Effects of a Selection Harvest in a Riparian Buffer Situated in Manitoba Forest Management License 01

By

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A practicum submitted in partial fulfillment
of the requirements for the degree of
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Abstract

Forest resource managers have become increasingly aware of the shortcomings of conventional riparian management. Riparian buffers may enter a decadent state as trees age and begin to acquire attributes that may reduce the effectiveness of the buffer and threaten non-timber resource values associated with the riparian zone. A selective harvest was designed and implemented in an experimental riparian buffer within FML 01, aimed at circumventing the problems associated with traditional riparian resource management. Due to the ecological and recreational value of the riparian resource, harvesting was designed with the intent to integrate various existing non-timber values. A pilot study was subsequently implemented to address the initial effects of selective harvesting on buffer floristics, the avian community, erosion and sedimentation and streamside aesthetics.

Primary data was collected for one year prior to, and for one year following, harvesting. Results pertaining to floristics suggested that the forest within the buffer was in a late seral state. Overstory vegetation throughout much of the study area was dominated by *Picea* sp. and *Abies balsamea* with an admixture of *Populus tremuloides*, *Fraxinus nigra* and other hardwood species. In accordance with harvesting design, all harvested blocks retained at least 30% of pre-harvest merchantable stems. Following harvesting, it was determined that the pre-harvest plant community may have played a stronger role in the floristics of the post-harvest understory than did cutting. Changes in species abundance were noted within the bird community following cutting. Various species with known mature stand affiliations were encountered less often while others affiliated with early forest seral stages were encountered more often. Avian diversity was, by a small margin, highest in harvested blocks. Factors not related to cutting, that may have been effecting abundance values, were not ruled out. Upland erosion was noted only on steep terrain in which the soil organic layer had been removed. Data and visual observations suggested that sedimentation was a natural and spatially variable process in the study area. Accumulations in sediment traps were felt to have originated from the stream bank rather than from upland locations. Periodic exposure of sediment traps, however, rendered sedimentation data unreliable, thereby precluding any meaningful inference. Harvesting

was found to have imparted a significantly negative impact on the streamside aesthetic. Comments provided by volunteers suggested that the appearance of trees played a strong role in scenic valuation. The presence of hardwood material may have, to a degree, ameliorated the effect of softwood removal in certain areas. The initial effects of harvesting on streamside scenery may, however, diminish with the continual regeneration of plant material.

Acknowledgements

Table of Contents **Page**

List of Tables

List of Figures

List of Plates

Glossary of Terms

Forest Resource Buffer:
A portion of a forest community within a managed area, left undisturbed to reduce or eliminate the effect of harvesting in an adjacent portion of the landscape, on non-timber values.

Chronosequence:
The mosaic of forest successional (seral) stages present across a given landscape, reflecting the disturbance history of that landscape (Barbour et al. 1999).

Commercial Clearcutting:
The complete removal of all merchantable species of timber (Wedeles and VanDamme 1995).

Relative Abundance:
The number of individuals encountered of a particular species expressed as a percent of the total number of individuals encountered of all species combined.

Riparian Zone:
'The interface between the aquatic and the terrestrial environment' (Gregory 1991),

The, *'portion of the terrestrial landscape from the high water mark towards the uplands where vegetation may be influenced by elevated water tables or flooding, and the ability of the soil to hold water'* (Naiman et al. 1993)

Selective Cutting:
The removal of timber that meet selected criteria including, species and diameter limitations (Wedeles and VanDamme 1995).

Selection Harvesting:
A form of un-even aged management involving the felling of a portion of merchantable trees in all age/size classes, either singly or in small groups (Wedeles and VanDamme 1995).

Silvicultural Clearcut:
A term referring to the complete removal of all trees (Wedeles and VanDamme 1995).

Un-even Aged Management:
A harvesting practice that promotes the maintenance of a diversity of tree age classes within a stand (Wedeles and VanDamme 1995).

Chapter 1: Introduction

1.1 Background and Issue

The responsibilities of forest managers in Canada have expanded beyond timber production to reflect a broad range of forest interests within the context of Sustainable Forest Management (Canadian Council of Forest Ministers 1997, 1998). In few places has this been more apparent than within the riparian zone. Riparian areas have received special management consideration within various landscapes, particularly during the latter part of the 20th century (Oakley 1988, Naiman and deCamps 1997, Naiman et al. 1993). A substantial body of literature has emerged during this period suggesting that riparian systems tend to exhibit a relatively diverse array of environmental processes and species (Naiman and deCamps 1997). Various authors have observed higher numbers of species and individuals of small mammals (Doyle, 1990) and non-game birds (LaRue *et al.* 1995, Gates and Giffen 1991, Oakley et al. 1985, Miller 1951, Stauffer and Best 1980, Thomas et al. 1979) in riparian as opposed to upland forests although similar work has not produced these observations (McComb and McGarigal 1992). Further, riparian vegetation is felt to play an important role in maintaining water quality (Anderson *et al.* 1976, Bilby 1988, Brown 1971) and streamside scenery in forested landscapes. Although little work has been conducted to delineate the role of the riparian zone in the boreal region, or the effect of streamside harvesting, riparian areas have been managed differently than boreal uplands as distinct landscape features.

Manitoba Conservation and the Pine Falls Paper Company (PFPC), the primary forestry interest within Manitoba Forest Management License 01, have recognized the value of riparian forests. Subsequently, the use of zero-disturbance resource buffers has been incorporated into standard operating procedures to reduce or eliminate the potential effects of adjacent upland commercial logging. By acting as a filter against potential upland erosion and acting to preserve riparian soil stability, riparian buffers serve to maintain water quality and protect fish resources. Furthermore, these areas assume added importance by maintaining the scenic value for water recreation and can act as travel corridors for wildlife moving between forest patches.

Although conventional riparian practices such as buffer implementation are felt to have been effective in ameliorating the potential effects of upland harvesting, forest managers have become increasingly aware of inadequacies concerning 'zero disturbance' buffer implementation. Buffered forests become susceptible to Spruce Budworm (*Choristoneura fumiferana*) and wind-throw (Keenan *et al.* 1990) as the forest continues to age. This can result in a deteriorating and visually unappealing buffer with reduced effectiveness. Wind thrown stems have the potential to accumulate in the aquatic environment as coarse woody debris. Although woody debris provides important structure for aquatic biota, excessive debris may negatively effect fish movements (DFO 1992, Narver 1971) in narrow streams and can be prohibitive to water recreation. Further, riparian buffers may provide a corridor through which Budworm infestations may spread and move into upland plantations and adjacent mature stands. Tree mortality, due either to Budworm damage or to wind-throw, may eventually lead to excessive fuel build-up making the buffer itself, and surrounding area, increasingly susceptible to fire.

One issue that emerges raises questions concerning the longevity and usefulness of zero-disturbance riparian buffers in a disturbance-driven ecosystem. The boreal landscape is a mosaic of plant communities that reflect both local environmental conditions as well as disturbance history (Alexander and Euler 1981, Barbour *et al.* 1999, Scott 1995). Riparian areas are not widely felt to have been exempt from the kinds of disturbance agents, such as fire, that return boreal stands to early seral stages (Agee 1988, Naiman et al. 1993, Naiman and deCamps 1997). In the absence of disturbance, forest stands can eventually reach a decadent state and accumulate many of the problems currently associated with riparian buffers. Subsequently, zero-disturbance protocol may, over time, be incompatible with the original intent of riparian buffer implementation, leading forest managers to seek alternatives for the future management of riparian areas.

To meet this challenge locally, the Pine Falls Paper Company (PFPC), in co-operation with the Manitoba eastern region Integrated Resources Management Team (IRMT), selected an existing riparian buffer in which to conduct an experimental harvest of softwood material. The IRMT operates regionally within the Manitoba government, as a

multi-disciplinary management body to which resource and land use issues within FML 01 are referred. The Province of Manitoba has promoted the integration of multiple resource values in managed landscapes through government policy (Manitoba Department of Natural Resources 1989b, 1996) and through the recent shift toward ecosystem based management within Manitoba's forestry sector (KPMG 1995). Subsequently, a selection harvest was chosen by both PFPC and the IRMT based on the potential for this system to be the least disruptive to existing non-timber resource values.

Current provincial forest management guidelines provide that limited harvesting may be conducted within a riparian management area pending regional approval (Manitoba Department of Natural Resources 1989a). Although harvesting has been conducted in the past within riparian management areas, there is a lack of information concerning its effects on associated non-timber resource values. In view of the variety of resource issues that converge in the riparian area, future initiatives in riparian management may be best served through a more informed approach. In this context, the PFPC has expressed a commitment to the development of a long-term integrated riparian buffer management plan designed to account for non-timber values and make better use of its timber resource.

1.2 Objective

The purpose of the study was to act as the initial phase in a sequence of monitoring to analyze the effect of a selection harvest within a riparian buffer, on various resources associated with the riparian environment.

1.2.1 Specific Objectives

Specifically, the objectives of the study were:

- to describe the riparian forest timber resource along an experimental riparian buffer, and design a selection harvest primarily targeting larger over-mature soft wood within this buffer;

- to provide wildlife resources information regarding the impacts of the selection cut design by examining avian abundance and diversity within the experimental buffer;

- to determine upland erosion and aquatic sedimentation with the aim of providing information on the impact of selection harvesting on water quality;

- to determine changes in frequency and composition of the forest understory and ground level vegetation in response to a selection cut;

- to determine the effect of a limited selection harvest within a riparian buffer on stream-side aesthetics and;

- to make recommendations for future considerations in the context of riparian resource buffer management within the PFPC Forest Management License 01.

1.3 Study Limitations

The purpose of this study was to act as a pilot project aimed at identifying the immediate impacts, within its components, of a limited selective harvest in a riparian resource buffer. Further, the results are aimed at providing critical baseline data which may be useful in subsequent follow-up research for the delineation of long term impacts. Intrinsic limitations to the study do, however, emerge. Such limitations are manifested in the following;

- The significance of riparian buffers in the boreal forest is not limited to the objectives undertaken within this study. This is particularly true from a wildlife perspective. Various wildlife groups such as amphibians, fish and reptiles, as well as certain fur-bearers such as the river otter (*Lutra canadensis*) are dependant upon aquatic habitat for at least part of their life cycle. Species which are less ecologically tied to the aquatic environment such as Moose (*Alces alces*), White-tailed Deer (*Odocoileus virginianus*), various species of rodents, bats and fur-bearers (Cross 1988) may also be present in riparian areas in greater densities than in upland settings (Raedeke *et al.* 1988). Riparian areas may also support unique aquatic and terrestrial invertebrate communities. Most insects, for example which are aquatic during early life stages, will spend their aerial or adult stage in the riparian zone (Voller 1998). Subsequently, the impacts of harvesting in riparian buffers which are addressed in this study are not exhaustive. Future considerations pertaining to the integration of timber and non-timber resources in riparian areas would be best served by examining a more complete range of potential impacts.

- The information and recommendations derived from this study may not be fully transferable into other riparian management settings. A variety of geomorphic conditions, forest types and associated wildlife communities may be found in areas subject to riparian management protocol in the Pine Falls FML. In addition, while rivers, streams and lakes are similarly subject to riparian management, they invariably differ hydrologically. Since it is reasonable to assume that the impact of harvesting will not be uniform among riparian resources across the boreal landscape, the

5

information yielded by the present study may have limited application in other riparian settings.

- The time frame utilized in the collection of data, was felt to have imposed certain limitations in identifying immediate impacts as well as the forecasting of long-term harvesting impacts within the various components of the study. Given the potential for bird populations to fluctuate over time, pre-harvest data spanning more than one year may have provided a more adequate frame of reference in determining the immediate effect of harvesting on the bird community (Bibby and Burgess 1993). Similarly, spatial variations in rate of erosion may be evident from one segment of the river to another prior to harvesting. A lack of adequate baseline information may confound attempts to separate natural variations in erosion from that which may potentially have been related to harvesting activities. Lastly, the pattern of natural mixed-wood forest succession can be complex and may exhibit broad changes in forest structure throughout the development cycle. Consequently, the long term successional trajectory is difficult to predict based on one year of post harvest data. Although limited inferences were made based on trends seen in the initial year following harvesting, existing literature was required to address possible long term impacts of harvesting on the plant community.

- The number of bird listening stations was limited by the operability of the study area. Avian data may not be sufficiently robust to determine conclusively the effect of a selective harvest on a mixed-wood forest bird community in the first year following a selective harvest. Rather, data resulting from this study may be useful in revealing which species of the pre-harvest bird community may continue to utilize the post-harvest stand, as well as those which may potentially be benefited. Existing literature further provided an important frame of reference in interpreting results.

Chapter 2: Literature Review

2.1 The Riparian Zone

The ecology and the management of riparian areas in forested landscapes has received substantial attention throughout the latter half of this century. Much of this attention has focused on landscapes subject to intensive commercial logging. Riparian zones are, in large part, limited by streamside topographical and geomorphic attributes and tend to occupy a relatively small portion of a given landscape. In terms of ecological importance however, riparian zones have been described as the "aorta of an ecosystem" (Wilson 1979) and an important source of biodiversity (Naiman *et al.* 1993). Subsequently, riparian areas have been treated separately as a unique landscape feature in managed forests.

The riparian zone has been defined broadly as 'the interface between the aquatic and the terrestrial' environment (Gregory *et al.* 1991). Specifically it is commonly held that the riparian zone extends onto the upland environment as far as the high water mark during periods of flooding, but includes the upland forest influenced by flooding (Meehan *et al.* 1977, Naiman & DeCamps 1997). Naiman *et al.* (1993) describe the riparian corridor as,

'...that portion of the terrestrial landscape from the high water mark towards the uplands where vegetation may be influenced by elevated water tables or flooding, and the ability of the soil to hold water.'

Drengson & Taylor (1997) address this upland zone of influence by separating the riparian ecosystem spatially into two zones; the *'riparian zone'*, defined as 'the wet forest' adjacent to lakes, wetlands and waterways, and the *'riparian zone of influence'*, defined as the drier upland forest. Bilby (1988) offers that the riparian zone is 'the direct expression of the influence of the aquatic system on the terrestrial'.

The riparian zone has been examined from the perspective of a variety of ecological parameters. Perhaps the most obvious expression of the aquatic influence is conveyed through the development of terrestrial plant communities along a particular land-water interface (Brinson and Verhoeven 1999). The influence of the aquatic environment on

7

that of the terrestrial diminishes laterally along a moisture gradient which is in turn reflected in terrestrial vegetation patterns and wildlife utilization (Bilby 1988). The influence of the water itself, though, is subject to local variations in geomorphology and perhaps explains more about the terrestrial ecology of the land–water interface than in merely examining riparian floristics.

Surface attributes such as topography and soil-type play an active role in determining the inland reach and gradient of the aquatic influence (Brinson and Verhoeven 1999). Neither can realistically be viewed separately. Topographical attributes of the landscape associated with the riparian environment typically include; the *active channel*, immediately adjacent to the aquatic environment itself, the floodplain, an area prone to periodic or seasonal inundation, and lower hill-slopes further inland which are not flooded but are influenced by flood events. The role of topography in determining the inland reach of the aquatic influence is, however, also linked to the surface type of the terrestrial component (Bilby 1988).

The geomorphic composition of the terrestrial surface can subtly modify the role of topography in determining the upland reach of the aquatic influence. A stream or lake-side surface composed primarily of exposed bedrock, as is frequently the case throughout the boreal eco-zone, commonly results in the riparian area ending abruptly where rock and water meet. The presence of a well developed soil layer, however, will permit a water table to extend laterally, at least until it is contained by underlying bedrock (Bilby 1988). The proximity of the water table to the rooting zone of vegetation, can subsequently influence terrestrial vegetation as far as topography will permit.

The boreal landscape is a mosaic of geomorphic discontinuities in both surface type and topography that can act either to facilitate or limit the inland expression of the aquatic system (Agee 1988, Naiman and DeCamps 1997, Naiman *et al.* 1993). The inland reach of the aquatic component can, and does, change dramatically over relatively short distances with variations in geomorphology. Consequently, the boundaries of the riparian zone itself can be highly irregular along any particular land-water association as is

expressed, perhaps most obviously, in riparian plant communities and floristic dynamics (Agee 1988).

2.2 Floristics of Riparian Resource Buffers:

The land managed by PFPC features a mosaic of riparian plant communities already buffered or that may, in the future, be subject to riparian buffer implementation. Depending upon local geomorphic factors, riparian 'leave-strips' may support plant associations that are both highly productive and structurally complex. Depending upon method of implementation riparian buffers may feature a mixture of vegetation communities, including significant portions of forests, beyond the reach of aquatic influence, that bear a closer resemblance to typical upland communities.

Riparian vegetation can be characterized as a series of plant communities or associations, that differ in structure and composition, blending into one another along the laterally diminishing influence of the aquatic environment (Bilby 1988, Hall 1988). Consequently, the floristic mosaic observed in the riparian area is not merely a result of the *presence* of water, but rather by the change in ground and surface water availability. This moisture gradient is itself modified by the topographical attributes of the managed landscape (Agee 1988, Barbour *et al.* 1999, Naiman *et al.* 1993). The commonly used template for riparian floristics categorizes riparian zones into distinct upland, floodplain and active channel plant communities (Agee 1988). This is, in part, a convenient way to categorize riparian floristic patterns, but also illustrates changes in vegetation patterns under the changing moisture regime driven largely by ground elevation.

Patterns in vegetation, however, also reflect the influence of disturbance events both related to the associated body of water (fluvial) and of past disturbance patterns more typical of boreal uplands (non-fluvial) (Brinson and Verhoeven 1999, Hall 1988, Naiman *et al.* 1993). Flooding events associated with spring melt water tend to impart seasonal disturbance patterns that are reflected in lowland floodplain riparian vegetation. Lower regions prone to seasonal or otherwise frequent flooding tend to support a hardwood canopy (Agee 1988). Many hardwoods, *Fraxinus nigra* being a good example, are

simply better suited to the year round wetter conditions of the rooting zone and are subsequently better able to compete (Agee 1988, Barbour *et al.* 1999) Further, softwoods such as *Picea glauca* and *Abies balsamea* begin metabolic activity much earlier in the spring than do most hardwoods (Agee 1988), and subsequently tend not to establish or persist in areas prone to spring flooding. Within the floodplain environment however, the duration, frequency and magnitude of flooding events is subject to variations most certainly related to shifts in climatic patterns (Agee 1988, Naiman and DeCamps 1997). It is conceivable therefore, that variations in the lateral extent of flooding over long periods of time, can create a spatial overlay of different aged stands along areas with a history of inundation. Higher reaches of the riparian lowlands that may be somewhat less prone to regular inundation will exhibit a floristic transition to riparian uplands.

Floristic composition, disturbance agents and forest succession in the higher reaches of the riparian zone differ from that of the floodplain, bearing an increasing resemblance to that of comparable non-riparian upland sites (Agee 1988). The lateral magnitude, frequency and duration of flooding diminishes in the riparian uplands and with it, the competitive advantage imparted on the hardwood communities found in the lower reaches. Softwoods become increasingly important in the riparian stand as the influence of the wetted channel diminishes (Agee 1988). Upland riparian forest composition and succession reflects not only the relatively drier moisture regime, but also the history of past non-fluvial disturbance agents such as fire, Budworm and blow-down (Brinson and Verhoeven 1999).

While it may be convenient to categorize riparian floristics into broadly defined community patterns, these patterns can be, and are, frequently modified (Naiman and DeCamps 1997). The meanderings of streams and rivers are characterized by the gradual removal of soil material from streambanks, and the subsequent deposition of this material elsewhere (Naiman and Decamps 1997). Older, softwood dominated stands, tend to occupy positions immediately adjacent to the gradually eroding outer edge of a river bend, while younger stands often occupy depositional areas on the inner side of a stream bend (Everitt, 1968). A thin band of shrubs or young hardwood tree species

may persist along the eroding edge of a stream bend due to favorable light conditions. The erosion process may however, prevent longer lived trees from becoming incorporated into over-story vegetation.

While implemented to protect the riparian zone itself, riparian buffers may also incorporate significant portions of non-riparian upland forests. Buffers are usually implemented in fairly uniform strips of forest, however the riparian zone itself, does not often reflect this spatial uniformity (Agee 1988). The precise spatial extent of lateral influence imposed by the aquatic environment on that of the terrestrial is rarely obvious (Naiman and Decamps 1997). That is to say, that the transition from "riparian" to "upland" is obscure. Where the water is met by exposed bedrock or sufficiently steep terrain the transition into non-riparian upland may occur at, or very close to, the water's edge. The design and implementation of riparian buffers was intended to capture spatial variations in the riparian environment along any particular land-water association as well as to preserve the various resources associated with riparian areas including streamside aesthetics, water quality and wildlife habitat (Campbell *et al.* 1996, Compton *et al.* 1988, Darveau *et al.* 1995, Davies and Nelson 1994, Richardson and Miller 1997). Subsequently, riparian buffers may contain riparian forests, as well as adjacent, non-riparian upland component. The structure and composition of the non-riparian forest component in riparian buffers can be as variable as the boreal forest itself.

2.2.1 Community Response to Harvesting

The notion that riparian forests are a significant source of floristic biodiversity and wildlife habitat, and that riparian buffers can help to maintain these attributes, is widely held. Consequently, the floristic response to timber extraction within riparian buffers is of considerable importance. Though riparian areas have, in the past, been described as 'sensitive to disturbance' (Thomas 1979), they are nonetheless themselves the 'product of disturbance' (Agee 1988). The implementation of buffer zones did not divorce these subsequent corridors of forest from the ecological role of disturbance that is pervasive throughout the forests of the boreal region. While the practice of maintaining riparian buffers as zones of zero-disturbance may warrant review, issues have emerged pertaining

to the ecological resilience to timber harvest, of forest historically driven by natural disturbances such as fire.

Although the effect of cutting on successional pathways in boreal riparian forests has not specifically received widespread attention, trends observed in non-riparian upland sites may potentially be comparable to logging in the upland reaches of riparian buffers (Fuller-DeWeile *et al.* 1998). Forest disturbance processes, structure and successional pathways typical of harvested upland sites may also be active in areas of riparian uplands. (Agee 1988, Bilby 1988, Cummins 1974). Subsequently, the response of plant communities to upland harvesting may be applicable to riparian uplands.

Historically, wildfire has imposed perhaps the most pervasive source of disturbance in the western boreal forest (Kimmins 1997), the effect of which serves as a useful frame of reference when examining the forest's resilience to logging. The resilience of a particular plant community is largely a function of the interplay between the nature of the disturbance itself and the life history attributes of affected vegetation (Ehnes 1998). In view of the historical role of fire in the boreal landscape, it is important to identify the ecological differences at play in post-logging plant communities and the potential long-term implications pertaining to forest recovery. Pickett *et al.* (1987) suggests three parameters, also reviewed by Ehnes (1998), that influence the nature of post disturbance communities: site availability, species availability and species performance.

The capacity for logging to create new sites for post disturbance colonization is considerably less than that of fire (Zasada 1986). Since communities exposed to fire suffer significantly greater plant mortality than do logged communities, fire will create a greater area available for colonization (Ehnes 1998). Depending on its intensity and frequency, fire will also significantly reduce the thickness of the soil organic layer, or eliminate it altogether. This generally increases the suitability for seed germination and seedling survival in many species and creates an overall greater variety of colonization sites. Logging can disturb the organic layer and expose the mineral A-horizon due to the

activity of machinery and timber extraction, however the areal extent of soil disturbance is considerably less, particularly during winter operations.

Post logging communities can be distinguished by the response of three types of species as described by Ehnes (1998). The First type of species are exotics which can be introduced on logging equipment. The second are species that survive logging or, as is the case with *Abies balsamea* (Rowe 1983), species that are less able to able to cope with fire and establish easily on logged sites. The third type of species distinguishing logged communities, is the absence of those which either require fire directly, or some aspect of the post fire environment. These species may exhibit poor natural regeneration after logging.

Differences in species performance relating to type of disturbance can be examined comparatively in terms of competition, exposure to sunlight and factors relating to soil processes (Ehnes 1998, Flinn & Wein 1977, Maclean *et al.* 1983). Plants that regenerate naturally *in situ* following a burn may exert a competitive advantage over invading species. This is contingent on increased radiant energy not exceeding the environmental tolerance levels of regenerating vegetation, subsequently inhibiting productivity (Ahlgren 1960). Harvesting, however, may actually reduce the intensity of competition that occurred prior to harvesting (Ehnes 1998). Light becomes less of a limiting factor following the removal of merchantable over-story species, initially alleviating competition for light. The stress of competition may also be reduced when harvesting induces the die-back of species either unable to tolerate the sharp reduction in humidity, or are simply less able to compete under higher light intensities (Crites & Dale 1995, Sims *et al.* 1989). Most plants, however, are suppressed by shade and are more productive under higher light intensities.

Plants exhibit a significant response to a change in the availability of light (Barbour *et al.* 1999). An increase in light conditions due to a decrease in canopy cover is perhaps the most obvious environmental change in both logged and burned communities. A burn will usually eliminate cover to a sufficient degree as to permit light to reach the ground

uninhibited by foliage. The shade cast by snags and any residual cover, if sufficient, may provide small patches of habitat suitable for plant species either intolerant of, or unable to compete in, the new light regime. Since logging results in considerably less plant mortality than does fire, post logging residual cover can modify the intensity of incoming radiant energy to the benefit of certain species at lower levels.

The nature of residual cover in post-logging communities relates strongly to the availability of light in the successional development of the forest at the time of logging. Succession in a post-fire mixed-wood community is usually characterized by an early, hardwood dominated upper canopy giving way over time to one that is increasingly dominated by conifers (Barbour *et al.* 1999). Conifers typically pose a greater barrier to light penetration than do deciduous species. As the forest ages, light subsequently becomes an increasingly limiting factor. Light passing through a deciduous canopy is sufficient to support well developed herb and shrub layers as well as the establishment of shade tolerant conifers (Scott 1995). As deciduous trees age and die back, conifers will acquire greater importance in the upper canopy and will exert shade conditions unsuitable to shrubs and most herbs. Shrub and ground level vegetation subsequently become increasingly restricted to patches of higher light intensity that persist under the remaining deciduous component of the upper canopy (Scott 1995). As the forest ages, an understory of *Abies balsamea* often emerges due to this species' ability to tolerate shade, and to take advantage of openings created in the canopy by dying hardwoods, wind-throw and retreating shrubbery. Since logging will disturb understory and ground layers to a far lesser degree than will a fire event, stage of succession at the time of harvest will consequently influence the structure and composition of residual cover.

The different effects of disturbance type on the physical, chemical and biotic properties of soil, can have a profound influence on species performance. The immediate physical effect of fire at the soil level is that litter, portions or all of the soil organic layer and components of live material are converted to ash. Several physical effects are related to this process. The removal of the organic layer creates a more suitable environment for seed germination. Soil temperatures in burned sites tend to be higher than in cleared but

14

unburned sites. When surface components are burned, the resulting layer of ash can reduce surface albedo thereby allowing the soil to become heated (Barbour *et al.* 1999). This may aid in seed germination and seedling productivity, but may also stimulate the activity of soil microbiota. Further, the elimination of bryophytes, which can retain large amounts of water may also subsequently lead to enhanced soil moisture. The presence of hydrophilic compounds in litter and standing vegetation, however, may ultimately impede soil moisture and lead to erosion if such compounds are released in significant quantities.

Fire has a profound effect on soil chemistry. Nutrients held in standing biomass are released in a form that is available to vegetation and move into the rooting zone through the leaching action of rainwater. Since plant colonization and regeneration is rapid following even relatively intense fires, the loss of nutrients by leaching is not significant (MacLean *et al.* 1983). Similarly, loss of nutrients through volatilization (nitrogen, organics) is more than offset by nutrients re-entering the soil (Ehnes 1998). Soil pH in mature coniferous communities can range at extremely low values (3-6) due to the acidity of accumulated litter (Gilliam 1991). Plant growth and nutrient release through decomposition is often impeded by high soil acidity. Soil pH will increase following fire due to the release of mineral bases in the soluble ash, consequently improving plant growing conditions. Organics will be destroyed if temperatures range between 200-300°C in the organic layer (Knapp *et al.* 1998). Nitrogen in its various forms (nitrates, ammonia amino acids) will volatilize between 100-200°C. Moisture in the organic layer, can play an important role in this respect because temperatures in the organic layer will not exceed 100°C until all water is burned off.

