EFFECTS OF MANAGED BUFFER ZONES ON FAUNA AND HABITAT ASSOCIATED WITH A HEADWATER STREAM IN THE INDIAN BAY WATERSHED IN NORTHEAST NEWFOUNDLAND

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Effects of managed buffer zones on fauna and habitat associated with a headwater stream in the Indian Bay watershed in northeast Newfoundland

By

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Abstract

The effectiveness of managed buffer zones in protecting an aquatic ecosystem during forest harvesting was studied for a two year period on a small headwater stream in northeastern Newfoundland, Canada. The study consisted of examining several components including abiotic (water temperature and sedimentation) and biotic (macroinvertebrates and salmonids). These components were studied pre- and post-harvest to determine the impact of the following riparian management schemes: 20 m no harvest buffer; 20 m buffer with 30 % of the basal area harvested; 30-50 m buffer with 30 % of the basal area harvested; and a no harvest 'control' site.

Sedimentation significantly increased for the 20 m buffer with selective harvesting. Water temperature was slightly impacted within the optimum temperature class for brook trout (*Salvelinus fontinalis*) only with a significant decrease for the 30-50 m buffer with selective harvesting and the 20 m buffer with selective harvesting. The stress and lethal temperature classes were not significantly different between pre- and post-harvest observations. The water temperature significantly increased within the upper and lethal temperature classes for Atlantic salmon (*Salmo* salar) within the 30-50 m buffer with selective harvesting.

The effects of selective harvesting on aquatic macroinvertebrates varied depending on the index and taxon. The number of Ephemeroptera, Plecoptera, and Trichoptera (EPT) divided by the number of Diptera index was not significantly affected by site and year. However, the number of total EPT, Ephemeroptera, Plecoptera, Trichoptera, Diptera (excluding Chironomidae) and Chironomidae were all significantly affected by site and year. The most notable difference between pre- and post-harvest occurred within the 20 m buffer, where a large increase in *Oxythira* sp., an algal consumer, was observed. The number of species observed for each of the sites was slightly greater post-harvest, however the differences were not significant.

Brook trout and Atlantic salmon populations significantly increased for all three experimental sites except for brook trout within the 20 m buffer site. The biomass of brook trout significantly increased within the 20 m buffer with selective harvesting, while all other differences for brook trout and Atlantic salmon biomass were not significant. Young-of-the-year salmonid populations increased for all three experimental sites, with the exception of brook trout within the 20 m buffer. Young-of-the-year salmonid biomass was not significantly different for any of the experimental sites. For year 1+ and older Atlantic salmon populations, the 20 m buffer displayed the only significant increase between pre- and post-harvest. The brook trout population estimates were not significantly different for any of the experimental sites as compared to the no harvest site.

The biomass of both salmonid species were not significantly different for any of the experimental sites.

Overall, the reach with the 30-50 m buffer with selective harvesting appeared to be the least impacted, specifically in terms of sedimentation and invertebrate community changes. These results suggest that managed buffers in this area of Newfoundland should be 30-50 m.

The results of this study should be cautiously interpreted, owing to the short post-harvest assessment, and longer term monitoring is recommended to assess the implications of harvesting with managed buffers.

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Chapter 1

General Introduction

Canada's boreal forest forms part of the world's largest forest ecosystem. This forest type makes up more than three-quarters of the forested landscape in Canada, representing one-quarter of the world's boreal forests (Canadian Senate Subcommittee on the Boreal Forest 1999). The boreal forest contains some of the largest river systems and also an estimated 1.5 million lakes (Canadian Senate Subcommittee on the Boreal Forest 1999). The boreal forest has been called the heart and lungs of North America.

However, this ecosystem is in danger for several reasons. The logging industry harvests millions of trees each year (Canadian Senate Subcommittee on the Boreal Forest 1999). Oil and gas development removes forest cover, and mining operations add waste to the rivers and lakes within the forests (Canadian Senate Subcommittee on the Boreal Forest 1999). Hydroelectric projects flood large areas and re-direct and/or alter the flow of many streams and rivers (Canadian Senate Subcommittee on the Boreal Forest 1999). Further, global warming has the potential to drastically alter the boreal ecosystem.

Of these pressing concerns logging is at the forefront due to its accelerating pace and because of its rapid advancements into fragile and slow growing northern regions. According to the Canadian Senate Subcommittee on the Boreal Forest

(1999), about 90 percent of logging is clearcutting, the legislated method of forest harvesting in Newfoundland and Labrador (Government of Newfoundland and Labrador 2002). Associated with logging induced changes to the boreal forest are changes to the riparian habitats that strongly influence the quality of stream habitat for aquatic organisms.

Much of the existing research on forestry-ecosystem interactions has been conducted either in the Pacific Northwest or the northeastern United States (Clarke et al. 1998; McCarthy et al. 1998; Scruton et al. 1998). These areas have environmental, biological, and ecological conditions differing greatly from those in Atlantic Canada (Clarke et al. 1998; McCarthy et al. 1998; Scruton et al. 1998). Throughout the 1990s there have been a few studies within Eastern Canada including Copper Lake and the Small Stream Buffer Study, Newfoundland, and Catamaran Brook and Hayward Brook, New Brunswick. This study is an extension of these other studies, with specific focus on the effects of selective harvesting within buffers in northeast Newfoundland. Within Newfoundland, forest harvesting has been occurring for nearly one hundred years and the effects of these activities on freshwater fish and invertebrate species still remains poorly understood (Scruton et al. 1998). Currently, it is believed by many researchers that provisions of buffer zones along waterbodies is one of the most important steps in reducing the harmful effects of forestry practices (Barton 1985; Murphy et al. 1986). As a result of requests for protection for fish and wildlife habitat, regulations have been implemented on timber harvesting activities (Scruton et al. 1997). The provision of buffer strips is one of the regulations implemented, and numerous studies have demonstrated the importance of buffer strips in reducing the negative impacts on fish habitat (Scrivener and Brownlee 1989). Buffer zones provide protection by keeping machinery and their associated sedimentation some distance from the stream (Scruton et al. 1997). Managed buffer zones differ from the conventional buffer zones in that they allow harvesting within the buffer. To maximize protection of waterbodies, the width of the buffer should be considered without greatly affecting the economics of harvesting (Scruton et al. 1997). The effectiveness of buffer zones is not dependent solely by the width, caution during logging activities greatly affect buffer zone efficiency (Scruton et al. 1997). For this study, the effectiveness of selective harvesting within a buffer zone was examined to determine whether significant changes to the adjacent stream occurred. A wider buffer strip was studied to determine whether a wider buffer further reduced the impact of harvesting near waterbodies, while permitting selective harvesting within the buffer to offset the economic losses to forestry due to the wider buffer width.

The ecological processes of a stream are intimately related to those of the surrounding terrestrial ecosystem (Garman and Moring 1991). Miller (1987) stated the quality of stream habitat for aquatic organisms is largely influenced by the riparian vegetation. This project was conceived to evaluate advancements in forest harvesting in Newfoundland, specifically the concept of 'managed' buffers. The practical nature of this project has allowed this new approach to be field-tested and comparisons made of the full cost and benefits of conventional and innovative harvesting practices in support of adaptive ecosystem management. It is vital to assess possible implications of implementing a new harvesting practice through the acquisition of new information and an improved understanding of the effect of these activities on various components of an ecosystem.

A study on various components of an aquatic ecosystem, including invertebrates and fish as well as abiotic components, makes it possible to draw conclusions from a more holistic viewpoint. In this thesis salmonid biomass and populations were studied to determine whether there was a significant response to forest harvesting. Salmonids are the most valued group of freshwater fish in Newfoundland and Labrador (Scruton et al. 1997) and are greatly dependent upon the conditions of the surrounding forests (Meehan 1991). The diet of salmonids is mainly composed of macroinvertebrates, specifically Ephemeroptera, Plecoptera, and Trichoptera (Wiederholm 1984; Gordon et al. 1992; Waters 1995), therefore invertebrate community changes as a result of forest harvesting could have implications for salmonids (Benke 1984). Macroinvertebrates are also well known indicators of water quality and macroinvertebrates respond quickly to changes within their environment due to their short life span (Reice and Wohlenberg 1993). Also, invertebrate community structure reflects conditions within a small spatial area and may reveal changes that salmonids would not display. Therefore changes to invertebrate community structure may indicate changes that would not be apparent by studying salmonids alone.

Furthermore, abiotic conditions such as sedimentation can impact salmonids directly through impaired vision from turbidity or abrasions, reduction in spawning and egg incubation success, and reduction in habitat quality (Cordone and Kelly 1961; Saunders and Smith 1965; Alexander and Hansen 1983; Wesche 1985; Furniss et al. 1991; Nelson et al. 1991; Waters 1995). As well, water temperature increases could result in an increase in disease or increases which could lead to physiological impairment or death (Lynch et al. 1984; Beschta et al. 1987; Gordon et al. 1992). Indirectly, increases in sedimentation and water temperature could negatively affect macroinvertebrate populations (Wiederholm 1984) thus decreasing food availability for salmonids (Gordon et al. 1992; Waters 1995).

The objective of this study was to assess whether managed buffers are a superior method of riparian zone management, protecting the ecological integrity of an area, while maintaining the total wood production for the forestry industry within these areas.

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Chapter 2

Effect of managed buffer zones on sedimentation

2.1 Introduction

Natural stream erosion is influenced by various factors such as water flow, channel morphology, substrate type, soil characteristics, and vegetation. It is known that forest harvesting can result in an increase in sediment input, resulting in an upset of the natural balance. This occurs usually as a result of bank erosion or influxes of sediment from upland sources (Gordon et al. 1992). Road building and clearcuts may cause or accelerate soil erosion resulting in increased sedimentation. Clearcuts increase runoff from other parts of the watershed, resulting in increased storm flows, road failures, channel scouring, bank failure, and debris accumulation (Toews and Brownlee 1981).

Mechanical operations by the forest industry can have a large impact on the forest floor through road building, landing construction, and forwarder tracks, exposing mineral soils to erosion resulting in increased sediment transport into streams (Toews and Brownlee 1981). The potential for surface erosion is directly related to the amount of bare compacted soil exposed to rainfall and runoff (Chamberlin et al. 1991). Chamberlin et al. (1991) also state that as a general rule, surface erosion results from mineral soil exposure.

Sedimentation affects several biotic communities at various trophic levels. It can lower the productivity of primary producers such as photosynthesizing plants, primary and secondary consumers such as benthic invertebrates, and secondary consumers, top carnivores and piscivores such as fish (Hartman et al. 1983; Waters 1995). Sediment accumulation can be sufficient enough to seriously reduce the available insect habitat. Recovery is usually not expected unless sediment on the surface and in the interstices can be removed by natural means (Slaney et al. 1977). Slaney et al. (1977) also stated that the lowest biomass and density of benthic invertebrates was found in the sections of streams which had the highest sediment concentration. Similarly, Lenat et al. (1981) have shown that as sediment is added to a stream, the area of available habitat decreased which corresponded to a decrease in the density of benthic macroinvertebrates.

The potential effects of sediment on benthic invertebrates include interference with respiration and the overwhelming of filtering insects such as caddisfly larvae that collect drifting food particles by using fine-meshed catchnets (Waters 1995). Another concern with the implication of increased sediment on invertebrates is the effect of changing invertebrate communities on fish populations (Waters 1995). One of the most obvious concerns is the change in invertebrate communities from EPT (Ephemeroptera, Plecoptera, and Trichoptera), which are common prey taxa for salmonids, to burrowing forms such as chironomids, midgefly larvae, and oligochaetes (Wiederholm 1984; Gordon et al. 1992; Waters 1995).

This chapter describes a study conducted to document the change in sedimentation, attributable to forest harvesting activities with different buffer widths. There is a need to research the effectiveness of forested buffer strips as "filters", and the influence of other factors within the watershed, since literature is relatively limited (Belt et al. 1992). One of the major benefits associated with implementation of managed buffer zones, in theory, is that there is a reduced chance of blowdown which can contribute to erosion, as a result of exposed soil from uprooting.

2.2 Methods

2.2.1 Study Area

The study area was located in northeast insular Newfoundland, Canada in the Indian Bay watershed, specifically within Hungry Brook (UTM coordinates: 21 5435423 N 700450 E for the upper reach to UTM coordinates: 21 5434294 N 698157 E for the lowest reach), a small second order tributary of Indian Bay Pond (Figure 2.1; Figure 2.2). This watershed is approximately 750 km^2 in total area and it is located in the central ecoregion (Damman 1983). Black spruce (*Picea mariana*) forest stands and trembling aspen (*Populus tremuloides*) dominate the ecoregion. Specifically, the study site is a productive black spruce forest with an average height of 12 m and approximately 82 years old (Dr. Gary Warren, Canadian Forest Service, pers. comm.). The topography varies from rolling to undulating, and the soils are classified as sandy loam (Damman 1983). The site has a slightly sloping terrain (gradient 1 % on average) with a north slope aspect.

