EVALUATING RIPARIAN AREA DYNAMICS, MANAGEMENT ALTERNATIVES AND IMPACTS OF HARVEST PRACTICES

Final Report to Minnesota Forest Resources Council
June 12, 2001

Minnesota Forest Resources Council Report
No. RR- 0601

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Overview and Synthesis

Introduction

Forested landscapes are highly productive and provide significant societal value. Riparian ecosystems are some of the most sensitive and most valuable elements of forested landscapes. These riparian systems and their associated water quality attributes are subject to degradation if timber is harvested using careless practices. Sustainable forest management practices maintain the flow of economic goods from the forested landscape and retain other values that society views as important (e.g., high biodiversity, high quality). The various parties with an interest in forest management in Minnesota have invested heavily in development and implementation of guidelines and prescriptions that promote sustainable forest management practices to maintain all of the values society assigns.

This four-year project was funded by the Minnesota Forest Resources Council to provide data, information and synthesized interpretations of the effects of timber harvest practices on riparian areas in two northern Minnesota forested landscapes. This study follows a previous two-year effort funded by the Forest Resources Council (Perry *et al.* 1998). The enti8er four-year effort was conducted in collaboration with a group analyzing timber harvest impacts on bird communities (http://oden.nrri.umn.edu/mnbirds/relatedprojects/riparian/riparian.htm). A range of sub-projects is included in this work. Each project group has developed its final report, summarizing the questions posed and the results obtained as well as providing an interpretation of the relationships between experimental manipulation of the forest and the effects measured. This Overview and Synthesis section describes the experimental design and common questions, and highlights significant findings from the four-year effort. Results from each of several studies are presented in 'Findings of Fact' sections; these results are measurements from the field sites and scientific interpretations based on those measurements. That information is followed by a section titled 'Interpretations for Riparian Management' which is more interpretive and is based on the best professional judgement of the scientists involved in this project.

Study Site Location and Experimental Design

We originally selected two watersheds in two distinct ecological systems: the **Pokegama Creek** system is on UPM Kymene Company land near Grand Rapids, in the Northern Minnesota Drift and Lake Plains Ecosystem. The **West Branch of the Knife River** is on State, St. Louis County, and Potlatch Corporation lands located west of Two Harbors. This site is in the Northern Superior Uplands Ecosystem. A third site, established in 1998 in the **Cloquet River** watershed received minimal attention in this study but represents additional information available from some of the Project investigators. This watershed includes the Langley River and Sullivan Creek in Lake County, on Potlatch Corporation and US Forest Service lands; it also is in the Northern Minnesota Drift and Lake Plains Ecosystem. Due to logistical difficulties in implementing the experimental harvests and complexity of responses, data presented here are almost exclusively from the Pokegama Creek basin.

This project is based on an experimental manipulation, using a randomized block design, with four treatments in each of three blocks. On the Little Pokegama Creek watershed (Fig 1), 12 treatment stands were established, each 4.8 ha (12 ac.) in size (2.4 ha, 6 ac. on each side of the stream). The upland portions of nine stands were clearcut, using either a cut-to-length (CTL)

system or a tree length feller-buncher grapple skidder (FT) system. The riparian portions of six of these nine stands also were cut, using the same system as in the upland. Our target residual basal area was 5.7 m^2/ha (25 ft^2/acre); harvests resulted in actual residual areas of 10 m^2/ha (44 ft^2/acre). The riparian areas of the remaining three cut stands (uplands cut) were left intact. The last three stands were true controls, with no harvesting in the uplands or riparian areas. In each stand, the fixed-width riparian management zone consists of a 61 m (200 ft.)-wide strip centered on the stream (30m, 100 ft. on each side). The length of stream contained in each stand ranges from 136-197 m (450 to 650 ft). Little Pokegama Creek sites were harvested in late summer-fall, 1997. Several of our plots are located on designated trout streams. In all cases, local Department of Natural Resources fisheries biologists participated in site selection and granted permission for the manipulations.

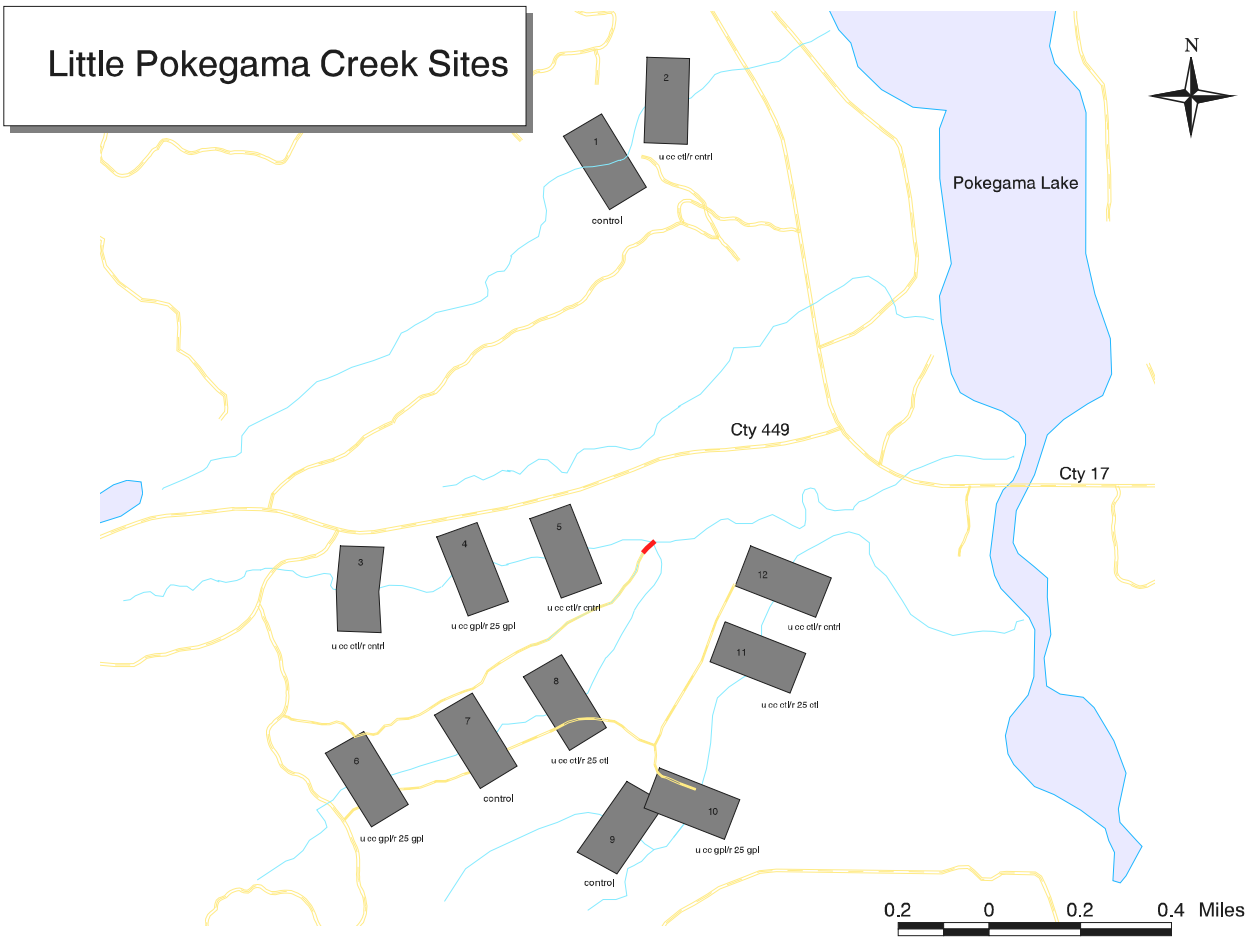

Figure 1 Displays the location of treatment sites and treatment type within the Pokegama Creek study area.

(■ =RCNTRL, ■ =CONTROL, ■ =CTL, ■ =FT)

At the Knife River site, 12 stands were established on one side of the river; riparian management zones (30m, 100-ft. wide) were delineated on that side. The length of stream channel contained in each stand was the same as at the Pokegama sites. Stand sizes varied from 2.4-6.4 ha (6-16 ac.). Treatments for the 12 stands were as follows: three true controls; three riparian controls (upland clearcut, riparian area uncut); three riparian clearcut (upland clearcut as well); three riparian residual (5.7 m^2/ha, 25 ft^2/ac in riparian area, upland clearcut). We used a feller-buncher with tree-length skidding (i.e., TL) harvesting system on all sites. This site used a modified block design; three blocks contained each of the three cutting treatments, while three uncut controls are located in a separate block. We experienced significant logistical difficulties (many of which were due to the vagaries of the weather) in establishing our treatments on the Knife River. Our harvests occurred during the winters of 1997, 1998 and 1999.

Our Cloquet River experiment included 12 stands established on one side of the river only. In this watershed, topography was used to define the riparian area (in contrast to using a fixed distance). Treatments were as follows: three true controls, three riparian controls (upland clearcut, riparian area uncut), three cut riparian areas having clumped residual distribution of approximately 11.4 m^2/ha (50 ft^2/ac), three cut riparian areas having uniform residual distribution of approximately 11.4 m^2/ha (50 ft^2/ac). The stands were harvested with a feller-buncher with tree length skidding (i.e., the TL system) and were harvested in the winter of 1999.

In summary, the following designations describe our treatments:

	Treatment	
Designation	**Upland**	**Riparian Area**
True Control	Unharvested	Unharvested
Riparian Control	Clear cut, either CTL or TL-system	Unharvested
CTL	Clear cut, either CTL or TL-system	CTL
TL-system	Clear cut, either CTL or TL-system	Tree-Length harvest

Findings of Fact: Pokegama Creek[1]

1. Riparian plant species biodiversity (Palik)

Riparian herbaceous vegetation was sampled along stratified transects, using floodplain, terrace, slope and upland as the stratifying variables. Biodiversity was expressed as species richness and compositional similarity. Riparian areas in these watersheds contribute uniquely to plant diversity, but the contribution is scale- and measure-dependent. At the plot-scale, riparian areas rarely had species richness values that exceeded the upland, however, when plots are pooled (reach scale), floodplains do have higher species richness than the uplands. At the watershed scale, both floodplains and terraces and slopes have richness values that exceed the uplands.

2. Riparian vegetation: biomass, regeneration and stem density (Palik)

Aspen suckers require high amounts of light to survive and grow. Some authors have questioned whether adequate aspen regeneration is possible with even modest amounts of residual overstory (e.g., Perala 1977 reported a 35-45% reduction on aspen regrowth with as little as 3.3-4.9 m^2/ha [10-15 ft^2/ac] overstory). We found that the two harvest systems (TL and CTL) resulted in equal biomass and stem densities of new suckers and that those densities and biomass estimates were much lower than in clearcut uplands. It is apparent that intolerant species such as aspen and paper birch will regenerate in partially harvested RMZs but both biomass and density might be as much as 50% less than that in a clear cut stand. The long term variability of regenerating aspen and paper birch within the RMZ is unknown. As crown closure within the RMZ occurs, it is unknown whether the regenerating aspen suckers will survive.

3. Leaf litter input to streams (Palik)

Nearly one-third of leaf litter originates more than twice the distance from the channel than would be expected from the literature (Perry *et al.* 1998). That trend continued to be apparent from our four-year data set. That leaf litter represents coarse particulate organic matter (CPOM), which serves as the food base for much of the aquatic ecosystem in these shaded forest streams (Shelly and Perry 2000). We found a linear decrease in stream-input of CPOM from the least to the most disturbed treatments; the riparian area sites that were most aggressively harvested (i.e., CTL and TL systems) received the least CPOM. Overall, we found that three years after harvest, CPOM flux to the streams was reduced 33-47% within the 30m (100 ft) riparian zone. It is unknown whether this trend will continue.

4. Windthrow (Hemstad and Newman)

Trees in forests are exposed to the forces of the wind and often fall during storms, causing forest gaps. This windthrow is a natural forest function, but may be increased under harvest conditions, as increased numbers of trees are exposed. We found that riparian windthrow significantly increased during our experiment and that the increase was significantly related to harvest treatment; windthrow was greatest in full tree harvest reaches and lowest in the control and riparian control treatments. There may be an (we did not test this – no data) interaction between windthrow propensity and soil saturation: trees will be more susceptible and windthrow will be more likely to increase when harvests occur in areas with saturated soils. It appears that soil

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[1]The report containing the detailed data for each suite of variables is referenced by authorship; individual reports follow this Overview and Synthesis

moisture and residual stand density, independent of RMZ width, controls the probability of windfall.

5. Large woody debris (LWD) in stream systems (Johnson, Stroom, Schomberg, Richards)

Fallen trees as well as branches and even the relatively large diameter portions of logging slash provide many positive functions in stream systems, including providing substrate, directing flow and diverting hydraulic energy, blocking transport and thus increasing CPOM storage and structuring local scale channel morphology. We did find that slash accumulation in stream channels was significantly higher in harvested riparian areas and higher with CTL harvesting than with other treatments (All other measurements (e.g., wood length, wood volume) were highly variable, often varying over two orders of magnitude among the sites. No significant differences were found among treatments and no patterns in space or time were evident. As windthrow trees begin to break and decompose, wood will be introduced to the channel and will form debris accumulations; long term followup studies are needed to completely assess the effects of harvest activities on wood standing stocks. Studies in the western US suggest that effects of harvest can be detected for more than a century (Richmond and Fausch 1995), and recovery to preharvest levels may take as long as 250 years (Murphy and Koska 1998).

6. Stream benthic organic matter (Johnson, Stroom, Schomberg, Richards)

Coarse Particulate Organic Matter ccumulates in streams and serves as a stored food for benthic invertebrates. We found that CPOM volumes in the Pokegama Creek stream sites were highly variables through time and space. CPOM accumulations at treatment sites tended to vary differently from control and riparian control sites, although trends were not significant ($p = 0.06$). As branches from windthrow trees begin fall into the stream and decompose, standing stocks of CPOM are expected to increase; increased standing stocks of wood also will increase CPOM retention.

7. Stream temperature and water quality (Fredrick and Perry, Verry)

Our experimental harvests opened up the forest canopy and increased the amount of incident light reaching the stream channel. That effect has been widely reported from other studies and has been widely shown to result in increased water temperatures, potentially threatening cold water species such as trout. We found no significant treatment-related effects and temperatures in all streams remained within the acceptable range for trout. All of our streams are groundwater fed and have water chemistries that are typical of the geology of the region (i.e., dominated by calcium magnesium and carbonates). We found no significant treatment effects on any water chemistry variable.

8. Stream substrate (Fredrick and Perry, Verry)

The geology of this region results in the stream substrates being primarily sand. There were large year to year changes in substrate composition and percent fine sediment increased the year after harvest across all sites, possibly due to climatic variability or to increased road use within the basin. We did find that some downstream sites had increased sand depth, reduced particle size and increased sedimentation. Those differences were not significantly related to our treatments but were related to the presence of road crossings, to total extent of forest harvest in the watershed and to the presence of harvests and roads on saturated soils.

9. Benthic macroinvertebrates (Fredrick and Perry)

We found evidence that the FT harvest system had the greatest impact on integrity of the macroinvertebate community; that impact was followed in severity the CTL and Riparian Control harvest treatments, respectively. The limited replication of our treatments (n=3) and the high natural variability in our dependent variables limited our ability to show site-specific impacts. In fact, the overall trend in reduced biotic integrity was observed at all sites, including true controls. These declines were correlated with the extent of overall disturbance in the watershed, indicating a need for further research at the watershed scale.

10. Fish Habitat (Hemstad and Newman)

As might be expected, overhead canopy cover was reduced in the two riparian harvest treatments and decreased further over time with increased windthrow. Slash (woody cover) increased in both riparian harvest treatment but more so with CTL harvesting. Harvest treament did not increase the amount of unstable bank (which increased in unharvested reaches). As noted above, fine sediments increased in all harvested reaches the year after harvest, but basin-wide, year to year differences in sediment appeared more important.

11. Fish (Hamsted and Newman)

As might be expected, overhead canopy cover was reduced in the two riparian harvest treatments and decreased further over time with increased windthrow. Slash (woody cover) increased in both riparian harvest treatment but more so with CTL harvesting. Harvest treament did not increase the amount of unstable bank (which increased in unharvested reaches). As noted above, fine sediments increased in all harvested reaches the year after harvest, but basin-wide, year to year differences in sediment appeared more important.

12. Archeology and cultural artifacts (Emerson)

In 1996 and 1997, we used replica archaeological deposits, using materials that closely replicate those typical of authentic archaeological sites in Northern Minnesota, to assess the effects of harvesting on cultural resources. Approximately 5% of the artifacts were lost or damaged during harvest of a plot. Horizontal displacement of shallow materials and breakage of fragile materials are the most likely effects of harvest activities; those effects tend to occur only in areas directly trafficked by heavy equipment. Detailed results for this part of the study were reported in the FRC I report (Perry *et al.* 1998) and are not repeated here.

13. Harvesting system efficiency and damage to residual trees (Thompson, Mattson, Blinn and Dahlman)

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Summary of Key Site_level Findings from Pokegama Creek

1. Floodplain species biodiversity was higher than within the uplands. Because species richness appears to stabilize around 30 m (100 feet) from Pokegama Creek, the width of that RMZ appears to be appropriate when species biodiversity is considered. That width may not be sufficient for other sites. The appropriate RMZ width for a site must be determined based on site-specific conditions.

2. It appears that harvesting segments up to 200 meters (600 feet) in length on both sides of a relatively small intermittent stream such as Pokegama Creek will not adversely effect stream temperature or water quality.

3. Regeneration of birch and aspen will be reduced by as much as 50 percent when 12.5 m^2/ha (44 ft^2/acre) of residual basal area is left within the RMZ. Aspen sucker density in the RMZs is well below levels considered to be adequate stocking at age three (Stone *et al*. 1999). Regeneration densities within the clearcut upland were adequate within this study.

4. On average, reduced aspen sucker biomass in the RMZs, as compared to the uplands, suggests the potential for reduced fiber production in these areas.

5. Partial harvesting to 12.5 m^2/ha (44 ft^2/acre) of residual basal area will neither result in quality aspen production nor the maintenance or increase of long-lived, late-successional species within the RMZ. While aspen will likely be a component of these RMZs in the future, its density will be much lower than found in single-cohort stands and lower than what is considered desirable from the standpoint of commercial production. A one-time entry into the RMZ for purposes of a commercial harvest may be inadequate for ensuring the regeneration success of long-lived, late-successional species.

6. There is a need for a comprehensive silvicultural plan for each RMZ prior to the initiation of harvesting activities. That plan should focus on ensuring adequate regeneration, growth, and survival of commercially desirable overstory tree species as well as the creation of stand structures that sustain or restore riparian ecological services.

7. Leaf litter input to streams is significantly reduced when harvesting occurs within a 30 m (100 foot) RMZ or when the upland is clearcut harvested within stands that are comparable in size to those assessed (4.8 ha or 12 acres). That reduction in leaf litter persists for at least three years.

8. Neither RMZ disturbance nor equipment treatments caused any significant impacts to stream benthic organic matter, water temperature, water chemistry, invertebrate taxa richness, fish species richness, or fish densities. While CPOM inputs to the stream were higher under the CTL system than for the TL system, there was no significant correlation between inputs and RMZ disturbance levels.

9. Significant effects of forest harvest treatments were found for late summer fish IBI scores, macroinvertebrate Biotic Index Scores, and % EPT Taxa. Lower IBI scores were found in

riparian-thin treatments relative to unharvested riparian areas in August when lower flow conditions likely limit habitat availability. These effects may be exacerbated by basin-wide increases in fine sediment and warrant further investigation. The relationship between upstream control and downstream treatment conditions as reflected by Biotic Index Scores and % EPT was affected by forest harvest. Both metrics displayed a decline in the condition of the selectively harvested sites relative to the riparian control. Additional significant changes in the invertebrate community, suggesting decreases in water quality, were observed to correlate with harvest throughout the basin.

10. Road crossings within a forest resulted in more input of sand and fine sediment to a stream than the actual timber harvesting operations. There is a need to minimize the amount and extent of road crossings within a watershed. There is also a need to more closely monitor the application of forest road guidelines within each watershed.

11. Because of their location within the landscape, riparian areas have a high probability of containing cultural resources. Harvesting activities within the RMZ can result in the horizontal displacement or breakage of such materials, especially within areas that are directly trafficked by equipment. It is important to continue emphasizing the need for avoidance of these areas during forest management activities (where it is practical and feasible) or the protection of these resources where avoidance is not possible.

12. The overall aerial extent of soil disturbance in the RMZ was less than in the adjacent upland. However, the incidence and severity of displaced soil was very similar, suggesting the increased sensitivity of the soil to operations near water. Anything that alters the hydrology (e.g., skidtrails, road crossings, soil compaction) within the RMZ changes the dynamics of the riparian area. This can affect the site's ability to support forest vegetation and other important functions. It is important to encourage use of low-impact equipment within RMZs and during periods when soil conditions can provide inadequate support to heavier equipment.

Other Key Findings

1. There is a need for additional research at the Pokegama Creek study site to assess long-term impacts of the treatments. Additional long-term studies need to be implemented elsewhere to assess both site-level and landscape-level impacts. We have clear indications of watershed scale effects but cannot yet quantify those.

2. The effectiveness of management within any particular RMZ depends on what is happening across the landscape. The types and intensity of land use practices within a landscape (e.g., the extent of harvest, conversion to non-forested status, development, number of road crossings) can have a greater influence on aquatic ecosystems than specific RMZ parameters. Management within riparian areas needs to consider both site-specific and broader landscape/cumulative effect perspectives.

Findings of Fact: Knife and Cloquet Rives

Harvesting and effects analysis on the Knife were confounded by logistical problems. Harvests were staggered in time due to weather conditions, site access and other influences. The streams in the Knife and Cloquet systems are much larger (i.e., wider, much greater volume) than those in the Pokegama and Knife River harvests were conducted on only one side of the stream. We found that variance among the sites was very high and we were did not detect any treatment effect on any of our variables. However, this does not suggest that effects were not present. We have reasons to believe that site-specific treatment effects from well-conducted harvesting at the scale of our plots (which were less than 6.5 ha, 16 ac) are less likely to be detectable in a stream the size of the Knife. However, we also feel that there is evidence from the literature and from our other work that would suggest that a large number of such plots spread throughout a watershed will result in significant cumulative impacts. Analysis of the effects of harvest treatments on stream fish and fish habitat will be conducted in Fall 2001 based on data collected from 1997 through summer 2001. An additional set of sites were sampled in 2000 and 2001 to relate larger temporal and spatial scale influences of riparian harvest to fish and fish habitat. Further work in larger systems like the Knife and the Cloquet should focus more on questions about percent of the watershed harvested and less on site-specific impacts.

Synthesis and implications for riparian management guidelines

Based on the findings reported above and in the associated papers and on our best professional judgment, the scientists collaborating in this project have drawn broader conclusions that are intended to facilitate the link between these research findings and their implications for management and policy.

Understanding site-specific effects

We found that the forested landscapes we studied are highly variable in space and time. That variance masked nearly all treatment effects we might have seen. As a result, we were able to show that some variables differed between the upstream and downstream sites surrounding a forest harvest, but we could not conclude that those differences were due to the harvest practice. Were we working in a less variable environment, we might have seen smaller upstream downstream differences (i.e., concluded there was no impact) or been able to show harvest-related effects. We did find that harvest reduced CPOM by 33-47% and that the CTL treatment resulted in more slash (fine wood) in the streams. We do not understand the ecological implications of those small wood increases. It is possible that the increases will have a positive effect on the invertebrates and minnows, and could have a negative impact on larger fish. Large wood provides a wide range of positive values in these landscapes; streams and their riparian zones will retain the highest ecosystem integrity if LWD inputs are relatively uniform through time. Increased probability of windthrow (as we saw on sites with saturated soils) will increase the temporal variability of LWD inputs. However, the diversity among plots and the dynamics of the wood and shrub communities make any long-term predictions of woody biomass highly imprecise.

Significantly, we could demonstrate few if any differences between the tree length (TL) and Cut

to Length (CTL) systems with the stream variables we measured in the Pokegama landscape. These systems also result in similar new cohort development in the partially-harvested riparian forests. In particular, aspen regenerated similarly with both treatments, but density and sucker biomass were greatly reduced, relative to the clearcut uplands. The CTL system did result in somewhat greater densities of shrubs and greater aboveground biomass of large regeneation (principally due to higher shrub species densities). These differences may lead to reduced commercial tree species success over time, due to greater competitive inhibition.

Understanding basin-wide effects

A variety of variables seem to be related to basin-wide rather than site-specific properties of the landscape. For example, changes in fine sediment appear to be explained by overall percent of a watershed harvested, rather than site-specific properties. Other authors have predicted that there would be a cumulative increase in species diversity as plots are accumulated through a watershed (Brinson and Verhoven 1999). We have documented that increase in species diversity and can use that information to further understand basin-wide impacts of management practice. We anticipate that a variety of impacts that were not significant at the site-specific scale here will be significant at the watershed scale. We also believe that site-specific activities that follow accepted forest management guidelines and disturbs $\leq$ 200 m (600 ft) on either side of the stream, and occurs on streams similar in size to Pokegama Creek will not be shown to have significant site-specific impacts.

Ensuring sustainable management of forested riparian zones

Several of the findings presented here have significance for development of Riparian Management Guidelines for Minnesota. It is apparent that neither a fixed width un-harvested riparian zone or a riparian zone with reduced residual basal area will protect all riparian and aquatic ecosystem functions. When forest harvest occurs in these landscapes, there will be demonstrable impacts. For example, CPOM inputs are likely to be reduced and water temperatures may change. It also is clear that geomorphology (e.g., channel shape, groundwater contact, soil saturation) will strongly influence the way a specific site responds to a harvest activity. However, it also is clear that riparian zones are highly productive and contain rare and commercialy valuable species. It appears that the current guidelines for trout streams (i.e., 60 m riparian buffers) adequately protect against water temperature change and provide minimal change in CPOM, riparian species dversity, and in-stream organic matter. We do not anticipate significant impacts on fish or invertebrate communities with such buffers. However, the current guidelines for non-trout species (i.e., 30 m) appear to adequately protect against temperature increases but do not protect against many other impacts such as COPM and LWD, and the resulting changes in biotic communities.

Future needs

It is clear from our work that some of our site-specific changes can best be understood at the scale of integrated effects in a basin or watershed (e.g., expressing independent variables as percent of a watershed harvested within a time frame). We feel that a more integrative, cumulative approach and/or examination using a choronosequence would be instructive in factoring out the effects of inter-annual variation and landscape heterogeneity. Another constraint evident from our work is that each variable seems to change on a specific space and time scale and local geomorphology will act to influence that way those changes are expressed; yet forest

management guidelines must be sufficiently generic that they can be applied relatively widely. There is a need to incorporate our results with a broader watershed perspective and develop a tool that is sensitive to fine scale landscape change yet can guide coarse scale management decisions.

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Effects of Riparian Management on Vegetation Variables: Herbaceous Plant Diversity, Coarse Particulate Organic Matter Flux, And Forest Regeneration.

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Abstract

As part of a larger integrated study of riparian management impacts, I quantified the contribution of headwater riparian areas to herbaceous plant diversity in a forested watershed, examined coarse particulate organic matter flux to headwater streams from harvested riparian management areas, and measured tree species regeneration dynamics, particularly aspen, under riparian management treatments that include retention of overstory competitors and different harvest technologies. I addressed these objectives within the context of a controlled experiment consisting of a randomized block design with 4 treatments, replicated in 3 blocks. In 1996, 12 treatment stands were established, each 4.8 ha (12 acres) in size (2.4 ha on each side of the stream). The upland portions of 9 stands were clearcut in summer-fall 1997, using either a cut-to-length logging system (CTL) or a whole-tree system (WT) consisting of a feller-buncher and grapple skidder. The riparian portions (30 m on both sides of the stream) of 6 of the 9 stands were cut, using the same system as in the upland, to a residual basal area of 10 m^2/ha (44 ft^2/acre). The riparian areas of the remaining 3 cut stands (uplands cut) were left intact (RC = riparian control). The last 3 stands were complete controls (C), with no harvesting in the uplands or riparian areas. Vegetation in all structural layers (trees, saplings, shrubs, seedlings, herbs) was sampled once before harvest, and three times after harvest (1998-2000) along a series of transects spanning the stream to upland gradient. Along each transect, sample points were stratified on major landforms (floodplain, terrace, slopes, upland). Overhead and lateral litter flux to the stream was collected in each of three years after harvest.

Individual riparian areas (within reach-scale) rarely had species richness values that exceeded the upland. At the among-reach scale, floodplains did have significantly higher species richness than other valley floor landforms (terraces and slopes combined) and the uplands. At the watershed scale, both floodplains and other valley floor landforms had richness values that exceeded the uplands. Species turnover was consistently high (50-90%) between floodplains, other valley floor landforms, and the upland at both within- and among-reach scales. The 30 m RMZ boundary used in this study fell on terraces or slopes 40% of the time and in the upland 60% of the time. These results suggest that: i) species richness contribution of individual riparian areas are not significant, but the contribution is evident when individual riparian areas are accumulated in a watershed; ii) at the watershed scale, riparian landforms other than the floodplain make a contribution to increased species richness; iii) the consideration of impacts to species richness should be focused less on site by site evaluations and more on minimizing the percent of a watershed in a disturbed condition; iv) high species turnover across the stream to upland ecotone is characteristic of riparian areas even at small scales; and v) fixed-width RMZ's often will subject landforms with unique species assemblages to direct impacts of logging, even if the RMZ itself is not logged.

Overhead and lateral CPOM flux to the stream decreased from the full control, to the riparian control (upland clearcut, RMZ uncut), to the partially harvested RMZ in each year. For overhead litter, the difference between the full control and the riparian control was significant in 1997 only, immediately after

harvest, while the difference between the full control and the partially-cut RMZ was significant in each year. Differences in lateral flux among treatments were never significant. These results indicate that a partially-cut RMZ with a residual basal area of 10 m^2/ha (44 ft^2/ac) does not sustain the functionality of an undisturbed riparian area, at least in terms of CPOM flux. Moreover, the data suggest that even an uncut RMZ, of 30 m width, may fail to maintain this functionality, at least in the first years after harvest.

Three years after harvest, composition of regeneration differed appreciably between the control plots (both RMZ and upland) and the harvested plots, but not between RMZ and upland plots. The primary differences were more aspen, paper birch, and mountain maple in the treated plots and more hazel in the control plots. Total density of regeneration remained relatively constant in the full control, but increased after cutting in the three RMZ treatments. Densities in 2000 (three years after harvest) were significantly higher in the harvest treatments than the control. Further, the RMZ harvest treatments had higher densities than the riparian control. Regeneration densities in the two RMZ harvest treatments did not differ significantly. Moreover, total regeneration densities of the two RMZ harvest treatment were similar in the clearcut uplands. Differences among RMZ treatments in above-ground biomass were similar to stem densities, increasing from the full control to the harvest treatments. Total above-ground biomass of regeneration in the uplands was nearly twice that in the RMZ's.