The ecology of soil microbiota following fire are only generally understood, but nonetheless have implications for post-fire plant species performance. Any changes in soil microbiota are most noticeable in upper soil levels (Wright & Bailey 1982). Broadly speaking, the effect of fire is to reduce fungi and increase soil bacteria and actinomycetes. While fungal succession is thought to be analogous to the succession of higher plants following fire, Barbour *et al.* (1999) infer that fungal pathogens in mature plant

communities may be destroyed if the upper soil level is sufficiently heated. Higher temperatures in the upper soil layers and the increase in soil pH and nutrient levels following a burn will enhance bacterial activity. This has important implications for plant productivity as nitrogen lost through volatilization can be recovered through the increased activity of free-living nitrogen fixing bacteria (Barbour et al. 1999).

Logging has an adverse effect on the local nutrient pool in forested communities. Timber harvesting has a capacity to cause a significant loss of nutrients through the export of biomass out of the system in post-logging communities (Kimmins 1977, Tamm et al. 1974). Whole tree harvesting (the removal of all above-ground tree biomass), had been subsequently modified with the intent of offsetting the loss of nutrients through timber extraction by leaving nutrient rich residual debris (branches, and crown foliage) on site. Though the foliage of residual debris may be rich in nutrients, there can be a considerable decline in the fertility of the soil in post logging communities even where such mitigative techniques are employed.

Various studies have revealed the leaching of soluble nutrients from the soil profile as a result of logging (Bormann and Likens 1974, Bormann and Likens 1979, Corbett et al. 1978, Dahlgren & Driscoll 1994, Tamm et al. 1974). Nutrient leaching in logged communities is strongly linked to soil pH. A low pH value indicates a lack of exchangeable base ions (i.e.: Ca^{2+}, Mg^{2+}, K^+, Na^+, and NH_4^+). This is a natural characteristic of mature coniferous communities. Post-fire soil pH will rise due to the massive re-entry of base ions into the soil. Logging does not increase soil pH, even where considerable residual debris has been left, because nutrient release is slow and lacks sufficient concentration. Meanwhile, increased post-logging soil temperatures enhances the activity of nitrifying bacteria in converting ammonium (NH_4^+) to nitrate. Nitrates are particularly mobile in the soil and more readily leached. Consequently, post-logging soil pH may actually decrease, as was the case at the Hubbard Brook experimental Forest (Bormann & Likens 1979), when leached nitrates are bound to basic cations (Dahlgren and Driscoll 1994, Likens et al. 1970, Reinhart 1973).

The argument might also be made, that the acid leachate of coniferous residual debris might also act to prolong post-logging soil acidity.

Long term attributes of forest succession in post-logging communities in boreal riparian or upland settings, are not well understood. Existing work has, however, revealed differences that pertain to rate of recovery. Recently, Ehnes (1998) showed that recovery in the understory and upper canopy was slower over the medium term (37 years) on logged sites than on those which had been burned. This was translatable into lower overall medium term community resilience to logging. These results were generally attributed to the failure of logging to rejuvenate the soil and stimulate seed dispersal and germination. Short term resilience (13 years) was, however, higher for herbs and understory species on logged sites, but declined by the medium term. Since many species are actually shade suppressed, the subsequent increase in light intensity, combined with reduced competition and poorer tree regeneration, may have imparted an initial head-start on these species over their conspecifics on burned sites (Ehnes 1998). Lower nutrient availability may have eventually offset the initial advantage created through canopy removal. It was therefore felt that lower soil fertility may have resulted in the decline in this initial higher rate of recovery.

While the over-story of logged communities may be slower to recover, there is also the potential that the relative species abundance of the over-story that eventually does emerge, may have shifted away from that of the original canopy. An Investigation by Yang and Fry (1981) into forest succession on 5-30 year-old cut-overs revealed that *Abies balsamea* was by far the dominant conifer while *Picea glauca*, *P. mariana* and *Pinus banksiana* exhibited relatively poor regeneration. The frequent inability of logging to create adequate site conditions for the establishment of *P. mariana* and *P. banksiana* is understandable since both species are post-fire thrivers and require relatively moist, exposed mineral soil to establish (Carleton and MacLellan 1994). Both *P. glauca* and *A. balsamea*, however, are intolerant of fire and must disperse into a burn from the adjacent forest. The propensity for *A. balsamea* to dominate in the degree found by Yang and Fry implies that regeneration is less tied to soil disturbance than is *P. glauca*. This is

17

somewhat evidenced during post-fire succession, as *A. balsamea* can dominate the understory in mixed-wood stands of advanced age by taking advantage of canopy openings created by wind-throw or tree decay (Palidwar 1997). Additional management considerations emerge where logging promotes the regeneration of *A. balsamea* due to the susceptibility of this species to Budworm infestation (DesGranges and Rondeau 1992).

2.3 The Avian Community

Riparian areas tend support greater numbers and diversity of breeding birds (Darveau *et al.* 1995, Oakley *et al.* 1985, Knopf and Samson 1988, Stauffer and Best 1980, LaRue 1990, Naiman *et al.* 1993, Naiman and Decamps 1997), although opposing trends have been observed in the Pacific northwest (McComb and McGarigal 1992). The effect of timber harvesting on riparian avifauna in the boreal forest has drawn little attention. The designated 'zero disturbance' status traditionally applied to riparian buffers throughout the region may not have translated into a sufficient need for this type of information. Subsequently, limited inferences must be made from the more widely examined response of bird communities to upland timber extraction. Although riparian floristics tend to differ from those of associated uplands, the current understanding of habitat selection by birds of forested landscapes may reasonably allow such inferences to be made.

2.3.1 Habitat Selection

Bird species, whether or not they are specifically affiliated with riparian forests, have certain habitat requirements. Productive foraging sites and suitable nesting cover are among those that are the most obvious (Cody 1985a). Certain species have relatively specific habitat requirements while others are less discriminating and may be observed throughout a broad range of habitat. The selection of habitat by any particular bird is undoubtedly not a response to any one particular attribute of the environment, but rather a response to a suite of characteristics. Some environmental cues may, however, play a greater role than others depending on the species being considered. For example, the presence of water riffles and suitable perching sites might be of greater importance to a

18

Belted Kingfisher (*Megaceryle alcyon*) than the presence of conifers in the canopy of the adjacent forest.

The cumulative effect of the many factors influencing avian habitat selection (floristics and structure, inter-specific competition, food supply, species morphology, etc.), may result in an infinitely complex overlay through which the perception of suitable habitat is filtered. Clearly within the context of the current discussion, an examination of the full range of environmental cues affecting habitat selection would neither be practical nor appropriate. The following overview will focus primarily on the role of floristic structure in habitat selection and the related impact of harvesting on forest bird populations.

2.3.2 Habitat Structure

Floristic structure is commonly perceived to elicit a strong influence on habitat selection in birds (Cody 1985a, Freedman *et al.* 1981, Temple *et al.* 1979, Welsh 1987). That is to say, the spatial distribution of both horizontal (landscape) and vertical (vegetation height, forest layering) floristic patterns offer structural cues that help indicate habitat suitability. The presence or absence of a particular bird species may be correlated to certain structural attributes within the forest. Canada Warblers (*Wilsonia canadensis*) in the boreal regions, for instance, are usually associated with dense hardwood understory (Wedeles and Van Damme 1995). Blackburnian Warblers (*Dendroica fusca*), on the other hand, are often associated with the presence of tall conifers, the middle and upper third being where this species spends much of its time (deGraaf and Rappole 1995, Morse 1976, 1968, Titterington *et al.* 1979, Wedeles and Van Damme 1995, Welsh 1992a). The modifications of stand structure as a result of timber extraction will tend to elicit changes in bird community composition and population densities (Welsh 1987). The magnitude of these changes tends initially to reflect the degree to which the stand itself has been modified as well as the range of structure that bird species in the affected community are able to exploit.

2.3.2.1 Stand Age: Shifts in Community Due to Logging

Harvesting has been likened to events, such as wildfire, which return mature forest stands to early seral stages (Welsh 1981). Many, although not all forest birds, are associated with a certain range along the structural continuum of forest succession (Anderson 1979, Shugart and James 1973, Welsh 1981, 1987, 1992a). Existing work has suggested that the extraction of mature trees from the forest canopy often results in a shift in bird abundance from species associated with the mature forest, to those associated with early stages of forest succession (Crawford and Titterington 1979, Welsh 1981, 1987, 1992a).

Much of the work aimed at addressing the role of stand age in boreal settings has been conducted in logged stands (Crawford and Titterington 1979, Welsh 1981, 1987, 1992a & b). Welsh (1981) found significant differences in bird species composition in mixed-wood stands that ranged from 1-220 years following last disturbance. His findings suggested that species typical of the early stages in post-logging stand succession included Alder Flycatchers (*Empidonax alnorum*), Mourning Warblers (*Oporornis philadelpia*), Chestnut-sided Warblers (*Dendroica pennsylvanica*) and the White-Throated Sparrow (*Zonotrichia albicollis*). Species associated with mid successional forests included the American Redstart *(Setophaga ruticilla)*, Red-eyed Vireo (*Vireo olivaceus*), Veery Thrush (*Catharus fuscenscens*), and the Black and White Warbler (*Mniotilta varia*). Species associated with late successional forests following logging or fire included the Bay-breasted Warbler (*Dendroica castenea*) and Golden-crowned Kinglet (*Regulus satrapa*).

The work of Titterington *et al.* (1979) produced similar results in *Abies balsamea* cut-overs of different ages in Maine. Species typical of 2-year-old cut-overs included Dark eyed juncos and White-throated Sparrows. Chestnut-sided Warblers and Mourning Warblers were more typical of 3-5 year old cut-overs characterized by low shrubs and young regenerating hardwoods. Red-eyed Vireos, American Redstarts and Black and White Warblers became more common with further hardwood regeneration between 7-12 years after cutting.

Many species are typically found over a broader range of the successional continuum with some occupying stands of all ages. Welsh (1981, 1987, 1992a & b) suggests that these species may be less affected by harvesting than those restricted to a certain range of forest succession. Welsh's findings suggest that the Magnolia Warbler (*Dendroica magnolia*), Canada Warbler, Black-capped Chickadee (*Parus atricapillus*), Ruffed Grouse (*Bonasa umbellus*) and Swainson's Thrush (*Catharus ustulatus*) seem to lack an affinity for any particular seral state. Titterington *et al.* (1979) made similar observations of Magnolia Warblers having found this species routinely occupying habitat ranging from newly cut as well as mature stands of *Abies balsamea*. The density of this Warbler did, however, vary with stand age.

Welsh offers two possible explanations for these observations. First, some species will simply utilize a relatively wide range of cover and food resources. The versatility of birds, wood warblers in particular, has been extensively documented (Cody 1981, Cody 1985b, Collins 1981,). Secondly, species not found to be sensitive to seral state may be responding to the presence of micro-habitat rather than stand age (Welsh 1992a). Presumably, small scale, structural vestiges of earlier seral stages may persist or develop throughout stand succession. Certain species may respond favorably to these fine-scale structural attributes within the maturing stand. Swainson's Thrushes, for example, were observed by Welsh (1987) to have almost identical habitat associations in very different seral stages.

Subsequently logging may not affect all forest species the same way. Species associated with early stages in forest succession may become more abundant as the forest is increasingly depleted of vertical structure. This trend is generally accompanied by a decrease in the abundance of species strongly linked to over-story foliage. The magnitude of the shift in species composition abundance driven by harvesting, depends largely on the amount of timber that is removed from the stand, or conversely, the structure that remains (Wedeles and Van Damme 1995). Bird habitat affiliations are diverse. Therefore, timber management practices seeking to minimize disruption of the pre-

harvest avian community would subsequently be well served by managing for post-harvest structural diversity as well.

2.3.2.2 Structural Heterogeneity

Previous work has revealed a strong relationship between the structural heterogeneity of a forest stand and avian diversity (DesGranges and Rondeau 1992, MacArthur and MacArthur 1961, Noon *et al.* 1979, Shugart and James 1973). MacArthur and MacArthur (1961) suggest that avian diversity and structural heterogeneity are linearly related although there are situational limitations that effect the generality to which this notion can be applied. The assumption is that a structurally diverse stand tends to offer a broader range of habitat thereby potentially meeting the requirements of a wider variety of birds. While harvesting may alter or deplete a forest stand of certain structural attributes, various species present in the pre-harvest stand may be retained if key habitat requirements remain in place (Cody 1985, Wedeles and VanDamme 1995). Consequently, the effect of harvesting on the diversity of the post-harvest avian community may depend on the method of harvesting that is employed.

From a structural perspective, even-aged harvesting practices tend, almost invariably, to simplify forest stands. Many studies have documented an immediate decrease in species diversity following even aged practices such as clear cutting (DeGraaf *et al.* 1993, Kendeigh 1947, Titterington *et al.* 1979, Wedeles and Van Damme 1995). Kendeigh (1947) for instance, noted 27 species per 100 acres in clear-cut sites, while adjacent mature forests were found to support an average of 35 species per the same unit area. The elimination of vertical structure is accompanied by the elimination of many species such as Cape May Warblers (*Dendroica tigrina*), Blackburnian Warblers and Golden-crowned Kinglets which are ecologically tied to coniferous foliage the mature mixed-wood or coniferous forest (Crawford and Titterington 1979).

Some work has produced results suggestive of the contrary. Welsh (1987) found relatively constant species density and diversity in different aged, post-harvest mixed-wood stands. He attributed his results, however, to the fact that the areas within which

the bird community was sampled, were commercially clear-cut. Subsequently, many species associated with pre-harvest structure were retained due to the presence of non-merchantable trees and, possibly, to the presence of merchantable species of unharvestable size. Based largely on this data, Welsh suggested that few if any entire guilds would be lost from mixed-wood stands in which softwoods such as species of *Picea* sp. were commercially clear-cut. Bird species, which rely on mature conifers, however, would either decline in abundance or be eliminated entirely.

Uneven-age management systems such as selection harvesting are widely considered to be the best able to promote the kind of fine-scale structural heterogeneity required to enhance or maintain pre-harvest composition and diversity (Crawford and Titterington 1979, DesGranges and Rondeau 1992, Frank and Bjorkborn 1973, Titterington *et al.* 1979). The selective removal of mature trees from various size or age classes tends to provide more canopy layers per unit area than even-aged systems, resulting in what has been described as, 'a mosaic of small even aged stands' (Wedeles and Van Damme 1995). The post-harvest stand consists of residual elements of pre-harvest over-story and mid-canopy vegetation, interspersed among regenerating vegetation in newly created canopy openings. Subsequently, uneven-age harvesting systems may circumvent the elimination of pre-harvest bird species linked to mid-canopy and over-story vegetation, while introducing habitat for species associated with earlier stages of forest succession (Titterington *et al.* 1979, Wedeles and Van Damme 1995).

Several authors have speculated on the response of boreal bird communities to selection harvesting. Welsh, (1981, 1987, 1992a) and Titterington *et al.* (1979) both suggest that the presence of mature conifers in post-harvest boreal stands may be particularly important in encouraging the continued presence of Blackburnian Warblers, Cape May Warblers and Golden-crowned Kinglets among others that depend on mature conifers. These species may return in lower numbers, however, largely due to the partial depletion of mature conifers in over-story vegetation. Species dwelling in the mid-low canopy strata including species such as Magnolia Warblers, Yellow-rumped Warblers (*Dendroica coronata*), Swainson's Thrushes and Blue-headed Vireos (*Vireo solitarius*)

should also continue to find suitable habitat in the post-harvest stand. Species that may find opportunities in newly created canopy openings include White-throated Sparrows, Dark-eyed Juncos (*Junco hyemalis*) and other species associated with early succession (Welsh 1987, Crawford and Titterington 1979).

Although most authors agree that selection harvesting may be the least disruptive to the pre-harvest bird community little work has been conducted to substantiate this assumption. In southern hardwood forests, Michael and Thornburgh (1971), failed to notice any significant effect on bird life initially after thinning. In northwestern Ontario, Darveau *et al.* (1995) noticed no significant difference in pre-harvest composition after the removal of 33% of mature trees in riparian buffers dominated by *Abies balsamea*. The number of breeding pairs in this study, however, declined by 20%. Franzreb and Ohmart (1978) also noted a decline in bird abundance in thinned hardwood/coniferous mixed-woods. Work in Alpine mixed-woods by Medin, (1985) and Medin and Booth (1989) revealed an increase in ground foraging species accompanied by lower numbers of foliage and bark gleaning species.

The importance of vertical diversity was illustrated through the work of Morrisson (1981) in *Pseudotsuga menzisii* plantations. A vertical separation in habitat was observed between Wilson's Warblers (*Wilsonia pusilla*) and MacGillavray's Warblers (*Oporornis tolmiei*) in which residual deciduous trees and shrubs were abundant. MacGillavray's Warblers were observed most frequently in the shrub layer while Wilson's Warblers seemed to be most active in the upper canopy. Predictably, MacGillavray's Warbler density did not change significantly after the thinning of residual canopy vegetation. Wilson's Warbler numbers, however, decreased dramatically. Interestingly, Orange-crowned Warblers (*Vermivora celata*), being able to utilize both shrub and canopy foliage, were also unaffected by thinning.

There is, however, variability in the way that selection harvesting may be operationalized. Single tree selection tends to favor horizontal continuity among the various forest strata (Wedeles and Van Damme 1995). As this practice is repeated

throughout the management cycle, forest gaps tend to be less pronounced and may be less attractive to early successional species. Group selection can, on the other hand, diminish the continuity of vertical layering but will enhance horizontal diversity across the managed area. Subsequently, group selection may provide habitat for species that prefer more open, sparsely treed areas. Canopy dwelling species would likely be absent from these patches but would be retained in unmanaged patches. The continuum between these two approaches may produce an intermediate range of mature tree density.

2.3.2.3 Tree Density and Forest Interior Birds

The response of birds to changes in tree density can be difficult to predict for species associated with mature stands (Cody 1985a). The removal of mature trees may simply translate into a reduction of habitat for canopy dwelling species accompanied by a fairly predictable reduction in the abundance of these birds. This kind of correlation has been clearly observed in Blackburnian and Bay-breasted Warbler populations (Apfelbaum and Haney 1981) and with Pine Warblers in Pine forests (Moore 1980). Studies have, however, revealed that species present in the pre-disturbance stand can respond differently to partial timber extraction.

Evidence suggests that certain forest dwelling bird species may in fact benefit from a reduction in tree density. Szaro and Balda (1979) observed an increase in a Yellow-rumped Warbler (*Dendroica coronata*) population with a reduction in mature conifer tree density. This species reached its highest densities on manipulated plots with tree densities being only 40% of that on control sites. It was assumed that high conifer density was prohibitive to this bird's preferred foraging pattern of 'hawking' for insects in forest openings. The population density of this species was predictably low on sites with tree densities only 10% that of control sites, implying that considerable timber extraction is required to completely eliminate this species from a forest stand.

Yet this threshold can be quite low for certain species. Szaro and Blada (1979) examined the response of birds to various degrees of thinning in pine forests. Most species inhabited all sites, although their densities tended to vary. The Red-faced Warbler,

25

however, was found occupying control sites but was absent from all thinned areas. Only 20% of mature trees were removed from manipulated plots subject to the lightest thinning, implying very specific density requirements for this species.

2.3.2.4 Species Interactions

Harvesting may produce indirect effects on the pre-harvest bird community, by altering species interactions. Most notably, the creation of edge habitat by logging may create conditions that favor brood parasitism by Brown-headed Cowbirds (*Molothurus ater*) as well as nest predation by other bird species (notably by species in the Corvid family) as well as terrestrial mammals Red Fox (*Vulpes vulpes*); Raccoons (*Procyon lotor*), Skunks (*Mephitis mephitis*) (Brittingham and Temple 1983, DesGrange and Rondeau 1992, Wedeles and Van Damme 1995). Thompson *et al.* (1993) suggest that bird species associated with the forest interior may be most susceptible to these factors. Wilcove (1985) found, however, that the incidence of brood parasitism and nest predation tends to decrease with distance from the forest edge. Yahner and Scott (1988), made similar findings using artificial nests in mature stands adjacent to clear-cut sites.

Little work has been conducted regarding the reproductive success of forest interior bird species in stands subject to uneven-aged management. Thompson *et al.* (1993) warn that selection harvesting may be a strong factor in selectively cut stands because the numerous canopy openings created by single tree removal may create more 'edge' per unit area than other methods of harvesting. Subsequently, the continued presence of pre-harvest species in stands subject to selection harvesting, may be undermined by poor reproductive success in the early years of post-harvest forest succession.

2.4 Timber Harvesting and the Aquatic Environment

Studies that have addressed the impacts of forestry on fish resources and other components of aquatic ecosystems date as far back as the 1930's. Since then, the logging industry has received considerable scrutiny pertaining to this facet of forest management. The potential for harvesting to affect aquatic ecosystems varies widely from region to region (Department of Fisheries and Oceans (DFO) 1992). This depends on a variety of

biophysical parameters including susceptibility to erosion and sedimentation, changes in temperature and stream velocity, the input of debris associated with logging operations and the ability of aquatic wildlife to tolerate changes in the aquatic environment.

2.4.1 Erosion and Sedimentation

Erosion and sedimentation occur naturally in most forested watersheds, however, commercial logging has been shown to accelerate this process in many areas to the detriment of aquatic habitat (Heatherington 1987, Newbold *et al.* 1980, Swanston 1971). Subsequently, mitigative efforts, such as riparian buffers, have been implemented to offset the potential impact of upland harvesting on the aquatic resource. Most information addressing this facet of forest management, however, has emanated from the western cordillera where erosion hazards are high as a result of steep terrain and high precipitation. In this region, logging activities are felt to have sharply impacted local fisheries (Hartman and Scrivener 1990, Swanston 1971). Relatively little information is available regarding the relationship between tree removal and erosion and its impact on aquatic ecosystems in other regions including the boreal forest. This is largely because the factors that have contributed to the problem in mountainous environments are not felt to be as active in the boreal region (Manitoba Model Forest 1995).

The potential of soil erosion to adversely effect aquatic ecosystems as a result of timber harvesting, varies widely both between and within different regions (DFO 1992). That is to say, erosion is largely a function of local or regional edaphic conditions which may not yield the same risks to aquatic habitat in all managed landscapes. The magnitude of erosion is largely a direct result of the interplay between slope and soil properties (Swanston 1971). Conditions that favor stability include a combination of gentle terrain and a favorable soil mixture of silt and clay, which together act as powerful binding agents (Anderson *et al.* 1976). Conversely, situations that tend to promote erosion include steep terrain and a shallow soil composed of unstable aggregates.

Yet the potential for erosion is generally not realized while the dominant vegetation layer remains intact. Even in forested landscapes where the potential for erosion is high, soils

27

anchored by tree roots may occupy terrain that might otherwise be prohibitively steep (Anderson 1971). In this respect, living root material acts as an important natural soil stabilizer. Tree mortality, either caused by logging or fire, and the subsequent disintegration of the roots of harvested individuals can trigger an erosion event, the nature of which depends largely on the site. 'Mass movements' are an extreme form of erosion typical of mountainous regions in which the soil layer of a large portion of a mountainside may detach from the bedrock substrate and gather at the foot of the slope. (Everest *et al.* 1987). In mountainous terrain, landslides, which are a type of mass movement, are 2-9 times more likely in a clear-cut area than on an undisturbed forest (Hammond 1992). Although smaller in scale, similar events can occur in boreal forest where terrain is sufficiently steep. Lengelle (1976) studied erosion in the Swan hills of Alberta as a consequence of logging that had occurred in the late 1950's. Logging had allowed the thin layer of soil to slide off the underlying bedrock, which had led to a serious erosion problem. An estimated 68,180 to 145,000 kg/ha of soil were being eroded on a yearly basis due to previous harvesting.

With the exception of the large soil movements described in Lengelle's work, logging is not commonly felt to be an important source of erosion or sedimentation in the boreal forest (Manitoba Model Forest 1995). The subtle topography and hummocky soil tends to reduce or intercept potential runoff. Further, riparian buffer strips are usually implemented to suppress the entry of sediment where upland erosion has occurred. Harvesting in riparian areas may contribute to sedimentation where the soil organic layer has been disturbed and the mineral layer is exposed to the erosive action of wind and rain (Hammond 1992, DFO 1992). Where slope is sufficient, this usually results in a surface movement of soil material. On high relief but stable terrain, ephemeral channels and gullies that had drained the cut-over prior to harvesting, will continue to do so afterward and may also contribute to increased sediment loads where runoff is substantial. This effect may be short lived, as deByle (1985) notes, where disturbed areas are quickly re-stabilized by colonizing vegetation. Hammond (1992) suggests, however, that natural regeneration can control any erosion that might occur but if the site is actively eroding, revegetation may be delayed thereby prolonging erosion.

Concern for fish stocks and aquatic habitat pertaining to erosion in the boreal region are most closely tied to factors other than the physical removal of timber. Roads and road crossings have been identified as the largest single source of erosion and sedimentation in the boreal forest of the three Canadian prairie provinces (Westworth *et al.* 1992). According to Anderson *et al.* (1982), as much as 90 percent of the erosion on forested lands in North America has been attributed to roads and stream crossings. Similarly, work by Megahan and Kidd (1972) revealed that sediment production from roads was 220 times greater on disturbed lands within managed forests in Idaho. Stream crossings in particular constitute a point source for the introduction of sediment into active channels.

Siltation generally results in a degradation of microhabitat for benthic organisms and can be particularly critical for developing fish. For species that spawn in gravel beds, or on sites that are comparatively coarse, siltation can decrease the survival of eggs and fry by reducing substrate permeability to water flow (DFO 1992, Phillips, 1971). The restriction of flowing water in the micro-habitat of the developing fish may reduce local oxygen concentrations and will prevent the removal of toxic metabolic waste material. Experiments by Hall and Lantz (1969) had also shown that heavy siltation may also present a physical barrier to alevins, causing them to be trapped within the sediment and to eventually starve. Sedimentation in river and stream environments has been of particular concern because many commercial and sport fish will swim out of lakes (or oceans) and into associated rivers to spawn.

The potential for sedimentation to adversely effect fish habitat depends largely on the tolerance of any given fish species (DFO 1992). Tolerance thresholds among the various species are poorly understood and most available information deals with fish species that are endemic to mountainous regions. Many species prefer gravelly sites in riverine environments in which to spawn. It is known, however, that certain species will tolerate other substrates if preferred habitat is not available. Sauger and Walleye, two species endemic to boreal forest lakes and rivers, will shed their eggs into the crevices of small rocks and boulders (Johnson 1961). However if gravelly habitat is not available, these

species will spawn on other bottom types including fine sand and/or muck. Egg survival, however, will vary depending on bottom type. Johnson (1961) demonstrated that egg survival for these species was poorest in a soft muck/detritus substrate and highest on a clean gravel bottom.

2.4.2 Turbidity

Finer particles that do not immediately settle on the bottom of a lake or stream tend to remain in suspension. The result is an increase in water turbidity which is a measure of the degree to which light is scattered or absorbed by suspended particles (DFO 1992, Lloyd 1987). An increase in water turbidity can effect aquatic organisms in variety of ways. The most inherently obvious effect is a reduction in available light which reduces primary productivity by reducing the depth to which photosynthesis can occur. Changes in primary productivity may be felt throughout aquatic food chains and , as a consequence, effect species at higher trophic levels.

Fish can also be physically effected by the increase in turbidity that accompanies higher loads of suspended sediments. Cedarholm and Salo (1979) noted structural damage to the gills of juvenile salmonids caused by the abrasion of particles in suspension. Sediment concentrations in suspension can be heavy enough to kill fish directly through gill damage, although experiments have demonstrated widely varying results pertaining to lethal levels. It was also noted that most fish required a chronic exposure to high turbidity over a period of many days before mortality was observed.

