 to allow for measurement of changes in river channel morphology over the course of dam removal. Ten additional transects were added in 2002 to fill in longitudinal gaps in areas of geomorphologic interest, and to extend the downstream zone. Thirteen transects were located in the upstream reference reach, 21 transects were located in the impoundment, and seven transects were placed in the downstream zone (Figure 1). Photographs, site descriptions, and latitude-longitude coordinates for all transects are archived at Michigan State University, Department of Fisheries and Wildlife. All transects were linked to real elevations above sea level based on a USGS monument. The distance of each transect from Stronach Dam was determined by floating the river in a canoe using the trip-log feature of several handheld Garmin® GPS units, on several occasions and averaging the results. All transects were surveyed annually from 1996 through 2006, during June to early-July of each year. At each transect, streambank elevations were recorded at points of inflection (appropriate for accurate documentation of streambank morphology). Streambed elevations were recorded every 0.61 m across the wetted width of the stream, and water surface elevation was measured once per transect. All measurements were taken at the same locations each subsequent year of the study. Elevations were recorded to the nearest 3 mm.

Top bank was delineated for each transect as the highest elevation during the entire time series in which water would just begin to overflow onto a floodplain. Cross-sectional areas, below top bank, were calculated for each transect in each year. The trapezoidal rule for numerical integration (Press et al. 1992) was used to estimate areas between each pair of cross-sectional points surveyed, and the sum of these was the transect cross-sectional area estimate. Change in transect cross-sectional area reflects the net amount of erosion or deposition that occurred between surveying events. Estimates of sediment volume transported during the dam removal were calculated using the trapezoidal rule for integration, given the amounts of cross-sectional area change at transects, and the distances between the transects. Gradient was calculated as the change in water surface elevation between two transects, over the river distance between those two transects. Width (w) and mean depth (d) were calculated for each transect, from the wetted channel dimensions. Bank slopes for each side of each transect were calculated as the difference in elevations from the top bank to the beginning of the streambed, divided by the lateral distance between those two points. The beginning of the streambed was determined to be the point, lower in elevation than the water surface, where bank slope was greatly diminished and the channel appeared to become mostly horizontal. These points, including the top bank elevations were allowed to change each year. The change in each bank slope at a transect, compared to the initial pre-dam removal slopes, were calculated and averaged for each transect.

Water velocity was measured at 10 of the permanent transects (Figure 1) annually from 1996 to 2006. From 1996 to 2000, a Marsh-McBirney Model 201 portable current meter was used. From 2001 to 2006, water velocity was measured using a Global Flow Probe Model FP101, impellor-style flow meter with a 4 cm diameter impellor. Water velocity was measured at 0.61 m intervals, starting at the water's edge on one stream bank and ending at the water's edge on the opposite stream bank. If water depth was less than 0.75 m, water velocity was measured at 60% of the water depth from the water surface. If water depth was greater than 0.75 m, water velocity was measured at 20% and 80% of the water depth from the water surface, and the two measurements were averaged (Gallagher and Stevenson 1999). Mean water velocities were calculated for each transect in each year. All point measurements of water velocity at each transect were aggregated for all transects in each study zone to examine changes in the distributions of all water velocities in a study zone. The

Kolmogorov - Smirnov two sample test (Steel and Torrie 1980) was used to test for differences between water velocity frequency distributions between years. The K-S test was used because it evaluates the observed difference between two distributions, not just differences in means.

Streambed substrate size composition was measured at 10 of the permanent transects in 1996, at each of the 31 original permanent transect sites annually from 1997 – 2006, and also at the ten newer additional sites from 2002 through 2006 (Figure 1). A modified pebble count method was used (Wolman 1954, Kondolf and Li 1992) to determine substrate size composition. We sampled 100 streambed particles systematically (instead of randomly) along each transect, measuring the intermediate axis, and assigning a size class code to each particle (from a modified Wentworth scale) (Wentworth 1922, Cummins 1962). Systematically sampling the substrate and recording the transect side started on, and the approximate distance between samples, allows linkage of the information gathered to other morphological data (e.g., size of substrate at places of erosion or deposition within a transect cross-section) in addition to providing a measurement of the size structure of the substrate. Median substrate size (D_{50}) for a transect was calculated after excluding "organic" or "trash" designations which did not have corresponding size classes. The Kolomogorov-Smirnov two sample test (Steel and Torrie 1980) was used to test for differences between substrate size frequency distributions between years.

In 1995, two years prior to the dam removal, bedforms (also commonly referred to as "meso-habitat" or "channel unit types") were mapped for all three study zones. This assessment involved the categorization of the stream into bedforms following the criteria developed by Hicks and Watson (1985). The length (along the thalweg), and widths of each bedform were measured, as well as periodically recording latitude and longitudes of selected bedforms for mapping purposes. In 2004, the year following complete removal of the dam, this assessment was repeated to document changes in the frequencies of bedforms in each study zone. This assessment was conducted at base flow levels.

RESULTS

Sediment Transport

During and following the staged removal of Stronach Dam, significant sediment transport occurred in both the impoundment and the downstream zones, while the upstream reference reach remained remarkably stable. In the upstream reference reach, net erosion occurred in some years while net deposition occurred in others (Figure 2). The average annual net sediment transport in this reach was 289 m^3 of erosion, with a standard deviation of 1,946 m^3. This translates to only ~4 mm average annual vertical change for each square meter of streambed in the upstream reference reach.

The total volume of sediment eroded from the impoundment during the staged removal through three years post removal was approximately 92,000 m^3. The annual volume of sediment eroded from the impoundment averaged 9,159 m^3 and was quite variable with a standard deviation of 6,947 m^3 and a range

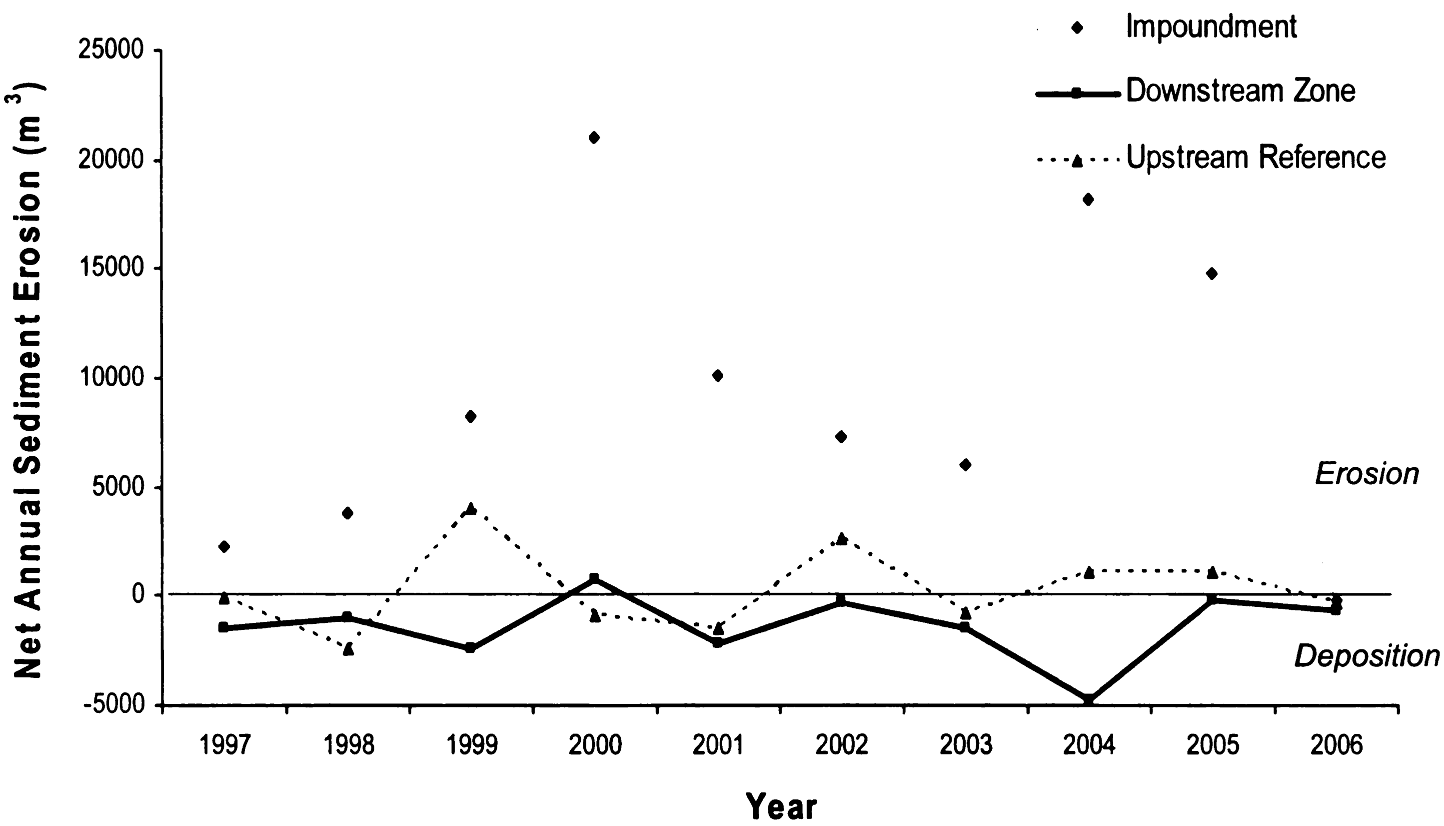

Figure 2. Net annual sediment erosion or deposition volumes during and after the staged removal of Stronach Dam (1997 – 2003). Positive values correspond to net erosion, negative values correspond to net deposition.

from approximately 0 – 21,000 m^3 (Figure 2). In 2006, no net erosion occurred in the impoundment. This volume of sediment, while large, is substantially less than the mean annual bedload estimated by Hansen in 1971 at the Stronach dam site. Hansen estimated that over three years, the mean annual bedload was 50,000 tons or approximately 28,000 m^3 (using a density of 1.8 g/cm^3 for sand (Morris and Fan 1998)). This makes the net total amount of sediment erosion from the former impoundment over ten years, roughly equivalent to about 3.5 years of annual sediment bedload during the time of Hansen's estimates.

The annual volume of sediment eroded from the impoundment was not correlated to the amount of dam removed between sampling events ($R^2 = 0.04$), annual mean flows ($R^2 = 0.07$), annual peak flows ($R^2 = 0.00$), days at or above bankfull discharge (1.5 year recurrence flood) ($R^2 = 0.01$), or approximate stream power in the impoundment reach (factoring in cumulative mean flows and changing stream slopes) ($R^2 = 0.12$).

The amount of erosion that occurred in the impoundment varied spatially with distance from the dam (Figure 3). In general, greater amounts of erosion occurred closer to the dam site, with the magnitude of erosion attenuating upstream. During the first several years of the removal, erosion progressed upstream only through the easily recognizable former reservoir (1.21 km), and it wasn't until approximately 2001 – 2002 when net erosion was documented at the furthest upstream extent of the impoundment, 3.89 km from the dam. A pair of transects, 0.39 and 0.50 km upstream of the dam, provide an exception to the longitudinal pattern of diminishing erosion with distance upstream of the dam.

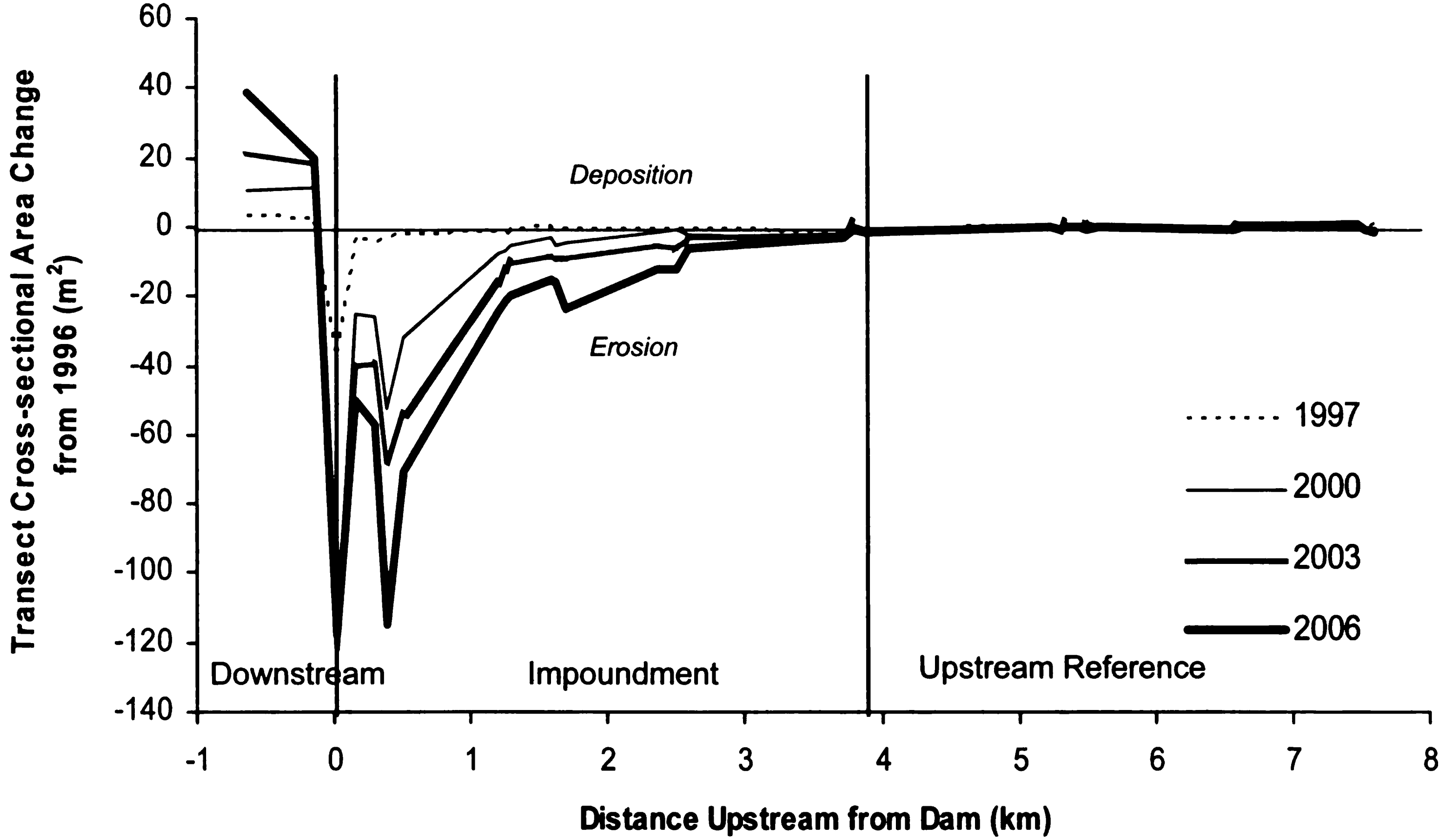

Figure 3. Longitudinal pattern of change in cross-sectional area, during the staged removal of Stronach Dam (1997 – 2003), compared to before dam removal (1996). Positive values correspond to deposition within a transect and negative values correspond to erosion at a transect.

These two sites experienced greater amounts of erosion than other sites closer to the dam due to large amounts of lateral erosion in addition to the typical vertical erosion.

Each year of the dam removal, with the exception of 2000, some amount of sediment moving downstream from the impoundment was deposited and retained in the first 0.63 km downstream of the dam (Figure 2). The volume of sediment that was retained and not transported further downstream varied considerably between years (average = 1,360 m^3, standard deviation = 1,518 m^3), totaling 13,599 m^3 by 2006. The remainder of the 92,000 m^3 of sediment eroded from the impoundment was either transported further downstream in the river, eventually being deposited out of the study area in Tippy Dam reservoir, or onto the floodplain downstream of the dam during high flow periods.

Channel Geometry

In the upstream reference reach, width and width to mean depth ratio (w/d) of the wetted stream channel remained stable (Figures 4 and 5). In the impoundment, the width of the wetted channel generally decreased (Figure 4), with the magnitude of this narrowing generally corresponding to the amount of erosion that occurred at a transect (greatest closest to the dam and diminishing upstream through the impoundment). Localized differences in geology and slope did alter this general pattern at some transects. Change in the w/d ratio of the impoundment did not show any easily discernable patterns (Figure 5). In the downstream reach, width and w/d ratio both increased (Figures 4 and 5).

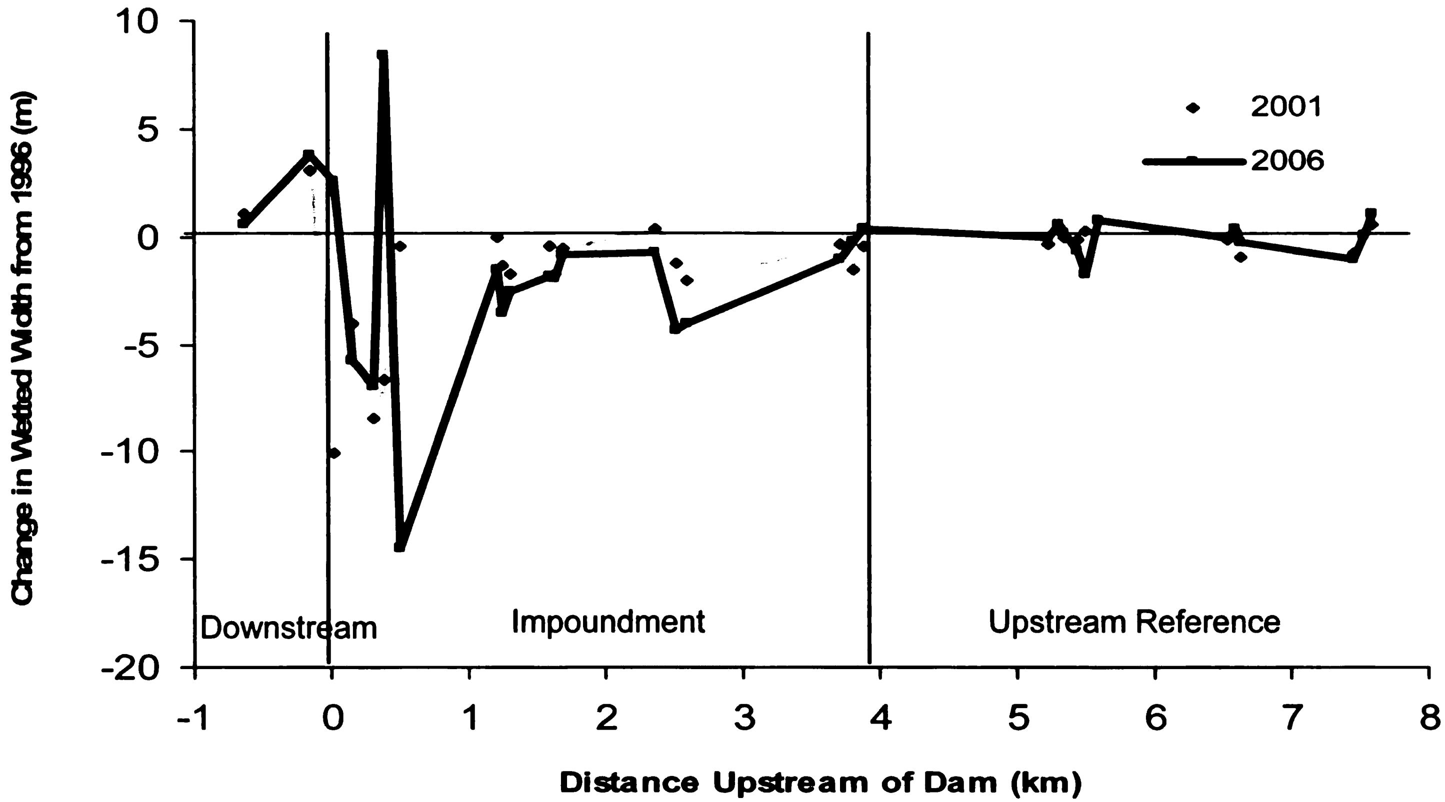

Figure 4. Longitudinal pattern of change in wetted stream width from pre-dam removal conditions in 1996. Only 2001 and 2006, five and 10 years after the beginning of the dam removal are shown for simplicity and clarity.

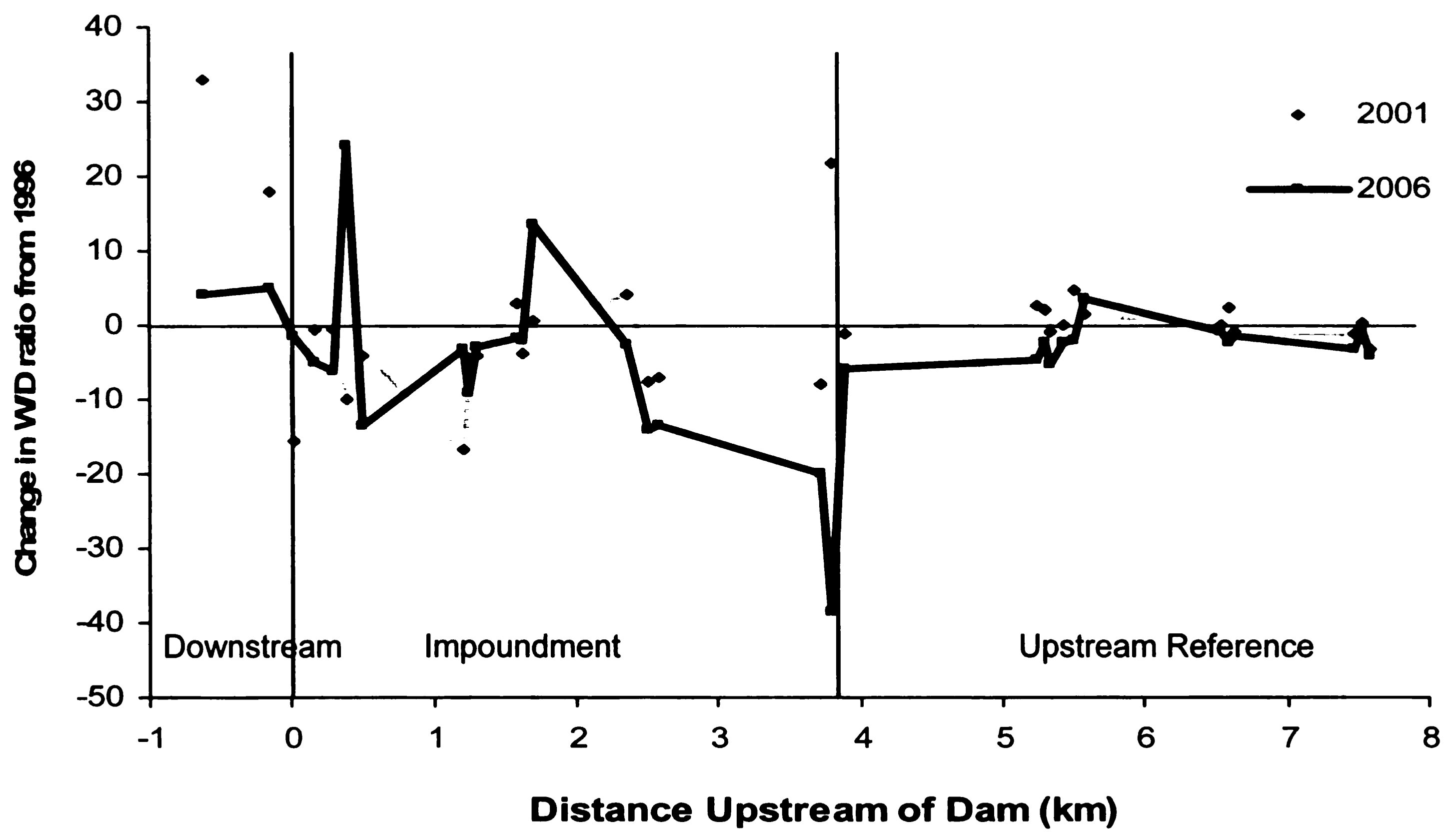

Figure 5. Longitudinal pattern of change in wetted stream width to mean depth ratio (w/d) from pre-dam removal conditions in 1996.

In the upstream reference reach, bank slopes at some transects remained stable, while several others were quite dynamic; showing both increases and decreases in slope (Figure 6). Since the streambed in this reach was remarkably stable during the study period, these changes are most likely due to lateral erosion processes, and are common to many streams in this area. Bank slopes in the impoundment gradually increased during the dam removal, with the greatest increases seen closest to the dam (Figure 6). In 2006, bank slopes further upstream in the impoundment reach began to decrease in slope, while those closer to the dam site remained steepened. In the downstream reach, bank slopes remained relatively stable (Figure 6).

Water Slope and Velocity

As a result of the sediment erosion and deposition processes that occurred, water slope in the impoundment and downstream reaches increased during the dam removal (Figure 7). Increases in slope were greatest in the first 1.59 km upstream of the dam removal, but were observed to a lesser degree in most of the former impoundment. Slope increased in the entire impoundment from 0.13% in 1996, to 0.21% in 2006 (with 0.26% slope in the first 1.59 km upstream of the dam site in 2006). The slope in the downstream zone also increased from 0.06% in 1996 to 0.10% in 2006. Slope in the upstream reference reach remained at 0.16% in both 1996 and 2006.

Water velocities were measured each year during transect surveying, during the month of June. In 1996, the average discharge during these

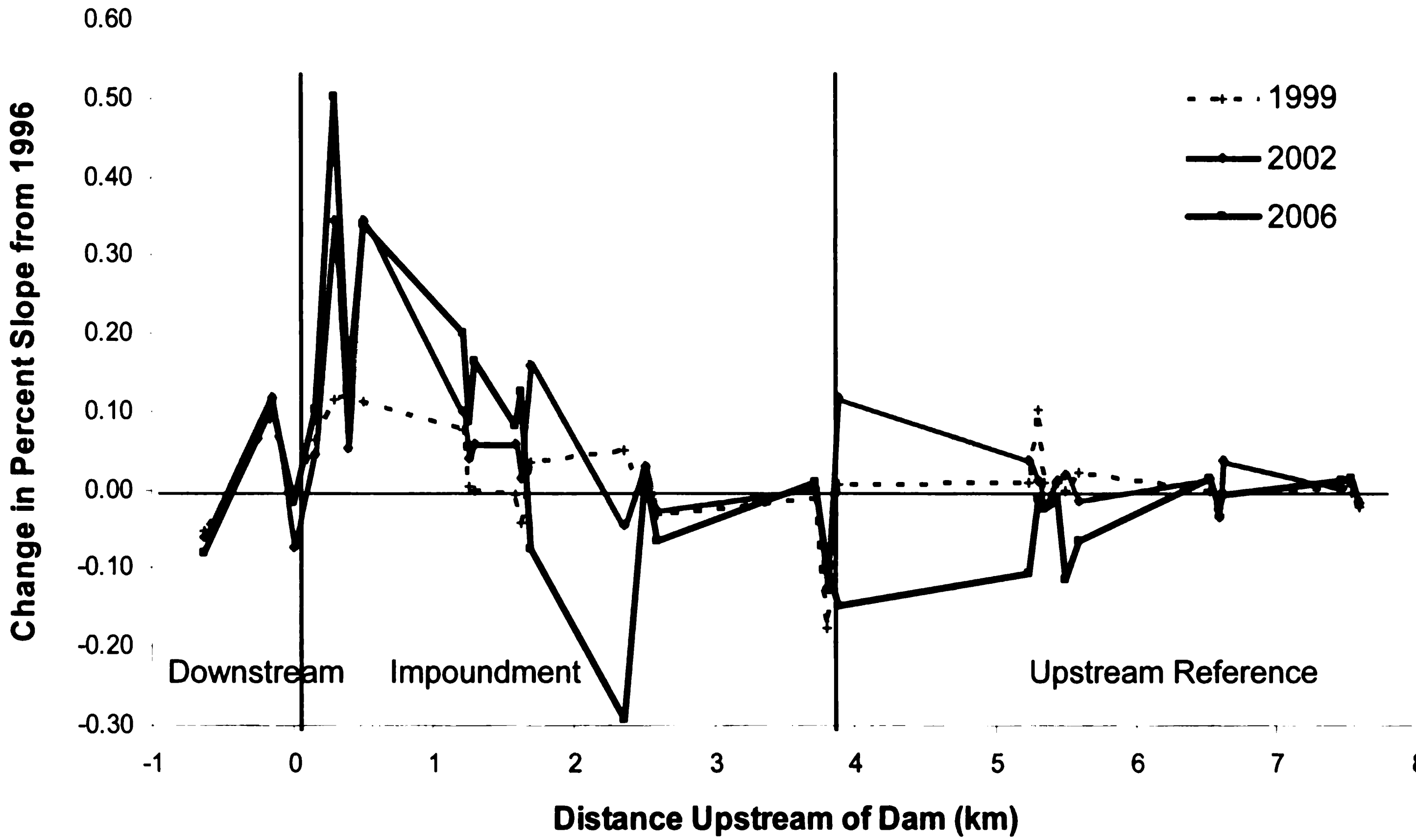

Figure 6. Longitudinal pattern of change in bank slopes from pre-dam removal conditions in 1996.