Aspen, paper birch, hazel, and mountain maple, showed consistent changes in density with treatment. Most notably, aspen and paper birch had higher densities in the two RMZ harvest treatments than in the riparian control, while the two harvest treatments did not differ significantly. Aspen and paper birch had significantly higher densities in the upland than the partially harvested RMZ's, while hazel and mountain maple had higher densities in the partially harvested RMZ's than the adjacent uplands. Mean three-year aspen sucker height ranged from 160 cm to 180 cm among the RMZ treatments, while mean sucker above-ground biomass ranged from 65 grams to 95 grams. Aspen sucker heights were not significantly different between the partially cut RMZ's and the clearcut uplands, but above-ground biomass of individual suckers averaged about 90 grams in the partially cut RMZ's, compared to 130 grams in the clearcut uplands. These results indicate that forest regeneration after three years does not differ between cut-to-length-processing and a whole-tree system, at least in terms of new cohort composition and growth. The results show also that intolerant species, including aspen and paper birch do regenerate in partially-harvested RMZ's, but their density, growth, and perhaps long-term survival, are greatly reduced relative to adjacent clearcut uplands. Specifically for aspen, above-ground biomass of suckers in RMZ's with 10 m^2/ha (44 ft^2/ac) residual overstory averaged only 64% of sucker biomass in the upland. Finally, high density of non-tree woody species (hazel, mountain maple) in partially-harvested RMZ's, along with low density of later successional tree species, makes long-term predictions of future overstory composition in the partially-harvested RMZ's difficult.

Effects of Riparian Management on Vegetation Variables: Herbaceous Plant Diversity, Coarse Particulate Organic Matter Flux, And Forest Regeneration

Introduction

Plant communities in forested riparian areas provide a number of ecological services. Broadly defined, one of these services is maintenance of biological diversity. For instance, riparian plants and plant communities can make important contributions to biological diversity by providing unique and heterogeneous habitat within watersheds (Brinson and Verhoeven 1999; Naiman et al. 2000). Many researchers believe that the contribution of riparian areas to overall plant diversity often exceeds the proportion of the landscape they occupy (Ferreira and Stohlgren 1999; Brinson and Verhoeven 1999; Crow et al. 2000); yet empirical tests of this hypothesis are rare. Those studies that have examined the contributions of riparian areas to plant diversity often have focused only on floodplains (e.g., Salo et al. 1986; Nilsson et al. 1994; Ferreira and Stohlgren 1999) and have ignored the potential contributions of other valley floor landforms to plant diversity. Because disturbance and hydrologic gradients vary depending on valley geomorphology, one would anticipate that in some landscapes valley floor landforms outside of the floodplain would contribute greatly to overall plant diversity. This is an important consideration when delineation of riparian management zones (RMZ) is based on fixed-width distance from the stream bank, without regard to geomorphic location within a stream valley. As such, an RMZ may fail to capture the functional extent of riparian services, including contributions to biological diversity, if non-floodplain valley floor landforms (terraces, hillslopes) fall outside of the RMZ boundary.

Riparian plants also are central to sustaining functional connections between forests and aquatic ecosystems through contributions of organic matter to streams (Vannote et al. 1980). Allochthonous organic matter, especially coarse particulate organic matter or CPOM, is an important energy source for many benthic invertebrates, particularly in small-order streams (Wallace et al. 1997). Annual flux of allochthonous CPOM into a stream from the adjacent riparian forest can be large, exceeding 500 to 600 $g*m^{-2}*yr^{-1}$ (Webster et al. 1990).

Riparian forests also are important ecologically because of the timber resources they support. This may reflect high productivity of riparian forests on fertile floodplains, or the occurrence of species that are rare elsewhere in the landscape (Palik et al. 1999). In water-rich regions riparian forests are valued simply because a large percentage of the timber resource is adjacent to water. For instance, in Minnesota 10% of commercial forests occur within 60 m of a lake or stream (Laursen 1996).

Ecological services of riparian forests are vulnerable to degradation by management practices. Riparian management guidelines are designed to protect ecological services of riparian areas, while allowing utilization of and management for timber resources. Current Minnesota management guidelines for riparian areas focus on width of the riparian management zone and amount of residual overstory to retain after harvest (Minnesota Forest Resources Council 1999). The primary intent of these guidelines is protection of water quality, but protection of other functions, including biological diversity and organic matter flux, is also considered.

Unfortunately, the efficacy of management guidelines at protecting riparian services is often untested. Additionally, the ability to manage riparian forests for timber, while adhering to the guidelines is questioned. This is of particular concern for intolerant species; management of aspen (*Populus grandidentata, P. tremuloides*) is an example. There is concern that adequate aspen regeneration is not possible with retention of even modest amounts of residual overstory. This concern agrees with findings

from a study which reported a reduction in aspen sucker growth of 35 to 45 percent with as little as 3.3 -3.4 m^2/ha (10-15 ft^2/ac) of residual overstory (Perala 1977).

It is examination of trade-offs between protection of ecological services provided by riparian vegetation and sustainability of timber management in riparian areas that is the intent of this research. Specific objectives and questions include:

Obj. 1. ***Quantifying the contribution of headwater riparian areas to herbaceous plant diversity in a forested watershed.***

Question 1. *How do valley floor landforms outside of the floodplain contribute to diversity?*
Question 2. *At what scale (within reach, among reach, or watershed) are diversity contributions most evident?*
Question 3. How often does an RMZ capture the full extent of increased riparian plant diversity?

Obj. 2. ***Quantifying coarse particulate organic matter flux to headwater streams from harvested riparian management areas.***

Question 1. *How is overhead and lateral litter flux affected by clearcut forest harvest outside of an uncut RMZ?*
Question 2. *How is litter flux affected by partial harvest within an RMZ?*

Obj. 3. ***Quantify tree species regeneration dynamics, particularly aspen, under riparian management treatments that include retention of overstory competitors.***

Question 1. *What tree species dominate the new cohort following partial harvest of an RMZ?*

Question 2. *How is aspen regeneration affected by a residual overstory in a partially harvested RMZ?*
Question 3. *How does harvest technology affects regeneration dynamics?*

I address these objectives and questions within the context of a controlled management experiment. The experiment compares responses of riparian and aquatic variables among RMZ management treatments that differ in amount of residual basal area and method of harvest.

Methods

Study Site Location and Experimental Design

The study area consists of a series of small watersheds drained by three small headwater streams (Little Pokegama Creek, Pokegama Creek North, Pokegama Creek South, Jack Irving Creek) in north central Minnesota, near Grand Rapids. The study area is entirely on UPM Blandin Paper Company lands. The forest ecosystems of the study area are mixed northern hardwoods-aspen occurring in the Sugar Hills end moraine. Soils are generally well -drained, fertile loams.

The experiment is a randomized block design, with 4 treatments replicated in each of 3 blocks. In 1996, 12 treatment stands were established, each 4.8 ha (12 ac) in size (2.4 ha on each side of the stream). The upland portions of 9 stands were clearcut, using either a cut -to-length logging system (CTL) or a

whole-tree system (WT) consisting of a feller/buncher and grapple skidder. The riparian portions of 6 of the 9 stands were cut, using the same system as in the upland, to a residual basal area of 10 m^2/ha (44 ft^2/acre). The riparian areas of the remaining 3 cut stands (uplands cut) were left intact (riparian control). The last 3 stands are complete controls, with no harvesting in the uplands or riparian areas. In each stand, the riparian area consists of a 60 m-wide strip (200 ft) centered on the stream (30 m on each side). The length of stream contained in each stand ranges from 135 to 200 meters (445-650 ft). Harvesting was completed in late summer-fall, 1997.

Vegetation Sampling

Transects were established in all stands perpendicular to stream direction at the transect. The number of transects ranges from 4 to 8 per stand. Permanent sample points are established along the transects, centered on landform (e.g. floodplain, terrace, hillslope upland). The numbers of points per stand ranged from 16 to 42, depending on the number of transects and landforms along a transect. At each point, herbaceous, seedling, shrub, sapling, and tree data were recorded as indicated in Table 1. Pre-harvest vegetation data were collected in summer 1997. Post-harvest data were collected in 1998, 1999 and 2000.

Table 1. Sampling methods used for each vegetation layer.

Vegetation layer	Sampling method
Herbaceous	0.5 m^2 plot at each point, ocular estimate of coverage by species (% cover); 6 classes used, class 1 = trace-1% through class 6 = 60-100%
Seedling	0.5 m^2 plot established at each point; # of seedlings for each species recorded.
Shrub	1.5 m radius circular plot established at each point; # of stems by species recorded.
Sapling and tree	Point-quarter sampling used, centered on each point.

CPOM flux to the streams was sampled in 1997, 1998 and 2000, using overhead and lateral litter traps located in the following treatments only; control, riparian control, basal area reduction using the whole-tree system. Five overhead and four lateral traps were located in or immediately adjacent to the stream in each stand. Overhead traps consist of wire tomato cages lined with a removable funnels made of nylon netting. Traps have a surface opening size of ~0.19 m^2 (2 ft^2). The legs of the cages are pushed into the ground or into lengths of PVC pipe that are then anchored into the ground or streambed. Lateral traps consist of a rectangular wooden box (46 cm (18 in) long by 20 cm (8 in) by 20 cm (8 in) that is open on two of the three long sides. One opening is covered with wire mesh. The box is placed on the ground with the opening oriented away from the stream. Traps were emptied periodically during each year and the contents oven dried, sorted to major life forms (wetland woody plants, upland woody plants, other), and weighed.

Analyses

Herbaceous plant diversity- These analyses utilize pre-treatment herbaceous plant data only. While collected within the context of the experimental design, they do no rely on the treatment structure for interpretation. Prior to analyses, individual sample plots along each transect were pooled into the following landform classes: floodplain, valley floor, or upland. The valley floor class includes low and high terraces, as well as slopes between terraces or between a terrace and the upland. I pooled these landforms into one class because transects and reaches did not consistently contain each of these landforms. Analyses of each landform class were conducted at three scales, including within individual reaches, among reaches, and at the watershed scale.

I summarized herbaceous diversity as species richness, the most direct and simplest way to measure species diversity (Whittaker 1972). Additionally, I calculated Jaccard's coefficient to quantify species overlap or turnover among landform classes. Jaccard's coefficient (J) was defined as $J = A/(A+B+C)$ where A = the number of species found in both paired landform classes, B = species in landform class 1 but not in landform class 2, and C = the number of species found in landform class 2 but not in landform class 1. The similarity coefficient ranges from 1.0 for complete overlap to 0.0, indicating no overlap.

I tested for differences in species richness among landforms using a complete block repeated measure ANOVA (Gill 1978, SAS Institut e 1989), where individual transects or reaches are the blocks and landform is the repeated measure. A repeated measure model is appropriate because we sampled the riparian ecotone at multiple locations in space and these locations are always arrayed in a non-random pattern (i.e., a floodplain is always positioned before a fluvial terrace, which is always positioned before the upland). I formulated a set of contrasts to test the following hypotheses at two scales including within individual reaches and amo ng reaches: 1) floodplains have higher species richness than the adjacent uplands; and 2) valley floor landforms have higher species richness than the adjacent uplands. I calculated the power of each ANOVA. I assumed contrasts with a power of 0.80 to have a sufficient sample size and requisite statistical power to detect differences among the landform classes.

I pooled data from each reach to examine differences in species richness among landforms at the watershed scale. From these pooled data, I estimated species-area curves for each landform class using the jackknife method described in Palmer (1990). Species richness (R) was calculated as:

$$R = S + r^1(n-1)/n, \qquad (1)$$

where S = the observed number of species, r^1 = the number of species occurring in one sample unit, and n = the number of sample units. This nonparametric sampling approach produces a more accurate and less -biased estimate of species richness when sub -sampling a larger area (Palmer 1990). I used PC -ORD software (McCune and Mefford 1999) to generate the species-area curves and to calculate the variance associated with each level of the species-area curve. I used these results to plot 95% confidence intervals (CI=1.96$\pm$sqrt[variance]) for each curve. I assumed species richness was si gnificantly different among different landform classes when confidence intervals did not overlap at or near the asymptote of the species -area curve. Additionally, I fit a regression line to each of the species -area curves using the natural log -log transformation of:

$$S=cA^z, \qquad (2)$$

where S is the expected number of species in area A, z is the instantaneous rate by which species richness increases with an incremental increase in area, and c is the expected number of species in a unit area (Arrhenius 192 1). I then compared z and c among curves within each valley type.

Finally, I overlaid the RMZ boundary in each stand on the distribution of landforms to determine the percentage of times a 30 m (100 ft)-wide RMZ captured areas of high herbaceous species diversity along the riparian ecotone.

CPOM Flux-I used a randomized block ANOVA to compare coarse particulate organic matter flux among treatments (control, riparian control, whole -tree harvest). The ANOVA's were run separately for overhead and lateral CPOM and by year. Variances were judged to be homogeneous in all cases. I used a one -sided Dunnett's test to contrast each of the two harvest treatments to the control. The hypothesis tested was that treatment reduced the flux of CPOM into the stream, r elative to the control.

Tree Regeneration-I analyzed regeneration data using detrended correspondence analysis (DCA) and ANOVA. I used DCA to characterize and contrast composition of tree regeneration among treatments. I ran separate ordinations usin g pre- and post-harvest data. Moreover, I separated the data into large ($\geq$ 1 m tall and < 2.5 cm diameter at 1.4 meters) and small (< 1 m tall) regeneration classes and ran analyses separately on each class. Input data for all analyses consisted of a sta nd (4 treatments x 3 reps = 12) by species matrix of square-root transformed stem densities. Only species occurring in at least three stands were included in the DCA analyses.

I compared various regeneration variables (stem densities, stem biomass, stem heights) among the four riparian treatments (control, riparian control, whole -tree harvest, cut-to-length harvest) using one -way ANOVA. Above-ground biomass (stems+leaves) of large regeneration (> 1m tall, < 2.5 cm diameter at 1.4 m) was determined using equations in Perala and Alban (1994). Examination of errors indicated that blocking did not improve the results of the ANOVA's, so I report results of completely randomized models. I assessed variance structure for each analysis and in some cases t ransformed the original data (square root, log) to better meet the assumption of homogeneous variance. I used orthogonal contrasts to test for differences among the following treatment combinations: i) control vs. pooled treatments (riparian control, cut-to-length, whole-tree); ii) riparian control vs. pooled riparian harvest (CTL, WT); and CTL vs. WT.

I used paired t-tests to compare responses between harvested RMZ's and the adjacent upland. For these analyses, I pooled the six harvested RMZ's (3 CTL, 3 WT). In some cases, data were transformed using log or square root transformations to better meet the assumption of normality.

Results

Herbaceous plant diversity

Within-Reach scale

There was considerable variability in species richness amon g landform classes in the study area (Table 2). The ANOVA results reflect this variability, as few significant differences at the individual reach scale were detected. Only two of the 11 individual reaches showed a significant difference in species richn ess between the floodplain and the uplands, or the valley floor and the upland ($p < 0.05$; Table 2).

Although there were few differences in species richness at the individual reach scale, there was significant species turnover among landforms (Table 3). Only reaches 8 and 9 had less than 50% species turnover from one landform class to another.

Among-Reach scale

When all individual reaches were pooled, there was a significant difference in species richness across the riparian ecotone ($p < 0.01$). Mean richness ($\pm$ 1 SE) of the floodplains was 7.9 (0.7) species per 0.5 m^2, compared to 5.7 (0.4) species per 0.5 m^2 in the adjacent upland (Table 2). Richness of the valley floor and the upland did not differ (Table 2). There was a different pattern in spec ies turnover among landforms, compared to the within -reach analysis, when individual reaches were pooled (Table 3). There was high turnover between the floodplain and the upland, but the valley floor and upland landforms had similar species compositions (species turnover < 50%; Table 3), compared to high turnover between these landforms when examined at the reach scale.

Table 2. Mean (± 1 SE) species richness and ANOVA results by landform within and among reaches.

Reach by Stream	Landform Class [a]			ANOVA[b]			
	FP	**VF**	**UP**	**n**	**F**	**FP v. UP**	**VF v. UP**
Reach 1[§]	5.2 (0.7)	3.1 (0.3)	3.3 (0.5)	6	8.07	9.17*	0.02
Reach 2	5.6 (1.0)	4.6 (0.4)	4.1 (0.6)	7	0.92	0.89	2.93
Reach 3[§]	6.7 (1.0)	5.0 (0.5)	4.6 (0.5)	7	2.69	2.95	0.68
Reach 4[§]	6.8 (0.8)	5.4 (0.5)	5.2 (0.6)	6	1.84	2.42	0.21
Reach 5	7.1 (1.0)	6.0 (0.7)	7.4 (0.7)	7	0.79	0.08	1.75
Reach 6	7.7 (1.0)	5.9 (0.4)	5.7 (1.2)	6	1.06	1.02	0.21
Reach 7	9.4 (0.7)	7.3 (0.4)	6.6 (1.4)	5	2.63	3.73	0.51
Reach 8	6.5 (0.8)	6.0 (0.5)	5.1 (0.6)	6	0.91	1.11	0.88
Reach 9	8.8 (1.5)	8.0 (0.1)	6.4 (0.6)	4	1.70	2.46	5.40
Reach 10	10.3 (0.5)	9.2 (0.7)	8.2 (1.2)	7	1.37	2.79	0.55
Reach 12[§]	12.8 (0.7)	7.1 (0.7)	5.8 (1.2)	6	26.15	16.71**	1.31
Among Reaches	**7.9 (0.7)**	**6.2 (0.5)**	**5.7 (0.4)**	**11**	**15.74**	**17.42****	**3.73**

[a]Landforms include: FP – floodplain; VF –other valley floor landforms (terraces and slopes); UP –upland. [b]Analysis of variance; * p <0.05; ** p <0.01. Values are landform comparisons are F -statistics. [§] power of ANOVA >0.80. Note: Reach 11 was excluded from analysis because its floodplain was too narrow to sample.

Table 3. Species turnover (%) between landform classes within and among reaches.

Reach by Stream	Species Turnover (%)[a]	
	VF-UP	FP-UP
Reach 1	81.1	78.7
Reach 2	64.9	97.5
Reach 3	65.2	92.9
Reach 4	64.3	94.1
Reach 5	64.4	86.0
Reach 6	49.8	76.6
Reach 7	68.3	97.4
Reach 8	36.8	77.2
Reach 9	41.3	79.5
Reach 10	98.3	73.7
Reach 12	92.9	94.0
Among Reaches	**47.0**	**80.3**

[a]Landforms include: FP – floodplain; VF –other valley floor landforms (terraces and slopes); UP–upland landforms

Watershed scale

Differences in species richness among landforms at the watershed scale were similar to those seen at the reach scale (Figure 1). The floodplain and valley floor had approximately 67 and 69 species, respectively, versus 60 for the uplands (Figure 1; Table 4). Although both the floodplain and valley floor landforms have similar initial species richness when a single 0.5 m^2 plot is examined (c = 25.04 and 23.49, respectively), the slope of the floodplain species-area curve is higher than the valley floor (z = 0.34 and 0.26, respectively), suggesting that initially, there is a higher rate of increase in ground-flora species per unit area on the floodplain than the other valley floor landforms (Table 4). Additionally, the floodplain and

upland landforms appear to increase in richness at similar rates (slopes are the same, Table 4), however, the floodplains have higher species richness when a single plot is examined (intercepts differ, Table 4).

Table 4. Jackknife estimates of species richness and regression coefficients.

Landform	Jackknife Estimate of Species Richness	Regression Coefficients c (intercept)	Regression Coefficients z (slope)
Floodplain	67.0	25.04	0.34
Valley floor	69.7	23.49	0.26
Upland	60.5	18.86	0.34

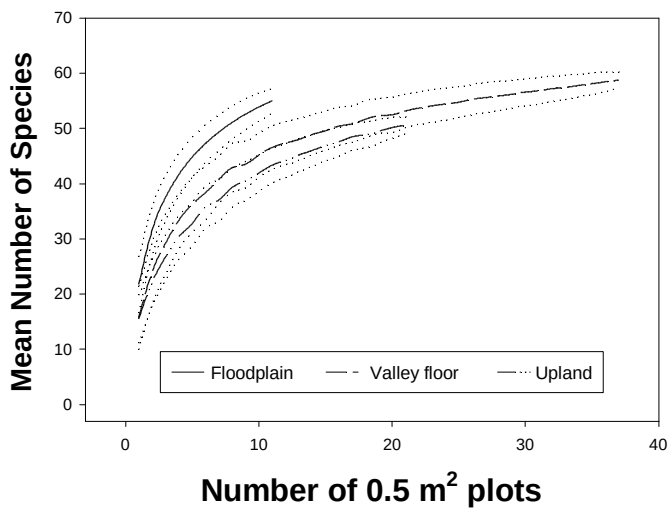

Figure 1. Species-area curves by landform. Dashed lines are 95% confidence intervals.

RMZ Boundary

The 30 m (100 ft) RMZ boundary fell in the geomorphically defined upland on about 60% of the transects (Figure 2). For the remaining 40% of the transects, the boundary occurred on either a t errace or slope. No boundaries occurred on the floodplain (Figure 2).

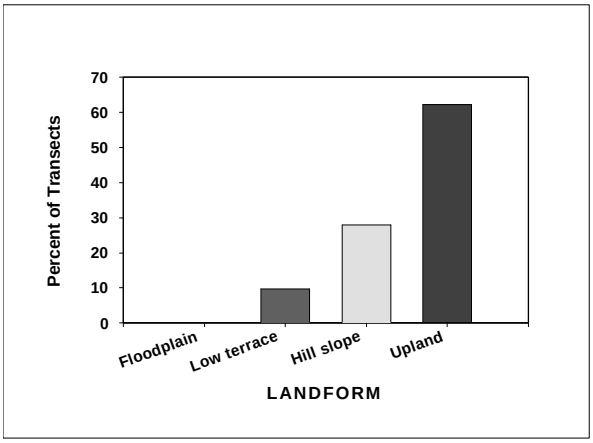

Figure 2. Geomorphic location of the 30 m (100 ft) RMZ boundary in stands of the study area.

Change in Overstory Structure

Pre-harvest basal areas were similar for all RMZ's, averaging 30 m^2/ha (120 $ft.^2$/ac) (Figure 3). Sugar maple (*Acer saccharum*), paper birch (*Betula papyrifera*), basswood *(Tilia americana)*, and aspen (*Populus tremuloides* and *P. grandidentata*) dominated the pre-harvest overstory (data not shown). Post-harvest basal areas of treated RMZ's (CTL and WT) averaged about 10 m^2/ha (44 $ft.^2$/ac) (Figure 3). Relative post-harvest overstory composition was similar to pre- harvest composition. In the harvested RMZ's, mean aspen basal area was reduced from 3 m^2/ha (13 ft^2/acre) to 0.8 m^2/ha (4 $ft.^2$/ac). Composition of the adjacent uplands was similar to the RMZ's (data not shown). However, total basal area (36 m^2/ha) and aspen basal area (13 m^2/ha) was higher in the upland than in the RMZ's.

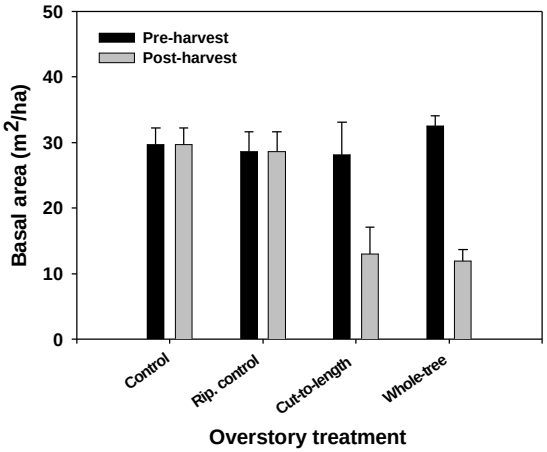

Figure 3. Pre- and post-harvest basal areas of study treatments.

CPOM flux

Overhead CPOM flux to the streams varied significantly among treatments in all years (Table 5; Figure 4). Each year, partially harvested RMZ's had lower overhead CPOM flux than the control (Figure 4). CPOM flux from the riparian control treatment (RC) was significantly lower than the full control in 1997 only (Figure 4, Table 5). Lateral CPOM flux declined from the full control to the riparian control to the partially harvested RMZ each year (Figure 4), but the differences were not significant (ANOVA results not shown).

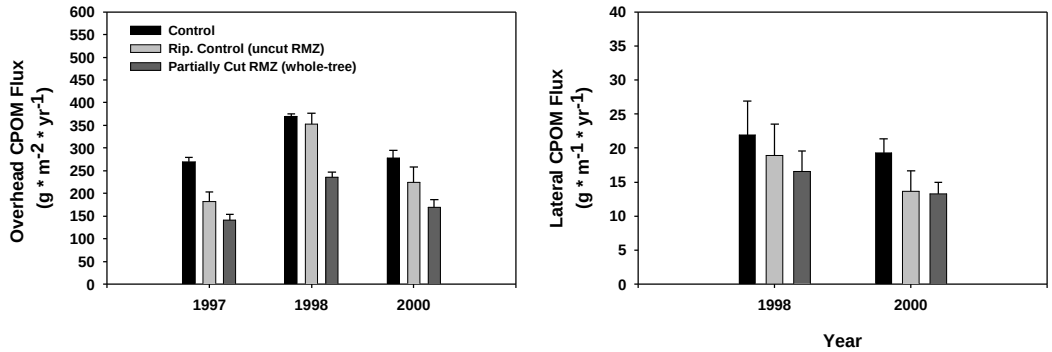

Figure 4. Overhead and lateral CPOM flux to the study streams.

Table 5. ANOVA results for overhead CPOM flux to study streams.

Year	Source	DF	MS error	F	Minimum significant difference	Difference between means	P
1997	Block	3	234.92				
	Treatment	3	8825.95	14.39			0.015
	Error	4	613.25				
	Contrast 1[a]	1			52.78	72.11	0.05
	Contrast 2[a]	1			" "	106.24	0.05
1998	Block	3	791.02				
	Treatment	3	15905.77	22.30			0.007
	Error	4	713.20				
	Contrast 1[a]	1			56.92	16.86	>0.10
	Contrast 2[a]	1			" "	133.70	0.05
2000	Block	3	2511.95				
	Treatment	3	8678.53	6.71			0.053
	Error	4	1293.99				
	Contrast 1[a]	1			76.67	53.56	>0.10
	Contrast 2[a]	1			" "	107.57	0.05

[a]Dunnett's test: contrast 1= control vs. riparian control; contrast 2= control vs. partially -harvest RMZ.

Forest regeneration

Compositional Responses to RMZ Treatments

The DCA joint plots show that composition of both the small and large regeneration classes were similar among treatments before harvest (Figure 5 and 6). Moreover, the RMZ's did not differ appreciable from the adjacent uplands (Figure 5 and 6). Three years after harvest, composition of the small regeneration class still did not differ appreciably among RMZ treatments or between the RMZ and upland (Figure 5).

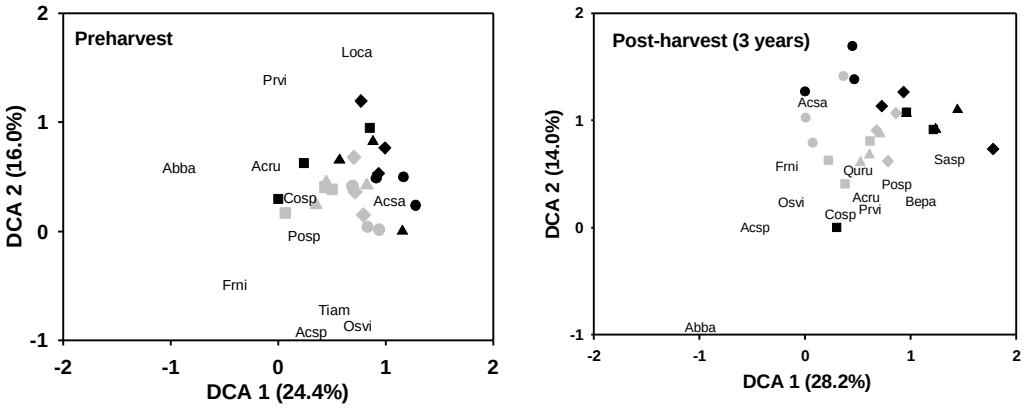

Figure 5. DCA joint plots of small regeneration composition. Dark symbols= upland plots, gray symbols = riparian plots. Circles=control; square=riparian control (RC); diamond=whole -tree (WT); triangle=cut-to-length (CTL).

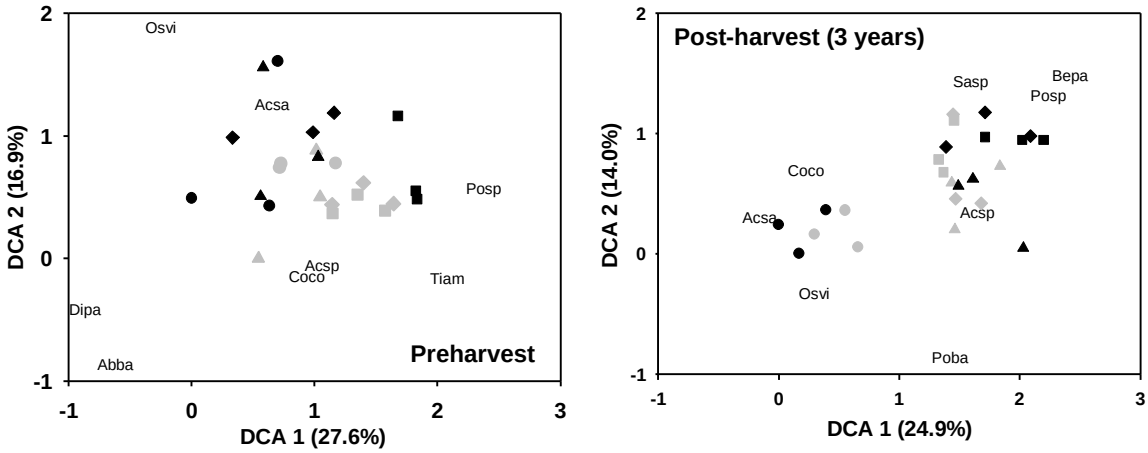

Figure 6. DCA joint plot of large regeneration composition. Dark symbols= upland plots, gray symbols= riparian plots. Circles= control; square= riparian control (RC); diamond= whole -tree (WT); triangle= cut-to-length (CTL).

The only notably differences (that were consistent among the replicates for a treatment) in the small regeneration class were more sugar maple (Acsa in the figure) in the control stands (both upland and RMZ) and more paper birch (Bepa) in the harvested stands (both RMZ and upland) (Figure 5). In contrast, composition of the large regeneration class differed appreciably between the control plots (both RMZ and upland) and the harvested plots (Figure 6). The primary differences were more aspen (Posp), paper birch (Bepa), and mountain maple (Acsp) in the treated plots and more hazel (Coco) in the control plots (Figure

6). Composition did not differ appreciably between the RMZ's and the upland for either the control plots or treated plots (Figure 6).