Research has suggested that fish behavior and predator-prey interactions may be effected by chronic turbidity. Cordone and Kelly (1961) noted that migrating salmonids in the southwestern United States will avoid turbid waters. Where suitable water turbidity cannot be located, migration may be aborted with obvious implications for future population recruitment. Sigler et al. (1984) also noted avoidance of unnaturally turbid waters by smolt. Avoidance of turbid water may be linked to decreased rates of predation. An increase in water turbidity is linked to poor visibility, subsequently effecting the ability of predators to locate prey items. The larvae of Striped Bass (*Morone saxatilis*)

were observed to consume over 40 percent fewer prey on average in waters with 200 and 500 mg/l suspended sediment concentration, than in waters with concentrations of 0-75 mg/l (Breitburg 1988). Poor visibility may also effect semi-aquatic wildlife such as mergansers and otter that hunt in the water.

2.4.3 Stream Hydrology

For a number of reasons, logging can yield a temporary increase in water yield and stream velocity (Rothacher 1971, DFO 1992). There are two major reasons for this that are identified in the literature. First, higher water yields is related to increased runoff due to the elimination of vegetation that might otherwise intercept water, to later be released through evapotranspiration. Secondly, the compaction of soil by harvesting machinery may reduce the porosity of soil, making it less permeable to water. The magnitude of the changes in stream-flow are, however, influenced by annual patterns in precipitation, local soil attributes as well as the species of tree that is harvested.

Intuitively, increased water levels in lakes and streams might be interpreted as a benefit to fish species, however, there are various negative impacts which are associated with the effect of increased water yield. Excessive current may create a hydraulic barrier to fish movement if water velocities overwhelm or prematurely exhaust migrating fish. Excessive water velocities have been noted in culverts which have been installed to facilitate drainage under forest roads. In response, research was conducted in Manitoba to establish critical (maximum) and intermediate (prolonged) swimming capabilities for fish species occurring within managed boreal forest watersheds (Fisheries Branch 1984, Katopodis 1991). It has been suggested that this work, combined with the more exhaustive work of Belford and Gould (1989) should therefore be useful in establishing a maximum limit culvert water velocity to help facilitate the passage of migrating fish.

Griffiths and Walton (1978) describe additional effects of excessive stream velocity on fish and fish habitat. The erosion of streambanks is a natural process, particularly where streambanks are composed of fluvial soils high in silt and clay content. This process may however become accelerated by the chronic exposure to high water velocities thereby

31

increasing the rate of sedimentation as well as turbidity. The effects of soil erosion and sedimentation have been noted previously but should also be taken in the context of the additive effects of potential upland erosion. High water speeds may also encourage the alteration of streambed micro-habitat through the shifting of natural sediment and in extreme cases, may dislodge benthic organisms and fish eggs deposited during the spring migration (DFO 1992).

Most information pertaining to the influence of logging on water yields has been obtained in the environment of the western cordillera of North America and may not be applicable in the context of the boreal forest (DFO 1992). Differences in climate, topography and soil are too vast to allow for meaningful extrapolation. The effect of flooding events that are driven by logging tend to be intensified by the relief associated with mountainous regions. Neither do climatic patterns in the boreal forest normally yield as much precipitation as is typical of the western cordillera. Spring melt-water in boreal watersheds accounts for a considerable quantity of water that enters boreal watersheds, however the water associated with spring flooding events is not typical water flows throughout the growing season. The manner in which the floristic attributes of a watershed can influence increased water yields, however, are more broadly applicable.

The presence of a band of vegetation between the logged area and an associated water system can effect the way increased water yields are transmitted into streams (DFO 1992). Peat bogs and fens which are common in boreal regions act as flood reservoirs due to the capacity of sphagnum moss to store water. If flooding is chronic and severe, the capacity of these areas to offset the impact of increased water yield may be short lived. Evapotranspiration in intermittent marshes and wetlands may dissipate increased runoff through evaporation (DFO 1992, Roulet 1987), however the effect may not be substantial unless higher water levels increase the cover of aquatic and semi-aquatic vegetation. These effects are not thought to completely offset the effect of timber extraction but may serve to dissipate increased water yields to varying degrees. Most likely, presence of well developed shoreline vegetation may act to reduce water velocity along the shoreline. This may pre-empt the erosion of the streambank by fast moving water.

The harvesting of softwood species may potentially yield greater runoff than hardwoods (Gifford *et al.* 1983). This may seem counter-intuitive given the generally larger leaf area index characteristic of deciduous species. At temperate latitudes, however, coniferous trees have a considerably longer growing season than do deciduous species. Most conifers will achieve optimal rates of photosynthesis at lower temperatures than deciduous species, subsequently allowing conifers to become metabolically active earlier into the spring and later into the fall. Consequently, conifers are able to sequester more soil water due to a longer growing season and may potentially release greater volumes of water through evapotranspiration. It should be noted, however, that coniferous foliage is extremely xerophytic implying an efficient use of water. As a result, the difference in water uptake and evapotranspiration between conifers and deciduous species may not be significant.

2.4.4 Logging Debris

Logging debris refers to the tops, limbs, leaves, bark and non-merchantable logs or trees that might accumulate in lake or stream ecosystems as a consequence of a logging operation (Narver 1971). The accumulation of logging debris in rivers and lakes may yield structural as well as chemical changes in fish habitat (DFO 1992, Narver 1971). Depending on the magnitude of input, changes in fish habitat associated with logging debris may yield both positive and negative effects on fish populations in forested watersheds.

The input of coarse (large branches, trees, etc.) and fine (small branches, leaves, needles, bark) woody debris is a natural process in forest ecosystems and plays an important role in providing fish habitat. Debris is a source of habitat structure both for predator and prey species seeking cover as well as a source of nutrients for lower trophic levels (Narver 1971). Arguably, the structural importance of debris for fish species may be analogous to the role of floristic structure for birds seeking suitable habitat. The extraction of timber may reduce the amount of material that may be available for aquatic structure. The accumulation of coarse and fine debris as a result of logging, however, may act to offset potential losses due to harvesting, and may improve existing habitat

structure. Hicks *et al.* (1991) note that the potential benefits to fish include a short term increase in food production and survivorship resulting from increased organic mater and improved cover. The initial benefits imparted on fish habitat by debris inputs may, however, become overwhelmed by the negative impacts associated with high debris loading.

A number of physical changes accompany excessive debris inputs which may be detrimental to fish and fish habitat. Organic debris may fill the interstices of gravel and rubble, subsequently reducing the suitability of certain sites for spawning fish, and possibly smothering eggs and developing embryos (DFO 1992, Narver 1971). In addition, coarse material may scour riverbanks and streambeds, consequently generating erosion and disrupting spawning habitat. Logging debris may impose a barrier to fish movement where high debris input has resulted in a 'log jam' (DFO 1992, Narver 1971). Log jams are perhaps the most obvious manifestation of excessive debris. Although accumulations sufficient to prevent fish movement are rare, it is a problem that has emerged in Coastal areas in Oregon and California (Holman and Evans 1964, Merrell 1951). Log jams are generally felt to be more likely where steep slopes encourage the entry of debris into associated water bodies (Narver 1971).

Excessive debris inputs may alter the chemical environment in aquatic ecosystems. The breakdown of organic debris produces high biological oxygen demand (BOD) caused by the respiration of bacteria, fungi and protozoa (Freedman 1989, Narver 1971). Further, soluble organic substances, such wood sugars which are leached from woody material exert a high chemical oxygen demand (COD). Reduced dissolved oxygen concentrations can adversely effect the swimming performance of juvenile and adult fish, may elicit avoidance reactions and halt migrations, may result in poor feeding behaviour and in extreme cases, may cause mortality (Bjornn and Reiser 1991, Davis 1975, DFO 1992,). Chemical changes in stream environments are most likely to be flushed away from input sources, particularly in streams which are fast moving. Additionally, the upstream environment may influence stream segments subject to debris input. Subsequently,

problems pertaining to water chemistry due to high debris input in rivers may tend to accumulate in lakes and wetlands.

2.4.5 Temperature

Water temperature in the freshwater environment is an important factor controlling the functions of fish and other components of freshwater ecosystems (Lantz 1971). Shoreline vegetation intercepts solar radiation, which helps to regulate overall water temperatures in forested watersheds. Timber harvesting in riparian areas may alter temperatures by increasing the exposure of the active channel to direct solar insolation (Brown 1971, DFO 1992).

The modification of water temperature in forested streams due to timber harvesting has been shown repeatedly in various forest landscapes. The most noticeable differences are generally observed in small, forested streams. Early research by Meehan *et al.* (1969) noted a maximum temperature increase of $9°F$ after logging in an Alaskan stream. Similar results were reported later by Patric (1969). Work in the Alberta 'Tri-Creeks' study area by Nip (1991) revealed an increase in July and August temperatures by $3.8°C$. Where streamside vegetation was retained in the form of a buffer strip, temperatures for the same time period increased an average of only $1.8°C$. In work by Brown and Krygier (1970) no increases in temperature attributable to logging could be detected in Oregon's Alsea Watershed Study Area where buffer strips were maintained. In the Severn Uplands situated in Ontario's boreal forest, water temperatures demonstrated a significant increase after logging (Nicolson 1975). Increased water temperature due to logging is felt to be most noticeable in small streams bordered by tall, mature or over mature forests.

Although research has shown that logging can produce higher average water temperatures, little information exists that addresses the specific impacts elevated water temperature on aquatic biota (DFO 1992). Unsuitable water temperatures may lead to disease outbreaks, changes in embryonic development and maturation in salmonids (Bjornn and Reiser 1991). This is particularly true of the boreal region. Information from other areas may be broadly applicable. Research in the Tri-Creeks area of Alberta yielded

no observable effect on the behaviour of local fish populations. Neither were there any observable effects on Coho salmon populations in a similar study by Holtby (1988) in British Columbia. Research by Newbold *et al.* (1980) revealed higher macroinvertebrate diversity and biomass in streams without buffer strips compared to streams for which protective vegetation was retained. This was attributed to higher primary production due to increased light intensity and warmer water. The implications of increased macroinvertebrate biomass for fish and other components of aquatic food chains were not quantified.

2.4.6 Natural Disturbances and Cumulative effects

Aquatic ecosystems in forested watershed may be influenced by natural disturbances in the terrestrial environment (Kimmins 1998). This is particularly true in the boreal regions where fire has historically been a frequent occurrence. In this context many of the impacts generated by logging may not be new to aquatic biota. Even moderate intensity fires will, in addition to causing tree mortality, leave mineral soil exposed and susceptible to erosion (Ehnes 1998). In addition, the elimination of streamside vegetation may produce higher water temperatures and a short term increase in water yield. It can be assumed that fire is be a source of coarse and fine debris in the aquatic environment however substantial amounts of organic matter would have been volatilized in the fire's heat.

While boreal watersheds are not new to the kinds of disturbances that may alter aquatic habitat, the cumulative effects of current forest usage may exceed the natural ability of aquatic ecosystems to absorb disturbance. The impacts that may be currently sustained by aquatic biota reflect not only those generated by logging, but those related to hydro development, mining, recreation, road construction, as well as fire (DFO 1992). Timber harvesting alone is not felt to be a significant source of aquatic habitat degradation in the boreal forest except where there has been excessive ground disturbance. Due to the number of development activities, however, the effects of logging may diffuse among the various mechanisms of habitat degradation (DFO 1992). Subsequently, the cumulative

effects of current activities in forested watersheds may potentially overwhelm the natural resilience aquatic ecosystems.

In summary, logging may impact the aquatic environment in a variety of ways including changes in the chemical and physical environment as well as through elevated water temperature. The potential for harvesting to affect fish populations varies widely from region to region depending site conditions and the tolerance of fish to the environmental changes associated with logging. Most research has emanated from mountainous regions where the effect of logging on fish populations may be severe due to edaphic conditions characteristic of the region. Fish populations in boreal regions are felt to be comparatively less susceptible to the effects logging due to typical soil and slope properties. Specific tolerance thresholds, however, are not well understood among aquatic wildlife, nor are the cumulative effects of logging and other forest uses.

2.5 Aesthetics and Forest Recreation

The continuing separation of people from wilderness, combined with the pace of urban lifestyle, is generally felt to have increased the recreational value of unmanaged landscapes (Kimmins 1997, Watson *et al* 1994). The natural aesthetic of wilderness landscapes is inarguably an integral part of the recreationists' experience. Boxall *et al* (1996), for example, found that canoeists in Nopiming Provincial Park consistently ranked landscape and scenery as the most important attributes of park environment. Considering the popularity of water recreation in many areas of the boreal landscape, riparian zones represent areas of maximum potential conflict between the logging industry and recreationists'. Recognizing this, forest managers have established protocol, where forest recreation and harvesting activities coincide, aimed at preserving the scenic asset of the landscape.

Little work, however, has been conducted to address the impact of limited harvesting on the scenic attribute of forested landscapes. The implementation of riparian and other resource buffers as a scenic barrier to upland management was invoked largely out of a perceived negative public reaction to logging, particularly in areas popular among

37

recreationists' (Boxall *et al.* 1995, 1996). The work of Boxall *et al.* (1996), in the Rocky Clearwater Forest in Alberta, for example, indicated that many users of local recreation areas did not view current forest management practices as sustainable and did not support traditional timber management practices. Considering the traditional zero-disturbance status of logging restrictions in most riparian buffers, research into the effect of limited harvesting in riparian zones may have lacked priority.

The effectiveness of riparian buffers in maintaining the streamside aesthetic was revealed in recent work by Boxall and others (1996). Surveyed recreationists in Nopiming Provincial Park in southeastern Manitoba, indicated that most canoeists had never encountered evidence of logging in backcountry canoe excursions. Furthermore, recreationists were unable to distinguish between logged areas and burned sites when shown photographic images of these kinds of forest disturbances. This indicated a general lack of familiarity among park users in general with logged landscapes. This was attributed not only to the presence and effectiveness of riparian buffers, but also to the notion that, few water recreationists seemed to venture more than 50 meters from the shoreline. This was noted by the researchers as casual observations. Subsequently little was revealed about the impact of harvesting on streamside scenery.

The known reaction of canoeist and other recreationists to burned areas may, however, provide a closer approximation of the impact of logging on the landscape aesthetic. Englin *et al.* (1995), used a discrete choice travel cost model and 2 years of canoe registrations from Nopiming Provincial Park to determine the effect of fire on the recreational welfare of park canoeists. Their results indicated a significant decrease in user welfare where canoe routes had passed through burned forest sites. Boxall *et al.* (1996), in their work within the same park, probed further into the importance of different landscapes by showing park users a series of photographic images of various forest types as well as clear-cuts and 13 year old burns. It was found that burned landscapes ranked consistently lowest in terms of preferred scenery. As previously noted, participants in this study were unable to detect difference between logged forests and those that were burned. Intuitively this would imply that clear-cut areas would be ranked similarly to

38

burned sites. The authors of the study, however did not indicate the nature of residual material within clear-cut areas. This is significant because the presence of mature, non-merchantable trees may modify the visual impact of harvested sites (Ohmann 1978).

The visual impact of timber extraction may also be inferred through the apparent preference of many backcountry recreationists for natural settings with little or no signs of human activity (Boxall *et al.* 1996, MacFarlane and Boxall 1996, Newton 1985). Preliminary research in Nopiming Park, showed that canoeists consistently ranked Park amenities such as fire pits, picnic tables and portage maintenance etc. as less important than park scenery. Further, research into forest and recreation management features indicated that park users in the Rocky Clearwater Forest would seek more primitive recreation sites if recreational features were to be expanded (MacFarlane and Boxall 1996). In addition, surveyed groups indicated that the presence of cottages detracted from their experience; with many indicating that they avoid areas where cottages were found or move through these areas as fast as possible. Another group of recreationists in the same study did not view current forest management practices as sustainable. Presumably, evidence of logging such as tree stumps and slash might significantly detract from the experience of backcountry recreationists.

Although research and general perceptions suggest that logging may have a negative impact on landscape aesthetics, research by Boxall *et al.* (1995, 96) in Nopiming Park produced results that were suggestive of the contrary. Their research was aimed at approximating the welfare gained or lost through the increase of various landscape feature including different forest types as well as logged and burned areas. Results indicated a low significant positive relationship between scenic preferences and logged areas. Although forested areas continued to provide better welfare, the results were viewed as problematic considering the negative public reaction to logging in the park as conveyed through the media. The slight positive relationship was attributed to the effect of logging on other forest amenities such as wildlife. It is known that logging may enhance habitat for certain wildlife species such as Moose (*Alces alces*) and White-tailed deer (*Odocoileus virginiana*). Subsequently, certain participants in their study may have

viewed the logged landscape in this context. It may also be suggested here, that mature residual trees and natural forest regeneration may have continued to impart scenic value to harvest sites. This information, however, was not provided in the published work.

Owing to the highly subjective nature of scenic valuation, a landscape subject to various intensities of timber management can be interpreted and valued differently depending upon the viewer. MacFarlane and Boxall (1996) address this notion through the theory of specialization. Specialization theory suggests that the participant's in an activity can be segmented based on their experience and psychological attachment to the activity. For example, a forest fire may be viewed positively by recreationists with some knowledge of fire ecology and systems renewal. Bryan (1997) noted that as specialization increases among anglers, attitudes shifted from the emphasis of catching fish, to concern for preserving stocks. Subsequently, the stigma attached to logging in forested watersheds may invoke a negative reaction among this segment of recreationists. Similarly, hunting enthusiasts may find a clear-cut or thinned forest, riparian or otherwise, an attractive landscape amenity. Conversely, Virden and Schreyer (1988) found that backpacking specialization (which can be reasonably viewed as being analogous to canoeing specialization), was positively correlated with preferences for natural settings, and negatively correlated with the presence of natural resource extraction activities.

Where harvesting has been, or may potentially be, perceived to impart a negative effect on scenery, the effect may eventually disappear through forest regeneration (Englin *et al.* 1996, Newton 1985). In addressing the temporal welfare effects of forest fires, Englin *et al.* (1996) suggested that the visual impact of a forest fire will fade over time during the course of natural forest succession. They were, however, unable to do more than speculate on linear analysis because welfare information was taken from only two points in time during early post-fire succession. Post-fire and post-logging succession may however differ (Ehnes 1998). Recovery on logged sites may be slower or may follow an entirely different successional pathway depending on the site and the pre-harvest forest type. It may be suggested here, that post-logging recovery that does not approximate pre-

harvest conditions may have implications for the scenic valuation of the affected landscape in the future.

Forest stands which are composed of species capable of quickly regenerating 'in situ' are generally easier to manage for aesthetics (Newton 1985). Harvesting in aspen stands, for instance, may only be evident for a relatively short period of time due to the ability of this species to exhibit abundant and rapid suckering (Graham *et al.* 1963, Newton 1985). Aspen and other hardwood tree species may also regenerate in gaps created by the selective removal of conifers from mixed-wood stands. Certain species, including some conifers, may exhibit poor regeneration within harvested areas. These species, and the scenic qualities they impart on a forest stand, may benefit from silvicultural assistance (Newton 1985).

Regardless of speed of recovery, or the ability of the harvested stand to re-approximate pre-harvest scenic attributes, the objective in key recreational areas is often to minimize the visual impact where forest management activities are conducted (Newton 1985, Ohmann *et al.* 1978). Perala (1977) and Ohmann *et al.* (1978) suggest, somewhat intuitively, that this is most likely to be accomplished through selective cutting or when 'attractive' or 'special interest trees' are left standing. Ohmann further states that this may encourage a diversity of age classes within the stand. In addition, the partial removal of conifers may, in certain cases, improve the aesthetic quality of the landscape when replaced by hardwood foliage. Newton (1985) emphasizes that, 'in foreground landscapes, mixed stands of aspen and conifers are probably the most visually pleasing'. Although these techniques have been widely applied in managed forests in Colorado, little or no information is available that specifically qualifies the temporal effect on public usage of managed sites.

41

Chapter 3: Methodology

3.1 Study Site

The study site was situated approximately 12 kilometers south of the community of Manigotagan, along a 4.8km segment of the Sandy River riparian buffer west of Manitoba provincial highway 304 (Figure 3.1). The buffer was established following upland harvesting in the early 1980's leaving an undisturbed zone of vegetation approximately 100m wide on either side of the river. The continuity of the buffered forest had been disrupted in certain areas by a fire that occurred in the late 1980's, and by the presence of a former river crossing situated near the center of the study area.

The study area was divided into alternating segments of experimental and control blocks. Treatments were then numbered 1-4 from west to east and labeled 'E' for experimental and 'C' for control (ie: C1, E1, C2, E2,.. etc.). A recent fire had removed most of the merchantable or otherwise mature trees on the north side of the river in blocks E1 and C1. Subsequently, sampling in these blocks was undertaken only on the south side of the river. The area covered by each study block is listed in table 3.1 below.

Table 3.1. Areal extent (in hectares) of experimental and control blocks further divided based on orientation to the active channel.

Experimental	Blk 1	Blk 2	Blk 3	Blk 4
North	-	10.08	7.56	10.68
South	5.16	10.32	6.21	18.23
Combined	5.16	20.4	13.77	28.91
Control	**Blk 1**	**Blk 2**	**Blk 3**	**Blk 4**
North	-	10.42	15.24	16.01
South	5.48	10.03	8.82	8.15
Combined	5.48	20.45	24.06	24.16

Figure 3.1. Location of the buffer management study site along a segment of the Sandy River situated west of Highway 304

43
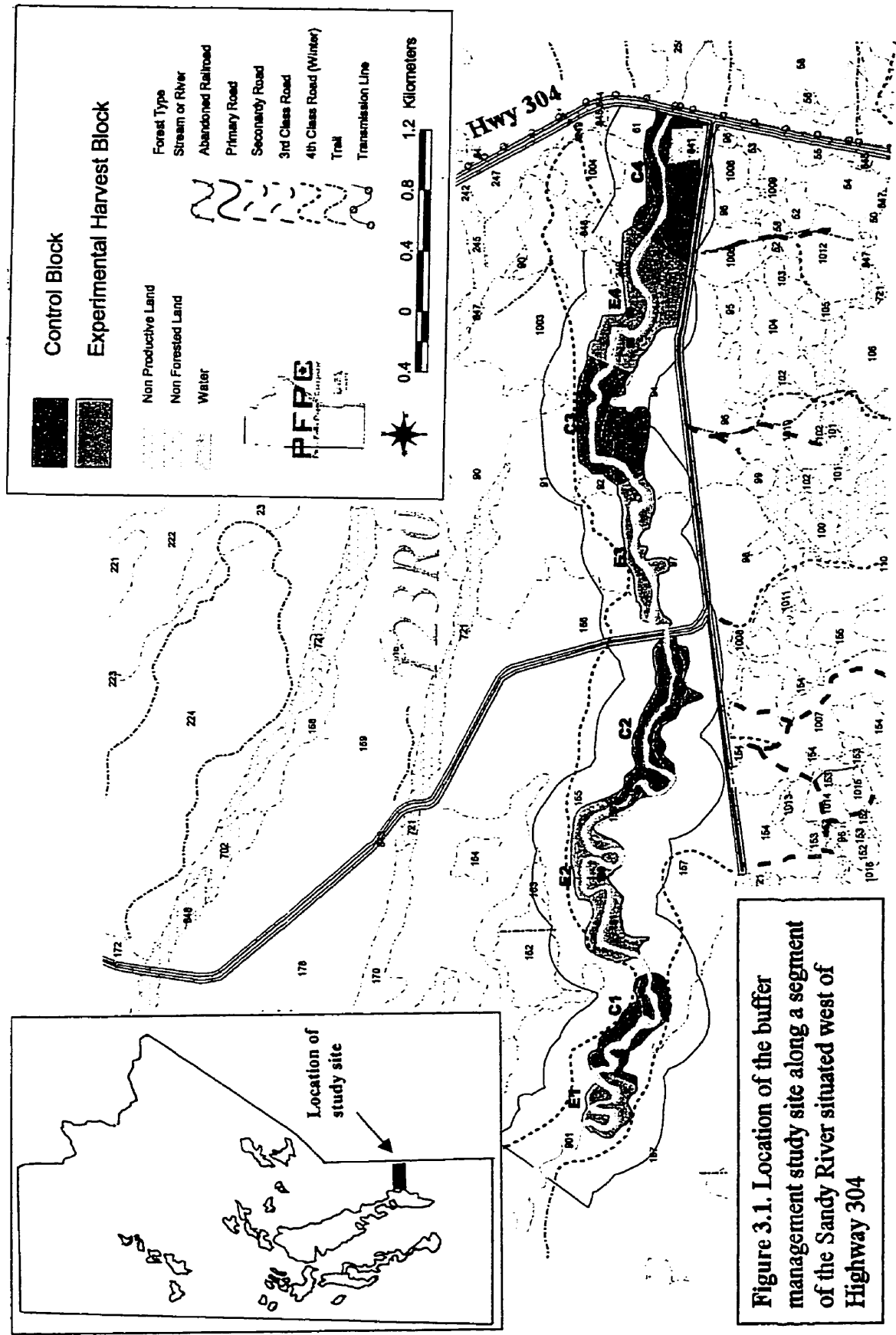
The buffer's plant community exhibited late successional mixed-wood forest attributes with discontinuous bands of *Fraxinus nigra* situated close to the riverbank. Over-story vegetation in the higher reaches of the buffer was primarily composed of *Abies balsamea* and *Picea glauca* with an admixture of *Populus tremuloides*. *Picea* sp. in this zone ranged from 76 - 124 years of age. The age of the forest as well as the abundance of large diameter P. *tremuloides* debris suggested that the community had been moving into late stages of mixed-wood forest succession. *Abies balsamea* as well as *Acer spicatum* were also observed to be important components of mid-canopy and shrub vegetation throughout much of the study area.

The dominant soil type within the study area was assessed to be a deep-moderately deep, gray luvisolic in accordance to Manitoba Forest Ecosystem Classification (FEC) (Zoladeski *et al.* 1995). The hand texturing of soil material obtained from soil pits, and exposed material at the base of wind-thrown trees, revealed that soils were predominantly clay in content. The depth of the organic layer (OL) ranged from 7-15.5cm. The 'A' horizon was determined to be primarily aggregate in structure with an approximate depth ranging between 10-15cm.

3.2 Harvest Parameters

Experimental blocks were subjected to a single-tree, selective softwood harvest which was conducted in early March 1998. Experimental blocks were split into two management zones running parallel to the shoreline. Stems found within 10m of the shoreline were harvestable based on the following criteria;

- tree DBH was = or > 25cm,
- tree exhibiting evidence of impact by disease,
- tree was leaning over the river ('leaner') with a perceived high potential for becoming river blockages in the near future

44

A machinery restriction was in place within the 10m zone. Skidder cables, however, were able to reach into this zone to remove felled timber. This 'ten-meter zone' was flagged prior to harvesting for the convenience of the harvesting crews.

Stems falling within areas beyond 10m of the shoreline were subject to a 10cm DBH cutting limitation while retaining a residual inventory of no less than 30% merchantable softwood material. To aid in achieving this, operators were instructed to leave standing, every 3rd - 4th merchantable softwood stem. Included within the target of 30% residual cover, were 2 stems/ha with a DBH of = or > 25cm. This was intended to promote age and size class diversity within the residual stand, and help maximize vertical structure.

Attention was given to the quality of merchantable stems left as residual cover. Emphasis was placed on avoiding high-grading and damage to residual material. The intent in doing so was to promote the incorporation of healthy and wind-firm stems into residual cover, and to discourage the over utilization of *Picea* sp. Residual stand damage, though difficult to avoid entirely, has been cited as an operational concern associated with various forms of selective harvesting (Wedeles and Van Damme 1995). The exposure of the vascular cambium, due to scrapes or otherwise, may act as a point of entry for plant pathogens. Subsequently, residual tree damage may compromise the value of the tree as habitat due to premature mortality, and may eventually lead to fiber losses throughout the rotation.