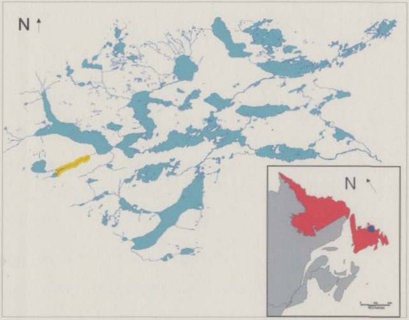

Figure 2.1. Map of Indian Bay watershed with a map of Newfoundland (inset) showing the location of the study.

The forest floor consists of predominantly feathermoss (*Pleurozium schreberi*) and is considered a moist site, with gleysol soils (Meades and Moore 1989; Bruce Roberts, Canadian Forest Service, pers comm.).

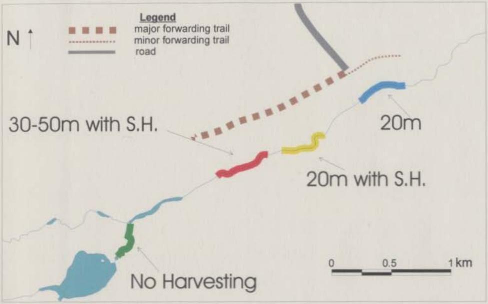

Figure 2.2 Map of Hungry Brook indicating the location of the experimental sites, the no harvest control site, the forwarding trails, and the road.

2.2.2 Study Sites

Data were collected in 2000 to represent baseline conditions pre-harvest. Three experimental buffers were established: (1) 20 m buffer, the current provincially legislated requirement for a brook of this size and location; (2) 20 m buffer with 30 % of the basal area harvested; and (3) 30-50 m varying width buffer with 30 % of the basal area harvested (Figure 2.3). To remove 30 % of the basal area, the diameter at breast height of trees were measured and marked, and totaled until 30 % of the buffer area for each site had been selected. A no harvest area was also established to serve as a 'control' against natural variation within this brook.

All of the sites were 500 m in length and harvesting was conducted with a mechanical harvester along one side of the brook during November 2000. Only one side of the brook was harvested thus removing the effect of stream crossings on the study's results. Data collected in 2001 after the establishment of these buffers represented the experimental data.

Hungry Brook has a mean wetted width of 4.8 m and has an average channel depth of 20.9 cm (summer data collection). The substrate consists mainly of cobble, rubble and gravel. The banks of the stream were considered to be stable with a small percentage of undercut.

Figure 2.3. Aerial photo of the varying 30-50 m buffer with selective harvesting at Hungry Brook.

2.2.3 Field and laboratory methods

Whitlock-Vibert boxes (Figure 2.4) were used to measure fine sediment accumulation, as described by Wesche et al. (1989). These boxes, typically

Figure 2.4. Whitlock-Vibert boxes used for sedimentation.

used for egg incubation, are 14 x 6.4 x 8.9 cm with 3.5 x 13 mm openings. The boxes were filled with cleaned gravel, approximately 25 mm in diameter. Duct tape was placed across the bottom of the box to prevent loss of fine particulates. Two Whitlock-Vibert sediment boxes were deployed at 100 m intervals, with placement beginning at 0 m, within each 500 m experimental and the no harvest site. Two sediment boxes were fastened to a wire rack (approximately 40 x 10 cm), with approximately 30 cm between the boxes and then anchored to the substrate to prevent loss or misplacement. Boxes were placed in riffle habitat and situated such that flow was not impeded by other instream obstructions, such as instream debris or large boulders. The boxes were changed again after the spring run-off in June 2000, and again late October 2000 prior to the harvesting commencing. During collection, each box was carefully detached from the apparatus and placed in plastic bags until analysis. For the post-harvest year, 2001, the sediment boxes were changed after the spring-run off in June and retrieved again in November.

The sediment boxes were opened and the contents were wet sieved through the following sieve sizes: 2.5, 1.4, 0.85, 0.50, and 0.09 mm and dried at 70 °C for 30 h, and then weighed. The sediment sizes were divided into two size classes: greater than 1.4 mm (represented by sediment collected from the 2.5 and 1.4 mm sieves) and less than 1.4 mm (represented by sediment collected from the 0.85, 0.50, and 0.09 mm sieves).

2.2.4 Statistical Analyses

The data were analyzed using the G statistic from a chi-square distribution with one degree of freedom (α=0.05). This statistic measures goodness of fit of chi-square to the data (Devore 1995). The quantity of sediment accumulated for each size class and the classes totaled within each site collected during 2000 (pre-harvest) and 2001 (post-harvest) were compared with the no harvest site to determine whether there was a significant difference between pre- and post-harvest.

2.3 Results

The only experimental site that differed significantly between pre- and post-harvest conditions, compared to the no harvest site was the 20 m buffer with selective harvesting (Table 2.1, Figure 2.5). It had significantly greater total accumulated sediment, for both sediment size classes.

The increase in sedimentation for the 20 m buffer (provincial legislation) was 39.95 g (from 286.96 (2000) to 326.92 g (2001)) (Figure 2.5). For the 30-50 m buffer with selective harvesting there was a 3.64 g decrease (from 35.92 (2000) to 32.28 g (2001)). However, the 20 m buffer with selective harvesting increased drastically by 289.85 g, (from 91.61(2000) to 381.46 g (2001)). The amount of sediment accumulated for the no harvest site was from 43.45 g (2000) to 63.09 g (2001), an increase of 19.64 g, which was not significant. Both 20 m buffers had greater sediment accumulated than the no harvest site, with two times greater sediment in the 20 m buffer, and 14.5 times greater in the 20 m buffer with selective harvesting.

Table 2.1. Comparison of total, >1.4 mm, and <1.4 mm sediment accumulation in the three experimental sites and the no harvest site and the p-values from the G statistic.

Site	Sediment size class	Sediment accumulated (g) 2000	Sediment accumulated (g) 2001	P-value
20 m buffer (provincial legislation)	Total	287.0	326.9	0.069
	> 1.4 mm	175.4	192.9	0.294
	< 1.4 mm	111.5	134.0	0.655
20 m buffer with selective harvesting	Total	91.6	381.5	<0.0001
	> 1.4 mm	57.8	214.8	0.017
	< 1.4 mm	33.8	166.7	0.0001
30-50 m buffer with selective harvesting	Total	35.9	32.3	0.125
	> 1.4 mm	20.5	14.8	0.354
	< 1.4 mm	15.5	17.5	0.655
No harvest	Total	43.5	63.1	NA
	> 1.4 mm	16.2	25.5	NA
	< 1.4 mm	27.3	37.6	NA

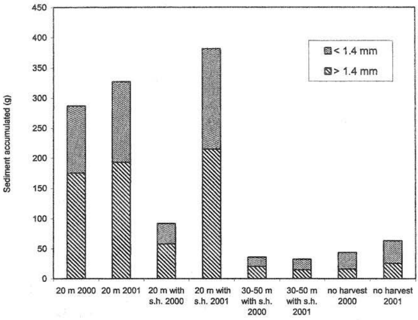

Figure 2.5. Comparison of total sediment accumulation (g) between the three experimental sites and the no harvest site separated into two size classes: (i) sediment greater than 1.4 mm and (ii) fine sediment (less than 1.4 mm).

<u>Discussion</u>

The effectiveness of buffer strips of various widths on filtering sediment overland has been investigated by various researchers. Belt et al. (1992) found that filter strips on the order of 200-300 feet (60-90 m) are effective in controlling sediment that is not channelized. Their findings appear to concur with the results of this study, where the only buffer that did not appear to be affected by overland flow of sediment entering the stream was the 30-50 m buffer. Much of the literature refers to logging roads as the main source of increased sedimentation. Observations during the study suggest the sedimentation problem originated from heavily used forwarding trails that had mineral soils exposed. This disturbance of the soil profile and exposure of mineral soils is similar to the 'grubbing' process common in road building; hence the similarity in the findings of Belt et al. (1992). There were also a few locations along this brook where point sources were obvious. The sediment source originated approximately 200 m from the stream, and sediment laden runoff was not completely filtered over a distance of 180 m that was clearcut but with slash remaining. Belt et al. (1992) also state that riparian buffer strip widths should be greater where slopes are steep. In this study, the gradient was very slight, mostly less than 1 %, and did not appear to play a role in the large sedimentation observed.

Salo and Cundy (1987) state that during timber harvesting, as long as trees are not felled directly into stream channels, the impact is usually small for erosion processes. They also state the impact of yarding operations on ground disturbance can be extensive when tractors are used, but when cable systems are implemented, that either fully or partially suspend the logs, the impact is minimal. In this study, logs and pulpwood were transported from the harvest site to a landing using a forwarder with a self contained loading rear rack for transporting wood. This forwarder was a six wheel drive unit with the two rear wheels on each side contained within a metal track, supposedly to reduce soil disturbance. Conversely, the effects of yarding using a forwarder were extensive with large ruts left from the established forwarding trails, that became a major source of sedimentation overland, and eventually into the stream. It is quite obvious there was excessive soil compaction as a result of these ruts in excess of 0.50 m in depth at times (Figure 2.6). This amount of compacted ground can

Figure 2.6. Photo of forwarding ruts in excess of 0.50 m as shown by the 0.60 m measuring stick in the right rut.

reduce access and capacity of subsurface channels, and increase soil erosion, and this can have long term consequences on the hydrologic characteristics of the soils (Salo and Cundy 1987).

The Copper Lake study conducted in western Newfoundland found that increased sedimentation was largely the result of road construction, specifically the installation of culverts (Clarke et al. 1998). A limited clear cut also resulted in increased sedimentation in addition to the accumulation attributed to culvert installation (Clarke et al. 1998). Within this study, road crossings were not present, however, increases in sedimentation did result.

Brownlee et al. (1988) found increased sediment loading four to twelve times greater than that observed in unlogged watershed. Erosion from skid trails was one of the many identified sources contributing to the increased levels of sediment. The results of this study found similar sources, except the source was forwarder trails, identifying the process of transporting timber as one of the major factors negatively influencing aquatic habitats.

An argument used against the use of buffer zones is that timber within the strips are subject to blowdown (Lantz 1971). One of the purposes of managed buffer zones is to allow the lower section of the wind profile to perforate through the buffer where trees had been selectively removed. Blowdown can result in the exposure of tree roots, and unstable soils, thus increasing sediment transport into streams. However, within the time frame of this study, differences between percent of blowdown was not determined. Lantz (1971) suggests blowdown is a local problem. One of the major problems with the longevity of a buffer is the amount of rot within the stand. Root and butt rot decay fungi kill the lateral root systems and decay the structural heartwood in the major root and butt section of living trees resulting in growth loss, tree mortality and windthrow. As wind moves across an opening such as a clear cut, and then comes into contact with a forest, the upper portion of the wind profile continues unimpeded above the forest crown. However, the lower portion of the wind profile meet with the branches and stems resulting in a great deal of stress for the trees that have recently been exposed to such winds for the first time and do not have the root systems established to withstand the wind. A pre- and post-harvest assessment was conducted to determine the percentage of trees within the experimental sites with rot present.

One of the limitations of a study design such as this, is confounding sedimentation. However, from my results, if sedimentation was confounding between sites, then the 20 m buffer (the site downstream from both the 20 m buffer with selective harvesting and the 30-50 m buffer with selective harvesting) should have displayed a large increase in sedimentation. However, only a small increase was observed for the 20 m buffer, therefore confounding sedimentation was minimal. The no harvest site displayed a very slight increase in sedimentation between pre- and post-harvest, indicating very little natural variation. Natural variation within a short term ecological study is a concern. However, this variation was not statistically different.

Future investigations into possible mitigation to prevent forwarding from causing such extensive damage to the soil, specifically in regions where soil characteristics are similar to those observed in this study site, needs to be investigated.

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Chapter 3

Effect of managed buffer zones on stream temperature regime

3.1 Introduction

The temperature of water in forest streams is an important determinant for various forms of life, including riparian vegetation, invertebrates, and fish (Gordon et al. 1992). An increase of stream temperature can increase the mortality of fish and other aquatic organisms, can cause temporary or permanent abandonment of habitat, as well as cause changes to community structure and increase interspecific competition (Gregory et al. 1987). It was not until the 1960's that the impact of forest harvesting on the temperature of forest streams was considered (Beschta et al. 1987). Previously, most research on the effects of forest harvesting focused mainly on changes to runoff and sedimentation levels (Beschta et al. 1987). The removal of the vegetated canopy often accompanies an increase in stream temperature (Gregory et al. 1987). Several other studies on the effects of timber harvesting on stream temperature have shown significant maximum stream temperature changes during the summer months (Brown and Krygier 1970; Lynch et al. 1977; Scruton et al. 1998).