Structural Response to RMZ Treatments

Total regeneration density varied considerably over the four -year study period. Small regeneration (< 1 m tall) density declined steadily over time in all RMZ treatments (Figure 7). There was no significant difference in density in 2000 (the last year of post -harvest sampling) among treatments (ANOVA results not shown). Density of large regeneration (≥ 1 ma tall, < 2.5 cm diameter at 1.4 meters) remained relatively constant in the full control, but increased after cutting in the three harvest treatments (Figure 7). Densities in 2000 were significantly higher in the harvested treatments than the control (Figure 7; Table 6). Further, the RMZ harvest treatments (CTL, WT) had higher densities than the riparian control (RC) (Figure 7; Table 6). Large regeneration de nsities in the two harvest treatments did not differ significantly (Figure 7, Table 6). Differences among treatments in above -ground biomass of large regeneration in 2000 were similar to differences in stem densities accept that the two RMZ harvest treatm ents differed significantly (Figure 8; Table 6).

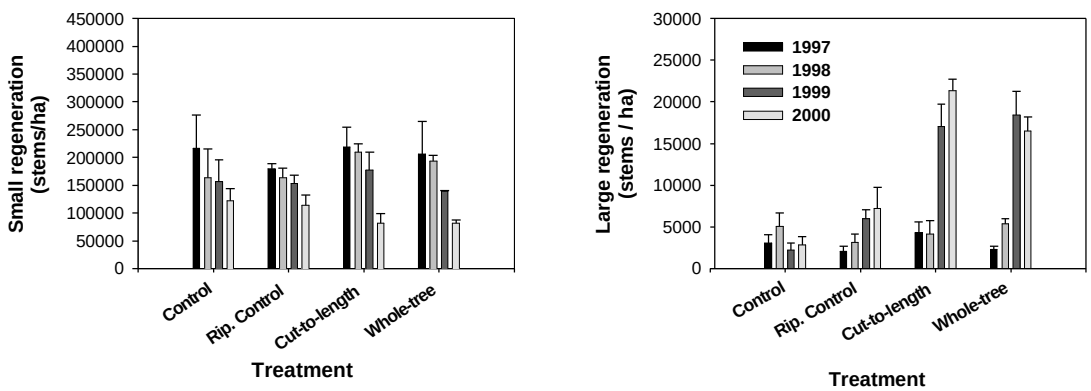

Figure 7. Regeneration density over time in the four RMZ treatments.

In the small regeneration class, only two species (paper birch, sugar maple) showed consiste nt (among replicates) density responses to the RMZ treatments by 2000 (Figure 9). Of these, only sugar maple densities differed significantly (marginally) among treatments (Table 6), having decreased densities in three RMZ treatments, compared to the control. The two harvested RMZ treatments did not differ significantly (Table 6). Four taxa in the large regeneration class including, aspen, paper birch, hazel, and mountain maple, showed consistent density responses to treatments by 2000 (Figure 10). Of t hese, only aspen (marginally) and paper birch has significant responses (Table 6). Both taxa had higher densities in the two RMZ harvest treatments than in the riparian control, while the two harvest treatments did not differ (Figure 10; Table 6).

Table 6. ANOVA results for RMZ treatment comparisons of selected regeneration variables.

Variable	Source	DF	MS error	F	P
Large regeneration density	Treatment	3	214516297.65	23.13	0.0003
	Error	8	9275425.95		
	Contrast 1[a]	1	334262055.64	36.04	0.0003
	Contrast 2[a]	1	273882267.55	29.53	0.0006
	Contrast 3[a]	1	35404569.75	3.82	0.0865
Large regeneration biomass	Treatment	3	1913374.79	52.58	0.0001
	Error	8	36392.49		
	Contrast 1	1	1861023.45	51.14	0.0001
	Contrast 2	1	1368701.67	37.61	0.0003
	Contrast 3	1	3757456.42	103.25	0.0001
Aspen density (large)	Treatment	2	1334.77	3.19	0.1140
	Error	6	418.86		
	Contrast 1		na[b]		
	Contrast 2	1	2600.82	6.21	0.0470
	Contrast 3	1	68.73	0.16	0.6995
Paper birch density (large)	Treatment	2	535536.54	5.24	0.0482
	Error	6	102135.45		
	Contrast 1		na[b]		
	Contrast 2	1	706642.53	6.92	0.0390
	Contrast 3	1	364430.54	3.57	0.1078
Sugar Maple density (small)	Treatment	3	2730187595.80	3.25	0.0812
	Error	8	840742771.00		
	Contrast 1	1	786916807.10	9.36	0.0156
	Contrast 2	1	6939709577.20	8.25	0.0207
	Contrast 3	1	320413998.80	0.38	0.5542

[a]Contrast 1: control vs. treatment (RC, CTL, WT); Contrast 2: RC vs. Harvest (CTL, WT); Contrast 3: CTL vs. WT. [b]Contrast 1 not performed due to missing values for some or all replicates.

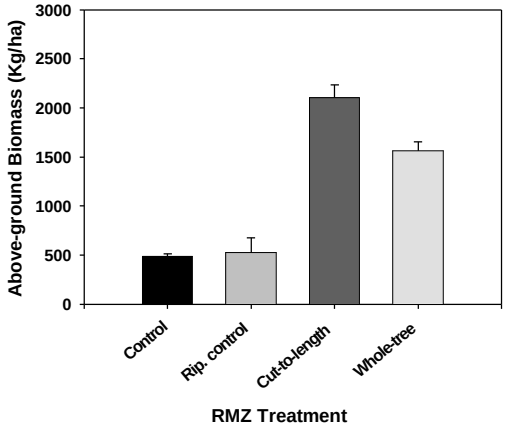

Figure 8. Above-ground biomass of large regeneration in 2000.

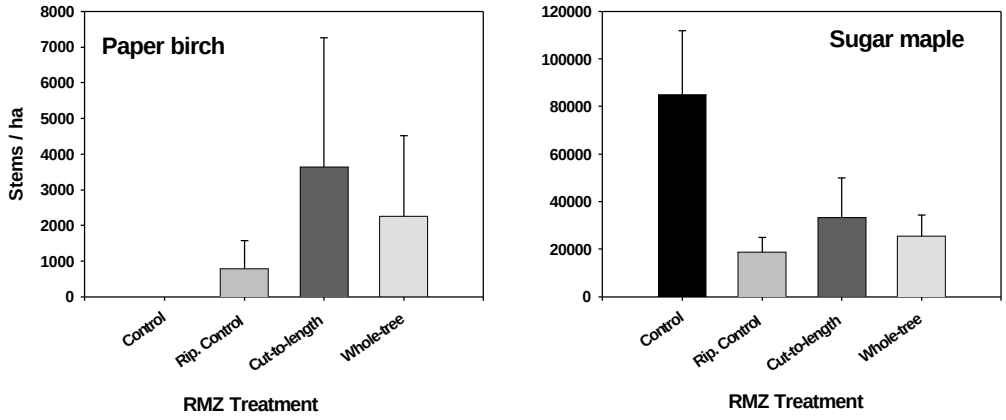

Figure 9. Paper birch and sugar maple densities in 2000 (small regeneration size class).

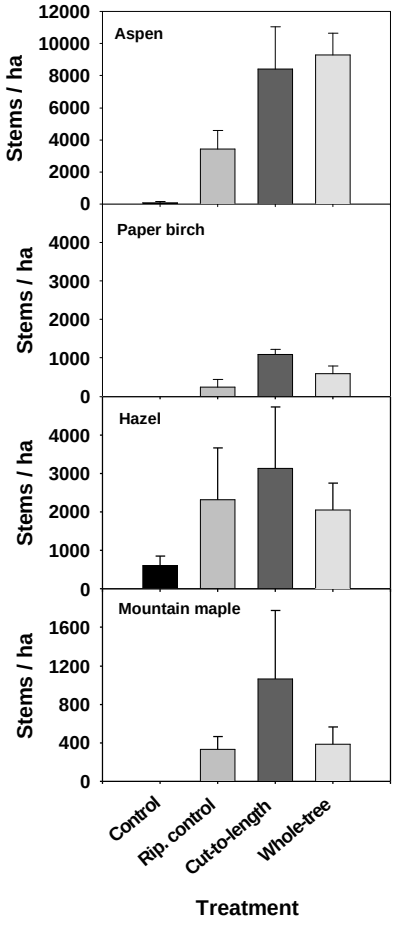

Figure 10. Large regeneration densities for selected taxa in 2000.

Mean aspen sucker ht (mean height of an individual sucker) in 2000 ranged from 160 cm to 180 cm among the RMZ treatments (control was excluded due to lack of suckers in this treatment), while mean aboveground biomass of suckers ranged from 65 grams to 95 grams (Figure 11). Both sucker height and biomass were higher in the two RMZ harvest treatments than the riparian control, but the differences were not significant (ANOVA results not shown). Further, there were no differences in sucker height or biomass between the two harvest treatments.

RMZ-Upland Comparisons of Regeneration Structure

Three years after harvest, total regeneration densities for the small and large size classes were higher in the uplands than in the adjacent partially harvest RMZ (Figure 12). However, the differences were not significant for the small size class (paired t -test: n=6, T=0.475, p=0.655) and only marginally significant for the large size class (paired t -test: n=6, T=2.071, p=0.093). Total above-ground biomass of large regeneration in the uplands was nearly twice that in the RMZ's (Figure 13). The difference was significant (paired t -test: n=6, T=2.765, p=0.040)

No species in the small regeneration class had any consistent or significant difference in densities between the partially harvested RMZ and the adjacent upland (data not shown). In the large regenerat ion class, aspen and paper birch had significantly higher densities in the upland (Figure 14) (paired t -tests: n=6, T=2.611, p=0.048 (aspen); T=2.418, p=0.060 (paper birch)).

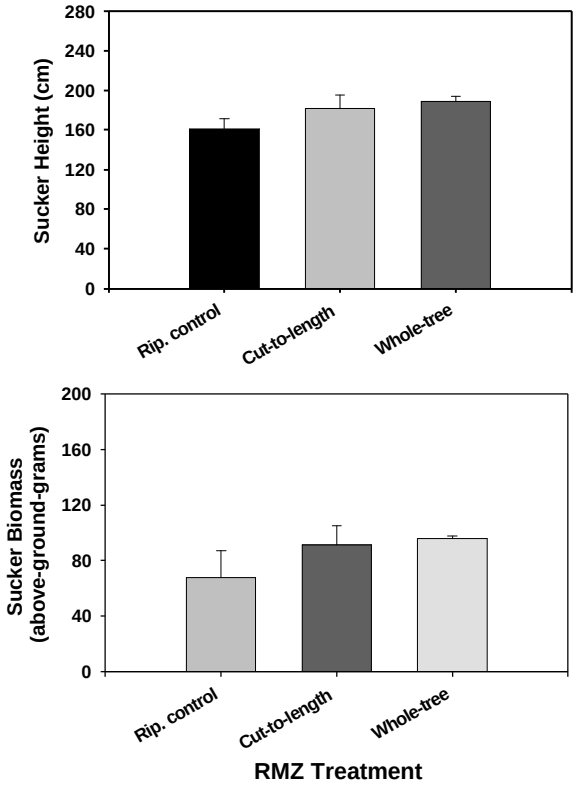

Figure 11. Mean aspen sucker height and above-ground biomass in 2000.

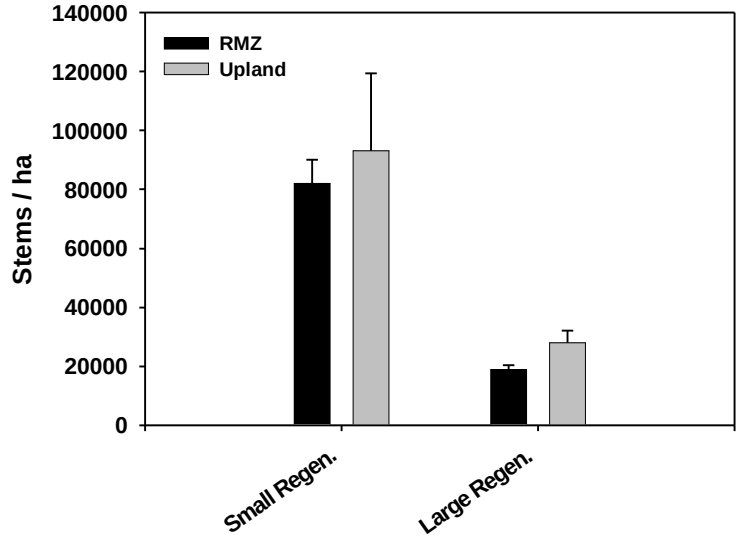

Figure 12. Regeneration densities in partially harvested RMZ's and the adjacent clearcut uplands in 2000, three years after harvest.

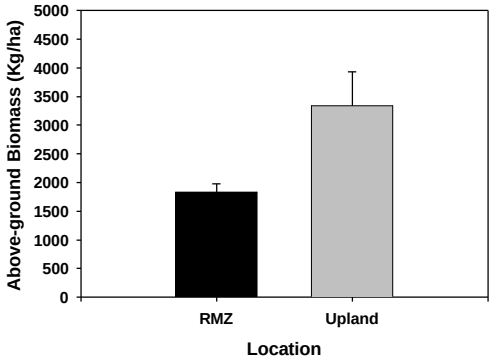

Figure 13. Above-ground biomass of large regeneration in partially harvested RMZ's and the adjacent clearcut upland in 2000, three years after harvest.

Both hazel and mountain maple had higher densities in the partially harvested RMZ's than the adjacent uplands (Figure 14). The difference was significant for hazel (paired t -test: n=6, T=3.466, p=0.018), but not for mountain maple (paired t -test: n=6, T=1.927, p=0.112).

Aspen sucker heights three years after harvest were not significantly different between the partially cut RMZ's and the clearcut uplands (paired t -test: n=6, T=0.888, p=0.415), averaging about 180 cm in both locations (Figure 15). Above-ground biomass of individual suckers averaged about 95 grams in the partially cut RMZ's and about 130 grams in the clearcut uplands (Figure 15). The difference was significant (paired t -test: n=6, T=15.979, p=0.0001).

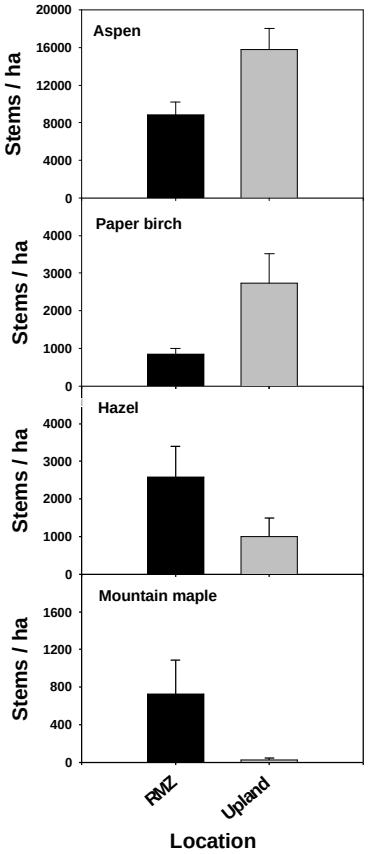

Figure 14. Densities of selected taxa in the large regeneration class in partially harvested RMZ's and the adjacent clearcut upland in 2000, three years after harvest.

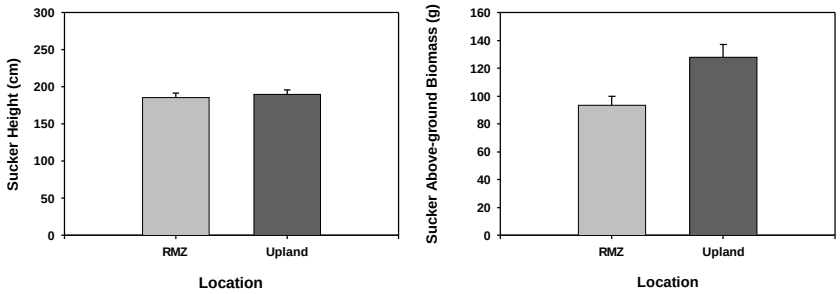

Figure 15. Mean heights and above-ground biomass of aspen suckers in 2000.

Discussion

Herbaceous Plant Diversity

There is little question that in many regions riparian areas, as a broad class of system, do contribute significantly to overall plant diversity of a watershed (e.g., Salo et al. 1986; Nilsson et al. 1994; Pollock et al. 1998; Ferreiera and Stohlgren 1999). Less is known about how individual riparian areas at the reach scale contribute to plant diversity, how landforms outside of the floodplain contribute, and at what scale these contributions become important. My results suggest that by themselves, individual riparian areas (as represented by an individual reach) may not significantly increase plant species richness

in the watershed. However, there does appear to be a cumulative effect of adding individual riparian areas together, as predicted by Brinson and Verhoeven (1999). The mechanisms responsible for these contributions are due largely to the high environmental heterogeneity associated with riparian areas when examined at the among reach and watershed scales, that are typically not present at the individual reach scale. For example, Spackman and Hughes (1995), Pollack et al. (1998), and Ferreira and Stohlgren (1999) all suggest that variability in duration, depth, and spatial extent of flooding may increase species diversity at the landscape scale by increasing landscape -scale heterogeneity. In other words, adding individual riparian areas together forms a longitudinal gradient of stre am valley conditions that increases environmental heterogeneity substantially above that occurring at small spatial scales. Thus, when riparian areas are viewed as the cumulative sum of all reaches in a watershed, the contributions to overall plant species richness provided by riparian areas are significant. Moreover, it is at the watershed scale that valley floor landforms other than floodplains can contribute significantly to species richness.

Individual riparian areas or reaches do make import ant contributions to plant diversity, even at the within- reach scale, in ways other than species richness. My results demonstrate that the composition (and likely the structure) of the ground-flora community often changes dramatically across the riparian ecotone. In all but a few instances, I observed significant species turnover from the floodplain to the higher terraces and slopes to the uplands. As both overstory and understory structure are important factors regulating the type and quality of habita t for many animals (Pregitzer et al. 2000), this species turnover across the riparian ecotone may be an important feature that regulates the diversity of other organisms, especially avian species.

In this study, the 30 m (100 ft)-wide RMZ boundary fell in the geomorphically defined riparian area 40% of the time. All of these occurrences were on either a terrace or slope, but never a floodplain. The significance of this for plant diversity, in terms of potential for impact from forest management, depends on the scale of consideration and the measure of diversity used. The contribution of terraces and slopes to increased species richness was only evident at the watershed -scale. Assuming no management in the RMZ, this suggest that direct impacts from forest management to ground-flora species richness will be minimal on an individual reach by reach basis, since at this scale increased richness was only evident on the floodplains, but the RMZ boundary never fell on this landform. In contrast, the watershed-scale contribution of terraces and slopes to increased species richness suggests that concern must be given to cumulative effects of management impacts on these landforms. In other words, one can expect a loss of species richness, on terraces and slopes, when a threshold level of disturbance in the watershed occurs. In contrast, generally high species turnover across the riparian ecotone, even within individual reaches, suggests that direct impacts to terraces and slopes, when they fall outside of the RMZ, will result in a loss of species that are unique to that riparian area. This discussion assumes no management within the RMZ. Management within the RMZ itself will greatly increases the potential for direct impacts to herbaceous plant diversity, particul arly if the floodplain harvested.

Coarse Particulate Organic Matter Flux

My results for CPOM flux to the streams are equivocal. In all years, overhead and lateral CPOM flux decreased from the control, to the riparian control (RMZ uncut, u pland clearcut), to the partially cut RMZ (with upland clearcut). However, the differences were consistently significant only for overhead CPOM when contrasting the control to the partially cut RMZ. The contrast between the control and the uncut RMZ (with upland clearcut) was significant only in 1997, immediately after harvest. Differences in lateral flux were never significant. I believe the trends observed are suggestive of actual patterns, but that inadequate replication or sub-sampling may have limi ted statistical power in most cases. With this caveat in mind, the data do show that a 30 m-wide RMZ, cut to a residual basal area of 10 m^2/ha (44 ft2/ac),

reduces overhead flux of CPOM to a first order stream by 33 to 47% over the uncut forest for at least three years after harvest. Moreover, the data show that CPOM flux from an uncut 30 m-wide RMZ (with adjacent clearcut) may be reduced by 33% below the control for at least one year after harvest.

Qualitative discussion suggests that most CPOM entering streams come from within one-half tree height of the channel (O'Laughlin and Belt 1995), or 17-19 m (55-60 feet) in this study. My results suggest that nearly one-third of CPOM comes from beyond 30 m from the channel (as indicated by the significant reduction in CPOM flux from the uncut RMZ compared to the control in 1997), but that rapid reestablishment of leaf area in the clearcut forest beyond this distance may lessen this impact after only one year. My results also show that retention of 10 m^2/ha (44 ft^2/ac) of overstory basal area within the RMZ is not sufficient for sustaining CPOM inputs at pre-harvest levels. These results are important for two reasons. First, they indicate that the functionally defined riparian area extends beyond 30 m (100-ft); exceeding at least some of the RMZ widths suggested in the current Minnesota management guidelines (Minnesota Forest Resources Council 1999). Secondly, they show that the low end of the basal area retention range for even age management (20-80 ft^2/ac), as suggested by the guidelines, falls well below the level needed to maintain functional similarity (in terms of CPOM input) to the uncut forest.

Several points temper these results. First, CPOM reductions are temporary. As the new forest develops, inputs will increase. However, the timing of this recovery is unknown. Second, watershed condition must be considered when speculating on the importance of CPOM reductions. A single 60 m reach with substantial reduction in litter inputs may be inconsequential if little of the watershed is in a young forest condition. Finally, the level of CPOM input to the stream that is required to sustain aquatic food webs in a pre-harvest condition is unknown.

Forest regeneration

My results, three years after harvest, indicate that the riparian treatments increase density and above-ground biomass of regeneration in the RMZ, compared to the control, and that most of this increase in concentrated in four taxa, including aspen, paper birch, hazel, and mountain maple. With the exception of sugar maple seedling density, which was much higher in the RMZ control than the three treatments, the regeneration responses was seen only in the large size class. This is probably because most individuals of the dominant regenerating taxa had grown out of the small size class by 3 years after harvest. Many regeneration variables were higher in the riparian control (RMZ uncut, upland clearcut) than in the full control, but they were always less than the RMZ harvest treatments.

There were some appreciable differences in regeneration response to harvest method in the RMZ. Total density and aboveground biomass of regeneration was higher with cut-to-length processing than with whole-tree harvesting, possible because of less soil disturbance with the former. The increase with CTL was due largely to greater numbers of woody shrubs, including mountain maple and hazel. Aspen regeneration, including sucker density and aboveground biomass, did not differ with RMZ harvest treatment. However, overtime, increased shrub competition in the cut-to-length treatments may result in reduced regeneration, survival, and growth of overstory trees, particularly later successional species.

Results from the RMZ harvest treatments become more meaningful when compared to responses in the adjacent upland. Keep in mind that for these comparisons the effects of overstory treatment (partial cut vs. clearcut) is confounded with location (riparian area vs. upland). I cannot distinguish between the influences of the two factors due to the nature of the experimental design. Never the less, the comparisons are useful because in everyday practice, RMZ's and uplands are often managed just this way, with partial cutting in the riparian area and clearcuting outside of the RMZ.

In these comparisons, total regeneration density, three years after harvest, is about the same in the upland and in the partially cut RMZ (90,000 and 30,000 stems/ha for the small and large size classes, respectively). However, above-ground biomass of large regeneration in the upland was nearly double that of the RMZ's, suggesting much greater competitive inhibition of growth in the new cohort by the residual overstory in the RMZ.

There also were significant differenc es between the partially cut RMZ's and the clearcut uplands in densities of several individual taxa. Most notably, aspen and paper birch stem densities (large size class) were 2 and 3 times greater, respectively, in the upland than the RMZ. In contrast, hazel and mountain maple densities were higher in the partially cut RMZ's than the upland.

Aspen sucker density (9000 stems/ha) in the RMZ treatments are well below levels considered to reflect adequate stocking at age three (Stone et al 1999), where as densities in the upland (16,000 stems/ha) are within this range. Heights of aspen suckers did not differ between the clearcut uplands and partially cut RMZ's, but mean above-ground biomass of individual suckers were less in the latter. On average, su cker biomass was 36% greater in the upland, compared to biomass of suckers in the RMZ. This, combined with the two-fold greater sucker density in the upland, indicates that aspen fiber production is greatly reduced in the RMZ's compared to the upland.

The reduction in aspen suckering and growth are not surprising, others have shown similar trends with overstory retention (Perala 1977). The value of my results is that they quantify the specific amount of aspen growth reduction that can be expected with the level of overstory retention used in this study, about 10 m^2/ha (44 ft^2/ac). Moreover, these results point to one potential successional pathway for the RMZ's, namely continued development of a non-tree understory dominated by hazel and mountain maple, along with some aspen and paper birch. However, the long-term persistence of the latter two taxa is questionable. Moreover, it remains to be seen how important slower growing, later successional species will become overtime as the new cohort in the RMZ continues to develop.

Application to Riparian Management Guidelines

Originally, this study was intended to test aspects of riparian management guidelines as specified in *Protecting water quality and wetlands in forest management: best management practices in Minnesota* (Minnesota Department of Natural Resources 1995). Specifically, RMZ width was set at 30 m (100 ft) and target residual basal area was 5.7 m^2/ha (25 ft^2/ac). The actual residual basal area achieved was 10 m^2/ha (44 ft^2/ac). There are two reasons for this difference. First, the Sate of Minnesota Fisheries Division required a one crown-width-wide leave strip along the stream banks in the harvest treatments. Trees in this ~3 m (10 ft) wide strip were not tallied when leave trees where mark ed in the RMZ before harvest. The effect was to underestimate residual basal area in the treatments. Secondly, the harvester operator had difficulty reaching some of the trees designated for harvest because of the distribution of inoperable site conditions (steep slopes, seeps, wet soil) in the harvest units. This increased residual basal area over the target. The final mean residual basal in the harvest treatments, 10 m^2/ha (44 ft^2/ac), is more applicable to current riparian management guidelines for M innesota (Minnesota Forest Resources Council 1999). These guidelines call for a residual basal area of 5.7 -18 m^2/ha (25-80 ft^2/ac) on non-trout streams and 13.6 m^2/ha (60 ft^2/ac) for trout streams, when even-aged management is the objective. As summarize d below, my results suggest areas where the current management guidelines are inadequate, relative to RMZ width and residual basal area requirements.

1. For the type of system studied, specifically first and second order streams that are 1 -1.5 m wide (3-5 ft), flowing through constrained valleys, a 30 m (100 ft) -wide RMZ will not consistently capture the full extent of the riparian area as determined by geomorphology. Suggested RMZ widths for similar sized non -trout

streams in the current guidelines are 15 m (50 ft) and 30 m (100 ft), for even-aged and uneven-aged management, respectively. For designated trout streams, widths are 45 m (150 ft) and 60 m (200 ft) for even-aged and uneven-aged management, respectively. Thus, based on geomorphology alone, RMZ widths as applied to non-trout streams are insufficient, while widths as applied to trout streams are sufficient to capture the full extent of the geomorphically-defined riparian area.

2. Data on herbaceous plant species richness and turnover suggest that RMZ's narrower than 30 m (100 ft) will not consistently capture unique aspects of plant diversity in riparian areas for the type of system studied. At the scale of entire watersheds, terraces and hillslopes contribute unique aspects to plant diversity through increased species richness and distinctly different composition that the upland. These geomorphic surfaces fall on or outside of a 30 m (100 ft) boundary 40% of the time in the study system, making them susceptible to direct and indirect impacts from intensive harvesting and management.

3. Upland clearcutting adjacent to a 30 m (100 ft)-wide, partially-harvested RMZ's (10 m^2/ha; 44 ft^2/ac) reduced coarse particulate organic matter (CPOM) flux to the streams by 33-47% over an intact forest, at least for the first three post-harvest years of the study. Upland clearcutting adjacent to a 30 m (100 ft) uncut RMZ reduced CPOM flux to the stream by 5-33% over three years. These results indicate that the combination of RMZ width and residual basal area used in this study will not protect against all functional impacts of harvesting and that even an uncut RMZ adjacent to a clearcut can result in a functional change in the stream. However, these levels of CPOM reduction may have little corresponding impact on aquatic organisms at the scale of the RMZ. Moreover, I do not know what percentage of a watershed must be similarly impacted to affect a response in the aquatic system. Finally, the changes in CPOM flux are temporary and will decline as the new forest develops. I do not know if these short-term changes have lasting impacts on stream biota.

4. The current guidelines use terminology to suggest that an even-aged stand can include 5.7-18 m^2/ha (25-80 ft^2/ac) of residual basal area after a regeneration harvest. By definition even-aged management, or more properly single-cohort management, leaves little if any residual overstory after a regeneration harvest. By definition, it is simply not possible to manage for even-aged stands while maintaining the levels of residual basal area included in the current guidelines.

5. The larger question of relevance to guideline implementation is whether intolerant tree species, specifically trembling and bigtooth aspen, regenerate under suggested levels of residual basal area (5.7-18 m^2/ha; 25-80 ft^2/ac). Results from this study show that these species do regenerate under high levels of residual overstory (10 m^2/ha; 44 ft^2/ac) in 30 m (100 ft)-wide RMZ's adjacent to upland clearcuts. Some aspen suckering occurred in the unharvested RMZ's, indicating that some of the response in the partially-harvested treatments resulted from increased side-lighting from the adjacent clearcut. However, sucker densities after three years in the partially-harvested RMZ's were reduced by about 50%, relative to the adjacent uplands, and aboveground sucker biomass was only about 70% of that in the upland. These disparities will likely increase over time as the effect of overstory competition accumulates.

6. Three years after harvest, no additional tree species, beyond trembling aspen, had increased in density in response to RMZ harvest treatments. Rather, the partially-harvested RMZ had increases in densities of woody shrubs, including mountain maple and hazel, relative to the upland. This response, along with reduced and perhaps declining aspen regeneration importance, calls into question the long-term productivity and sustainability of the partially-harvested RMZ's. As such, it suggests the need for a comprehensive silvicultural plan for the RMZ's, focused on ensuring adequate regeneration, growth, and survival of overstory tree species. A one-time entry for a commercial harvest may be inadequate for insuring regeneration success and a desirable species mixture.

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Effects of Riparian Management on Coarse Particulate Organic Matter in Low Gradient Streams

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Effects of Riparian Management on Coarse Particulate Organic Matter in Low Gradient Streams

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Introduction:

Large wood is a critical component of small to medium streams, directly influencing stream geomorphology and many ecosystem properties and processes (Harmon *et al*.1984). Habitats created by large wood are varied (e.g., plunge pools, backwaters, eddies, interstices of debris dams and individual logs and are critical for fish as well as invertebrate species. They provide flow and predation refugia for fish, oviposition and pupation sites, a feeding platform for invertebrates, and a substrate for biofilm production (Sedell et al.1988, Shearer & Webster1988). Increased channel complexity due to large wood is linked to habitat and biotic diversity (Angermeier & Karr 1984) as well as increases in the retention of organic matter. Changes in retention of CPOM alter nutrient fluxes through the biota and affect biotic processes and communities. Land use changes (including forest harvest), through their effects on the riparian zone, alter detrital inputs and thereby stream community structure and function. Forest harvest often results in a long-term decrease in the quantity and quality of coarse particulate organic matter and standing stock, size, and orientation of large wood in the stream channel (Hogan 1987, Hedin *et al*. 1988, Murphy & Koski 1989, Golladay, et al. 1989, Ralph et al. 1994).