Contracted cutting crews were briefed, and provided with a detailed field copy of the cutting guidelines described above. Appropriate residual trees in large sections of two experimental blocks were identified and marked with paint. This was carried out to help cutting operators visualize the intended pattern and quality of residual cover.

3.3 Vegetation Measurements

Most of the vegetation data was collected with the use of fixed points established at 50 meter intervals on *both sides* of the river along a series of experimental and control transects. Each point was flagged and acted as 'plot-center' around which was

conducted. Fixed points were established in early July concurrent with ground vegetation surveys and served as the source from which all related forest vegetation data was to be obtained. While all sampling points were intended to remain in place throughout the study, it was predicted that most points established within experimental blocks would require re-flagging following the harvest.

3.3.1 Identifying the Timber Resource

To adequately determine the timber resource within the study area, a forest inventory 'cruise' was conducted. To accomplish this, both control and experimental blocks were cruised for volume, starting in August 1997 and ending in January of the following year. The actual path of the cruise lines were established by photo-interpretation and subsequent ground verification. The wedge- prism technique was used to identify harvestable timber within the allowable radius (Manitoba Department of Natural Resources 1992). Diameter breast height (DBH) and species were recorded for each tree tallied as part of the sample. Calculations were then performed to determine volumes per hectare of all tree species.

3.3.2 Floristics

Ground vegetation was sampled in July and August during both pre- and post-harvest seasons using a 0.5m x 0.5m variation of the Daubenmire quadrate. Sampling was conducted at 50m intervals roughly 30m from the waters edge on both the north and south sides of the river with the exception of blocks E1 and C1. The average percent cover of each species found growing within the quadrat was estimated using the Daubenmire Index (Daubenmire 1959) which consisted of a series of percentage intervals. Species frequency in each study block was also determined from this data.

Ideally the sampling intensity and size of the quadrat should be adapted to the characteristics of the community being sampled (Barbour *et al.* 1999,). With greater floral diversity and horizontal heterogeneity, sampling should be more rigorous, conducted either at a higher frequency or with a larger quadrat. While the buffer along the Sandy River experimental area was floristically diverse, it was also somewhat

consistent. By increasing sample size (over 300 samples taken) and utilizing a somewhat smaller sampling quadrat, trends in dominant vegetation could effectively be identified in 'pre' and 'post' management years. Furthermore, estimating percent coverage using the Daubenmire Index increases in accuracy as the quadrat decreases in size (Smith 1980).

Typically, quadrat shape may also be tailored to the need of a particular study (Barbour *et al.* 1999). For example, where vegetation is being sampled along an environmental gradient, such as that within a riparian area, a rectangular quadrat is often used oriented lengthwise along the gradient (Lindsey *et al.* 1958). Vegetation sampling in this study, however, was not aimed at capturing the floristic changes accompanied by the change in moisture availability. Rather, sampling was aimed at determining changes in the floristics of the buffer merely within the zone of merchantable material. Subsequently the use of a square quadrat was determined to be the most appropriate

3.3.3 Shrub and Tree Measurements

The line intercept technique was used to identify species and quantify the height and average cover of vegetation >1 meter in height. The line intercept method is essentially a quadrat that has been reduced to a single dimension and is useful for measuring dense vegetation (Barbour *et al.* 1999). The line has been described as, 'one edge of a vertical plane that is perpendicular to the ground. All plant canopies projecting through the line can be tallied using this technique. The line used for the present study was 2 meters long (one meter on either side of the pre-established point), oriented parallel to the direction of travel. Vegetation failing to intercept this line within one meter of plot center were not recorded. The foliage of stems engaging this line within one meter of plot center were measured in their entirety (ie: a stem with 300 horizontal centimeters of foliage was given a value of 300 regardless of how much of the foliage fell within 1m of the center point). The Line intercept data categorized into 1-2m, 2-10m and >10m strata.

To determine stand characteristics such as relative density and relative dominance of tree species occurring within the study area, the plotless 'point-quarter' method was utilized (Smith 1980). The diameter breast height (DBH), distance from center as well as species

47

were recorded for all stems recorded using this technique. Point-quarter measurements were taken for all tree species with a DBH of at least 2.5 cm. Further measurements were taken for the over-story (>9m) component. Over-story measurements were taken at 150m intervals. Forest ecosystem classification was carried out at each point along experimental and control transects using the FEC manual developed for Manitoba (Zoladeski *et al.* 1995). Procedures for classification followed the criteria described in the manual based on canopy closure using a 10m x 10m plot.

3.5 Sampling of the Avian Community

Avian survey listening stations were established throughout the study site on alternating sides of the river approximately 30m from shore. Bird data was collected using the Indice Pontual d' Abondance (IPA) technique established by Blondel *et al.* (1970) and recorded as unlimited distance point counts. Species were recorded by mapping their approximate location relative to the listening station. To avoid counting individual birds more than once, listening stations were situated approximately 250m apart (Berger pers. comm. 1997, Bibby and Burgess 1993). Plots were also situated at least 100m from the interface of control and experimental blocks to avoid the overlap of avian data between adjacent control and experimental blocks.

Surveys were conducted during the months of June and early July in both pre- and post-harvest years, according to the peak vocalization period for most breeding forest bird species. Each listening station was sampled twice within this period. Surveys were conducted in the morning one half hour prior to sunrise (approx. 0410h in June) until approximately 0900h. Surveys were not conducted during rain or during weather which may have otherwise been prohibitive to the detection of bird vocalizations.

The bird community was sampled for no longer than 10 minutes following a 1 minute period of silence to permit bird behavior to re-stabilize. Effort required to encounter new individuals and species increases with time. Subsequently, while the effectiveness of the survey may be enhanced with increasingly long listening periods, efficiency invariably diminishes. Efficiency was critical during surveys due to the degree of difficulty

experienced in accessing many of the listening stations. Further, the chance of 'double-counting' an individual bird tends to decrease as survey time is shortened. A 10 minute listening period represented a feasible trade off between survey efficiency and effectiveness for surveying birds in forested landscapes (Bibby and Burgess 1993, Berger pers. comm. 1997).

3.6 Erosion and Sedimentation

3.6.1 Upland Erosion

Upland erosion was measured, utilizing the techniques employed in the previous work of Schneider-Vieira (1996). To assess changes in ground height, erosion pins (61cm X 1.27cm diameter circular steel bars) were vertically inserted into the ground. A steel washer was initially fitted over the bar and placed flush with the ground surface. The initial distance from the washer to the top of the bar was measured and recorded. The ground height at all erosion pins was measured twice thereafter. Erosion pins were installed in transects of three pins each, arranged in a line along sloped terrain. Each pin was labeled either 'ridge', 'center' or 'bottom' according to its position along the transect. Erosion pins were installed immediately after the harvest in both control and experimental blocks with the intent of capturing spring runoff conditions. All transects were left in place throughout the spring and into the fall. The selection of plot locations fell roughly into regular intervals. Within experimental blocks, however, the specific locations of plots were chosen with the aim avoiding patches of residual or undisturbed foliage.

A series of erosion pins were installed along the naturally eroding portions of the stream bank environment in blocks E2 and C2. These areas were perceived as being somewhat more vulnerable to the erosive action of rain and runoff than logged upland sites, most which retained protective organic layers and logging debris.

3.6.2 Sedimentation

Stream-bed sedimentation was measured with sedimentation traps fashioned out of ABS tubing after the technique described by Bloesh and Burns (1980) and also used by

Schnieder-vieira (1996) within the Manitoba Model Forest. Trap dimensions were 31.75 cm (12.5 inch) in length with a 3.18 cm (1.25 inch) opening at the top using the 10:1 height:width ratio recommended by Bloesch and Burns (1980). Sediment traps were inserted into the streambed having left approximately 5cm above the substratum. Due to the inherently 'patchy' nature of sedimentation in riverine systems (Schnieder-vieira, 1996), each sampling point consisted of 2 sediment traps, the values of which were later averaged for analysis. All sediment traps were placed within 1m of the streambank.

Sedimentation was measured in both control and experimental blocks. A segment of river immediately upstream of control block C4 was incorporated into control data, within which 12 points were established. Sampling intervals along a particular segment of river was somewhat irregular, however, due to the presence of debris, rocky outcroppings, or other shoreline obstructions.

Initially sedimentation traps were to have been in position before fall freeze up to remain in place during the winter to better expose the study to the erosion potential of spring runoff. Many of the flags necessary to relocate erosion traps had been lost, most likely due to unforeseen animal activity. For this reason, sediment traps were replaced when the river ice was thin enough to break. As a result, however, the effect of the spring runoff could not be captured within the data. It was felt, however, that spring melt water may exert only a small effect on sedimentation data as the ground likely remained frozen and largely unsusceptible to erosion until after most of the spring melt water had disappeared.

3.7 Aesthetics

The aesthetic component of the study was conducted through the use of photographs. Two sets of photographs were used; 'series 1' and 'series 2' images. Series 1 images were taken with the use of fixed locations, or 'photo-stations', which were established along shorelines within experimental and control blocks in the pre-harvest year. Photographs were taken at each photo-station in the late summer of both pre- and post-harvest years. Pre-harvest and post-harvest photographs of any one photo-station were

taken at roughly the same angle in an attempt to capture the same scene in pre- and post harvest images. Series 2 images consisted of photographs taken in experimental and control blocks in the post-harvest year. The intent in doing so was to record visual impact which, by chance, may not have been captured in series 1 images. Photographs were randomly presented to a group of volunteers who were asked to rank, using a number scale from 1-10, the scenic value in each photograph. Volunteers were asked to have had interest and experience in canoeing in boreal landscapes. Poorly exposed photographs were not selected to be used for scenic evaluation. The results of Series 1 and 2 images were then analyzed separately.

Given the subjective nature of aesthetics, effort was made to reduce the effect of variables that might influence the opinion of individuals surveyed. Subsequently, photographs were taken in the same season and under roughly similar weather conditions. The pre- and post-harvest images for any particular series 1 photo-station were used if both images were of similar quality (i.e.: if one image of a series 1 photo-station was markedly over or under exposed, the photo-station was not entered into the evaluation process). Furthermore, the prime investigator presumed that knowledge of the logging aspect of the study might have introduced a bias on the part of the volunteers. This was considered to be important given the perceived negative response by the public to logging which is often conveyed through the media. Consequently, individuals surveyed were informed of the role of logging after the data had been recorded.

3.8 Data Analysis

The floristic relationships among the various study blocks in both pre- and post-harvest years were explored using correspondence analysis (CA). As an ordination technique, CA yields a lower dimensional representation of a series of points in ordination space. The closeness of points, representing study blocks, indicates the degree of floristic similarity among the various study blocks. The position of points is based on the degree of duplication or correlation in species combinations found within the sampled plant community (Barbour *et al.* 1999). In doing so, CA extracts an ordinal ranking of causal variables measured by the dispersion of points along a series of axes which are displayed

two at a time in order of decreasing influence (Kenkel, pers. comm. 1999). The investigator can then infer a possible underlying ecological gradient based on the dispersion of points along the various axes (the First axis being the most influential). Correspondence analysis is well suited to non-linear data structures (i.e.: abundance of '0' values in the data set), making this ordination technique useful for abundance or average cover data (Kenkel 1997).

Correspondence analysis was applied to interpret bird species site relationships. Line intercept data was used for site characteristics in forest structure. Bird species such as White-winged Crossbills (*Loxia leucoptera*), Evening Grosbeaks (*Coccothraustus vespertinus*), and Pine Siskins (*Cardeulis pinus*) which tended to rove in foraging flocks were not included in the analysis. Further, species such as the Canada Goose (*Branta canadensis*), Mallard duck (*Anas platyrhynchos*) and Sandhill Crane (*Grus canadensis*) were also not included in the analysis. Unlike vegetation analysis, the north and south side of each study blocks was not treated separately. As with vegetation data, pre- and post-harvest study blocks are represented as points in ordination space. The relative position of points indicates the degree of similarity among the various blocks in terms of their respective bird communities.

The Shannon-Weaver Index of Diversity was utilized in addressing the impact of harvesting on bird diversity. Diversity indices are a measure of the heterogeneity of a given community, combining richness (number of species) and evenness (number of individuals of a given species). The Shannon Weaver Index of Diversity (H) is calculated as the negative value of the sum for each species in the sample. H values are logarithmic measures of diversity in which less common species are emphasized and dominant species are de-emphasized. This may be beneficial as some less common species may have relatively more specific habitat requirements.

Paired T-tests were administered to sedimentation, erosion and aesthetics data to compare the mean values in experimental and control data. A single value (weight in grams) was used for each sedimentation point by calculating the mean weight of sediment among the

two traps in each sampling point. A paired T-test was also performed on shoreline erosion data. Upland erosion data was examined for trends for a possible net gain or loss of ground height prior to further statistical analysis. A statistical comparison of mean values among pre- and post-harvest aesthetic results were also analyzed using paired T-tests. Series 1 and 2 images were analyzed separately.

Chapter 4: Results

4.1 Harvesting Observation

Harvesting in experimental blocks commenced early in march of 1998 and was completed within the following three weeks. Tree species targeted for extraction were *Picea glauca, P. mariana* and *Abies balsamea*. Trees were felled using a feller-buncher in blocks E4, north and south. In all other harvest blocks, cutting was carried out manually with the use of chainsaws. Skidders were used in all harvest blocks to bring harvested stems to nearby wood piles situated outside of the study area. The delimbing of felled trees was conducted either at the wood pile, or through the use of 'rub trees'. The use of rub trees resulted in the removal of bark and the subsequent exposure of vascular tissue for trees used in this manner, many of which were incorporated into residual cover (Figure 4.1.)

4.2 Vegetation

A total of 306 pre-harvest vegetation plots were established. Out of this total, 166 were situated throughout experimental blocks and 140 plots in controls. A total of 305 post-harvest vegetation plots were established, with 166 situated throughout experimental blocks and 139 in controls. Vegetation plots were not established within previously disturbed areas that intersected sampling transects. These sites included river crossings, boat launches, burned sites and hunting trails.

4.2.1 Forest Ecosystem Classification

A total of ten identifiable FEC vegetation types were encountered throughout the study site (Table 4.1. and Appendix A). Vegetation types were distinguished primarily by the presence or relative abundance of *Picea glauca, Abies balsamea, Populus tremuloides* and *Fraxinus nigra* in the over-story. Classification data revealed the over-story in over 80% of FEC plots to be dominated by *Picea* sp. with varying admixture of *A. balsamea* and *P. tremuloides*. Vegetation types representing a pure hardwood or hardwood dominated over-story occurred most frequently within Blocks C3, E4 and C4 north respectively. On four occasions, categorization could not be made due to a gap in the

FEC system of classification. Plots that eluded classification were characterized by a pure *A. balsamea* over-story.

Plate 4.1. Damage to a residual stem used as a 'rub tree' during the delimbing process.

Table 4.1. Forest Ecosystem Classification (FEC) data for control (C) and experimental (E) blocks.

Treatment:	Experimental							Control						
Block:	E1s	E2s	E2n	E3s	E3n	E4s	E4n	C1s	C2s	C2n	C3s	C3n	C4s	C4n
Number of Plots (N):	10	33	35	21	20	21	23	17	26	31	17	19	15	15
FEC Vegetation Type v21	6	30	24	17	16	10	9	9	17	22	11	3	5	4
v13	4	3	14	4	1	8	8	8	7	9	4	8	6	6
v10	-	-	-	-	3	-	6	-	-	-	-	-	-	-
v9	-	-	-	-	-	2	-	-	-	-	1	-	-	-
v8	-	-	-	-	-	1	-	-	-	-	-	-	-	1
v7	-	-	-	-	-	-	-	-	-	-	-	-	-	1
v6	-	-	-	-	-	-	-	-	1	-	-	4	2	2
v5	-	-	-	-	-	-	-	-	-	-	-	2	1	-
v2	-	-	-	-	-	-	-	-	1	-	-	1	-	-
v*	-	-	-	-	-	-	-	-	-	-	1	1	1	1

* Vegetation types not described in FEC classification guide

4.2.2 The Timber Resource

4.2.2.1 Density, Relative Dominance and Volumes

Tree species (DBH > 2.5cm) that occurred on at least 11 of all experimental and control blocks in the pre-harvest season included *Picea glauca*, *Abies balsamea* and *Populus tremuloides*. *Fraxinus nigra* were encountered as trees on 4 experimental and 5 control blocks. *A. balsamea* demonstrated the highest densities (Stems/ha) in all pre-harvest experimental and control blocks but, due to relatively low basal area values, was the dominant tree in only half of all blocks (Table 4.2.). Post-harvest experimental blocks were dominated by residual tree stems. The noticeable decline in *A. balsamea* density is likely the result of gaps in the understory created by skidder trails and/or unavoidable damage during the felling of large trees. Predictably, post-harvest non-merchantable species increased in relative dominance in nearly all experimental blocks.

Similar patterns were observed within the over-story (>9m) (Table 4.2.). *Abies balsamea* demonstrated the highest densities in all pre-harvest blocks with the exception of E4 north were it was replaced by *Picea glauca* by a small margin. *P. glauca* and *A. balsamea* were the only canopy species subject to density measurements that occurred in every block. *Populus tremuloides* were recorded in every block with the exception of control block C1 south. The highest densities of deciduous trees such as *P. tremuloides* and *Fraxinus nigra* were recorded for experimental blocks E4 north and C3 north. The

relative dominance of *A. balsamea* and non-merchantable species increased after the removal of merchantable stems.

The cumulative density of *Abies balsamea* and *Picea* sp. at the >9m stratum was 97 –150 stems/ha in combined control blocks, 138 – 168 stems/ha in pre-harvest experimental blocks and 42 - 73 stems/ha in post-harvest experimental blocks. Residual stems at this stratum were within the target of 30% residual cover in all blocks. Data listed in Table 4.2. suggests, however, that the density of *Picea* sp. was disproportionately low following cutting compared to that of *A. balsamea*.

Over-story stems were distributed among 3 height classes (Table 4.3.). Most *Abies balsamea* stems occupied the 9-12 meter range in both pre-harvest and post-harvest years. Most *Picea glauca* and *Populus tremuloides* stems occurred at heights greater than 12 meters. Non-merchantable species appeared more frequent in experimental sites following harvest due to the removal of merchantable Spruce.

Timber volumes for both control and experimental blocks in the pre-harvest year are illustrated in figure 4.2. *Picea* sp.-yielded the highest volume in nearly all blocks with the exception of blocks C4 south and E4 south where *Abies balsamea* produced the highest volumes by a slight margin. The largest volumes of *Picea* sp. were tallied within the western portion of the study area comprising blocks E2 and C2. *A. balsamea* volumes were more consistent, ranging from 15 –51 m³/ha. Hardwoods produced notably smaller volumes, however tallies produced *P. tremuloides* volumes that surpassed those of *A. balsamea* in both E4 and C4 north.
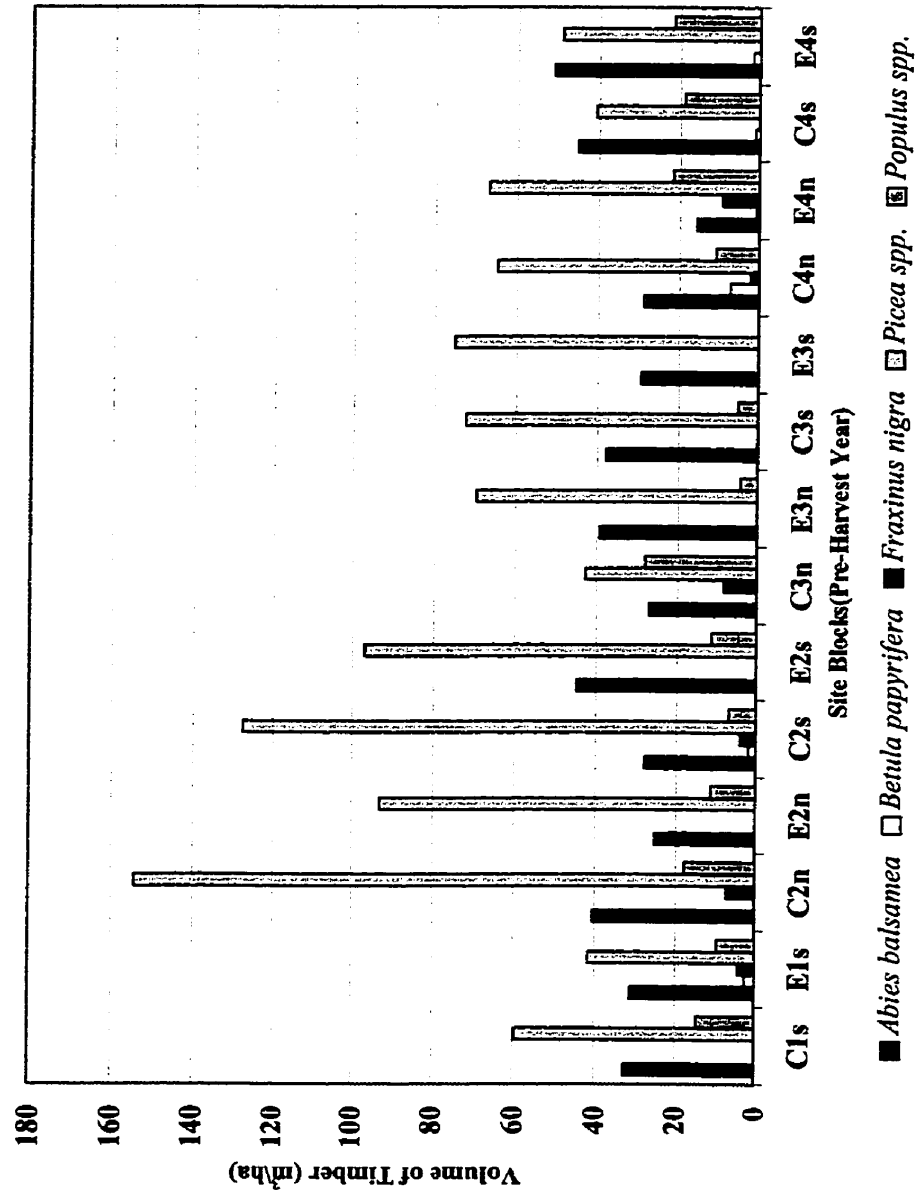
Figure 4.1. Timber volumes for five tree species measured in m³\ha in pre-harvest study blocks. Volume data represents trees with a diameter breast height (DBH) of no less than 10cm using the prism sweep technique (BAF 10). *Populus balsamifera* and *P. tremuloides* are represented collectively as *Populus* sp. while *Picea glauca* and *P. mariana* are represented by *Picea* sp.

58

Table 4.2. Density and relative dominance for tree species >2.5cm dbh and for tree species occurring in the over-story (>9m) based on point-quarter data.

Treatment:	Pre-harvest							Post-harvest							Control						
Density (Stems ≥2.5cm DBH/Ha):	E1	E2S	E2N	E3S	E3N	E4S	E4N	E1	E2S	E2N	E3S	E3N	E4S	E4N	C1	C2S	C2N	C3S	C3N	C4S	C4N
Abies Balsamea	1746	3771	2089	3917	1644	913	702	480	351	293	455	309	494	116	2261	2345	2478	1196	779	2803	1434
Betula papyrifera	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	37
Fraxinus nigra	.	66	91	.	.	101	117	.	3	32	.	6	30	53	25.4	351	37	21	103	65	.
Picea glauca	119	500	431	148	365	264	604	45	33	23	17	53	92	72	297	117	204	239	311	326	367
Picea mariana	.	.	23	99	.	40	39	.	.	.	.	.	.	.	.	.	18	.	.	.	.
Populus Balsamifera	.	.	.	.	52	.	.	.	26	6	15	18	3	8	178	58	.	21	26	65	.
Populus tremuloides	119	66	113	.	26	345	156	45	16	50	38	65	195	68	76	29	74	.	494	391	294
Ulmus americana	.	.	.	.	.	.	.	.	.	.	.	.	20	.	.	58	18	.	.	.	.
Total Stems	1984	4403	2747	4164	2087	1663	1618	570	429	404	525	451	834	317	2736	2958	2829	1477	1713	3658	2132
Relative Dominance (Basal Area):																					
Abies Balsamea	41.2	46.9	39.1	69.1	55.5	31.6	17.7	63.1	54.7	46.5	48.5	56.8	32.7	17.4	53.2	38.9	38.8	42.1	44.5	61.8	33.3
Betula papyrifera	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	0.3
Fraxinus nigra	.	0.3	0.5	.	.	0.6	11.8	.	.	8.4	.	0.2	0.9	3.1	0.3	17.3	0.1	0.1	1.0	0.2	.
Picea glauca	45.7	37.5	40.3	30.1	31.5	35.3	61.3	13.6	19.9	15.5	8.0	20.3	16.5	24.6	43.6	26.6	40.9	57.4	41.5	12.5	44.7
Picea mariana	.	4.3	.	0.8	.	4.6	5.6	.	.	.	.	.	.	.	.	.	4.8	.	.	.	.
Populus Balsamifera	.	.	.	.	8.6	.	.	.	20.6	5.3	13.7	0.5	3.1	2.7	0.1	10.5	.	0.2	0.1	3.9	.
Populus tremuloides	13.1	10.6	14.0	.	4.4	27.6	14.0	20.9	3.9	23.8	29.4	20.8	46.5	52.2	2.3	3.7	13.9	.	11.7	19.2	21.3
Ulmus americana	.	.	.	.	.	.	.	.	.	.	.	.	0.9	.	.	0.3	0.1	.	.	.	.
*Canopy Density (Stems > 9-10m/Ha):																					
Abies Balsamea	91	82	96	101	98	79	61	68	35	53	61	51	57	26	83	75	91	62	58	77	69
Betula papyrifera	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.
Fraxinus nigra	5	.	.	.	.	.	16	11	.	.	.	.	.	7	2	5	.	.	12	.	.
Larix laricina	.	.	.	.	.	.	.	.	.	:	:	1	1	1	1	1	1	1	.	.	.
Picea glauca	63	57	57	44	68	73	69	.	15	11	12	15	10	16	52	49	73	48	39	73	65
Picea mariana	.	.	.	.	.	16	8	4	.	.	.	.	.	.	.	.	.	.	.	.	.
Populus Balsamifera	.	8	5	.	5	.	.	.	12	7	.	12	.	.	.	.	.	.	.	5	.
Populus tremuloides	8	12	14	5	8	12	18	10	16	15	18	16	19	22	.	5	19	19	32	8	11
Ulmus americana	.	.	.	.	.	.	5	.	.	.	.	.	.	7	.	.	.	.	.	.	.
Total Stems	167	159	172	150	179	180	177	93	68	94	91	94	86	78	137	134	183	131	146	158	145
*Canopy Relative Dominance (Basal Area):																					
Abies Balsamea	32.7	40	30.8	47.7	36.0	41.2	17.1	59.4	46.7	53.0	72.0	48.0	56.1	26.0	53.1	33.0	34.7	37.0	29.0	41.2	43.8
Betula papyrifera	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.
Fraxinus nigra	8.3	.	.	.	.	.	23.8	11.2	.	.	.	.	.	17.2	6.0	3.1	.	.	9.1	.	2.6
Larix laricina	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	1.6	.	.	.
Picea glauca	46.0	38.9	41.1	49.1	42.9	39.0	32.0	.	15.2	9.8	11.4	17.6	12.0	8.1	41.9	50.9	51.5	48.2	33.9	39.8	32.1
Picea mariana	.	.	.	.	.	5.0	3.1	5.1	.	.	.	.	.	.	.	.	.	.	.	.	.
Populus Balsamifera	.	5.0	9.7	.	4.0	.	.	.	17.0	10.0	.	13.0	.	.	.	.	.	.	.	11.0	.
Populus tremuloides	13.01	15.7	18.5	3.7	17.0	14.7	20.6	24.3	21.4	27.0	16.6	21.3	31.7	32.4	.	14.0	13.9	12.7	17.0	18.7	21.6
Ulmus americana	.	.	.	.	.	.	3.5	.	.	.	.	.	.	16.3	.	.	.	.	.	.	.