Stream temperature is an important ecological factor because basic metabolic processes of living organisms are temperature dependent and temperature can also affect the holding capacity for oxygen (Miller 1987). Water temperature is an important determinant defining the geographical distribution of brook trout

(*Salvelinus fontinalis*) (Eschner and Larmoyeux 1963; MacCrimmon and Campbell 1969). One of the factors contributing to low salmonid populations is extremes in water temperature and streamflow (Ensign et al. 1990).

Smaller streams are highly susceptible to the effects of timber harvesting as a result of changes to several parameters. Two of the most important parameters that change in association with timber harvesting are light intensity and stream temperature (Lynch et al. 1984). Removal of riparian vegetation results in an increase of light intensity which could then result in a deterioration of trout habitat by decreasing food availability through decreased dissolved oxygen and/or reduced drift from light intensity on macroinvertebrates (Lynch et al. 1984). The management of the riparian zones of headwater streams is very important in maintaining community structure in a watershed. The ecological processes of streams are intimately related to the surrounding terrestrial system (Gregory et al. 1987; Garman and Moring 1991) with this relationship being even more important for smaller streams and tributaries, thus increasing vulnerability to changes to riparian habitats. The objective of the work in this chapter was to examine how forest harvesting affects temperature regimes and the implications for brook trout and Atlantic salmon in Hungry Brook.

3.2 Methods

3.2.1 Study Area

Refer to section 2.2.1

3.2.2 *Study Sites*

Refer to section 2.2.2

3.2.3 *Field methods*

Water temperature was recorded from January to October 2000 to determine the temperature regime pre-harvest. A Vemco Minilog-TR thermograph was deployed at the upper and lower reach of the experimental sites and the control sites. The thermographs have a temperature range between -5 and 35 °C, with a 0.2 °C resolution. The thermographs were programmed to record the water temperature every hour over that period. On 23 January 2000, ten thermographs were deployed as described and then were subsequently retrieved and replaced on 23 October, 2000, just prior to the commencement of harvesting. The thermographs were retrieved and replaced again in June and November 2001.

Hourly recordings were used to calculate monthly means, minima, and maxima and the daily temperature regimes for the months of June through September, inclusive. During these months water temperatures often reach annual maxima. Placement of thermographs at the upper and lower reach of each experimental and no harvest site, allowed temperature dynamics and the ability of the riparian vegetation to thermo-regulate stream temperatures to be examined.

Specific attention was given to mean and maximum summer monthly temperatures in the upper and lower stations for each of the three experimental buffers and the no harvest site, and daily summer temperature regime (minimum, maximum, and mean) for the upper and lower stations for the three experimental buffers and the no harvest site.

For each thermograph, the total number of hours during the summer months (June 1 to September 30, 2928 hours for pre- and post-harvest) was separated into five temperature classes (Table 3.1). The temperature classes were defined based on the thermal requirements of brook trout (Power 1980; Raleigh 1982; Lynch et al. 1984; Jirka and Homa 1990; Scruton et al. 1998) and Atlantic salmon (Gibson 1978; Dwyer and Piper 1987; Chiasson et al. 1990; Elliott 1991; Siemien and Carline 1991;) and were as follows:

Brook trout

a) <11°C (Lower; below optimum but not stressful);
b) 11 to 16°C (Optimum; good growth);
c) 16.1 to 20.9°C (Upper; above the optimum range but no induced stress);
d) 21 to 23.9°C (Stress; temperature range posing stress and increased susceptibility to disease); and
e) >24°C (Lethal; lethal if exposed for a period of time).

Atlantic salmon

a) <15°C (Lower; below optimum but not stressful);

b) 15 to 20°C (Optimum; good growth);

c) 20.1 to 24.9°C (Upper; feeding ceases but no thermal stress behaviour);

d) 25 to 30°C (Stress; temperature range exhibiting thermal stress behaviour); and

e) >30°C (Lethal; lethal if exposed for a period of time).

3.2.4 Statistical Analyses

To determine whether there was a significant difference between pre- and post-harvest for five temperature classes the data were analyzed using the G statistic from a chi-square distribution with one degree of freedom ($\alpha=0.05$). This statistic measures goodness of fit of the data (Devore 1995). The values obtained for the number of hours in each temperature class for the upper and lower thermograph were averaged for each site to obtain one value for each site per study year.

3.3 Results

Daily summer temperature regimes (minimum, maximum, mean) for the upper and lower stations of each experimental and no harvest site were graphed to compare the daily regimes between sites and within sites (i.e. upper versus lower station) (Figure 3.1 and 3.2). The number of hours in the five temperature classes for brook trout (lower, optimum, upper, stress, and lethal) differed between the experimental and no harvest sites (Table 3.1). There was an increase in the number of hours in the lethal class for both 20 m buffers (Figure 3.1) during the post-harvest study period, however the 30-50 m buffer with selective harvesting did not show any increase. Statistically, there was no difference in the number of hours in the lethal class for the experimental sites compared to the no harvest site (Table 3.2).

Not obvious in Figures 3.1 and 3.2, is a significant difference in the optimum temperature class for the 20 m with selective harvesting and the 30-50 m with selective harvesting (Table 3.2). There was a significant decrease in the number of hours in the optimum temperature range for the 20 m with selective harvesting, whereas the 30-50 m with selective harvesting showed a significant increase. For the 20 m buffer, none of the five temperature classes significantly differed from the no harvest site between the pre- and post-harvest (Table 3.2, $p=0.0320$, $p=0.0404$, respectively).

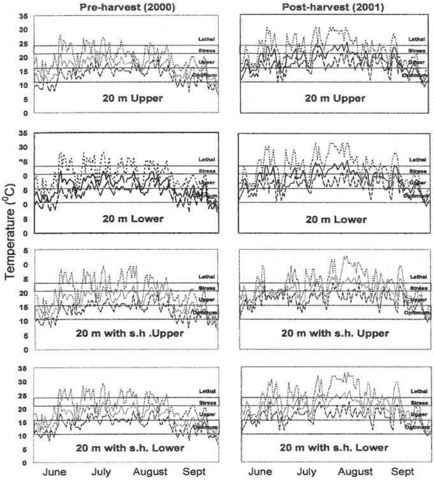

Figure 3.1. Daily summer temperature regime (minimum, maximum, mean) for the upper and lower stations of the 20 m buffer and the 20 m buffer with selective harvesting for pre- and post-harvest conditions for brook trout thermal requirements.

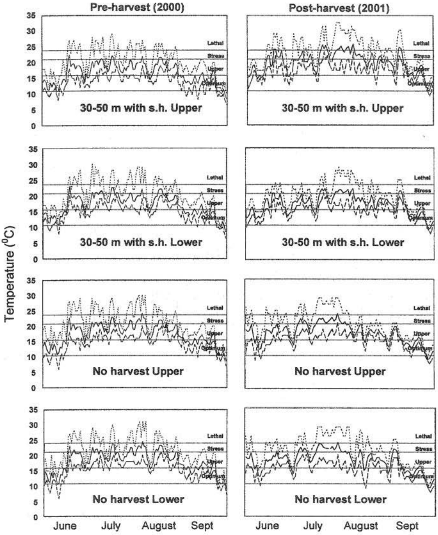

Figure 3.2. Daily summer temperature regime (minimum, maximum, mean) for the upper and lower stations of the 30-50 m with selective harvesting and no harvest site for pre- and post-harvest conditions for brook trout thermal requirements.

Table 3.1. Number of hours and percentages (in parentheses) of summer (June 1 to September 30) water temperatures for each buffer zone for pre- and post-harvest conditions, in each of five temperature categories according to brook trout temperature requirements.

Site	Lower (<11°C)	Optimum (11-16°C)	Upper (16.1-20.9°C)	Stress (21-24°C)	Lethal (>24°C)
20 m L (2000)	238 (8)	1366 (47)	1081 (37)	133 (5)	110 (4)
20 m L (2001)	120 (4)	1449 (49)	1110 (38)	154 (5)	95 (3)
20 m U (2000)	215 (7)	1342 (46)	1076 (37)	155 (5)	140 (5)
20 m U (2001)	92 (3)	1396 (48)	1168 (40)	160 (5)	112 (4)
20 m with s.h. L (2000)	210 (7)	1319 (45)	1039 (35)	154 (5)	206 (7)
20 m with s.h. L (2001)	83 (3)	1325 (45)	1181 (40)	169 (6)	170 (6)
20 m with s.h. U (2000)	167 (6)	1259 (43)	1116 (38)	185 (6)	201 (7)
20 m with s.h. U (2001)	62 (2)	1225 (42)	1303 (45)	181 (6)	156 (5)
30-50 m with s.h. L (2000)	170 (6)	1260 (43)	1110 (38)	189 (6)	199 (7)
30-50 m with s.h. L (2001)	64 (2)	1241 (42)	1278 (44)	185 (6)	160 (5)
30-50 m with s.h. U (2000)	118 (4)	1175 (40)	1241 (42)	214 (7)	180 (6)
30-50 m with s.h. U (2001)	61 (2)	1100 (38)	1408 (48)	211 (7)	148 (5)
No harvest L (2000)	141 (5)	1079 (37)	1236 (42)	203 (7)	269 (9)
No harvest L (2001)	79 (3)	1108 (38)	1294 (44)	224 (8)	223 (8)
No harvest U (2000)	125 (4)	911 (31)	1296 (44)	279 (10)	317 (11)
No harvest U (2001)	73 (2)	1041 (36)	1282 (44)	260 (9)	272 (9)

Table 3.2. The p-values obtained from the G statistic for five temperature classes defined by brook trout thermal requirements for each of the experimental sites compared with the no harvest for pre- and post-harvest. (* indicates $p<0.05$)

Experimental Site	**Temperature Classification**	**P-value**
20 m (provincial legislation)	Lower	0.2815
	Optimum	0.6547
	Upper	0.4543
	Stress	0.5716
	Lethal	1.0
20 m buffer with selective harvesting	Lower	1.0
	Optimum	0.0320*
	Upper	0.1380
	Stress	1.0
	Lethal	1.0
30-50 m buffer with selective harvesting	Lower	0.1871
	Optimum	0.0404*
	Upper	0.0571
	Stress	1.0
	Lethal	0.7184

The number of hours in the five temperature classes for Atlantic salmon (lower, optimum, upper, stress, and lethal) were graphed with the daily summer temperature regimes (minimum, maximum, mean) for upper and lower stations of each experimental and no harvest site (Figure 3.3 and 3.4). The number of hours differed between the experimental and no harvest sites (Table 3.3). There was a significant increase in the number of hours in the upper and lethal temperature class for the 30-50 m buffer with selective harvesting (Table 3.4).

The temperature recordings were also used to compare the water temperature difference between the upper and lower reaches of the study sites, on a monthly basis. The mean monthly temperatures for three experimental sites and the no harvest site show a slight decrease in water temperature over each of the experimental sites for pre- and post-harvest observations (Figure 3.5). There was also an overall decrease in the water temperature from the upper site to the lowest site over the entire study area (see Figure 2.2; Figure 3.5).

The summer average maximum monthly temperatures for three experimental sites and the no harvest site for the upper and lower reaches show slightly different results than the mean monthly temperatures. The 20 m buffer with selective harvesting differed from the other sites by having the lower reach greater for August than the upper reach for the pre- and post-harvest study period (Figure 3.6).

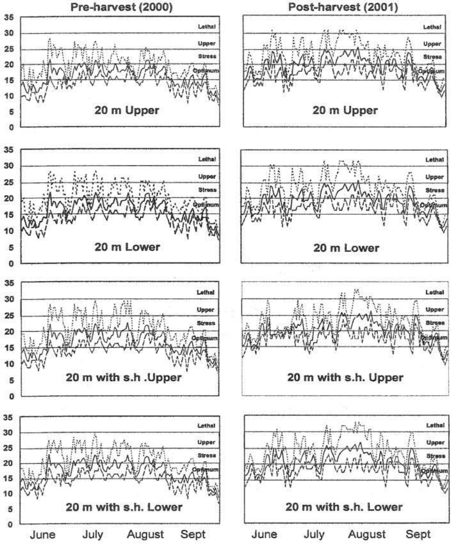

Figure 3.3. Daily summer temperature regime (minimum, maximum, mean) for the upper and lower stations of the 20 m buffer and the 20 m buffer with selective harvesting for pre- and post-harvest conditions according to Atlantic salmon thermal requirements.

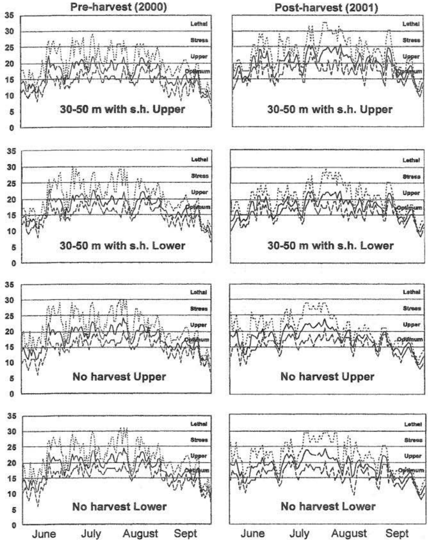

Figure 3.4. Daily summer temperature regime (minimum, maximum, mean) for the upper and lower stations of the 30-50 m buffer with selective harvesting and the no harvest site for pre- and post-harvest conditions according to Atlantic salmon thermal requirements.