In forests of the Upper Great Lakes region, very little is known about the characteristics of wood and other CPOM in streams. Although much of the region is in forested land cover, few investigations have attempted to quantify characteristics of CPOM in forested streams and the relationships between CPOM and other stream ecosystem components. Further, almost no information exists regarding the effects of forest harvest on the standing stocks of large wood or CPOM in Midwestern streams. Thus, the objectives of this study were to:

1) Quantify standing stocks of large wood and coarse particulate organic matter (CPOM) in a low gradient stream system in the Upper Midwest;

2) Quantify the effects of forest harvest and riparian management on large wood and CPOM standing stocks in these streams;

3) Quantify the effects of two forest harvest techniques on large wood and CPOM standing stocks.

Methods

Study Design: See overview of project in main document and Figure 1.

Wood and CPOM Assessments

We quantified standing stocks of coarse particulate organic matter (CPOM) and large wood at harvested areas in tributaries of Pokegama Creek (Jack Irving Creek, and North and South Branch of Pokegama River, and Little Pokegama Creek). CPOM sampling and wood assessments were performed during low flow conditions during the summer of 1997,1998, 1999,

and 2000. Wood volume was measured using the line transect method (Wallace and Benke 1984). Three random transects were established in each flow regime (e.g., pools, riffles) represented within the sample reach in 1997 and 1998; in 1999 and 2000, only erosional sites were sampled. Logs $\geq$ 5 cm diameter and 1 m length that intersected the transect and fell within the bank-full channel were measured. Wood volume per unit area was calculated based on the formula:

$$\hat{X}_v = (\pi^2/8L) \sum^{n} d_i^2$$

where L = transect length, and d = stem diameter intercepted by the transect. Volume per unit area was calculated for each transect and an average was derived for each reach and for erosional and depositional habitat types. In addition to volume measurements, counts of the total length of large wood $\geq$ 5 cm diameter and $\geq$ 1 m in length were made at 10 m intervals within the reach and summarized as total meters of wood per m^2 of stream bottom (m/m^2) for each site. Debris accumulations $\geq$ 1 m^2 in area were classified by type and counted; data are shown as # / 100m. Benthic particulate organic matter (CPOM) > 1 mm was collected from three locations on each transect using a 0.1 m^2 Surber sampler. Samples were dried, weighed, ashed, and reweighed to determine the organic matter content of each sample.

Distributional properties of all variables were assessed and appropriate transformations were applied to non-normal variables. Three levels of control were used in the experimental design: time, site and watershed. The pretreatment year 1997 provided the time control, the upstream reach provided the site-level control, and the control sites (n = 3) provided the watershed-level control. Control sites (1, 4, 9) did not have site-level replication during 1997 and 1998. To enable us to perform a repeated measures analysis of variance (ANOVA), missing data were estimated by taking average of the three control site samples for 1999 and 2000.

A repeated measures ANOVA was performed on each variable, lumping downstream and adjacent together. The factors were <u>time</u>, with 4 levels (1997, 1998, 1999, and 2000) and <u>treatment</u>, with 4 levels (control, riparian control, cut-to-length, and grapple skidder). A contrast between the lumped control treatments and lumped logged treatments was also analyzed, which looked at differences within each year between logged and control sites. The time * treatment interaction and treatment effect were of main importance, indicating whether the values changed differently between treatments over time and if the treatments alone showed any differences, respectively. This analysis method has been recommended for use with before-after control-impact study data (McDonald et al. 2000). Response variables included: wood volume, total wood length, # accumulations, and g CPOM/m^2. The data used in the analysis was the difference from the upstream reach for a given treatment and year, which standardized each value for the current upstream condition in each watershed and set up a before-after control-impact study design (Conquest 2000, McDonald et al. 2000). The downstream and adjacent reaches were subtracted from the upstream reach value to obtain the values used. These values were analyzed for normality, and found to be excessively skewed and/or kurtotic. A log

transformation was applied, which increased the normality of the data (reduced the skewness and kurtosis in particular; Sokal and Rohlf 1995).

Results and Discussion:

Standing Stocks of large wood and CPOM

Debris dams were most abundant in the grapple-skidder reaches. The other three treatment types had similar numbers of debris dams. An aspen forest in New Mexico was reported to have between 0 and 10 debris dams per 100 m; this value is considerably lower than the range observed for the Pokegama streams (13.4 - 26 overall). A similar density was observed in 1st and 2nd order streams in the Hubbard Brook Experimental Forest (Bilby and Likens 1980, Hedin et al. 1988). In comparison to undisturbed streams in the Coweeta Hydrologic Laboratory, which had 1.2 - 2.4 /100m, Pokegama streams had much higher densities of debris dams (Webster et al. 1992).

Highest wood volumes were found in the cut-to-length treatment; the other three treatment groups had similar levels of large wood when measured using transect-based methods. Reach-scale measures of large wood (total wood length per reach area) were greatest in the grapple-skidder treatment, followed by the cut-to-length reaches. Both control sections had similar standing stocks of wood across the reach. Standing stocks of large wood in the Pokegama streams are comparable in volume to those reported in several old growth forests in Oregon, Washington State, California, Montana Alaska, and Tennessee (summarized in Gurnell et al. 1995). All of these forest, however, were dominated by coniferous species that are not found in this study area.

CPOM showed the largest variation across years of all measures; a 6 to 9-fold increase was observed across years within several of the treatment reaches. Large benthic organic matter (> 1 mm diameter) standing stocks in streams of the Coweeta Hydrologic Laboratory in Georgia were 124 to 255 g/m^2 in the impacted reaches and 213 to 244 in the reference reaches (Golladay et al. 1989). These values are well within the range observed in the Pokegama streams, although some of the reaches had very low standing stocks of CPOM (e.g., riparian control, 1997, 1998; grapple-skidder 1997, 1998).

Effects of forest harvest on CPOM and large wood

There were no obvious trends in the data between the control year, reaches, or sites (Tables 1, 2). CPOM, debris dams, wood volume, and total wood length displayed both increases and decreases through time and across reaches and treatments (Figs. 2-5). CPOM standing stocks varied over two orders of magnitude difference across treatments and reaches (Fig. 2). The sites with the highest CPOM levels were upstream reaches in the grapple skidder treatment sites, but overall, CPOM was unaffected by the treatments. The trend of upstream reaches having higher CPOM levels did not hold outside of the grapple skidder treatment. Total wood length data remained fairly consistent over the sample period (Fig. 4), as did the number of

debris dams (Fig. 3). Wood volume was consistent in all but the cut-to-length treated sites (Fig. 5), where values were higher than the other treatments and showed greater variability.

The modified data, subtracted from the upstream values to standardize for the within-watershed conditions at each site, display little difference between treatments and years (Fig. 6). The grapple skidder sites had greater amounts of CPOM than the others, for the first 3 years of the study, but showed no significant change from the control year (1997) until the year 2000.

The repeated measures analysis of variance (ANOVA) was designed to look at differences between treatments over all years (treatment effects), differences in how the treatments changed between years (time * treatment interaction), and differences between lumped control vs logged treatments within each year (contrasts). The analysis indicates no significant treatment effect or time * treatment interactions (Table 2), implying that none of the treatments differed across the years, nor did they differ in how they changed from year to year. The analysis does find a significant difference between control and logged treatments in the pre-treatment year of the study (1997) for CPOM, debris dams, and wood length, and for wood length again in 2000 (Figs. 7, 8).

Conclusions

Forest harvest did not appear to have a direct effect on benthic organic matter or large wood. This analysis did not account for the effects of harvest on blowdown, since wood that did not contain some portion within the bank-full channel was not included in the surveys. As the trees that currently span the channel decay and become entrained in the channel, they will significantly augment the standing stocks of large wood. These large logs are expected to have a dramatic effect on the channel, diverting flow, creating backwaters and plunge pools. Such habitat should provide enhanced habitat for fish and invertebrates; however, the effects of the open canopy on stream temperature could offset any benefits to the fish community. A more open canopy could also enhance primary production within the stream by providing more light and increased water temperatures. Increased wood retention and standing stocks will increase CPOM retention as well. In the Colorado Rocky Mountains, streams whose riparian areas were harvested over a century ago still have lower standing stocks of wood, and wood is smaller than nearby streams that were not harvested (Richmond and Fausch 1995). In Alaska, recovery of wood frequency to preharvest levels is predicted to take more than 250 years (Murphy and Koski 1989). These trends suggests that the long-term effects of harvest on benthic organic matter may not yet be realized in the Pokegama system, and require a longer term study.

March 20, 20001 Pokegama Creeks Summary Sandy Verry

A Rose by any other name

There has always been some confusion about the names of the various streams. The DNR names some creeks by the lake they run into. Thus, Pokegama Creek runs into Pokegama Lake, and Little Pokegama Creek runs into Little Pokegama Lake. The creek that runs through Plots 1 and 2 does not have a DNR name except "unnamed"; however, the local historians do have a name for it. For what it is worth, the names I use for the various streams are:

Plots 1 and 2	Jack Irving Creek
Plots 3,4,5	Pokegama Creek, North Branch
Plots 6,7,8	Pokegama Creek, South Branch
Plots 9, 10,11,12	Little Pokegama Creek

Jack Irving homesteaded on Sugar Lake to the West and North at the turn of the last century. John Otis documents the Jack Irving Road and Creek in his History of Sisseebakwet (Sugar Lake) with a sketch map. The Jack Irving Road is still there running along side the creek through Plots 1 and 2 and runs all the way to the large wetland and then to Jack Irving Lake where it originates. The road then turns North, Northwest and ends up at Sugar Lake. On the East end of the road, Jack Irving Road ends at Sherry's Arm of Pokegama Lake about 2 ¾ miles North of the County 17 Bridge. Travelers from Sugar Lake then used a boat to travel North in Sherry's Arm across Pokegama Lake to the current Golf Course Road and then by road again into Grand Rapids.

Pokegama Creek enters the South end of Sherry's Arm of Pokegama Lake (500 feet North of the County 17 Bridge), while Little Pokegama creek enters Smith Creek near the outlet of Little Pokegama Creek about 2000 feet South of County 17.

Since streams integrate various inputs in a serial fashion in the downstream direction, I organized my stream condition data by stream rather than plot replicate. This may be a useful alternative (adjunct) to testing various other data as well.

Data

The attached Share file in Excel contains these tabbed sheets:

- Water Chemistry 1997 (this was handed out years ago, but here it is again)
- Water Temperatures in 2000 (other years are similar, relatively speaking between plots, but slightly warmer or cooler)
- Percent sand, gravel, cobble, and boulder in the middle of each plot (97)
- Depths of sand in the channel of each plot in 97 and 98
- Total Pool depth, total riffle depth, and residual pool depth for each plot

- A series of survey points in plan view (a Map of each plot for the stream part and most vegetation transects
- A series of Longitudinal profiles for each stream plot showing pools, riffles, etc. and various stream slopes in or above the plots

Results and Some Interpretations

Water Chemistry

These waters are typical calcium (magnesium) bicarbonate waters. Little Pokegama Creek, (Plots 9,10,11,12) has the highest calcium and magnesium concentrations and is also the coldest of the four streams reflecting a greater contribution of groundwater to this branch than the other branches, although all are groundwater fed to some extent. Pokegama Creek, South Branch (Plots 6,7,8) is borderline with specific conductance values ranging from 91 to 128. I generally consider 100-150 as the point where water in the stream (lake or wetland) is about half and half groundwater and half surface water. Pokegama Creek, South Branch probably has the largest watershed, thus surface water is compensating for groundwater occurring more strongly in the other streams.

Water Temperature

The chart shows the range of summer time maximum daily values in dark red, along with the preferred, tolerable, and lethal ranges for brook trout. All the plots meet tolerable brook trout temperatures. Plot 6 (the one with more surface water) has the highest temperature, while Plots 10, 11, and 12 the ones with the strongest groundwater input have the lowest temperatures. The tributary to Pokegama Creek, South (Plot 9) is smaller in width than the plots farther downstream (it is a lower order stream (1) compared to the other plots (2)).

The largest Plot treatment effect on stream temperature was the Tributary to Plot 3. The chart shows temperatures in the 12 – 14 Cº range in the tributary above the cut part of the plot, while the tributary in the cut portion of Plot three runs from 15 to 22 Cº, and exceeds the tolerable range. The tolerable temperature range boundaries (yellow lines) are the temperatures that will cause brook trout to move around looking for either warmer or cooler temperatures.

The largest impact on stream temperatures is below the study plots where the remnant beaver pond warms surface water temperature into the 18 – 22 Cº range. This persists below the dam ranging from 19.5 to 23 Cº. It cools slightly after entering the forest canopy area below the beaver dam (traveling 200 feet into shade), and reaches tolerable levels at the wooden bridge site East of Plot 5 and North of Plot 12.

There was no Plot treatment effect, except the tributary on Plot 3. Stream temperature is controlled by the position of the plots in the landscape. The streams begin picking up groundwater about 200 feet below the Sugar Hill Moraine peak (top of Big Thunder

Peak). The spring water exiting to the streams is sheltered by the rather narrow and deep stream valleys in the moraine.

Percent of Sand, Gravel, Cobble, and Boulders

The chart depicting the composition of total stream sediment illustrates that sand overwhelms the sediment make up --- generally above 50% of the total sediment. This derives from surface soils characterized as the Goodland sandy loam. The sand is mostly fine sand in texture. These data are from pebble counts (after Wolman). Notable exceptions to the dominance of sand are Plots 9 and 10 where gravels and cobble are nearly as important or even dominate. Plot 10 is the strongest site for groundwater input and this continually flowing source may act to perpetually winnow sands from the channel. Note, that Plot 10 may be one of the most severe blow down plots, and this may derive from the wide spread occurrence of spring water entering the creek and at the edges of the small floodplain there.

Thalweg Sand Depth

The Wolman pebble count does a good job of characterizing sediment composition, but more importantly to stream habitat is just how much sand is in the channel. To measure this in 1997 and again in 1998 we used a simple measure of sand depth above underlying gravel and cobble in the thalweg of the channel.

There were small amounts in Little Pokegama Creek (Plots 9, 10, 11, and 12). When we first started this procedure, we measured it only on Plot 12 where it was 0.1 to 0.2 feet deep on average. In Jack Irving Creek (Plots 1 and 2) and Pokegama Creek S. Branch (Plots 6,7,8) and Pokegama Creek N. Branch (Plots 3,4,5), sand depths are generally deeper than 0.1 feet, much deeper, up to 1.1 feet deep. It should be noted that these measure of sand depth are taken from the thalweg of the channel, sand is deeper near the channel sides.

Jack Irving Creek experienced a lowering of the sand depth in the channel between 1997 and 1998. This basin has no road crossings on the channel. We first measured the channel on Plot 1 in June 1997 and discovered that the channel was not a single-thread channel (stream types E, or C), but a multiple-thread channel (stream type DA). Puzzled by why this channel should be multiple-thread, I walked the entire channel in July 1997 beginning at the wetland below Jack Irving Lake suspecting a beaver dam had blown out There was an old dam on the wetland, but it had been inactive for half a century or more, and there was no evidence of a dam break. Instead I found a normal E channel type below the wetland until I got about 200 feet above Plot 1 where a recent, mid-July storm had blown out the banks and the channel changed from a width of 4.5 feet to 11 feet wide. Farther downstream, just above the Plot 1 boundary, the stream had spit much of the eroded bank material onto the floodplain and reshaped itself closer to the normal E channel type.

Subsequent channel measurements in 1998 well above Plot 1 revealed where other blowouts had occurred and partially repaired themselves. I attribute the shallower sand depths in Jack Irving Creek to the self-cleaning of the system following the large July and large fall storms in 1997. These storms were bankfull in size or slightly higher, not large floods at all (the 2 to 3 inch rainstorm size). It is likely that the blowouts are in response to the amount of aspen harvesting occurring in the basin (about 70%) in the previous 12 to 15 years. This change in canopy cover can increase bankfull discharges by about 2 times and cause excessive undercutting of banks. Over the years the extra sand in the channel bottom forces the channel to "flood" more often and cut new, smaller, multiple channels across the flood plain.

In contrast, the basins for Pokegama Creek North, and Pokegama Creek South have not had wide spread harvesting in the past, but the two channels do have road crossings. The North channel (Plots 3,4,5) has two road crossings above the plots, and the South channel has one crossing above Plot 6. These channels received extra sand during and subsequent to the harvesting of the riparian plots. The chart shows significantly higher sand depths in 1998 compared to 1997. Plot 4 is an exception, with the same sand depth in both years, but in both years the sand depth is high, and I suspect it was just full.

The South Branch, with one crossing, has experienced several road prism washouts over the years and these plots (6,7,8) show a decreasing trend in sand depth downstream of the crossing in both years.

Repeated trips across the stream crossing road surface, tends to grind gravel into sand that subsequently erodes down the road surface into the channel. It can be a significant factor in changing the amount of available pool depth and the amount of sand embeddedness for gravels and cobble on the riffles.

UPM Kymmene is well aware of the road erosion problems at our plot stream crossings. In 1998 and 1999 they began a process of major stream crossing restorations in the Sugar Hills Moraine area. Other streams in the area included Morrsion Brook to the South, and Smith Creek to the Southeast. The Pokegama Creeks crossings were reconstructed in August and September of 2000 with larger culverts, sized to match the width of the stream, and with crushed rock surfacing the road approaching the crossings from either side. Earlier they began using brush hogs to control roadside brush rather than a grader blade that initiates more down-road or down-ditch erosion.

Channel Structure (Riffles and Pools)

Pool and Riffle depth is the distance from the bankfull elevation to the thalweg elevation of the pool or riffle. They indicate the depth of water available during bankfull flow conditions, and thus the swimming room available during spawning runs.

The residual pool depth is the distance between the downstream lip of the pool (top of a riffle) and the thalweg bottom of the pool. It is the water depth in pools when the stream just stops running. It represents the amount of space available during low water in summer and over winter during low water. Stream widths are similar between most plots varying between 4.5 and 8 feet with most in the 5 to 7 foot wide range. Plot 9 (a tributary) ranges between 3.6 and 4.9 feet in width.

Little Pokegama Creek Plots (10,11,and 12) have pool and riffle depths of about 2.6 and 1.6 respectively, and a residual pool depth of about 0.95 feet. Plot 9, about 2/3rds the width of the other streams, has respective depth values about 2/3rds less than Plots 10, 11, and 12.

Jack Irving Creek (Plots 1 and 2) has the lowest pool, riffle and residual pool depths at about 1.4, 0.8 and 0.6 feet respectively. The repeated channel erosion and channel self-repair in this basin has led to the multiple thread channels and probable frequent over-bank flows because of excessive sands in the channel bottom. The lower pool and riffle depths, and the second highest sand depths indicate that the channel and floodplain in the lower quarter of this basin are in an unstable condition. The presence of four channel types found in a short distance (E, C, DA, and B types) also indicates a channel system undergoing relatively rapid change.

Pokegama Creek Plots (3,4,5 and 6,7,8) are intermediate in pool, riffle, and residual pool depth. Plots 3 and 4 below the double road crossings has the shallowest pools (residual pool depths of about 0.5 feet) and reflects the presumed heavier sand inputs at the two road crossings.

Percent Sand, Gravel, Cobble, and Boulders

The chart of these percentages shows considerably more cobble and boulders in Little Pokegama Creek than in the other streams. Plots 3, 5, 7, and 8 are also relatively higher in gravels. Plot 10 is the highest in gravels reflecting the groundwater (a likely interface of a gravel lenses with the channel that is delivering the groundwater at relatively high rates). Plots 3,4,and 5 are just about as cool as Plots 10,11,and 12 and thus have groundwater entering as well (see also their calcium and magnesium levels), and they have 10% to 25% gravel in their sediment mix. Plot 4 is the exception, but this is likely caused by the drop out of sand at this point. Plot 3 may benefit from high exit velocities at the old, small, culvert at the road above Plot 3 that flushed the sand farther downstream. Sand in Little Pokegama Creek averages about 50%, while sand in the other streams averages 60% to 80%. Remember that the moraine material is a fine sandy loam texture, thus high sand amounts are to be expected even in Little Pokegama Creek.

Longitudinal Profiles

The longitudinal profiles show slope changes within each of the plots. They range from 0.3 to 5.0 percent slope. A general slope figure for this moraine system is 1.4% slope. The longitudinal profiles show where unusually deep pools occur, some in plots with high sand depths.

General Thoughts on Analysis

The physical, chemical, and temperature data indicate that each of the stream channels can also be considered a "treatment". They could be thought of as blocks, or just as separate treatments to the harvesting patterns and methods. Physically, the story is that on Pokegama Creeks N and S, the road crossings have added significant amounts of sand, and that additional road traffic in late 1997 and 1998 added more sand to the channel. Jack Irving Creek is undergoing accelerated channel changes in its lower quarter as relatively rapid changes in the canopy cover occurred in the last 12 to 15 years. Little Pokegama Creek is mostly normal for its size, parent material composition, and land cover conditions. Temperature wise there was little impact. The beaver dam was the biggest, and the cutover tributary on Plot 3 also warmed up. By and large, the deeply cut, narrow valleys and the strong groundwater emerging about 200 feet below the highes hill elevations over rode any lack of shade effect. The residual BA shade may also play a role though.

These physical conditions suggest additional analyses for other data types. For instance, invertebrate and fish communities may differ between the four stream channels, since physical differences in sand percent and depth are an a priori condition, not to mention the beaver dam on the Pokegama Creeks. Floodplain herbaceous species (or percent make up of species) may be different between streams. The greater sand depths (and lower pool and riffle depths) on channels other than Little Pokegama Creek suggest that over bank flows are more frequent there and thus smaller summer storms could move seed more frequently across the floodplain than in the Little Pokegama Creek floodplain.

The occurrence of the "big" pools may be related to LWD in the various channels in spite of much of each channel being buried in sand. Blowdown might be related to the intensity and occurrence of spring water on the plots. Differences in stream water temperature (even a degree or two) might correlate with higher amounts of blowdown.

Whether any of these possible cause and effect links can be shown is speculation on my part, but knowing more about each channel now suggests its worth a look.

Impacts of Forestry Best Management Practices on Water Quality and Benthic Macroinvertebrates

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Abstract

We experimentally determined the impact of selectively cut buffers using Full Tree and Cut to Length harvesting strategies and uncut forest buffers. Our dependent variables were stream macroinvertebrates and their habitats. We experimentally harvested the riparian zones of Little Pokegama Creek. We then measured the impacts of those harvests on benthic macroinvertebrates and their habitats over four years. Three replicates of four riparian treatments were compared using a randomized block design. One year's data collected prior to harvest were combined with three years' data following treatment. There were significant increases in fine sediment and intolerant taxa below harvested sites. Significant differences in yearly EPT were found in the Full Tree riparian treatment and the uncut riparian treatment. Differences also were seen in the Cut To Length riparian harvest treatment, although they were not statistically significant ($p>0.10$). True controls, with both the riparian and uplands uncut, also showed significant year to year differences. In an attempt to control for variation among sites, analysis of covariance was used to compare harvest treatments. In this analysis, riparian controls were significantly less impacted than other treatments when impact was expressed as %EPT or Family-Genera Biotic index. Immediate site-specific impacts varied but system wide impacts are occurring. Analysis of cumulative impacts at the watershed scale should be investigated over a longer period of time.

Introduction

Riparian ecosystems perform a vital role in maintaining water quality. Riparian ecosystems have been shown to affect the bank stability, sediment delivery, water temperature, hydrology, nutrient balance and the biotic community of stream systems (O'Laughlin and Belt 1995). The positive influence of functioning riparian areas on streams has led to efforts to protect these systems. Best management practices (BMPs) in riparian zones often are used in an attempt to mitigate the effects of logging and other anthropogenic impacts on stream systems. Forestry BMPs were developed to protect and

maintain water quality and have been shown to be highly effective in high gradient streams; their effectiveness in low gradient stream systems has not been thoroughly investigated. Buffer widths of at least 15-30m are widely recommended, although site-specific conditions may need to be considered (Castelle *et al.* 1994). The size of the riparian management zone (RMZ), the type of forest harvest and the extent of harvest within that zone all affect the cost of implementing these BMPs and the water quality benefits they provide. For this reason, it is important to determine the extent of impacts and how they are mitigated by various management practices.

Biological monitoring has been shown to be an effective way to assess water quality change. The organisms living within riparian communities depend on certain physical and chemical properties of the stream for their survival. One of the strengths of biological monitoring is that the stream organisms integrate water quality conditions through their life cycle. By monitoring changes within communities in aquatic habitats, we gain valuable insight about in-stream conditions over the lives of the organisms. Macroinvertebrates have been extensively used and investigated as a monitoring tool in lotic systems. These organisms are known to react in predictable ways to disturbance and to changes in water quality. Taxa Richness, %EPT (i.e., Ephemeroptera-Plecoptera-Trichoptera), and metrics that express the richness of groups of known tolerance have been shown to be effective measures of community health (Courtemanch 1996). This study uses biological monitoring to assess changes in physical and chemical properties of a low gradient stream ecosystem as a measure of impacts that may have resulted from various forest harvest treatments.

The state of Minnesota is currently examining its BMPs; those BMP guidelines are due to be re-issued in 2003. With a better understanding of their effectiveness, revisions of these guidelines may be able to better meet management goals and maintain the economic use of riparian lands.

Materials and Methods

Effects of forest harvest were analyzed on 12, 4.86 ha. treatment stands which were centered on first order stream reaches within the Little Pokegama Creak watershed. Three replicates each of four different treatments (Riparian Control, Cut to Length, Full Tree and True Control) were assigned using a restricted randomized block design. The position of the sites in the watershed is shown in figure 1.

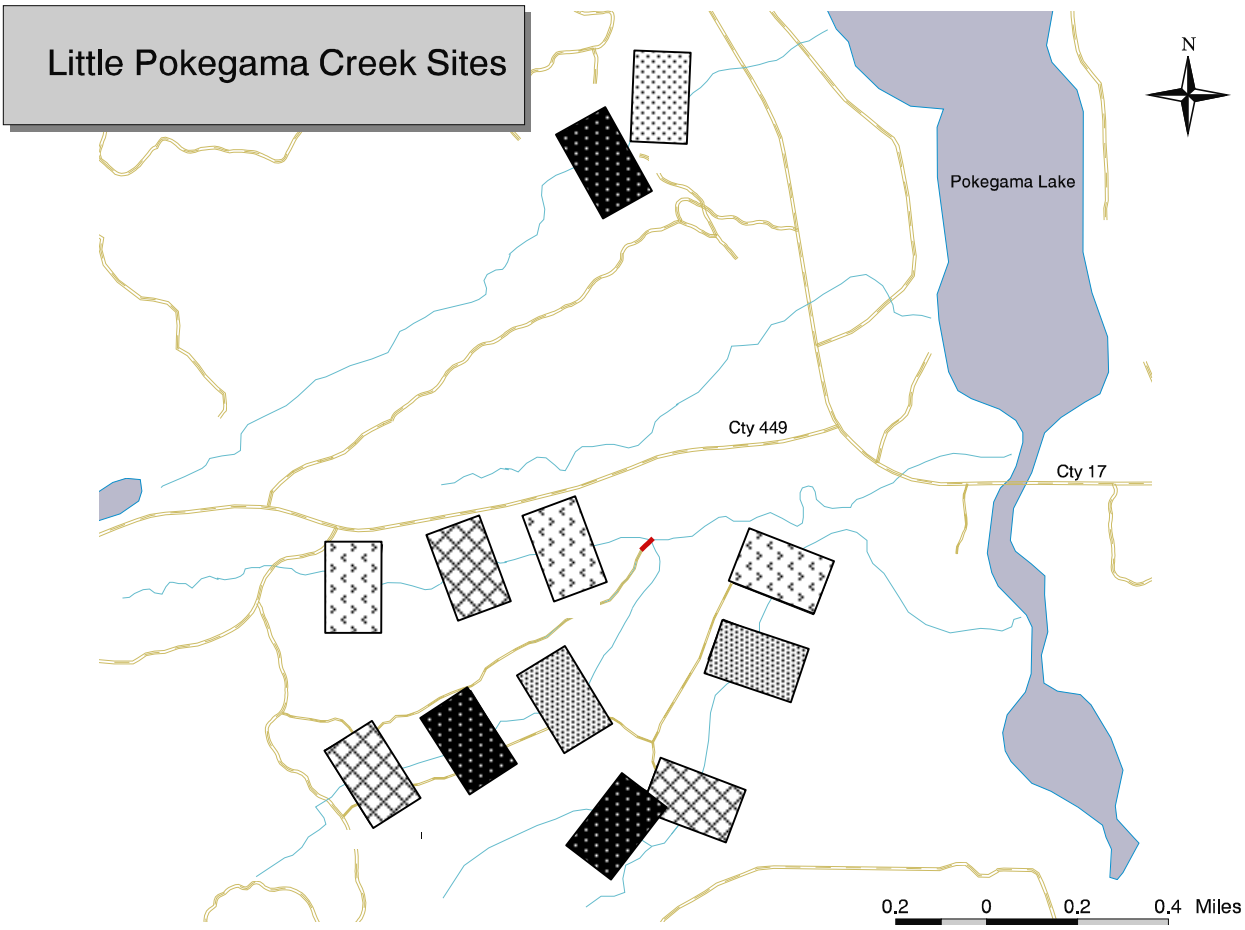


Fig 1 Displays the location of treatment sites and treatment type within the Pokegama Creek study area. (=RCNTRL, =CONTROL, =CTL, =FT)

No harvesting was conducted on the True Control (TC) stands. The upland area of the remaining nine stands was cut using either a Cut to Length (CTL) or Full Tree (FT) harvesting system. A 60m wide riparian management zone centered on the stream was left uncut in the Riparian Control (RCNTRL) stands. In the CTL and FT stands, selective harvest within the 60m wide riparian zone was conducted leaving a target residual basal

area of 5.7 m^2/ha based on earlier BMPs. Post harvest survey showed that 10m^2/ha residual basal area remained. This may imply significant variance in post harvest condition no matter what the target may be.

At each treatment stand, we sampled invertebrates in riffles in each of two 100m reaches. One reach was centered on the downstream boundary and was designed to detect effects of harvest at that stand. The other reach, designed as a control, started at the upstream boundary and extended upstream. At the TC sites, samples were taken within a representative 100m reach (Fig 2).