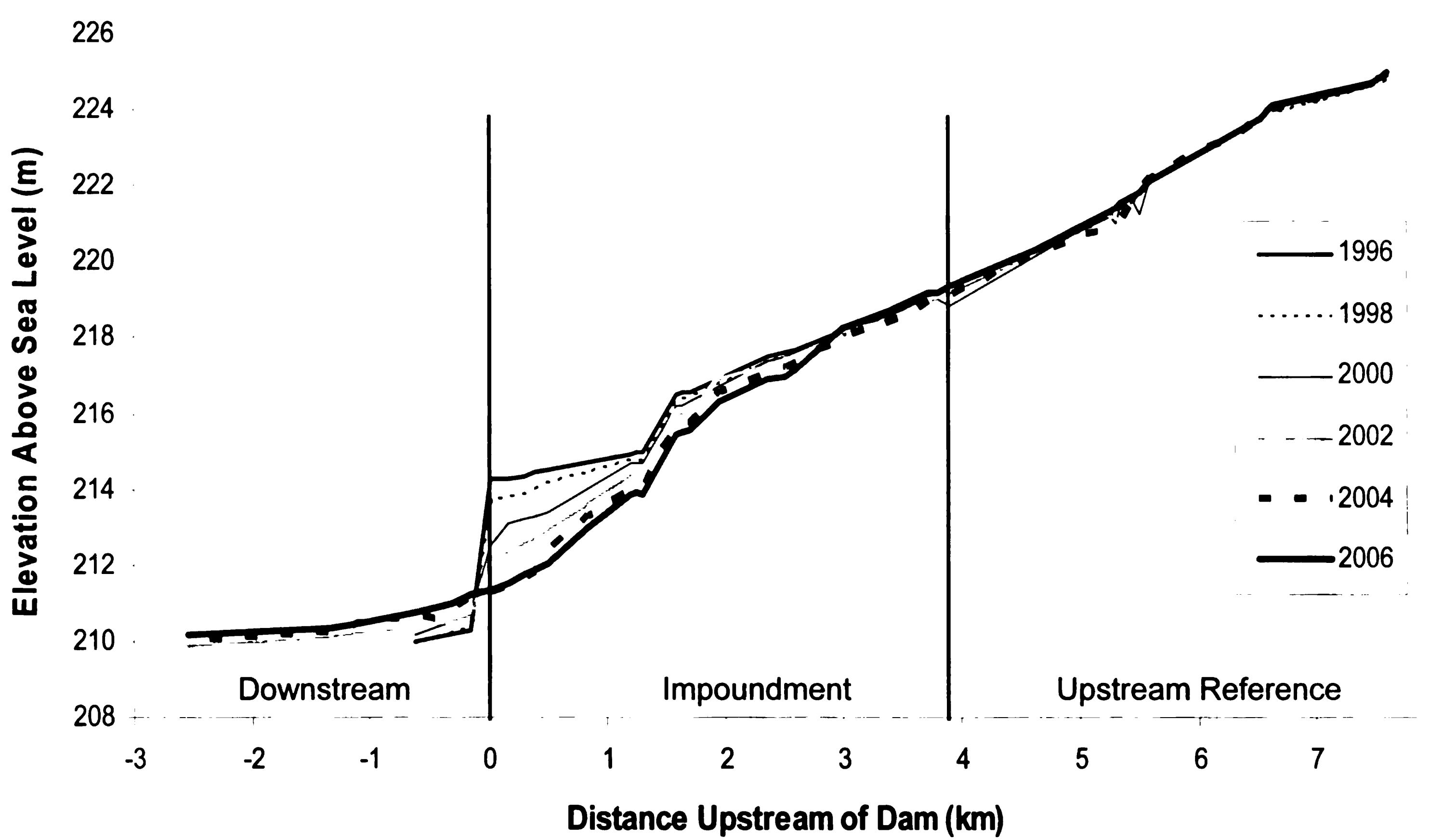

Figure 7. Longitudinal profile of water slopes or gradients. Vertical scale exaggerated relative to horizontal scale.

measurements was 7.70 m^3/s. In 2006, due to continuing precipitation events, the measurements were taken at an average discharge of 9.29 m^3/s, significantly higher than 1996. Due to the influence of discharge on water velocity, we chose to use measurements taken in 2005, when discharge was 8.07 m^3/s, for comparison with 1996 data. Water velocity was not surveyed in all downstream zone transects in 1996 or 1997, so data from 1998, when discharge was 6.94 m^3/s, where used for comparison purposes for the downstream zone.

Prior to dam removal, mean water velocities generally decreased in a downstream direction through the impoundment and downstream zone (Figure 8). With the dam removal and consequent increased slopes, mean water velocities generally increased in both the lower impoundment and the downstream zone, with some of the highest mean water velocities found in the impoundment after the dam removal. Due to localized differences in slope and channel morphologies, there is more variability in the mean water velocities in the impoundment than previously observed.

The frequency distributions of water velocities were compared for each zone, in both the first and last year of sampling. The water velocity frequency distribution for the upstream reference reach was not significantly different in 2005 as compared with 1996 (Reference: K-S test $D_{max} = 0.227$, $n = 77$, $p>0.01$). Water velocity frequencies in the impoundment were significantly faster in 2005 than in 1996 (Impoundment: K-S test $D_{max} = 0.247$, $n = 137$, $p<0.01$). The water velocity frequency distribution for the downstream zone became significantly faster in 2005 than in 1998 (Downstream: K-S test $D_{max} = 0.235$, $n = 107$,

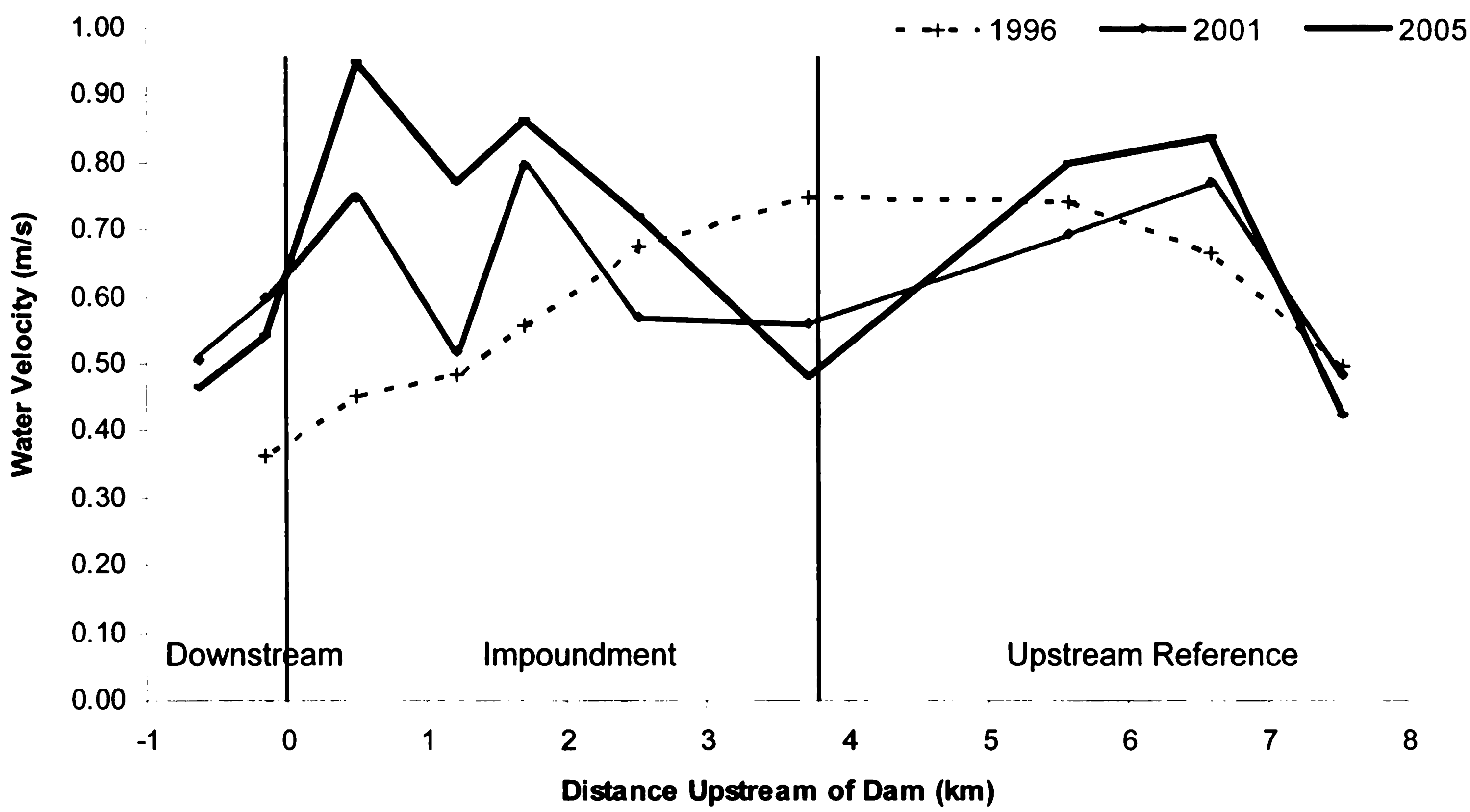

Figure 8. Longitudinal pattern of mean water velocity (m/s) in the Pine River.

p<0.01). In the impoundment and downstream zones, frequencies of water velocities less than 0.3 m/s were relatively unchanged (representing the slower water velocities found at the stream margins), while thalweg water velocities increased in magnitude, with the highest water velocities increasing (Figure 9). The former impoundment is now the only study reach to contain water velocities greater than 1.2 m/s.

Substrate

Median (D_{50}) substrate sizes and substrate size frequency distributions were compared for each zone, in both 1997 – 1998 and 2005 - 2006, the first two years and last two years of sampling. The first and last two years were averaged to reduce the influence of annual variability in substrate composition measurements in comparisons. Before, during and after the removal of Stronach Dam, there was considerable spatial variability or patchiness in substrate size compositions in the Pine River (Figure 10). Prior to removing the dam, substrates were generally coarser and more heterogeneous in the upstream reference reach than in the impoundment and downstream. The impoundment was mostly fine sediments such as sand, with a few patches of small gravel, while the downstream zone was almost totally dominated by sand. During the dam removal process, median substrate size in the reference decreased slightly (Reference: D_{50} 1997-98 = 34.8 mm = "very coarse gravel", 2005-06 = 29.5 mm = "coarse gravel") but the overall substrate size frequency distribution did not change significantly (Reference: K-S test $D_{max} = 0.063$, $n_1 = 989$ $n_2 = 1122$, p >

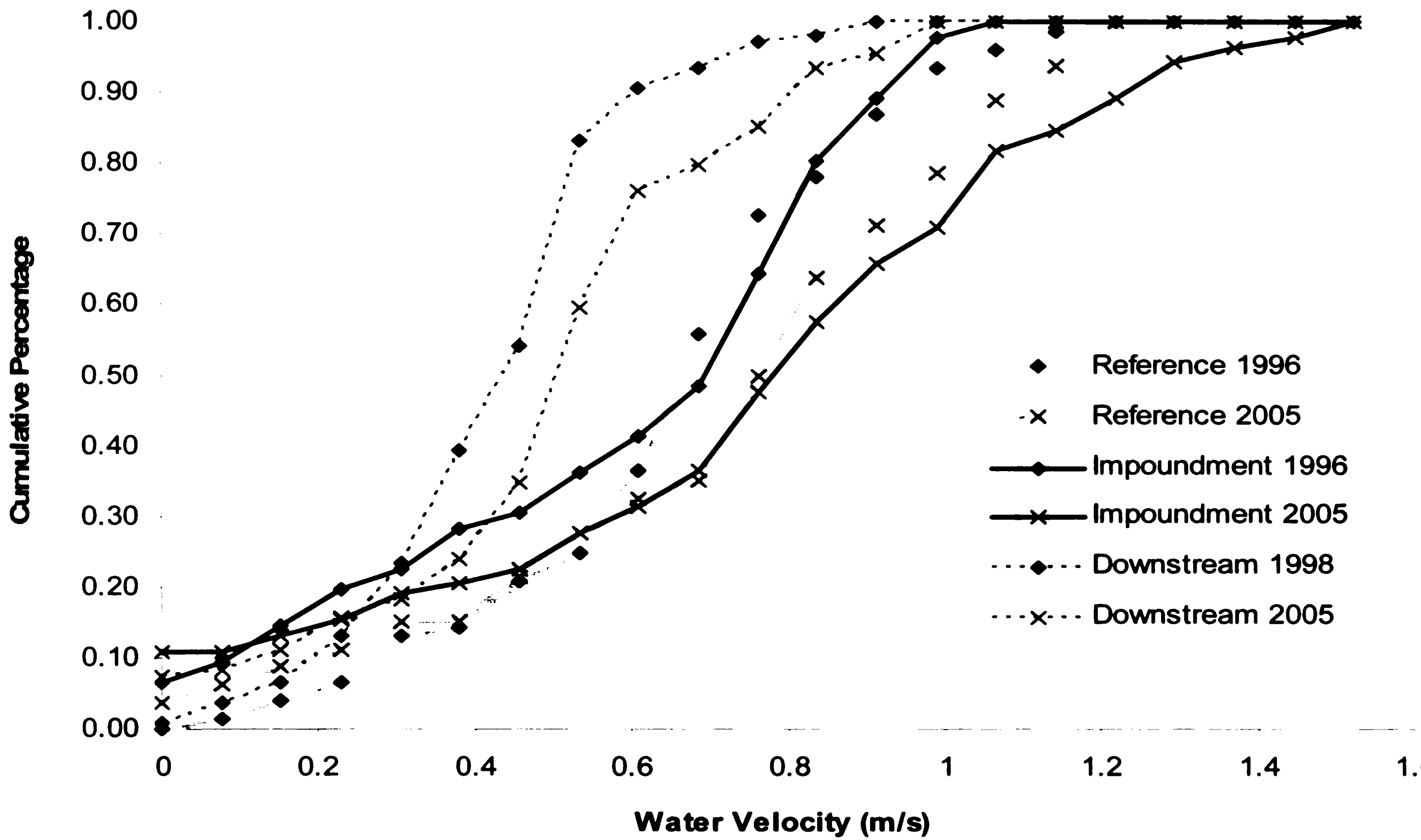

Figure 9. Cumulative percent frequency distributions for water velocities in each study zone of the Pine River.

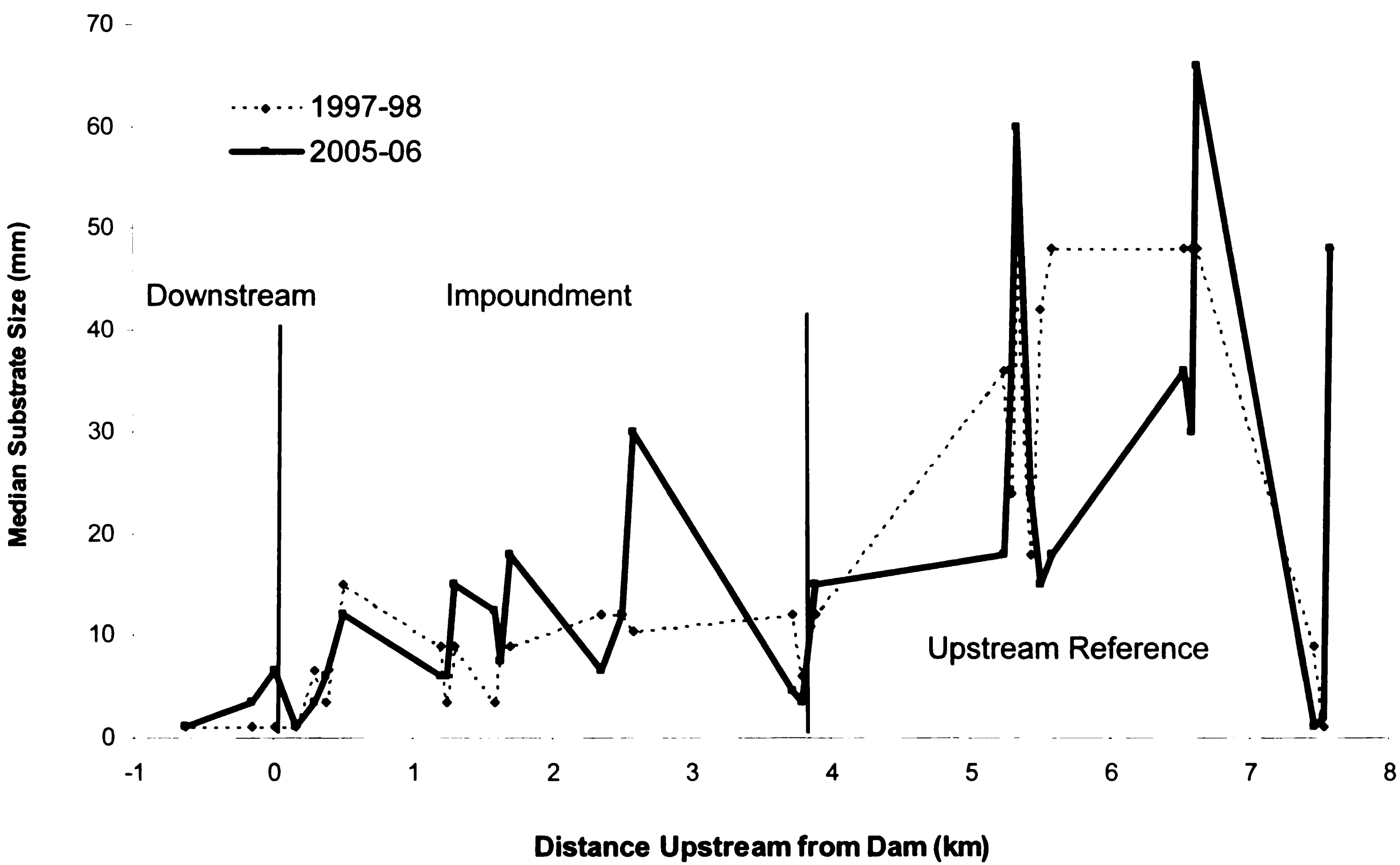

Figure 10. Longitudinal pattern of median substrate size.

0.01) (Figure 11). Median substrate size increased slightly in the impoundment (Impoundment: D_{50} 1997-98 = 7.9 mm = “fine gravel”, 2005-06 = 9.7 mm = “medium gravel”) (Figure 10). The substrate size frequency distribution for the impoundment was significantly coarser in 2005-06 than for 1997-98 (Impoundment: K-S test $D_{max} = 0.087$, $n_1 = 1574$ $n_2 = 1567$, $p < 0.01$), with increased frequencies of large gravel (12 – 48 mm) (Figure 11). Median substrate size increased slightly in the downstream zone (Downstream D_{50} 1997-98 = 1.0 mm = “sand”, 2005-06 = 2.3 mm = “very fine gravel”) but the overall substrate size frequency distribution for the downstream zone did not change significantly (Downstream: K-S test $D_{max} = 0.119$, $n_1 = 191$ $n_2 = 184$, $p > 0.01$) (Figures 10 and 11).

Larger transient changes in median substrate size were seen during the dam removal. It appeared as though during years when less than roughly 15,000 m^3 of sediment were eroded from the impoundment, substrate coarsening progressed in both the impoundment and downstream reaches (Figure 12). However, during years when large volumes of sediment (>15,000 m^3) were eroded from the impoundment (1999-2000 and 2003-2004), this sediment (mostly sand), was transported through these reaches, covering up previously coarsened substrate, and decreasing median substrate size (Figure 12). An exception to this trend occurred in 2006, three years after the removal, when no net sediment erosion occurred but median substrate size still decreased.

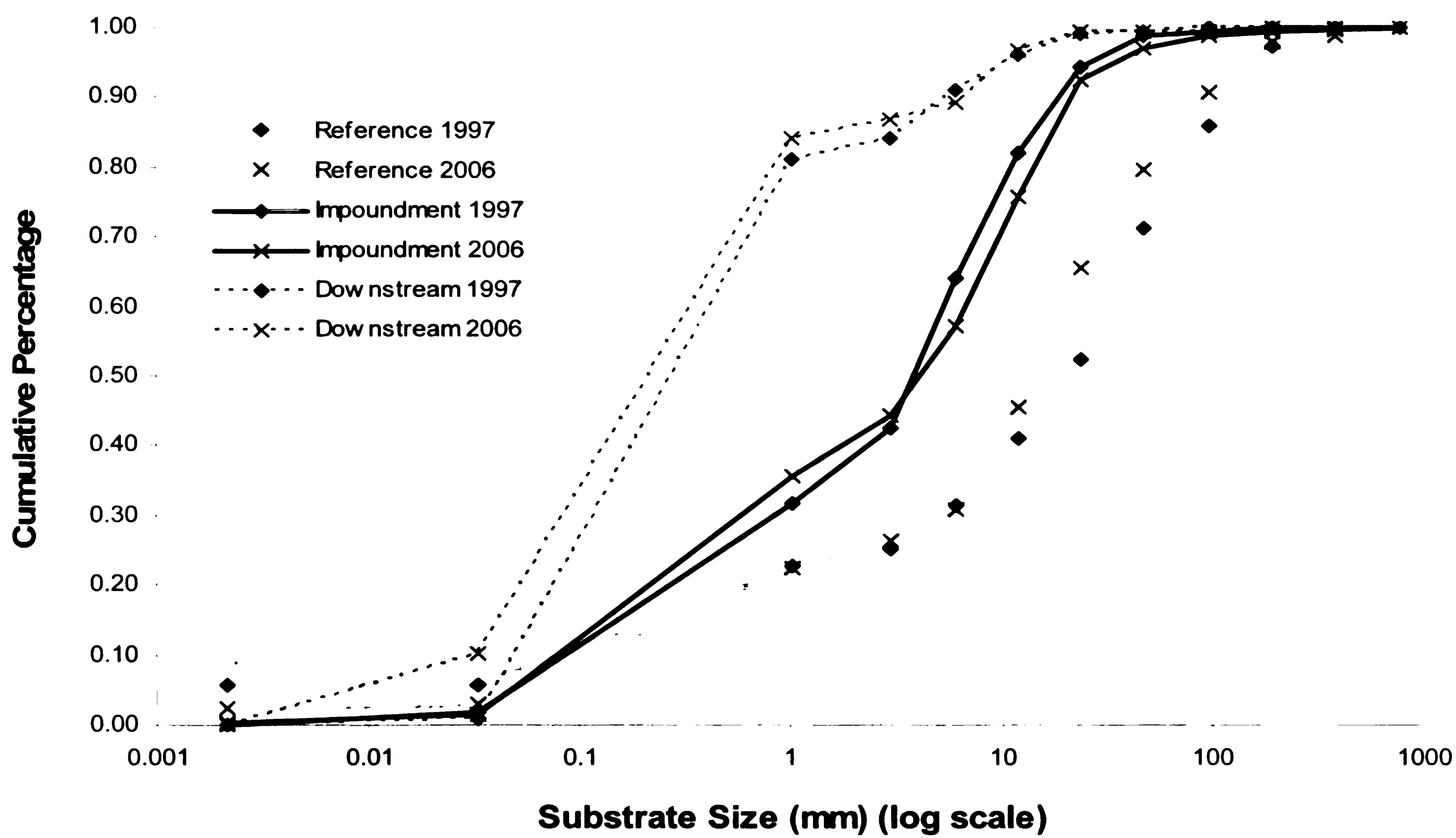

Figure 11. Cumulative percent frequency distributions for substrate size compositions in each study zone of the Pine River.

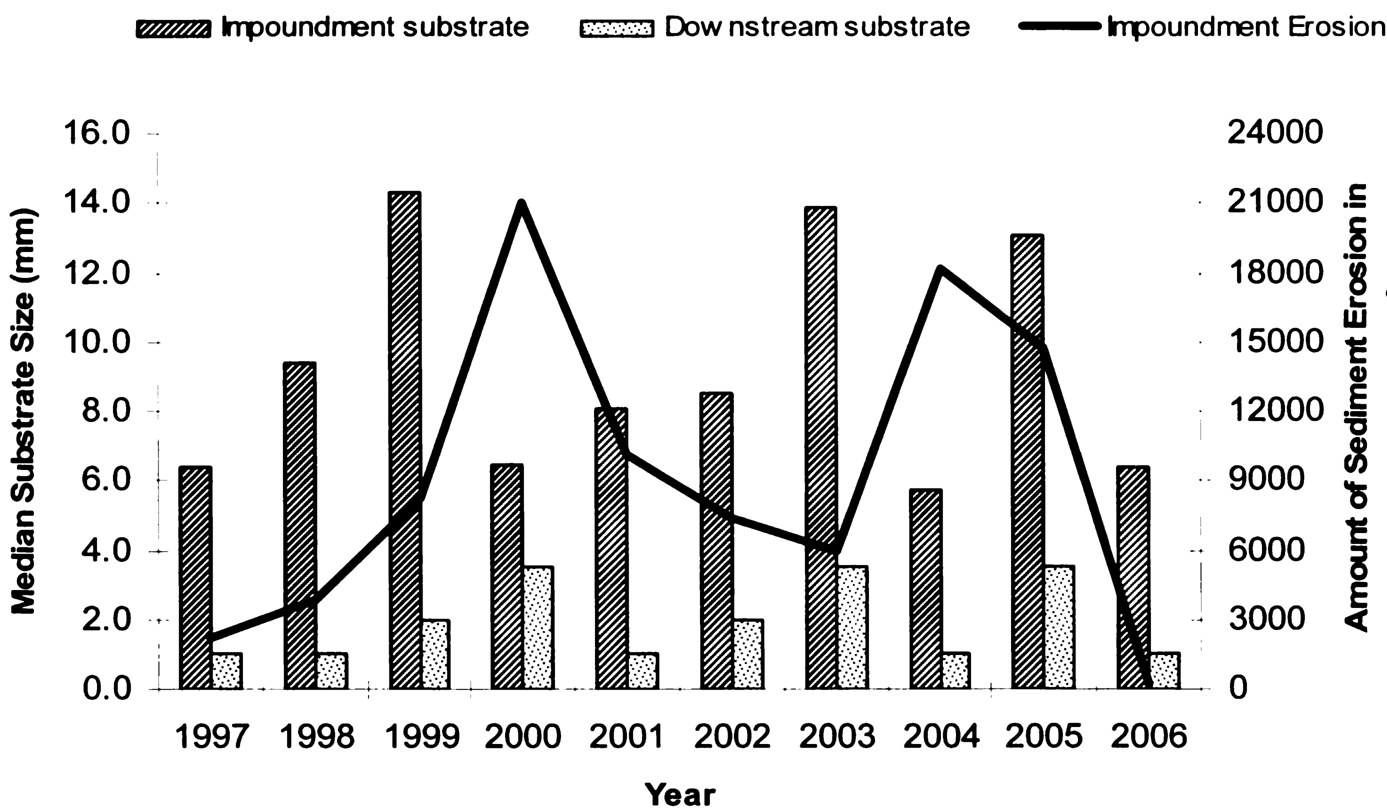

Figure 12. Annual median substrate sizes in the impoundment and downstream study zones shown in relation to the annual amounts of sediment erosion occurring in the impoundment zone each year of the staged dam removal.

Bedform

In 1995, the upstream reference reach contained the highest heterogeneity of bedforms, and both the impoundment and downstream zones were comprised almost exclusively of run bedforms. In 2004, the reference reach had changed little from the 1995 survey but there were some differences in the percentages of the pool/complex and rapid designations, and minor differences in the percentages of run bedforms (Table 2). These changes were most likely due to differences between sampling crews in the designations of complex bedforms versus run or pool bedforms, and do not likely represent any real changes. In 2004, the impoundment had higher percentages of riffles and pools, and lower percentages of runs than in 1995 (Table 2). The downstream zone remained overwhelmingly run bedform in the 2004 survey, gaining only one pool unit (Table 2).

Table 2. Percentages of bedforms types found in each study zone in 1995 - prior to dam removal, and 2004 - after dam removal. Pool and complex bedforms were aggregated due to issues in the repeatability of complex bedform delineation.

Study Zone		**Run**	**Riffle**	**Pool / Complex**	**Rapid**
Upstream Reference	1995	44.0	32.9	16.3	6.8
	2004	41.5	32.8	23.6	2.1
Impoundment	1995	96.4	1.4	2.2	0
	2004	68.3	13.8	17.9	0
Downstream	1995	100	0	0	0
	2004	96.9	0	3.1	0

DISCUSSION

Impoundment

Each stage of the removal of Stronach Dam increased slope in the area immediately adjacent to the dam, leading to high water velocities over the downstream edge of the reservoir sediment fill at the site of the dam. The fine-sized sediments deposited in the impoundment and held by the dam were then subjected to increased shear stress and stream power, and were subsequently eroded. As this sediment erosion progressed upstream, it led to a dissipation of the slope difference caused by the dam removal, distributing increased slope upstream over longer sections of stream. Erosion continued due to this section of stream having increased slope, increased water velocity, and higher stream power, but still flowing over a channel of fine sediment that had accumulated in the slow water of the impoundment. This process should have continued to occur until the stream came into equilibrium with the new slope, sediment discharge from upstream, and coarsened substrate (Lane 1955). In the case of the staged removal of Stronach Dam, this equilibrium was not likely reached before successive stages of the removal occurred. An implication of this is that sediment erosion will continue to occur upstream of a dam removal, not only due to headcutting and immediate slope differences, but also at mean flows, until the stream channel reaches an equilibrium with its new slopes, substrate sizes, and increased sediment discharges coming from upstream eroding sites.

Relatively small amounts of sediment eroded during the first two years of the removal, likely due to the low slope of the impoundment at that time. As the removal progressed, the slope increased in the impoundment and on average more sediment erosion resulted each year, with the second and third highest amounts of erosion coming in the first two years following completion of the dam removal. These amounts did not correlate well with simple mean flows, peak flows, time duration over bankfull flow, approximate stream power (discharge and slope) or even the height of dam removed during each stage. Further explanation of the temporal variability of sediment erosion would likely require a site-specific approach, incorporating transect level differences both in space and time.

Despite the variability and complexity in estimating yearly amounts of erosion that will occur in impoundments behind dam removals, the total amount of erosion that is likely to occur following a dam removal seems to be fairly easily estimated. Not all of the sediment in a reservoir will be mobilized. In the Pine River the size of the reservoir that was filled with sediment was 789,428 m^3. However, as the stream channel eroded vertically through this sediment fill, the width of the wetted channel decreased and became very close to the average width of the upstream reference. The underlying slope of the reservoir was only slightly higher than the upstream reference, and consequently the mean width of the stream channel in the impoundment (17.6 m) became similar to the mean width of the stream in the reference (16.9 m). Therefore, the volume of sediment to be eroded due to dam removal could be estimated by (H*L/2)*W where; H is maximum height of the sediment fill, L is the longitudinal distance of the sediment fill, and W is the average width of stream immediately upstream of impoundment effects (Figure 13). For reservoirs such as Stronach Dam, where the reservoir is completely filled with sediment, H would equal the dam height and L would equal the length of the impoundment (delineated with a bedform survey). For reservoirs not completely filled with sediment, H would equal the maximum height of the sediment fill, usually at the downstream or leading edge of the sediment delta, and L would equal the length of the sediment fill (upstream extent delineated with bedform survey and downstream extent with a bathimetric survey). H*L is divided by 2 to produce an area estimate for the triangular shaped sediment fill or delta. In the case of Stronach Dam, this yields an estimate of (3.66 m*3800 m/2)*16.9 m = 117,522 m^3. Through 2006, approximately 92,000 m^3 eroded from the impoundment, with no new net erosion occurring during 2006. This estimation method over-predicted the amount of sediment eroded, due in part to the upper portion of the Stronach Dam impoundment not approximating a perfect triangle or sediment wedge, and partly due to the rectangular cross-sectional shape our approximation assumes (Figure 13). However, this simple estimation method allows for a conservative (in regards to pre-cautionary management) and useful pre-dam removal estimate of the volume of sediment that is likely to erode. If significant tributaries entered the impoundment, this method would have to be applied to these as well, and the results combined with the estimates from the main river channel.