*Canopy density and relative dominance is based on point quarter data taken at 150m interval

Table 4.3. Density of over-story species (>9 meters) distributed among three height classes (9-12m, 12-15m, and >15m) for control blocks and pre- and post-harvest experimental blocks.

Treatment:	Control			Pre-Harvest Experimental			Post-Harvest Experimental		
Height (m):	9 to 12	12 to 15	>15	9 to 12	12 to 15	>15	9 to 12	12 to 15	>15
Abies balsamea	59	13	-	76	11	-	44	6	-
Betula papyrifera	-	-	-	-	-	-	-	-	-
Fraxinus nigra	2	-	-	2	2	-	3	-	-
Larix laricina	-	2	-	-	-	-	-	-	-
Picea glauca	14	35	7	9	35	17	5	7	-
Picea mariana	-	-	-	-	3	-	-	-	-
Populus balsamifera	-	5	-	-	2	-	2	2	-
Populus tremuloides	4	5	2	3	7	1	3	13	-
Ulmus americana	-	-	-	1	-	-	1	-	-

* taken from over-story point quarter data

4.2.3 Overall Floristic Patterns

A total of 143 taxa were encountered throughout the study site in both years combined (Appendix B). Out of this total, 113 were encountered in the pre-harvest year and 128 were encountered during post-harvest sampling. Vegetation could be broken down into five functional groups; forbs (including ferns, clubmosses and horsetails), graminoids, bryophytes and lichens, shrubs, and trees. Out of the total number of species encountered for both years, there were 79 forbs, 10 graminoids, 15 bryophytes and lichens, 20 shrubs, 9 trees, 5 ferns, 3 horsetails and 1 species of clubmoss. Certain mosses were difficult to distinguish at the generic level of classification. For this reason, *Brachythecium* sp. and *Eurhynchium* sp. were linked into one sub-group among bryophytes while *Mnium* sp. and *Plagoimnium* sp. were grouped into another.

A number of taxa were widespread. Of the total number encountered in the pre-harvest year, 26 (18% of total) were encountered in at least 10% of plots, 11 (8% of total) were encountered in at least 25% of plots and 6 (4% of total) were encountered in at least 40% of plots. Sixty nine (62% of total) occurred on both experimental and control sites in the pre-harvest year. In the post-harvest year, 74 (58% of total) occurred in both treatments.

Species which consistently demonstrated high frequency and cover values throughout all pre-harvest study blocks are listed in Appendix C. At the ground-1 meter stratum, *Rubus pubescens*, *Cornus canadensis* and *Arailia nudicaulus* were among forbs ranking within

the 20 highest average cover values for each block. *Fragaria virginianum, Mertenzia paniculata, Mitella nuda* and *Linnaea borealis* fell within this range for at least 12 of the 14 blocks. *Acer spicatum* fell within this range in 9 blocks. The combined cover values of the wood/bark mosses *Brachythecium* sp. and *Eurhynchium* sp. ranked no less than the second highest of all bryophytes in all study blocks except C4 south where they were replaced by *Pleurozium schreberi* and *Hylocomium splendens* respectively. Understory at the 1-2 stratum was dominated by *Abies balsamea* and A. *spicatum* in nine of 14 blocks with each occurring individually in no less than 11. *Corylus cornuta* had the second highest overall cover among shrubs at this stratum largely due to exceptionally high values in control block C3 north. Vegetation at the 2-10 meter stratum was dominated by *A. balsamea* in all except 4 blocks. Cover values vegetation at heights greater than 10 meters were greatest among *A. balsamea, Picea glauca* and *Populus tremuloides* in most blocks. *Fraxinus nigra* had higher average cover than *A. balsamea* in Blocks C3 and E4 north. Only in blocks E4 north, C3 north and C2 south did the average cover of *P. tremuloides* exceed that of *P. glauca.*

The correspondence analysis site ordination of pre- and post-harvest control and experimental blocks is represented in Figure 4.2a. Clustering patterns in the scattergram suggested that the floristics of many study blocks differed marginally prior to cutting. The separation observed in blocks C3 and E4 north is driven mainly by deciduous shrub and tree species, suggesting a slight gradient in deciduous vegetation along the primary axis. The species exerting the most influence over the position of these two blocks included *Acer negundo, Amelanchier alnifolia, Apocynum androsaimifolium, Corylus cornuta, Ribes lacustre, Rosa acicularis, Ulmus americana, Viburnum rafinesquianum* and *V. trifolium* (Figure 4.2b).

The distribution of study blocks along the secondary axis prior to cutting appears linked to the average cover of coniferous vegetation, particularly that of *Abies balsamea*, in upper forest strata. Blocks C1 south, C2 north, E3 south, E2 north and E1 south, demonstrated the highest average cover values for *A. balsamea* while blocks E4, E3 and C4 north ranked the lowest. Much of the vegetation affecting the position of blocks C3

and E4 north occur at relatively low frequencies with the exception of *C. cornuta* at the 1-2 meter stratum. This suggests that the distribution of these blocks may reflect floristic anomalies encountered along respective sampling transects.

Correspondence analysis identified movement in all experimental blocks along the secondary axis in ordination space, with comparatively minor re-ordering along the primary axis (See figures 4.2. & 4.3.). This suggests that the floristic variability inherent to the study site was a more important variable in the distribution of study blocks in ordination space than was the initial impact of harvesting. According to frequency criteria, the most common species present in the pre-harvest community remained so in the post-harvest community. Post-harvest species composition within the various strata are listed according to frequency criteria in Appendix D. Movement of control blocks in the post-harvest year was not appreciable. Data indicated that the movement of experimental blocks was elicited by a marked reduction in *A. balsamea* and increased cover among post harvest colonizers. A slight re-ordering of experimental blocks was a result of crossovers between blocks E1 and E2 south and between E3 south and E2 north. Re-ordering of experimental blocks following harvest was influenced by the occurrence of species not present prior to harvest or had occurred at considerably lower frequencies.

Relatively few understory species exhibited a noteworthy increase or decrease in frequency following the harvest (Appendix E). A number of species demonstrated either an increase or decrease in frequency in at least three study blocks for an overall change in frequency of at least 25%. Among forbs, *Linnaea borealis, Mitella nuda, Pyrola asarifolia, P. secunda* and *Trientalis borealis*, declined the most in overall average cover. Forbs that demonstrated the largest increase in cover included exotic species *Circium arvense* and *Sonchus arvensis*. Native species that demonstrated a noticable increase in frequency included *Aster ciliolatus, Corydalis aureus, Gallium borealis, Lathyrus ochroleucus* and *Epilobium angustifolium*. Most mosses demonstrated a marked decrease in average cover with the exception of *Climacium dendroides* which increased in post-harvest experimental blocks. *Schizachne purpurescens* was the only grass to demonstrate a noteworthy decrease in frequency and average cover while *Bromus ciliatus* and

Oryzopsis asperifolia both increased. The only shrubs to decline at the ground - 1 meter stratum were *Rhamnus alnifolia* and *Viburnum edule* while five, including *Diervilla lonicera, Corylus cornuta* and *Rosa acicularis* demonstrated an increase. Tree species *Populus tremuloides* and *Betula papyrifera* both demonstrated an increase at the ground-1 meter stratum cover while *Abies balsamea* declined. *Corylus cornuta* was the only species to demonstrate an increase within forest strata higher than 1 meter. At heights greater than 1 m, decreases in average cover, particularly among softwood material, is likely attributable to mechanical disturbance during the harvesting operation.

Several species typical of disturbed or open areas had exhibited an increase in frequency average cover which was relatively small in the first year, but could be more meaningful in future years. Native species included *Achillea millefolium, Agropyron trachycaulon, Agrostis scabra, Atriplex patula, Erigeron canadensis, Geranium bicknellii, Heuchera richardsonii, Hieracium* sp., and *Thalictrum venulosum*. An exotic species worth noting was *Plantago major*

Axis 2

4N●

3N●
3S■

4S●

4N■

4S■

3N■

2N■
2N
2S■
3S●
2S● ■1S
1S

Axis 1
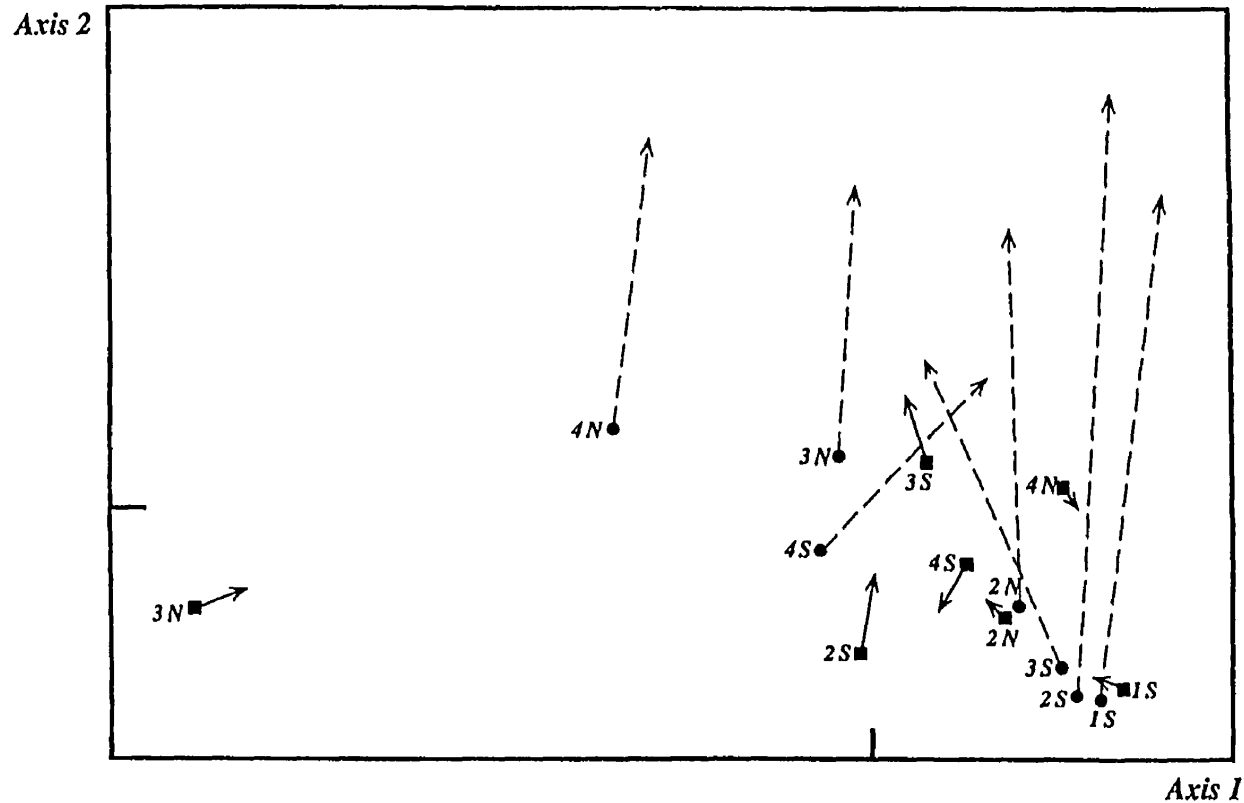
Figure 4.2a. Correspondence analysis site ordination of control and experimental blocks in the pre- and post-harvest year. Circles (●) indicates experimental cut blocks while squares (■) indicate the position of control blocks. Arrows represent the movement in ordination space of study blocks from their pre-harvest position. Dashed lines indicate the movement of experimental blocks while solid lines represent the movement of control blocks. Axis 1 & 2 indicate respectively, the First and second most important causal variables influencing floristic patterns. Hatch marks found on either axis indicate the location of the centroid in ordination space.
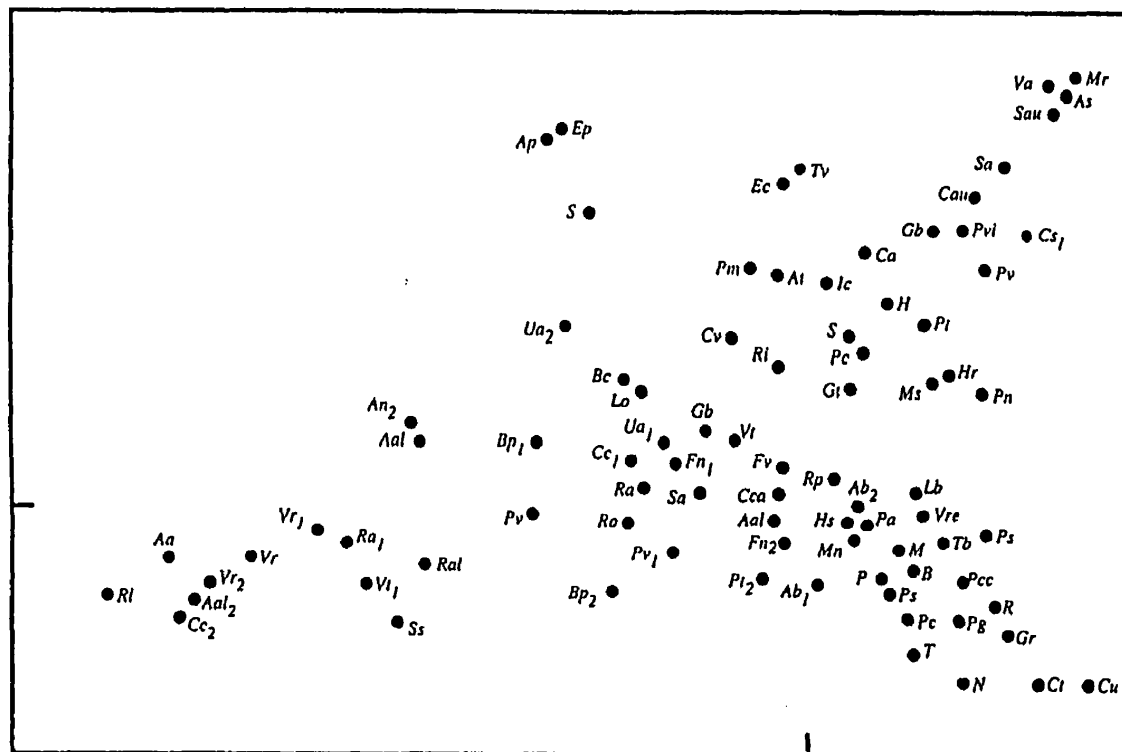
Figure 4.2b. Correspondence analysis species ordination linked to site ordination in Figure 4.1(a) Axis 1 & 2 indicate First and second most important causal variables influencing floristic patterns. Hatch marks found on either axis indicate the location of the centroid in ordination space. Plant species corresponding to acronyms seen within the chart are as follows (Subscript '1' indicates the presence of the species at the 1-2m statum while a subscript of '2' indicates the occupation of the species within the height range of 2-10m); *Aa Apocynum andorsalmifolium, Aal Amelanchier alnifolia, Ab Abies Balsamea, Ac Aquelegia canadensis, An Acer negundo, Anu Aralia nudicaulus, Ap Atriplex patula, As Agrostis scabra, At Agropyron trachycaulon, B Brachythecium/eurhynchium sp., Bc Bromus ciliolatus, Bp Betula papyrifera, Ca Circium arvense, Cau Corydalis aureus, Cc Corylus cornuta, Cca Cornus canadensis, Cs Cornus stolonifera, Ct Coptis trifolum, Cu Chimaphilla umbell, Cv Cerastium vulgatum, Ec Erigeron canadensis, Ep Equisetum pratensis, Fn Fraxinus nigra, Fv Fragaria virginiana, Gb Geranium bicknellii, H Heiracium sp., Gr Goodyera repens, Hr Heuchera richardsonii, Hs Hylocomium splendens, Ic Impatiens capensis, Lb Linnaea borealis, Lo Lathyrus ochroleuchus, Gt Gallium trifolium, M Mnium/Plagiomnium sp., Mn Mitella nuda, Mr Muhlenbergia racemosa, Ms Medicago sativa, N Neckera sp., Pa Pyrola asarifolia, Pc Polygonam cilinode, Pcc Ptillium crista-castrensis, Pch Pyrola chlorantha, Pg Pyrola grandiflorum, Pm Plantago major, Pn Potentilla norvegica, Ps Pyrola secunda, Psc Pleurozium schreberi, Pt Populus tremuloides, Pvi Prunus virginianum, Pv Petasites vitifolia, R Rhytadiadephus sp., Ra Rosa acicularis, Ral Rhanmus alnifoia, Ri Rubus ideaus, Rl Ribes lacustre, Ro Ribes oxyacanthoides, Rp Rubus pubescens, S Solidago sp., Sau Senecio aureus, Sa Sonchus arvensis, Se Scutellaria epilobiifolia, T Thuidium recognitum, Tb trientalis borealis, Tv Thalictrum venulosum, Va Viola adunca, Vr Viburnum rafinesquianum, Vre Viola renifolia, Vt Viburnum trilobum, Ua Ulmus americana*
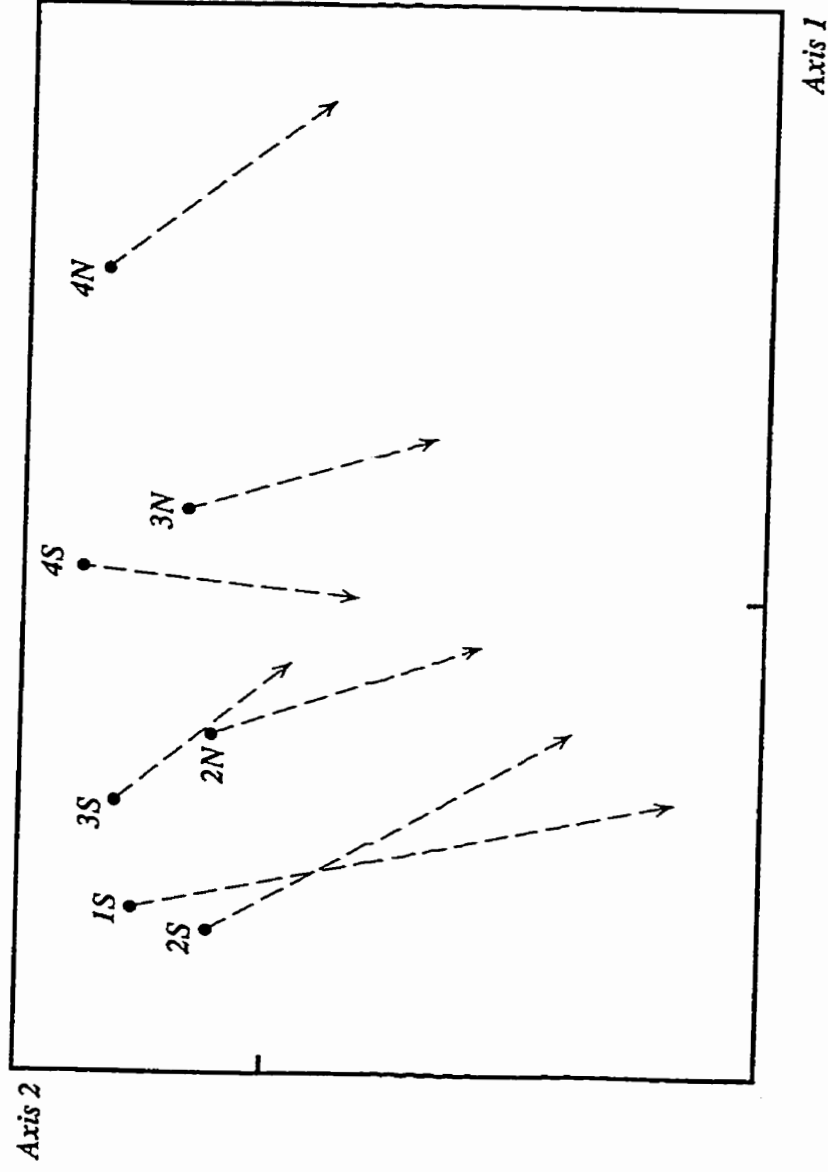
Figure 4.3a. Correspondence analysis site ordination of experimental blocks in the pre- and post-harvest year. Circles (●) indicate the pre-harvest position of experimental blocks in ordination space. Arrows represent the movement in ordination space of study blocks from their original pre-harvest position. Axis 1 & 2 indicate First and second most important causal variables influencing floristic patterns. Hatch marks found on either axis indicate the location of the centroid in ordination space.

66
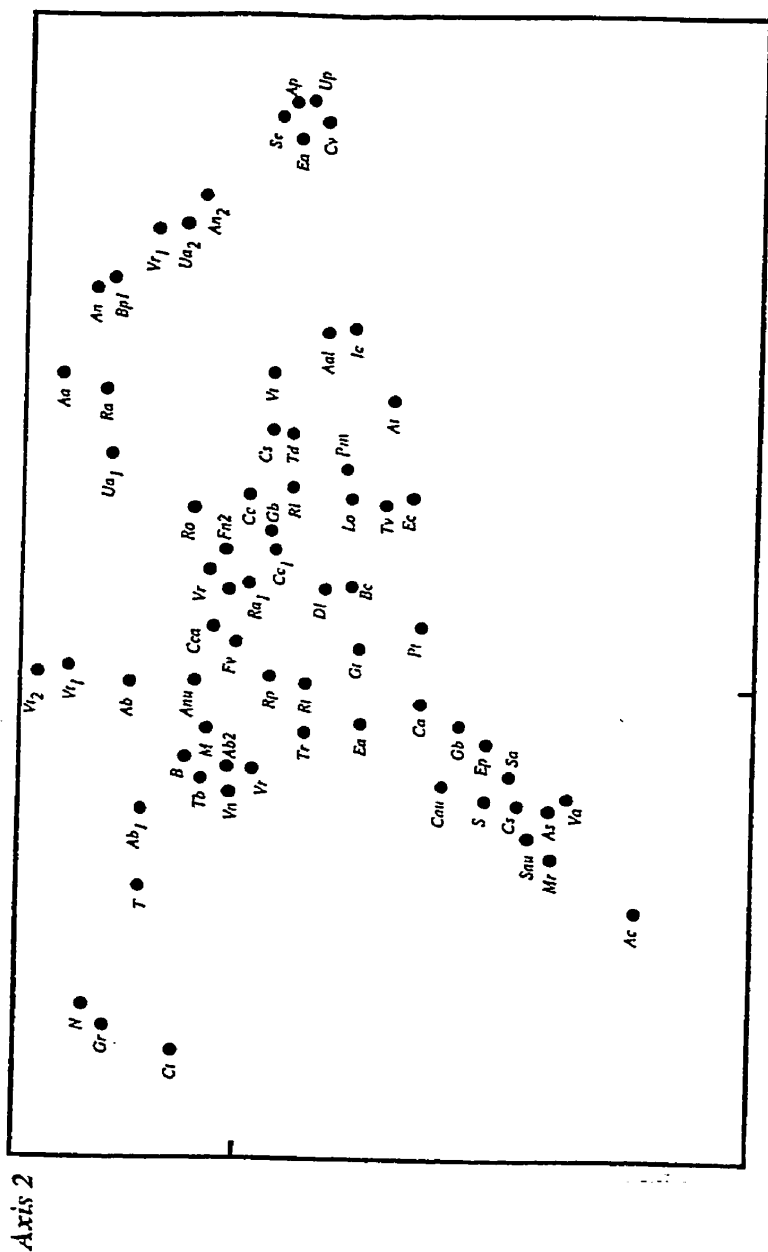
Figure 4.3b. Correspondence analysis species ordination linked to site ordination in Figure 4.2(a). Axis 1 & 2 indicate First and second most important causal variables influencing floristic patterns. Hatch marks found on either axis indicate the location of the centroid in ordination space. Plant species corresponding to acronyms seen within the chart are as follows (Subscript '1' indicates the presence of the species at the 1-2m statum while a subscript of '2' indicates the occupation of the species within the height range of 2-10m) ; *Aa Apocynum andorsaimifolium, Aal Amelancher alnifolia, Ab Abies Balsamea, Ac Aquelegia canadensis, An Acer negundo, Anu Aralia nudicaulus, Ap Atriplex patula, As Agrostis scabra, At Agropyron trachycaulon, B Brachythecium/eurhynchium sp., Bc Bromus cillolatus, Bp Betula papyrifera, Ca Circium arvense, Cau Corydalis aureus, Cc Corylus cormuta, Cca Cornus canadensis, Cs Cornus stolonifera, Ct Coptis trifolum, Cu Chimaphilla umbell, Cv Cerastlum vulgatum, Ec Erigeron canadensis, Ep Equisetum pratensis, Fn Fraxinus nigra, Fv Fragaria virginiana, Gb Geranium bicknellii, H Heiracium sp., Gr Goodyera repens, Hr Heuchera richardsonii, Hs Hylocomium splendens, Ic Impatiens capensis, Lb Linnaea borealis, Lo Lathyrus ochroleuchus, Gt Gallium trifolium, M Mnium/Plagiomnium sp., Mn Mitella nuda, Mr Mühlenbergia racemosa, Ms Medicago sativa, N Neckera sp., Pa Pyrola asarifolia, Pc Polygonam cilinode, Pcc Ptillium crista-castrensis, Pch Pyrola chlorantha, Pg Pyrola grandiflorum, Pm Plantago major, Pm Potentilla norvegica, Pr Pyrola secunda, Prc Pleurozium schreberi, Pt Populus tremuloides, Pvi Primus virginianum, Pv Petasites vitifolia, R Rhytadiadephus sp., Ra Rosa acicularis, Ral Rhamnus alnifolia, Rl Rubus idaeus, Rl Ribes lacustre, Ro Ribes oxyacantholdes, Rp Rubus pubescens, S Solidago sp., Sau Senecio aureus, Sa Sonchus arvensis, Se Scutellaria epilobifolla, T Thuidium recognitum, Tb trientalis borealis, Tv Thalictrum venulosum, Va Viola adunca, Vr Viburnum refinesquianum, Vre Viola renifolia, Vt Viburnum trilobum, Ua Ulmus americana*

4.3 The Bird Community

A total of 30 songbird listening stations were established within the study area with 15 being located in both control and experimental treatments. All listening stations were successfully sampled twice in each study season. Sampling in the pre-harvest season commenced in the first week of June and was completed in the first week of July. Post-harvest season sampling commenced in the last week of May and was also completed in the First week of July. A total of 83 species were encountered in the study area including casual observations (Appendix F).

Bird species relative abundance values for control and harvest blocks in pre-and post-harvest years are listed in Appendix G Non-forest dwelling species or those detected in transit are not listed. Species in the pre-harvest community with relative abundances of greater than 3.0 % in both harvest and control blocks included the American Redstart, Blackburnian Warbler, Magnolia Warbler, Nashville Warbler (*Vermivora ruficapilla*), Northern Parula Warbler (*Parula americana*), Pine Siskin (*Carduelis pinus*), Red-eyed Vireo, Tennessee Warbler (*Vermivora peregrina*), White throated Sparrow and White-winged Crossbill. Bay-breasted Warblers, Black and White Warblers and Evening Grosbeaks (*Coccothraustes vespertinus*) had relative abundance values of slightly more than 2.5%.

Species in post-harvest experimental blocks with the comparatively high relative abundance included the Blackburnian Warbler (6.48%), Magnolia Warbler (4.10%), Nashville Warbler (4.78%), Northern Parula Warbler (4.78%), Ovenbird (*Seiurus aurocapillus*) (4.10%), Red-eyed Vireo (6.83%) Swainson's Thrush (3.75%), White-throated Sparrow (9.56%) and the Winter Wren (3.41%).