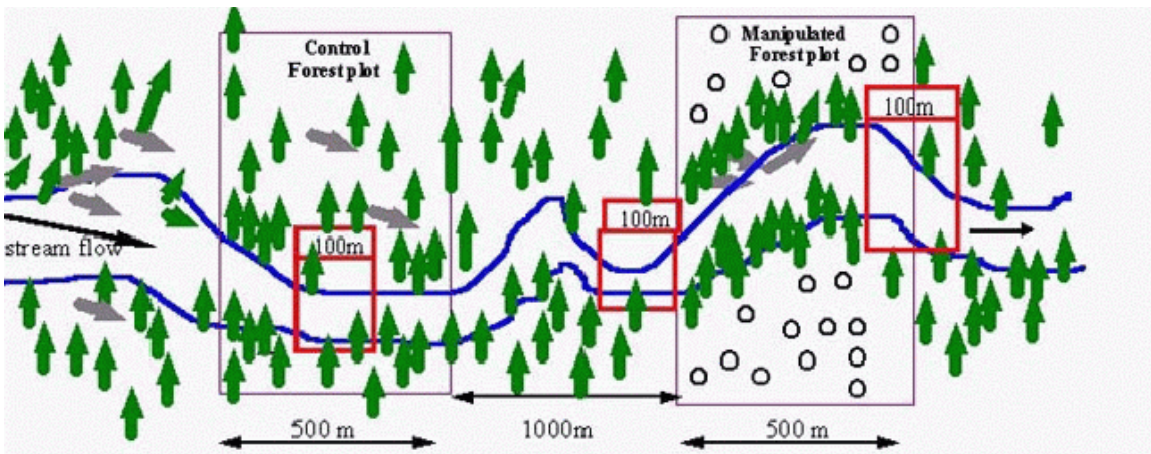


Fig. 2 Sampling locations at control (representative 100m reach) and treatment sites (100m treatment reach centered on the downstream boundary of the site and control reach extending 100m from the upstream boundary of the site) (Perry *et al*. 1998).

Within each of these reaches, two invertebrate samples were taken at random locations using a Waters-Knapp Hess sampler. The streambed was sampled to a depth of 10cm; all material collected was preserved in 95% ethanol and returned to the laboratory. Samples were collected three times in 1997 and 1999 (June 15, July 15, August 15) and twice in 1998 and 2000 (June 15 and August 15). Site-specific grab samples for temperature, conductivity, and dissolved oxygen were collected with each invertebrate sample. Continuously recording temperature loggers also were placed throughout the stream. Each invertebrate sample was "clean picked" and invertebrates were identified to the lowest practical taxon (usually genus) using McCafferty (1983), Hilsenhoff (1995), Brown (1976), Stewart *et al.* (1988), Wiggins (1996), Thorp and Covich (1991) and

Merritt and Cummins (1996), as appropriate. Nineteen biotic metrics were analyzed including %EPT, Taxa Richness, abundance, Family-Genera Biotic Index, percent of individuals per feeding guild and percent of individuals per similar family group (Figure 7). Treatment impacts were tested using these metrics in a one-way Analysis of Variance (ANOVA). Downstream reaches of each treatment were compared on a pre- vs. post-harvest basis, seeking individual differences among all four years. Upstream and downstream reaches also were compared among the three years after harvest, using each metric. Analysis of Covariance (ANCOVA) also was used to assess the impact of forest harvest practice on macroinvertebrates (following Lyons *et al.* 2000). In the ANCOVA model, the downstream value of a metric was the response variable, treatment the factorial predictor and the upstream value of a metric the second predictor. All models began with each of these terms and an interaction term. Terms were then eliminated until the best model was reached. A second ANCOVA was conducted using preharvest data as a covariate, year and treatment as categorical predictors, and the difference between the upstream and downstream condition as the response variable. Terms were then eliminated until the best model was found. Finally, Cluster Analysis was used to group sites based on the biological metrics fish IBI, Family Genera Biotic Index, %EPT, and Taxa Richness. This analysis was conducted with PC-ORD using the Euclidean distance and furthest neighbor method.

Results

Analysis of the data started with a direct comparison of control and treatment conditions. Two tests were used in this initial analysis. A one-way ANOVA compared the upstream control with downstream treatment condition; a second ANOVA compared downstream conditions among the four years of the study. Strong trends were observed among the four years of the study for all treatment types including the True Control but only one treatment, and only two of the 19 metrics, showed significant differences when the upstream and downstream sites were compared. This result provided evidence that system-wide impacts were occurring but site-specific impacts were not occurring, or not being detected. It was hypothesized that the variability among sites was interfering with the ability of the one-way ANOVA to detect differences due to forest harvest. Cluster

Analysis was used to ask if the upstream and downstream components of a site were more closely related to each other then the average condition. Using this information, an Analysis of Covariance (ANCOVA) model, with the upstream condition as a covariate, was constructed to control the variability between sites. This model asks if forest harvest caused the decline in water quality over the last four years.

Site-specific Impacts on Invertebrate Communities

One-way ANOVA failed to show any significant difference between control and downstream condition (Fig. 3, P >.05) with two exceptions. The Full Tree harvesting treatment showed a significant increase in the Family-Genera Biotic Index (ANOVA, p=.013, Fig 4) and a decrease in %Trichoptera, reflecting a decline in water quality. In addition, the RCNTRL displayed a significant difference in %Gastropoda due to a few unusual samples. Significance values for each of the treatments and metrics tested can be found in figure 7.

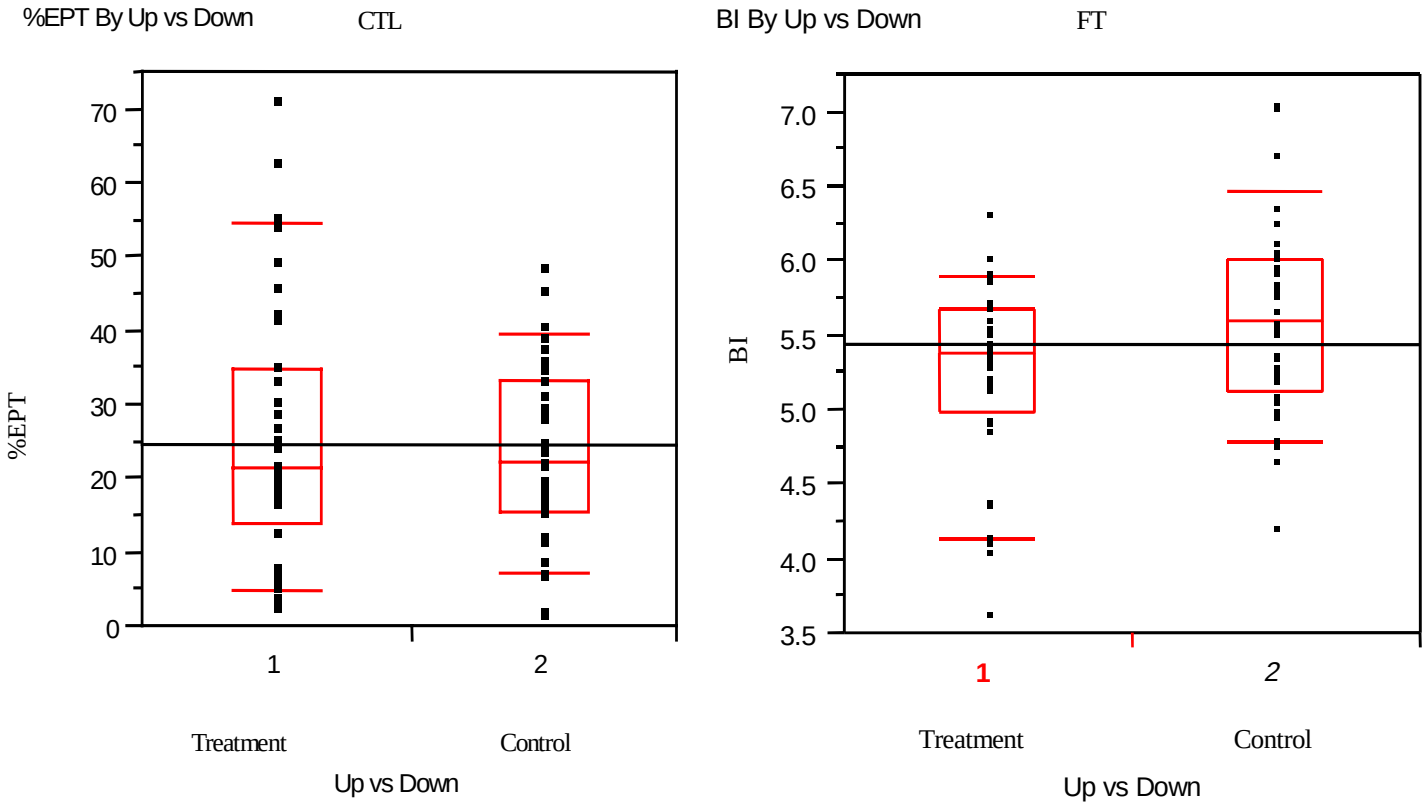
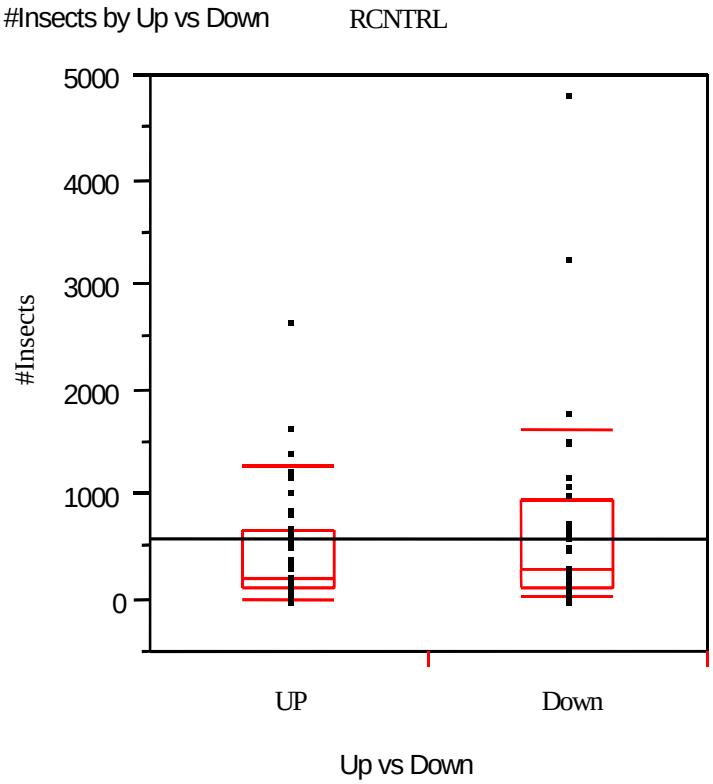


Fig 3,4 No treatment effect was seen between control and treatment reaches for each treatment type and metric combination (CTL and %EPT shown). FT harvest was the only treatment to show an impact (Biotic Index, %Trichoptera).



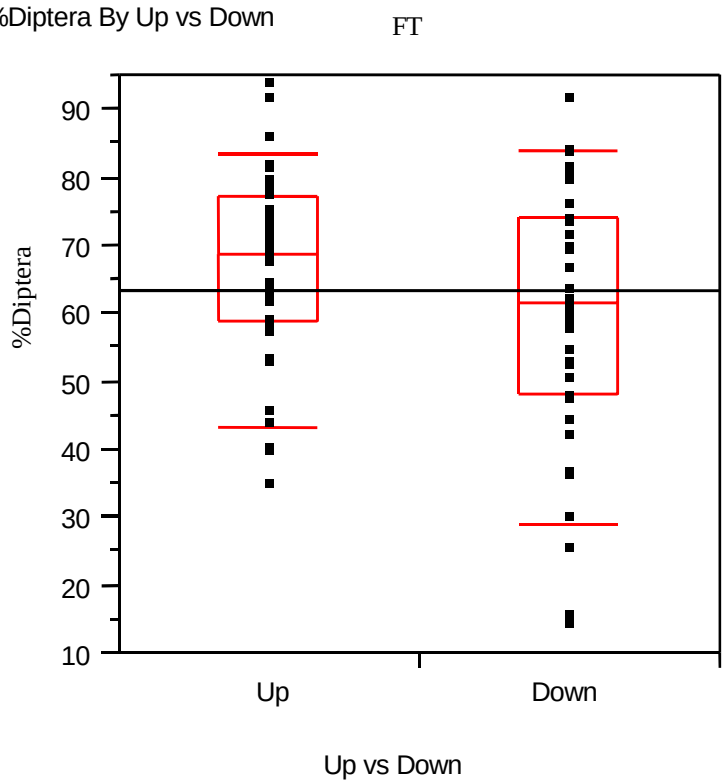
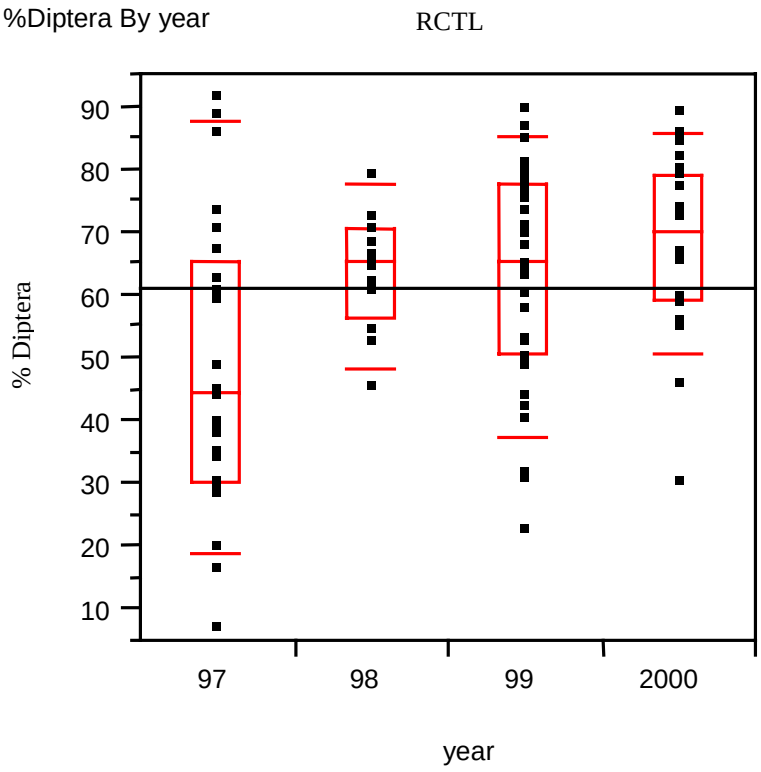


Fig. 5, 6 No significant treatment effect was seen for FT and RCNRTL sites when upstream and downstream reaches were compared (RCNTRL and #Insects, FT and % Diptera). Outliers were removed from data sets only when they changed the outcome of analysis.

	CTL	RCNTRL	FT	TC
%EPT	0.544	0.597	0.136	na
%Diptera	0.352	0.559	0.075	na
BI	0.471	0.536	0.018	na
Abundance	0.397	0.257	0.323	na
Taxa Richness	0.982	0.237	0.543	na
%collectors	0.928	0.831	0.521	na
%filterers	0.633	0.381	0.802	na
%predators	0.636	0.257	0.977	na
#Insects	0.384	0.267	0.258	na
%Shredder	0.187	0.119	0.519	na
%Gatherer	0.648	0.444	0.863	na
%gastropoda	0.834	0.045	0.326	na
%Coleo/Hemeptera	0.961	0.237	0.493	na
%Odonata	0.186	0.344	0.306	na
%Simuliidae	0.453	0.330	0.628	na
%Chironomidae	0.672	0.872	0.658	na
%Ephemeroptera	0.503	0.769	0.142	na
%Plecoptera	0.761	0.251	0.706	na
%Trichoptera	0.919	0.975	0.002	na

Fig. 7 Displays significance values for each of the treatment types and several of the metrics tested. Shaded boxes represent significant differences between upstream and downstream reaches for the three years following treatment.

Significant declines in water quality following treatment for all treatment types were observed in the invertebrate metrics. Significant results were also detected among the four years for the true control sites. Forty of the 76 tests exhibited significant differences between the pre-harvest condition and at least one of the years following treatment with a significance level $p<.05$. An additional 7 metrics are significant with a significance level of $p<.10$. Figures 8, 9, 10 and 11 display some of the trends evident following harvest.



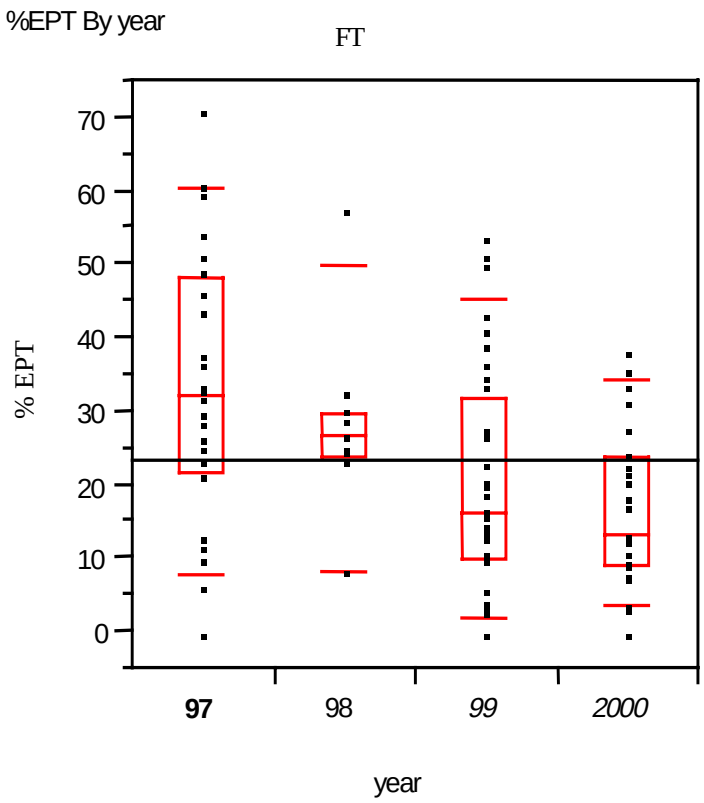
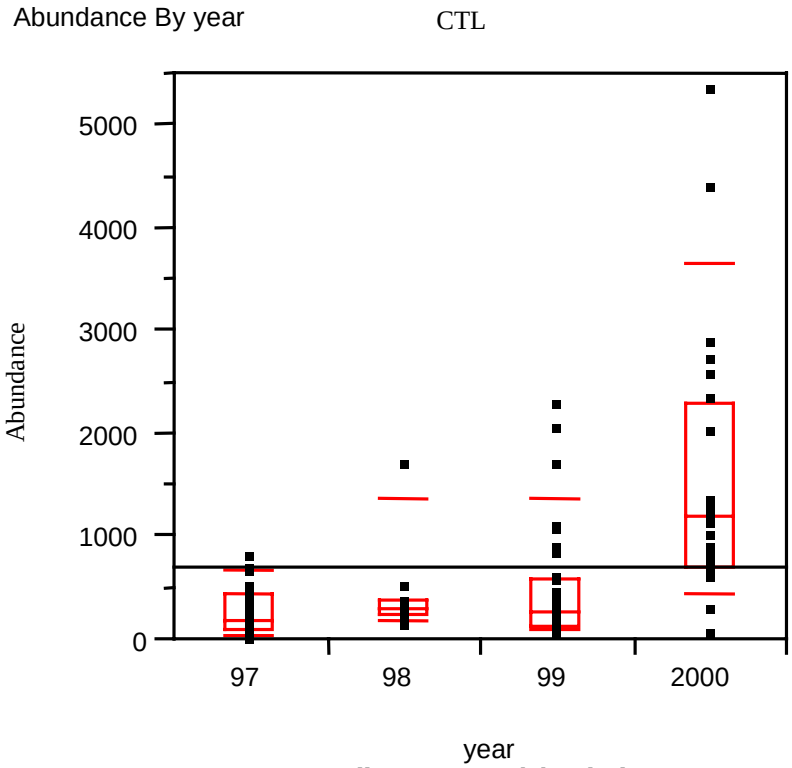


Fig. 8,9 Most metrics displayed changes in the community following forest harvest in the winter of 1997-1998. The generally tolerant Order Diptera responded positively after harvest (RCTL 1997 significantly different than 1998, 1999, and 2000) while the more sensitive orders that compose %EPT declined steadily following harvest (FT 1997 significantly different than 1999 and 2000).



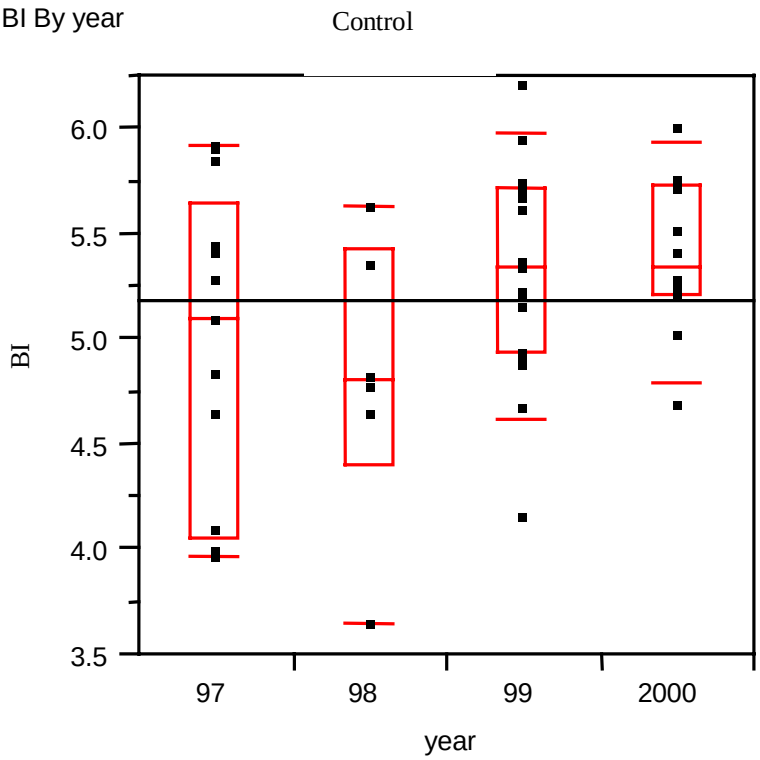


Fig. 10, 11 All treatments exhibited a large increase in abundance (CTL 1997, 1998, and 1999 significantly different than 2000). BI was a more conservative metric (TC) which did not significantly change over the four years of the study.

	CTL	RCNTRL	FT	TC
%EPT	0.346	0.009	0.033	0.004
%Diptera	0.361	0.030	0.003	0.000
BI	0.528	0.064	0.178	0.076
Abundance	0.001	0.000	0.001	0.000
Taxa Richness	0.193	0.290	0.921	0.015
%collectors	0.014	0.079	0.013	0.340
%filterers	0.000	0.012	0.000	0.000
%predators	0.003	0.173	0.000	0.000
#Insects	0.001	0.000	0.001	0.001
%Shredder	0.652	0.599	0.065	0.276
%Gatherer	0.000	0.001	0.000	0.000
%gastropoda	0.054	0.327	0.981	0.775
%Coleo/Hemeptera	0.294	0.403	0.615	0.054
%Odonata	0.260	0.653	0.814	0.336
%Simuliidae	0.007	0.637	0.074	0.058
%Chironomidae	0.045	0.091	0.083	0.001
%Ephemeroptera	0.314	0.064	0.034	0.001
%Plecoptera	0.229	0.148	0.008	0.605
%Trichoptera	0.951	0.296	0.050	0.051

Fig. 12 Displays significance values for each of the treatment types and several of the metrics. Significant values are shaded and represent treatment sites that displayed changes between at least two years of the study.

Analysis of Covariance was used to factor out some of the natural variation among the three sites receiving a given treatment so the effect of harvest could be more clearly seen. The upstream condition was selected as a covariate because it reflects the condition of the stream entering each treatment area and it is often very closely related to the downstream condition. In fact Cluster Analysis showed that upstream downstream site pairs sorted in the same cluster with the exception of site 8 (figure 13). The lower reach of site 8 sorted with sites 6 and 7 (C) while the upper reach was independent of all other sites (E). Clusters seem to correspond well with stream origin. However, both site 3 reaches cluster with sites 1 and 2 in Jack Irving Creek (A), and site 9 clusters with the North Branch of Pokegama Creek (B) rather than Little Pokegama Creek (D). In spite of that, a general pattern can be seen.

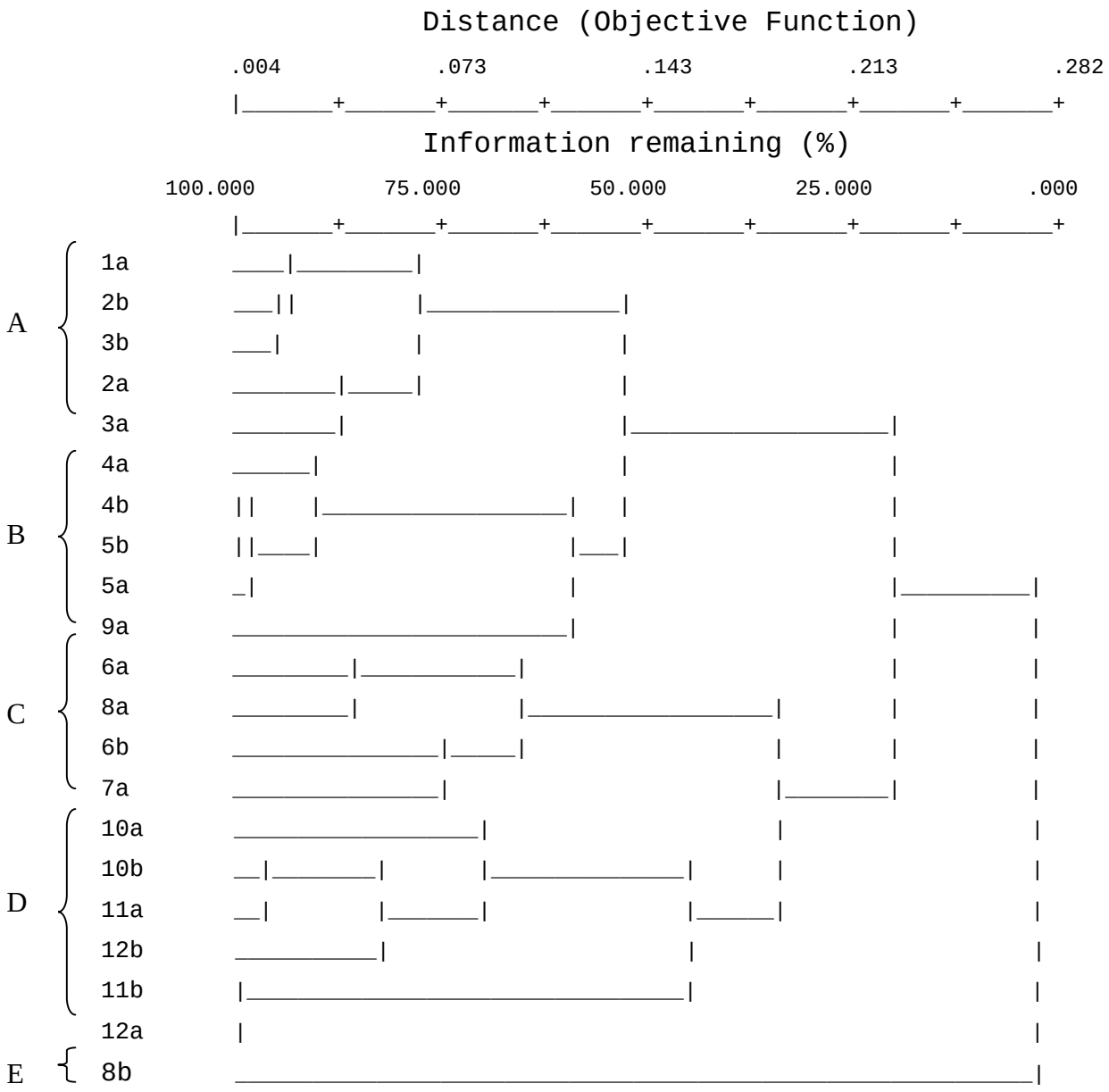


Fig. 13 Cluster analysis showed that upstream and downstream sites are similar. A pattern of sorting by stream origin can be seen.

The metrics %EPT, Taxa Richness, and Family-Genera Biotic Index were chosen for use with ANCOVA because the data fit the assumptions of the model. The best model in the Analysis of Covariance for the metrics %EPT and Biotic Index include both the treatment and upstream predictors but not the interaction term. The effect of Taxa Richness was not significant in the regression but it displays the same trend. In the

%EPT and Biotic Index models, the significant difference between treatments can be displayed as the vertical distance between regression lines (or the intercepts) in the model. Since the interaction term was not significant in the model ($p>.05$), the three regression lines ($r^2= 0.30$) are parallel but the intercepts are significantly different($p< .05$). The vertical distance between any two lines is the effect of treatment on the relationship between upstream and downstream sites. Figure 14 displays %EPT in this format allowing direct visualization of the model.