Table 3.3. Number of hours and percentages (in parentheses) of summer (June 1 to September 30) water temperatures for each buffer zone for pre- and post-harvest conditions, in each of five temperature categories according to Atlantic salmon temperature requirements.

Site	Lower (<15°C)	Optimum (15-20°C)	Upper (21.1-24.9°C)	Stress (25-30°C)	Lethal (>30°C)
20 m L (2000)	997 (34)	1386 (47)	516 (19)	29 (2)	0 (0)
20 m L (2001)	782 (27)	1489 (52)	595 (21)	14 (<1)	0 (0)
20 m U (2000)	953 (33)	1366 (47)	564 (190)	45 (2)	0 (0)
20 m U (2001)	753 (26)	1501 (52)	599 (21)	36 (1)	0 (0)
20 m with s.h. L (2000)	960 (33)	1318 (45)	555 (19)	95 (3)	0 (0)
20 m with s.h. L (2001)	792 (28)	1358 (47)	634 (22)	105 (4)	0 (0)
20 m with s.h. U (2000)	879 (30)	1334 (46)	637 (22)	78 (3)	0 (0)
20 m with s.h. U (2001)	626 (22)	1498 (52)	722 (25)	41 (1)	4 (<1)
30-50 m with s.h. L (2000)	879 (30)	1333 (46)	642 (22)	74 (3)	0 (0)
30-50 m with s.h. L (2001)	632 (22)	1484 (52)	725 (25)	45 (2)	3 (<1)
30-50 m with s.h. U (2000)	755 (26)	1404 (48)	689 (24)	79 (3)	1 (<1)
30-50 m with s.h. U (2001)	843 (29)	1657 (58)	1000 (35)	105 (4)	4 (<1)
No harvest L (2000)	710 (24)	1370 (47)	717 (24)	124 (4)	7 (<1)
No harvest L (2001)	550 (19)	1444 (50)	792 (28)	103 (4)	1 (<1)
No harvest U (2000)	612 (21)	1293 (44)	857 (29)	152 (5)	14
No harvest U (2001)	560 (19)	1317 (45)	871 (30)	139 (5)	2 (<1)

Table 3.4. The p-values obtained from the G statistic for five temperature classes defined by Atlantic salmon thermal requirements for each of the experimental sites compared with the no harvest for pre- and post-harvest. (* indicates p<0.05)

Experimental Site	**Temperature Classification**	**P-value**
20 m (provincial legislation)	Lower	1.000
	Optimum	0.4386
	Upper	0.5271
	Stress	0.3711
	Lethal	1.000
20 m buffer with selective harvesting	Lower	0.2733
	Optimum	0.6547
	Upper	0.3173
	Stress	1.000
	Lethal	1.000
30-50 m buffer with selective harvesting	Lower	0.3173
	Optimum	0.0578
	Upper	0.0042*
	Stress	0.6547
	Lethal	0.0052*

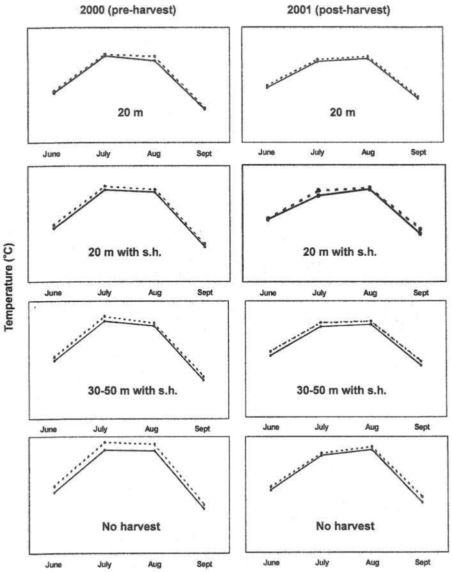

Figure 3.5. Summer mean monthly temperatures for three experimental sites and a no harvest site for pre- and post-harvest for the upper (dotted line) and lower (solid line) stations.

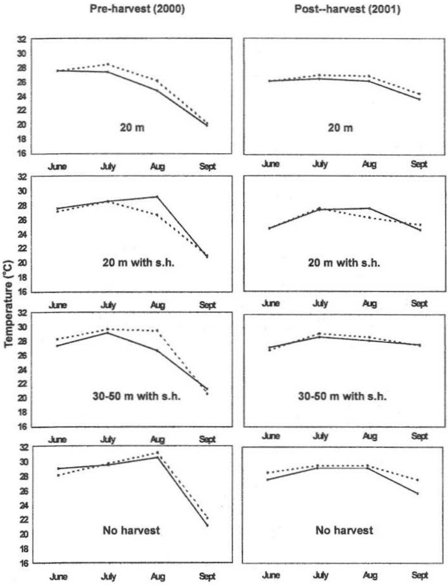

Figure 3.6. Summer maximum monthly temperatures for three experimental sites and a no harvest site for pre- and post-harvest for the upper (dotted line) and lower (solid line) stations.

Discussion

One of the effects of increased water temperature on salmonids is a decrease in growth/potential. The growth efficiency is highest at low temperature, but the activity of fish also becomes depressed during these lower temperatures, thus depressing feeding (Beschta et al. 1987). The optimum stream temperature for fish occurs when activity levels are high enough to ensure feeding, and good metabolic conversion efficiency (Beschta et al. 1987). Higher water temperatures also result in an increase of metabolic rate/oxygen consumption, and thus increased food consumption (Gordon et al. 1992).

The potential effects of harvesting on stream temperature indicated in the three experimental buffers implemented in this study showed no significant change to the overall thermal regime for brook trout. The number of hours and percentage of summer hours in various water temperature classes changed significantly only in the optimum category with a decrease for the 20 m buffer with selective harvesting and the 30-50 m buffer with selective harvesting. Therefore, there does not appear to be any reason to suspect there would be a large negative impact on brook trout since there was no significant change in the number of hours post-harvest in the stress and lethal temperature classes. A slight decrease in growth potential could be expected due to the decrease in the number of hours of optimum class temperatures in these two sites.

The effect of the three experimental buffer sites on the thermal regime of Atlantic salmon was significantly different within the 30-50 m buffer with selective harvesting. The upper and lethal categories significantly increased within the 30-50 m site. However, the lethal category only observed an increase of three hours, while the no harvest site observed a decrease of nine hours, resulting in the significant increase in the lethal category. A slight decrease in growth potential could be expected as a result of the increased number of hours in the upper and lethal categories for Atlantic salmon.

The dynamics of the temperature regime in this headwater stream were evident when the mean monthly water temperatures were calculated. Usually as water flows downstream, it equilibrates with the air temperature. The air immediately in contact with the water surface is determined by the stream's shading among other environmental factors (Beschta et al. 1987). From the results for the mean monthly stream temperatures there was a cooling trend over each of experimental sites for pre- and post-harvest, i.e. the upper thermograph for each of the sites was slightly greater than the lower thermograph within the same site. Furthermore, there was also a cooling trend over the entire section of stream in the study area. This small stream has streamside protection along its entire length, which maintains the temperature. The harvesting in the buffer also did not appear to have been intense enough to alter the shading provided by the streamside vegetation or affect the groundwater influence, therefore maintaining temperature regimes. The most intense treatment occurred within the 20 m buffer with selective harvesting, however the streamside vegetation provided sufficient shade to maintain temperature regimes similar to pre-harvest.

The maximum monthly temperatures were very similar for the pre- and post-harvest study periods. The ability of a buffer strip (30 m) to effectively maintain maximum temperature has been shown by Lynch et al. (1984). For my study, a 20 m buffer, and a 20 m with selective harvesting also effectively maintained maximum temperatures similar to those observed post-harvest.

Other studies on water temperature changes due to logging have shown increased summer temperatures (Lynch et al. 1977; Holtby and Newcombe 1982; Scruton et al. 1998). My study did not find an overall increases in summer temperature. However, decreases and increases were observed for specific temperature classes for brook trout and Atlantic salmon. The provision of buffer strips along streams is the one of the most important steps in reducing the effects of forest harvesting (Barton et al. 1985; Murphy et al. 1986). Within my study, the provision of buffer strips, at least 20 m in width, did not produce any adverse effects on stream temperature suggesting that buffer strips provide adequate protection for this component of an aquatic ecosystem. However, it is difficult to base the adequacy of a buffer on two years of data (pre- and post-harvest). The effect of these buffers may not become apparent until after several years of temperature data collection.

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Chapter 4

Effect of managed buffer zones on macroinvertebrates

4.1 Introduction

One of the primary concerns associated with forest harvesting is the adverse effects these activities may have on aquatic resources. The current common logging practice in Newfoundland and Labrador is clearcutting (Government of Newfoundland and Labrador 2002). This type of logging can affect the physical and biological conditions in streams, including increases in sediment, nutrients, and debris, and alterations to the hydrological and temperature regime. Changes to any or all of these together could result in a change in aquatic production (Murphy and Hall 1981). A change in aquatic primary production could result in a change to higher trophic levels such as fish, if prey populations are altered (Hartman et al. 1983; Waters 1995). Therefore, the structure of the aquatic macroinvertebrate community can reflect the state of the entire ecosystem (Benke 1984).

Benthic macroinvertebrates offer many advantages in biomonitoring directly related to the biological characteristics of this group of organisms (Rosenberg and Resh 1993). Biological characteristics of macroinvertebrates important to this project are: 1) macroinvertebrates can be affected by various environmental perturbations including sedimentation and react to them quickly;

2) macroinvertebrates are ubiquitous, abundant and relatively easy to collect; 3) macroinvertebrates are basically sedentary which permits effective spatial analysis of local conditions; and 4) the macroinvertebrate community is very heterogeneous, consisting of several taxa; therefore there exists a high probability some of the taxa will respond to environmental perturbation (Rosenberg and Resh 1993).

Sedimentation effects on aquatic macroinvertebrates were examined to determine whether sedimentation was severe enough to affect the macroinvertebrate community. High levels of sediment in streams have been shown to reduce the diversity and/or density of benthic macroinvertebrates (Lenat et al. 1981). Sedimentation can be sufficient enough to seriously reduce the available insect habitat and recovery is usually not expected unless sediment on the surface and in the interstices is removed (Slaney et al. 1977). The potential effects of sediment on benthic invertebrates include interference with respiration and the overwhelming of filtering insects such as some caddisfly larvae (Waters 1995).

Most invertebrate-forestry interaction studies have been conducted on the Pacific coast of North America or the central United States (Erman et al. 1977; Newbold et al. 1980; Murphy and Hall 1981; Blosser 1984; Carlson et al. 1990; Hartman and Scrivener 1990; Kohlhepp and Hellenthal 1992). These regions differ in many respects from Newfoundland's environmental characteristics, including biogeography, climate, soil conditions, fauna, and forest type. The exceptions are the Copper Lake buffer zone study conducted in western Newfoundland and the Newfoundland Small Steam Buffer Study, both of which included a forestry-invertebrate component, however there is a need to further this research to determine how other watersheds and their associated aquatic ecosystems in different eco-regions respond to managed buffer zones, and increased harvesting intensity. This knowledge could then be incorporated into sustainable forestry management for regions with similar environmental characteristics.

Studies as to whether buffer strip widths are effective as filters are also very limited (Belt et al. 1992). Furthermore, they also state that the literature is limited on the advantages of varied-width buffers compared with fixed-width buffers, and it is acknowledged that variable-width buffers allow the riparian zone to simulate a more natural edge. The objective of the study in this chapter was to determine if the ecological integrity of areas adjacent to forest harvesting activities was maintained by three different buffer zones treatments, two of which had selective harvesting within the buffer. A variable-width buffer was also created for one of the selective harvesting treatments. Erman et al. (1977) found that streams that had selectively cut riparian areas had less severe impacts on invertebrates than those where clearcutting without bufferstrips occurred. Subsequently, field experiments were designed specifically to test for effects of clearcut logging under various riparian zone treatments on stream benthos.

4.2 Methods

4.2.1 Study Area

Refer to section 2.2.1

4.2.2 Study Sites

Refer to section 2.2.2

4.2.3 *Sedimentation Analysis*

Refer to section 2.2.3

4.2.4 Invertebrate Sampling

Twenty four artificial substrates (rock bags) consisting of approximately 7.2 kg of 3.5 to 5.0 cm washed cobble encased in plastic Vexar mesh (1.5 cm, stretch measure) (Rosenberg and Resh 1982; Merritt and Cummins 1996) were placed in the three experimental buffers and the no harvest site within Hungry Brook within riffle habitat and were exposed to the stream flow, i.e. were not placed near large instream debris or boulders. The substrates (six bags in each of the four sites) were deployed May 11 for the pre-harvest and post-harvest and retrieved three weeks after deployment.