In the case of Stronach Dam, only about 12% of the reservoir sediment was mobilized and transported. Similarly small percentages of reservoir sediment fill mobilization were found by Evans et al. (2000) in an Ohio dam

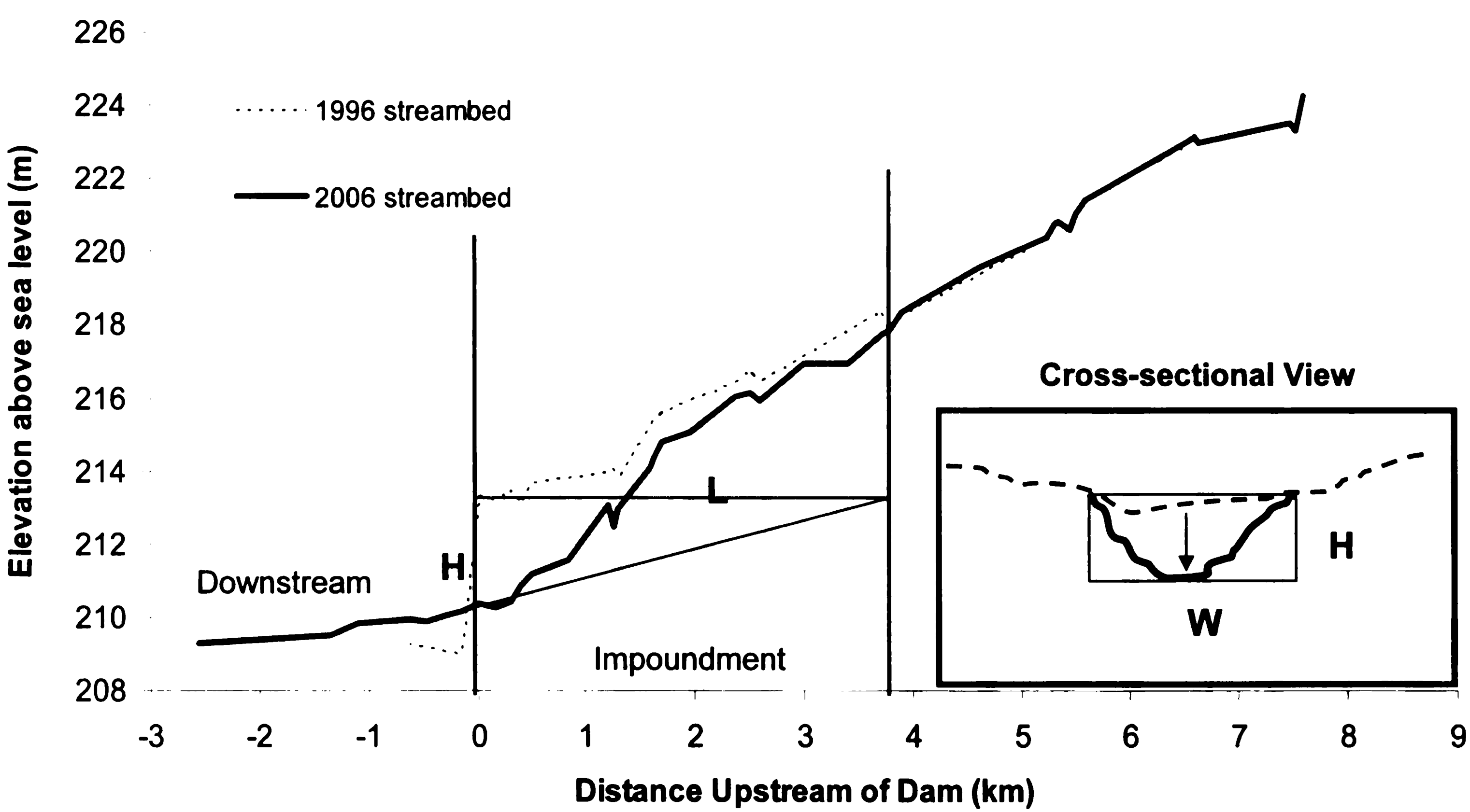

Figure 13. Longitudinal profile of the Pine River streambed before and after dam removal. The triangle approximates the area of the sediment fill, and along with average stream width upstream of the impoundment, can be used to estimate the volume of sediment that can be mobilized at a dam removal.

failure (9 – 13 %) and Doyle et al. (2003) in a Wisconsin dam removal (8 – 14%). The amount of sediment to be mobilized with a dam removal is an important consideration during the planning and decision making process in removing dams (Doyle et al. 2002, Randle 2003, Rathburn and Wohl 2003). In situations where sediment transport downstream from dam removals is undesirable, sediment removal or management can add considerably to the expense of removing dams. Given the small percentage of the reservoir sediment fill that will likely be transported following a dam removal, it should be more cost-effective to manage sediment transport downstream of a dam removal (e.g., through the use of sediment traps or collection devices) than to remove larger amounts of sediment from the impoundment prior to dam removal (e.g., through dredging).

As the streambed incised in the impoundment and slope increased, the wetted width of the stream also generally decreased. Differences in the localized geology and slope between transects affected how much stream width decreased, but the average stream width in the impoundment became remarkably similar to the average stream width of the reference. These localized differences in geology and slope exerted greater influence on water depth at transects, and led to large differences in w/d ratio changes between transects due to the dam removal.

Changes in the slopes of stream banks following dam removal have been predicted to follow Channel Evolution Models (CEM's) (Pizzuto 2002, Doyle et al. 2002). These models were developed from incising channels and predict that following dam removal, bank slopes in impoundments should increase along with vertical incision (and so should be steeper, initially, closest to the dam). Banks should continue to steepen with further incision, until a point is reached where the slope is too great for the cohesive forces of the sediment or vegetation to continue holding it together, causing slumping and a reduction in bank slope, allowing for the development of equilibrium channel dimensions. In the Stronach Dam impoundment, bank slopes did increase gradually during the dam removal, with the first and greatest increases occurring closest to the dam. Bank slopes continued to increase and during the last year of the study decreases in bank slopes were observed only at sites further upstream in the impoundment, where streambed erosion had ceased. However, bank slopes in the reference reach also exhibited changes of similar magnitude (albeit different direction) during the study, indicating bank slopes in the Pine River may be naturally dynamic and variable. While some patterns consistent with the CEM seem discernable from this data, the natural variability in bank slopes makes interpreting the significance of those patterns difficult.

As the stream channel slope increased in the impoundment following the dam removal, so did the mean water velocities. This is not surprising, but perhaps more interesting is how the frequencies of water velocities changed in the impoundment. The frequency of slower water velocities, (i.e., less than 0.30 m/s) did not change substantially. These slower water velocities primarily occurred near the stream margins. Despite mean velocities increasing, and increased thalweg velocities (e.g., as seen in the creation of faster velocities not previously observed in the impoundment), similar amounts of slower stream margin water velocities still exist. Water velocities therefore not only increased but became more diverse. Changes of this nature will have important implications for sediment and nutrient transport dynamics, and should be beneficial in providing diverse habitat conditions for different species and life stages of aquatic invertebrates and fish.

Average substrate size increased throughout most of the impoundment in response to higher slope and water velocities. Overall increases in the proportions of rocky substrate were similarly observed following the removal of the Woolen Mills Dam on the Milwaukee River (Kanehl et al. 1997). However, substrate frequencies also showed some changes similar to those of water velocities. In the impoundment, where substrates had been dominated by sand before dam removal, frequencies of sand decreased due to the dam removal, but frequencies of silt did not decrease. Similar results were reported by Stanley et al. (2002), shortly after the removal of two low-head dams on the Baraboo River, Wisconsin. The frequency of slower water velocities at the stream margins stayed constant, allowing the retention of finer substrates such as silt. As sand decreased in frequency, so did several size classes of the smaller gravels, and they were replaced by larger size gravels. This shift corresponds with the thalweg velocities increasing in magnitude, not becoming more frequent. Again, this has important implications on stream biota, because the homogenous sand substrate prior to dam removal was not replaced with equally homogenous larger substrate, but with a greater diversity of substrates. At the conclusion of this study, we predict substrate size composition will continue to coarsen in the impoundment area. While substrates are already somewhat coarser, they are still smaller than predicted to be stable, even under typical flows (Burroughs unpublished data). This means that while substrate is coarser, it is still considered unstable, and not as beneficial to stream biota as possible (Gordon et al. 2004).

The alternating patterns of riffles, runs, and pools in mixed gravel streams is seen as a way rivers self-adjust to regulate energy expenditure, and are very important to the biological productivity of streams (Gordon et al. 2004). These bedforms can be created by localized scour during normal flows, at river bends or by wood debris, but are normally formed by rare high flow events (Petts and Foster 1985, Knighton 1984, and Beschta and Platts 1986 as cited in Gordon et al. 2004, and Pizzuto 2002). During the period of the Stronach Dam removal, a 1 in 5 year flood occurred, but a 1 in 10 year flood did not. Despite this, some new riffle and pool bedforms formed in the impoundment reach. In 2004, the diversity of these bedforms was not as high as seen in the reference reach, and may not be realized in the impoundment until very high flows are experienced. Long time durations for the complete recovery of bedforms were predicted by Pizzuto (2002) based on short time duration empirical studies and laboratory flume experiments. Bushaw-Newton et al. (2002) also noted that riffle-pool bedforms had not reformed in the impoundment with one year of the removal of Manatawny Creek Dam. This has important implications for the functioning of streams as bedform diversity influences sediment transport and sorting, nutrient cycling, and is crucially important to the habitat suitability of stream biota (Gordon et al. 2004). Consideration of the long timeframe for bedform restoration may lead to innovative ways of actively helping stream rehabilitation following dam removals. For example, if another water control structure existed upstream of a dam removal site, water releases could be negotiated to allow pool-riffle forming flows; or various structures such as wood debris or gravel bars could be added to the stream to aid bedform formation. Managing for these bedforms may lead to a faster realization of benefits of stream rehabilitation through dam removal.

Another insight derived from the consideration of bedform diversity, concerns the delineation of the impoundment effects boundary. If the top height elevation of the dam was followed upstream, it would roughly show the boundary of the formal reservoir, where water widths would likely be very wide and water impoundment would be most noticeable. However, we used a simple means of delineating bedform types, and performed this mapping of bedforms for many kilometers upstream of the dam. Through this, it became apparent that bedform diversity was lacking for a considerable distance upstream of the formal reservoir. In addition, it was noticeably sand-dominated in this area of little bedform diversity. The furthest upstream extent of this sandy run habitat became our upstream boundary of impoundment effects, and the extent of where we expected to see changes due to the dam removal. This method, along with the use of aerial photos, proved quite accurate in delineating the furthest upstream extent of impoundment effects and changes from the dam removal. We recommend this as an easy, and cost-effective technique to predict how far upstream changes may take place following dam removal. This information can be important in the early dam removal planning process for assessing possible impacts (e.g., infrastructure concerns, mitigation measures, and landowner impact assessments).

Downstream

As each stage of the dam was removed, the drastic difference in elevation from the top of the sediment fill to the downstream side of it, created a situation where water velocities were extremely high and flow became supercritical. Transport of the sandy substrate proceeded with antidune formation. Further downstream where slope and water velocities decreased, sediment transport continued with dune and ripple formations (Gordon et al. 2004). The transport of this sediment aggraded the streambed, increasing the slope of this section of stream downstream of the dam, decreasing the water depth, and slightly increasing the stream width. These changes led to higher sediment transport ability, but the sediment eroded from the impoundment was in excess of the transport capacity of this stretch of stream. At the end of this study, approximately 14% of the sediment eroded from the impoundment was retained and stored in the first 1 km of river downstream from the dam. The rest of the sediment was either transported further downstream, forming a sediment delta at the confluence of the next reservoir downstream, or deposited on the the floodplain during high flows. As the stream channel downstream from the dam aggraded, the elevation difference between the stream and the floodplain decreased. During high flows, suspended sediments were deposited onto the floodplain, vertically raising the top bank by as much as a 0.50 m at one of the transects.

The implications of this could be important in considering the impacts from dam removals. During the erosion process upstream of dams, the streambed is lowered drastically, and can reduce connectivity with adjacent floodplain wetlands. With dam removal permitting, the removal of wetlands could be seen by permitting agencies as needing remediation (even if the wetlands were created by the dam construction). However, if a river valley downstream of a dam removal is not steep and narrowly confined, floodplain connection and recharge (frequent overbank flooding) in this stretch of stream could be enhanced, leading to the recharge of historic wetlands or the creation of new ones. Frequent overbank flooding was observed in the downstream reach of the Pine River following the dam removal, and has been predicted to occur following the removal of other dams (Stoker and Harbor 1991, Randle 2003).

The sediment deposition in this downstream reach resulted in a substantial decrease in water depth and an increase in width, together greatly increasing in the w/d ratio of this reach. This increase in w/d ratio reached a peak during the later stages of dam removal, and has begun to decrease during the last few years after the removal was completed. At the conclusion of this study, the w/d ratios were only slightly higher than pre-removal levels.

With the increased slope in the downstream zone, water velocities also increased. Average water velocities increased in this section as well as frequency distributions changing significantly. As with the impoundment, water velocity frequencies in the downstream zone, changed in a manner as to increase the variability of velocities. Frequencies of slower water velocities (<0.3 m/s) increased as the stream channels became wider and shallower. At the same time, faster water velocities became faster, leading to an overall greater diversity of water velocities.

Substrate size increased only slightly in the downstream zone. This section of stream, while having faster velocities and higher slopes, continued to receive sediment from the eroding impoundment. The median size of substrate increased very slightly but the frequencies of substrate did not change significantly. The substrate composition of this zone will likely stay dominated by sand until the former impoundment section reaches an equilibrium. At that time, the downstream reach should experience some substrate coarsening immediately downstream of the dam. However, this section may stay relatively sand dominated due to the close proximity of the next downstream reservoir and its impoundment effects.

Changes in bedform diversity downstream of Stronach Dam may never occur, or may take a very long time to happen. This section is much lower gradient due to the downstream impoundment behind the Tippy Dam, and even during rare high flows may not have the power to scour pools. This section was all run bedform before dam removal, and remained largely run bedform at the end of this study. Even if pools were scoured, until the substrate becomes more diverse, including more gravel and cobbles, those pools may not be easily maintained in a stretch of stream with easily movable sand.

Synthesis

This study achieved its objectives of documenting the spatial and temporal dynamics of sediment erosion, transport and deposition following the removal of Stronach Dam. These processes and the subsequent changes in river morphology, were spatially and temporally variable in magnitude and extent, but in general were clearly understandable using the principles of fluvial geomorphology. At the end of this study it is apparent that changes are still occurring in the Pine River due to the removal of Stronach Dam, 10 years after its initiation. While sediment incision in the impoundment seems to have finally slowed, three years after dam removal, lateral erosion, substrate coarsening, and bedform formation will likely continue in the Pine River for many years. It now appears that dam removal does have the potential to be an effective tool for stream rehabilitation, but many of the outcomes make take relatively long periods of time to be realized. It is our hope that this research serves as valuable starting point for future research on the effects of dam removal, and as a tool to improve the effectiveness and efficiency of future dam removals.

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CHAPTER 2

THE EFFECTS OF THE STRONACH DAM REMOVAL ON FISH IN THE PINE RIVER, MANISTEE COUNTY, MICHIGAN

INTRODUCTION

Dams affect river systems in a myriad of direct and indirect ways including disrupting the flow of water, energy, sediment, nutrients, and biota (e.g., Hammad 1972, Petts 1980, Williams and Wolman 1984, Cushman 1985, Bain et al. 1988, Ward and Stanford 1989, Benke 1990, Ligon et al. 1995). These changes impact lotic fish communities both through habitat alteration and fragmentation (Hayes et al. in press). Habitat alteration occurs upstream and downstream of dams, but in fundamentally different ways. Upstream from dams, the flow of water, sediment, and nutrients is slowed, creating impoundments and converting lotic habitat to lentic habitat (Petts 1980, Ward and Stanford 1989). This decreases habitat suitability for lotic species, and often leads to the juxtaposition of lentic fish communities in impoundments with upstream resident lotic species. Downstream of dams, habitat is altered through the reduction of sediment supply, subsequent erosion, water temperature changes and flow variability (Williams and Wolman 1984, Cushman 1985, Ligon et al. 1995, Collier et al. 1996). This often leads to the displacement of resident fish species and the colonization of other invasive or non-native fish species (Martinez et al. 1994, Quinn and Kwak 2003). From a fisheries management perspective, changes in resident fish communities can be seen as either detrimental or beneficial depending on conservation status and fishery values of the fish species being lost or gained.

Dams also impact fish communities through habitat fragmentation. All fish species need access to habitats essential for reproduction, feeding and survival. The placement of dams on rivers prevents or impedes movements for many fish species. For diadromous fishes, dams can block essential fish reproductive migrations, and the migration of juveniles to feeding habitats, often with severe consequences for these fish populations (Benke 1990, Pringle et al. 2000). Many non-diadromous riverine fish species also make substantial migrations critical to their life histories and survival (Auer 1996, Northcote 1998, Burrell et al. 2000). These movements include downstream drift of juveniles, movements to and from over-wintering habitat, movements to thermal refuges, migrations to preferred spawning habitat, and searching movements crucial for individual fish to locate optimal areas for feeding and holding. While not historically recognized as being as important as migrations of diadromous fishes, these upstream and downstream movements by riverine fishes may also be vital to the sustainability of these fish populations.

For numerous reasons (reviewed by Pohl 2002, Heinz 2002, Stanley and Doyle 2003, and Burroughs 2007) dam removal has recently become a popular stream restoration method for remedying both fish habitat alteration and fragmentation, as well as rehabilitating overall river ecosystem form and function. The popularity of dam removal as a river restoration technique stems almost entirely from the well-documented negative impacts of dams and hypotheses about the reversal of these impacts following dam removal. Empirical information on the effects of dam removal on fish is very limited. Several qualitative observations exist that anadromous fish have been seen to migrate upstream past dam sites following removal (American Rivers 1999, Smith et al. 2000), and one recent study documented and quantified the successful spawning of striped bass (*Morone saxatilis*), American shad (*Alosa sapidissima*) and hickory shad (*Alosa mediocris*) upstream of a dam following its removal (Burdick and Hightower 2006). However, despite the more than 400 dams removed in the United States (Pohl 2003), very few published studies exist where the effects of dam removal on fish populations were quantified. Hill et al. (1994) found that following the removal of Chipola Dam in Florida, largemouth bass (*Micropterus salmoides*) recruitment, while becoming highly variable, also increased substantially on average due to restored flow variability. Migratory striped bass were also seen using the river as thermal refuge and the total number of species present in the river upstream of the dam increased from 34 to 61 following dam removal. Kanehl et al. (1997) documented large increases in the recruitment and density of smallmouth bass (*Micropterus dolomieu*) in the Milwaukee River upstream of the Woolen Mills Dam removal site, a decrease in the density of common carp (*Cyprinus carpio*), and an increase in fish community biotic index scores. While these studies provide unique and valuable insight into the effects of dam removal on fish communities, they do not fully represent the range of variability that exists in North American rivers and their fish communities. For example, no published quantitative studies currently exist documenting the effects of dam removals on fish in a coldwater stream. The current state of knowledge regarding the effects of dam removal is limited, providing only a precursory understanding of the outcomes of removing dams on fish. Fishery resources are an important consideration in dam removal decision-making processes. Given the current void in our understanding of this emerging and important topic, additional information on this subject is needed to inform future decision making regarding removing dams.

The "staged" or gradual removal of Stronach Dam, on the Pine River, in Manistee Co. Michigan, created an opportunity to gain insight into both the effects of habitat alteration on a fish community following dam removal and also the effects of restored connectivity and subsequent fish movements and species invasions. The Pine River is a coldwater stream with another moderate –sized reservoir located just downstream from the Stronach Dam removal site and thus has coldwater, coolwater and warmwater fish species utilizing the river, at least seasonally. Self-sustaining populations of resident brown trout (*Salmo trutta*) and rainbow trout (*Onchorynchus mykiss*) provide a valuable sport fishery upstream of the Stronach Dam site, and habitat alteration following the dam removal was anticipated to improve habitat conditions for trout throughout the approximately 4 km long impoundment. Downstream of the dam, a different fish community existed including 18 species of fish not found upstream of the dam. Due to the staged nature of this dam removal, several years of habitat alteration occurred during the early stages of the dam removal before fish passage was possible. This allowed insight into the effects of habitat alteration following dam removal, before connectivity was eventually restored during the final stage of dam removal.

The specific research objectives of this study were to: 1) document the changes in fish habitat that occurred due to dam removal, 2) document changes in the distributions of fish species in the Pine River following the dam removal, 3) document density changes for brown trout, rainbow trout, brook trout (*Salvelinus fontinalis*), white sucker (*Catastomus commersoni*), and shorthead redhorse sucker (*Moxostoma macrolepidotum*) occurring before, during and after the dam removal and 4) document changes in the size structure of those species of fish. Objective 1, the documentation of changes in fish habitat due to the dam removal is covered in detail in Burroughs (2006), and will be summarized here.

Site Description

Stronach Dam is located on the Pine River, a tributary to the Manistee River, in the northwestern Lower Peninsula of Michigan (Figure 1). The Pine River is 77 km long, is a fourth order stream, and drains a 68,635 ha watershed dominated by sandy glacial outwash plains, recessional moraines, and areas of consolidated clay (Hansen 1971, Rozich 1998). It carries a high bedload of sand due to the local geology and extensive logging operations of the late 1800's, which created unstable banks along the river. The Pine River is a coldwater stream, dominated by groundwater input, and rarely exceeds 21° C. Mean daily discharge recorded at two U.S. Geological Survey gaging stations on the Pine

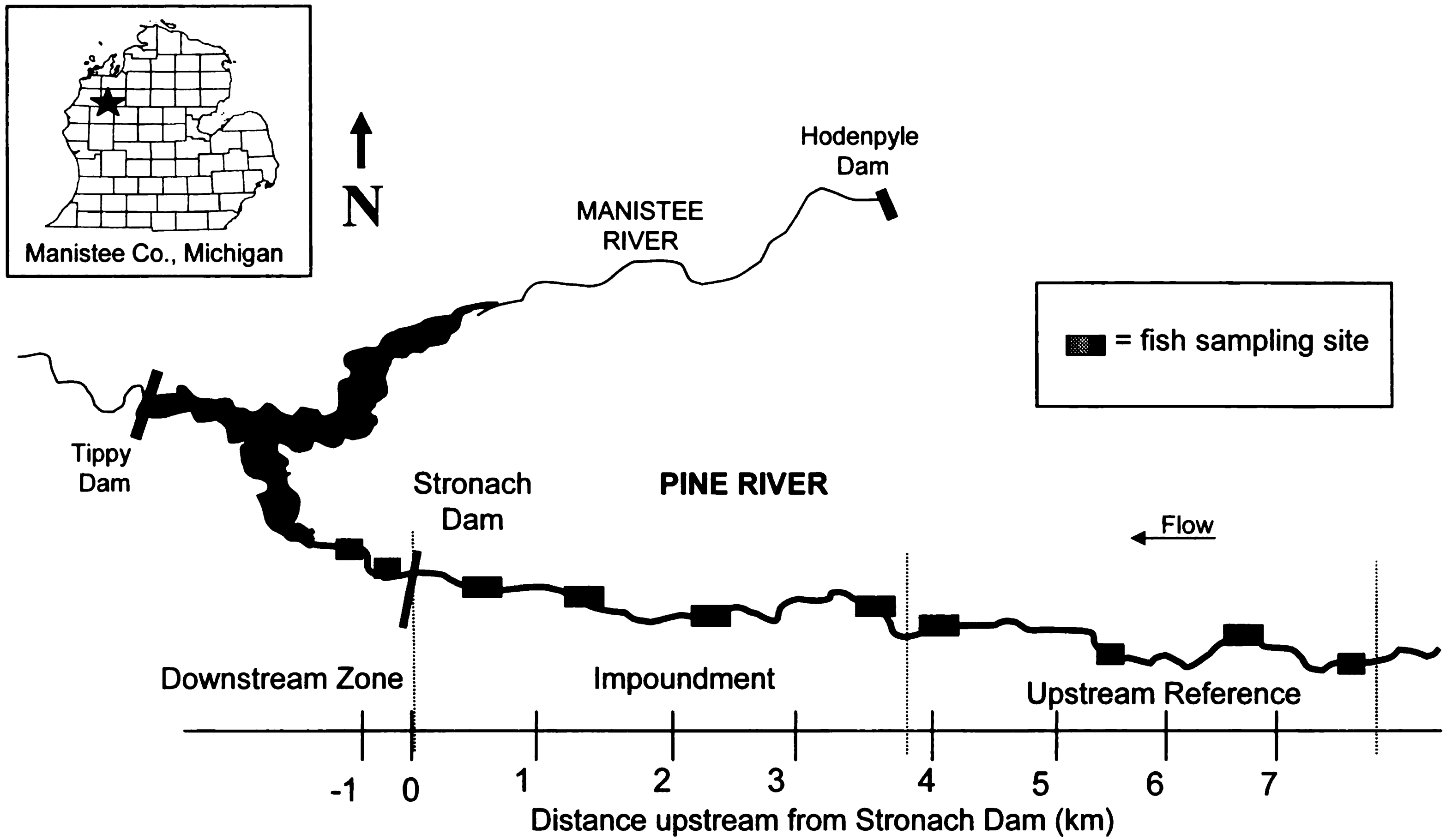


Figure 1. Locations of Stronach Dam and the Pine River in relation to the State of Michigan; and the location of permanent fish sampling sites within the study area of the Pine River.

River averaged 8.10 m^3/s during 34 years of record, with an average annual ratio of low to high mean monthly flows of 2.02, indicating "stable to very stable" flows (Rozich 1998). The Pine River is a riffle-pool stream with an average gradient of 2.8 m/km. The section of river impounded by Stronach Dam historically had a gradient of 4.7 m/km, and was reported to be the best fish spawning area of the river (Rozich 1998).

Stronach Dam was constructed from 1911 to 1912, 5.6 km upstream from the confluence of the Pine River and the Manistee River (Figure 1). Stronach Dam was an earth embankment dam with a concrete corewall; a 4.57 m fixed-concrete spillway section with 0.91 m of flashboards on top of the spillway; a concrete and brick powerhouse with two turbine bays; and an upstream fish ladder (Consumers Power Company 1994). Stronach Dam, with 5.49 m of head height possible, was operated mostly around 5.18 m of head. This created a 26.7 ha reservoir with a 789,428 m^3 volume capacity (Consumers Power Company 1994, Hansen 1971). Tippy Dam (17.07 m head height) was constructed in 1918 immediately downstream of the confluence of the Pine and Manistee Rivers (Rozich 1998). It created a 428 ha, 48,722,530 m^3 reservoir which impounded water upstream to Stronach Dam and blocked all upstream fish migration from Lake Michigan (Figure 1).

Due to the Pine River's large sediment load, the reservoir quickly filled with sediment and problems arose with the operation of the dam's turbines. Attempts were made in the 1930's to remove the accumulation of sediment behind the dam. These efforts were only marginally successful and dredging eventually became uneconomical (Consumers Power Company 1994). In 1953, 41 years after the dam's construction, Stronach Dam was decommissioned by the owner, Consumers Power Company. The generator rooms were demolished, the fish ladder was removed, and the river flow was directed over the spillway. The spillway flashboards were removed gradually over the following years; the last was removed in 1983 (Consumers Power Company 1994).

In the early 1990's, removal of Stronach Dam was negotiated as part of a Federal Energy Regulatory Commission (FERC) agreement in the relicensing of Tippy Dam. Removal of Stronach Dam began in the spring of 1997 and was completed in December of 2003. A "staged" or gradual removal was decided upon in order to allow gradual river channel adjustments with the least amount of environmental impact, at the lowest cost, and without impacting the operation of Tippy Dam (Battige et al. 1997). In 1996, a 3.6 m high "stop-log" structure was installed in the old powerhouse to allow a gradual drawdown of the river. The stop-log structure consisted of hollow metal pipes (15 cm diameter) stacked one on top of another, with a metal grate called a "trash-rack" immediately upstream to protect the stop-logs from debris impingement. The original removal schedule called for one stop-log to be removed every three months, for a total of 0.60 m per year, over the course of six years; with corresponding trash-rack removal. This plan was altered due to recreational safety concerns, feasibility issues, and technical difficulties with removal (Battige personal communication). Table 1 shows the actual sequence of the staged dam removal.

Table 1. Schedule of removal events during the staged removal of Stronach Dam on the Pine River, Manistee County, Michigan. Stop-logs are 15.24 cm diameter hollow metal pipes stacked on top of one another. Trash-rack removal estimates are approximate. Cumulative meters removed are in parentheses. (Dave Battige, Consumers Energy, personal communication 2003).

Date	**Number of Stop-logs removed**	**Meters of Trash-rack removed**
March 17, 1997	1 (0.15)	0 (0)
June 5, 1997	1 (0.30)	0 (0)
June 16, 1997	2 (0.61)	0 (0)
June 24, 1997	2 (0.91)	0 (0)
September 15, 1997	1 (1.07)	0 (0)
December 15, 1997	1 (1.22)	0 (0)
March 16, 1998	1 (1.37)	0 (0)
May 7, 1998	0 (1.37)	1.83 (1.83)
May 29, 1998	0 (1.37)	0.30 (2.13)
June 15, 1998	1 (1.52)	0 (2.13)
September 8, 1998	1 (1.68)	0.30 (2.44)
December 14, 1998	1 (1.83)	0.30 (2.74)
March 15, 1999	1 (1.98)	0 (2.74)
May 11, 1999	1 (2.13)	0 (2.74)
September 13, 1999	2 (2.44)	0 (2.74)
September 16, 1999	0 (2.44)	0.61 (3.35)
April 17, 2000	2 (2.74)	0 (3.35)
October 2, 2000	2 (3.05)	0 (3.35)
October 5, 2000	0 (3.05)	0.61 (3.96)
May 8, 2001	2 (3.35)	0 (3.96)
September 8, 2001	2 (3.66)	0 (3.96)
November 11, 2002	0 (3.66)	1.52 (5.49)
December 2003	Remaining spillway and dam superstructure removed	

METHODS

In 1995, two years prior to the commencement of dam removal activities, the Pine River was assessed to document the spatial extent of impoundment effects due to Stronach Dam. This assessment involved the surveying and description of physical characteristics, including categorization of the stream into bedform units of runs, riffles, pools, or rapids, following the criteria developed by Hicks and Watson (1985). This survey allowed detection of impoundment effects well upstream of the readily noticeable reservoir area. This "impoundment" area of the river extended for 3.89 km upstream of Stronach Dam and was relatively wide, slow-flowing, sand-bottomed, and generally consisted of only run bedform units. An upstream reference reach was chosen, extending for 3.70 km upstream from the upstream boundary of the impoundment. This study zone was chosen as a reference reach because no effects on river morphology were evident. The river was narrower, faster-flowing, had coarser substrates, and showed high bedform heterogeneity. A third study zone was chosen downstream of Stronach Dam, where the river was wide, very slow-flowing, sand-bottomed, and consisted entirely of run bedforms. Prior to the removal of Stronach Dam, water was impounded in this study zone from Tippy Dam Reservoir, and the zone extended for only 0.63 km downstream of Stronach Dam.