Changes in abundance were noted among various birds species in the post-harvest year (Appendix H). The Cape May Warbler, Bay-breasted Warbler and Boreal Chickadee (*Parus hudsonicus*) were encountered fewer times in post-harvest experimental sites and more times in control sites for the same year. The Song Sparrow, Common Snipe (*Gallinago gallinago*), Winter Wren and Dark-eyed Junco were encountered more

often on post-harvest experimental blocks than in control blocks during the same year. Ovenbirds were encountered more often in control blocks in both years and were encountered less often in pre-harvest experimental blocks. Various species did not appear to exhibit a noticeable change in abundance. These included Blackburnian Warblers, Golden-crowned Kinglets, Red-eyed Vireos and Rose-breasted Grosbeaks (*Pheucticus ludovicianus*), Least Flycatchers and Nashville Warblers.

Several species were exhibited changes in abundance for post-harvest experimental sites but trends were obscured by similar changes in post-harvest control blocks (Appendix H). This strongly suggested the possibility of natural fluctuations in populations caused by factors, either not germane to the study area, or not directly related to logging. The Tennessee Warbler, White-winged Crossbill, Red-Breasted Nuthatch and Pine Siskin were encountered considerably less often on both treatments in the post-harvest year than during the previous year. To a lesser extent, this was also seen in the abundance of American Redstarts and Black and White Warblers . An opposite trend was seen in the post-harvest abundance of Canada Warblers, Yellow-bellied Sapsuckers (*Saphrypicus varius*), and American Robins (*Turdus migratorius*), which were encountered more often for both treatments. All three, however, were encountered more often on harvested sites.

The number of species encountered for both treatments for both years were; 51 and 49 in pre-harvest and post-harvest control blocks respectively and 55 and 51 in pre-harvest and post-harvest experimental blocks respectively. Differences for Shannon-Weaver Diversity Index values (*H*) for both treatments in both survey years did not exceed 0.18. (Table 4.4.). The post-harvest experimental block was the highest of the four values while the pre-harvest value for experimental blocks produced the lowest. The difference, however, was not significant ($P > 0.25$). A total of 15 species included in the analysis which occurred in the experimental blocks during the pre-harvest year did not return the following year. A total of 8 species were encountered in post-harvest experimental blocks which were not encountered in the previous year on these sites. Nine species which were encountered on control sites during the pre-harvest year did not return to these sites in the following year.

Table 4.4. Diversity values (H) derived from the Shannon-Weaver index for each year in both control and experimental blocks. Values for each treatment represents combined data from blocks within that treatment.

Treatment	Control	Exp.	Control	Exp.
Year	Pre-Harv.	Pre-Harv.	Post- Harv.	Post-Harv.
H	3.51	3.38	3.49	3.56

An ordination using canonical correspondence analysis (CCA) was performed as an aide in interpreting species-habitat relationships in the study area (Figure 4.4). Waterfowl and other species actively roving in foraging flocks during the sampling period were not included in the analysis. The distribution of species and treatment blocks relative to the First axis (Axis 1) suggested that harvesting was the most important variable affecting the bird community. The second axis (Axis 2) may have corresponded to a gradient in hardwood material within the forest over-story.

The scattergram produced by CCA (Figure 4.4) provided an indication of the response of bird species to selective harvesting. Several species appeared to associate, to varying degrees, with the position of post-harvest experimental blocks. Notably, these included the Song Sparrow, Common Snipe, American Robin, Dark-eyed Junco, Olive-sided Flycatcher (*Contopus borealis*), Yellow-rumped Warbler, and Black-backed Woodpecker (*Picoides arcticus*). The position of several other species in ordination space suggested a slight, but somewhat less obvious, affiliation with the post harvest stand. These species included the Mourning Warbler, Northern Flicker (*Colaptes auratus*), Winter Wren, Chipping Sparrow (*Spizella passerina*), Canada Warbler and the White-throated Sparrow.

The position of certain species relative to control or pre-harvest experimental blocks suggested a preference for coniferous foliage in unharvested sites. Species whose position in ordination space suggested a preference for unharvested sites, included the Bay-breasted Warbler, Black-capped Chickadee, Boreal Chickadee, Cape May Warbler, Hermit Thrush (*Catharus guttatus*), Northern Three-toed Woodpecker (*Picoides*

Axis 2
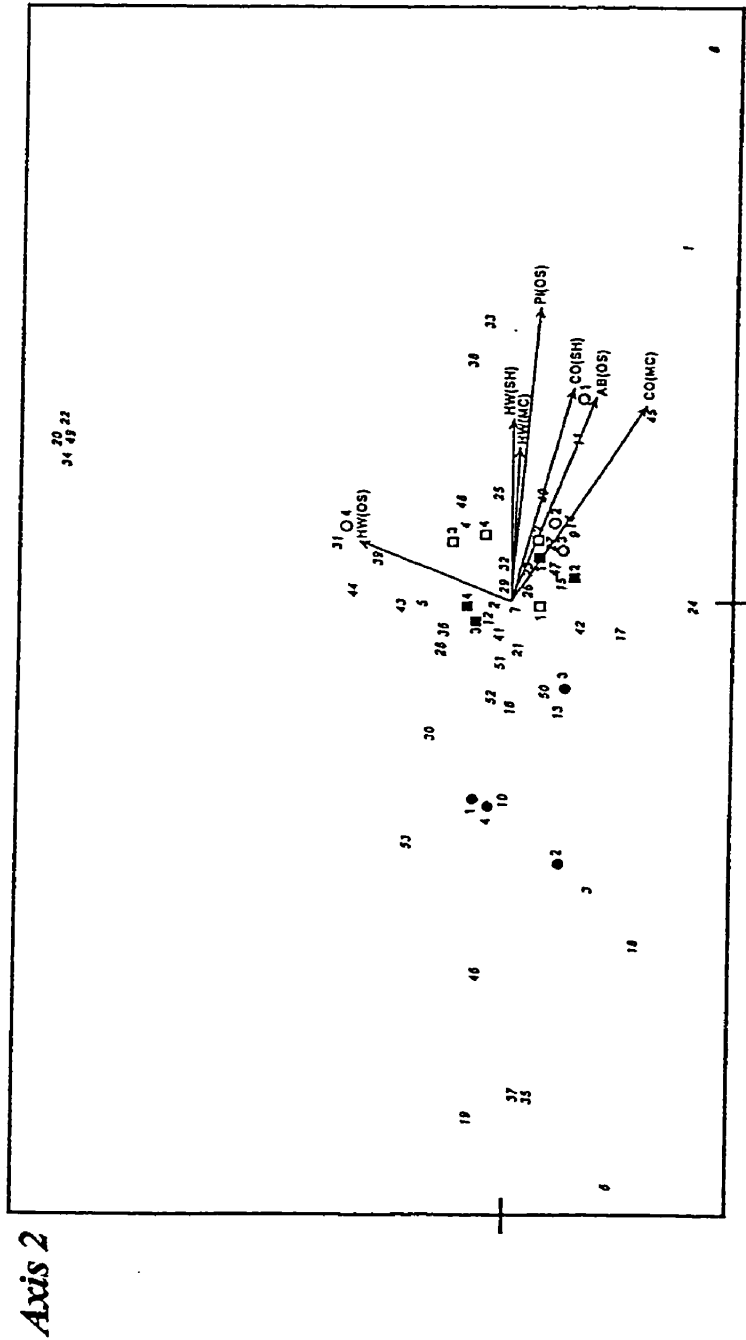
Axis 1

○ Pre-harvest experimental cut blocks □ Pre-harvest control blocks ● Post-harvest experimental cut blocks ■ Post-harvest control blocks

Figure 4.4. Positions of study blocks in relation to the abundance values of 53 bird species along the First two axes of an ordination using correspondence analysis. Habitat parameters are based on Average % cover data in appendices C & D and are described as follows; over-story (OS), mid-canopy (MC), Shrub (SH), hardwood (HW), conifer (CO), *Picea* sp. (PI) and *Abies balsamea* (AB). Shrubs vegetation falls within the 1-2 meter vertical stratum, mid-canopy within 2-10 meters and over-story being greater than 10 meters. Arrows indicate habitat vectors of increasing importance within forest structure. Axis 1 appears to reflect the distribution of birds in response to harvesting, while axis 2 suggests a gradient in deciduous material, particularly within the over-story. Bird species are denoted by italicized numbers as follows: *1.* Alder Flycatcher, *2.* American Redstart, *3.* American Robin, *4.* Bay-breasted Warbler, *5.* Black and White Warbler, *6.* Black-backed Woodpecker, *7.* Blackburnian Warbler, *8.* Black-Capped Chickadee, *9.* Black-throated Green Warbler, *10.* Blue Jay, *11.* Boreal Chickadee, *12.* Brown Creeper, *13.* Canada Warbler, *14.* Cape May Warbler, *15.* Chestnut-sided Warbler, *16.* Chipping Sparrow, *17.* Common Merganser, *18.* Common Snipe, *19.* Dark-eyed Junco, *20.* Downy Woodpecker, *21.* Eastern Wood Peewee, *22.* Eastern Phoebe, *23.* Golden-crowned Kinglet, *24.* Hairy Woodpecker, *25.* Hermit Thrush, *26.* Least Flycatcher, *27.* Magnolia Warbler, *28.* Mourning Warbler, *29.* Nashville Warbler, *30.* Northern Flicker, *31.* Northern Oriole, *32.* Northern Parula Warbler, *33.* Northern Three-toed Woodpecker, *34.* Northern Waterthrush, *35.* Olive Sided Flycatcher, *36.* Ovenbird, *37.* Philadelphia Vireo, *38.* Pileated Woodpecker, *39.* Purple Finch, *40.* Red-breasted Nuthatch, *41.* Red-eyed Vireo, *42.* Rose-breasted Grosbeak, *43.* Ruby-crowned Kinglet, *44.* Ruffed Grouse, *45.* Blue-headed Vireo, *46.* Song Sparrow, *47.* Swainson's Thrush, *48.* Tennessee Warbler, *49.* Veery Thrush, *50.* White-Throated Sparrow, *51.* Winter Wren, *52.* Yellow-Bellied Sapsucker, *53.* Yellow-rumped Warbl]

71

tridactylus), Pileated Woodpecker (*Drycopus pileatus*), Red-breasted Nuthatch (*Sitta canadensis*), Blue-headed Vireo (*Vireo solitarius*), and Tennessee Warbler.

A number of species were relatively infrequent during the two years of surveys. Several of these species appeared as outliers within the scattergram (Figure 4.5) These included four species that, following CCA, appeared to be associating strongly with hardwood over-story such as the Alder Flycatcher (*Empidonax alnorum*) Downy Woodpecker (*Picoides pubescens*), Eastern Phoebe (*Sayornis phoebe*), Northern Waterthrush (*Seiurus noveboracensis*), Veery Thrush (*Catharus fuscescens*),. Other species included the Black-capped Chickadee, Olive-sided Flycatcher, Philadelphia Vireo, (*Vireo philadelphicus*), Purple Finch (*Carpodacus purpureus*), and the Blue-headed Vireo. Black-backed Woodpeckers were encountered only twice in the study. Although certain species were encountered relatively infrequently, they were nonetheless, considered capable of yielding potentially meaningful information and were therefore included in multi-variate analysis.

4.4 Erosion and Sedimentation

4.4.1 Upland Erosion

Erosion pins were installed in April and early May following harvesting. This was necessary in order to avoid the puncturing of skidder tires and/or dislodging of erosion pins. Pins were installed in both control and experimental blocks with 30 in control and 30 in experimental. An additional 2 transects were placed in experimental blocks on skidder trails, where terrain was steep and mineral soil was exposed due to repeated use. All transects were measured twice after installation. Terrestrial measurements for harvested and control sites are summarized in Appendix I. Negative values indicate a decrease in ground height relative to the original measurement. Positive values indicate where ground height had exceeded that of the original measurement.

Measurements on harvested and control sites revealed that ground height fluctuated in both directions between measurements. On control blocks, ground measurements ranged from -0.60cm to 1.70cm, -1.20cm to 1.00cm and -1.10cm to 1.40cm for ridge, center and bottom pins respectively. On harvested sites, ground measurement ranged from -1.55cm to 1.70, -1.00cm to 1.50cm and -1.10cm to 1.40cm on ridge, center and bottom pins respectively.

These data suggest that changes in ground height are most likely attributable to the flexing of the organic layer during changes in moisture conditions. Fluctuations in ground height were usually uni-directional across all pins within a particular transect. Overall changes in ground height were extremely small. A slight increase in overall average ground height was noted for ridge and bottom pins for both control and harvest blocks. This may reflect the typically dry conditions of late spring during which many of the pins were installed. Average changes among center pins for control blocks were slightly negative. This change was relatively small for the center pins in harvest blocks (-0.01cm). The average increase in ground height measured for ridge and bottom pins was slightly more for control blocks than for harvest blocks.

73

A continual decline in ground height during both observations was noted for transects [#]31 and [#]32 in particular, as well as transect [#]14. In all three transects the organic layer had either been compromised or, in the case of transects [#]31 and [#]32, eliminated. Center pins in transect [#]31 also exhibited a continual decline however the reverse was noted for the center pins of transect [#]32. Ground height measurement for bottom pins in transect [#]32 indicated a slight accumulation. No accumulation could be inferred for transect [#]31 because the 2nd reading indicated a relatively substantial decline in ground height. This was, again, most likely attributable to changes in the thickness of the organic layer during periods of wetting and drying

Shoreline erosion pins were installed later during the month of June and only in blocks E2 and C2. Blocks E2 and C2 were selected because shoreline attributes were more amenable to assessing the movement of exposed mineral soil. A total of 22 pins were installed in block E2, while 20 pins were installed in block C2.

Table 4.5. Difference in ground height measurements (cm) for control and experimental shoreline erosion pins.

(N):	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19
Harvest:	-0.2	NA	-.25	0.00	-0.35	-0.10	-0.10	-0.05	-0.15	0.00	0.05	-0.30	0.00	-0.15	-0.20	-0.20	0.00	-0.05	0.00
Control:	-0.10	-0.25	-0.05	0.00	-0.05	-0.05	-0.10	-0.10	-0.05	-0.10	0.00	-0.20	-0.10	-0.15	0.00	0.00	-0.20	0.00	0.00

Table 4.5. continued...

(N):	20	21	22
Harvest:	-0.20	-0.05	0.00
Control:	0.00	-	-

Some pins did not yield a measurable change and were given a value of 0 (Table 4.5.). Due to disturbance caused by a River Otter or Beaver, measurements at one pin were not taken. Most measurable changes were negative indicating a decrease in soil height. A positive value was noted for one pin located in the harvest block. This would imply the movement of soil material due to the observed absence of an organic layer. The

difference in ground height were not significant (P< 0.01) however the average changes were slightly larger for harvested sites.

4.4.2 Sedimentation

Sediment traps where installed after harvesting following the break-up of river ice. The original intent was to leave sediment traps in place until the fall before 'freeze-up'. The continual decrease in water levels through out the summer, however, exposed many of the sampling points entirely or in part. As a result, many sampling points were re-installed in the active channel. A segment of Sandy River situated east of highway 304, and adjacent to the study site was utilized as an extension of control block C4. A total of 11 sampling points were installed in the extended area.

Results are presented in Figure 4.5. Average sediment was higher in experimental blocks E2 and E4 than in corresponding control blocks. Average sedimentation was highest among blocks E2 and C2, while experimental and control blocks along segments 3 and 4 produced comparatively little sediment. Control block C3 produced slightly higher sedimentation than E3. Differences in average sedimentation between treatments within
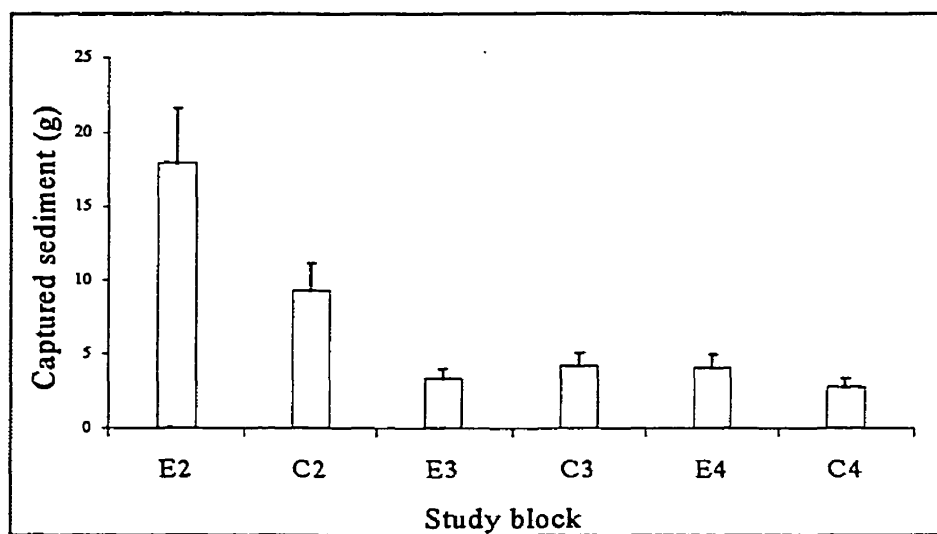
Figure 4.5. Average weight (g) of sediment captured in control and harvest blocks (N: E2=31, C2=32, E3=18, C3=19, E4=16, C4=20)

any one segment were found to be significant only for Blocks E2 and C2 (P<0.01). The exposure of sampling points, however, may have contributed to differences in the accumulation of sediment thereby rendering sedimentation results to be largely unreliable. When sampling points were removed in the fall, 9 points were found at least partially exposed in Control block C2, while 3 were found in this condition at the time of removal in experimental block E2.

4.5 Streamside Aesthetics

A total of 57 photo-stations were established throughout the study area. Of this number, 29 were taken in control blocks and 28 in experimental blocks. Photo records were taken in late summer during both field seasons in both control and harvest blocks. Pre-harvest and post-harvest photographs were matched for each photo-station. The photograph pairs that were used for analysis were selected for comparable quality (i.e.: exposure, angle of view, etc. to avoid creating a bias in panel grading). A total of 21 photo-stations were selected for analysis; 10 taken in experimental blocks and 11 in controls (Series 1). An additional 16 photographs were taken in the year following the harvest (Series 2). Out of these, 8 photographs featured harvested scenes while the remaining 8 were taken in control blocks. The intent of the 16 supplemental shots was to capture scenic attributes of harvested sites that would be seen by water recreationists, but may not have been captured by the pre-established photo-stations

A total of 18 volunteers comprised the judging panels used to grade selected photographs, and were encouraged to provide comments. Volunteers were informed that they would be shown a series of pictures of rivers in the boreal forest which they would be asked to grade on a scale of 1 – 10. The results for photographs taken at photo-stations are presented in table 4.6a. Differences among pre- and post harvest control photo-stations were not significant. Significant differences were found for pre and post-harvest values among the series 1 photo-stations situated in harvest blocks (P < 0.01). Nearly all post-harvest values were lower for each photo-station situated in harvest blocks. Only for photo-station F was the average post-harvest grade higher than the corresponding pre-

harvest value. Values of harvest photographs among the 2nd series were graded significantly lower (P< 0.01) than control photographs (Table 4.6b.). Photographs of logged areas in series 2 photographs ranged in value from 3.0 to 5.9. Average grades for photographed control sites ranged from 8.8 to 7.6. Series 2 harvest photographs yielded, almost consistently, the lowest scenic values.

Table 4.6a. Average scenic values acquired by Series 1 control and experimental scenic values in pre- and post-harvest years.

Control (N):	A	B	C	D	E	F	G	H	I	J	K	Mean d
Pre-harvest 1997:	8.3	8.4	7.9	7.8	8.2	7.2	7.4	6.8	7.6	7.1	6.3	
Post Harvest 1998:	8.1	8.0	7.0	8.5	7.9	7.4	7.6	7.2	6.4	8.0	6.8	
Difference (d)	+0.2	+0.4	+0.9	-0.6	+0.3	-0.2	-0.2	-0.4	+1.2	-0.9	-0.5	+0.02
Experimental (N):	A	B	C	D	E	F	G	H	I	J	-	Mean d
Pre-harvest 1997:	8.1	8.4	8.5	7.4	8.2	7.4	8.5	7.1	7.3	7.5	-	
Post Harvest 1998:	6.0	7.0	6.0	6.3	6.0	7.8	8.4	5.4	6.7	6.1	-	
Difference (d)	-2.1	-1.4	-2.5	-1.1	-2.2	+0.4	-0.1	-1.7	-0.6	-1.4	-	-1.27

Table 4.6b. Series 2 photographs featuring harvested and control scenic values during the post-harvest season

(N):	A	B	C	D	E	F	G	H	Mean d
Control 1998:	8.2	8.1	8.7	7.7	8.2	8.4	7.6	8.8	
Experimental 1998:	3.0	3.8	5.9	4.5	3.0	5.7	5.3	4.4	
Difference (d)	-5.2	-4.3	-2.8	-3.2	-5.2	-2.7	-2.3	-4.4	-3.76

Relatively few comments were provided by volunteers. A list of comments and the photographs to which they correspond are provided in Table 4.7. The comments that were provided implied that the scenic attributes that tended to effect grading included; floristic diversity, tree health, lushness of vegetation, stream blockages, and 'wildness'. Several volunteers suggested that the landscapes featured in some photographs had been subject to logging or some activity involving machinery. Scenic attributes that appeared to produce consistently low marks in post logging harvest blocks were: trees that appeared sick or sparsely foliated, low tree density, and signs of disturbance. Photographs of unharvested sites, yielded negative comments where poor tree health or stream blockages were noted, although these attributes did not appear to have led to substantially lower grading.

Logging did not appear to negatively impact the scenery featured in photo-stations F (Plates 4.5a &b) and G (Plates 4.6a & b) of series 1 images. Several factors may have been at play in producing these results. Comments that were linked to these photo-stations suggested that the trees featured in these images appeared healthy. It is possible that the presence of deciduous foliage may have offset the visual impact of softwood removal. Harvested areas lacking deciduous material tended to acquire lower grades by volunteers, particularly in series 2 harvest blocks. Further, *Abies balsamea* may have provided a visual substitute for harvested *Picea* sp. A comparison of pre-and post harvest images of photo-station G, reveals that nearly all Spruce stems were removed. Nearly all conifers in the post-harvest image are *A. balsamea.* The relatively minor decrease in the average value of G may also be, at least in part, due to the angle of the photograph. Impacted sites are not visible from the angle at which the photograph was taken. This may have played a role for the post-harvest values given to photo-station F as well.

Table 4.7. Summary of comments and corresponding photographs in Series 1 and Series 2 images.

Comments	Photo-stations (Series 1)			Supplemental Photographs (Series 2)	
	Pre-harvest	Post-harvest	Controls	Harvest	Control
bare banks,	.	.	.	A	.
blockage, [poor] entry point, challenge to paddle	.	A, E(3)	G(3), F	.	.
contrast in vegetation, depth of forest, nice mix	D	.	B	.	.
Dark	.	.	.	.	B, G
diverse [Uneven-aged]	J	.	D	.	.
[lot] of living trees, healthy	.	F	D	.	.
lush, etc.	D	.	G	.	B
mixed canopy [scenic], diverse canopy, etc.,	D, F(2)	E(2),F,B	B	.	C
not a forest, barren, thin, scraggily, sparse	.	A(2), D(4), J(2)	.	A,E(2),H(5), D(3), G	.
not boreal enough	.	C	.	C	.
not scenic	.	J,I,A(2)	.	H(3)B, G(3), A(2), D	.
pretty, beautiful, lush, dramatic, etc.	F, G(5), J(4),	F(3)	D, F(2), A(4), B(3), E(4)	.	A(3),B,D,E,(2)
scenic, good route, etc.	F(2), G(2)	D, F, G(3), I(2)	E(2),F, A, D(2), G	.	A(2), D, E(3),H(2),
short [Spruce]	.	.	.	A, B, E,	.
sickly trees, poor health, tree death?, etc.	H	B(2),C, A(3), D(2)	E	E(3),A(5)C, B(3), F(4), H(4), G	.
Spruce showing chlorosis			C		
spiritual [inspirational]	E	.	.	.	.
tall, Good height, towering Spruce etc.	.	I	A (2)G(4)	.	A
too much broadleaf	C	D	.	.	.
too much uniformity	.	.	C	.	A
too open [exposed]	.	.	K	.	.
trails visible, disturbed	.	.	.	A(4)	.
tree stump? Logged?	.	.	.	A(2), F	.
trees thin with patch (foliage) on top	.	.	.	F, B	.
veneer-like	.	.	.	H, D	.
Wild, looks remote	G(2)				
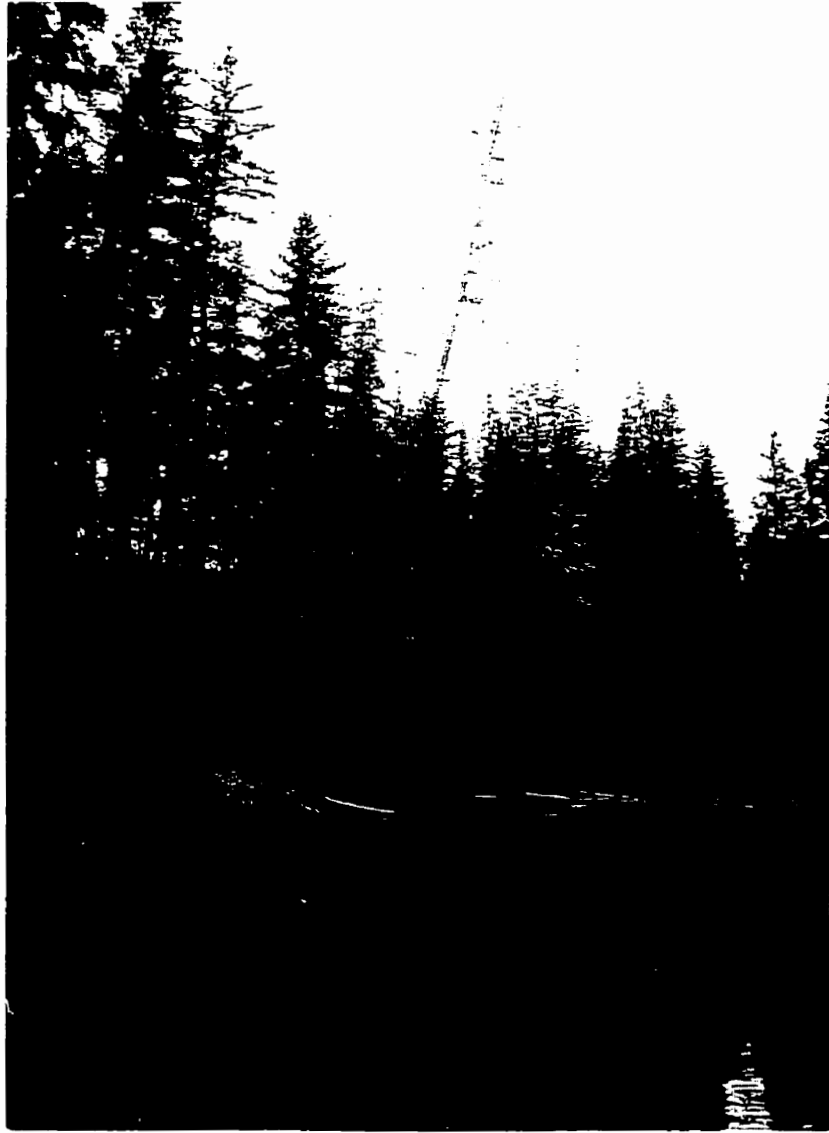
Plate 4.2a. Series 1 pre-harvest image of photo-station 'A'.

Plate 4.2b. Post-harvest image of Series 1 photo-station 'A'.

Plate 4.3a. Pre-harvest image of Series 1 photo-station D.

Plate 4.3b. Post-harvest image of Series 1 photo-station D.

Plate 4.4a. Series 1 control photo-station E. Relatively high scenic value was acquired by this image (note arboreal lichen in upper right region of image).

Plate 4.4b. Series 1 control photo-station C. Although this image received relatively high scenic value, it drew one comment that referred to potential chlorosis (note off colored tree toward the center of the image).