When lifting the rock bags from the stream, special care was taken to capture invertebrates that may swim or drift away by making a sweep of the immediate area with an aquatic sampling net. Each rock bag was shaken vigorously in a five gallon bucket of water for 60 seconds to remove organisms and then visually inspected to insure complete removal of all colonized invertebrates visible to the naked eye. The water was then filtered through a 500 μm Nitex screen and all organisms retained were preserved in 95% ethanol. Twelve of the twenty four rock bags (three from each of the four sites) were randomly selected and analyzed due to the time and effort required for this component of the study.

4.2.5 Species Identification

Each specimen belonging to the Orders Trichoptera, Ephemeroptera, and Plecoptera were classified to the species level, with a few exceptions to the genus level using Merritt and Cummins 1996; Larson 1997a, b; Larson et al. 2000. Specimens belonging to the Family Chironomidae were not classified to a lower taxon, and all other Diptera were classified only to order. These orders were given special attention based on the observation that the majority of taxa in these orders are pollution sensitive (Lenat 1988).

4.2.6 Statistical Analyses

The sediment data was analyzed using the G statistic with α=0.05. The quantity of sediment accumulated within each site collected during pre-harvest and post-harvest was compared to determine whether there was a significant difference.

For the insect data, the generalized linear model, with poisson errors, and log link was used (SAS 1988; McCullagh and Nelder 1989) with α=0.05. The assumptions of this model are that the residuals are homogeneous, normal, and independent. All of the assumptions were met. The number of Ephemeroptera, Plecoptera, and Trichoptera (EPT) were totaled and divided by the number of Diptera (including Chironomidae) (EPT/D) index that were observed for each sample (Resh and Jackson 1993). This index is based on the rationale that taxa belonging to EPT are pollution sensitive (Lenat 1988; Resh and Jackson 1993) and that members of Diptera are pollution tolerant, specifically Chironomidae when compared with members of EPT (Resh and Jackson 1993). A stressed community will reflect an imbalance within the invertebrate community (Resh and Jackson 1993). The EPT/D index was compared between the four sites for pre- and post-harvest using the explanatory variable of site and year as an interaction term. This model was also used to determine whether there was a statistical difference between pre- and post-harvest for the following response variables using the interaction term site and year: number of EPT, number of

Ephemeroptera, number of Plecoptera, number of Trichoptera, number of Diptera (excluding Chironomidae), and the number of Chironomidae.

For the purposes of the study design the interaction term site and year was of interest. If the response variable was found to be statistically significant, then from the parameter estimates it could be determined which buffer treatments were significantly different relative to the no harvest site (McCullagh and Nelder 1989). For this study design there were two years of data, with the method being the same for both years, and three explanatory variables: experimental treatment, year (pre- or post-harvest), and the natural changes between the two years for each of the experimental treatments – interaction term. If the interaction term was significant then the overall means between the treatments and the years could not be interpreted individually. Furthermore, by using this type of analysis and using the interaction term as the explanatory variable it is possible to take into consideration inter-annual variation, relative to the no harvest site.

The data were also analyzed using Krebs' (1991) RAREFACT to determine rarefaction values. Rarefaction gives an estimate of how many species are likely to be represented in a sample of a given size, thus allowing sample size to be standardized between treatments so that species diversity can be compared on an even basis between samples of variable sample size. It is empirical and can not be extrapolated beyond the maximum observed species numbers for a sample. The rarefaction values for the number of species for the sites were compared using the G statistic with $\alpha=0.05$ to determine if there was a significant difference between pre- and post harvest samples.

Results

Sediment accumulation increased significantly for the 20 m buffer with selective harvesting (Figure 4.1). This increase was significant ($p<0.0001$) and was four times greater post-harvest. The post-harvest results from other sites, including the no harvest, was not significantly different from the pre-harvest results.

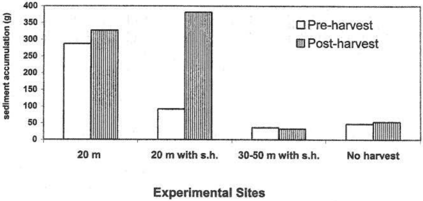

Figure 4.1. Comparison of total sediment accumulation between three experimental sites and the no harvest site pre- and post-harvest. Sites were 20 m buffer (20 m), 20 m buffer with selective harvesting (20 m with s.h.), and 30-50 m buffer with selective harvesting (30-50 m with s.h.).

For the EPT/D index the interaction term of site and year was not significant ($p=0.4995$, Table 4.1, Table 4.2). The number of EPT was significantly related to the interaction term of site and year ($p<0.0001$) and all three experimental buffers were significantly different from the no harvest site (Table 4.1, Table 4.2). The percent change for the number of EPT increased for the 20 m buffer while for the 20 m buffer with harvesting and the 30-50 m buffer with harvesting there was a decrease in percent change from pre- to post-harvest conditions (Table 4.1, Figure 4.2).

Table 4.1. Comparison of the number of each taxa and the number of species for each taxa for each of the experimental and no harvest sites for pre- and post-harvest. (N.B. Number of Diptera excludes Chironomidae).

	20 m (pre)	20 m (post)	20 m with s.h. (pre)	20 m with s.h. (post)	30-50 m with s.h. (pre)	30-50 m with s.h. (post)	No harvest (pre)	No harvest (post)
N EPT	72	86	188	42	172	70	185	73
N sp EPT	21	18	20	20	15	23	30	18
N trichoptera	23	26	82	9	162	59	115	34
N sp trichoptera	12	6	9	8	8	10	14	6
N plecoptera	4	9	3	2	1	2	18	17
N sp plecoptera	2	3	1	2	1	2	5	6
N ephemeroptera	45	51	103	31	8	11	11	6
N sp ephemeroptera	7	9	10	10	6	11	11	6
N chironomids	400	390	266	301	99	246	285	196
N diptera	22	15	144	28	17	283	711	241

Table 4.2. The p-values from the generalized linear model for several response variables for the experimental sites that differed from the no harvest (control) site.

Response variable	P-value	Sites statistically different from the no harvest site for the interaction term site and year
Number of EPT/D	0.4995	NA
Number of EPT	<0.0001	20 m; 20 m with harvesting; 30-50 m with harvesting
Number of Ephemeroptera	<0.0001	20 m; 20 m with harvesting; 30-50 m with harvesting
Number of Plecoptera	0.0001	20 m; 20 m with harvesting
Number of Trichoptera	<0.0001	30-50 m with harvesting
Number of Diptera	<0.0001	20 m; 20 m with harvesting; 30-50 m with harvesting
Number of Chironomidae	<0.0001	20 m; 30-50 m with harvesting

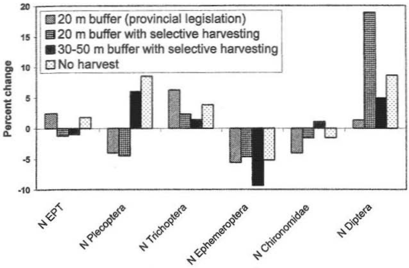

Figure 4.2. Percent change in various taxa between pre- and post-harvest observations for three experimental buffers and a no harvest (control) site. (N.B. Number of Diptera excludes Chironomidae).

The number of Ephemeroptera was significantly different for all three of the experimental buffers ($p<0.0001$, Table 4.1, Table 4.2) when compared with the no harvest site for the interaction term – site and year. The numbers observed decreased for each of the experimental sites, with the greatest decrease observed for the 30-50 m buffer with harvesting (Table 4.1, Figure 4.2).

The number of Plecoptera in both 20 m buffers (with and without selective harvesting) also differed significantly ($p=0.0001$, Table 4.1, Table 4.2) from the no harvest site for the interaction term site and year with the percent change of Plecoptera decreasing for both 20 m buffers (with and without selective harvesting; Figure 4.2). The 30-50 m with harvesting site was not significantly different from the no harvest site.

The number of Trichoptera was significantly related to the interaction term site and year ($p<0.0001$, Table 4.1, Table 4.2) with the number of Trichoptera increasing for all sites including the 'control'. The only site that differed significantly from the no harvest site was the 30-50 m buffer with harvesting ($p<0.0001$, Table 4.2). The largest percent increase was observed for the 20 m buffer (Figure 4.2). Upon examination of the raw data, a single genus, *Oxyethira* sp. was responsible for this large increase in the 20 m buffer site, pre- and post-harvest, increasing 52 times (n=3 pre-harvest; n=155 post-harvest) (Appendix 1).

The number of Diptera (excluding Chironomidae) was also significantly related to the interaction term site and year ($p<0.0001$) with all three experimental buffers differing significantly from the no harvest site (Table 4.1, Table 4.2). All sites, including the control, demonstrated increased numbers of Diptera with the largest percent increase observed with the 20 m buffer with harvesting; an 18.9 % increase (Table 4.1).

The number of Chironomidae was also significantly related to the interaction term site and year ($p<0.0001$) with the 20 m buffer and the 30-50 m buffer with harvesting both significantly different from the no harvest site. The 30-50 m buffer with harvesting demonstrated the only positive percent change, while the 20 m buffer demonstrated the greatest negative percent change in the number of Chironomidae (Table 4.1, Figure 4.2).

As shown by the values obtained from RAREFACT, the number of taxa of macroinvertebrates expected to be collected from samples of various sizes was greater for each of the experimental sites and the no harvest site during the post-harvest year when compared with the values obtained for the pre-harvest year (Figure 4.3). However, the difference observed was not statistically significant for any of the experimental sites. Thus, even if the sample size had been much greater than the effort in this study, the species diversity would not have significantly differed.

Discussion

The literature on the effects of forest harvesting on invertebrates provides conflicting results. For example, several studies state that higher densities of benthic invertebrates were observed for streams that had been logged (Newbold et al. 1980; Murphy and Hall 1981). Conversely, Smith (1980) and Trayler and Davis (1998) noted reduced invertebrate abundance following stream-side harvesting.

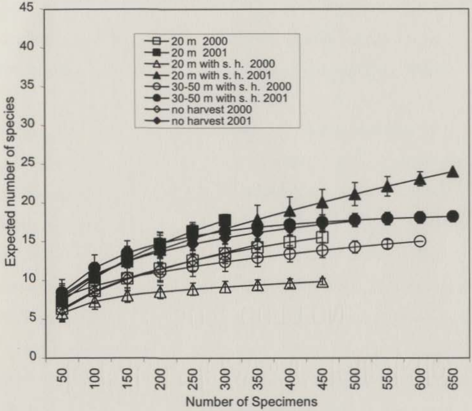

Figure 4.3. Rarefaction curves of expected number of species of macroinvertebrates to be collected from various sample sizes for the three experimental buffers and the no harvest sites, pre- and post-harvest.

Lenat et al. (1981) found that when sediment was added to a stream, the habitat available to invertebrates on rocks decreased resulting in a decrease in benthic density, however community structure did not change. Sedimentation affects several biotic communities at various trophic levels. It lowers the productivity of primary producers such as photosynthesizing plants, primary and secondary consumers such as benthic invertebrates, and secondary consumers, top carnivores and piscivores such as fish (Hartman et al. 1983; Waters 1995).

Physical changes to invertebrate habitat that may have caused the observed responses by invertebrates are not clear. Sediment accumulation increased for the 20 m buffer with selective harvesting for Hungry Brook post-harvest. It is known that populations of macroinvertebrates respond in various ways to disturbance. The Copper Lake study conducted in western Newfoundland did not show any clear trend related to increased sediment accumulation (Clarke et al. 1998). It has also been shown that macroinvertebrate biomass may increase with sediment addition with the proliferation of sediment tolerant taxa (Blosser 1984). Blosser (1984) found gradually decreasing numbers of *Hydropsyche* sp. with increasing fine sediments. Similarly, Barton (1977) found reduced densities of *Hydropsyche slossonae* with increased sediment. In contrast, Kohlhepp and Hellenthal (1992) found significantly greater densities of *Hydropsyche morosa* with increased sediment. In my study, in the 20 m buffer with selective harvesting, the number of *Hydropsyche betteni* was equal for pre- and post-harvest (Appendix 1). The number of *Hydropsyche betteni* in the 20 m buffer in pre-harvest decreased to zero post-harvest (Appendix 1). This site displayed an increase in sedimentation, however the increase was not significant. The 30-50 m buffer also displayed a decrease in the number of *Hydropsyche betteni* (Appendix 1), however sediment accumulation at this site, from pre- to post-harvest conditions, was almost unchanged. Furthermore, the large increase in sedimentation for the 20 m buffer with selective harvesting did not affect the number of Diptera (excluding Chironomidae) which increased dramatically (Appendix 1).

The number of Plecoptera decreased for both 20 m buffers, whereas the other sites increased. This decrease corresponds with the observed increases in sedimentation for the 20 m buffers, specifically the 20 m buffer with selective harvesting. Similarly, Carlson et al. (1990) found that Plecoptera decreased for logged sites, however the difference was not significant. Plecoptera are typically intolerant of pollution and their presence indicates good water quality (Lydy et al. 2000).