Bedform frequency (also referred to as meso-habitat), latitudinal and longitudinal channel morphology, water velocity, and substrate size composition were documented in the three study zones, annually from 1996 through 2006 (see Burroughs 2007 for full review of the methodology used).

Fish were sampled in the Pine River from 1997 through 2006 with a 17-foot Smith-Root Cataraft® electrofishing boat. The electrofishing boat was set to deliver pulsed DC (40% cycle duty) on low range (50 – 500) volts at 4 – 6 amps. Fish community composition and species abundances were sampled at 10 sites along the river (Figure 1), once per year (mid-July to early August), from 1997 through 2006. Four sites were located in the upstream reference reach, four sites were located in the impoundment, and two sites were located in the downstream reach. Each site was enclosed with block-nets and multiple pass removal sampling was conducted in order to estimate fish population sizes (VanDeventer and Platts 1983). A minimum of three passes were made at each site; occasionally, additional passes were made in order to achieve a clear depletion pattern in catch. Fish captured were identified and total length was measured to the nearest millimeter.

Abundance was estimated for brown trout, rainbow trout, brook trout, white suckers, and shorthead redhorse suckers. Preliminary analyses suggested lower gear selectivities for smaller fish. As such, abundance estimates were conducted by size class. For brown trout the two size classes were <130 mm and ≥130 mm total length. For rainbow trout the two classes were ≤100 mm and >100 mm, for brook trout the classes were ≤110 mm and >110 mm, and for white suckers the classes were ≤100 mm and >100 mm. One size class was adequate for shorthead redhorse suckers since few fish less than 300 mm were captured.

Catch patterns were tabulated for each size class for each species, and abundance estimates were calculated using the equations of Seber (1982) for triple removal pass population estimates (Junge and Libosvarsky 1965, Seber 1982). For sites and species size classes in which catch patterns produced unreliable abundance estimates (due to low catches or irregular depletion patterns) the average gear selectivity, or catchability, for that species size class, for all sites in all years, in all passes, was used to estimate the abundance, using the following formula: $[Y(1+(1-q)^3]$, where Y= the total catch over three passes and q= the average catchability or probability of an individual being captured during a sampling pass (Seber 1982). Abundance estimates for each length group for each species were then combined to produce an overall abundance estimate for that species, for each site in a given year. The abundance estimates were then converted to density estimates using sample site average width and length information. A one-way analysis of variance was used to test for significant year effects in fish densities, for each species in each of the three study zones. The Kolmogorov-Smirnov two sample test (Steel and Torrie 1980) was used to test for differences between fish length-frequency distributions between years and study zones, for each species.

Brown trout density estimates for several other Michigan trout streams, during the study period from 1997 – 2006, were estimated as part of ongoing monitoring by the Michigan Department of Natural Resources. These estimates were calculated using the Petersen mark-recapture estimation method (Seber 1982), utilizing electrofishing equipment. These data were analyzed and presented in this study as a secondary form of reference for changes in trout densities during the study period.

Morista's similarity index was used to compare the proportional numeric composition of the fish community between study zones (Morista 1959, Krebs 1989). An index value of 0.00 indicates complete dissimilarity, a value of 1.00 indicates complete similarity, and values greater than 0.60 are generally interpreted as "similar" (Angradi and Griffith 1990). The Shannon-Weaver diversity index (H')(Shannon and Weaver 1949, Ricklefs 1990) was used to estimate the fish species diversity in each study zone. This index considers both the number of species present and evenness of numerical proportions of each species, rewarding higher diversity values to species compositions not numerically dominated by a just a few of the species present.

RESULTS

Fish Habitat

From 1997 through 2005, substantial changes occurred to the fluvial geomorphology of the Pine River, due to the removal of Stronach Dam. Those changes are described in detail by Burroughs (2006), and are reviewed here briefly to provide context for understanding changes to the fish community mediated by habitat alteration following dam removal. In the reach immediately upstream of Stronach Dam, progressive erosion of the accumulated reservoir sediment fill occurred with each subsequent stage of dam removal. This erosion increased gradient throughout the entire former impoundment (~4 km), increased water velocities, and eroded large volumes of sediment as a new channel was carved downward through the reservoir sediment fill. This resulted in a narrower river channel, with steepening banks, increasing frequency of riffle and pool bedforms, and slightly increased median substrate size composition (Table 2). Large amounts of the reservoir sediment fill were transported downstream of the dam, eventually being deposited at the next impoundment downstream. Significant amounts of this sediment were also deposited in the river channel along the way and a smaller portion was deposited onto floodplains during high flow events. Sediment deposition downstream of the dam resulted in streambed aggradation and increases in stream width and decreased water depth. Gradient was increased downstream of the dam removal through sediment deposition, water velocities increased slightly and substrate size remained small (sand dominated). Bedform diversity did not increase in this reach during the study period, and remains almost entirely run bedforms (Table 2).

Fish Community

A total of 35 fish species were identified during sampling in the Pine River. Prior to dam removal, a coldwater fish community existed upstream of Stronach Dam, numerically dominated by slimy sculpin (*Cottus cognatus*), brown trout, rainbow trout, and white suckers (Appendix A). Downstream of Stronach Dam, a coolwater fish community existed, numerically dominated by white sucker and shorthead redhorse sucker, but also with frequent smallmouth bass, northern pike (*Esox lucius*) and brown trout. Most of these fish utilized the reservoir ~2 km

Table 2. A summary of key fish habitat changes that occurred during the Stronach Dam removal (1997 -2003). Fish passage past the dam site was not possible until 2003. Bedform frequencies were surveyed in 1995 and 2004.

Habitat Characteristic	**1996**	**2002**	**2006**
Wetted Stream Width (mean, m)			
Reference	17.0	17.0	16.9
Impoundment	19.9	17.4	17.6
Downstream	32.7	34.9	34.8
Gradient (% slope)			
Reference	0.159	0.157	0.155
Impoundment	0.128	0.181	0.206
Downstream	0.061	0.074	0.104
Water Velocity (mean, m/s)			
Reference	0.66	0.68	0.69
Impoundment	0.54	0.71	0.86
Downstream	0.15	0.63	0.57
Substrate Size (median, mm)			
Reference	35.6	39.0	25.2
Impoundment	6.4	8.5	6.4
Downstream	1.0	2.0	1.0
Bedform Frequency			
Reference	High diversity		no change
Impoundment	Run bedforms		more pools and riffles
Downstream	Run bedforms		no change

downstream of Stronach Dam, and were found in the downstream reach of the Pine River seasonally. Prior to the removal of Stronach Dam, 18 fish species were found only downstream of the dam, 14 species were found both upstream and downstream of the dam, and three species were found only upstream of the dam (Table 3).

Within three years after the dam removal (2006), 17 of the 18 species previously found only downstream of the dam had been found upstream of the dam site. Only one of the three species previously only found upstream of the dam, had been discovered downstream of the dam site. Prior to the removal of Stronach Dam, the fish community in the impoundment was intermediate in similarity between the reference and the downstream reaches while the reference and downstream reaches were highly dissimilar (Table 4). By 2006 all three zones became highly similar. This increase in fish community similarity was manifested through changes in the species compositions of all three study zones, resulting in a homogenization of species compositions among zones of the river (Table 4). However, species diversity also increased in all three zones following the dam removal, with all three study zones having higher species diversity than observed in any zone prior to the dam removal (Reference H' 1997 = 1.31, 2006 = 1.81; Impoundment H' 1997 = 1.55, 2006 = 1.99; Downstream H' 1997 = 1.63, 2006 = 2.25).

Table 3. Fish species occurrences in the Pine River, before and after the removal of Stronach Dam. Arrows and italics represent fish species that were not found both upstream and downstream of the dam prior to removal, but were found both upstream and downstream following the dam removal.

Downstream of Dam Only	**Upstream and Downstream of Dam**	**Upstream of Dam Only**
		Brook stickleback
		Banded killifish
	Blacknose dace	◀ Blacknose dace
Common carp		
Largemouth bass ▶	*Largemouth bass*	
Trout perch ▶	*Trout perch*	
Rock bass ▶	*Rock bass*	
Pumpkinseed ▶	*Pumpkinseed*	
Emerald shiner ▶	*Emerald shiner*	
Blackside darter ▶	*Blackside darter*	
Logperch ▶	*Logperch*	
Chestnut lamprey ▶	*Chestnut lamprey*	
Walleye ▶	*Walleye*	
Central mudminnow ▶	*Central mudminnow*	
Silver redhorse ▶	*Silver redhorse sucker*	
Shorthead redhorse ▶	*Shorthead redhorse sucker*	
Golden shiner ▶	*Golden shiner*	
Yellow bullhead ▶	*Yellow bullhead*	
Johnny darter ▶	*Johnny darter*	
Northern pike ▶	*Northern pike*	
Yellow perch ▶	*Yellow perch*	
	Common shiner	
	American brook lamprey	
	Longnose dace	
	Creek chub	
	Bluegill	
	Mottled sculpin	
	Slimy sculpin	
	White sucker	
	Brown trout	
	Rainbow trout	
	Brook trout	
	Black bullhead	
	Spottail shiner	
	Smallmouth bass	

Table 4. Morista similarity index values for each study zone before and after dam removal (1997 and 2006). Morista similarity index compares the proportional numerical species composition; 0.00 indicates complete dissimilarity, 1.00 indicates complete similarity, and values greater than 0.60 generally indicate "similar" fish species compositions.

	Reference 1997	Impoundment 1997	Downstream 1997	Reference 2006	Impoundment 2006
Reference 1997	-	-	-	-	-
Impoundment 1997	0.79	-	-	-	-
Downstream 1997	0.11	0.62	-	-	-
Reference 2006	0.70	0.55	0.64	-	-
Impoundment 2006	0.49	0.65	0.72	0.95	-
Downstream 2006	0.30	0.55	0.73	0.80	0.87

Fish Populations

At the beginning of the staged dam removal, brown trout density was low throughout all three study zones of the Pine River, and remained low for the first several years of the dam removal (1997 -1999: Reference mean = 46 fish/ha., st.dev. = 19; Impoundment mean = 33 fish/ha., st.dev. = 9; and Downstream mean = 14 fish/ha., st.dev. = 13). After the initial three years, brown trout density began to increase in both the reference and impoundment zones, with large differences in the magnitude of response among sites within zones (Figure 2). This increasing trend continued through the end of the study in 2006 (2004 - 2006: Reference mean = 129 fish/ha., st.dev. = 26; Impoundment mean = 157 fish/ha., st.dev. = 40; and Downstream mean = 48 fish/ha., st.dev. = 34). Brown trout density in the downstream zone remained relatively low throughout most of the study period, increasing only in the last several years of the study (Figure 2).

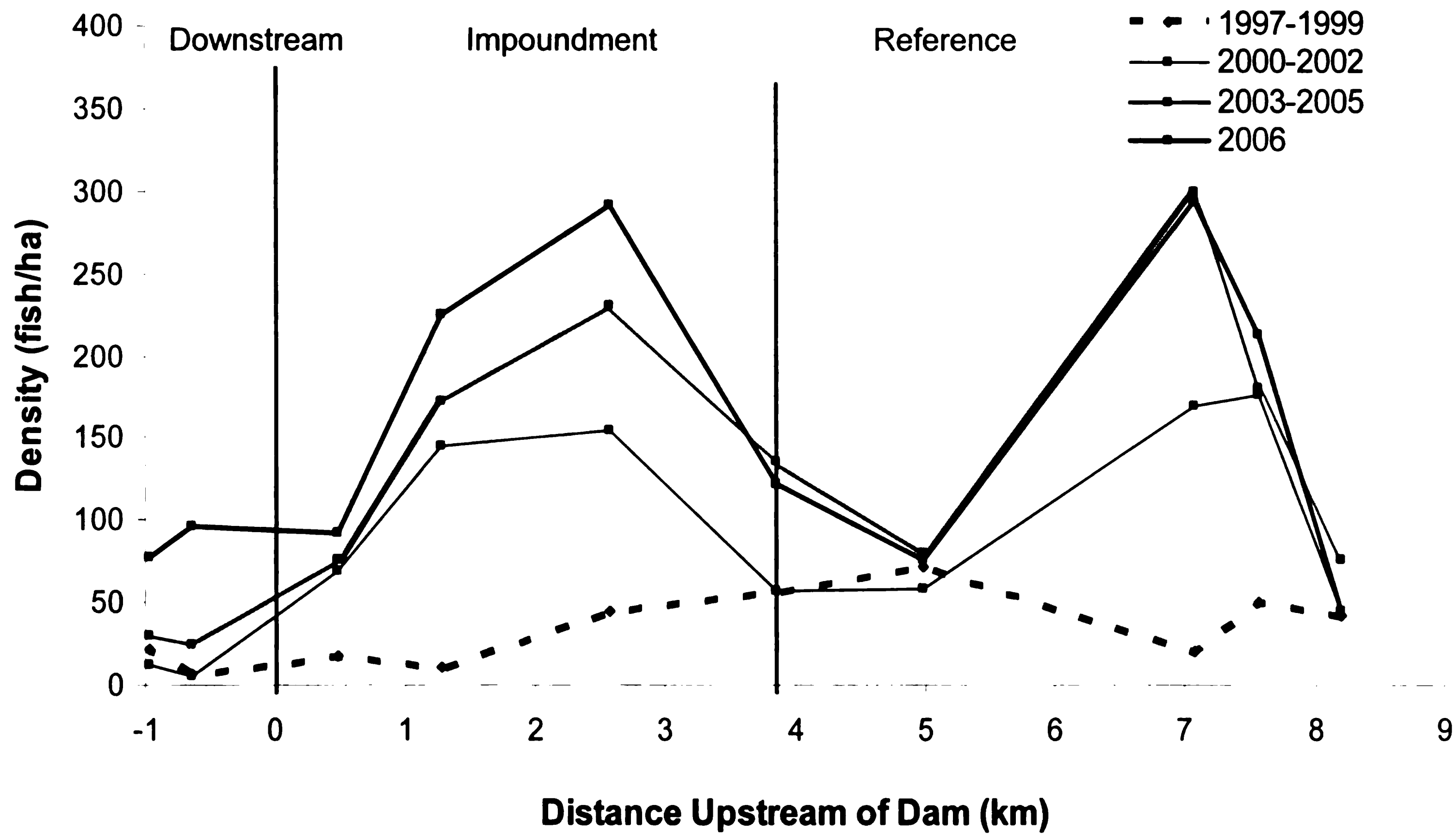


Figure 2. Brown trout density (number of fish per hectare), longitudinally in the Pine River. Densities averaged in three year blocks to minimize year to year variability, and for presentation simplicity.

While brown trout density increased in all three zones from the beginning of the dam removal to the end of the study period, one-way analysis of variance revealed a statistically significant Year effect for the Impoundment ($F = 4.00$, $p = 0.002$, $df = 39$) and the Downstream zones ($F = 3.70$, $p = 0.027$, $df = 19$), but not the Reference zone ($F = 1.82$, $p = 0.105$, $df = 39$).

Rainbow trout density was also low throughout all three study zones during the first several years of the dam removal (1997 – 1999: Reference mean = 49 fish/ha., st.dev. = 9; Impoundment mean = 24 fish/ha., st.dev. = 6; and Downstream mean = 1 fish/ha; st.dev. = 1). After the initial three years of the dam removal, rainbow trout densities increased in both the reference and impoundment zones, and continued to increase through the end of the study period (Figure 3). This increasing trend, while somewhat slower than for brown trout, also showed similar spatial variability between sites in the magnitude of density increases. In particular, two sites within the former impoundment, and within the reference increased greatly while other sites increased only slightly, or not at all (Figures 2 and 3). Rainbow trout density downstream of the dam removal, while increasing slightly, has remained relatively low throughout the entire study period. By the end of the study, average rainbow trout density in each of the zones was higher than at the beginning of the study (2004 - 2006: Reference mean = 128 fish/ha., st.dev. = 37; Impoundment mean = 107 fish/ha., st.dev. = 35; and Downstream mean = 8 fish/ha., st.dev. = 6), but a significant Year effect was found only for the impoundment zone ($F = 5.32$, $p = 0.0002$, $df =$

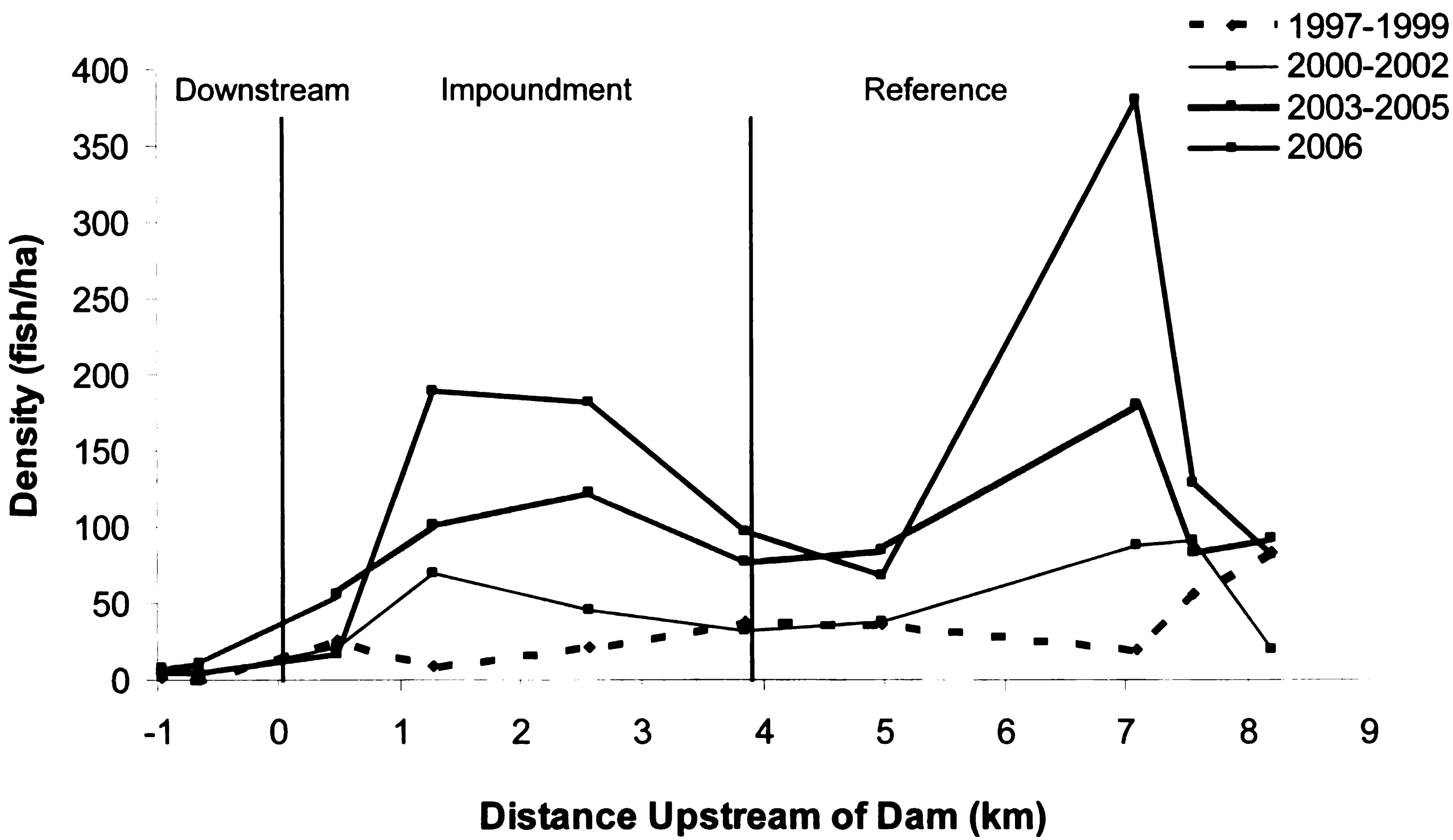


Figure 3. Rainbow trout density (number of fish per hectare), longitudinally in the Pine River. Densities averaged in three year blocks to minimize year to year variability, and for presentation simplicity.

39) (Reference: $F = 1.67$, $p = 0.14$, df = 39; Downstream: $F = 2.90$, $p = 0.056$, df = 19).

Because brown trout and rainbow trout densities increased in both the impoundment and reference study zones during and following the Stronach Dam removal, data on the brown trout density trends in other similar Michigan trout streams were acquired from the Michigan Department of Natural Resources. Rainbow trout density trends were not available for other Michigan trout streams, because the Pine River's rainbow trout population is uniquely non-migratory, whereas most other populations of rainbow trout in Michigan streams are migratory (i.e., "steelhead"). Brown trout density data for the other Michigan streams were derived from estimates at single sites on other streams. Density of brown trout in the Pine River was lower than sites on other streams for which data were available. Many of the other streams also experienced increases in brown trout density during this study period. The increase in density in the Pine River, expressed on an arithmetic scale, was not unique (Figure 4). However, when expressed as a rate of increase, Pine River brown trout density increases were substantially greater than the other trout streams, and this increase was greatest in the Pine River former impoundment reach (Figure 5).

Prior to the dam removal, size structure of brown trout did not differ significantly between the reference and the impoundment (K-S test, $D_{max} = 0.18$, $p > 0.05$, n = 78, 71). The downstream zone contained so few brown trout that annual estimates of the size structure were not reliable. Following dam removal, the abundance of brown trout of nearly all length classes increased in all three

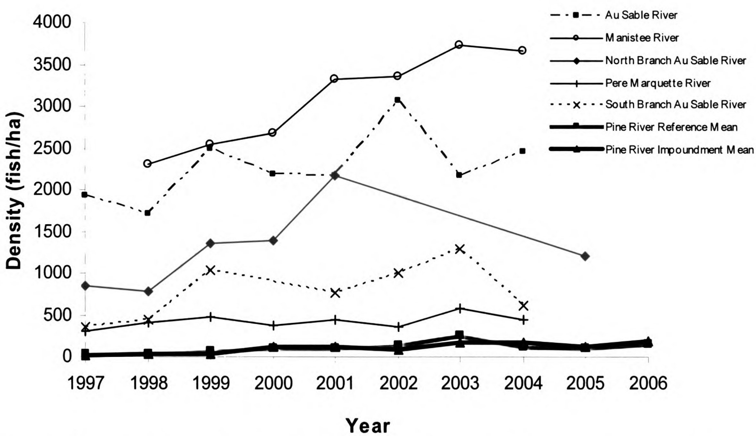


Figure 4. Brown trout densities (number of fish per hectare) in several Michigan trout streams and in the reference and impoundment study zones of the Pine River. Density was estimated for the other MI trout streams using mark-recapture estimation, while the Pine River estimates were derived from depletion sampling.

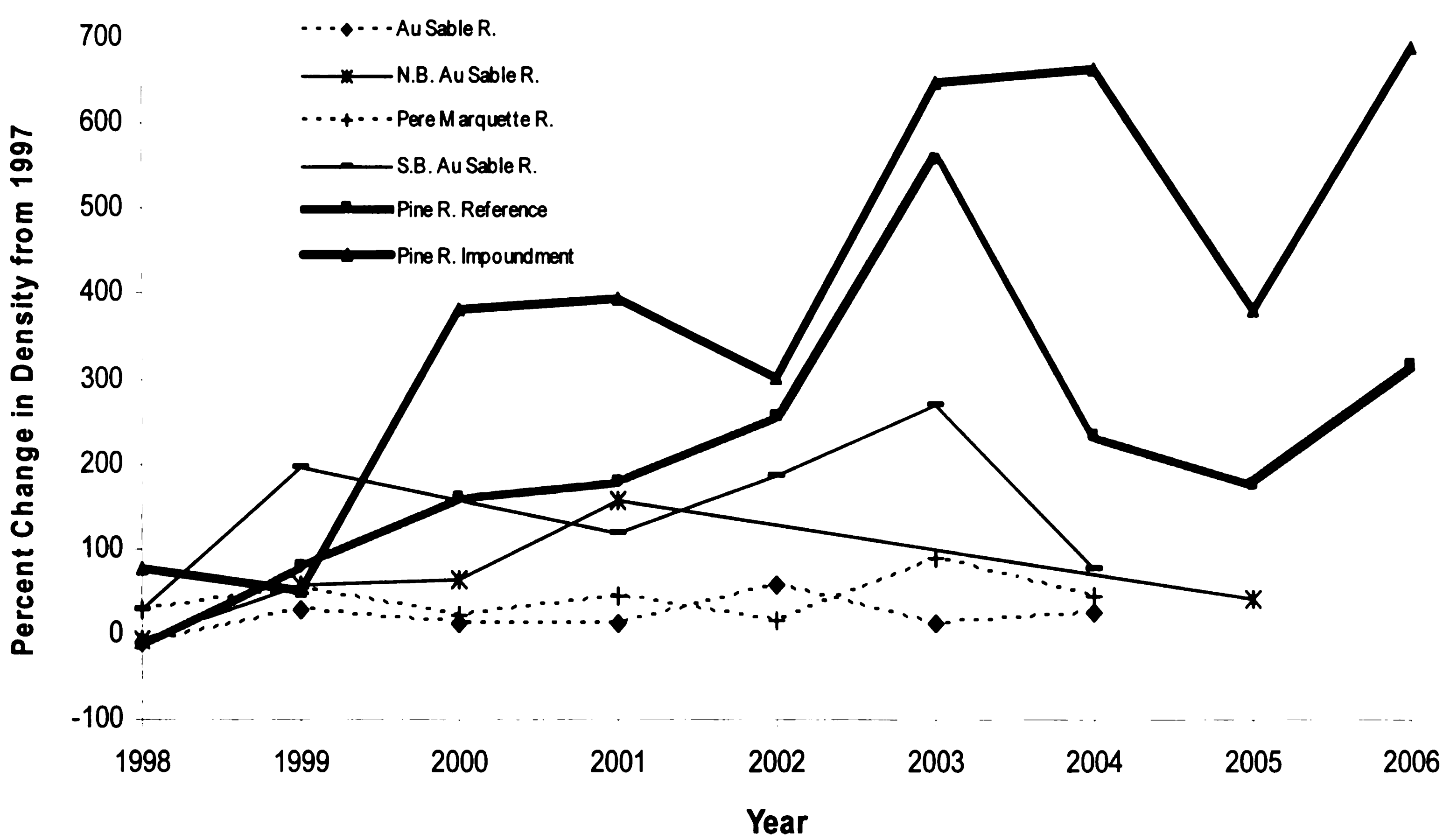


Figure 5. Population growth rate, or percent change in the density (number of fish per hectare) of Pine River brown trout, from 1997, compared to the percent changes in brown trout density of several other Michigan trout streams of comparable size.

study zones (Figure 6). The size structures of brown trout in the reference zone in 1997 and 2006 were not significantly different (K-S test, $D_{max} = 0.18$, $p > 0.05$, $n = 78, 143$), and neither were the size structures of brown trout in the impoundment zone in 1997 and 2006 (K-S test, $D_{max} = 0.15$, $p > 0.05$, $n = 71, 292$). Although the size structure of both zones did not change significantly over time, the difference in size structure between the reference and impoundment zones became significant by 2006 (K-S test, $D_{max} = 0.19$, $p < 0.01$, $n = 143, 292$), indicating a smaller proportion of individuals over 200 mm in length in the reference. Downstream of the dam, more brown trout were seen in 2006 than in 1997, and a significant change in size structure occurred (K-S test, $D_{max} = 0.56$, $p < 0.05$, $n = 8, 32$). Brown trout in this zone were still relatively low in number compared to the other study zones, but ranged in size from approximately 50 – 550 mm in length.

Rainbow trout in the reference and impoundment zones had significantly different size structures prior to dam removal (K-S test, $D_{max} = 0.49$, $p < 0.01$, $n =38, 51$), but both zones were characterized by having relatively few rainbow trout, and those present were mostly from 200 – 400 mm in length (Figure 7). The size structures of these zones were significantly different between 1997 and 2006, mainly due to much higher frequencies of rainbow trout <200 mm length (Reference: K-S test, $D_{max} = 0.49$, $p < 0.01$, $n = 38, 102$) (Impoundment: K-S test, $D_{max} = 0.58$, $p < 0.01$, $n = 51, 200$). Very few rainbow trout were captured in the downstream zone in either 1997 or 2006.

Figure 6. Brown trout length frequency distributions, for each study zone of the Pine River, before (1997), during (2001), and after (2006) the removal of Stronach Dam.