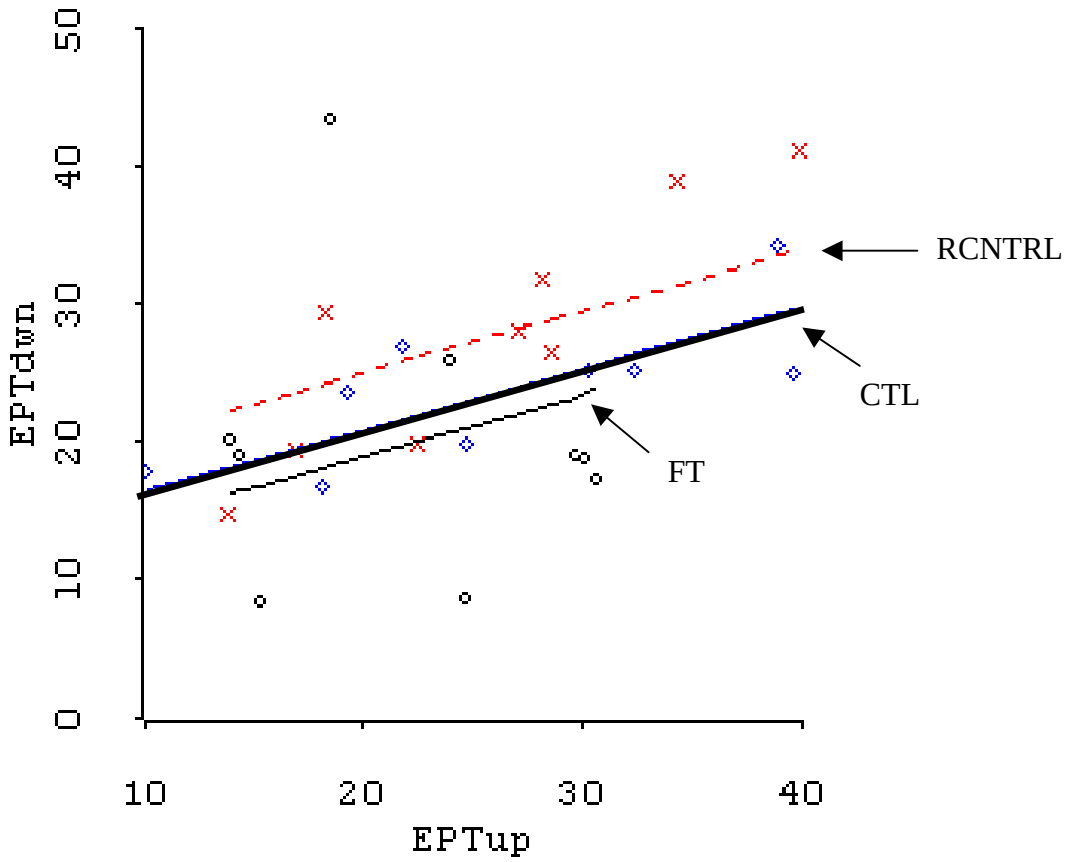
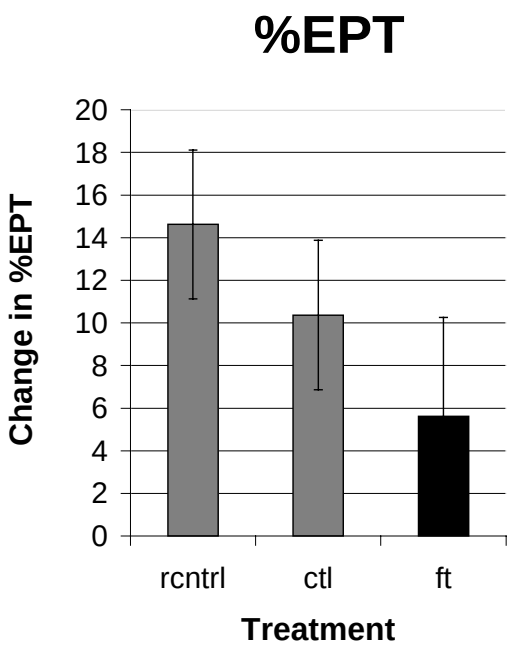
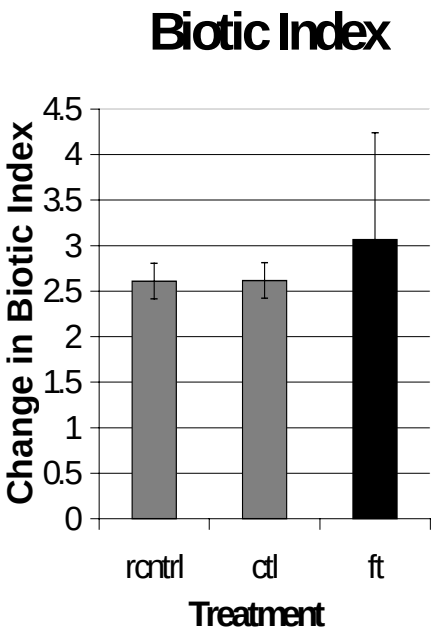
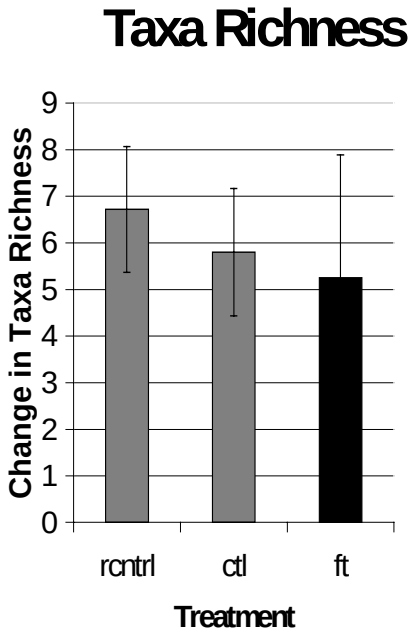


Fig. 14 Represents the result of the ANCOVA regression between upstream controls and downstream treatments ($r^2= .3$) The slope of the lines does not differ ($p>.05$) but the intercept is significantly different. Dashed line (x); Riparian Control, Bold line (◇); Cut to Length, Standard line (○); Full Tree

The intercepts for %EPT were the Riparian Control (representing our reference condition) with 16.09(+/-5.34), followed by the Cut to Length at 11.78(+/-5.35) and the Full Tree at 9.33(+/-5.41) (fig 15). Treatment effects for Biotic Index and Taxa Richness are shown in Figures 16 and 17, respectively.







Figures 15, 16, 17 display the estimates of the intercepts which represent the effect of forest harvest on the treatment variable in the ANCOVA.

The preharvest(1997) data were also used to minimize the amount of natural variation between sites with ANCOVA. This ANCOVA was conducted using the difference between the upstream and downstream values of a metric as the response and the 1997 condition as the covariate. Only the Family- Genera Biotic Index and Taxa Richness metrics fit the assumptions of the model. Of these two, the Family Genera Biotic Index model could be reduced to the effect of year alone and Taxa Richness failed to display a significant relationship. This second ANCOVA, with the 1997 data as a covariate, did not show any effect of treatment on macroinvertebrate metrics.

Impacts of Surrounding Conditions

Macroinvertebrates and fish were both used as measures of changing water quality in this study. Both groups of animals depend on habitat condition and predator prey relationships, and may react in similar or correlated ways. Sites were separated into two groups due to the presence of trout, which strongly effect IBI scores; one group included sites 10-12 and the other included sites one through eight (without trout). Site nine was eliminated due to a lack of fish. R^2 values indicated that there was not a linear relationship between IBI scores and any macroinvertebrate metric ($r^2<.01$). P-values exceed 0.2 for all regressions between macroinvertebrates and IBI scores. The scale of our analysis could have affected the correlation between fish and macroinvertebrates; fish were sampled over 100m reaches while macroinvertebrates were sampled in select riffles.

Leaf litter inputs were shown to vary by harvest treatment and are depended upon by many macroinvertebrates. The sum of overhead leaf litter (litter entering traps from above) and lateral leaf litter (entering traps laterally) was significantly but weakly correlated with IBI scores and abundance ($r^2= .15$ in each case). Fish IBI scores and macroinvertebrate Taxa Richness were also correlated with overhead leaf litter ($r^2=.16$ and .10). Taxa Richness was the only metric significantly related to lateral leaf litter ($r^2=.14$). Taxa richness was consistently positively correlated with leaf litter input but abundance and IBI score were negatively correlated.

Decreases in percent gravel and increases in percent sand were seen at all sites between pre-harvest and post-harvest years ($p<.05$). Substrate and other habitat variables

were not correlated with invertebrate metrics. Habitat metrics tested included %embedded, % fines, % coarse, %riffle, % pool, depth, course woody debris, canopy cover, overhanging vegetation, unstable banks, and boulder pockets.

Discussion

Downstream conditions (i.e., below harvest treatment) were significantly impacted after three years of forest harvest. However, a similar trend was shown for True Control sites. Our ANOVA analyses of conditions upstream and downstream of forest treatments failed to implicate site-specific forest harvest as the cause of invertebrate community change. (The two significant site-specific changes, FT sites with %Trichoptera and %EPT, could have been due to chance considering the large number of analyses conducted.)

Large variation in the invertebrate community among stream sites may have had an impact on our ability to detect upstream downstream changes using ANOVA. We used Analysis of Covariance, with the upstream condition as a covariate, in an attempt to control this variability and get a more accurate estimate of the impact of forest harvest. Cluster Analysis verified the argument that upstream sites were more similar to their downstream counterparts than other reaches in general. Treatment type was a statistically significant predictor in %EPT and Biotic Index models. Both of the models suggests that the Riparian Control was least impacted while the Cut to Length and Full Tree harvesting methods were progressively more severely impacted. Treatment effects on Taxa Richness, while not a significant predictor, displayed the same trend. These ANCOVAs, with the upstream condition as the covariate, show that impact to the invertebrate community was dependent on forest harvest. These statistical tools also allow the direct comparisons between harvesting methods. Functional feeding groups have been shown to have a weak relationship to changes in water quality (Palmer *et al.* 1996). Several metrics (including functional feeding groups) are yet to be analyzed using ANCOVA and will be reported elsewhere.

System wide impacts were evident over the four years of the study. ANCOVA, with the upstream condition as the covariate, showed that some of those impacts were attributable to forest harvest. For example, significant treatment effects on the Family

Genera Biotic Index and %EPT metrics were detected. The riparian control treatments were the least impacted for each of the metrics tested, implying that there is an environmental cost associated with the selective harvest of riparian lands. The size of the harvest effect, especially with the Cut to Length harvesting strategy, is most often relatively small and within the 90% confidence interval. However, in sensitive systems, any impact could alter the system and should therefore, be considered in management decisions.

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EFFECTS OF FOREST HARVEST ON STREAM FISH AND HABITAT

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ABSTRACT

The response of fish communities and stream habitat to four logging treatments was studied in north central Minnesota. Logging treatments included (three replicates each): (1) control (no harvest), (2) riparian control (30 m unharvested buffer, upland clearcut), (3) cut-to-length (CTL) riparian thin (30 m buffer cut to 25 square feet basal area using CTL logging system) and (4) full tree (FT) riparian thin (30 m buffer cut to 25 square feet basal area using FT logging system). Fish and habitat data were collected from 50 m reaches above, within and below each treatment site, one year before harvest and for three years after harvest. Analysis of covariance (pre -harvest data used as covariate) revealed some different effects due to harvest treatment. Canopy cover was reduced within the two riparian thin treatments, but not below. The reduction continued in the years following harvest due partly to increased vulnerability to windthrow. Coarse woody debris (CWD) increased at within reaches in the two riparian thin treatments. The highest increases of CWD were in CTL treatments where slash was removed immediately after trees are felled. Increases of CWD continued throughout the study in the riparian thin treatments due to windthrow. Fine sediments tended to increase in harvested treatments while remaining the same in controls following harvest. There was, however, large year to year variation in fine sediments across all sites that may be more influential than harvest effects. Fish community Index of Biotic Integrity (IBI) scores were higher in controls and riparian controls than in riparian thin treatments three years after harvest during August surveys. Brook trout declined in riparian controls while s howing no change in riparian thin treatments. While there were some significant harvest effects, large year to year variation on most measured variables throughout the system suggest that factors operating at a broader scale might influence fish and habit at more than local harvesting.

INTRODUCTION

The hydrology of forested stream systems can be altered due to forest harvest operations and subsequent forest succession (Verry et al. 2000). Removal of trees usually results in evapotranspiration decreases (C ampbell and Doeg 1989), which can lead to changes in the timing and level of stream flow. Paired watershed experiments have shown that rainfall peak discharges and storm flow volumes have initially increased following forest harvest in Minnesota (Verry et al. 1983), but the cause of the flow changes are confounded by many factors such as change in vegetative cover, soil storage properties, changes in the conveyance system of the watershed (roads and skid trails), and extent of surface erosion (Brooks et al. 1997). Removal of trees can also alter the thermal regime and quantity of organic inputs to streams. Harvesting trees near streams eliminates the shading effect provided by the canopy; this can lead to increased stream temperatures (Rishel et al. 1982, Bowlby and Roff 1986).

Canopy removal results in the temporary reduction of allochthonous leaf inputs (Webster et al. 1983), and tree removal reduces the future contribution of large CWD to the stream (Murphy and Koski 1989), although immediate increase s in smaller CWD (slash) can occur. Finally, transport of sediments to streams has occurred following forest harvests. The primary source of sediment is from forest roads, including skid trails and stream crossings (Eagan 1999). While some effects relat ed to tree removal (change in

flow, temperature increase, reduction in leaf inputs) are usually short term and decline as new trees grow, sediment inputs due to roads (Zimmer and Lisle 1998) and reduced inputs of large CWD (Murphy and Koski 1989) can last well into the future.

The effects of forest harvest practices on stream fish have been well documented (e.g., Campbell and Doeg 1989, Meehan 1991), although most studies have been conducted in high gradient systems. Generally, forest clearing negatively affects fish (Campbell and Doeg 1989). Fish can be both directly and indirectly affected by the alterations in stream hydrology due to forest harvest. Direct effects on fish include: (1) removal of fry from streams due to increase flow volumes (Scrivener and Anderson 1984), (2) changes in fish physiology due to increases in stream temperature, (3) changes in fish habitat due to decreasing large CWD inputs (Murphy and Koski 1989, Bilby and Ward 1991), and (4) reductions in egg and fry survival (Scrivener and Brownlee 1989) due to filling of interstitial substrate spaces with sediment transported from harvested areas and forest roads (Rothwell 1983).

Indirect effects of logging on fish communities usually operate through the macroinvertebrate forage base. Fish communities can be affected by changes in macroinvertebrate community composition and abundance. Reductions in allochthonous leaf material and increases in sunlight penetration can alter the macroinvertebrate functional community composition from one dominated by shredders to one consisting primarily of grazers (Carlson et al.1990, Stone and Wallace 1998). Also, filling interstitial substrate spaces with sediment, and reduced inputs of CWD both result in reductions of habitat that macroinvertebrates utilize (Lenat 1981). Reductions in the macroinvertebrate food base can reduce fish abundance and growth. Generalist fish that are able to switch prey type will outcompete specialist fish that feed only on the declining prey species (Garman and Moring 1993).

A common approach used to mitigate the effects of logging on instream habitat and biotic communities is to protect a portion of the riparian vegetation (Clinnick 1985). These protected areas near streams and lakes are called buffer strips. Buffer strips provide a canopy that continually shades the stream, and provides allochthonous leaf inputs. Also, trees in the buffer strips provide a constant supply of CWD. Finally, buffer strips help stabilize banks, and serve as filters that collect sediment before it enters the stream (Davies and Nelson 1994, Lowrance 1998). However, buffer strips are limited in their ability to influence timing and volume of flow (Barton et al. 1985). Although it has been shown that buffer strips can mitigate negative logging effects (Newbold et al. 1980, Moring et al. 1994) the land dimensions and residual tree densities needed for effective buffer strips are still being debated.

The purpose of this study is to examine the effects of forest harvest practices on fish communities and instream habitat in northern Minnesota streams. Results from this study will be used to evaluate the effectiveness of various forest harvest management practices. Specifically, residual tree density and mechanical forest harvest techniques were evaluated on a low gradient, coldwater stream system. This study on fish and instream habitat represents one component of a much larger integrated study. Other aspects being studied in response to local forest harvest are water quality, stream benthos, birds, organic matter, tree regeneration, susceptibility of replica artifacts, and cost effectiveness of harvesting systems.

The initial results (one year preharvest, one year post harvest) of harvest effects on fish and instream habitat were presented by Merten (1999). Few habitat changes occurred due to harvest, although canopy cover was reduced and residual trees were more susceptible to windthrow in riparian zones where trees were thinned (Merten 1999). Fine sediments increased throughout the stream system one year after harvest. Merten (1999) suggested that increased traffic on road crossings located upstream from the study area was a likely source for this sediment. Fish Index of Biotic Integrity (IBI) scores decreased, and species richness increased throughout the system one year after harvest, likely due to the increase in sediments. These changes in fish communities suggested an overall decline in the "quality" of a coldwater stream, however, Merten (1999) suggested that additional years of study were needed to properly asses stream impacts.

METHODS

Study Site

Pokegama Creek is a coldwater stream system located near Grand Rapids in north central Minnesota. This is part of the Northern Lakes and Forest ecoregion (Omernik and Gallant 1998). The system includes four small (mean width ~ 1 m, mean depth ~ 0.12 m) first and second order streams that flow into Lake Pokegama. The four streams are named: Little Pokegama Creek, Pokegama Creek North, Pokegama Creek South, and Jack Irving Creek. Sugar maple (26%), paper birch (20%) and aspen (11%) are the major overstory trees (Palik 1998). Terraces are present on both sides of the streams, and soils are well-drained loams.

Study Design

The approach of this study was to assign replicated harvest treatments to sites along the four streams. Study sites were about 12 acres (4.8 ha) each (6 acres on each side of the stream). Three replicates of four different harvest treatments (12 sites total) were used (Table 1). These treatments use two different forms of mechanical harvest, the conventional full tree (FT) system, and the "light on the land" cut-to-length (CTL) system. The FT system uses a feller/buncher to harvest trees and a grapple/skidder to drag the trees to a landing site where limbs and tops are removed. The CTL system uses a harvester to fell the trees, remove the limbs and tops, and cut into desired length. A forewarder then loads the cut timber and takes it off site. The CTL should result in less soil disturbance to the clearcut area because the vehicles drive on the slash mat produced from the harvester (Richardson and Makkonen 1994). Also, forewarders carry more wood than grapple skidders resulting in fewer trips.

The four treatments include unharvested control, riparian control, CTL riparian thin, and FT riparian thin. No harvesting was conducted in the control sites. Riparian controls had upland areas clearcut with CTL while a 30 m buffer strip on each side of the stream was left unharvested. The upland of CTL riparian thin treatments was clearcut with the CTL system and the buffer strip was harvested to 44 ft^2/acre (10 m^2/hectare) basal area. The upland of FT riparian thin treatments was clearcut with the FT system and the buffer strip was harvested to 44 ft^2/acre basal area. All harvesting took place in the fall and winter of 1997. The harvest treatments were assigned to sites using a stratified restricted randomization. The restriction was that no control sites could be downstream

from more than one harvested site and treatm ents were stratified among streams. Fish and habitat data were obtained from three 50 m reaches per study site (Figure 1). The "upstream" serves as a local control for each site while the "within" and "downstream" represent potentially impacted reaches.

Fish Sampling

Fish were collected during the past four years (1997, 1998, 1999 and 2000). Fish were sampled twice each year (June and August), with a Wisconsin AbP -3 backpack electrofisher. At each site fish were collected from the three 50 m reaches wit h two consecutive passes. During the first pass fish were identified to species, measured (total length), weighed (only in June), marked with a caudal fin clip and returned to the stream. Fish caught during the second pass were measured and weighed (reca ptured fish were only measured). Scale samples were collected from all trout for age determination.

Fish abundance was estimated two ways: two-pass depletion (removal) or mark-recapture (PopPro, Kwak 1992). The removal estimate was used if catchability was $\geq$ 0.8. When the catchability was < 0.8 and the ratio of recaptured fish to total second -pass fish (r/c ratio) was $\geq$ 0.2, the mark-recapture estimate was used. If the catchability was < 0.5 and the r/c ratio was < 0.15, abundances were not calculated. Values this low appear to violate removal or mark-recapture assumptions. The sum of pass 1 and pass 2 captures was used when both estimates failed these cutoffs and there were ten or fewer fish. Index of Biotic Integrity (IBI) values were calculated according to Mundahl and Simon (1998). This index assesses the environmental health of coldwater streams using fish communities. Each reach was assigned a score from 0 to 120 based on 12 metrics. The metrics are based on fish abundances, species richness, indicator species and trophic function. Higher scores are indicative of higher quality reaches. Species richness was analyzed in addition to species abundances and IBI scores.

Instream Habitat Evaluations

Stream habitat measurements were obtained duri ng July of 1997, 1998, 1999 and 2000. Stream discharge was estimated at five transects evenly spaced throughout the 50 m reach. Depth and velocity were recorded at four evenly spaced points along each transect except when the stream was less than 1 m wide , in which case a measurement was obtained every 30 cm. Final discharge values were calculated by measurements of depth, width and velocity.

Habitat was assessed with a modified version of the protocol developed by the Minnesota Pollution Control Agency (Baily et al. 1993). Habitat assessments were conducted along the three 50 m reaches, usually at three 17 -m intervals in each reach. Percent canopy cover was estimated using a spherical concave forest densiometer in four directions in each 17 m section (Lemmon 1957). Overhanging vegetation was estimated as the percentage of stream channel shaded by vegetation within 1 m of the water. Percentage boulder pocket cover was estimated as the amount of stream channel occupied by boulders and the disturbed flow produced from them. Percentage unstable bank was determined by estimating the amount of both banks with fine bare soil. Banks covered with roots or larger stable substrates were not considered unstable. Percentage woody cover (WC) was estimated as the amount of bankfull channel containing woody material

of any size (including twigs and branches) that could provide cover for fish. Windthrow trees were also counted in 1998-2000.

Stream substrates were characterized by visually estimating the percentages of each substrate size class (clay, silt, sand, gravel, cobble and boulder) and by ranking the top three substrates present. In each reach, 35 randomly placed circular (28cm diameter) quadrats (7 within each 10 m interval) were placed for substrate characterization. Along with substrate, percentage coverage of macrophytes and visible periphyton was estimated. Depth was recorded at each quadrat to obtain a depth profile for the entire reach. Substrates were summarized into two categories: percentage fine substrate, and percentage coarse substrate. Percentage fine substrate was calculated by summing the average percentage of clay, silt, and sand in the 35 quadrats. Percentage coarse substrate was calculated by summing the average percentage of cobble and boulder in the quadrats.

Statistical Analysis

A factorial ANCOVA design was used for the analyses. The 1997 (preharvest) data were used as a covariate in order to remove the effect of natural variation occurring across the study sites before harvest took place. The three factors included year (97, 98, 99, 00), reach (within, downstream), and harvest treatment (control, riparian control, CTL, FT). Response variables included the measured habitat and fish variables (Table 2). The response variables were calculated by subtracting the "upstream" value from the related "within" or "downstream" value. For example, if WC cover was 25% at the "upstream", 30% at the "within", and 20% at the "downstream" reach at a site, the response for the ANCOVA would be: "within" = 5, and "downstream" = -5. This difference value made use of the local control provided by the "upstream" data and also tended to reduce heteroscadasticity that was present when "upstream" data were not used. In this paper when it is stated that a variable increased or decreased it means that there was an increase or decrease in either the "downstream" or "within" reach compared to the "upstream" reach. Table 3 shows an example ANCOVA output from percentage fine substrates. In this example the covariate is significant (all values reported significant are $p < 0.05$), so it would be appropriate to use the adjusted mean scores for tests and analyses. The figures in this report use the adjusted means unless otherwise noted. Also, there is a significant year main effect, and a year by treatment interaction. Because the three way interaction is not significant it would be removed and the analysis repeated. If there were no significant two way interactions the model would be reduced to only main effects. Tukey's HSD comparisons (alpha = 0.05) were used to test for pairwise treatment mean differences.

All study sites were included in analyses of habitat variables. Analyses of fish variables were divided into two sets because of differences in fish communities among the streams. Sites 1-8 contained similar fish communities and were analyzed together. Each harvest treatment was represented twice in the analyses. This replication allowed for separate analyses for June and August data. The "within" reaches at both site 1 (control) and 2 (CTL) were generally dry by August so these data are excluded from all August analyses leaving only one data value for control and CTL treatments. Northern redbelly dace (*Phoxinus eos*) and finescale dace (*Phoxinus neogaeus*) were difficult to distinguish and were grouped together (*Phoxinus*) for all analyses. Sites 10-12 contained brook trout

and rarely had other fish so they were analyzed separately. Sites 10 -12 only contained one case for each harvest treatment so the data from June and August were combined for analyses. Fish were seldom present at site 9 (control) so it was not used in the analyses.

RESULTS AND DISCUSSION

Canopy Cover

The canopy cover responded in a predictable manner to the harvest treatments. All main effects and two -way interactions were significant for the ANCOVA on canopy cover (Table 4). The riparian thin treatments showed a reduction in canopy cover over time, while the controls and riparian controls did not (Figure 2). The FT treatment did not show an immediate reduction, but declined two years following harvest. This is likely due to windthrow. The decrease in canopy cover was only observed along "within" reaches, and did not extend to the "downstream" reaches (Figure 3).

WC

The full ANCOVA model did not produce any significant effects on WC, however the normality assumption was violated, due to outliers. Most of these outliers were from the "within" reach at site 11. Removal of all outliers produced an ANCOVA with all main effects and the harvest treatment by reach, and harvest by year interactions significant (Table 4). WC increased after harvest along both riparian thin treatments, but more so in CTL (Figure 4). This increase persisted through 2000 in the CTL, but decreased t o levels only about 5% greater than preharvest along the FT sites. A large increase in WC in the "above" reach at site 10 (FT) in 2000 is responsible for the decline in WC difference values in FT harvest treatments in 2000. If this increase above site 10 had not happened, the FT treatments would have leveled out at the 1999 values. The increases of WC along the CTL sites were localized to the "within" reaches (Figure 5). The CTL "within" reaches were found to be significantly different from all other re aches with an HSD comparison. Most of the WC appeared to be small (branches and tree tops). Since the CTL harvest system removes branches and tops where the tree is felled rather than skidding the full tree to a landing (i.e., FT system), this suggests t he increase was due to slash falling in the stream. The more gradual increase in WC in the FT treatments is likely due to the yearly increase in windthrow trees.

It is unclear why site 11 (a CTL site) did not exhibit the same increases in WC that the other CTL treatments did. The percentage of WC at site 11 did not change much either way. One possible reason could be that the thinned trees happened to be harvested at a greater distance from the stream channel than at the other CTL sites. If this were th e case, most branches and tops would not fall into the stream. Johnson et al. (2001) did not detect a significant harvest effect on CWD in these study streams. Wood needed to have a minimum diameter of 5 cm to be identified as CWD by Johnson et al. (2001), while bankfull channel containing woody material of any size that could provide cover for fish was used in this study. Much of the slash was less than 5 cm in diameter.

The results from this study agree with other findings that show increases in small forms of WC following logging. Large amounts of small WC can make it difficult for large fish to migrate, however, in these streams it is likely that the WC is providing extra habitat for the minnow species. If logging continues and trees are harvested b efore they

fall into the streams there can be a general trend toward less WC in future years. This is one of the primary arguments for buffer strips, to provide a constant supply of WC. If buffer strips are not designed properly, windthrow can remove buf fer strip trees causing an initial increase in large WC and reducing the ability of the buffer strip to provide long term inputs of WC.

The number of windthrow trees that had fallen in or across the streams was counted at all sites starting in 1998. An AN OVA on a square root transformation of these data (same setup as ANCOVAs, except no 1997 data) shows that the number of windthow trees varied significantly by harvest treatment, reach and moderately (P = 0.07) by year. Windthrow trees increased with year since harvest (Figure 6). An HSD comparison (alpha = 0.05) did not find any differences among the years, but if an alpha of 0.10 was used it would be likely that the number of windthrows in 2000 would be different from 1998 and 1999. The FT treatment had the most windthrows (Figure 7). There was no significant difference between the FT and CTL treatment, but the FT value was significantly higher than the control and riparian control treatment. Windthrow occurred significantly more in "within" than "downs tream" reaches (Figure 8). The more frequent occurrence of windthrows in the FT treatment indicates that thinned buffer strips probably do not provide a dense enough shield to protect trees from strong winds; the riparian control had little increase in wi ndthrow. It is also reasonable to expect excess inputs of large WC due to windthow near thinned buffer strips. Buffer strip design should try to minimize unnatural tree loss due to windthrow. This will ensure a constant supply of WC well into the future.

Percent Fine Substrate

Both the year main effect and the year by harvest treatment interaction were significant for fine substrate percentage (Table 4). Fines increased throughout the system in 1998, decreased in 1999, then increased again in 2000 (Fi gure 9). The only significant difference occurred between 1999 and 2000. Fines substrates tended to increase across all sites receiving harvest treatments while remaining unchanged at true controls the year following logging (Figure 10). Two years after logging, fines decreased across all harvest treatments while increasing slightly at true controls. Fines increased at all sites three years after logging. The only significant pairwise difference occurred between the control in 2000 and the riparian con trol in both 1999 and 2000. Overall, Little Pokegama Creek had lower percentages of fine substrates and lower sand depths (Verry 2001) than the other three streams.

The pattern of increased fines at all harvested sites, but not true controls, following logging suggests that fines might originate from the upland clearcuts, and not from within the 30 m riparian zones. These upland areas receive more logging traffic than the riparian zones. This increased traffic could lead to ground disturbance which could facilitate sediment transport to the streams. Further examination of the data show that the year effect likely explains more of the variation in fines following logging than the harvest treatment. If fines were being transported only from the upland clea rcut areas, there should not be an increase in fines in reaches located upstream from the harvests. The "upstream" reaches at site 2 and 10 are both located above any harvest. Fine substrates

increased in both of these reaches one year after harvest (Fig ure 11), indicating that fines are not only originating from the clearcut areas, but from elsewhere in the basin.

Road crossings located upstream from the study sites have been suggested as a likely source (Merten 1999). These crossings are likely to b e a source of fine sediments for sites 3-8, but if roads were the only source there should not be an increase at sites 1 and 2 where there are not any road crossings and sites 9 -12 which do not have any road crossings immediately upstream. Fine sediments were found to increase at these sites following harvest, suggesting fines are originating from elsewhere in the basin (Figure 12). An ANOVA on these data (from Figure 12) show a significant year effect ($P < 0.001$), with significant differences between 199 7 and all other years, and also between 1998 and 1999. Another possible explanation is that the observed fluctuation in fines is due to yearly flow conditions. It is possible that flows in 1997 or previous years were high and removed many of the fine sub strates, and lower flows in 1998 tended to deposit more. The observed decrease in fine substrates across all streams in 1999 is probably due to a big storm that occurred about a week before the substrate data were collected. The high flows likely had eno ugh energy to scour channels, removing fine sediments. The fluctuations in fines tended to be greater in Little Pokegama Creek and Jack Irving Creek where roads do not provide an extra source of sediments (Figure 12).

Embeddedness

There was a significan t year effect on embeddedness (Table 3). Embeddedness decreased for two years following harvest (Figure 13) and then increased in 2000. The 1999 level was significantly lower than both the 1997 and 2000 levels. The level in 2000 was about the same as the preharvest embeddedness. Embeddedness is highly correlated with the percentage of fine substrates ($R^2 = 0.66$, $P < 0.001$) so it is unclear why there was a decrease in 1998 when fines sediments increased (Figure 9).

Unstable Bank

There were significant t reatment main effects, and treatment by reach and treatment by year interactions with unstable bank (Table 3). Unstable bank tended to decrease with time in the treatments receiving any kind of harvest, while initially decreasing and then increasing in th e control treatments (Figure 14). The only significant pariwise difference on unstable bank was with the control in 2000 and the CTL in 1999 and 2000. The erratic pattern of the controls is likely due to the fluctuations in flow occurring at sites 1 and 9. Decreases in flow at these two shallow control sites caused some flow to be interstitial which increased the area of unstable bank in dry periods. The reduction in unstable bank in the riparian thins (Figure 14) combined with decrease (CTL) or no change (FT) in the unstable bank at “within” reaches (Figure 15) demonstrates little or no bank disturbance from riparian harvesting.

Boulder Pocket, Percentage Coarse Substrates, Overhanging Vegetation

Boulder pocket, percentage coarse substrates, and overhanging vegetation all appear to violate the equal slopes assumption for the ANCOVA (significant interaction with the covariate). The P-values given in table 2 are from the ANCOVA and are

probably unreliable. These habitat variables will not be discussed until a proper analysis technique is determined.

Stream Fish Communities

Fish communities varied among streams (Figure 16). Brook trout (*Salvelinus fontinalis*) were the dominant species in Little Pokegama Creek, and were not found in any other stream. Pokegama Creek North and South had similar assemblages of fish consisting of *Phoxinus*, central mudminnows (*Umbra limi*), brook sticklebacks (*Culaea inconstans*), and fathead minnows (*Pimephales promelas*). Jack Irving Creek contained primarily *Phoxinus* and mudminnows.