It has been suggested that buffer zones act primarily as "policeman" against logging near stream banks, or even more detrimental acts such as forwarding through streams (Newbold et al. 1980). Furthermore, the experimental buffer in my study that received the most intense harvesting (20 m buffer with selective harvesting) exhibited a large increase in sedimentation, thus suggesting that buffers of wider width may be more effective in "filtering" by reducing some of the overland flow of sedimentation from forwarding trails and possibly reducing the effects observed in this study on the invertebrate community.

Members of the orders EPT are known to be sensitive to pollution, so it is expected that the numbers of individuals in these orders would decrease with a decrease in water quality (Norris and Georges 1993). However, Murphy and Hall (1981) and Murphy et al. (1981) found that the density of invertebrates is higher in clearcut sites. This is similar to the findings of this study where the percentage change in the number of EPT only increased for the 20 m buffer and this response was driven by the large percentage increase in *Oxyethira* sp. Changes within the invertebrate community following forest harvesting have been attributed to changes within the food pathways which results in a change within the primary producer biomass (Erman et al. 1977; Vannote et al. 1980; Murphy et al. 1981; Gregory et al. 1987). Erman et al. (1977) suspected the changes in the invertebrate communities within logged streams was the result in a change of the stream's energy budget as a result of increased nutrients and light. These results might indicate a possible increase in primary production possibly from an increase in nutrient input within Hungry Brook.

Oxyethira sp. was also responsible for the large percent change increase between pre- and post-harvest for the number of Trichoptera in the 20 m buffer. This species is often associated with filamentous algae which it consumes (Winterbourn and Gregson 1989). Several authors have reported increased primary production after clearcuting when compared to forested areas (e.g. Gregory 1976; Johnson et al. 1986) and even thinning of the riparian canopy allowed for increased solar radiation with subsequent increased primary production (Burns 1972). In my study a treatment involving canopy removal was not included, and the other buffer treatments maintained full canopy closure, thus an increase in direct sunlight on Hungry Brook did not occur as a result of harvesting (pers. obs.). Therefore the dramatic increase in *Oxyethira* sp. is not likely related to increased primary production caused by an increase in solar radiation. Bormann et al. (1968) observed increased nutrient runoff in watersheds post-harvesting and if such an increase occurred in my study, it may have resulted in increased primary production, possibly resulting in the increased number of *Oxyethira* sp.

The findings of my study for Trichoptera numbers is consistent with Carlson et al. (1990) who observed caddisflies to be more numerous at logged sites. Trichoptera increased in my study in the two selective harvesting experimental sites, but the increase was less than that observed for the no harvest site. The 20 m buffer observed the greatest increase and was greater than the no harvest site.

The number of Diptera (excluding Chironomidae) displayed the largest percent increase of all invertebrate orders. Carlson et al. (1990) also found that true flies were significantly more numerous at logged sites. Diptera increased at all sites, including 'control' and only the 20 m with harvesting exhibited increases greater than the no harvest site. The vast majority of Diptera were members of the family Simuliidae and this taxa was responsible for the large increase observed for the 20 m buffer with selective harvesting. Similarly, Newbold et al. (1980) found large numbers of *Simulium* sp. at a logged site compared with none at the paired control site, however this taxa was sparse at another paired set of study sites.

The number of Chironomidae did not display any pattern with respect to pre- and post-harvesting. Both 20 m buffers and the 'control' displayed a decrease in percentage change, whereas the 30-50 m buffer with harvesting exhibited a slight increase. On the contrary, Erman et al. (1977) found increased numbers of Chironomidae in logged streams within their study.

The results of the rarefaction analysis indicated no significant difference in the species diversity between pre- and post-harvest conditions. Blosser (1984) found no clear relationship in the study they conducted between diversity and sediment content. Conversely, Newbold et al. (1980) found invertebrate diversity in unprotected streams to be lower than in controls. Erman et al. (1977) found similar results with control streams having much greater invertebrate diversity than logged streams. It appears the response of the macroinvertebrate community to forestry harvesting may be site specific, largely dependent on the resident fauna and the degree of disturbance to the stream. This clearly indicates the need to incorporate local data in forest management practices, particularly when considering the diverse and often contradictory findings in the literature on invertebrate response to disturbance.

Erman et al. (1977) concluded that buffers less than 30 m were ineffective as protective measures, however buffers greater than 30 m provided protection equivalent to conditions in unlogged streams. The results of my study are similar in that the buffer with the least amount of change overall was the 30-50 m buffer with selective harvesting.

My results suggest that increased sediment accumulation does not necessarily decrease the number of macroinvertebrates, however, a change in the taxonomic composition was found. As well, the potential for increased primary production, as suggested by the increase in algal consumers for the 20 m buffer in my study needs more investigation in future studies with similar environmental characteristics as those in Newfoundland.

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Chapter 5

Effects of managed buffer zones on salmonid population and biomass

5.1 Introduction

Salmonids depend greatly on the condition of the adjacent forests (Meehan 1991). The habitat requirements are precise, and activities that alter water quality, can have implications for fish productivity (Meehan 1991). Studies on the overall impact of forest harvesting on salmonids in streams has produced contradictory results (Burns 1972; Murphy and Hall 1981; Murphy et al. 1981; Bisson and Sedell 1984). Negative effects from stream-bed sedimentation on the survival of salmonid embryos and larvae have been reported (Coble 1961; Cordone and Kelley 1961; Shelton and Pollock 1966; Phillips et al. 1975; Hausle and Coble 1976; Alexander and Hansen 1983; Wesche 1985; Berkman and Rabeni 1987; Nelson et al. 1991; Waters 1995). Sedimentation rates are often accelerated as a result of forest harvesting (Shapley and Bishop 1965). Salmon and trout are primarily sight feeders therefore suspended sediment can influence feeding ability (Scruton et al. 1997). Streams must allow sunlight to penetrate for algae to grow, which are important for the insect population, a food source for trout and salmon (Scruton et al. 1997).

An increase in sedimentation can have irreversible effects on a fish population. Suspended sediment can cause difficulty for developing eggs and fry (Waters 1995). Sedimentation will reduce dissolved oxygen levels if water cannot percolate throughout the substrate of spawning beds (Wesche 1985; Nelson et al. 1991; Waters 1995). This movement of water is also imperative in removing metabolic waste (Wesche 1985; Waters 1995). It appears that any sediment less than 3 mm can cause adverse effects on salmonid production. For example, Phillips et al. (1975) found an increase in 1 to 3 mm sand in the spawning gravel from 20 % to 30 % decreased the emergence of coho salmon from 65 % to 40 %. Additionally, Shapley and Bishop (1965) concluded that salmon production is inversely related to percentage of stream substrate less than 0.833 mm diameter.

Fry and juvenile habitat can be affected by increased sediment deposition. Sediment also fills interstitial spaces which are vital for winter survival of fry (Furniss et al. 1991; Waters 1995). Bustard and Narver (1975) demonstrated that salmonid fry show a strong preference for clean cobble as compared to silted cobble. Some studies have suggested that a temporary increase in productivity can be expected after logging, as long as no major changes occur to the stream channel (Bisson and Sedell 1984), thus no increase in sedimentation would occur if the stream channel is not eroded.

In this chapter, the impact of forest harvesting on salmonid populations and biomass was the focus, with consideration given to the potential role of sedimentation and macroinvertebrate production on salmonids. As well, population estimates and biomasses were studied separately for both species of young-of-the-year compared with 1+ fish.

5.2 Methods

5.2.1 Study area

Refer to section 2.2.1

5.2.2 Study sites

Refer to section 2.2.2

5.2.3 Electrofishing methods

Two electrofishing sites were established within each of the three experimental sites and the no harvest site. Each electrofishing site had two barrier nets installed to barricade a 200 m^2 section (50 m-length x 4 m-width) of the stream from immigration and emigration of fish, with the downstream net being installed first. The barrier nets were made of black fly screen with mesh size used to prevent young of the year salmonids from escaping or entering the electrofishing section. Rubble and small boulders were positioned along the bottom of the net to secure the net to the substrate. The area was then intensely electrofished using a Smith-Root, Type VIIIA electrofisher making a minimum of three sweeps of the section (Scruton and Gibson 1995). Electrofising was conducted between August 29 and September 6, 2000 (pre-harvest), and between August 22 and

August 26, 2001 (post-harvest). The removal method was used to estimate population (Zippin 1958). The time was recorded in seconds according to the electrofisher's internal timer to ensure consistent effort on each run. The electrofishing team consisted of three people, one person operating the electrofisher, the second person with a dip net, and the third person carrying a live well in which captured fish were placed prior to processing. The fisher started at the downstream end of the station and slowly fished across the stream in standardized widths, gradually moving upstream towards the upper barrier net. Between each run, all salmonid specimens collected during the sweep were identified to species, measured (to the nearest mm), and weighed (to the nearest gram). All collected specimens were then released downstream from the electrofishing site.

5.2.4 Statistical Analyses

The population estimate for each site was obtained from Microfish 3.0 which uses a maximum likelihood estimator (Van Deventer and Platts 1983). The average of the two population estimates within each experimental site was compared with the estimate for the no harvest site for pre- and post-harvest. The data were analyzed using the G statistic from a chi-square distribution with one degree of freedom, with α=0.05. This statistic measures goodness of fit of the data (Devore 1995).

Biomass was determined by multiplying the population estimate for each site by the average weight (g) of a given species captured within the site to yield an estimated biomass per electrofishing site. This number was then divided by the surface area of the electrofishing site to obtain a biomass estimate in g/m^2. The G statistic was then used to determine whether the biomass was significantly different for pre- and post-harvest for each experimental site compared with the no harvest site.

To separate fish into young-of-the-year and 1+ and older, for brook trout, any fish less than 70 mm fork length was considered to be young-of-the-year. All remaining brook trout were considered to be 1+ and older. Similarly, the same method was used for Atlantic salmon, except the maximum fork length for young-of-the-year was 60 mm.

5.3 Results

The population estimate of Atlantic salmon significantly increased for all three experimental buffers compared to the no harvest (Figure 5.1, Table 5.1). However, Atlantic salmon biomass decreased in the 20 m buffer site, and increased within the 20 m with selective harvesting and 30-50 m with selective harvesting, but not significantly (Figure 5.2, Table 5.2).

The population estimate of brook trout increased for all three experimental sites, but the increase was only significant for the 20 m with selective harvesting and the 30-50 m with selective harvesting (Figure 5.3, Table 5.1). The 20 m buffer with selective harvesting and the 30-50 m buffer with selective harvesting both displayed increases in biomass, however only the biomass increase in the 20 m buffer was significant (Figure 5.4, Table 5.2). The 20 m buffer observed a slight decrease in biomass (Figure 5.4).

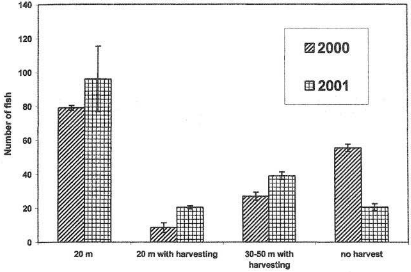

Figure 5.1. Atlantic salmon population estimates for three experimental sites and the no harvest site for pre- (2000) and post-harvest (2001). Error bars = standard error.

Table 5.1. The p-values obtained from the G statistic for population estimates for the experimental sites compared with the no harvest site for pre- and post-harvest for both species of salmonids.

Experimental Site	**Species**	**P-value**
20 m (provincial legislation)	Atlantic salmon	<0.0001
	Brook trout	0.3428
20 m buffer with selective harvesting	Atlantic salmon	<0.0001
	Brook trout	0.0007
30-50 m buffer with selective harvesting	Atlantic salmon	0.0001
	Brook trout	0.0059

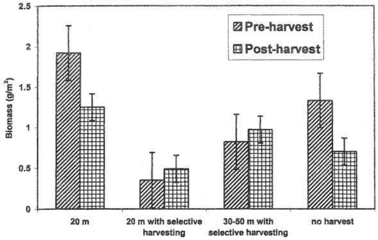

Figure 5.2. Biomass of Atlantic salmon for the three experimental sites and the no harvest site pre- (2000) and post-harvest (2001).

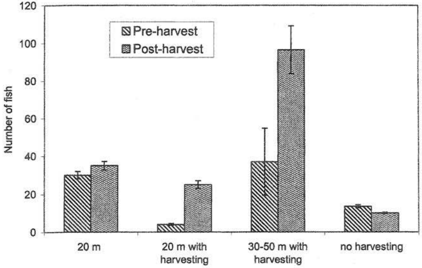

Figure 5.3. Brook trout population estimates for three experimental sites and the no harvest site for pre- (2000) and post-harvest (2001). Error bars = standard error.