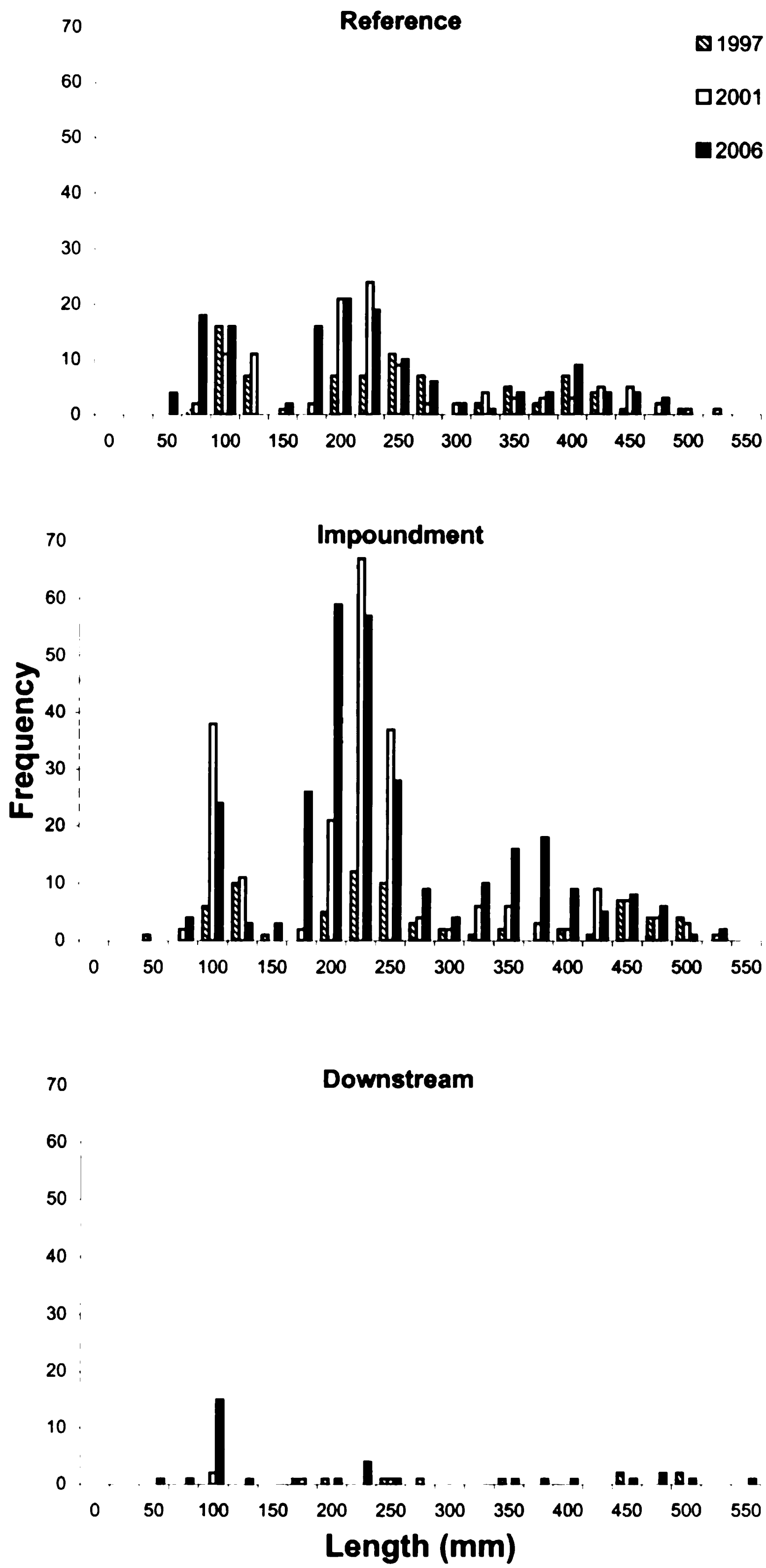
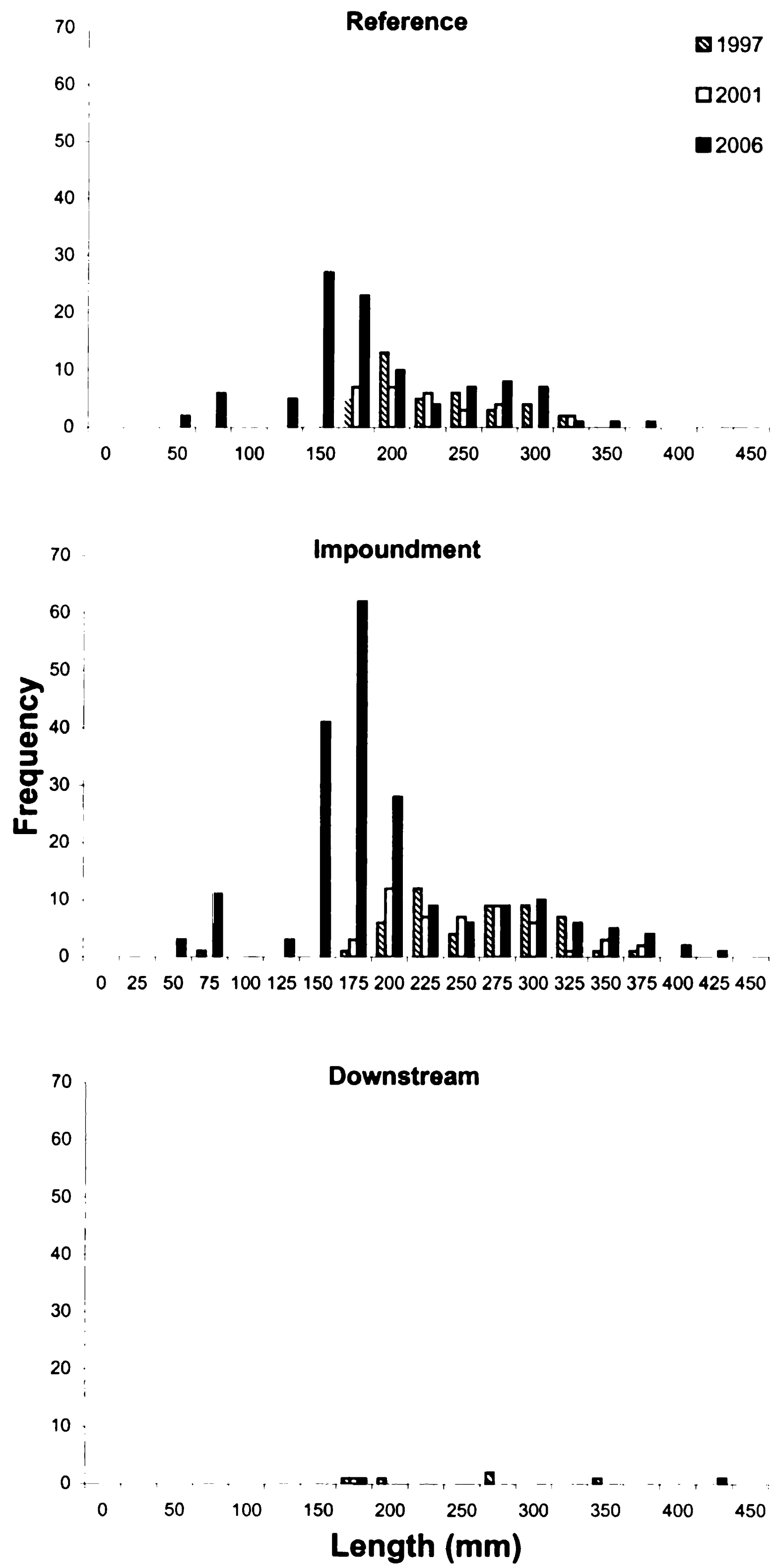


Figure 7. Rainbow trout length frequency distributions, for each of the study zones of the Pine River, before (1997), during (2001), and after (2006) the Stronach Dam removal.



Density of brook trout generally declined in the downstream direction through the study zone of the Pine River, both before and after the dam removal (Figure 8). Similar to the brown trout and rainbow trout, brook trout density was low in all three study zones of the Pine River during the initial stages of the dam removal (1997-1999: Reference mean = 40 fish/ha., st.dev. = 12; Impoundment mean = 11 fish/ha., st.dev.= 1; Downstream mean = 0 fish/ha., st.dev. = 0), but remained low throughout the entire study period (2004 – 2006: Reference mean = 20 fish/ha., st.dev. = 2; Impoundment mean = 7 fish/ha., st.dev. = 1; Downstream mean = 0 fish/ha., st.dev. = 0). While average brook trout density appeared to have decreased slightly within the two upstream zones, densities at individual sites within zones were highly variable and showed no consistent trends among sites within zones (Figure 8). The low density of brook trout restricted comparisons of the size structures of this species in the three study zones before and after dam removal.

Fish passage during the staged drawdown was limited until 2003. Before that time, white sucker density was relatively high downstream of the dam, due to influxes of spawning adults from the reservoir located immediately downstream. Densities were consistently much lower in both the impoundment and reference zones upstream of the dam (Figure 9) (1997-1999: Reference mean = 19 fish/ha., st.dev. = 12; Impoundment mean = 41 fish/ha., st.dev. = 36; Downstream mean = 134 fish/ha., st.dev. = 155). After 2003, spawning white suckers from the downstream reservoir, previously prevented from accessing potential spawning habitats upstream of Stronach Dam, were allowed access

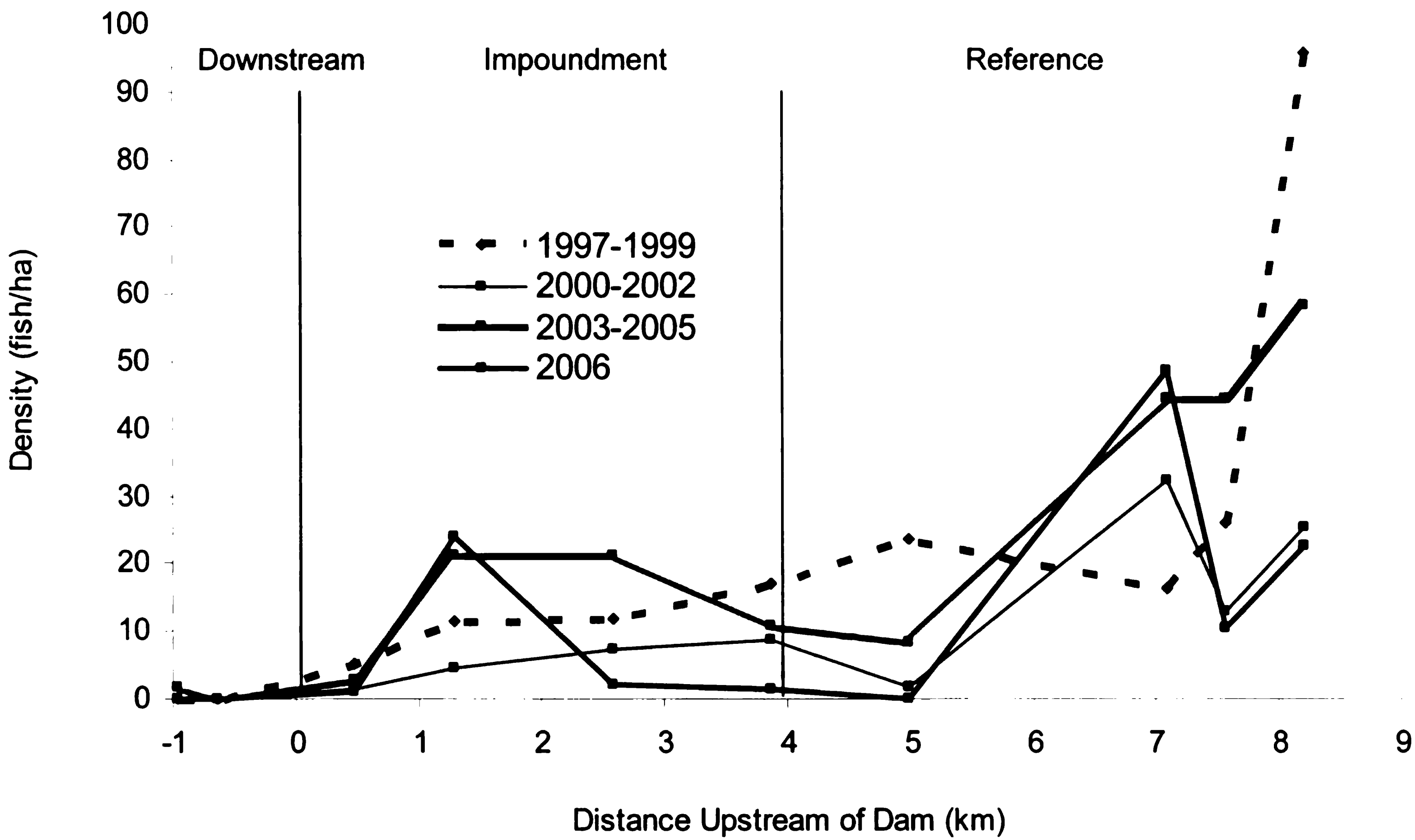


Figure 8. Brook trout density (number of fish per hectare), longitudinally in the Pine River. Densities averaged in three year blocks to minimize year to year variability, and for presentation simplicity.

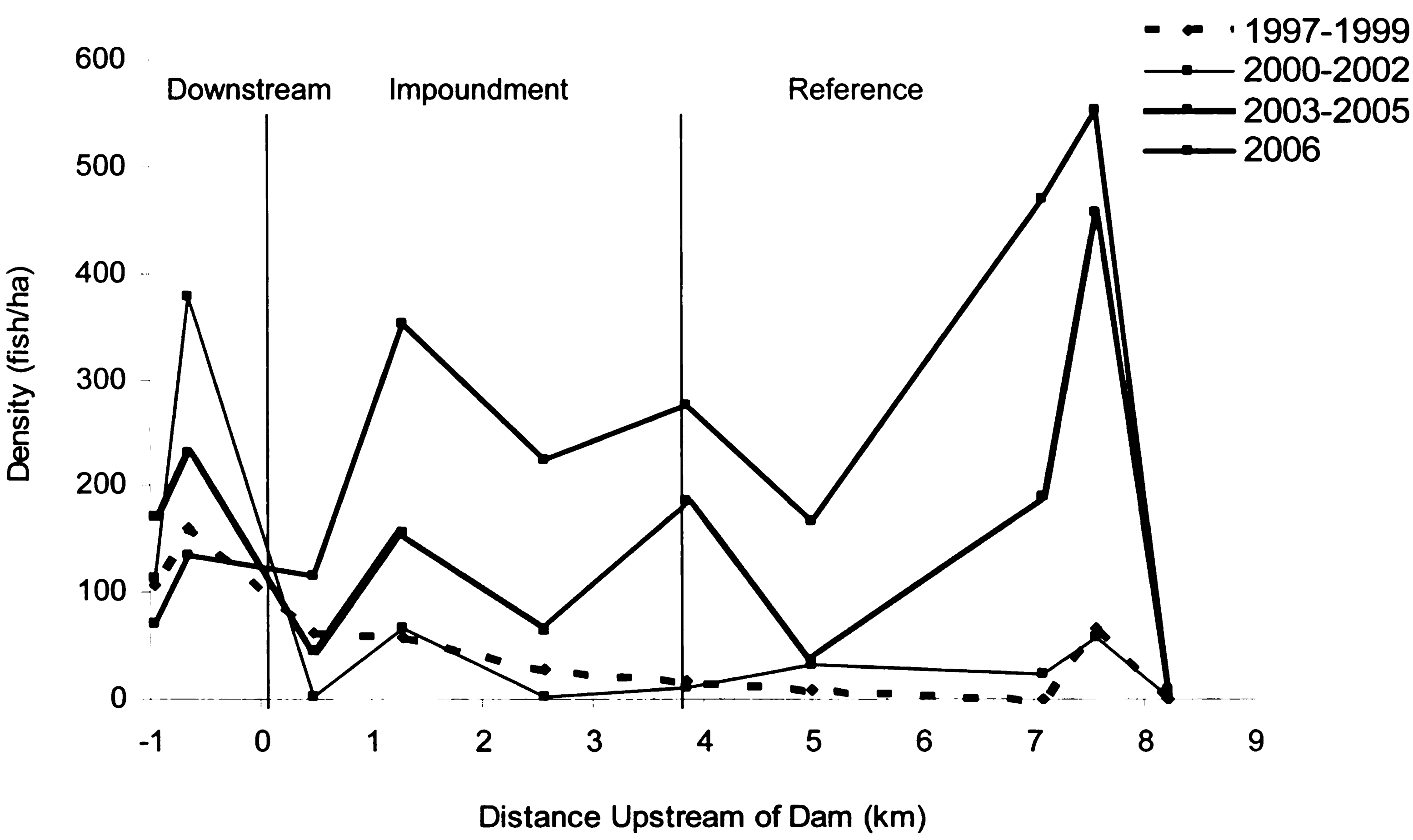
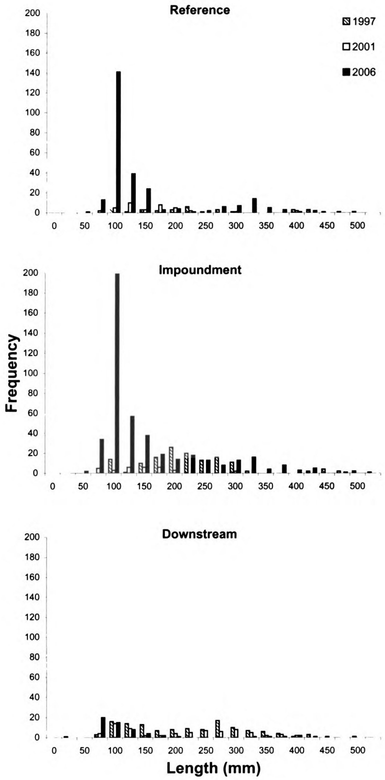


Figure 9. White sucker density (number of fish per hectare), longitudinally in the Pine River. Densities averaged in three year blocks to minimize year to year variability, and for presentation simplicity.

due to the dam removal. Consequently, density of white suckers increased substantially in both the impoundment and reference zones, while remaining relatively high in the downstream zone (Figure 9) (2004 - 2006: Reference mean = 240 fish/ha., st.dev. = 52; Impoundment mean = 174 fish/ha., st.dev. = 65; Downstream mean = 161 fish/ha., st.dev. = 53). Large changes in the size structure of white sucker in the Pine River were also observed. Prior to the dam removal, the downstream zone had a relatively uniform size distribution from ~100 – 500 mm, the impoundment had relatively low frequencies of white suckers of intermediate lengths (~100 – 350 mm), and the reference had few white suckers of any length (Figure 10). Following dam removal, the size structure downstream of the dam changed significantly (1997 vs 2006) (K-S test, $D_{max} = 0.50$, $p < 0.01$, n = 128, 56) and only contained individuals from ~100 – 200 mm length. Upstream of the dam removal, size structures also changed significantly, through the addition of large frequencies of juvenile white suckers (~75 – 200 mm) (Reference: K-S test, $D_{max} = 0.58$, $p < 0.01$, n = 33, 266) (Impoundment: K-S test, $D_{max} = 0.55$, $p < 0.01$, n = 140, 452).

Prior to 2003 shorthead redhorse suckers were found only downstream of the dam. The density of this species downstream from the dam was variable during the study period (Figure 11). From 2003 through the end of the study in 2006, shorthead redhorse suckers were found in relatively low densities throughout both the impoundment and reference zones (Figure 11). The few individuals that have been sampled upstream of the dam were greater than 200 mm in length. Downstream of the dam, this species is more abundant than

Figure 10. White sucker length frequency distributions for each study zone of the Pine River, before (1997), during (2001), and after (2006) the Stronach Dam Removal.



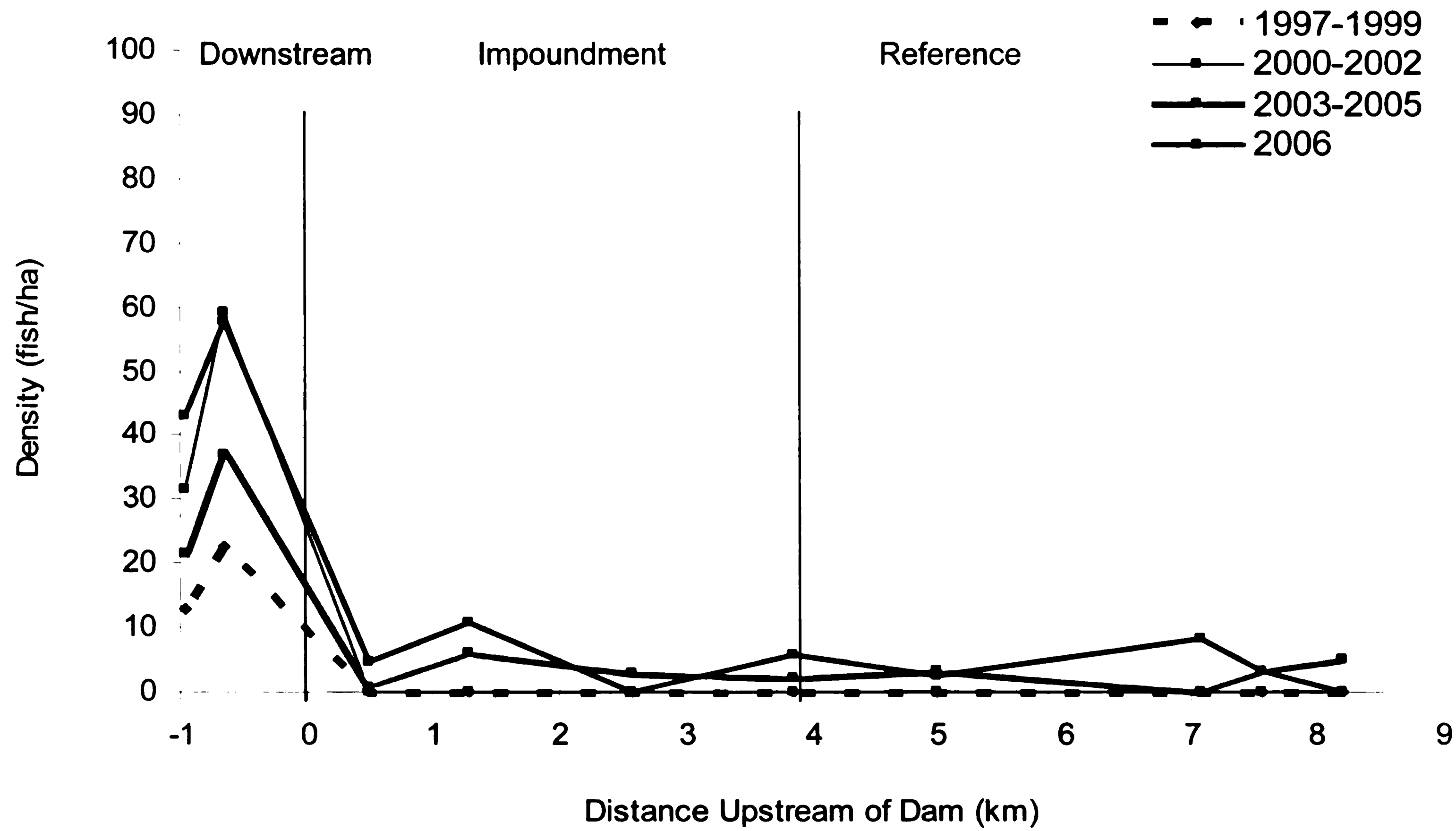


Figure 11. Shorthead redhorse sucker density (number of fish per hectare), longitudinally in the Pine River. Densities averaged in three year blocks to minimize year to year variability, and for presentation simplicity.

upstream, but still relatively low in abundance. Here, the size structure changed significantly (K-S test, $D_{max} = 0.63$, $p < 0.01$, $n = 28, 21$), with higher frequencies of larger fish than were present before the dam removal (Figure 12).

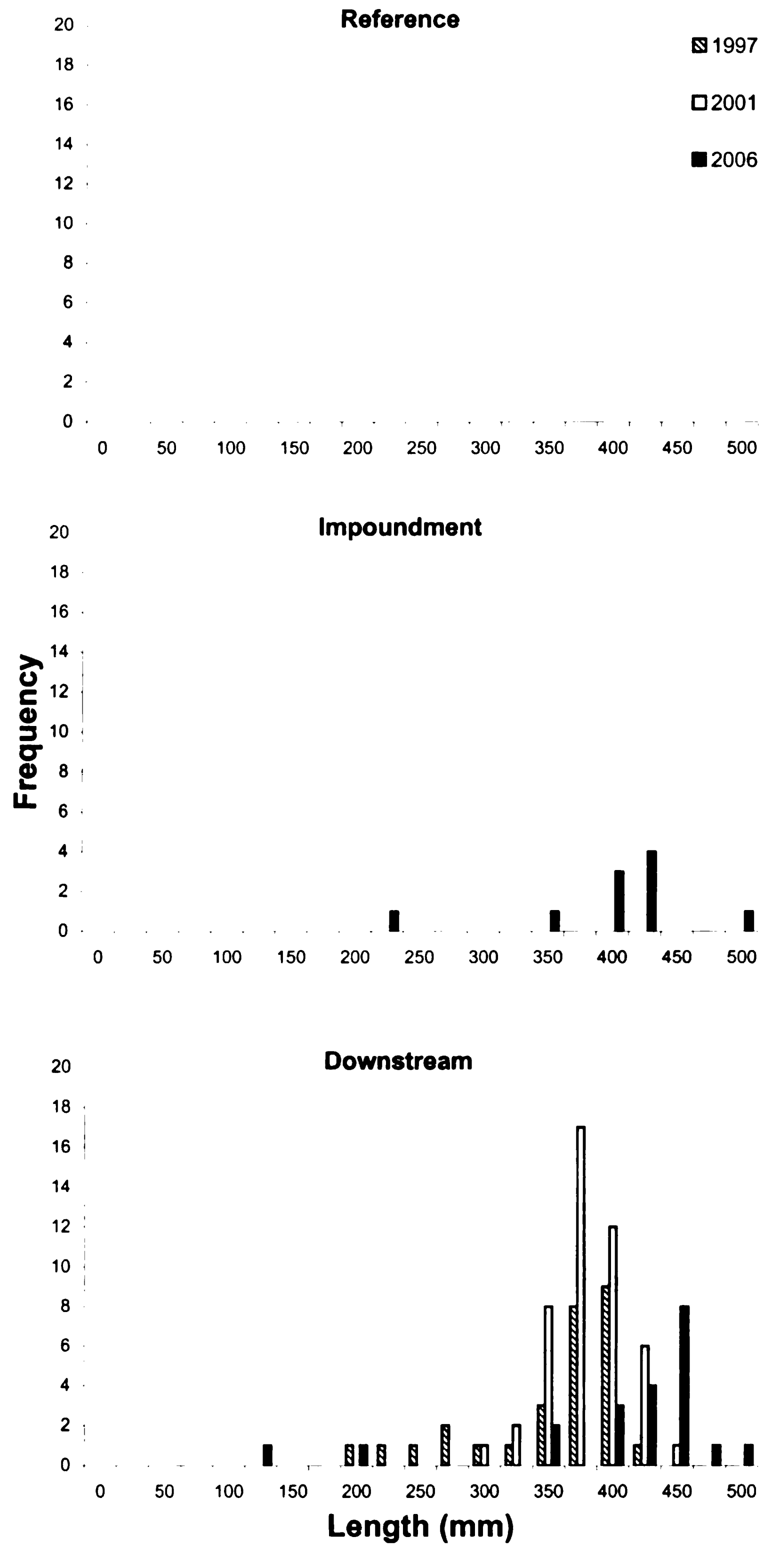
DISCUSSION

Fish Habitat

Fish habitat was altered greatly due to the dam removal. In the former impoundment, habitat quality generally improved for lotic fish species. Prior to the dam removal, this section of river was characterized by low gradient and wide, sand-dominated run bedforms. During the removal process substantial amounts of sediment erosion occurred leading to drastic changes in the habitat characteristics of this section of river. Gradient increased substantially, leading to faster and more diverse water velocities, narrower stream width, substrate coarsening, and a higher diversity of bedforms (i.e., more riffles and pools). While these changes represent significant improvements in the heterogeneity and quality of lotic habitat, this section of stream was not restored to habitat condition levels seen in the reference reach of the Pine River.

Downstream of the dam removal, habitat quality for most lotic species was generally degraded during the dam removal. This section of river received large amounts of sediment deposition and transport due to the dam removal. This created unstable and shifting fine substrates and eliminated deeper water habitats. This section of river was characterized as overly wide and shallow, sand-dominated, run bedform habitat throughout the duration of this study. While

Figure 12. Shorthead redhorse sucker length frequency distributions for each study zone of the Pine River, before (1997), during (2001), and after (2006) the Stronach Dam removal.



some of these impacts to the downstream section of river may be transient and reversed after sediment erosion in the former impoundment ceases, those changes were not observed during this study. One incidental benefit to the fish community downstream of the dam removal, may occur due to the streambed aggradation that occurred. With a streambed higher in elevation, overbank flooding requires less discharge to occur and happens more frequently, recharging adjacent floodplain wetlands and ponds. Fish species utilizing these habitats may benefit from this, but those habitats were not directly sampled in this study.

Fish Community

Prior to the removal of Stronach Dam, the three study zones had distinctive fish communities. The species composition of the impoundment was intermediate to both the reference and the downstream, but the reference and downstream zones were highly dissimilar. These differences were the result of both habitat differences between the zones, and the effects of the dam on connectivity between the zones. The reference and impoundment differed in habitat conditions, but possessed connectivity that allowed fish to move freely between the two zones. The impoundment and downstream zones had similar habitat conditions, but only possessed limited connectivity in the downstream direction. The reference and downstream zones possessed neither habitat similarities nor significant connectivity (Figure 13).

The dam removal resulted in habitat changes to the impoundment and downstream zones, and restored connectivity between all three zones. This led to changes in the fish species compositions of all three study zones. Nearly all of the fish species (17 of 18) found only downstream of the dam prior to removal were found upstream of the dam following its removal. However, many of these species remained in low abundance upstream of the dam due to differences in their habitat preferences and the habitat characteristics of the upstream zones. Only one of three species of fish found only upstream of the dam prior to its removal was found downstream of the dam following its removal. These species were likely to have been found only upstream of the dam due to habitat differences upstream and downstream of the dam and not due to lack of connectivity. Following dam removal, the fish communities found within each of the three zones became similar, and the diversity of fish species within in each zone increased. In light of this, it is important to recognize that dam removal did not necessarily result in "restoration" of the fish community, but did increase fish diversity in each zone, while increasing homogenization of the fish community throughout the river.

Fish Populations

For the first three years of the study, brown trout densities in all three study zones of the Pine River remained relatively low and stable. Between 1999

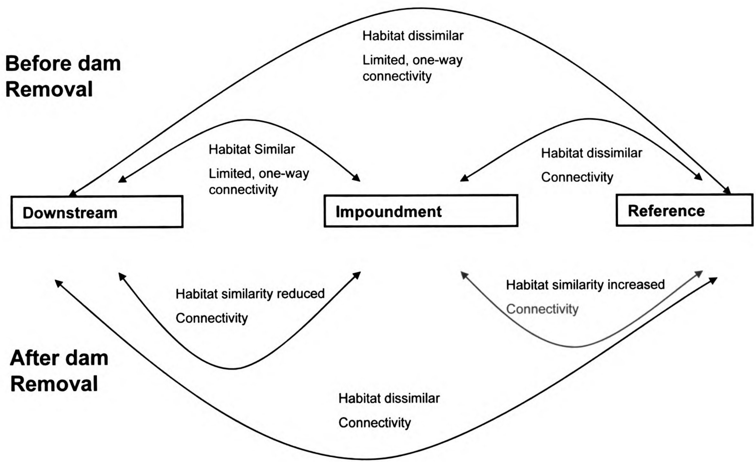


Figure 13. Conceptual diagram depicting the influences on fish species composition similarity between zones, before and after dam removal.

and 2000, substantial erosion and habitat changes were observed in the former impoundment and brown trout density began to increase. Brown trout density continued to increase, with variability, through 2006, when density was 450% higher than it was at the beginning of the dam removal. Examination of length-frequency distributions showed that recruitment increased significantly. As recruitment increased, the abundance of each size class increased, and by the end of the study all size classes were more abundant, but the shape of the length-frequency distribution was the same as before the dam removal.