Plate 4.4c. Series 1 control photo-station B. This image received favorable comments pertaining to plant diversity.
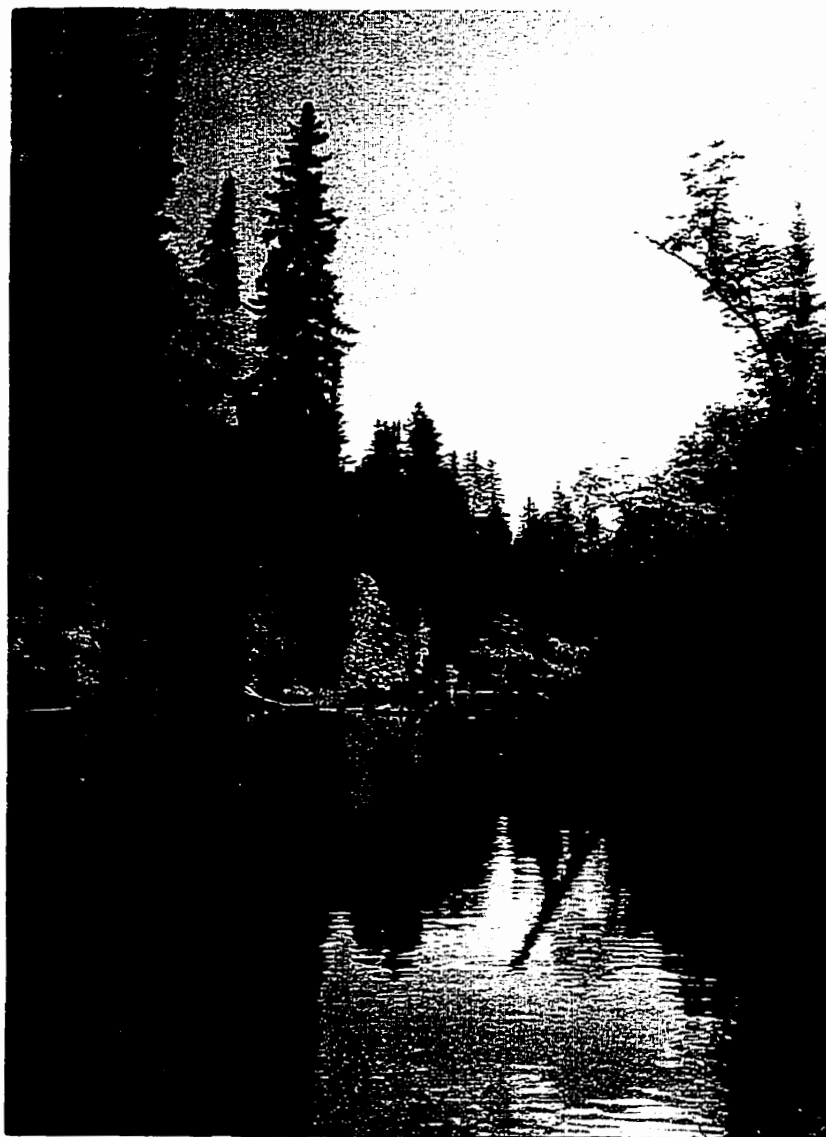
Plate 4.4d. Series 1 control photo-station G. This scene was noted on several occasions by viewers as a 'poor entry point' due to the blockage seen in the foreground.

Plate 4.4e. Series 1 post-harvest image of photo-station B. A favorable comment pertaining to canopy diversity was acquired by this image.
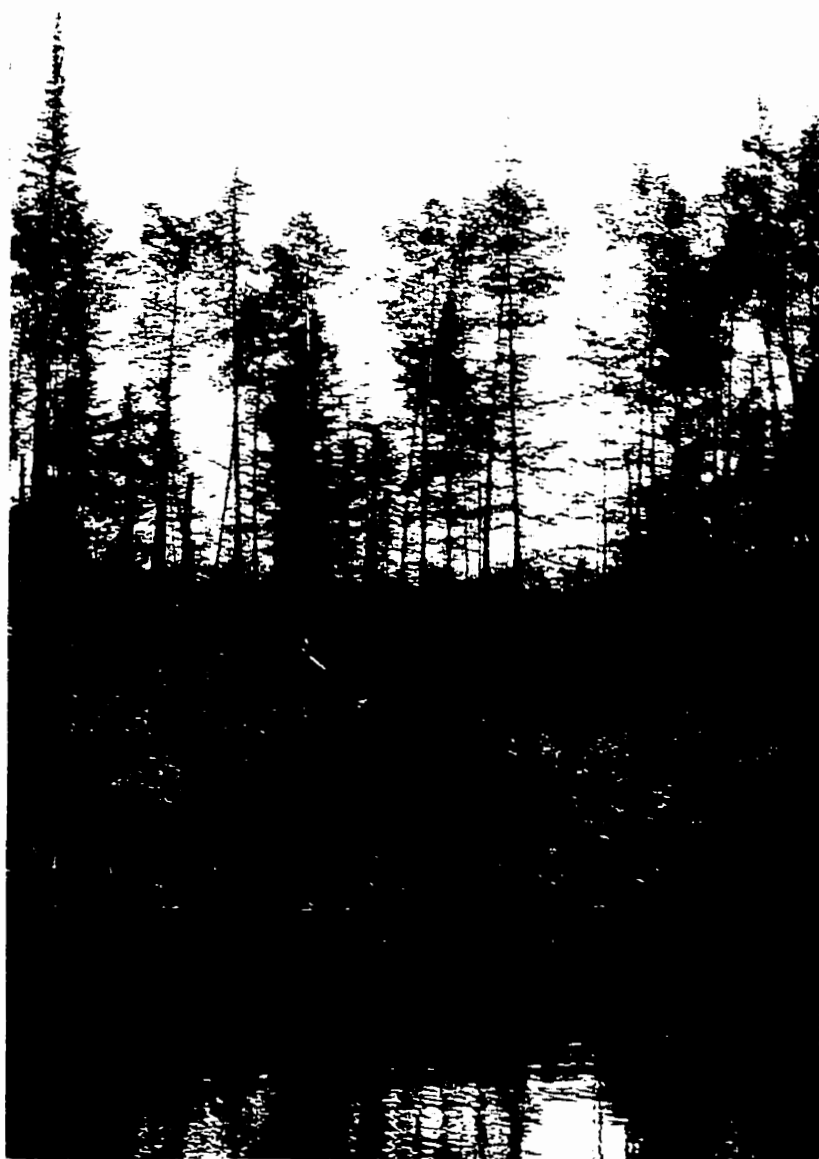
Plate 4.4f. Series 2 post-harvest image H; The forest in this image was called 'veneer-like' on one occassion and repeatedly drew comments referring to poor tree health and sparseness of trees and foliage.

Plate 4.4g. Series 2 post-harvest image A. Logging or other human activity was suspected in this image (note skidder trails, slash and small tree stump situated near the center-right of the image). The possibility of poor tree health was also noted by viewers.

Plate 4.5a Pre-harvest image of Series 1 photo-station F.

Plate 4.5b Post-harvest image of Series 1 photo-station F.

Plate 4.6a. Pre-harvest image of Series 1 photo-station G.

Plate 4.6b. Post-harvest image of Series 1 photo-station G.

Chapter 5: Discussion

5.1 The Timber Resource

The forest within the sampled region of study site exhibited floristic patterns typical of a boreal mixed hardwood\softwood community. Scott (1995) describes *Picea glauca, P. mariana, and Abies balsamea* together with an admixture of *Populus* sp. and *Betula papyrifera* as one of the 'classic' expressions of a closed mature mixedwood forest. This type of stand is often the product of successional processes on clay soils in which the hardwoods of early seral stages gradually die off and give way to shade-tolerant softwood species (Zoladeski et al. 1995). Over time this can generally lead to an increase in the importance of softwood material in the over-story, coinciding with a decrease in the importance of hardwoods.

The distribution of relative dominance among stems >9 meters in height, combined with the age of dominant stems and FEC data, suggested that the forest in much of the study area was in an advanced stage of succession. Over-story vegetation in both control and experimental blocks were dominated by either *Abies balamea* or *Picea* sp. with an admixture of *Populus* sp. Due to variations in terrain, vegetation transects would occasionally pass through areas of relatively higher moisture levels, which would account for the presence of *Fraxinus* sp. in over-story vegetation measurements. Slightly higher *P. tremuloides* stems were noted for block C3 north due in large part to the presence of a small, pure stand of this species which was encountered during sampling.

Older mixed hardwood\softwood stands, may eventually exhibit well developed softwood vegetation at the understory and mid-canopy strata. *Abies balsamea* often emerges as the dominant softwood at this stratum due to the propensity of this species to persist and develop in the shade of a softwood dominated canopy. This may result in what can be described as a 'second' forest within the forest's over-story. The high density of *A. balsamea* stems at 2.5cm dbh or more, combined with a relative dominance which was comparable to that of *Picea* sp., suggested a well developed layer of *A. balsamea* in the understory. Presumably, *Picea* sp. in the softwood component of the over-story would eventually be replaced by *A. balsamea* as the former, over time, succumb to age.

95

The increase in the relative dominance of hardwood species following cutting in experimental blocks was predictable. Shifts in the relative dominance of softwood species, however, suggested that *Abies balsamea* may have been under-utilized in favour of *Picea* sp. *Picea* sp. is a preferred softwood for harvesting purposes within the management license, presumably because harvesting and processing machinery is not suited for resinous softwood species such as *A. balsamea*. Although harvesting parameters provided that roughly every 3rd or 4th merchantable stem remain as mature residual cover, the relative proportion of mature residual stems was not distinguished by species. Failure to specify the preferred relative proportion of species to be incorporated into residual cover may have positioned *Picea* sp. to be over-utilized in view of its desirability among merchantable species within the study area. The pattern of post-harvest relative dominance values among the major softwood species in the study area may, however, also have been related to factors pertaining to the operability of the site.

The harvesting operability of a resource buffer can be limited by economic feasibility and presumably may result in the concentrated use of a preferred fiber source. The criteria used in determining the merchantability of a particular stand, or group of stands, consists of an overlay of factors including disturbance history, stand age and the potential yield per unit area. The minimum volume of timber per hectare needed to be ecomically feasable was roughly 50m^3/ha (Swanson 2000). *Picea* sp. alone achieved this minimum in all but three study blocks, one of which was a harvest block. Buffer management objectives aimed minimizing the impact to non-timber resources while circumventing the proposed short comings of zero-disturbance management may endure costs not typically associated with upland practices. Harvesting within an established buffer carries the added operational overhead associated with the mobilization of equipment for a second visit to the site. This can be exacerbated by the opportunity cost posed by residual, or 'leave' trees when the site is harvested selectively. This and the potentially limited volume of merchantable timber within the buffer, depending on its areal extent and topography, may promote the under-utilization of less desirable species to meet the efficiency needs of the operation.

Nonetheless, an inventory of *Abies balsamea* and *Picea glauca* had been maintained at heights ranging between 15m and 9m in the post harvest over-story. Although both of these species where encountered at heights that were greater than 15m in the pre- harvest canopy of experimental blocks, data does not clearly suggest that trees in this height class did not persist following harvesting. The lack of *Picea* sp. in the >15m height class in post-harvest experimental blocks may be more reflective of this species possibly being over-utilised. That is to say, *A. balsamea* with a height >9m were simply more likely to be encountered. This would explain the higher number of post-harvest *A. balsamea* stems that were greater than >9m in height relative to the pre-harvest canopy.

5.2 Vegetation Patterns

The data presented in this study suggested that natural variability within the study site had a stronger influence on post-harvest floristics than harvesting during the first year after cutting. The relatively minor reordering of harvest blocks along the primary axis using correspondence analysis suggested that key pre-harvest floristic attributes had been retained. While post-harvest experimental blocks continued to be dominated by pre-harvest vegetation, increases in frequency and average cover were noted for colonizing and certain shade suppressed species. Species typically associated with shaded or humid environments were maintained within harvest blocks but decreased in frequency and average cover.

Logging will invariably impose changes in the forest environment which may be reflected by changes in the pre-disturbance plant community. Changes in the environment of the pre-harvest plant community may favour the growth and expansion of certain plant species and may result in the decline of others. A number of factors will influence post-logging floristics including. These include the attributes of pre-harvest species, as well as changes in site availability and light intensity. The degree to which the post-harvest community resembles the pre-harvest stand is largely a function of the interplay among these factors (Ehnes 1998).

The results presented in this study appear to support previous findings that pre-harvest composition can exert a strong influence on the post-harvest community (Ehnes 1998, Cayford and MacRae 1983, Johnston and Elliot 1995). Forest communities generally retain a large proportion of pre-disturbance species shortly after logging (Payette 1992, Ehnes 1998) largely because relatively few species, with the exception of commercial species, suffer above or below ground mortality (Ehnes 1998). Logging, especially when conducted during winter months, usually has little effect on soil organic layers. Subsequently, damage to stem and other tissue in non-commercial species is usually offset by the persistence of regenerative tissue beneath the soil. Sufficient above ground tissue damage can result in the temporary removal or reduction of taller growing individuals at forest strata above that of the ground level. Reduction of *Acer spicatum*, *Prunus virginiamum* and *Abies balsamea*, for instance is mainly attributable to mechanical damage as opposed to changes in light conditions or increased competition.

The degree to which pre-harvest vegetation may contribute to the post harvest community depends on the effect of logging on species performance. The removal of over-story vegetation invariably results in an increase in the amount of light available to understory vegetation. Most boreal vascular plants persist in the forest in a shade suppressed state and will therefore readily take advantage of canopy openings and the subsequent increase in sunlight (Bell 1991, Sims *et al.* 1990, Viro 1974). Several species demonstrating better performance in this study have also been noted elsewhere in the literature as benefiting from the effects of canopy removal. The improved performances of species such as *Aster ciliolatus* and *Oryzopsis asperifolia* are well documented in logged communities (Ehnes 1998, Rowe 1983, Viro 1974). In addition, Ehnes (1998) found that species including *Agrostis hyemalis*, *Gallium boreale*, *Corylus cornuta* performed better in 13 year old cut-overs than in burned communities of the same age. The better post-harvest performance observed of species such as *Heuchera richardsonii*, *Corydalis aureus*, *Solidago sp.*, *Thalictrum sp.*, *Heiracium sp.*, *Polygonam cilinode*, *Bromus ciliolatus* and *Viola adunca*, which favor open or disturbed sites (Johnson *et al.* 1995), is somewhat predictable although not widely noted in the literature in the context of post-logging communities.

Although logging generally results in limited direct mortality of non-commercial species, logging may indirectly reduce the cover of a few species not able to tolerate the change in light conditions (Ehnes 1998, Johnston and Elliot 1995). The increased exposure to light and wind results in the desiccation of most forest bryophytes. With the exception of a few rock dwelling species, bryophytes generally prefer the humid and well shaded micro-environment associated with the forest ground stratum. The decline of nearly all mosses on harvested sites was therefore likely attributable to the increased exposure of their preferred microenvironment. The decline in average cover of flowering species such as *Mitella nuda, Trientalis borealis* and *Pyrola secunda* was expected because they are typically associated with a mature closed canopy (Ehnes 1998, Rowe 1983). The decline of these species may have been attributable either to an inability to tolerate high light intensities, or to exclusion by species better able to compete in improved light conditions.

The persistence of shade dwelling species such as *Mitella nuda, Trientalis borealis*, and mosses such as *Brachythecium* sp., *Hylocomium splendens*, and *Pleurozium schreberi* in harvested blocks, is largely due to the abundance of residual foliage. In post-harvest communities, residual shrubs and saplings can act as a canopy for ground layers by intercepting a portion of incoming radiation thus maintaining habitat for shade dwelling species. This effect can be enhanced when elements of the over-story remain intact. Residual conifers which are less permeable to light (Barbour *et al.* 1999), may be particularly effective in providing shaded habitat for mosses and species likely to be out competed in better light conditions. In this respect, residual cover in logged areas may help maintain post-harvest floristic diversity.

Relatively few new species were encountered within harvested blocks, and most acquired little coverage. This may be in part due to lack of seed source but also tends to reflect the notion that logging does not typically promote the colonization of many species not present in the pre-harvest community (Johnston and Elliot 1995, Rowe 1983). The rapid regeneration of shade released species may have been prohibitive to the establishment of new species (Ehnes 1998, Viro 1974). Average cover of vascular cover in most ground

quadrats, however, rarely approached 100% implying that competition may not have been a strong factor in limiting the establishment of new species. Many species may have been prevented from establishing due to limited number and variety of available sites for colonization. The post-logging environment may provide available sites for certain species merely through the increase in light and the subsequent die back of shade plants. Seed germination in most species, however, is favored by the exposure of mineral soil (Ehnes 1998). The organic layer remained largely intact in harvested blocks, however, the seedlings of species requiring soil disturbance for germination may have been able to establish on isolated sites of disturbance (skidder trails), or in microenvironments otherwise suitable for germination.

Localized sites of soil disturbance may have similarly improved opportunities for two species, *Epilobium angustifolium* and *Geranium bicknellii*, which thrive in the post-fire environment, and which had occurred in harvested blocks. The former had increased substantially, however the latter had been encountered only once. Logged communities are frequently distinguished by the absence of post-fire thrivers, or the limited establishment of species that may otherwise benefit from some attribute of the post-fire environment (Ehnes 1998). Known post-fire thrivers such as *Pinus banksiana*, and *Picea mariana* require the direct effect of fire to either stimulate seed dispersal or trigger seed germination. In addition, fire reduces or eliminates soil organic layers, thereby exposing mineral soil and providing a suitable germination medium for many species including *P. mariana* and *P. glauca*. *Geranium bicknellii* has been cited as a species requiring the direct effect of fire for dispersal and germination (Abrams *et al.* 1985, Ehnes 1998), however both this species and *E. angustifolium* are also frequently found in disturbed sites, forest clearings, road sides and waste ground (Johnson *et al.* 1995). Subsequently the presence of both species is likely attributable to ground perturbations in harvested sites.

Ehnes (1998), cites exotics as among the kinds of species that may distinguish post-harvest communities. An increase in the cover of exotics, particularly of *Sonchus arvensis* and *Circium arvense*, was noted within harvested sites. Most exotic species, and

some native species such as *Hieracium* sp. and *Atriplex patula*, are ecologically aggressive, establish quickly, and tend to flourish in well lit disturbed sites (Barbour *et al.* 1999). These species may invade a site from the seed bank if there is a nearby seed source, or may be transported into a logged site on forestry machinery. Both factors likely played a role in the case of this study. Thick stands of *Sonchus* sp. and *Circium* sp. were noted along an existing logging road to the north of the buffer in the vicinity of blocks E2 and C2, and at the river crossing situated near the center of the study area. Subsequently, it is of little coincidence that most individuals of these species were encountered in harvest blocks E2 and E3. The presence of exotic species may be of concern due to the prolific nature of seed production typical of many weedy species. In some situations, the establishment of a few species either present within the pre-disturbance community or associated with adjacent areas, can generate atypical post-disturbance community with implications for long term forest succession (Abrams 1985).

The increase in average cover of *Populus tremuloides* and *Betula papyrifera* among ground level taxa, suggests that over time, these species may eventually become re-incorporated into the over-story of the subsequent post harvest community. This is not surprising, particularly for *P. tremuloides*, because the rapid regeneration of this species within harvested sites in well documented (Ehnes 1998, Graham *et al.* 1963, Day and Harvey 1981, Gordon 1981, Kimmins 1997, Yang and Fry 1981). Below ground suckering from rhizomatous tissue is often stimulated by increased light intensity as well as by the release from hormonal suppression where above ground tissue have been removed or extensively damaged. Although mature residual hardwoods will eventually succumb to decay and wind-throw, the presence of hardwoods in the over-story is very likely to be maintained by advanced hardwood regeneration or newly emerging saplings at the ground stratum.

Although many stems of *Abies balsamea* had been removed from the understory due the activity of harvesting machinery and the felling of large merchantable stems, the abundance of this species in residual cover will most likely ensure its presence in the post harvest community. As will likely be the case with *P. tremuloides*, advanced

regeneration will likely ensure the presence of *A. balsamea* in the over-story after the inevitable decay of mature residual stems. Conifers do not typically regenerate from underground organs although layering is common in *A. balsamea*. Therefore this species may not immediately return to localized sites from which it had been removed. Like *P. glauca*, *A. balsamea* cannot tolerate fire (Rowe 1983). Its seeds, however, may germinate more successfully on logged sites than those of other 'fire avoider' species including *P. glauca* (Yang and Fry 1981). A substantial decrease of *A. balsamea* was noted within the ground stratum, perhaps mostly due to factors related to mechanical damage. Further assessments would likely be required to determine seedling recruitment for this species on harvested sites within the study area.

The presence of mature residual *Picea glauca* and perhaps *P. mariana* in the post harvest over-story will help ensure a continued seed source for these species into the future. *Picea mariana*, however, is partially dependant on fire to aid in seed dispersal although fallen cones may open with the aid of the sun's heat. Both species prefer exposed mineral soil for seed germination although some literature has suggested that *P. glauca* may be somewhat less dependant on this condition than *P. mariana* (Alexander and Euler 1981, Yang and Fry 1981). An adequate description of the response of these species to harvesting in the study area, however, cannot be made in one season of post-harvest sampling.

5.3 The Bird Community

Sampling conducted in the pre-harvest year in the present study produced similarities to a survey conducted by Wildlife Resource Consulting Services and Silvitech Consulting (1996) along a segment of Sandy River adjacent and east of the study site. Out of 119 species listed in the 1996 survey, species such as the Black and White Warbler, Magnolia Warbler, Nashville Warbler, Red-eyed Vireo, Swainson's Thrush, Tennessee Warbler White-Throated Sparrow and White-winged Crossbill ranked among the 20 most abundant species. This was a pattern comparable to relative abundance values produced by species encountered in the present study. Notable differences seen in the 1996 survey included relatively high Song Sparrow and Common Yellowthroat abundances. Further,

Red-breasted Nuthatches and Bay-breasted Warblers were comparatively less abundant than those seen in the present study. The similarities observed between the two studies, however, tend to support the findings pertaining to the pre-harvest community in the present study.

Changes in the bird community following the selective harvest were, however, difficult to identify clearly. For various reasons that may or may not relate directly to breeding habitat, breeding bird populations may fluctuate to varying degrees over time (Bibby and Burgess 1993). Subsequently, the time frame utilized for surveys, combined with relatively low sampling rigor, can make interpreting the results somewhat problematic. This, combined with the fact that the harvest did not result in the complete extraction of merchantable timber, can act to blur the possible impacts of the harvest on the bird community.

Bird species present in the pre-harvest community may continue to occupy harvested sites provided that key habitat requirements remain in place (Cody 1985, Micheal and Thornburgh 1971, Webb *et al.* 1977, Wedeles and Vandamme 1995). The results presented in this study tend to support the notion that stands subject to uneven-aged management tend to retain species present in the mature pre-harvest plant community (Crawford and Titterington 1979, Micheal and Thornburgh 1971, Webb *et al.* 1977, Wedeles and Vandamme 1995, Welsh *et al.* 1981, 1987, 1992). The continued presence of the Bay-breasted Warbler, Blackburnian Warbler, Cape May Warbler and Golden-crowned Kinglet, on harvested sites was likely attributable to the continued presence of mature *Picea* sp. and *Abies balsamea* stems in residual cover vegetation. Residual softwood material in the over-story and mid canopy may also have encouraged the continued presence of known mid-canopy dwelling species such as the Swainson's Thrush and Magnolia Warbler (Crawford and Titterington 1979, deGraaf and Rappole 1995). The position in ordination space of Bay-breasted Warblers, Cape May Warblers, Golden-crowned Kinglets, Magnolia Warblers and Red-breasted Nuthatches, are fairly consistent with habitat descriptions in existing literature in that they do not appear to be

associated with harvested areas (Crawford and Titterington 1979, deGraaf and Rappole 1995, Wedeles and Vandamme 1995).

The partial elimination of conifers from the pre-harvest canopy, however, constitutes a loss of habitat for many birds associated with late successional mixed woods. Subsequently, these species may occupy managed sites in relatively fewer numbers (Welsh, 1987). The slight reduction in the abundance of Cape May Warblers in harvested blocks was not unexpected given the preference of this species for relatively dense stands of mature *Picea* sp. (deGraaf and Rappole 1995, Titterington *et al.* 1979, Welsh 1981). Descriptions of Bay-breasted Warbler habitat suggest a possible preference for the foliage of *Abies balsamea*. Presumably, this is a result of its propensity for Spruce Budworm, a major food item of this species. The partial depletion of *A. balsamea* stems in the post-harvest environment may have subsequently elicited the slight reduction in Bay-breasted Warbler abundance in experimental blocks. Blackburnian Warblers, although usually associated with mature mixedwood or coniferous forests (deGraaf and Rappole 1995, Wedeles and Vandamme 1995), may be somewhat less dependent on conifer density than the Cape May Warbler. That is to say, relatively few mature conifer stems may result in the presence of a single Blackburnian Warbler (Berger 1999, pers. comm). Blackburnian Warblers, theoretically, may not respond as strongly as Cape May Warblers to partial softwood removal, but may decline nonetheless.

The reduction in the number of Northern Parula Warblers in harvested blocks was not unexpected. One of the key habitat requirements for this species, according to Sealy (1997, pers. comm.) as well as Rappole and others (1983), is the presence of arboreal lichens for use as nesting material. As a casual observation, the Beard lichen *Usnea* sp. was growing abundantly on coniferous foliage, the removal of which might have played a role in post-harvest Northern Parula Warbler abundance. Northern Parula Warblers, however, were also encountered less often in control blocks following harvesting. This suggested that a natural seasonal variations may also have played a role in post-harvest abundance.

Annual fluctuations in abundance may similarly explain the concurrent decreases in both experimental and control blocks in the post-harvest year demonstrated by several species. While episodic weather can cause local shifts in breeding populations during migration, fluctuations can also be attributed to changes in food availability within potential breeding ranges. The decline of White-winged Crossbills throughout the study area in the post harvest year, may have been a function of the irruptive nature of their populations (deGraaf and Rappole 1995, Wren et al 2000, Berger 2000, pers. comm.). Populations of White-winged Crossbills tend to follow the availability of conifer seeds and may fluctuate widely with the availability of this resource. The Pine Siskin, like the White-winged Crossbill, is a cone-dependant seed eater and, presumably, populations may vary with the cone crop. A similar inference may be made of the Red-breasted Nuthatch which may demonstrate irruptive patterns of yearly populations at a local level (Berger 2000, pers. comm.). The Tennessee Warbler, while not regularly noted as an irruptive species, is a Spruce Budworm specialist and may relocate to areas of budworm outbreaks (Cody 1985, deGraaf and Rappole 1995). This explanation may seem somewhat less plausible given the slight increase in the abundance of other budworm specialists (such as the Cape May Warbler and Bay-Breasted Warbler) in control blocks following cutting.

Several bird species demonstrated post-harvest increase in abundance which may also be due to factors either not germane to the study area, or to factors largely unrelated to logging activities. Canada Warblers, Mourning Warblers and Winter Wrens were detected more often in both control and experimental blocks in the post-harvest season. This suggested that seasonal fluctuations in breeding passerine populations had played a stronger role for these species than had harvesting activities. Further, Canada Warblers tend to prefer dense deciduous foliage in the understory and mid-canopy the post harvest environment (deGraaf and Rappole 1995, Wedeles and VanDamme 1995). While the post harvest environment may have been suitable in fostering this type of habitat, it is unlikely that sufficient time had elapsed for this to have occurred.

Although the distribution of Mourning Warblers and Winter Wrens in the post-harvest year are difficult to interpret, the increased abundance of these species in experimental blocks is, nonetheless, not suprising. Crawford *et al.* (1979) reported increases in both species within the first two years following clearcutting in a Spruce/Fir forest in Maine, although Mourning Warblers were most abundant within 3-5 years following cutting. While Mourning Warblers are usually associated with edge and early successional habitat following fire or logging (deGraaf and Rappole 1995), Winter Wrens are usually associated with mature conifer dominated forests (Green and Niemi 1980, Merrill *et al.* 1998). Crawford *et al.* suggested that Winter Wrens had been utilizing the abundance of Logging slash in the post-harvest environment for foraging purposes. This species in also known to nest in log and slash piles (Berger 2000 pers. Comm.). The presence of cull trees and residual patches of *Picea* sp. and *Abies* sp. in their study area, however, likely continued to play an important role in post-harvest Winter Wren habitat. This, and the additional presence of merchantable softwoods after selective harvesting, may have played a similar role in the current study.