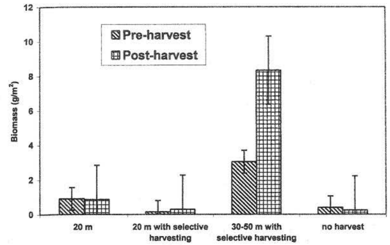

Figure 5.4. Biomass of brook trout for the three experimental sites and the no harvest site pre- and post-harvest. Error bars = standard error.

Table 5.2. The p-values obtained from the G statistic for biomass for each of the experimental sites compared with the no harvest site for pre- and post-harvest for both species of salmonids.

Experimental Site	**Species**	**P-value**
20 m (provincial legislation)	Atlantic salmon	1.0
	Brook trout	0.8875
20 m buffer with selective harvesting	Atlantic salmon	0.7083
	Brook trout	0.0191
30-50 m buffer with selective harvesting	Atlantic salmon	0.6892
	Brook trout	0.5598

The population estimate for young-of-the-year Atlantic salmon increased significantly for all three experimental sites (Figure 5.5; Table 5.3). However, the biomass for young-of-the-year Atlantic salmon were not significantly different compared to the no harvest site (Figure 5.6; Table 5.4).

The population estimate for young-of-the-year brook trout significantly increased for the 20 m buffer with selective harvesting and the 30-50 m buffer with selective harvesting (Figure 5.7; Table 5.3). The biomass for young-of-the-year brook trout was not significantly different from the no harvest site for any of the experimental buffers (Figure 5.8; Table 5.4).

The population estimate for 1+ and older Atlantic salmon increased for all three experimental sites, however it was only significant for the 20 m buffer (Figure 5.9; Table 5.5). The biomass for 1+ and older was not significantly different from the no harvest site for any of the three experimental sites (Figure 5.10; Table 5.6).

For 1+ and older brook trout, all three experimental sites displayed increases in population estimates, however, none of the increases were significant (Figure 5.11; Table 5.5). The biomass for 1+ and older brook trout remained relatively unchanged for both 20 m buffers, however an increase was observed for the 30-50 m buffer, but it was not significant (Figure 5.12; Table 5.6).

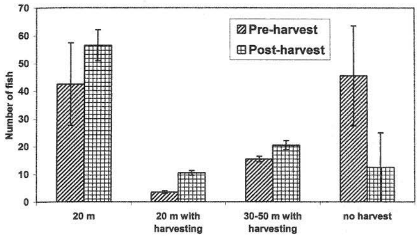

Figure 5.5. Population estimate of Atlantic salmon young-of-the-year for the three experimental sites and the no harvest site pre- and post-harvest. (Error bars=standard error).

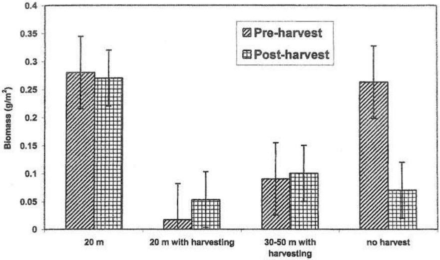

Figure 5.6. Biomass of Atlantic salmon young-of-the-year for the three experimental sites and the no harvest site pre- and post-harvest. (Error bars = standard error).

Table 5.3. The p-values obtained from the G statistic for population estimate of young-of-the-year for the experimental sites compared with the no harvest site for pre- and post-harvest for both species of salmonids.

Experimental Site	Species	P-value
20 m (provincial legislation)	Atlantic salmon	<0.0001
	Brook trout	0.5271
20 m buffer with selective harvesting	Atlantic salmon	0.0002
	Brook trout	0.0002
30-50 m buffer with selective harvesting	Atlantic salmon	<0.0001
	Brook trout	0.0002

Table 5.4. The p-values obtained from the G statistic for biomass for young-of-the-year for the experimental sites compared with the no harvest site for pre- and post-harvest for both species of salmonids.

Experimental Site	Species	P-value
20 m (provincial legislation)	Atlantic salmon	0.8003
	Brook trout	1.0
20 m buffer with selective harvesting	Atlantic salmon	0.7850
	Brook trout	0.7970
30-50 m buffer with selective harvesting	Atlantic salmon	0.8176
	Brook trout	1.0

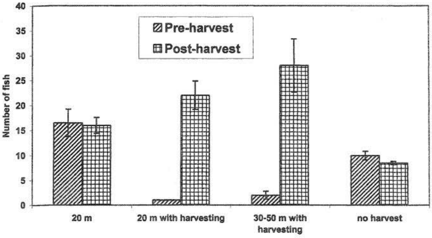

Figure 5.7. Population estimate of brook trout young-of-the-year for the three experimental sites and the no harvest site pre- and post-harvest. (Error bars=standard error).

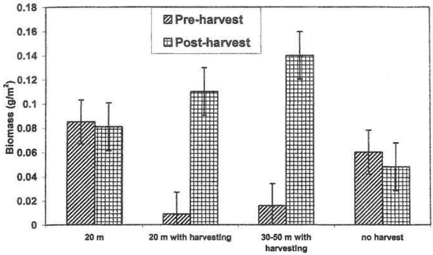

Figure 5.8. Biomass of brook trout young-of-the-year for the three experimental sites and the no harvest site pre- and post-harvest. (Error bars=standard error).

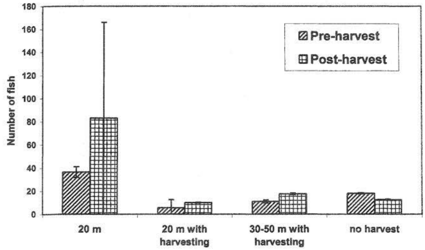

Figure 5.9. Population estimate of Atlantic salmon 1+ and older for the three experimental sites and the no harvest site pre- and post-harvest. (Error bars = standard error.

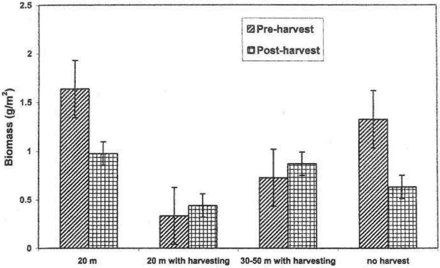

Figure 5.10. Biomass of Atlantic salmon 1+ and older for the three experimental sites and the no harvest site pre- and post-harvest. (Error bars=standard error).

Table 5.5. The p-values obtained from the G statistic for population estimate for year 1+ for the experimental sites compared with the no harvest site for pre- and post-harvest for both species of salmonids.

Experimental Site	Species	P-value
20 m (provincial legislation)	Atlantic salmon	0.0038
	Brook trout	0.2367
20 m buffer with selective harvesting	Atlantic salmon	0.1380
	Brook trout	0.2733
30-50 m buffer with selective harvesting	Atlantic salmon	0.1069
	Brook trout	0.0943

Table 5.6. The p-values obtained from the G statistic for biomass for year 1+ for the experimental sites compared with the no harvest site for pre- and post-harvest for both species of salmonids.

Experimental Site	Species	P-value
20 m (provincial legislation)	Atlantic salmon	1.0
	Brook trout	0.8875
20 m buffer with selective harvesting	Atlantic salmon	0.2367
	Brook trout	0.8415
30-50 m buffer with selective harvesting	Atlantic salmon	0.6892
	Brook trout	0.5839

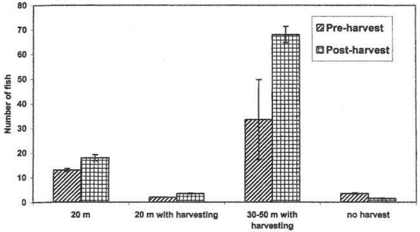

Figure 5.11. Population estimate of brook trout 1+ years for the three experimental sites and the no harvest site pre- and post-harvest. (Error bars=standard error).

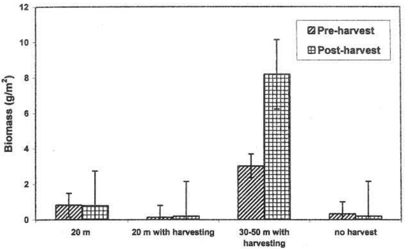

Figure 5.12. Biomass of brook trout 1+ years for the three experimental sites and the no harvest site pre- and post-harvest. (Error bars= standard error).

5.4 Discussion

Bisson and Sedell (1984) found that salmonid density in streams within clearcuts was greater than streams in old-growth forests by a factor of 1.6. Murphy et al. (1981) found that trout (cutthroat and rainbow) had greater abundance and biomass in clear-cut sites than in forested sites. Similarly, increases in population estimates of brook trout were found in my study. The 20 m buffer displayed an increase of 1.2 times post-harvest, the 20 m buffer with selective harvesting increased 6.25 times, and the 30-50 m buffer with selective harvesting increased 2.6 times. Increases in population estimates of Atlantic salmon were also observed. The 20 m buffer increased 1.2 times, the 20 m buffer with selective harvesting increased 2.4 times, and the 30-50 m buffer with selective harvesting increased 1.4 times.

The Copper Lake study conducted in western Newfoundland found brook trout densities decreased (Clarke et al. 1998). Young-of-the-year brook trout significantly decreased in density within one of the affected streams (Clarke et al. 1998). Clarke et al. 1998 also state that the effects on brook trout populations became more evident and greater if the sediment source was in close proximity. Bisson and Sedell (1984) noted that salmonid biomass was greater in streams in clearcuts than streams in forested sites with an average biomass 1.5 times greater. The only increase in biomass (for all fish or for both age classes) that was significant in my study was for brook trout in the 20 m buffer with selective harvesting, with 2.1 times greater biomass post-harvest.

Increased sedimentation can have a devastating effect on incubating eggs. Sediment can decrease the permeability of oxygen, carbon dioxide, and other metabolites, thus decreasing survival (McNeil and Ahnell 1964; Scrivener and Brownlee 1989). Further, if sediment pores are too small, then alevins may become restricted preventing intergravel movement (Phillips 1971; Scrivener and Brownlee 1989). For my study the population estimate for young-of-the-year Atlantic salmon increased significantly for all three experimental sites. The biomass for young-of-the-year Atlantic salmon increased slightly for the 20 m buffer with selective harvesting and the 30-50 m buffer with selective harvesting, and the 20 m buffer showed a slight decrease, however, none of these biomass changes were statistically significant.

The population estimate for young-of-the-year brook trout also increased significantly for the 20 m buffer with selective harvesting and the 30-50 m buffer with selective harvesting. The biomass of young-of-the-year brook trout was not significantly different post-harvest for any of the experimental sites. The 20 m buffer displayed a very slight decrease, while both the 20 m buffer with selective harvesting and the 30-50 m buffer with selective harvesting both displayed a large increase post-harvest.

From these results, it appears that there was no negative impact on salmonid emergence or development for young-of-the-year.

The population estimate of 1+ and older Atlantic salmon increased for all three experimental sites, however only the 20 m buffer increase was significant. Biomass of 1+ and older Atlantic salmon displayed increases for the 20 m buffer with selective harvesting and the 30-50 m buffer with selective harvesting, and the 20 m buffer observed a decrease, however none of the changes observed were significant.

The population estimate of 1+ and older brook trout increased for all three experimental sites, however none of the increases were significant. Biomass of 1+ and older brook trout displayed increases for the 20 m buffer with selective harvesting and the 30-50 m buffer with selective harvesting, and the 20 m buffer observed a very slight decrease, but none of these changes were statistically significant.

There may be several factors that are contributing to this increase in population of salmonids. The results from another component of this project on macroinvertebrates (Chapter 4) indicated there may have been an increase in nutrient input, with an associated increase in algal production, resulting in increased macroinvertebrate production as potential "fish food", which in turn could have resulted in increased salmonid production.

Another component of research in this project was sedimentation. Despite large observed increases in sedimentation for the 20 m buffer with selective harvesting, salmonid population estimates and biomass did not appear to be negatively affected. The results of this component of the study demonstrated no negative impacts on salmonid population estimates. Despite decreases in salmonid biomass for some of the experimental sites, the decreases observed were not significant. Further research is required to determine whether the increase in algal consuming macroinvertebrates was the result of increased primary production which could provide more insight into possible reasons for the observed salmonid increase.

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Chapter 6

General Discussion

6.1 Overview of the impact of harvesting with managed buffer zones

Riparian vegetation plays a large role in determining the health of the associated aquatic community. The ecological processes of a stream are intimately related to those of the surrounding terrestrial ecosystem (Garman and Moring 1991). Therefore, a compromise has to be reached that permits the retrieval of natural resources without reducing the ecological integrity of the area; a delicate balance. The primary purpose of my research project was to determine whether harvesting within buffer zones resulted in adverse effects to aquatic fauna and habitat. The results from the components of the study yielded varied results on the impact of harvesting.