Given that changes were also observed in the reference zone, it is possible that other factors also contributed to the observed increases. Three plausible hypotheses for the substantial increase in brown trout recruitment and density are: 1) favorable environmental influences during the study period, 2) potential changes in trout harvest during the study period and, 3) the dam removal improved and increased spawning habitat for brown trout in the river. The first hypothesis is partially supported by the observation that brown trout density increased in both the impoundment and the reference reach, suggesting system-wide changes unrelated to the dam removal. Several pieces of evidence suggest this is not entirely responsible for the trout density increases. First, the reference reach was chosen to be immediately adjacent to the impoundment, because no effects of the dam on habitat conditions were seen in the reference reach. This proved to be a very effective reference reach for habitat conditions, as no changes in habitat conditions associated with the dam removal were seen in this zone. However, tagged and marked trout were seen to frequently move between the sites (Burroughs unpublished data). Also, at the beginning of the dam removal, trout density and size structure of each zone were similar. Together, this suggests that the trout in each of the two zones were acting as one population due to their close proximity and connectivity, and as such, changes affecting trout in one of these study zones should also impact the trout in the other zone. Therefore, the large increases in recruitment and density of trout in the former impoundment could also be expected have contributed to increased density of trout in the reference zone. In future studies on the effects of dam removals, we therefore suggest that additional fish reference areas be located further from the influence of the dam removal, possibly including other independent reference streams.

Following the removal of the Woolen Mills Dam in Wisconsin, Kanehl et al. (1997) also observed significant increases in smallmouth bass density in both the former impoundment and the adjacent reference reach, but not in another nearby reference stream. We examined the trout dynamics in other similar Michigan trout streams, where data were available as part of ongoing population monitoring by the Michigan Department of Natural Resources. These data suggested that abundance of brown trout generally increased during much of the study period in many other Michigan streams. However, the population growth rate observed rate in the Pine River was substantially greater than all other populations. Therefore, environmental factors may have contributed to the observed response, but the increases are likely to be at least partially due to the dam removal.

Starting in the spring of 2000, trout harvest regulations were altered on many Michigan trout streams, including the portion of the Pine River encompassing the study area. From the beginning of the study through 1999, there was a 203 mm (8") minimum length and 10 fish per day creel limit on all three species of trout. In the spring of 2000, the regulations were changed to 5 fish per day, 203 mm minimum length, with no more than three over 381 mm (15") in length. In 2001, the regulations for trout harvest were changed to 254 mm (10") minimum length for brook trout, 305 mm (12") minimum length for brown trout and rainbow trout, and 5 fish per day creel limit with no more than 3 fish over 381 mm in length. These regulation changes may have increased the survival, and subsequently the abundance, of larger sized brown trout in the study zones of the Pine River. However, if this had occurred, a significant shift in the proportion of fish over the regulatory minimum lengths would have been expected in the length frequency distributions. This was not observed, however trout harvest in the Pine River is thought to be low compared to other local rivers (M. Tonello, Michigan Department of Natural Resources, personal communications), thus the influence of harvest in explaining the trout density increases in the Pine River is doubtful.

Habitat conditions in the former impoundment changed significantly during the dam removal, and those changes were generally favorable to the spawning requirements of stream trout. Perhaps most influential were the observed increases in water velocity, substrate size, and the frequency of riffle bedforms. Average water velocity, the diversity of water velocities, and the occurrence of faster water velocities all increased. Additionally, median substrate size increased from “small gravel” to “medium gravel”, and the diversity of substrate sizes increased due to increased proportions of large substrates, and less sand. As an example, most brown trout redds contain substrate between 18 – 30 mm, and larger substrate (>50 mm) is commonly selected for (Grost et al 1991), which was rare in the impoundment zone prior to dam removal, but significantly increased in abundance following the removal (Burroughs 2007). Additionally, this coarse substrate also provides cover for trout fry by offering shelter from high water velocities (Heggenes 1988). Moreover, Zimmer and Power (2006) found that brown trout favored riffles over pools, over runs for redd construction; and the frequencies of riffles and pools increased in the former impoundment section. These changes in water velocity, substrate coarseness, and bedform frequencies likely improved spawning conditions for trout in the impoundment zone, and provide evidence for explaining the increased recruitment of brown trout in the Pine River.

Rainbow trout density showed a remarkably similar pattern to that of brown trout. However, the steady increase in density of rainbow trout did not begin until almost 2003. At the end of the study in 2006, rainbow trout density had increased to 300% of the density observed at the beginning of the dam removal. Analysis of the size structure of the rainbow trout in both the impoundment and reference zones indicated that recruitment and the proportion of juvenile rainbow trout increased substantially, with the frequency of larger rainbow trout increasing only slightly. Rainbow trout prefer spawning substrate between 15 – 60 mm (Raleigh and Hickman 1984), which was rare in the impoundment zone prior to dam removal, but significantly increased following the removal. This is likely to have improved spawning conditions for rainbow trout and contributed to improved recruitment. The lack of proportional increases in the abundance of larger adult rainbow trout could result from the relatively recent increases in rainbow trout density and recruitment, with those effects not carrying through to the older age groups yet.

While average brook trout densities for both the reference and impoundment zones seemed to decline from the beginning to the end of the study, this could be attributed to factors other than the dam removal. Generally, in rivers with coexisting populations of brook trout, brown trout, and rainbow trout, upstream areas are typically characterized by brook trout, while brown and rainbow trout are found more often downstream (Vincent and Miller 1969, Gard and Seegrist 1972, Magoulick and Wilzbach 1997). This pattern is thought to stem from differences in competitive abilities (Rose 1986, Lohr and West 1992), or the adaptation to and selection of different environmental conditions (Cunjak and Green 1983). For example, where habitat is sub-optimal for brook trout, brown trout have been shown to exclude brook trout from preferred resting positions (Fausch and White 1981). Also, there is evidence that rainbow trout dominance over brook trout can result from reduced brook trout fecundity or year class failures giving rainbow trout a competitive advantage (Clark and Rose 1997). It is possible that this section of the Pine River was suboptimal habitat for brook trout, and the increased brown trout and rainbow trout densities eventually lead to the overall decreased density of brook trout.

White suckers were found upstream and downstream of the dam throughout the study period. However, the density of this species was relatively low and stable in both the reference and impoundment zones from 1997 – 2002, the period in which fish passage upstream past the dam was restricted. Downstream of the dam, white sucker density was variable, but consistently higher than seen upstream of the dam, due to the influx of spawning adults from the reservoir located immediately downstream. These spawning adults were prevented from accessing upstream habitats that might have been used for spawning. In 2003, the last phase of the dam removal was completed, and fish passage past the dam site was possible. Spawning adult white suckers then moved upstream in the Pine River, and length composition of white suckers downstream of the dam was limited to small individuals. Those adult spawning fish utilized both the reference and the impoundment for spawning, and the overall density of this species upstream of the dam site increased approximately 550% from 1997 levels. While the abundance of juvenile fish increased tremendously, similarly low frequencies of adult fish were seen in both of these upstream zones in 1997 and 2006. This suggests that white suckers in the reference and impoundment zones may be limited by adult habitat availability. The dam removal allowed this species to access habitats available in the river system that are beneficial to different life stages and resulted in higher productivity and abundance of this species, through the use of available spawning habitat. A similar type of response may also be expected from many other fish species that make spawning migrations in streams, but have been prevented from accessing suitable spawning habitat.

Prior to dam removal, shorthead redhorse suckers were found only downstream of Stronach Dam. Density levels of this species downstream of the dam were also influenced by the influxes of spawning adults from the downstream reservoir. In the spring, this species migrates out of large bodies of water into smaller rivers or streams to spawn (Scott and Crossman 1973). Meyer (1962) found that in Iowa, shorthead redhorse suckers became sexually mature at age 3, corresponding to approximately 300 mm in length. In the Downstream zone of the Pine River, shorthead redhorse suckers less than 300 mm in length were rarely sampled. After 2003, this species was found widely distributed throughout both the reference and impoundment zones, but in very low densities. Juveniles of this species were not sampled in the Pine River, as with the case of white suckers, suggesting that the lotic habitat of the Pine River is not the preferred habitat of juvenile shorthead redhorse suckers. This species is likely not unique in that dam removal allowed them to access habitats that may benefit certain life history stages, but the lotic habitat of the Pine River is not likely preferred for a majority of the life history, and thus this species was not found to substantially increase in density in the lotic habitat, following dam removal. Many of the other 17 species of fish found only downstream of the dam prior to removal, may benefit from the dam removal in a similar way (e.g., northern pike, trout perch (*Percopsis omiscomaycus*), walleye (*Stizostedion vitreum*)).

Synthesis

Removal of the barrier to fish migration alleviated many of the impacts of habitat fragmentation. As fish in the Pine River were allowed to freely move between areas of the river, species diversity both upstream and downstream of the dam increased. Productivity of the fish community also increased as fish were able to choose and access those habitats that best fulfill their life history requirements.

Dams alter the habitat for lotic fish in streams. In the former impoundment zone of the Pine River we observed habitat conditions improve for brown and rainbow trout and documented improved reproductive success and significant increases in the density of these important sport fishes. With the dam removal, habitat conditions in the downstream zone worsened through the deposition and transport of large quantities of fine sediment. Fish populations in this zone did not benefit as observed upstream of the dam removal. However, three years after the dam removal was completely finished, habitat conditions were still changing. While conditions improved significantly in the former impoundment, they were not restored to reference levels. The last year of this study, 2006, was the first year in which no new net erosion occurred in the former impoundment (Burroughs 2007). In the future, without this influx of sediment, habitat conditions in the downstream zone may also begin to improve. The extent to which these habitat characteristics will be restored to reference levels, and the timeframes needed to realize these benefits of dam removal are still uncertain, but the potential for dam removal to be a useful tool for improving riverine fish communities appears strong.

APPENDIX

Appendix A. Number of fish caught and their numeric proportions, for each study zone in the Pine River, before and after the Stronach Dam removal.

			Reference Zone				Impoundment Zone				Downstream Zone			
			1997		2006		1997		2006		1997		2006	
Fish Species Common Name	Genus	species	catch	%	catch	%	catch	%	catch	%	catch	%	catch	%
White Sucker	*Catostomus*	*commersoni*	33	5.39	266	28.85	140	30.50	452	30.54	128	58.18	56	28.14
Slimy Sculpin	*Cottus*	*cognatus*	368	60.13	258	27.98	157	34.20	217	14.66	0	0.00	15	7.54
Brown Trout	*Salmo*	*trutta*	78	12.75	143	15.51	71	15.47	292	19.73	13	5.91	33	16.58
Rainbow Trout	*Oncorhynchus*	*mykiss*	38	6.21	102	11.06	51	11.11	200	13.51	5	2.27	2	1.01
Logperch Darter	*Percina*	*caprodes*	0	0.00	85	9.22	0	0.00	126	8.51	4	1.82	38	19.10
Brook Trout	*Salvelinus*	*fontinalis*	68	11.11	10	1.08	31	6.75	12	0.81	0	0.00	0	0.00
Longnose Dace	*Rhinichthys*	*cataractae*	23	3.76	17	1.84	3	0.65	57	3.85	1	0.45	1	0.50
Shorthead Redhorse	*Moxostoma*	*macrolepidotum*	0	0.00	3	0.33	0	0.00	10	0.68	28	12.73	21	10.55
Pumpkinseed	*Lepomis*	*gibbosus*	0	0.00	3	0.33	0	0.00	46	3.11	0	0.00	3	1.51
Troutperch	*Percopsis*	*omiscomaycus*	0	0.00	0	0.00	0	0.00	31	2.09	8	3.64	6	3.02
American Brook Lamprey	*Lampetra*	*appendix*	2	0.33	10	1.08	3	0.65	8	0.54	0	0.00	2	1.01
Bluegill	*Lepomis*	*macrochirus*	0	0.00	15	1.63	0	0.00	2	0.14	0	0.00	0	0.00
Northern Pike	*Esox*	*lucius*	0	0.00	0	0.00	0	0.00	4	0.27	6	2.73	7	3.52
Yellow Perch	*Perca*	*flavescens*	0	0.00	5	0.54	0	0.00	6	0.41	0	0.00	2	1.01
Silver Redhorse	*Moxostoma*	*anisurum*	0	0.00	0	0.00	0	0.00	0	0.00	8	3.64	1	0.50
Smallmouth Bass	*Micropterus*	*dolomieu*	0	0.00	0	0.00	0	0.00	0	0.00	7	3.18	2	1.01
Largemouth Bass	*Micropterus*	*salmoides*	0	0.00	0	0.00	0	0.00	5	0.34	0	0.00	2	1.01
Blackside Darter	*Percina*	*maculata*	0	0.00	2	0.22	0	0.00	1	0.07	1	0.45	1	0.50
Rock Bass	*Ambloplites*	*rupestris*	0	0.00	0	0.00	0	0.00	2	0.14	1	0.45	2	1.01
Black Bullhead	*Ameiurus*	*melas*	1	0.16	0	0.00	2	0.44	0	0.00	0	0.00	1	0.50
Chestnut Lamprey	*Ichthyomyzon*	*castaneus*	0	0.00	0	0.00	0	0.00	2	0.14	2	0.91	0	0.00
Common Shiner	*Luxilus*	*cornutus*	0	0.00	0	0.00	0	0.00	1	0.07	2	0.91	1	0.50
Spottail Shiner	*Notropis*	*hudsonius*	0	0.00	0	0.00	0	0.00	0	0.00	4	1.82	0	0.00
Creek Chub	*Semotilus*	*atromaculatus*	1	0.16	0	0.00	1	0.22	0	0.00	0	0.00	1	0.50
Emerald Shiner	*Notropis*	*atherinoides*	0	0.00	1	0.11	0	0.00	2	0.14	0	0.00	0	0.00
Walleye	*Stizostedion*	*vitreum*	0	0.00	0	0.00	0	0.00	1	0.07	2	0.91	0	0.00
Central Mudminnow	*Umbra*	*limi*	0	0.00	1	0.11	0	0.00	1	0.07	0	0.00	0	0.00
Johnny Darter	*Etheostoma*	*nigrum*	0	0.00	0	0.00	0	0.00	0	0.00	0	0.00	2	1.01
Blacknosed Dace	*Rhinichthys*	*atratulus*	0	0.00	0	0.00	0	0.00	2	0.14	0	0.00	0	0.00
Golden Shiner	*Notemigonis*	*crysoleucas*	0	0.00	1	0.11	0	0.00	0	0.00	0	0.00	0	0.00
Sum			**612**		**922**		**459**		**1480**		**220**		**199**	

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CHAPTER 3

THE REHABILITATION POTENTIAL OF DAM REMOVAL: TEMPORAL PERSPECTIVE FROM MICHIGAN'S PAST DAM REMOVALS

INTRODUCTION

Dam removal has become a popular technique for stream restoration in the United States. The optimism for this technique to achieve restoration of stream ecosystems is based on the knowledge that dams critically alter the form and functioning of rivers. Often, the removal of dams is assumed to restore the original form and functioning of river systems, ameliorating the negative impacts of the dam's existence. To date, the strength of this assumption has largely overshadowed the need for evidence of the benefits of dam removals, as the rate of removing dams continues to increase and little empirical evidence of the benefits of dam removal exists. If the practice of removing dams is to be justified as a tool for stream improvement, and advances made in dam removal methods, then knowledge of the outcomes of dam removals needs to be documented.

Currently, only a few studies exist that document the outcomes of dam removals. The existing studies have focused on changes in river morphology, fish, aquatic insects, and nutrients, and have documented that substantial changes occur immediately following a dam removal (Hill et al. 1994, Kanehl et al. 1997, Evans et al. 2000, Bushaw-Newton et al. 2002, Stanley et al. 2002, Chaplin 2003, Burroughs 2007a, Burroughs 2007b). However, all of these studies have examined the effects of dam removal over relatively short time periods following removal. The financial commitment needed to study a dam removal for several years prior to removal, during the removal phase, and for several years following a removal, can be large and limit the feasibility of evaluating long-term response (Doyle et al. 2002, Pizzuto 2002, Gordon et al. 2004).

Large changes in streams are expected to occur immediately following a dam removal. Many of these changes are expected to continue for long periods of time, acquiring knowledge about the eventual outcomes of dam removal and the time durations to achieve these outcomes, is a very challenging using conventional Before-After-Control-Impact (BACI) study design. Some of the important questions regarding dam removal may require decades to be answered. How long after a dam removal will sediment continue to be transported? Will the river channel in the formerly impounded area and downstream area ever resemble the original river channel? Will the dam removal be beneficial or detrimental for resident fishes? Knowledge of the eventual outcomes of dam removal, and the time required to achieve these outcomes is needed to develop realistic expectations and evaluate removal methods.

Dams have many different impacts to river systems. Upstream of dams, water is impounded, gradient is reduced, water velocity is slowed, river channels widen, and fine sediment is deposited (Petts 1980, Ward and Stanford 1989). Downstream of dams, temperature of water released is often warmer or colder than the normal river water temperature. Sediment supply downstream of dams is greatly reduced, often leading to channel degradation, a lowering of gradient and water velocity, narrowing of the river channel, and coarsening of the substrate (Williams and Wolman 1984, Cushman 1985, Ward and Stanford 1989, Ligon et al. 1995, Collier et al. 1996, Lessard 2000, Shields et al. 2000). Fish species respond in varying ways to the habitat alteration and fragmentation caused by dams (e.g., Holden 1979, Bain et al. 1988, Hayes et al. in press). Non-native species often flourish in modified habitats (Martinez et al. 1994) and overall species richness and diversity can decrease (Quinn and Kwak 2003). Native species often decrease in abundance due to loss of habitat, loss of access to habitat, or competition with non-native species (e.g., Benke 1990, Pringle et al. 2000).

In impoundments, sediment erosion occurs following dam removal, which increases gradient and water velocity, and leads to river channel narrowing, bank steepening, substrate coarsening, and some reformation of pools and riffles (Kanehl et al. 1997, Stanley et al. 2002, Burroughs 2007a). Downstream of dam removals, substantial amounts of sediment transport and deposition can occur, which increase gradient and water velocity, aggrade the streambed, increase stream width, decrease water depth and fill in pools and riffles (Burroughs 2007a). However, due to the relatively short periods of time post-dam removal that these changes have been studied, it is currently unknown how long they will take to reach an equilibrium, and what equilibrium conditions will be (Francisco 2004, Doyle et al. 2005).

Changes to river morphology, along with the opening of fish passage, will alter conditions for the fish community. Fish species will be able to access habitats not previously available to them, and interact with species they did not previously interact with. Changes in fish habitat conditions will alter the suitability of sections of the river to the benefit of some species and the detriment of others. With each stream possessing a unique fish species composition and unique habitat conditions, generalizations about the effects of dam removals on fish are difficult. Given this variability of fish communities among rivers, are there still some generalizations that will hold true for most dam removals? After a dam removal, a larger number of fish species will be able to access all habitats within a river system. Burroughs (2007a) suggests that even within a few years after dam removal, habitat conditions will become more diverse and heterogeneous due to dam removal. Fish communities have been observed to become more diverse and overall productivity increased (Burroughs 2007b).

The difficulty of answering many of the crucial questions regarding dam removal lies in the relatively long periods of time that many of these changes may take to occur. Gathering this information could take scientists decades if not longer, using BACI experimental designs. However, the practice of removing dams continues to grow in frequency each year (Pohl 2003) despite the lack of this important information. These dam removals may have detrimental effects, or at least could be conducted more efficiently and effectively if more knowledge of the effects of dam removal was available. In this study we attempt to answer some of the crucial questions surrounding the eventual outcomes of dam removal, by surveying several past dam removals in Michigan that occurred from 2 - 39 years ago.

The goal of this study was to provide insight into the extent of the changes following dam removal and the duration of time required to realize those changes. Specifically, the study objectives were to:

1) document differences in water temperature, slope (gradient), width/depth, sinuosity, substrate size composition, and bedform frequencies, between upstream reference reaches and former impoundment and downstream reaches of dam removals, across a temporal scale of approximately 0 - 40 years post dam removal.
2) document differences in fish community composition and fish abundance between the reference reaches and former impoundments and downstream reaches, across a temporal scale of approximately 0 - 40 years post-dam removal.

METHODS

Study Design

At each past dam removal site, attributes of the fluvial geomorphology and fish community were surveyed in the former impoundment and downstream of the dam removal site. These were then compared to the same attributes surveyed in an upstream reference reach, outside of the influence of the dam on stream fluvial geomorphology. This upstream reference reach served as a surrogate for what the river in the former impoundment and downstream reaches would look like if the dam and the dam removal had not occurred. As such, we assume this provides a target for rehabilitation of river form and functioning. Natural differences between these three river reaches, not attributable to the presence of the dam or the occurrence of the dam removal might be expected (e.g., many dams were built on high gradient sections of stream to maximize storage potential and energy generation, and the underlying impoundment reach would have been higher gradient than an upstream reference reach). However, the reference reach conditions provide reasonable benchmarks for assessing how much rehabilitation was achieved in the two zones impacted by the dam and dam removal. If rehabilitation occurred in the area impacted by the dam and its subsequent removal, attributes of fluvial geomorphology in these zones would be expected to closely approach those seen in the reference reach of the river (Figure 1). Fish community attributes of a reference reach however, could not be considered rehabilitation benchmarks since with restored fish passage following dam removal, the fish community in the reference reach would also change. In this situation, differences in fish community attributes can only be compared between the three zones to assess whether any of these zones consistently ranks lower than the others, indicating impairment.

A list of all known dam removals that have occurred in the state of Michigan was obtained from the Michigan Department of Natural Resources, Fisheries Division. From this list of past dam removals, candidate study sites were selected to attain as wide and uniform of a temporal distribution possible. Additionally, sites were eliminated as candidates if the removal date was

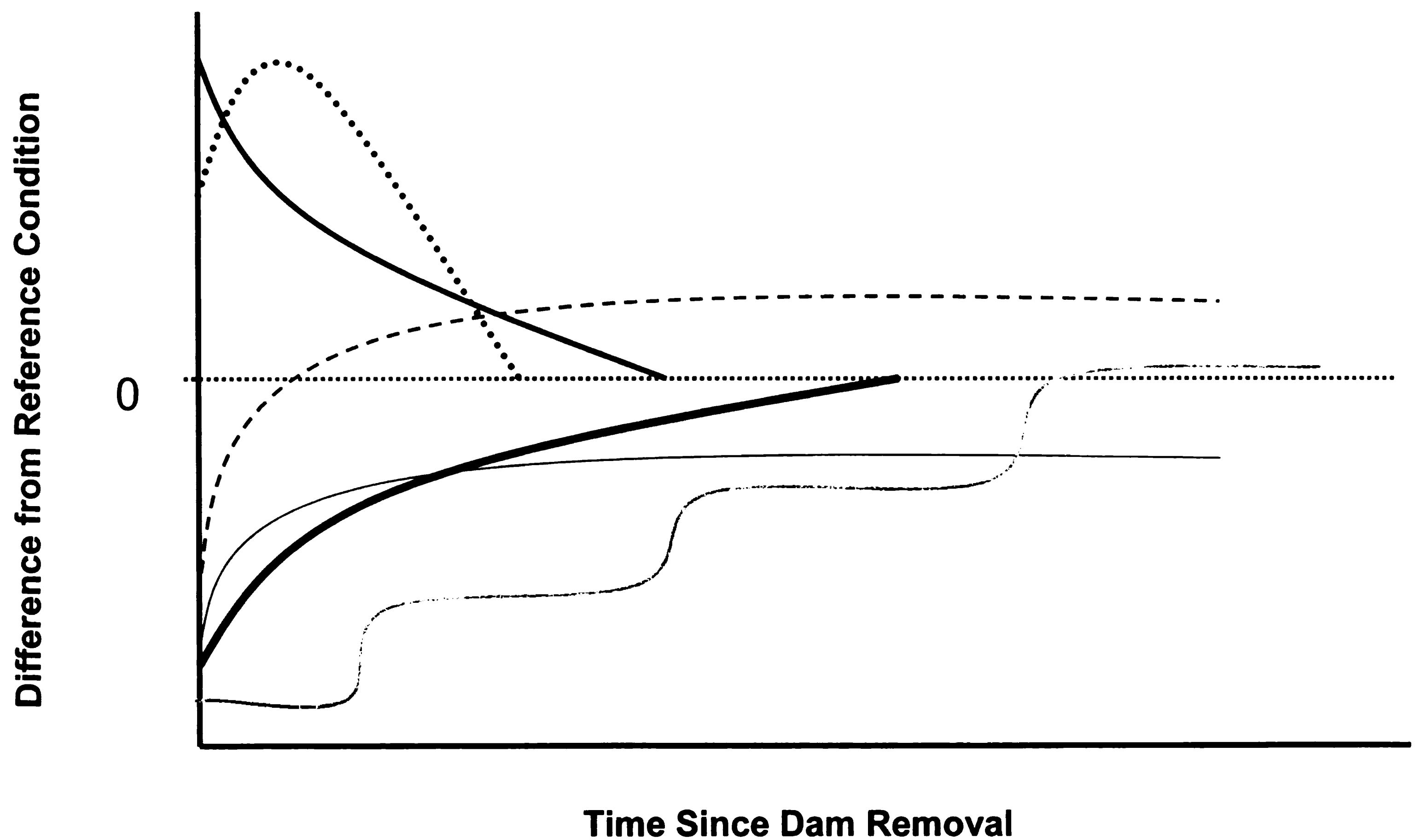

Figure 1. Hypothetical trajectories for rehabilitation of river attributes following dam removal.

unknown or uncertain. Scouting trips were made to many of the remaining candidate sites, and final study site selections were made based on whether or not the former dam site could be located with certainty, logistic considerations, and the value of the dam removal to the temporal distribution of past dam removals (Table 1 Figure 2). Historic aerial photos were used to help identify former dam sites and the upstream extent of the historic impoundments. These photos were used in combination with the Delorme® digital topographic map program to determine latitude and longitude coordinates for pertinent landmarks, and to select study sites prior to sampling, to maximize the spatial distribution within each study zone and eliminate site selection bias.

Table 1. Past dam removal study site information.

Dam Name	Randall	Stronach	L'Anse	Salling	Smyrna	Wacousta
River Name	Coldwater	Pine	Falls	AuSable	Flat	Looking Glass
County	Branch	Manistee	Baraga	Crawford	Ionia	Clinton
Year Removed	2002	1997 - 2003	1998	1991	1973	1966
Dam Height	3.7 m	3.7 m	?	5.2 m	?	1.4 m
Constructed	1912	1912	?	1914	1908	1860

Each study stream was broken into three study zones; the former impoundment, the upstream reference, and a zone downstream of the dam site. The former impoundment was delineated through the use of historic aerial photos and maps, and confirmed in more recent dam removal sites, through use of on-ground visual cues (former impoundments length range 318 – 5869m). The upstream reference was chosen to start immediately adjacent to the former impoundment and extend upstream for as long a distance as logistically feasible

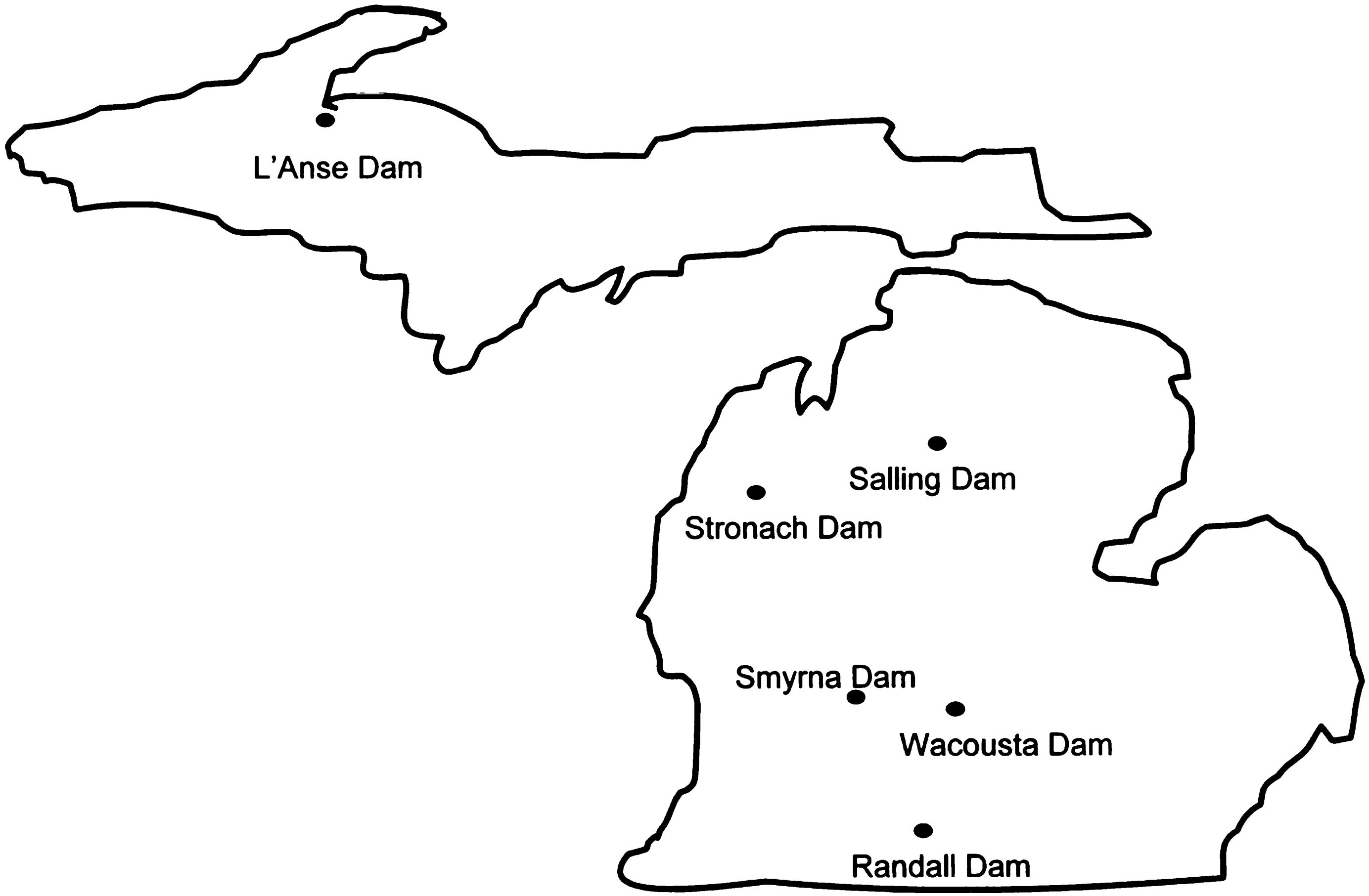

Figure 2. Location of study sites within the state of Michigan.