The ANCOVA analysis on fish variables produced significant year and harvest treatment main effects, and year by harvest treatment interactions (Table 5). A reach main effect or any interaction containing reach was never significant. This suggests that fish are just as likely to be affected by potential changes in habitat due to forest harvest whether they are immediately downstream from the harvested area or within the harvested zone.

For abundance, ANCOVAs were only conducted on *Phoxinus* and mudminnow abundance in the three non-trout streams. These two species were well represented in all three streams and therefore had a better chance to show changes due to harvest. ANCOVAs conducted on other species produced heterogeneous variances, likely due to patchiness and under-representation.

Mudminnows did not show any significant effects in June (Table 5), however there were significant harvest and year main effects, and a significant harvest by year interaction in August (Table 5). An outlier from a CTL site containing a high abundance of mudminnows in 1998 was removed to produce normal residuals.

The abundance of mudminnows by year (Figure 17) seems to follow the pattern of fine sediments. There was an increase one year after harvest, then a decrease in 1999 followed by an increase in 2000. There were significantly more mudminnows in 2000 than in 1997. This shows an overall increase in mudminnows three years after harvest. Since mudminnows are considered a tolerant species, increases could indicate degraded conditions.

Mudminnows increased across all treatments except CTL (Figure 18). CTL treatments contained significantly fewer mudminnows than the other three harvest treatments. It is possible that the mudminnows moved out of the CTL reaches due to the increase in small WC produced from slash, however if that were the case they should not have returned in 1999 (Figure 19). The only significant difference among treatments occurs in 1998 (Figure 19). The CTL treatment contains significantly fewer mudminnows than both the FT and riparian control. It is this year that is controlling the significant treatment main effect (Figure 18).

There was a significant year main effect, and a year by harvest treatment interaction on *Phoxinus* abundance in June (Table 5). There was only a year by harvest treatment significant interaction in August. *Phoxinus* abundance increased by a factor of 10 at some of the control reaches in 2000. This large yearly difference in abundance did not allow for transformations to cure inherent heteroscadascity. The individual outliers could not all be removed from the analysis because too many degrees of freedom would be

lost. As a result of this, the ANCOVAs on *Phoxinus* only included three years of data (1997-1999).

June *Phoxinus* decreased in the two years following harvest (Figure 20). There was only a significant difference between 1997 and 1999. The only significant difference in harvest treatments occurred in 1999 (Figure 21). The control contained fewer *Phoxinus* than the riparian control and FT treatments. The August sample follows the same pattern (Figure 22) except only the control and riparian control were different. There is no reasonable explanation why the *Phoxinus* would move out of the control reaches.

There were significant differences in IBI scores in August (Table 5). Both year and harvest treatment main effects and the year by harvest treatment interaction were present. IBI scores were highest in the controls (Figure 23). The control treatments had significantly higher IBI scores than the other three treatments. The riparian controls had the second highest IBI scores, which were significantly higher than the two riparian thin treatments. There were no differences among the riparian thin treatm ents. The year effects on IBI (Figure 24) displayed a pattern similar to fine sediments and mudminnows (Figure 17). IBI scores decreased following harvest, increased in 1999 and then decreased again in 2000. The IBI scores in 1999 were significantly dif ferent from all other years. The IBI scores in 2000 were also significantly higher than in 1998. There were no significant effects on June IBI scores. Lack of effects in June could be due to water level. Water levels in early summer are generally highe r than in late summer, providing more fish habitat along the stream. As water levels decline, fish will likely move out of poor quality habitats and into high quality habitats. The excess habitat provided by higher water levels in early summer could have been overriding any negative forest harvest effects. This would indicate that harvest effects are not constant throughout the year.

The harvest treatments all had similar IBI scores before harvest (Figure 25). The control and riparian control treatments scores increase with time, while the FT scores decrease and the CTL scores fluctuate. Three years after harvest both the control and riparian control treatment had higher IBI scores than the riparian thin treatments, but the only significant difference oc curred with the FT treatment. One possible explanation for the decrease in IBI scores in the riparian thin treatments is the observed increase in fine sediments. The riparian thin treatments had IBI scores that decrease with increases in fine sediments (Figure 10 and 25). This explanation does not hold for the control treatment where increases in IBI scores followed increases in fine sediments (Figure 10 and 25). Fredrick and Perry (2001) also observed the same pattern of higher macroinvertebrate biotic integrity with riparian control > CTL > FT.

Some caution should be used with interpretation of the IBI results. Many of these IBI scores were computed on 10 or fewer fish. Mundahl and Simons (1999) recommend that at least 25 fish be used to compute an IBI score. An IBI score is suppose to produce a value that reflects the reach health, while integrating information about all species present. If only a few fish are used, the IBI score might not give a reliable value. In this study IBI scores based on a few fish tended to produce low values. This is a desired result because few fish is usually indicative of impaired systems.

Species richness displayed a significant year by harvest treatment interaction in June, and a significant year effect in August (T able 5). Species richness remained stable in

the control treatment while changing with no apparent pattern in the other treatments (Figure 26). A large increase in richness in the riparian control sites in 1998 is what produced the significant interaction. This richness value was found to be significantly higher than then the CTL richness values in 1998 and 1999, and the FT richness value in 2000.

The species richness values in August do not change much in the first two years (Figure 27). There is a drop in richness in 1999 and a subsequent increase in 2000. The only significant difference occurs between 1999 and 2000. The drop in 1999 represents a loss of about one species system wide. The increase in 2000 was a gain of about 1.7 species. These are quite large fluctuations considering the average August richness value is 2.8. It is possible that the richness fluctuations are being partially controlled by the same processes controlling the fine sediments.

Brook trout abundances had a moderately significant (P = .07) year main effect and a significant year by treatment interaction. The general trend seems to be a decrease in brook trout with time since harvest (Figure 28). The HSD comparison did not find any differences among these means at P = 0.05, but at P = 0.10 there was a difference between the preharvest abundance and the 1999 and 2000 values. Examination of Figure 29 shows that the decrease in brook trout with year is due to what is happening at the riparian control site. Brook trout abundance shows a significant decline following harvest in the riparian control treatment. The numbers continue to decline in 1999 and 2000. The high 1997 abundance is due to the large number of small (likely age 1) brook trout encountered in the downstream reach during both seasons. These juvenile trout were not present in following years and thus the steep decline in abundance was seen. We have not yet determined if larger trout exhibited the same changes. The two riparian thin treatments fluctuate, but return to preharvest levels by 2000. No significant differences were found among the riparian thin treatments.

Temperature and Fish

Stream temperature measurements were continuously recorded from July 1997 through September of 2000. Temperature data were not collected in the same three reaches (“upstream”, “within”, and “downstream”) that fish and habitat data were collected from. The temperature data were collected from the middle of each study plot, and in most cases above all plots, excluding the true controls. As a result of this discrepancy, only data representing similar reach locations were used in analyses. All sites had similar “within” reaches, and most had similar “above” reaches with the exception of the controls. The only available data from a “downstream” reach came from site 2.

An average temperature value for the months of July and August (approx. July 10 – August 31) was calculated. ANOVA indicated that there were significant temperature differences (P < 0.0001) among streams. Little Pokegama Creek and Pokegama Creek North had similar temperatures while Pokegama Creek South and Jack Irving Creek also had similar temperatures (Figure 30). Little Pokegama Creek and Pokegama Creek North had significantly colder temperatures (HSD = 0.05). The maximum temperatures observed during July and August also follow the same pattern. Little Pokegama Creek and Pokegama Creek North had similar maximum temperatures which were lower than Pokegama Creek South and Jack Irving Creek, which also had similar maximum

temperatures (Figure 30).

Spearman rank correlations were used to measure the association among the fish response variables and temperature. Temperature was significantly associated with most fish response variables when all streams were analyzed together (Table 6). As temperature increased, IBI scores decreased, brook trout decreased, and species richness (only in June) increased. This represents the expected pattern in a cold water stream, as water temperature increases, the few types of thermally intolerant species decline, and are replaced with a greater number of species that are tolerant of warmer temperatures. Thus, an increase in fish diversity is usually associated with a decline in the biotic quality of a cold water stream.

It appears that the presence or absence of brook trout is driving the high associations of temperature with the fish response variables. When Spearman rank correlations were conducted on streams not containing brook trout (Jack Irving Creek, Pokegama Creek North and Pokegama Creek South) few associations were present, even though there was a range of temperature values (Table 2). The only significant associations in these streams were opposite of what would be expected. Species richness decreased with increasing temperature, and IBI (only in June) increased with temperature. So in the absence of brook trout the expected patterns seem to fade.

These streams all have average July-August temperatures within the brook trout tolerance range, but Pokegama Creek South and Jack Irving Creek have high daily maximums which may be to warm for brook trout. The absence of brook trout from Pokegama Creek North is likely due to a lack of coarse substrates which are present in Little Pokegama Creek. The distribution of brook trout suggests that if canopy cover is reduced by excessive riparian thinning or windthrow, stream temperatures could rise, and brook trout might leave the area. Also, brook trout would likely leave the area if fine sediments increased as a result of harvest. The most common source of sediments would be from forest roads.

SYNTHESIS AND RELATIONS TO RIPARIAN ZONE MANAGEMENT

Riparian harvest treatments did produce significant changes in some aspects of stream habitat and fish communities. Riparian thinning lead to reductions in canopy cover in both FT and CTL treatments, and increases in woody cover in CTL treatments. The initial reduction in canopy cover due to thinning (Hemstad and Newman 2001) was not enough to significantly raise stream temperatures (Verry 2001). Therefore, leaving a residual basal area of 10m^2/ha (44 ft^2/acre) should protect similar, ground water-fed streams from increases in temperature which can fish and invertebrate communities. However, 10m^2/ha residual basal area does not appear to provide adequate protection from windthrow, especially in areas with high levels of ground water. In the three years following harvest, windthrow increased in the riparian harvest treatments. This increase in windthrow further reduces canopy cover, which could lead to future increases in stream temperature. No increase in windthrow was seen in the riparian control, which suggests that residual basal area rather than RMZ width was the main factor in windthrow. Residual tree densities, therefore, may need to be increased in areas vulnerable to windthrow, to ensure that residual trees are protected.

Woody cover significantly increased in CTL treatments. The origin of the wood was likely from the branches and tops of trees (slash) that were immediately removed with the CTL system. These smaller branches are not included in estimates of coarse woody debris, which did not vary by treatment (Johnson et al. 2001). Future increases in woody cover are likely due to the increase in windthrow. Slash represents smaller woody material than the boles from windthrow. It is unknown how the smaller slash will affect the fish community, but in shallow reaches with homogeneous sand cover, even small woody material should create a more diverse habitat by altering flow regimes. Also, the wood represents additional substrate that macroinvertebrates can utilize. However, large inputs of slash, which were occasionally observed, could clog small stream channels impeding fish migrations. If it is desired to reduce slash inputs when riparian areas are harvested, trees should be felled and slash removed away from the channel or the residual tree density should be increased.

One of the goals in protecting a riparian zone is to provide a constant supply of allochthonous organic material (leaves and wood). Thinning the riparian zone to 10m^2/ha seems to alter this constant supply. There was an initial increase in slash followed by an increase in larger wood due to windthrow. This represents a significant increase in the first few years following harvest. Windthrow will continue to contribute wood debris as these trees drop to the bankful channel (Johnson et al. 2001). It is, however, likely that long-term inputs of large wood will decrease due to reduction in riparian trees from both harvest and windthrow. Protecting against windthrow should help to provide a more constant supply of woody material.

The 30 m width of the riparian zone seems to be appropriate for protecting trees when there is no riparian thinning. Using a wider width could help protect riparian trees when they are thinned. Also, a wider riparian zone might be needed in order to supply a constant amount of leaf inputs during the first years after harvest (Palik 2001). More investigation of the interactive effects of RMZ width and basal area remaining is warranted.

Filter strips are designed to reduce fine sediment input to streams. We saw increases in fine sediment one-year post harvest associated with riparian harvest treatment, however, there was considerable year-to-year variation in fine sediment across all sites. Fine sediment increased the year after harvest and varied thereafter. It appears that year-to-year variation in sediment input, or basin-level increases in sediment due to logging activities or traffic are more important than local riparian harvest. We found no increase in unstable bank associated with riparian harvest. Increased fine sediment is deleterious to coldwater fish and year to year changes in fine sediment were negatively related to fish biotic integrity, underscoring the importance of minimizing fine sediment input to streams. Within the study system, brook trout were only present in streams with a high percentage of cobble and boulder; they were absent from streams with high percentage sand and fine sediment. The cumulative effects of forest harvest and road activities, even those off site, should be considered in forest harvest guidelines.

Stream temperature was not affected by forest harvest treatment (Verry 2001), however, temperature, along with substrate, may be the main factor limiting the distribution of brook trout. Although the mean summer and maximum temperature differences among the four streams were not large, these small differences in combination

with fine sediment likely determine the presence of brook trout. The effects of harvest to a basal area less than $10m^2$/ha on stream temperature should be evaluated and practices to prevent increases in fine sediments must be encouraged.

Fish responses to harvest treatments were variable. Mudminnows, which are tolerant to high temperature and low oxygen, increased in full tree harvest sites but decreased in cut-to-length sites. Dace, which are also tolerant, tended to decrease at unharvested control sites and varied less at the harvested sites. The fish Index of Biotic Integrity in August did decrease with level of harvest: unharvested controls > riparian controls > CTL > FT. A similar pattern was seen with benthic invertebrates (Fredrick and Perry 2001) and the changes in IBI and mudminnows over time mirrored changes in fine sediment. Whole stream increases in fine sediment were associated with increased mudminnows and decreases in IBI, and mudminnows decreased and IBI increased with decreases in fine sediment. These results suggest some local effects on stream fish and invertebrate communities with increasing degree of harvest disturbance, but the mechanisms of these changes are unclear. It is likely that combinations of changes in substrate, canopy cover, woody material and invertebrate community composition are affecting stream fish biotic integrity. For example, the distribution of the IBI scores in August of 2000 suggests that the FT system, which had the highest levels of windthrow, had the greatest decrease in IBI difference scores, followed by the CTL. Increased windthrow may increase light and degrade instream habitat. Year-to-year differences, partly associated with changes in fine sediment, appear equally important. The relatively short distance of the riparian harvests (135 to 200m of stream; Palik 2001) probably minimized the effects on fish communities and length of harvested reach should be considered in future investigations.

In conclusion, the goal of creating protected riparian areas is to maintain the natural processes of these transitional areas between land and water. Thinning riparian zones to a basal area of $10m^2$/ha with a width of 30 m does not appear to fully protect all natural processes in this stream system. One problem appears to be the less than adequate protection from windthrow, which increases with time. Changes in canopy coverage, litter inputs, fine sediments and invertebrate communities may be influencing fish community biotic integrity. Because harvest occurred on both sides of the stream we conducted a rigorous test of the guidelines; harvests of similar lengths on only one side of the stream should show lesser effects. Two major factors not addressed in this study were system-wide or cumulative effects of harvest and length of streamside harvested. These factors may be more important to maintaining stream communities than local impacts at least for harvest lengths < 200m.

CONCLUSIONS

Forest harvest significantly affected some fish and habitat variables. Canopy cover decreased in both CTL and FT treatments, and WC increased in CTL treatments. Windthrow trees were most numerous at FT treatments, but also occurred at CTL treatments. Fines substrates tended to increase in CTL, FT and riparian controls while remaining unchanged in controls one year after harvest. Bank stability was not affected by riparian thinning, but did decrease in controls. IBI scores were highest in controls,

followed by riparian controls three years after harvest during August collecti ons. Mudminnows decreased in CTL treatments one year after harvest.

Overall, undisturbed riparian zones (controls, riparian controls) seem to be more effective at reducing disturbances to instream habitat and fish communities than riparian thins (CTL, FT). Establishing an undisturbed riparian zone near streams eliminates canopy cover loss, inputs of WC from slash, and reduces susceptibility to windthrow. Also, undisturbed riparian zones appear to provide habitat for higher quality (as determined by IBI scores) fish communities than riparian thins in late summer. If the increase in fine sediments seen at all harvested sites one year after logging is originating from the upland clearcut, a 30 m buffer strip does not seem to be wide enough to reduce sediment inputs.

Harvest treatments produced changes in some habitat variables that have straightforward explanations (e.g., canopy cover and WC). Other habitat variables showing a treatment effect (either main or interaction) do not have an easy explanation (e.g., percent fines and unstable bank). In cases without a clear explanation for the harvest effect, it is likely that year is causing most of the variation. Year has a significant effect (either main or interaction) in all variables (Tables 4 and 5). Th is also seems to be the case for some fish variables, where there were not any clear explanations due to harvest treatment, although significant harvest treatment effects do exist.

There does seem to be a larger scale year effect occurring throughout the system. Percentage fine substrates, embeddedness, mudminnow abundance and August IBI scores all seem somewhat related. It is possible that changes in fine substrates are causing the changes in these other variables. If this is the case, then what is causing the changes in fines? Road crossings above some study sites are a possible explanation, but not all sites have crossings. Another possible explanation is annual differences in stream flow and the associated sediment load transported. Finally th ere is the treatment effect. It is likely that a combination of these factors is causing the effect, but when there is not a clear explanation for treatment effects year effects are probably causing more variation.

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Table 1. Description of harvest treatments and corresponding study sites. FT = full tree system, CTL = cut-to-length system

Harvest Treatment	30 M Riparian zone	Upland	Site #	Stream
Contol	No cut	No cut	1 7 9	Jack Irving Creek Pokegama Creek South Little Pokegama Creek
Riparian Control	No cut	Clearcut with CTL	3 5 12	Pokegama Creek North Pokegama Creek North Little Pokegama Creek
CTL Riparian Thin	Thinned to 44 ft^2/acre basal area with CTL	Clearcut with CTL	2 8 11	Jack Irving Creek Pokegama Creek South Little Pokegama Creek
FT Riparian Thin	Thinned to 44 ft^2/acre basal area with FT	Clearcut with FT	4 6 10	Pokegama Creek North Pokegama Creek South Little Pokegama Creek

Table 2. Habitat and fish variables used in analyses. J = June sample, A = August sample.

Habitat	
% Canopy Cover	% Boulder Pocket
% Unstable Bank	% Embeddedness
% WC	% Overhanging vegetation
% Fine Substrates	% Coarse Substrates
Windthrow trees	
Fish	
J IBI	J Mudminnow Abundance
A IBI	A Mudminnow Abundance
J Species Richness	J *Phoxinus* Abundance
A Species Richness	A *Phoxinus* Abundance
	Brook trout Abundance

Table 3. Sample ANCOVA output.

Source	Nparm	DF	Sum of Squares	F Ratio	Prob>F
treatment	3	3	288.888	1.0965	0.3572
reach	1	1	0.015	0.0002	0.9895
year	3	3	860.737	3.2671	0.0270
treatment*reach	3	3	270.488	1.0267	0.3869
reach*year	3	3	9.827	0.0373	0.9903
treatment*year	9	9	1777.582	2.2491	0.0299
treatment*reach*year	9	9	123.851	0.1567	0.9974
97fines (covariate)	1	1	13275.929	151.1765	<.0001

Table 4. P-values of main effects and 2 way interactions of habitat ANCOVAs. Bold indicates significance (P < 0.05). All 3 way interactions were non-significant and were removed (models subsequently reduced). The covariate is the preharvest data (1997). H = harvest treatment, Y = year, R = reach, WC = coarse woody debris, - means no significant 2 way interactions so model reduced to main effects.

		Main effects			2 Way Interactions		
Habitat Variable	Covariate	H	Y	R	HxY	HxR	YxR
% Canopy Cover	**<0.0001**	**<0.0001**	**0.0450**	**<0.0001**	**0.0003**	**<0.0001**	**0.0043**
% Unstable Bank	**<0.0001**	**0.0112**	0.5285	0.3445	**0.0011**	**0.0192**	0.9955
% WC	**<0.0001**	**<0.0001**	**0.0016**	**0.0040**	**0.0181**	**0.0002**	0.5186
% Fine Substrates	**<0.0001**	0.3066	**0.0164**	0.9889	**0.0146**	0.3358	0.9886
% Coarse Substrates[1]	**<0.0001**	0.8559	0.2943	0.9775	**0.0158**	0.8471	0.9562
% Boulder Pocket[1]	**< 0.0001**	0.1141	**0.0016**	0.6401	-	-	-
% Embeddedness	**<0.0001**	0.1749	**0.0058**	0.7302	-	-	-
% Overhanging vegetation[1]	**<0.0001**	0.3114	**0.0172**	0.2794	-	-	-

1 Possible violation of equal slopes assumption for ANCOVA

Table 5. P-values of main effects and 2 way interactions of fish ANCOVAs (non trout streams). Bold indicates significance (P < 0.05). All 3 way interactions were non-significant and were removed (models subsequently reduced). The covariate is the preharvest data (1997). A = August sample, J = June sample, H = harvest treatment, Y = year, R = reach, IBI = Index of Biotic Integrity, - means no significant 2 way interactions so model reduced to main effects.

		Main effects			2 Way Interactions		
Fish Variable	Covariate	H	Y	R	HxY	HxR	YxR
IBI J	0.7827	0.1808	0.5218	0.5925	-	-	-
IBI A	**0.0038**	**0.0006**	**<0.0001**	1.000	**<0.0001**	0.7251	0.9201
Species Richness J	**0.0558**	0.3808	0.8448	0.8820	**0.0472**	0.7439	0.8868
Species Richness A	**0.0004**	0.1344	**0.0055**	0.7381	-	-	-
Mudminnow J	0.6253	0.1741	0.6496	0.9988	-	-	-
Mudminnow A	**0.0015**	**0.0004**	**0.0066**	0.1131	**<0.0001**	0.7395	0.2619
Phoxinus J[1]	**0.0022**	0.5652	**0.0367**	0.7638	**0.0048**	0.9764	0.6032
Phoxinus A[1]	**0.0003**	0.2215	0.4274	0.2866	**0.0010**	0.8996	0.1823

1 used only 3 years (97-99)

Table 6. Spearman' s rank correlations with temperature and fish variables across all four streams. IBI = Index of Biotic Integrity, bk = brook trout abundance, bs = brook stickleback abundance, mm abundance = mudminnow, total = total fish abundance, rich = species richnes s, 1 = June sample, 2 = August sample.

Fish Variable	Spearman's Rho	Prob >|Rho|
IBI1	-0.2213	0.05
IBI2	-0.3719	0.0007
bk1	-0.481	<.0001
bs1	0.2662	0.0177
mm1	0.2823	0.0117
total1	0.1094	0.337
rich1	0.1973	0.0814
phox1	0.163	0.1511
bk2	-0.5103	<.0001
bs2	0.1676	0.1398
mm2	0.2307	0.0408
total2	-0.0126	0.9123
rich2	0.008	0.9443
phox2	0.0188	0.8691

Table 7. Spearman's rank correlations with temperature and fish variables across the three stream without brook trout. IBI = Index of Biotic Integrity, bs = brook stickleback abundance, mm abundance = mudminnow, total = total fish abundance, rich = species richness, 1 = June sample, 2 = August sample.

Fish Variable	Spearman's Rho	Prob > >|Rho|
IBI1	0.2611	0.0542
IBI2	0.0488	0.7237
bs1	-0.1483	0.28
mm1	-0.1965	0.1505
total1	-0.3055	0.0233
rich1	-0.3366	0.012
phox1	-0.2144	0.116
bs2	-0.2047	0.1339
mm2	-0.0984	0.4746
total2	-0.2489	0.0669
rich2	-0.3376	0.0117
phox2	-0.328	0.0145

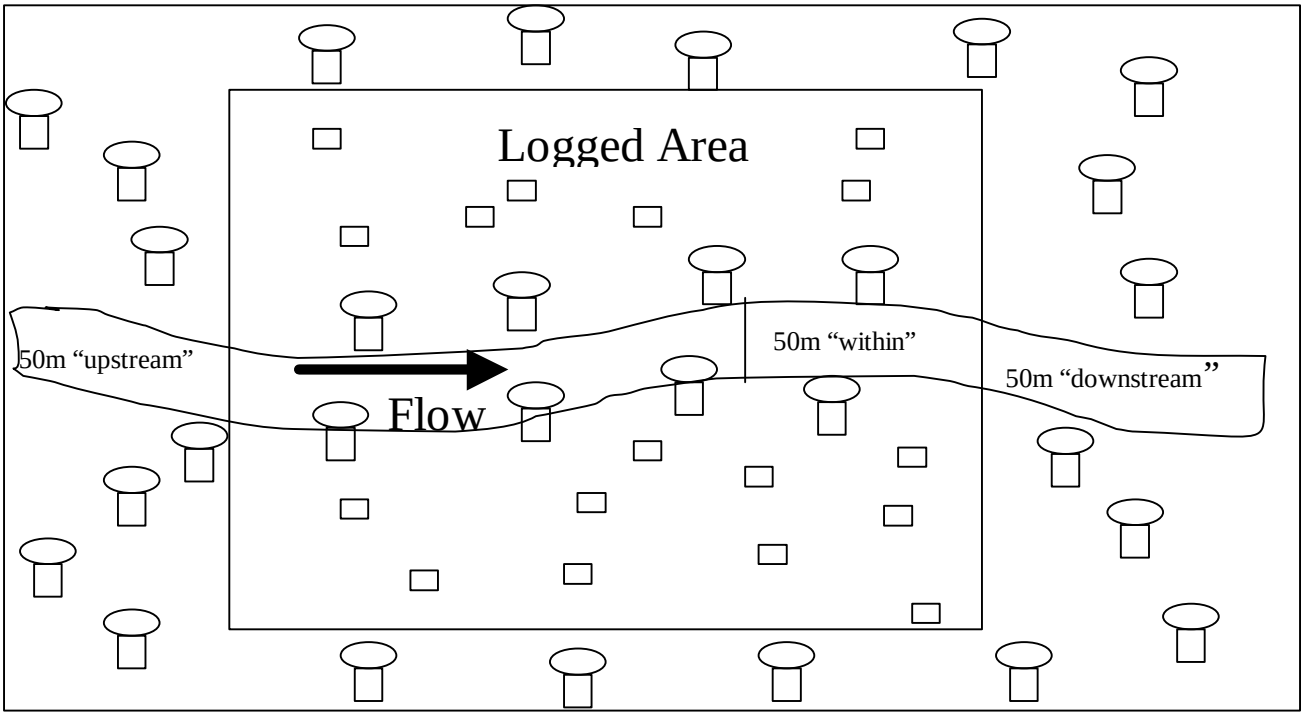

Figure 1. Reach locations along a hypothetical site.

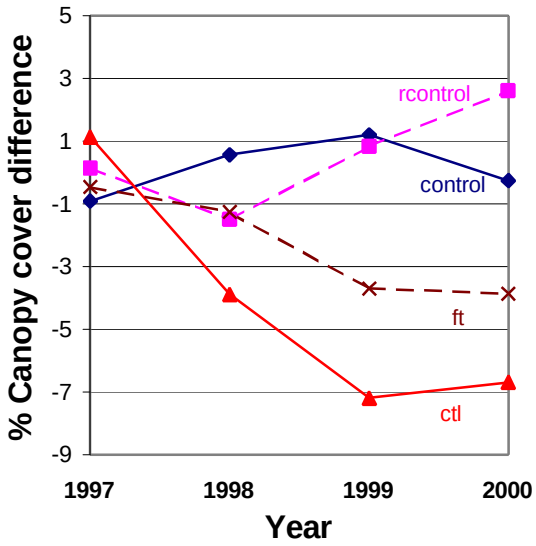

Figure 2. Percentage canopy cover by year and harvest treatment. Y axis is calculated by subtracting "upstream" reach values f rom "within" and "downstream" values.

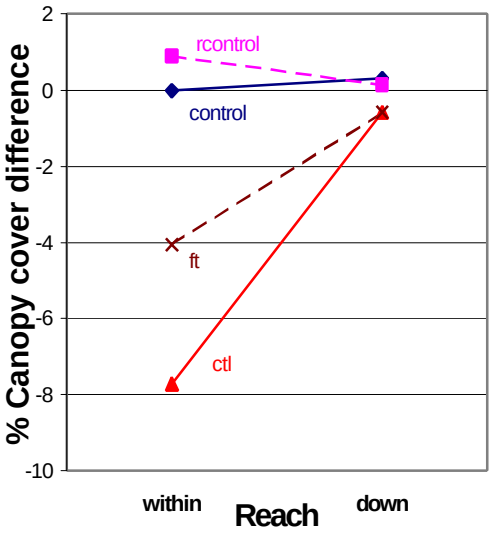

Figure 3. Percentage canopy cover by reach and harvest treatment. Y axis is calculated by subtracting "upstream" reach values from "within" and "downstream" values.

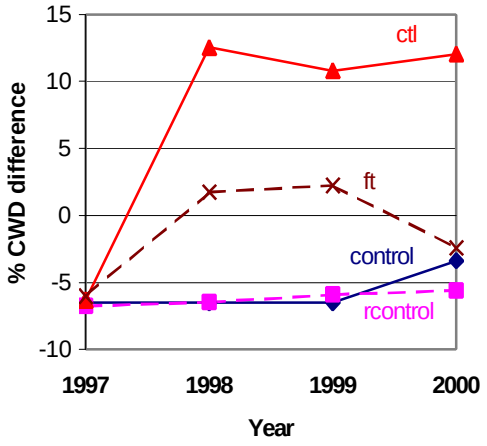

Figure 4. Percentage woody cover (WC) by year by reach and harvest treatment. Y axis is calculated by subtracting "upstream" reach values from "within" and "downstream" values. ctl = cut-to-length, ft = full tree, rcontrol = riparian control.

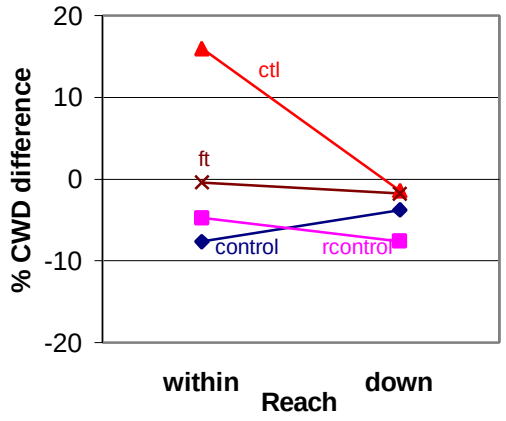

Figure 5. Percentage woody cover (WC) by reach and harvest treatment. Y axis is calculated by subtracting "upstream" reach values from "within" and "downstream" values. ctl = cut -to-, length, ft = full tree, rcontrol = riparian control.

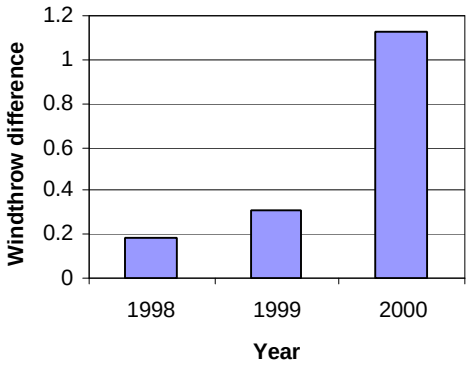

Figure 6. Year main effect on windthrow trees. Y axis is calculated by subtracting "upstream" reach values from "within" and "downstream" values.