Sediment deposition within the 20 m buffer with selective harvesting was significantly affected, however the 30-50 m buffer with selective harvesting did not appear to have been affected, as indicated by the slight change between pre- and post-harvest conditions. In the 20 m buffer with selective harvesting the increase in sediment was 14.5 times greater post-harvest, an indication that buffers of this width are insufficient in 'filtering' overland flow containing sedimentation from entering a stream. The implication of this increase within this site could adversely affect the aquatic habitat and fauna as sediment is known to affect the productivity of an aquatic environment at various trophic levels.

The effect of this large increase in sedimentation within the 20 m with selective harvesting site was not determined. The number of Plecopterans, known as indicators of water quality, decreased the greatest within both 20 m buffers, while the 30-50 m with selective harvesting site increased similarly to the no harvest. Plecoptera are typically intolerant to pollution, such as increased sediment, and their presence indicates good water quality (Lydy et al. 2000). This suggests 20 m buffers, with or without selective harvesting may not be sufficient to maintain the macroinvertebrate community. However, the 30-50 m buffer with selective harvesting appeared to have been sufficient in terms of not affecting the macroinvertebrate community.

Conversely, the number of Diptera exhibited a large increase within the 20 m buffer with selective harvesting, much greater than the other experimental sites. Slaney et al. (1977) found that Dipterans increased with sedimentation, therefore the increase in Dipterans observed in Hungry Brook may be the result of increased sedimentation. The other significant change within the macroinvertebrate community was a 52 times greater number of *Oxyethira* sp. post-harvest compared with pre-harvest observations for the 20 m buffer. This genus is an algal consumer. It may be that this increase indicated an enhanced nutrient input from increased nutrient runoff as observed by Bormann et al. (1968).

The impact of increased sedimentation on the salmonid fishes appeared to have been negligible. Increases in brook trout and Atlantic salmon population estimates were significant for all three experimental sites except for brook trout within the 20 m buffer site. The biomass of brook trout did not appear to be negatively affected by the increased sedimentation for any of the experimental sites. The biomass of Atlantic salmon did exhibit a decrease within the 20 m buffer, however it was not significant. The young-of-the-year salmonid populations increased for all three experimental sites, with the exception of brook trout within the 20 m buffer, which was relatively equal between pre- and post-harvest observations. The young-of-the-year salmonid biomass was not significantly different between pre- and post-harvest for any of the experimental sites. If there was any effect of sediment on egg incubation and hatching success, it would have been reflected in young-of-the-year populations. For 1+ and older Atlantic salmon population estimates, the 20 m buffer displayed the only significant increase between pre- and post-harvest. The brook trout population estimates were not significant for any of the experimental sites. As well, the biomass of both salmonid species were not significantly different for any of the experimental sites.

The impact the changes within the macroinvertebrate community will have on the salmonids may not be evident until 2002 or later, therefore caution is warranted in drawing conclusions as to the impact on the salmonid populations. Changes appear to have had minimal impact on salmonid productivity during the two year time frame of this study. However, effects of harvesting on fish population, the highest trophic level in the aquatic food chain of Hungry Brook, would be expected to lag observed changes in habitat and at lower trophic levels (e.g. macroinvertebrates).

Water temperature is an important determinant of the occurrence and distribution of various aquatic fauna (Gordon et al. 1992). Within this study, emphasis was placed upon the potential impact of water temperature regimes on brook trout and Atlantic salmon. Changes in water temperature and other water quality parameters have restricted brook trout within headwater streams (MacCrimmon and Campbell 1969), and several studies on the effects of timber harvesting on stream temperature have shown significant maximum stream temperature increases during the summer months (Brown and Krygier 1970; Lynch et al. 1984; Scruton et al. 1998).

The results of my study do not suggest any adverse effects of harvesting on water temperature regimes for brook trout. The number of hours within various water temperature classes, defined by brook trout thermal requirements, changed significantly only in the optimum category with a decrease in the 20 m buffer with selective harvesting and an increase in the 30-50 m buffer with selective harvesting. The most notable observation was the lethal and stress temperature classes were not significantly affected. A slightly better growth potential could be expected within the 30-50 m buffer with selective harvesting due to the increase in the number of hours for the optimum class, and in contrast, for the 20 m buffer with selective harvesting a slight decrease in growth potential could be expected. Growth potential could also affect biomass, however the biomass of brook trout increased in both of these sites, with a significant increase observed for the 20 m buffer with selective harvesting.

For Atlantic salmon, within the 30-50 m buffer with selective harvesting, there was an increase in the number of hours within the upper and lethal thermal regimes. All other sites and temperature classes were not significantly different. It is possible the increases in the upper and lethal thermal categories could result in a decrease in growth potential, and maybe a slight decrease in population.

The findings of my study can only be considered within the two year time frame of the research. For this study, only one year of post-harvest data was collected. Therefore any observations should be cautiously assessed with respect to assessing harvesting effects with managed buffer zones. It is possible some of the effects, negative and positive, may take longer than one year to become evident. Over the longer term, some of the components of this project are being studied by the Department of Fisheries and Oceans. These include sedimentation, temperature regime, salmonid population estimates and biomass.

6.2 Future research

Further research on the effects of timber harvesting within buffers need to focus on two areas: the impact of forwarding and nutrient runoff. Firstly, the impact of forwarding on soil compaction, and subsequent sub-surface hydrological characteristics needs to be investigated further since field observations from my study suggest this may be causing long term damage and may be the principal source of sedimentation. The increase in algal consuming Trichopterans in the 20 m buffer site indicates an increase in nutrient runoff. This study did not investigate the nutrient levels within the experimental buffers, however it needs to be investigated to determine whether there could be an increase in primary production from increased nutrient runoff. And if there is an increase, to what extent could the nutrient input increase primary production, within the nutrient poor waters of Newfoundland.

With limited research on these potential infractions, and Newfoundland's unique environmental conditions, the effect of this method of buffer zone management should be studied over a longer term. More data on sedimentation could assist in determining whether increased sedimentation occurs over a long time frame or just immediately post-harvest. As well, the impact of these buffers on water temperature in the longer term should be studied. Furthermore, changes to salmonid populations and biomass should be studied to determine the long term effects of this method of buffer zone management.

The initial study design for this project included two no harvest site replicates. Despite immense effort by the researcher to maintain these two areas, there was partial non-compliance, consequently one of the areas was harvested, removing a no harvest replicate. Therefore, any future research collaborating with the forestry industry must include assurance that harvesting plans will be strictly adhered to, and this compliance needs to be enforced.

6.3 Recommendations

For this study, a six wheel drive Fabtek forwarder was used with "Eco-tracks" to enclose the two rear wheels on each side. It was observed that compaction and rutting was minimal when the forwarder drove over the slash beds left by the harvester. However, compaction and rutting were severe in numerous areas as a result of the machine making numerous passes over the same area. In these established paths, large ruts were evident. Also, extensive rutting was very noticeable in the areas that had wet terrain. Whenever the machine did not make repetitive passes over the same area, and when wetter terrain was avoided, the effect was minimal.

These deep ruts, in excess of 0.50 m resulted in water becoming channeled and confined to these paths, and following repeated passes created very turbid water with high concentrations of suspended sediment. The channeled water eventually would overflow these ruts and the result was suspended sediment flowing across the slash covered cutover, and entering the stream. Throughout the landscape there are numerous small tributaries entering headwaters streams, and it was observed that some of these tributaries became 'feeder' tributaries for these large ruts. These ruts then become "artificial streams" that begin to dominate the landscape.

My recommendation to ameliorate these negative impacts includes altering forwarding schedules to minimize soil disturbance. Winter harvesting in areas near waterbodies could result in negligible soil disturbance. Furthermore, by forwarding during winter months, frozen soil conditions would minimize the impact that forwarder ruts and soil compaction had when conditions were wet. The findings and observations of this study suggest that concentrated forwarding within watersheds should be a concern. Subsequent rutting and siltation were observed as problems even when they were in excess of 200 m from Hungry Brook. Frequently used forwarding trails with mineral soils exposed in the ruts appeared to be the major problem particularly on sustained slopes of stream valleys. Further, methods to mitigate sediment transport from forwarder trails to streams should be developed.

Of the different experimental buffers for this study, the 30-50 m buffer with selective harvesting appeared overall to have had the least impact, specifically in terms of sedimentation and invertebrate community changes. The 20 m buffer and the 20 m buffer with selective harvesting showed a decrease in the number of Plecoptera, possibly indicating a decrease in water quality. Further, the 20 m buffer with selective harvesting site exhibited a large increase in sedimentation.

The concept of ecoforestry is based on the understanding that there are thousands of life forms that form complex communities and which live within forests, or are intimately dependent on forests such as aquatic ecosystems. These life forms have intrinsic value which needs to be recognized and respected (Thom 1997). Ecoforestry attempts to maintain these complex interactions while harvesting forest resources to meet the requirements of humans over the long-term, in other words, sustainable forestry (Thom 1997). It places priority first on maintaining the ecological integrity of forests, so that they can provide the economic needs of humans and the forestry industry. To this end, if future research suggests the 30-50 m buffer with selective harvesting is superior to the current method of riparian protection, more widespread application of this method to provide adequate protection along waterbodies should be considered. To maintain sustainable forests maybe we should be attempting to leave as much value as we can in a forest, instead of taking as much value as we can out of a forest; a new concept that is long overdue within a century old industry in Newfoundland.

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Appendix 1

Site	Species	Number
20 m (pre)	*Habrophlebia vibrans*	15
	Eurylophella prudentalis	1
	Acerpenna pygmaeus	27
	Drunella cornuta	2
	Hydropsyche betteni	9
	Ceraclea sp.	1
	Micrasema sp.	4
	Polycentropus sp.	2
	Pycnopsyche sp.	1
	Oxyethira sp.	3
	Leuctra ferruginea	4
	Aeshna eremita	1
	Anabolia sp.	2
	Hydroptila metoeca	1
	Chironomids	400
	Simuliidae	22
20 m (post)	*Oxyethira sp.*	155
	Leptophlebia cupida	3
	Simulliidae	17
	Drunella cornuta	1
	Ceraclea sp.	1
	Chironomids	99
	Habrophlebia vibrans	1
	Eurylophella prudentalis	2
	Anabolia sp.	1
	Hydroptila metoeca	1
	Acerpenna pygmaeus	1
	Leuctra ferruginea	1
20 m with s.h. (pre)	Chironomids	390
	Simuliidae	15
	Acerpenna pygmaeus	40
	Drunella cornuta	7
	Habrophlebia vibrans	3
	Eurylophella prudentalis	1
	Leuctra ferruginea	7
	Hydropsyche betteni	23
	Oxyethira sp.	4
	Anabolia sp.	1
20 m with s.h. (post)	Chironomids	246
	Simuliidae	283
	Oxyethira sp.	23
	Hydropsyche betteni	19

	Acerpenna pygmaeus	27
	Drunella cornuta	10
	Pycnopsyche sp.	2
	Habrophlebia vibrans	3
	Ceraclea sp.	8
	Podmosta macdunnoughi	1
	Eurylophella prudentalis	2
	Ephemerella subvaria	1
	Cheumatopsyche pettiti	1
	Diplectrona sp.	1
	Leuctra ferruginea	1
	Lype diversa	3
30-50 m with s.h. (pre)	Chironomids	266
	Simuliidae	144
	Habrophlebia vibrans	51
	Eurylophella prudentalis	2
	Drunella cornuta	11
	Acerpenna pygmaeus	38
	Hydroptila metoeca	30
	Hydropsyche betteni	42
	Oxyethira sp.	7
	Leuctra ferruginea	3
	Leptophlebia cupida	1
	Pycnopsyche sp.	2
	Glossosoma nigrior	1
30-50 m with s.h. (post)	Chironomids	285
	Simuliidae	711
	Drunella cornuta	14
	Oxyethira sp.	52
	Hydropsyche betteni	18
	Hydroptila metoeca	22
	Leptophlebia cupida	3
	Acerpenna pygmaeus	19
	Podmosta macdunnoughi	2
	Ceraclea sp.	10
	Pycnopsyche sp.	1
	Eurylophella prudentalis	6
	Leuctra ferruginea	16
	Polycentropus sp.	6
	Habrophlebia vibrans	10
No harvest (pre)	Chironomids	301
	Simuliidae	28
	Acerpenna pygmaeus	11
	Habrophlebia vibrans	9
	Leptophlebia cupida	7
	Isoperla transmarina	1

	Leuctra ferruginea	1
	Oxyethira sp.	1
	Cheumatopsyche pettiti	1
	Pycnopsyche sp.	3
	Polycentropus sp.	2
	Polycentropus sp.	1
	Hydroptila metoeca	1
	Aeshna eremita	1
No harvest (post)	Chironomids	196
	Simuliidae	241
	Lype diversa	2
	Podmosta macdunnoughi	15
	Leuctra ferruginea	3
	Leptophlebia cupida	6
	Habrophlebia vibrans	3
	Pycnopsyche sp.	6
	Acerpenna pygmaeus	10
	Oxyethira sp.	23
	Aeshna eremita	1
	Hydropsyche betteni	1