(references length range 1155 – 5177m), in which the river maintained similar characteristics (e.g., it would not extend upstream of the confluence of significant tributaries). The downstream study zones extended from the former dam site downstream for as long as logistically feasible (downstreams length range 624 – 3286m), while still maintaining similar characteristics (e.g., not extending downstream of significant tributaries, or into downstream impoundments).

Data Collection & Analysis

Bedforms (i.e., riffles, runs, and pools) were mapped throughout the entire study area of each river. Bedforms were delineated following the general guidelines of Hicks and Watson (1985), and were classified as runs, riffles, pools, rapids, waterfalls, or complexes (a designation where more than one bedform applied). Widths and length of each unit were measured using a Nikon® laser rangefinder (+/- 0.5m accuracy). Areas of rivers impacted by dams were expected to be relatively homogenous run bedforms compared to upstream reference conditions (Burroughs 2007a), and it was hypothesized that riffle and pool bedforms would be formed following dam removal. The diversity of bedforms in the former impoundment and downstream zones were compared to the diversity of bedforms in reference reaches. The Shannon-Weaver diversity index (H')(Shannon and Weaver 1949, Ricklefs 1990) was used to estimate the bedform diversity in each study zone. This index considers both the number of bedform types present and the evenness of the numerical proportions of each bedform, rewarding higher diversity values to study zones not numerically dominated by a just one bedform type (e.g., run).

Water temperature was recorded in each study zone using Onset® water temperature data recorders (HOBO Water Temp Pro, +/- 0.174° C). All water temperature recorders were placed at the downstream boundaries of each study zone (to reflect temperature effects of that zone), and recorded water temperature every 30 minutes, from July 29, 2005 through October 2, 2005.

Three sample sites, approximately 5-7 stream widths in length, were located within each study zone. River channel cross-sectional morphology was surveyed at both the upstream and downstream boundaries of each sample site, and the difference between the water surface elevations at each boundary cross-section was surveyed in order to calculate slope for the sample site. At each boundary cross section (n = 6 per zone), elevations were recorded to the nearest 3 mm, at 0.61 m intervals across the wetted width of the stream, and at points of inflection on the streambanks (appropriate for the accurate mapping of bank morphology), using a level and stadia rod. Width to depth ratio (W/D) was calculated for each cross section using the width and mean depth of the wetted channel.

Sinuosity of each study zone was estimated from aerial photos as the straight-line valley length of a section of river, divided by the actual length of the water flow through that section of river. River measurements were made with ARC MAP (by ESRI) software. Aerial photographs were compiled from the Michigan State University Remote Sensing and Geographic Information Science

(RSGIS) Research and Outreach Services aerial imagery archive. If needed, photos were rectified using a Michigan roads shapefile from the Michigan center of geographic information website. Shapefiles of points were added to represent the three stretches of the river being measured; each stretch of the river was measured three times and averaged. For many of the older dam removal study sites, aerial photos had been taken in several different years since the dams were removed. In these cases, sinuosity was repeatedly measured at several points in time since dam removal, for individual study sites.

Substrate size composition was measured at each cross-section using a modified pebble count method (Wolman 1954, Kondolf and Li 1992, Burroughs 2007a). systematically sampled 100 substrate particles, from water's edge to water's edge across the wetted width of the channel. The substrate particles were measured along the intermediate axis and assigned a size class code (from a modified Wentworth scale) (Wentworth 1922, Cummins 1962). The median substrate size (D50) was calculated for each cross-section and averaged for each study zone.

Thorough one-pass sampling of the fish community was conducted of all habitats within each sampling site. The sites were unblocked, and effort (time actively sampling) was recorded. Backpack, tow barge, and boat electrofishing equipment were used as appropriate for each river, but the equipment used was constant within each river. All fish sampled were netted, identified to species, and released (with the exception of "voucher specimens" that were retained for further identification in the laboratory). Total species richness (number of fish species) and relative numeric abundance of all fish species were calculated for each study zone. The similarity of fish communities in each study zone of a river was evaluated using the Morista similarity index (Morista 1959). This index takes into account both the differences in which species are present and differences in the numeric proportions of those species present. An index value of 0.00 indicates complete dissimilarity, a value of 1.00 indicates complete similarity, and values greater than 0.60 are generally interpreted as "similar" (Angradi and Griffith 1990).

One of our study rivers, the Pine River, was part of a long-term investigation, and provided comparable but more detailed data. The data collection methods utilized were identical to those reported for this study (Burroughs 2007a, Burroughs 2007b). Data were selected from that dataset to match those acquired at that other past dam removals in this study (e.g., one pass fish sampling, matched cross-section sites), for both fluvial geomorphology and fish community characteristics. Repeated samples for each characteristic were drawn from the Pine River dataset from before and after the dam removal.

RESULTS

Fluvial Geomorphology

Only small differences in water temperature were observed between study zones following dam removal (Figure 3). Water temperatures in both the former impoundment zones and downstream zones showed very little deviation from water temperatures seen in the upstream references. This was observed for all

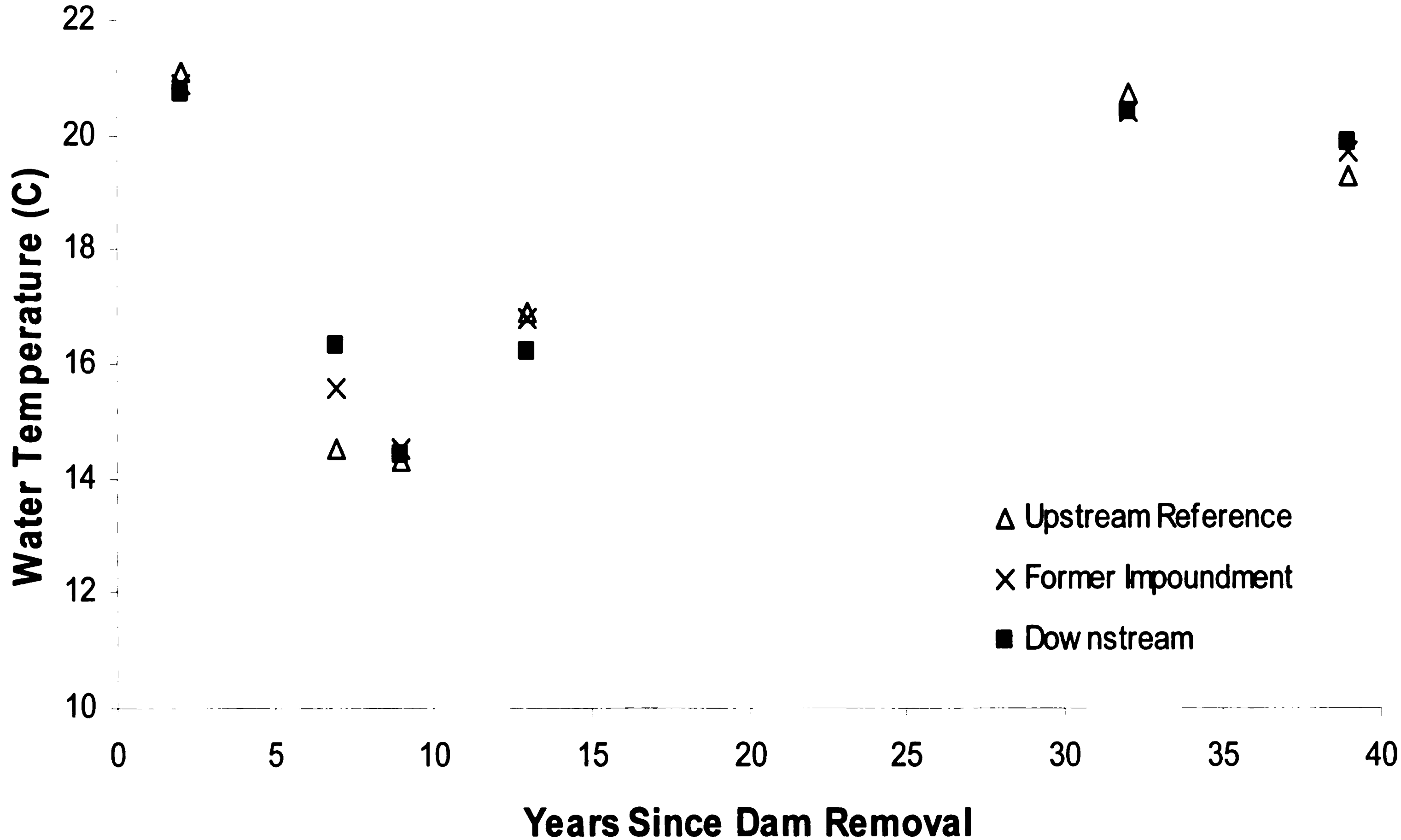

Figure 3. Mean water temperature for each study zone, recorded from 7/29/05 – 10/2/05.

past dam removals studied, indicating that once a dam and impoundment were removed, its effects on water temperature were almost immediately ameliorated.

In former impoundment reaches, slope was similar to or greater than reference levels, in all but one case, and this process appeared to occur within just a few years following dam removal (Figure 4). Differences in slope downstream of dam removals appeared more variable. Several downstream reaches had slopes greater than reference levels, while others had lower slopes than reference levels even 8 -15 years after dam removal. In the two oldest dam removals, greater than 30 years post-dam removal, slopes of both former impoundments and downstream reaches were similar or greater than reference reaches.

Width to mean depth of the wetted channel (W/D) is a unitless ratio describing the shape of the river channel cross-section morphology; higher W/D indicates a wide shallow river channel and a lower W/D indicates a narrow and deep river channel. Former impoundment zones generally had higher W/D than reference zones, but appeared to decrease in W/D following dam removal, approaching reference level W/D by approximately 10 years (Figure 5). Downstream of dams, sediment transport and deposition appeared to increase W/D of the wetted channel immediately following dam removal, but eventually reference level W/D were achieved by 30 years post dam removal (Figure 5).

Sinuosity of the river channel in all three study zones was measured for multiple years at each dam removal site, and no changes were observed (Figure 6). At some rivers, both former impoundment and downstream zones were more

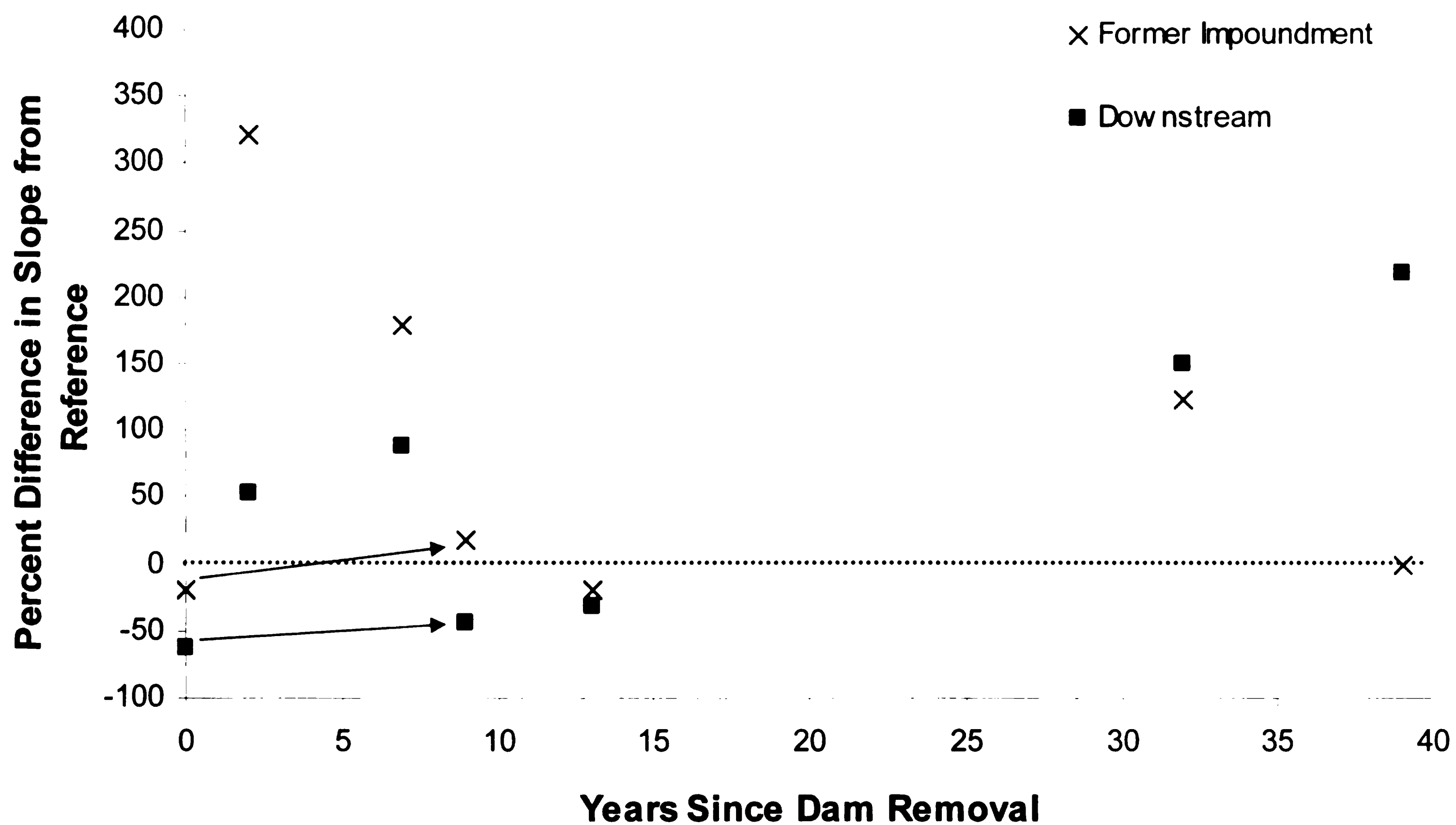

Figure 4. Percent difference in slope for former impoundment and downstream zones, compared to slope in reference zones. Arrows connect data points from the Pine River, indicating the trajectory observed before to after the removal of Stronach Dam.

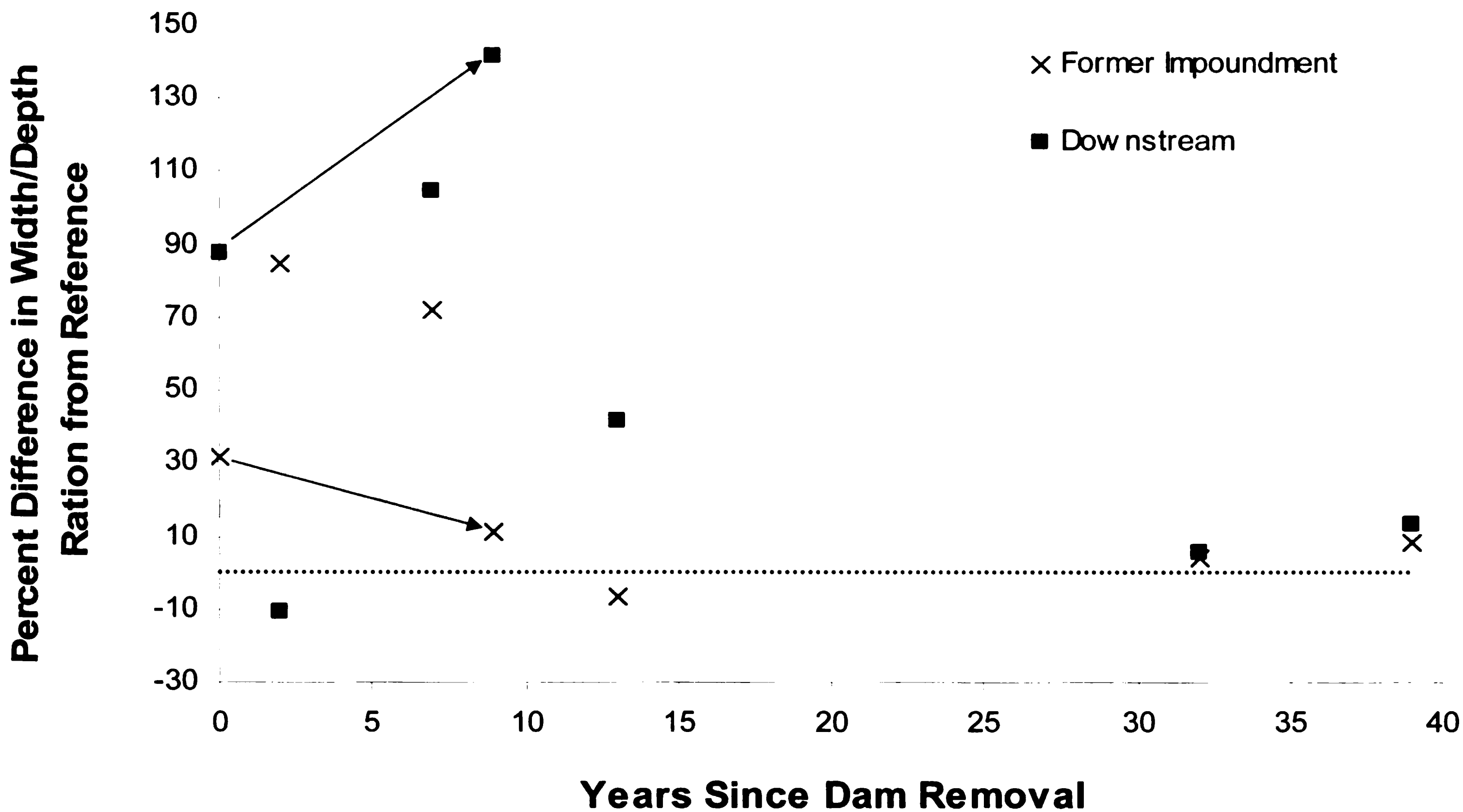

Figure 5. Percent difference in wetted width to depth ratio (w/d) for former impoundment and downstream zones, compared to w/d in reference zones. Arrows connect data points from the Pine River, indicating the trajectory observed before to after the removal of Stronach Dam.

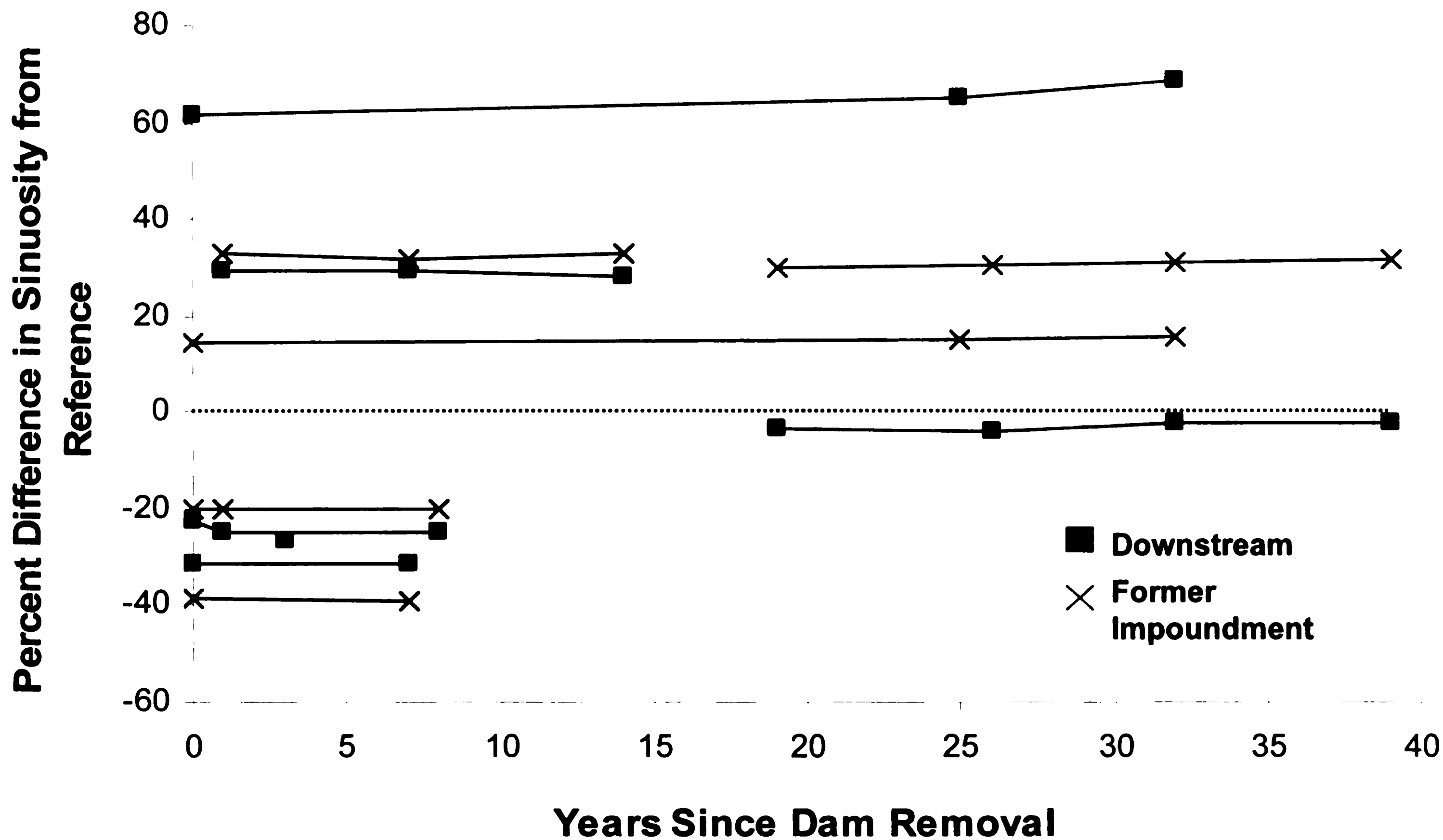

Figure 6. Percent difference in sinuosity for former impoundment and downstream zones, compared to sinuosity in reference zones. Connected data points indicate the sinuosity of the same study zone, measured at different dates through aerial photos.

sinuous than reference zones, and in other study rivers they were less sinuous. However, in all cases the sinuosity of the river channel in the former impoundment and downstream zones remained stable following dam removal.

Median substrate sizes in former impoundments and downstream zones were generally smaller than reference zones shortly following dam removal. One exception occurred where substrate in the former impoundment and downstream had coarsened to levels slightly higher than in the reference, just two years following dam removal, and in another case (excluded from graphical presentation) median substrate size was far greater than reference levels due to frequent boulders and bedrock in the waterfall-filled former impoundment and downstream zones. On average, substrate appears to begin coarsening slightly in the former impoundment immediately following dam removal, while the downstream zone shows very little substrate coarsening (Figure 7). The oldest dam removal cases studied, greater than 30 years post-removal, showed median substrates sizes in both former impoundments and downstream zones that were larger than those of the reference zones, indicating that restoration of substrate coarseness is possible following dam removal.

Bedform diversity was generally lower in the former impoundment than reference zones, following dam removal (Figure 8). The diversity of bedforms appears to increase following dam removal, with a greater rate of increase in the former impoundment zone than for the downstream zone. However, the diversity of bedforms in the former impoundment zones was still lower than reference zones even after 30 years post-dam removal. In the downstream zones,

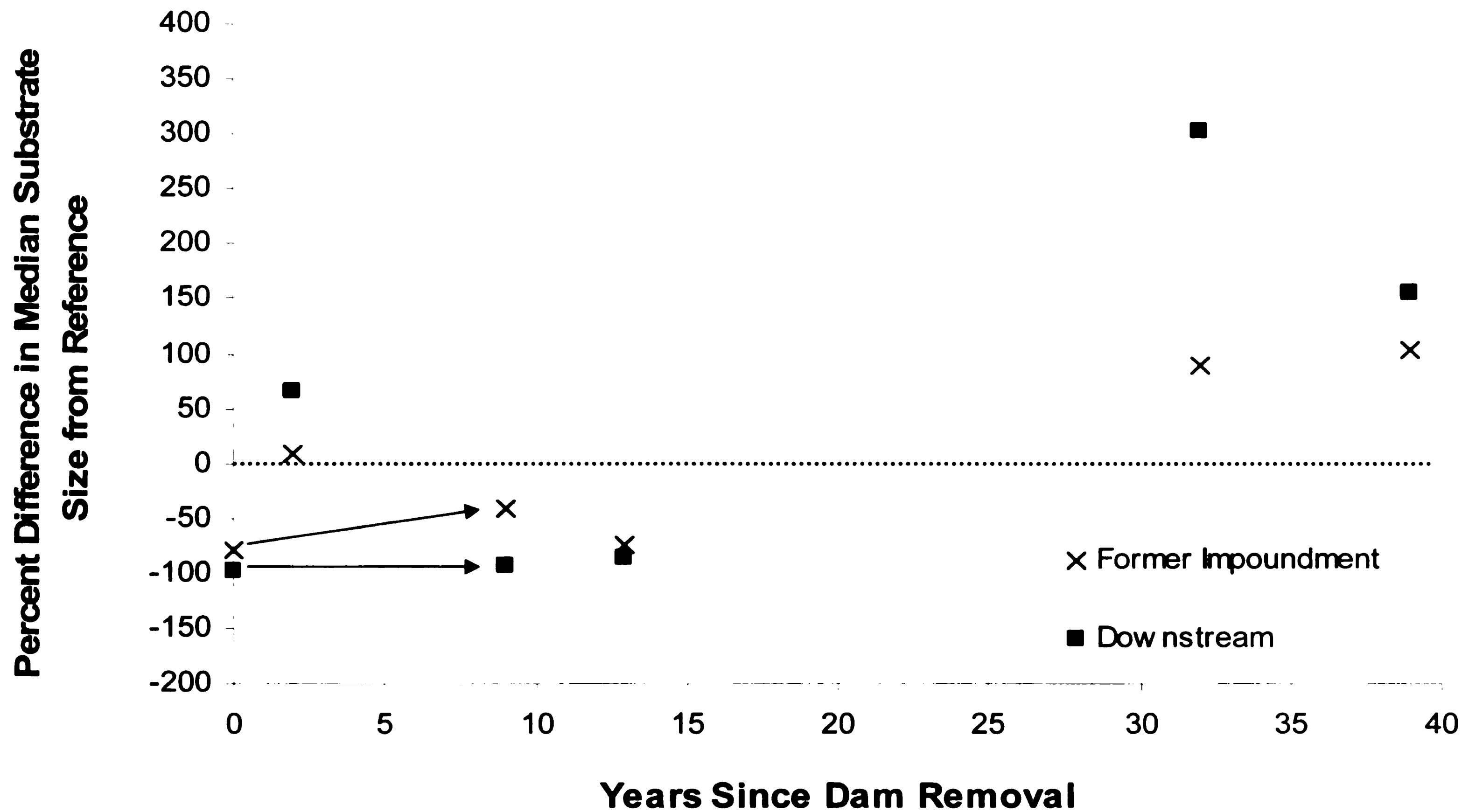

Figure 7. Percent difference in median substrate size for former impoundment and downstream zones, compared to median substrate size in reference zones. Arrows connect data points from the Pine River, indicating the trajectory observed before to after the removal of Stronach Dam.

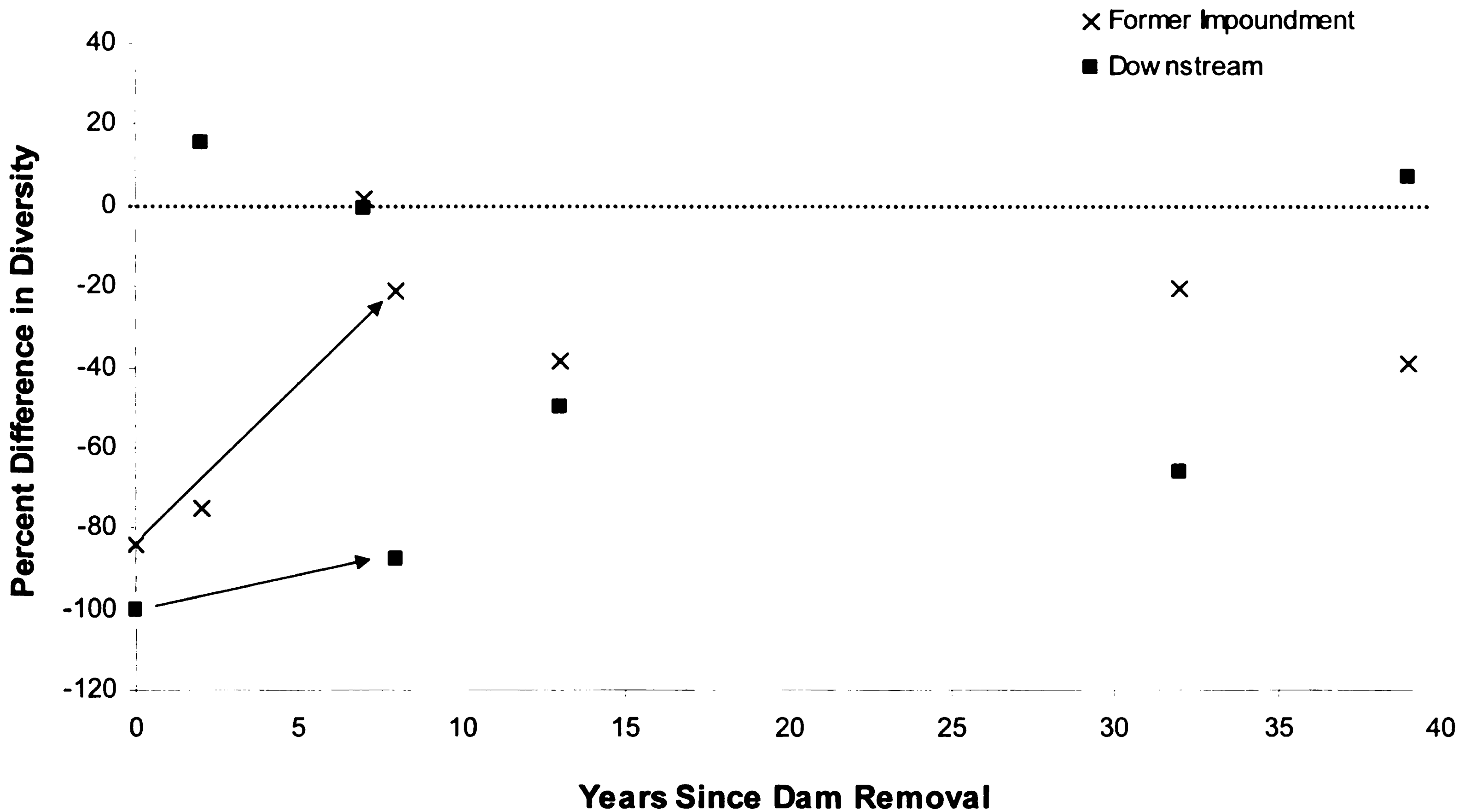

Figure 8. Percent difference in the diversity of bedforms for former impoundment and downstream zones, compared to reference zones. Arrows connect data points from the Pine River, indicating the trajectory observed before to after the removal of Stronach Dam. Shannon-Weaver diversity index used.

bedform diversity was more variable, with many of the sites attaining reference-level bedform diversity, while others remained less diverse. The seven-year post-dam removal study site, the Falls River, remained an exception to the general trends (excluded from graphical presentation). In this river, the reference zone was lower gradient and contained more run bedforms, while the impoundment and downstream zones were extremely high gradient, and contained extensive waterfall bedforms.