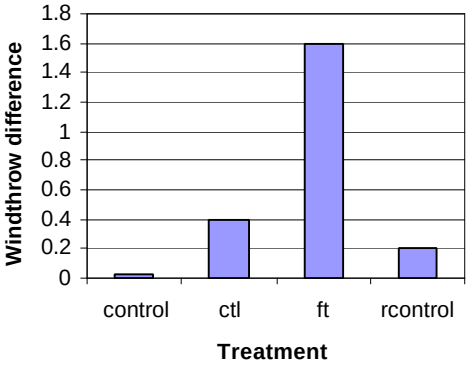

Figure 7. Harvest main effect on width row trees. Y axis is calculated by subtracting "upstream" reach values from "within" and "downstream" values. ctl = cut-to-length, ft = full tree, rcontrol = riparian control.

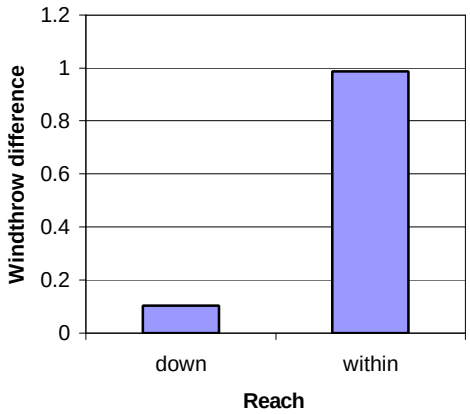

Figure 8. Reach main effect on windthrow trees. Y axis is calculated by subtracting "upstream" reach values from "within" and "downstream" values.

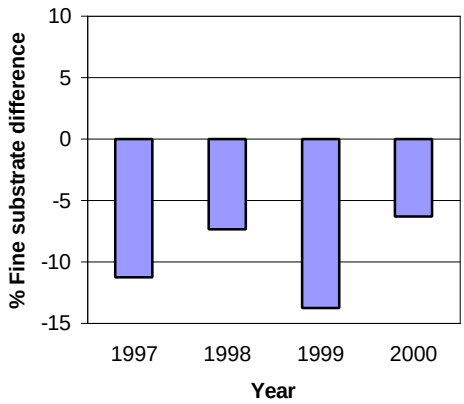

Figure 9. Year main effect on percent fine substrate. Y axis is calculated by subtracting "upstream" reach values from "within" and "downstream" values.

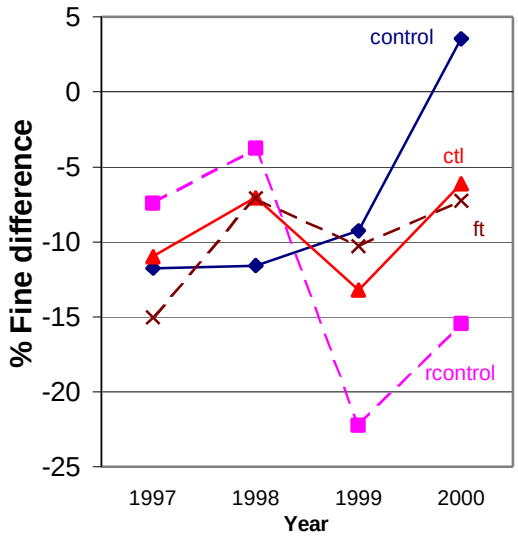

Figure 10. Percent fine substrate by year and harvest treatment. Y axis is calculated by subtracting "upstream" reach values from "within" and "downstream" values. ctl = cut-to-length, ft = full tree, rcontrol = riparian control.

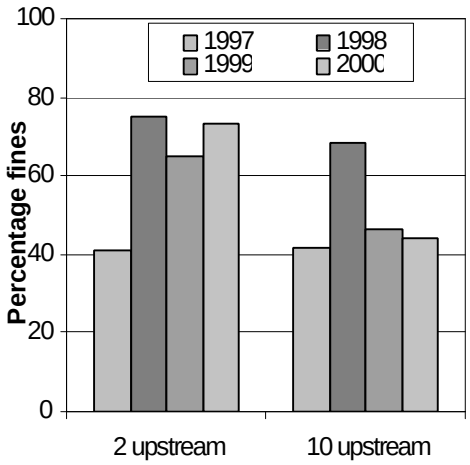

Figure 11. Percentage fine substrates by year at two sites located above any harvest.

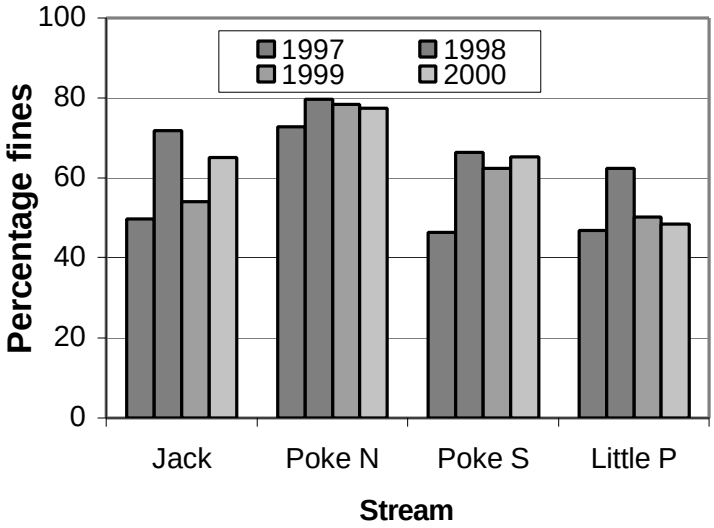

Figure 12. Percentage fine sediments by year in the four study streams. Jack = Jack Irving Creek, Poke N = Pokegama Creek North, Poke S = Pokegama Creek South, Little P = Little Pokegama Creek.

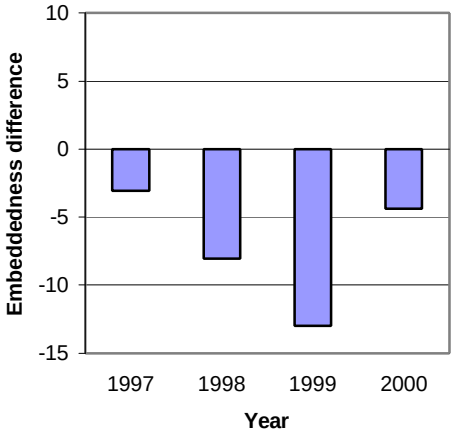

Figure 13. Year main e ffect on embeddedness. Y axis is calculated by subtracting "upstream" reach values from "within" and "downstream" values.

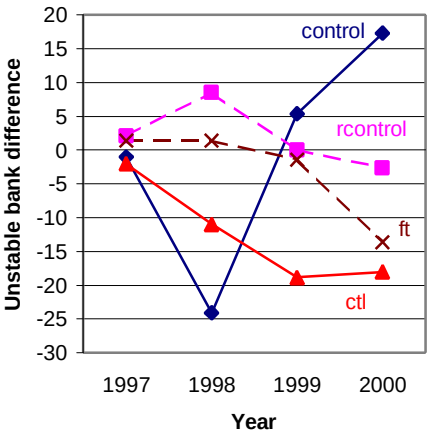

Figure 14. Unstable bank by harvest treatment and year. Y axis is calculated by subtracting "upstream" reach values from "within" and "downstream" values. ctl = cut-to-length, ft = full tree, rcontrol = riparian control.

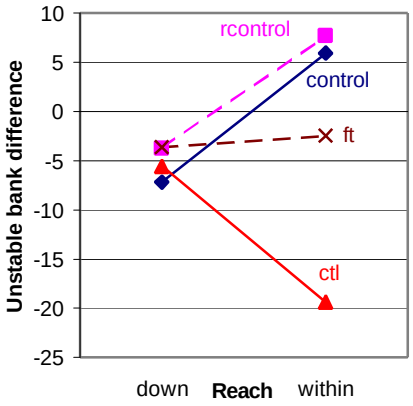

Figure 15. Unstable bank by reach and harvest treatment. Y axis is calculated by subtracting "upstream" reach values from "within" and "downstream" values. ctl = cut-to-length, ft = full tree, rcontrol = riparian control.

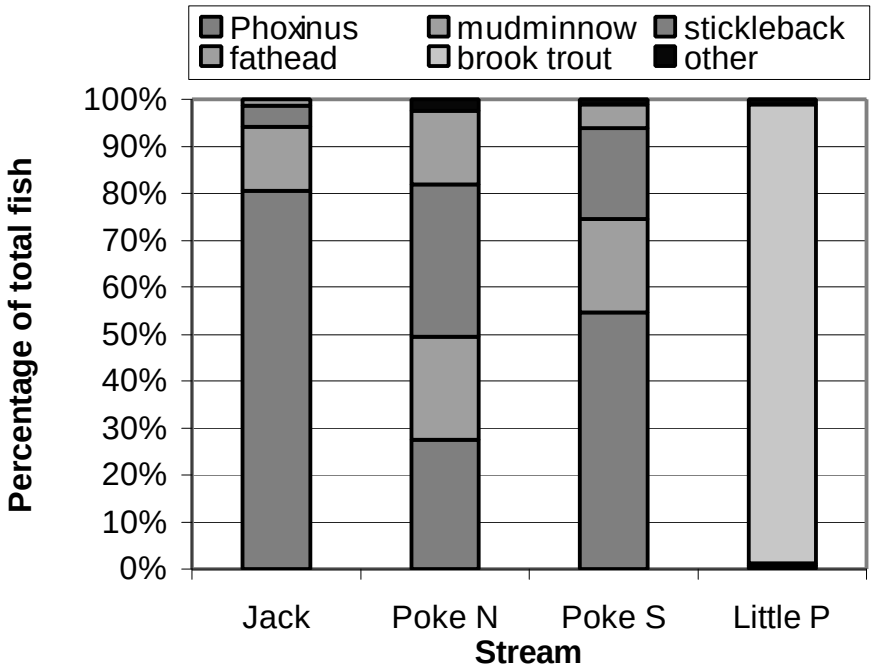

Figure 16. Fish communities in the four study streams averaged over all four years. Jack = Jack Irving Creek, Poke N = Pokegama Creek North, Poke S = Pokegama Creek South, Little P = Little Pokegama Creek.

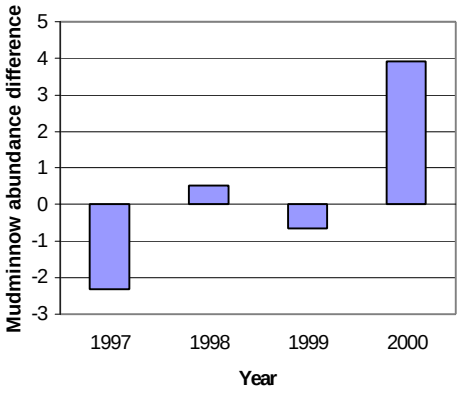

Figure 17. Year main effect on mudminnow harvest abundance. Y axis is calculated by subtracting subtracting "upstream" reach values from "within" and "downstream" values. August sample.

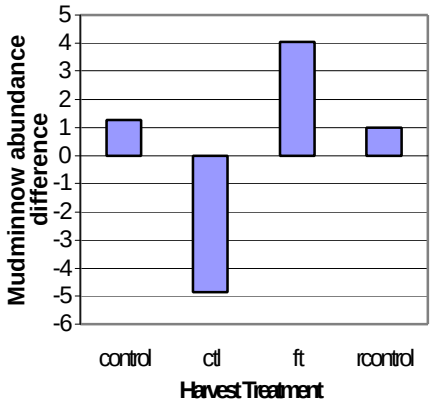

Figure 18. Mudminnow abundance by treatment. Y axis is calculated by "upstream" reach values from "within and "downstream" values. ctl = cut-to-length, ft = full tree, rcontrol = riparian control. August sample.

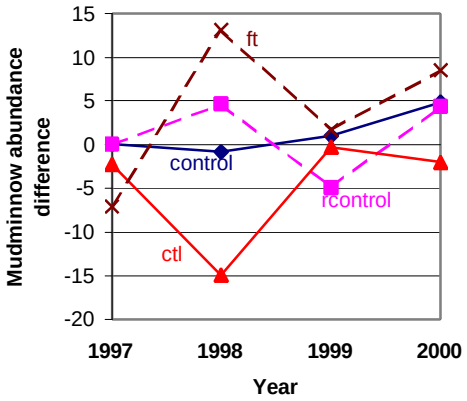

Figure 19. Mudminnow abundance by y ear and harvest treatment. Y axis is calculated by subtracting "upstream" reach values from "within" and "downstream" va lues. ctl = cut-to-length, ft = full tree, rcontrol = riparian control. August sample.

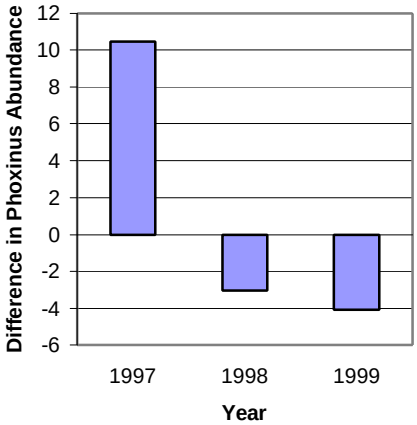

Figure 20. *Phoxinus* abundance by year. Y axis is calculated by subtracting "upstream" reach values from "within" and "downstream" values. June sample.

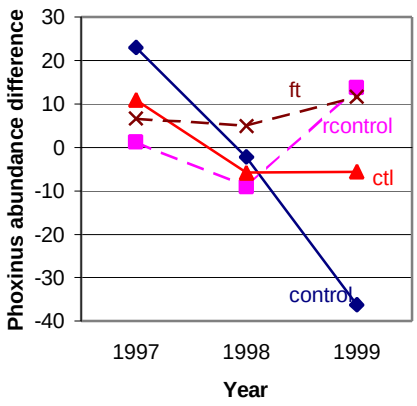

Figure 21. *Phoxinus* abundance by year and harvest treatment. Y axis is calculated by subtracting "upstream" reach values from "within" and "downstream" values. ctl = cut-to-length, ft = full tree, rcontrol = riparian control. June sample.

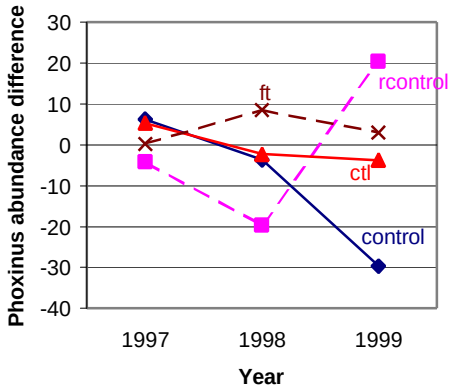

Figure 22. *Phoxinus* abundance by year and harvest treatment. Y axis is calculated by subtracting "upstream" reach values from "within" and "downstream" values. ct l = cut-to-, length, ft = full tree, rcontrol = riparian control. August sample.

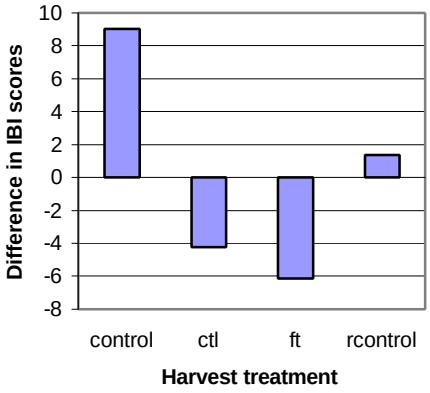

Figure 23. Index of Biotic Integrity (IBI) scores by harvest treatment. Y axis is calculated by subtracting "upstream" reach values from "within" and "downstream" values. August sample.

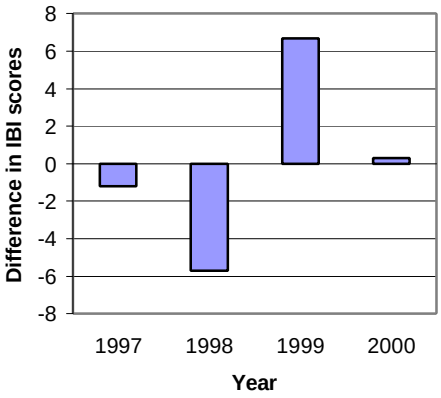

Figure 24. Index of Biotic Integrity (IBI) scores by year. Y axis is calculated by subtracting "upstream" reach values from "within" and "downstream" values. August sample.

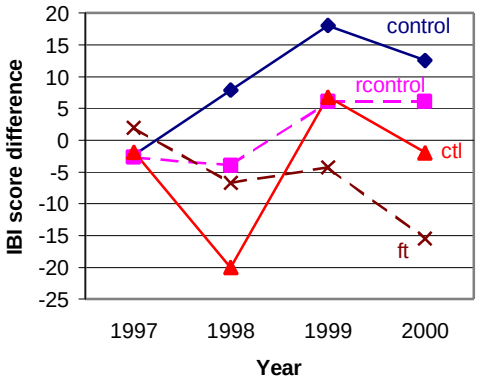

Figure 25. Index of Biotic Integrity (IBI) scores by year and harvest treatment. Y axis is calculated by subtracting "upstream" reach values from "within" and "downstream" values. ctl = cut-to-length, ft = full tree, rcontrol = riparian control. August sample.

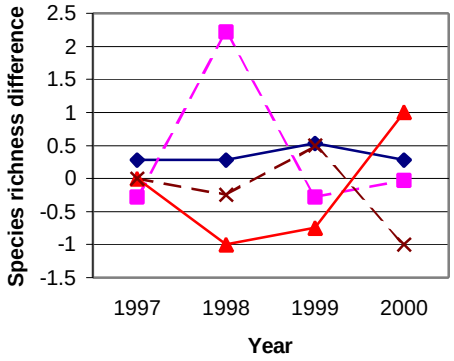

Figure 26. Species richness by year and harvest treatment. Y axis is calculated by subtracting "upstream" reach values from "within" and "downstream" values. ctl = cut-to-length, ft = full tree, rcontrol = riparian control. June sample.

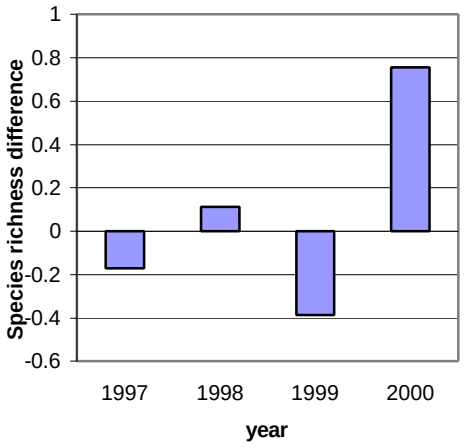

Figure 27. Species richness by year. Y axis is calculated by subtracting "upstream" reach values from "within" and "downstream" values. August sample.

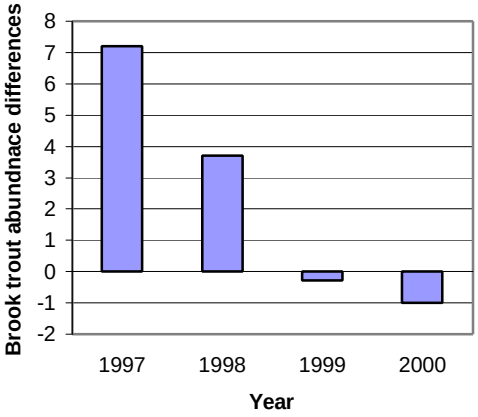

Figure 28. Brook trout abundances by year. Y axis is calculated by subtracting "upstream" reach values from "within" and "downstream" values. Combination of both June and August samples.

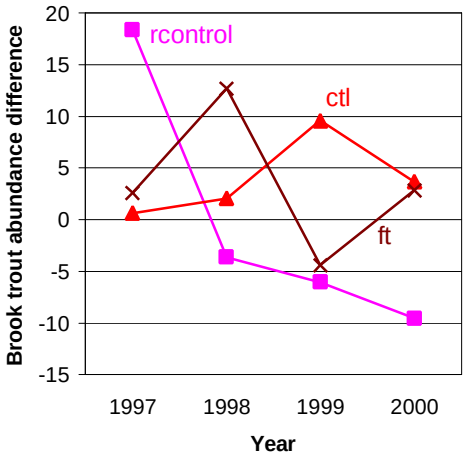

Figure 29. Brook trout abundance by year and harvest treatment. Y axis is calculated by subtracting "upstream" reach values from "within" and "downstream" values. ctl = cut-to-length, ft = full tree, rcontrol = riparian control. Combination of both June and August samples.

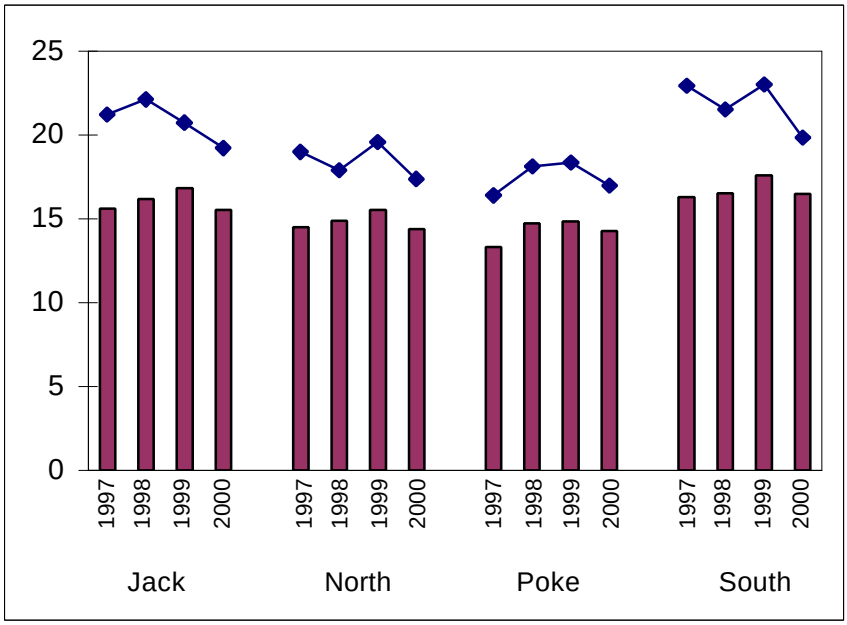

Figure 30. Comparison of average yearly temperatures and maximum temperatures at all four study streams. Jack = Jack Irving Cr eek, North = Pokegama Creek North, Poke = Little Pokegama Creek, South = Pokegama Creek South, bars = average yearly temperature, line = maximum daily temperature. All temperatures from July -August.

TIMBER HARVESTING EFFICIENCY AND STAND DAMAGE

James A. Mattson and Michael A. Thompson

Abstract

The full-tree (FT) harvesting system exposed more organic and mineral soil than did the CTL system. The CTL system left considerably more slash on the site. The full-tree system traffics a greater portion of the plot. Untrafficked slash is more evident on the CTL plots. There is evidence of greater disturbance (e.g., rutting) in limited situations. Analysis is ongoing to identify the harvest and traffic patterns that lead to that level of disturbance.

Introduction

Assessing the effects of forest operations on the site is an important first step in evaluating the significance of impacts on the health of the forest ecosystem. The majority of the effort devoted to operational questions in phase I of this project were devoted to studying the operational site impacts of the harvesting operations. Most riparian areas have a gradient of site conditions ranging from upland conditions away from the water body to conditions that may be characterized as wetlands near the water. Therefore, the most serious constraints placed on harvest system selection and operation, with the possible exception of unstable soils on steep slopes, are the conditions occurring closest to the water body where high soil moisture conditions are the predominant consideration.

Minimizing damage to residual trees and plants is critical in all-age management systems where partial cuts are applied on a regular basis. Logging damage most often occurs in the first log of the tree, which is usually the most valuable. The trees selected for saving in a partial cut are usually those species and individuals that have the most potential for growing high valued products. Any logging damage incurred can then have a significant impact on the long-term economics of the management system.

Objectives

We posed two objectives:

1. Determine if the imposition of additional constraints in the riparian area would have an effect on the operational efficiency of the harvesting systems, and
2. Assess the extent and nature of damage to residual trees and the surface soils of sites harvested in different ways.

Methods

Site impacts were studied over a gradient of conditions from the stream to the upland areas of the study plots, with the emphasis on the riparian areas.

Harvesting System Efficiency

The harvesting systems used in this study were a conventional full-tree (FT-)system consisting of a feller-buncher, grapple skidder, and roadside slasher, and a cut–to-length (CTL) system consisting of a rubber-tired single grip harvester and a rubber tired forwarder. The specific equipment used in the full-tree system were a Timbco 425B tracked feller-buncher with a Quadco 22 inch high-speed saw head, and a John Deere 648E grapple skidder. The cut-to-length system used either a Valmet 546 Woodstar Series II harvester, or a Ponsse Cobra HS10

harvester, in conjunction with a Valmet 546 Woodstar Series II forwarder.

Three different applications of the harvesting systems were evaluated. In the first treatment (plots 4, 6 and 10), the FT-system was used to harvest both the upland and riparian areas of the study plot. In the second treatment, the CTL system was used to harvest both the upland and riparian areas of the study plot. This system was used for plots 2, 8, and 11, except that the FT-system was used to harvest the upland portion of plot 2. This plot was harvested last, after the harvesting equipment had been moved to other jobs, and the contractor requested that we use the FT-system on the upland. In the third treatment (plots 3, 5 and 12), the CTL system was used to harvest only the upland portion of the study plot, leaving the riparian area uncut as a control.

Standard time and motion study methods were used to study a sample of the harvesting operation. The large size of the overall project (i.e., approximately 6000 cords of wood harvested, a large number of machines operating simultaneously) made it impractical to attempt to do a 100 percent time study. Most of the actual harvesting operation was a fairly straight-forward application of standard equipment, and the productivity and cost of the operation would not be expected to vary from established industry standards for productivity and cost. It was originally envisioned that the riparian area and the upland area in each study plot would be harvested separately, and the productivity and cost of the operation could be determined separately for the two areas. However, it quickly became obvious that the most efficient approach for the contractor was to simultaneously harvest the riparian area along with the associated upland area. Data were collected on a sample of the operations in both the upland and riparian areas. This information and supporting information on the normal efficiency of these systems and the additional constraints that working in a riparian area place on the harvesting system allow us to develop estimates of the productivity and costs of the operation, and to identify any additional costs incurred as a result of harvesting in the riparian area.

Residual Stand Damage

After logging was completed in the study plots that had a harvesting operation in the riparian area, a 100 percent inventory of the residual trees in the riparian area was conducted. Damage to the crowns, boles, and roots was documented on all trees 5 inches dbh and larger. Damage to a total of 321 trees was recorded. Analyses to categorize all damage with respect to location, type of damage, and severity are ongoing. Ultimately, plot by plot analyses will look at differences that may be attributed to the harvest system.

Site Impacts

We employed a field method we previously developed specifically for application to these settings. We visually assessed soil disturbance resulting from forest operations to evaluate the site impacts created by the harvesting operation. The method relies on three levels of assessment, with an associated height or depth measurement for some disturbance categories. The first level of assessment uses variables visible on the soil surface (e.g, litter, slash, mineral soil). The second level attempts to answer the question *What happened here?*, using categories such as undisturbed, trafficked and rutted.. The third level of assessment attempts to identify larger features of the site such as landings, roads and skid trails that define the patterns of impacts on the site. The method can be used with any number of sampling schemes and is reproducible by different observers. The method does not produce absolute measures of site impacts, but is

particularly appropriate for making comparative evaluations of different systems, such as was done in this project.

The impact evaluations done in this project were based on a series of transects established in each study plot. Twelve to sixteen transects, each nominally 100 meters long, were established in each study plot. Each transect began at the stream and extended out approximately parallel to the plot boundary. Transect locations were chosen to coincide with the transects established for measurement of vegetation on the plots so that collaborative analyses could be done between harvesting impacts and vegetative response. Along each transect at intervals down to 0.1 meter, the visible layer, disturbance evaluation, and main site features were recorded.

Results

Preliminary information on the soil visible layer is summarized in Table 1. It appears evident that the FT-system exposed more organic and mineral soil than did the CTL system. This would be reasonable to expect since the grapple skidding involves dragging loads of full-trees or tree-length stems on the ground which would tend to scrape away the litter layer and expose the underlying soil layers. A second evident trend is that the CTL system left considerably more slash on the site. Slash, in this assessment system, is defined as accumulations of logging debris so thick that the underlying soil surface cannot be seen. This is typical of the operation of CTL systems (i.e., "windrows" of slash are left in the stand due to the pattern that the harvesters will typically follow as they work their way through the stand).

Table 1. Soil visible layer by treatment (percent of total transect length)

		Riparian: Control Upland: CTL	Riparian: FT Upland: FT	Riparian: CTL Upland: CTL
Litter	Riparian Upland	94.7 41.7	78.4 51.7	67.0 49.2
Organic	Riparian Upland	0.1 7.4	6.7 15.7	5.3 5.7
Mineral	Riparian Upland	0.0 0.5	1.9 3.1	0.4 0.4
Slash	Riparian Upland	1.7 49.4	9.5 28.4	23.8 43.7
Non-soil	Riparian Upland	3.5 1.0	3.5 1.0	3.5 1.1

Preliminary information on the disturbance evaluation is summarized in Table 2. Again, some trends appear evident. The FT-system, again because of the skidder travel, tends to traffic a greater portion of the plot. Skidder loads are smaller, requiring more trips and, particularly in the uplands where travel is not restricted, skidders will tend to vary their traffic patterns, always minimizing their travel distance. Untrafficked slash is more evident on the CTL plots, a product of the more patterned approach generally taken by a CTL system.

A third category undergoing further analysis is the one termed “disturbed” in Table 2. This category includes the most severe forms of site impacts such as rutting. The amounts of disturbance that were recorded appear to be within reasonable limits for typical harvesting operations (i.e., three percent or less). However, the FT-system did appear to generate about twice the amount of disturbance that the CTL system. This category analysis will provide the opportunity to develop mitigation strategies for any harvest effects that are identified.

Table 2. Disturbance evaluation by treatment (percent of total transect length)

		Riparian: control Upland: CTL	Riparian: FT Upland: FT	Riparian: CTL Upland: CTL
Undisturbed litter	Riparian Upland	93.9 11.5	61.9 11.6	45.5 18.9
Trafficked	Riparian Upland	0.0 47.0	20.9 67.2	20.8 38.1
Disturbed	Riparian Upland	0.0 1.7	3.0 3.4	1.6 1.6
Scarified	Riparian Upland	0.9 9.2	4.2 4.0	9.0 10.2
Untrafficked slash	Riparian Upland	1.7 29.7	6.5 12.8	19.5 30.1
Non-soil	Riparian Upland	3.5 1.0	3.5 1.0	3.5 1.1