Fish Community Composition and Productivity

Fish species richness differed between study zones for several years following dam removal, but by approximately 5 years post-dam removal fish species richness was similar between all three zones (Figure 9). Morista similarity index values also were similar for all zones within only a few years (Figure 10). The 7 year post-removal site remained an exception to fish community changes, due to waterfalls in the former impoundment and downstream zones which affected fish species distributions and prevented the influx of several species from Lake Superior.

The total number of fish sampled in a study zone, adjusted for effort, was used as a surrogate for the productivity of the fish community. Total number of fish in former impoundment zones was relatively variable among rivers for the first 15 years or so, but across all rivers from 0 – 39 years post-removal, shows a gradual increasing trend, from slightly less fish than reference levels to slightly more than reference levels for the dam removals >30 years old (Figure 11).

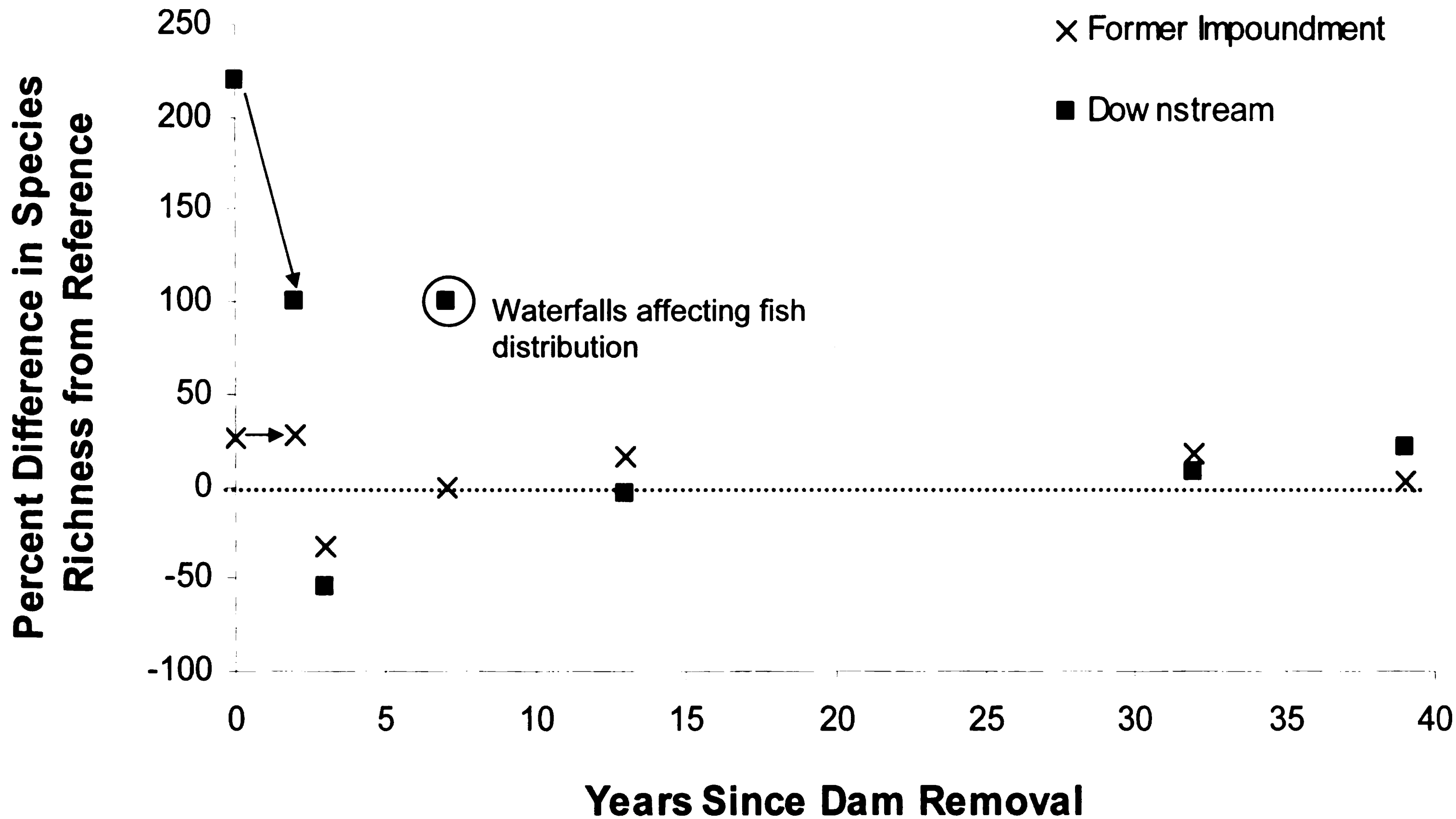

Figure 9. Percent difference in fish species richness for former impoundment and downstream zones, compared to fish species richness in reference zones. Arrows connect data points from the Pine River, indicating the trajectory observed before to after the removal of Stronach Dam.

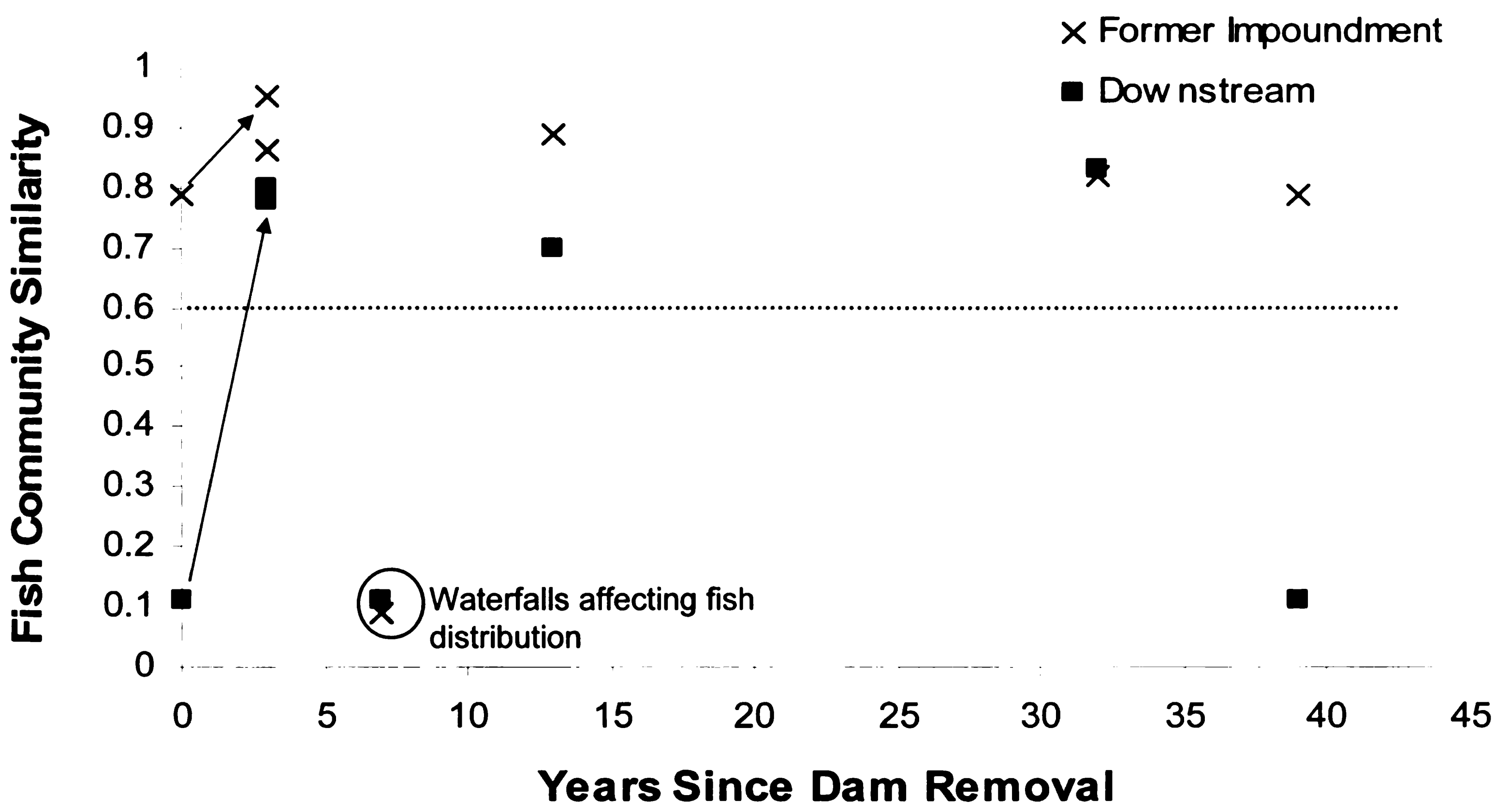

Figure 10. Morista similarity indices of fish community numeric composition, for former impoundment and downstream zones, compared to reference zones. Similarity indices of 0.00 denote complete dissimilarity, 1.00 denotes complete similarity, and indices greater than 0.60 indicate "similar" fish community compositions. Arrows connect data points from the Pine River, indicating the trajectory observed before to after the removal of Stronach Dam.

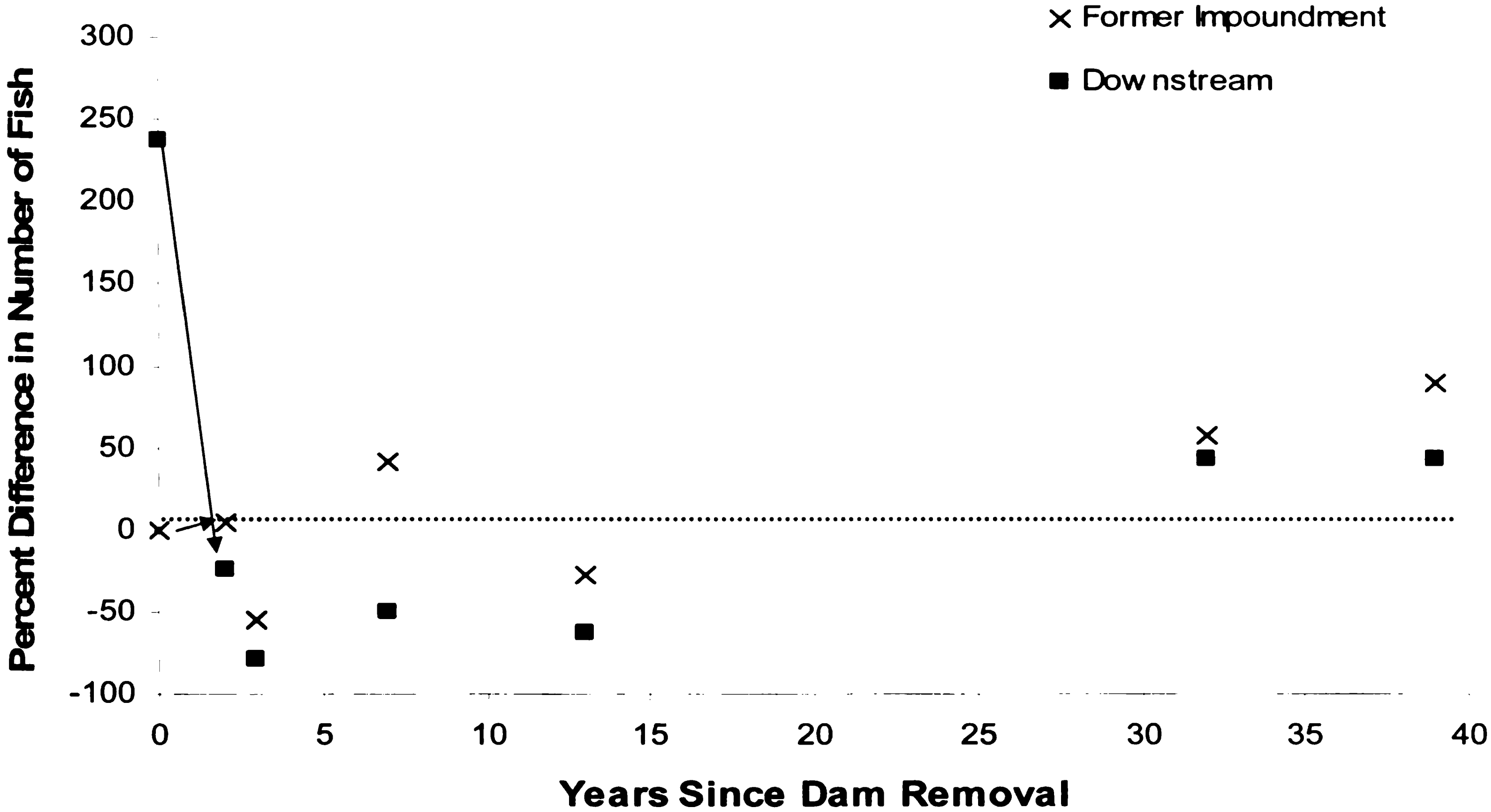

Figure 11. Percent difference in fish abundance (number of fish) for former impoundment and downstream zones, compared to fish abundance in reference zones. Arrows connect data points from the Pine River, indicating the trajectory observed before to after the removal of Stronach Dam.

Downstream zones show a more distinct trend in lowered fish abundance immediately following dam removal, with lower fish abundances for approximately 15 years post-removal, and higher fish abundances than reference zones by >30 years post dam removal.

DISCUSSION

Fluvial Geomorphology

Dams can alter the river temperature in numerous ways (Ligon et al. 1995, Collier et al. 1996). Lessard and Hayes (2003) found that in Michigan coldwater streams, water temperature downstream of dams was on average, 2.7° C warmer than upstream reference zones. These effects were thought to be caused by the warming of water retained in the impoundment, and its subsequent release downstream. Dams can also release water from the hypolimnion of an impoundment, lowering water temperature substantially relative to upstream. The removal of a dam ameliorates effects on water temperature almost immediately, as only very small differences in water temperature were observed in any past dam removals in this study. The water temperature effects of dams can be a major motivation and consideration in deciding whether to remove dams. Dams can alter water temperatures in ways both detrimental and beneficial to fisheries management. In some cases, water temperature can be warmed by dams to levels that will not support coldwater fisheries resources (Lessard and Hayes 2003, Hayes et al. in press). In other cases, coldwater releases downstream of dams can create water temperatures cold enough to support coldwater fisheries not otherwise possible, but displace native fish species not adapted to the colder water temperatures (e.g., Martinez et al. 1994, Quinn and Kwak 2003). The results of this study suggest that dam removal will restore water temperatures in impoundments and downstream sections of rivers to levels similar to upstream references. Whether this outcome of dam removal is viewed as desirable will depend on the fish community goals, and will likely vary with the uniqueness of each river and its fish community.

Slope is a main variable driving many of the characteristics of a river channel and fish habitat. Slope directly affects water velocity, thereby influencing channel morphology and substrate size composition. Dams generally decrease water slope in reservoirs through the impoundment of water and aggradation of sediment (Lane 1955, Gordon et al. 2004). Sections of rivers downstream of dams also generally experience decreased slope, through the degradation of the streambed immediately downstream of dams. Following dam removal, slope increases through a reversal of these sediment processes, as sediment degradation (i.e., incision) occurs in former impoundments and sediment aggradation occurs downstream of dams (Figure 12). The recovery rate of slope following dam removal will depend on how much sedimentation occurred in a reservoir and how sediments are managed. In reservoirs with very little sediment fill, dewatering the impoundment during removal will restore the slope of the former impoundment nearly immediately. Reservoirs that contain large amounts of sediment fill must undergo significant incision that could take several years to reach an equilibrium. In this study, we observed sites in which slope was

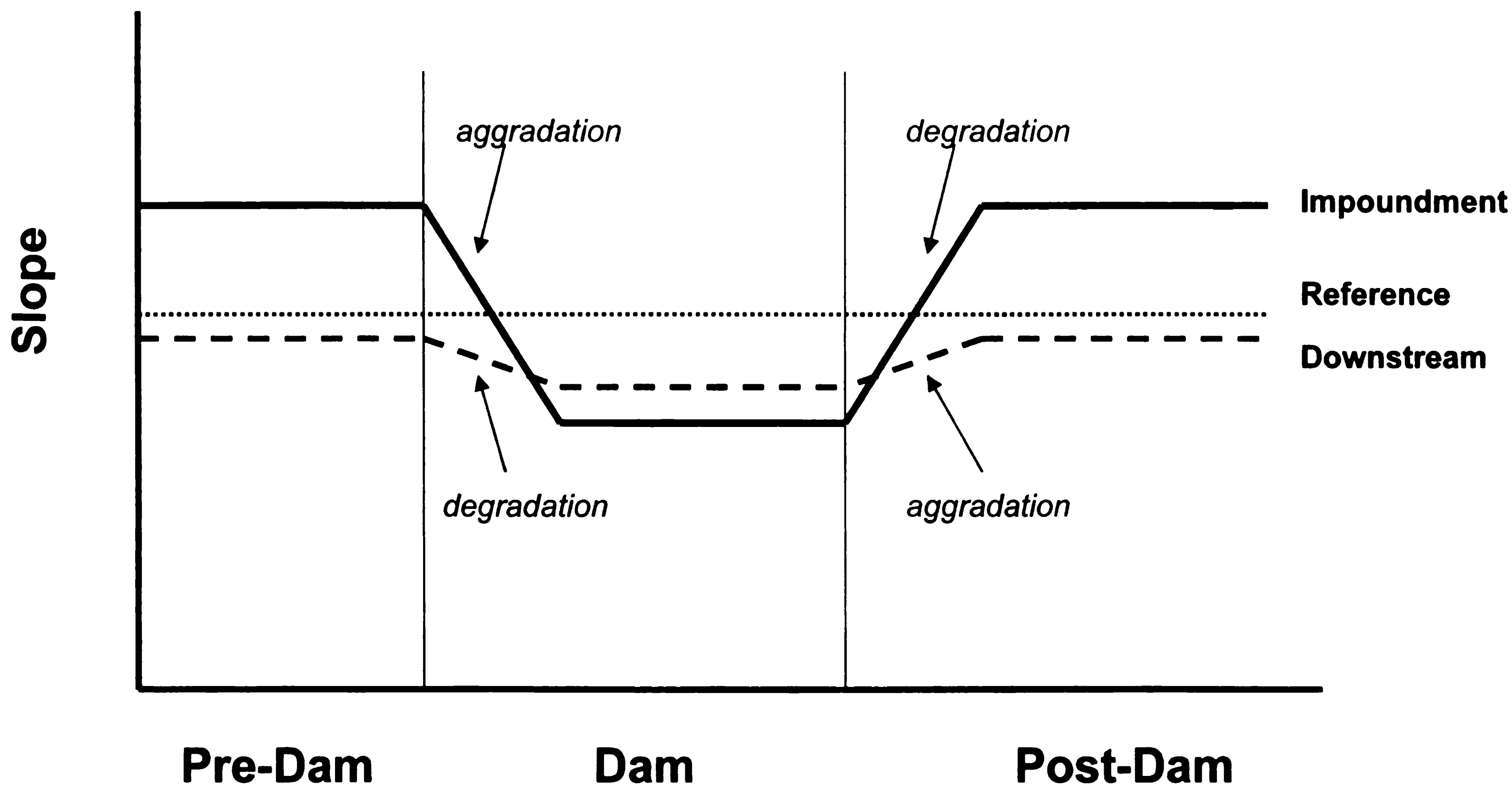

Figure 12. Hypothetical changes in the slope of sections of rivers affected by dams and dam removals, and the sediment processes causing the changes in slopes.

restored quickly and others where slope recovered more slowly. These differences could be explained by the amount of sediment contained in the impoundments prior to dam removal, natural variation in the underlying slope of the river reaches the impoundments were constructed on, or possibly by the methods used to remove the dams (e.g., staged versus all-at-once dam removal). Understanding the influence of each possible explanation will be difficult given the lack of pre-dam removal information available for these past dam removal study sites. However, it appears that slope generally reaches reference levels within 2 to 5 years following dam removal.

Immediately following dam removal, the lack of water impoundment immediately decreases wetted width of the river channel and often the water depth. Further, in impoundments that contain significant amounts of sediment fill, incision occurs vertically downward through these sediments, forming a narrower river channel, and further decreasing the width of the river channel and the W/D ratio. The results of this study suggest that the equilibrium slope and W/D ratio will eventually become very similar to reference conditions, but may take more than a decade for this new equilibrium channel to form. Downstream of dams, increased sediment supply following dam removal generally aggrades the streambed and increases both slope and W/D of the river channel. So whereas the impoundment zones move toward restoration of channel form immediately following dam removal, the downstream zones often move away from reference conditions immediately following dam removal. The results of this study suggest that river channels downstream of dam removals eventually move toward reference conditions, but require more time to achieve these conditions, perhaps 20 years or more. Reference conditions were observed in the downstream zones for the dam removals >30 years old, indicating that restoration of channel form is possible. One notable exception occurred with implications for how dams are removed. The Randall Dam on the Coldwater River was removed in 2002, during which a sediment collection basin was operated immediately downstream of the dam (D. Zebell *personal communication*). This prevented large amounts of sediment from impacting the downstream zone of this river. Consequently, many of the negative impacts of sediment deposition, including increased W/D of the river channel were not observed.

Sinuosity in former impoundments and downstream zones did not change following dam removal. Despite large amounts of vertical incision observed at some dam removals (Stanley et al. 2002, Burroughs 2007a), substantial lateral erosion through the floodplain was not observed. Sinuosity of a river channel influences the slope and subsequently the equilibrium channel shape. Sinuosity also affects the formation of pool bedforms at the outside corners of meanders, and affects the rate of wood debris recruitment and retention in rivers. It still remains uncertain whether river channels will revert to pre-dam removal river channels or whether new river channels will form through accumulated reservoir sediment fill. Managers may decide upon desirable locations of new river channels formed following dam removal, and the equilibrium slope and sinuosity of these post-dam removal channels, based on various management objectives. However, it is important to realize that sinuosity does not seem to change significantly following dam removal, so desired sinuosity should be considered prior to dam removal as part of an integrated dam removal plan.

As slope increases in former impoundment and downstream zones, water velocity increases, and substrate size composition coarsens slightly. This begins soon after dam removal in the impoundment zone, and later in the downstream zone if the reservoir sediment fill is allowed to be transported and deposited downstream. However, the median substrate size of reference zones is reflective of higher, more infrequent flow events. Even as substrate size coarsens in impoundment and downstream zones, the median substrate size would not be expected to coarsen to reference levels for many years or decades, until several high flow events have occurred. Results from this study suggest that substrate in former impoundments and downstream zones can become as coarse as seen in reference zones following dam removal. This substrate coarsening could be achieved shortly after dam removal if active sediment management efforts are employed (i.e., dredging, or passive collection), or if a large, infrequent flow event happens to occur shortly after the dam removal. In most cases however, it appears as though several decades may be required in order to achieve the full restoration of substrate coarseness following dam removal.

Similar to substrate coarseness, the creation of bedforms is controlled by infrequent high flow events, often 5 -10 year recurrence interval floods, that possess sufficient energy to scour pools and transport and deposit large substrates, forming riffles (Petts and Foster 1985, Knighton 1984, and Beschta and Platts 1986, as cited in Gordon et al. 2004, and Pizzuto 2002). Bedforms can also be created secondarily by other localized structures or events such as wood debris or bank slumping. In this study we observed an increase in bedform diversity in former impoundment and downstream zones following dam removal, but full restoration of bedform frequency was inconsistent, even decades following dam removal.

In the case of the Stronach Dam removal on the Pine River, the former impoundment was almost entirely run bedform prior to dam removal (Burroughs 2007a). Eight years after the dam removal, the proportion of run bedform had decreased with increases in riffle and pool bedforms, but the diversity of bedforms was still less than observed in the reference zone. The results of this study suggest that while bedform diversity may improve in former impoundments following dam removal, the process will be slow, and reference-level bedform diversity may not be achieved. Several of the downstream zones in this study did attain reference-level bedform diversity. The two downstream study zones that remained low in bedform diversity were both low gradient and emptied into other impoundments. This situation may reduce sediment transport ability during high flows and slow or prevent the reformation of diverse bedforms downstream of dam removals.

Bedform diversity influences sediment transport and sorting, nutrient cycling, and is important to the habitat suitability of stream biota (Gordon et al. 2004). Given the lack of bedform restoration in impoundments following dam removal, managers may need to consider actively restoring bedforms following dam removal.

Fish Community Composition and Productivity

While many aspects of the fluvial geomorphology of rivers seem to be restored eventually following dam removal, changes that occur to the fish community can not necessarily be seen as restoration. With the initiation of fish passage following dam removal, often quite different fish assemblages are allowed to intermix, and interact with varying habitat conditions. The result is a new and different fish community following dam removal, throughout all of the zones.

Hill et al. (1994) found that species richness increased in the Chipola River, Florida following dam removal. Kanehl et al. (1997) found index of biotic integrity scores increased in the Milwaukee River, Wisconsin following dam removal; and Burroughs (2007b) found fish species richness, diversity, and similarity of numeric community composition increased in all three zones (reference, former impoundment, and downstream) following the removal of Stronach Dam on the Pine River, Michigan. Results from this study support these other findings, in that the similarity of species richness and numeric composition between study zones increased very quickly, within just a few years following dam removal. Again, these changes are not necessarily interpreted as restoration of original “pre-dam” fish communities, but rather homogenization of fish assemblages within a river. Fisheries managers faced with potential dam removals will need to decide whether these changes are desirable for a specific river.

The trends in fish abundances after dam removal follow many of the changes in fish habitat observed in this study. Many aspects of fish habitat were observed to change immediately following dam removal, such as water temperature and slope. These changes will immediately alter the suitability of the habitat in former impoundment and downstream zones, leading to changes in the fish communities and fish abundances in each zone. These immediate changes alter the former impoundment to reflect more reference-like lotic conditions, while downstream zones are often inundated with sediment transport and deposition from the incision process in the former impoundment. This leads to wider and shallower river channels, and smaller sized, unstable substrates. Correspondingly, fish abundances in downstream zones were lowered to levels far below those of the reference zones, taking several decades to recover. Bushaw-Newton et al. (2002) also observed decreased fish abundance downstream of a dam removal in Pennsylvania immediately following dam removal.

Other important aspects of fish habitat were observed to take much longer amounts of time to recover after dam removal. The frequency of different bedforms such as riffles or pools are fundamentally important to the survival and feeding needs of different life stages of almost all stream fishes. Substrate coarseness is also related to the spawning requirements of many stream fishes, the survival needs of many juvenile fish as holding cover, and the productivity of macroinvertebrate communities which serve as food for many stream fishes. These important attributes of fish habitat were observed to require long periods of time post-dam removal to be improved or restored. It should not be surprising then, that fish community productivity as measured by fish abundance, also required a long period of time (~20 – 30 years) post-dam removal in order to reach levels seen in upstream reference habitat. Fisheries managers hoping to improve the productivity of a fish community through dam removal, may need to consider active restoration of key fish habitat attributes such as substrate composition and bedform diversity, if the full potential benefits of dam removal are to be achieved within short management timeframes.

Synthesis

While the results of this study were derived from a limited number of dam removal case studies, they provide the first empirical estimates of the timeframes required for various attributes of rivers and their fish to recover or change following dam removal, and also generate novel insights into the trajectories these changes may follow. Many of the aspects of river form and function appear to be restored through dam removals. However, many of the most important aspects of fish habitat may take several decades to be restored, and the full benefits of dam removal won't be achieved until those aspects of fish habitat are recovered.

These results give guidance on where to best spend limited funds directed at dam removals and associated active restoration. Sinuosity did not change following dam removal, and bedform diversity and substrate coarseness took long amounts of time to improve or be restored. If active restoration is feasible following dam removal, efforts aimed at creating meanders, pool and riffle forming structures and coarsening substrate, could help to speed the restoration of these important habitat attributes and lead to a faster realization of the full potential benefits of dam removal.

Deciding on whether dam removal is good for a fish community will always require consideration of the uniqueness of the river, the fish assemblages surrounding the dam removal in question, and their preferred uses and states. Dam removal allows mixing of species from downstream and upstream of dams. This will allow all fish to access and sample all habitats available to them and select those that best fulfill their life history requirements. This appears to lead to higher species richness, higher similarity of fish communities upstream and downstream of dams, and higher diversity. While Hill et al. (1994), Kanehl et al. (1997) and Burroughs (2007b) all observed increases in abundances of desirable sportfishes, along with increased species richness following dam removal, changes in fish habitat and new species interactions could have detrimental effects for some desirable fish species. In order to make a decision on the desirability of suspected outcomes of potential dam removals, fisheries managers will need to consider the fish species present in the river, those fishes' habitat requirements, and the projected changes to habitat expected due to dam removal.

Surveying the outcomes of past dam removals, even with limited sample size, has revealed valuable insights, including the extent of restoration possible with dam removal, and the timeframes required for restoration. Pohl (2003) estimated that more than 400 dams have been removed in the U.S. to date. There is great potential for researchers to apply this methodology to the study of other past dam removals. As more of these past dam removals are studied, more reliable estimates of the timeframes of restoration of different attributes will be possible, the timeframes could be calibrated to specific geological or ecological regions, and the influence of starting conditions on time-to-restoration could be examined. For example, if particular starting conditions exist prior to dam removal, and they are known to lead to faster restoration times, decisions can be made whether to conduct active restoration or whether passive restoration would be sufficiently fast. Also, as more dams are removed in the future with active site restoration conducted (e.g., sediment collection and removal, natural channel design), the effectiveness of these efforts can be evaluated by whether or not they lead to faster restoration than average timeframes with just passive, natural restoration; and cost-effectiveness of dam removals can be improved.

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