The distribution of study blocks and bird species along the primary axis following correspondence analysis suggested that certain species appeared to exhibit a preference for logged habitat. Increases in the abundance of White-throated Sparrows and Dark-eyed Juncos were not surprising due to the known affinity of these species for early and mid successional habitat (Welsh 1981, 87, 92, Titterington *et al.* 1979, Telfer 1992). Titterington *et al.* (1979) found White-throated Sparrows, Dark-eyed Juncos to be most abundant in 1-2 year old *Picea* sp./*Abies* sp. clear-cuts. This was attributed to the cover which was offered by logging slash combined with the habit of these species to feed and nest on the ground. Dark-eyed Juncos are also noted by Wedeles and Vandamme (1995) to respond positively to presence of logging slash. Yellow-bellied Sapsuckers were also detected more often in both treatments in the post-harvest season, however were encountered more often in harvested blocks. This species may have benefited both from the provisions made to avoid the felling of existing snags as well as from forest openings (James 1984)

Harvesting in the streamside environment may have provided increased habitat opportunities for Song Sparrows and Common Snipes (*Gallinago gallinago*). Little information exists that addresses the impact of cutting on Common Snipe populations. Partial forest clearings may however be advantageous to this species due to its propensity to occupy open areas, particularly during spring courtship. Similarly, little information exists regarding the response of Song Sparrows to harvesting. In Ontario, Song Sparrows were not found utilizing upland boreal mixed-wood cut-overs (Welsh and Fillman 1984), but were, however, found newly occupying hardwood cut-overs adjacent to wetlands in Connecticut. The differing attributes of southern hardwood forests and the geographical distance involved, makes extrapolating this information onto post-harvest boreal riparian bird communities problematic. This information may be significant, however, due to the abundance of Song Sparrows in the boreal regions, and their known habitat affiliations. Both Cuthbert (1978) and Taylor (1983) consider the Song Sparrow to be fairly common in the southeastern regions of the Manitoba boreal forest, preferring shrubby habitat near water. Forest songbird studies in the Manitoba Model Forest yielded the classification of Song Sparrows as habitat generalists with a preference for forest stands with moderate tree height and low to moderate canopy coverage. (Wildlife Resource Consulting Services and Silvitech Consulting 1996). It was, however, found that this species was more strongly oriented toward lakes and marshes, having occurred less frequently near rivers.

The influence of vegetation situated outside of harvested sites most likely extended into harvested areas and complicated the impact of harvesting on some bird species. Some of the individuals detected during pre-harvest surveys may have occupied territory that was either partially or entirely situated within areas outside of the buffer. Most vegetation growing outside of the buffer would have been recovering from 12-15 year old cuts and would therefore have been early successional. Subsequently, the abundance of species such as the American Redstart, Black and White Warbler and Chestnut-Sided Warlber, which are often associated with early successional habitat, would appear to have changed little. Presumably, the thinning of the forest may have also better enabled the listener to detect individuals associated with habitat outside of the buffer. As a result, any

disruption in the abundance of early seral species within the buffer, due to logging activities, may have been somewhat offset by the enhanced ability to hear individuals outside of the managed area.

Harvesting is known to effect the quality of habitat for various species residing in adjacent unharvested forests (Burke & Nol 1998, Couturier 1999, Wedeles and VanDamme, 1995). Although not conclusively demonstrated by data emanating out of the current study, cutting in experimental blocks may have exerted an influence on components of the bird community in control blocks. As noted previously, White-winged Crossbills are an irruptive, cone-dependant species. The partial depletion of cone bearing trees in experimental blocks may have generally discouraged further use of the area as a whole, or at least accelerated a possible down-swing in the irruptive phase for the Sandy River area. That is to say, even the remaining foraging habitat located in control blocks may not have translated into sufficient habitat.

The influence of harvesting may have extended into the bird community of control areas, however, the overall effect may have been small. Few of the birds occurring in control blocks in the pre-harvest year have been listed as area sensitive, or are known to actively avoid edge habitat. Tennessee Warblers, for example, which showed among the most marked declines, are associated with open areas and shrubs in a variety of forest types in the boreal region (Rimmer 1998, deGraaf and Rappole 1995). Further, area sensitivity scores produced by Couturier (1999) for boreal forest bird species yielded a value of 3 out of 5 for Tennessee Warblers with a value of four or greater indicating area sensitivity. Species such as the Cape May Warbler, which is perhaps more dependant on over-story conifer density than many of passerines occurring in the study area, were encountered slightly more often in control blocks in the post-harvest year. Red-breasted Nuthatches, which declined in control blocks, may have retreated from the edge of control areas as a whole thereby yielding lower numbers. Golden-crowned Kinglets, , however, which have been listed as 'area sensitive' in Ontario (Couturier 1999), were encountered the same amount of times in control blocks during the pre- and post-harvest years. The

numbers of this species were never high, possibly due to the area encompassed by the pre-harvest buffer, thereby making results difficult to interpret.

The increase in the abundance of Bay-breasted Warblers, Cape May Warblers and Swainson's Thrushes in control blocks in the post-harvest year, accompanied by the simultaneous decrease of these species in harvested blocks suggests that individuals present in harvest blocks prior to cutting may have relocated to control sites after the harvest. In a similar study, Darveau *et al.* (1995), noted higher bird densities in unthinned forest stands immediately following harvesting in adjacent areas, and cited crowding as a possible cause. Crowding would be most likely to occur where forest remnants are situated geographically close to managed areas (Darveau *et al.* 1995), a situation akin to the design of the Sandy River study area. Crowding is not likely to persist due to the short life span of passerines (Newton 1989, Darveau *et al.* 1995).

Crowding, if in fact this had occurred, may have contributed to the decline of certain species in control blocks due to increased competition. Competitive exclusion would be most probable among species with extensively overlapping food and habitat requirements. The decline of Tennessee Warblers in control blocks may be linked to the increase in Cape May Warblers and Bay-breasted Warblers. All three species are foliage gleaning insectivores and, as budworm specialists, are known to respond to budworm outbreaks (deGraaf and Rappole 1995, Wedeles and Vandamme 1995). In addition, these three species are typical of mixed-wood forests, however, Tennessee Warblers are more strongly linked deciduous foliage than Bay-breasted Warblers or Cape May Warblers (deGraaf and Rappole 1995, Sealy 2000, pers. comm.). Subsequently it may be possible to suggest that Tennessee Warblers had been, at least temporarily, partially displaced. Such inferences, however, are highly theoretical because little direct evidence exists to support the theory of competitive exclusion within animal populations. Circumstances other than those related to harvesting may have contributed to trends in the bird community within the duration of the study.

The species that appeared as outliers in ordination space following correspondence analysis are a result of low frequency in the data set. That is to say, species that were rarely encounters were placed, through ordination, in the location of 'best fit' relative to sampled habitat parameters. Subsequently, these species tended to be placed in extreme locations within ordination space. The location of various outliers did not completely contradict known habitat affiliations. The Black-Backed Woodpecker, whose position indicated a strong preference for harvested blocks, is known to utilize logged areas as well as recent burns (Peck and James 1983, Berger 2000, pers. comm.). The position of the Pileated Woodpecker (*Drycopus pileatus*) though encountered only once, is consistent with notion that this species is not typically affiliated with logged forests (Couturier 1999, deGraaf and Rappole 1995, Peck and James 1983). The Olive-Sided Flycatcher, responds positively to forest openings or areas with a low percentage of canopy cover (deGraaf and Rappole 1995) with a particular preference for burned-over areas with tall snags. It may be interesting to note, that in the current study, this species was encountered in a harvested block, singing from a tall Spruce snag. The location of the Downy Woodpecker on the scattergram is fairly consistent with the preference of this species for mature deciduous woodlands.

The position of some rarely occurring species in ordination space may have appeared contradictory in view of known habitat affiliations. This may be a result of individuals being detected from areas in which habitat structure was not sampled. This included foliage in the lower reaches of the buffer, as well as upland habitat situated outside of the study area. The Eastern Phoebe was situated high along the habitat vector describing over-story hardwoods. Literature suggests that the Eastern Phoebe tends to occupy a wide variety of vegetation structure provided that natural or artificial ledges are available near fresh running water (deGraaf and Rappole 1995). Presumably, this species had been occupying habitat of this nature adjacent to, but not dependant upon, forest with a relatively high hardwood component in its over-story. The same may be said of the Alder Flycatcher which is not typically associated with dense mid-canopy conifers as was suggested by correspondence analysis. This species had most likely been detected calling from preferred habitat such as shrub thickets, perhaps situated adjacent to sampled

110

vegetation (deGraaf and Rappole 1995, Peck and James 1983). The Philadelphia Vireo, which appeared to be strongly associated with harvested blocks, will utilize burned or cutover areas but requires young early successional deciduous regeneration (deGraaf and Rappole 1995, Peck and James 1983). This species may more likely have been utilizing this kind of habitat associated with the upland cutover.

The effect of selection harvesting, or techniques that otherwise promote uneven-aged forest structure, is broadly felt to enhance bird diversity by enhancing vertical structural heterogeneity (Crawford and Titterington 1979, Desgranges and Rondeau 1992, Wedeles and VanDamme 1995). Theoretically, the single tree selection system provides more canopy layers throughout the management cycle, per unit area, than does other harvesting systems. This provides a greater diversity of habitat and, in turn, can translate into greater avian diversity.

Diversity values obtained using the Shannon-Weaver index of diversity suggested that the diversity of the avian community had increased marginally in experimental blocks following the harvest. This is generally consistent with literature that addresses the impact of uneven-aged management on the bird community, although the difference between pre- and post-harvest H values was not significant (Crawford and Titterington 1979, Desgranges and Rondeau 1992, Merrill *et al.* 1998, Welsh 1981). A total of 15 species in pre-harvest experimental blocks were not detected following harvesting. A total of 10 species not previously encountered in pre-harvest experimental blocks, were detected in these block following cutting. Most species which were lost to harvest blocks had registered low relative abundance values in the pre-harvest year. The Shannon-Weaver Index accounts for both species richness and evenness. All species in post-harvest experimental blocks, including those species that were gained, demonstrated higher relative abundance values than most of the species that were not encountered following cutting. This suggested that the possibility of greater evenness in the post harvest community may have more than compensated for the slight loss in richness.

The notion that harvesting, using uneven-age systems or otherwise, could substantially increase bird diversity in a riparian buffer, would seem to be counter-intuitive. Riparian areas are often characterized by high bird diversity driven by small-scale habitat diversity due to the changing influence of the aquatic environment. Post-harvest canopy openings in the upper reaches of harvest blocks may have nonetheless created opportunities which were previously absent, or at least less abundant, for some species. This would particularly be true for riparian buffers in which habitat diversity was limited by terrain (ie: extent to which water can influence adjacent terrestrial vegetation).

In view of existing literature, the key factor driving the diversity of the bird community in harvested blocks was the retention of mature, residual softwood stems (Crawford and Titterington 1979, Desgranges and Rondeau 1992, Merrill *et al.* 1998, Welsh 1981). The boreal forest is a mosaic of different forest types in different seral stages of development. While many bird species may occupy a broad range of seral stages, others will utilize a narrower range along the successional continuum. The creation of forest openings while simultaneously retaining attributes of the pre-harvest over-story, creates the kind of small-scale heterogeneity needed to attract species typical of early seral stages as well as those of the mature pre-harvest stand. Much has been written on the desirability of leaving mature residual cover. Merrill *et al.* (1998) found significantly higher abundances of forest birds associated with mature hardwood canopies in clearcuts bearing residual trees ('residual clear cuts') as opposed to clearcuts without residual patches. This had contributed to significantly higher diversity values among 'residual clearcuts'. Welsh (1981) also writes that commercial clearcutting a mature mixedwood forest (hardwood and softwood) would not likely result in the total elimination of any one foraging guild in particular. He adds that certain species which are strongly tied to mature coniferous foliage may, however, be eliminated from the post-harvest community entirely. These species include Blackburnian Warblers, Cape May Warblers, and Golden-crowned Kinglets.

Few comparable studies have been conducted in the boreal forest uplands or areas associated with riparian areas. Most existing work has either taken place in different

forest settings or have implemented different thinning regimes. Darveau *et al.* (1995) found that the removal of 33% of merchantable stems from *Abies balsamea* stands in boreal riparian buffers did not significantly affect bird diversity although the number of breeding pairs declined by approximately 20%. Studies that have specifically addressed the effect of thinning in Hardwood communities (Webb *et al.* 1977, Freedman *et al.* 1981, Dellasalla and Rabe 1987) and in coniferous forests (Virkkala 1990, Virkkala and Liehu 1990) have shown that even at intensities of 50% thinning does not result in declines in density and diversity of the same amplitude as does reducing the width of the buffer. The work of Darveau and his colleagues also revealed that interior species such as the Golden-crowned Kinglet, Swainson's Thrush, Blackpole Warbler and the Black-throated Green Warbler became nearly absent from strips reduced to 20 meters.

Although this study did not provide for a direct comparison of the effect of different harvesting methods on bird communities, the findings of this study seem to support the notion that uneven-aged selective logging may be the least disruptive to pre-harvest bird species composition and abundance. The impact of harvesting on the bird community depends largely on the amount and diversity of residual structure. Silvicultural clear-cuts (complete removal of all trees of merchantable size), for instance, will initially tend to result in relatively low species diversity and the complete removal of all canopy dwelling species (Titterington *et al.* 1979, deGraaf 1987, Thompson *et al.* 1993, Wedeles and deGraaf 1995). Commercial clear-cutting may however leave considerable structure in the form of non-merchantable species. Welsh (1981, 87), for example, notes that few foraging guilds would be lost entirely if all conifers in a mixed-wood stand were extracted from the canopy in a commercial clear-cut. He adds however, that late successional species including Blackburnian Warblers, Cape may Warblers and Golden-crowned Kinglets, which depend on mature coniferous foliage, would be substantially reduced or eliminated from the post-harvest bird community entirely. The persistence of these species, as was evident in harvested blocks, can be encouraged (albeit perhaps at lower numbers) by maintaining an inventory of merchantable conifers.

While harvesting method utilized in this study was most likely better able to attract pre-harvest species than even-aged harvesting, most authors caution that even small openings in the forest canopy may jeopardize the reproductive success of several species (Brittingham and Temple 1983, DesGrange and Rondeau 1993, Wedeles and Vandamme 1995). Forest openings tend to create edge habitat, which is recognized as poor nesting habitat for interior forest dwelling passerines. This is due to the high incidence of brood parasitism by Brown - headed Cowbirds or nest predation by corvids. Although these interactions tend to decrease with distance from the edge (Wilcove 1985), the high number of forest openings in a selectively logged stand may actually create more edge habitat than would be found in a clear-cut or burned area. (Wedeles and Vandamme 1995).

The absence of Brown-headed Cowbirds during both pre-and post-harvest years, however, suggests that brood parasitism may not be an immediate issue in the study area. This species has been regarded as rare in the Manigotagan/Sandy River after recent surveys throughout the Forest Management License (Wildlife Resource Consulting Services and Silvitech Consulting 1997). The Brown-headed Cowbird, however, is perceived to be expanding range in the province from the southwest in a northern and northwesterly direction (Berger 2000 pers. comm.). This species has been observed in the Pine Falls FML, utilizing non-riparian stands of *Pinus banksiana* and edge habitat affiliated with beaver floods and burned stands (Wildlife Resource Consulting Services and Silvitech Consulting 1997). Although Brown-headed cowbirds in the Pine Falls FML appear to be most common in southeastern areas, the continued expansion of this species' range could eventually result in higher frequencies in areas where it is now considered rare.

5.4 Erosion and Sedimentation

Upland terrestrial measurements yielded no observable soil erosion in any transect, with the exception of two sites where mineral soil had been exposed (Appendix I). Shoreline measurements suggested that greater soil movement had occurred in harvested blocks than within control sites although the differences were not significant. Aquatic

114

measurements revealed sedimentation in all sampled locations. Substantially more sediment had accumulated in experimental and control blocks, E2 and C2 than in sampled areas elsewhere. Differences in the weight of accumulated sediment were significant for E2 and C2. Differences were not significant between paired segments E3 and C3 or between E4 and C4.

The results of terrestrial measurements in this study tend to support the findings of similar work within the Manitoba Model Forest. The bi-directional movement in ground height frequently noted in the results was most likely attributable to the expansion and contraction of organic matter in the upper soil layers rather than to the movement of the mineral component. This explanation is based largely on the work of Schnieder-Vieira (1996) in Manitoba FML 01, who made similar observations where post logging organic layers remained intact. The flexing of the soil organic layer is most likely due to the shrinking and expanding of soil organic material as well as bryophyte vegetation during intermittent periods of drying between rain showers. This explanation seems plausible because the complete removal of organic layers was rarely noted. Further, where organic layers remained intact, mineral soil would not likely be exposed to the erosive action of rain.

The regeneration of herb and shrub species on harvested sites may also have contributed substantially to the perceived overall lack of upland soil erosion (See Plate 5.1.). Terrestrial vegetation can anchor soil material on steep slopes thereby lessening the chance of a soil 'mass movement' event (DFO 1992). Regenerating plant material may, in addition, act to reduce the erosive action of water by intercepting water droplets as well as surface flow. Presumably, plant foliage may also act as a shelter to exposed soil material by reducing wind speed at the ground level.

Changes in ground height on sites where the organic layer had either been substantially compromised or eliminated, was likely attributable to surface movements in the mineral layer. Exposed mineral soil was sampled for changes in ground height on two occasions, both of which yielded a decrease in ground height for both 'ridge' pin measurements. The

Plate 5.1. Plant growth surrounding upland erosion pin on a harvested site.

consistent decrease in ground height indicated by negative values suggested that changes were most likely due to soil movements in the exposed mineral layer. Schnieder-Vieira (1996) also noted evidence of upland soil erosion on sites where the mineral layer had been exposed. She attributed the exposure of the mineral soil layer, however, to post-harvest activities related to site preparation. Disruption of the organic layer in isolated areas of harvested blocks within the Sandy river buffer, however, was likely due to the repeated use by skidders dragging felled timber. The erosion of exposed soil was, however, greater than that which was observed by Schnieder-Vieira. This may have been attributable to factors relating to slope as grades were steeper where erosion was detected in this study, than in any of the sampled sites in Schnieder-Vieira's work.

Interestingly, the erosion that was observed within the 2 upland erosion transects was greater than any movement detected by shoreline pins which were installed in the exposed soil of naturally eroding streambanks. Residual shoreline vegetation may have played a role in this difference by at least partially intercepting rainfall. It was noted that upland sites, on the other hand, were completely defoliated and therefore fully exposed. Nevertheless, no movement of soil material was noted in upland transects, or in ephemeral streams. Eroding soil material in upland sites would most likely have been eventually intercepted by organic layers left in place, or would have been deposited in flatter terrain. Subsequently, the exposed soil of the streambank is the most probable source of the soil material that had accumulated in the sediment traps.

The deposition of mineral soil material in all sediment traps, including upstream locations grouped into control block C4, suggested that sedimentation was partially a result of processes not related to harvesting activities in the study site. Erosion and sedimentation is a natural process in rivers where streambanks are composed of erodible material in the form of glacial till and/or fluvial deposits (Cole 1983). The hydraulic forces associated with the outside shoreline of river bends will, over time, deepen the river bend by dislodging soil material to be later deposited in backwater eddies or on the inside curve of downstream river bends. The floristic diversity of many riparian areas is actually a result of erosion and the subsequent deposition of material. The gradual accumulation of

117

sediment, for example, will eventually act as a suitable rooting medium for shoreline emergents and eventually larger hardwoods such as *Fraxinus nigra*. The data presented in this study also seem to reflect the variability inherent to the rate of sedimentation that can be found along a particular segment of shoreline.

The relatively small amounts of material collected along river segments passing through blocks C4, E4, C3 and E3 compared to the amounts collected within blocks C2 and E2, indicated that site conditions effecting erosion and sedimentation varied along the length of the river. The commonly exposed, poorly vegetated streambanks of blocks C2 and E2 indicated that the shoreline in these blocks had been actively eroding prior to harvesting. This was particularly noticeable on the 'outside' shore of river-bends where hydraulic forces causing shoreline erosion would have been most strongly exerted. The absence of colonizing vegetation in these sites was an indication of the instability of the soil along these banks. Shorelines within blocks E3, C3, E4 and C4 did not show obvious signs of erosion. The shoreline environment was usually well colonized by vegetation and often displayed a gradual slope into the active channel. Subsequently, erosion may have been occurring at an overall slower rate along these river segments. Sedimentation was nonetheless noted along these sites. The presence of extensive emergent vegetation late in the spring and throughout the summer may have locally reduced the river's velocity and thereby facilitated the removal of sediment from the water column. For this reason, greater sedimentation might have been expected in these areas rather than along the shorelines of blocks E2 and C2 which could be characterized as usually lacking any emergent vegetation. Water velocity may, however, have been sufficiently slow to allow sedimentation in blocks E2 and C2 to surpass that of other sites.

The difference between the amounts of sediment collected in experimental block E2 and control block C2, were difficult to interpret, particularly in the absence of adequate baseline data which would better identify the role of natural variability in sedimentation. Notwithstanding natural variability, however, several factors pertaining to harvesting as well as to sampling error may have contributed to the observed differences. Although shoreline pins did not yield a significant difference in erosion, data implied that slightly

118

greater soil movement had occurred along streambanks situated in harvested block E2. Greater erosion along harvested shorelines would likely have been due to increased exposure bare streambank soil to the erosive action of rain. Subsequently, this may have contributed to greater sedimentation within the harvested block. In addition, harvesting would have left the channel more exposed to wind. The subsequent increase in wave action may have led to greater perturbation of shoreline material, resulting in greater input into the active channel*.

The exposure of sediment traps during sampling, however, may have contributed to the observed differences. Observed differences in blocks E2 and C2 were likely due to a combination of the above factors. The exposure of sediment traps, however, precludes the opportunity to make meaningful inferences into the effect of harvesting on sedimentation. It was noted that a total of 9 were found exposed in control block C2 as opposed to 3 found exposed in experimental block E2. Whether or not observed differences in sediment accumulations would have been significant in the absence of sampling error remains undetermined. The absence of baseline data however may also further circumvent any attempt to draw meaningful conclusions.

5.5 Streamside Aesthetics

Significant differences were found between photographic images featuring harvested scenes, and those featuring unharvested scenes for both series 1 and series 2 photographs. Out of 10 'Series 1' photo-stations, all but one post-harvest image ranked lower than the corresponding pre-harvest image. Significant differences were stronger between harvest and control images in Series 2 photographs than for Series 1. Predictably, values for photo-stations situated in control blocks did not differ significantly between pre- and post-harvest years. The comments that were provided by participants indicated that the

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*sediment traps often had to be reinstalled when disturbance of the shoreline during initial installation (by canoe) resulted in sediment entering the traps prematurely.

scenic attributes which appeared to be playing a role in scenic valuation included; tree health, lushness of foliage, diverse and contrasting vegetation (diversity) signs of human disturbance, and possibly, tree density.

Considering the results presented here, it may be reasonably assumed that selective tree removal had, in nearly all cases, a negative impact on the aesthetic quality of the streamside environment in harvest blocks. Images of unharvested sites clearly drew higher marks in terms of scenic value which were often supported by positive comments. Comments such as 'scenic', 'dramatic', 'pretty', and 'spiritual', however, reveal very little about the specific scenic attributes which the viewer found pleasing. On occasion, comments such as lush, tall and diverse, would provide better insight into which attributes were favorable to the viewer. More perhaps is revealed about the aesthetic preferences through comments linked to images featuring harvested scenes.

Taken alone, the lower grades acquired by harvest images, would suggest intuitively that the quantity of timber removed was the main factor. That the quantity of remaining trees had an influence on scenic valuation can, however, only be inferred. This is mainly due to the absence of comments directly referring to tree quantity. Certain comments and adjectives such as 'barren', 'thin', 'bare banks', 'veneer' and 'not a forest', and the values with which these comments where often associated, tend to imply that the quantity of stems removed had played a role in scenic valuation. Likewise, no comments addressing tree sparsity were directed at images of unharvested sites. Series 2 harvest images A and E give the impression that a lack of mature trees was reflected in the values these images had attained (see images in results section). Worthy of mentioning is that both photographs A and E had been referred to as 'not a forest' while photograph A had drawn the remark referring to 'bare banks.'

Most comments linked to scenes of harvested sites referred to the quality rather than the quantity of remaining vegetation. Remarks pertaining to the health of trees and the sparseness of foliage reoccurred in harvest scenes in both series 1 and 2, and on three occasions in scenes of unharvested sites. Comments pertaining to poor tree health in

120

unharvested sites involved a direct reference to chlorosis on one occasion. Others comments may have been in response to the presence of 'beard' lichen (*Usnea* sp.) which is an arboreal species not uncommon to the study area. The presence of sick or dying trees in unharvested scenes apparently did not have a substantially negative effect on the overall appearance of these images.

It is arguable, however, whether or not many of the residual trees in harvested scenes were actually 'sick'. There are a number of alternative explanations for the sparseness of foliage and/or branches. One possible explanation is that the branches of some residual trees were most likely knocked off during the felling of neighboring trees. The size of many trees that were felled and the density of the forest made damage to residual material difficult to avoid. An equally likely explanation pertains to the effect of poor light availability on plant foliage. Plants, or portions of plants existing in poor light conditions, tend to produce sparse foliage (Barbour *et al.* 1999). Although the growth habit of conifers results in prolific branching throughout most of the length of the stem, foliage may be concentrated near the top of the stem in dark forests (see comments for series 2 harvest images) . To the viewer, the removal of neighboring trees may enhance the visibility of this growth pattern. The post-harvest image of series 1 photo-station A implies that the presence of snags and other dead material may have further compounded this effect.

Deciduous trees may be less susceptible to the effect of shade in the pre-harvest stand, because most foliage normally inhabits the tree's apex regardless of ambient light conditions in the mid canopy. The shade cast by neighboring conifers may however influence the abundance of hardwood foliage in mid canopy levels. The post-harvest image of photo-station D, for instance, in which hardwoods dominated the foreground, drew comments referring to unhealthy trees on two occasions. This may also have been linked to damage caused during the felling of neighboring conifers.

The presence of deciduous trees may, on some occasions, have played a role in offsetting the visual impact of conifer removal. Harvested sites lacking hardwoods tended to

acquire lower scenic values. This is well illustrated, once again using the example of series 2 harvest images E and A, where hardwoods were likely not present in the pre-harvest community. Managing for streamside aesthetics in these situations may be problematic considering the relatively low values given to these sites by study participants. Conversely, many of the scenes in the post-harvest photographs in series 1 featured abundant mature hardwoods. The presence of hardwood may subsequently have contributed to higher post-harvest values in series 1 compared to those of series 2. This is perhaps best illustrated using the example of photo-station D where the amount of remaining softwood material was comparable to that of high impact scenes in series 2. The abundance of *Fraxinus* sp., *Acer spicatum* and various tall shrubs in the foreground likely allowed the post harvest image to acquire a considerably higher value than if the hardwoods were otherwise absent.

In certain cases the angle of the photograph may have moderated the visual impact of harvesting. This may have been a factor in the valuation of the post-harvest image of photo-station G in series 1 photographs. A close look at the pre- and post-harvest images for this photo-station, reveals that nearly all *Picea* sp. had been removed leaving only *Abies balsamea*. The values for both images, however, were nearly identical. A comparison of the pre- and post-harvest images for this photo-station suggests that, from the angle at which the image was taken, *A. balsamea* and *Picea* sp. may have been visually substitutable. This may not have been the case if the photograph had been taken perpendicular to the shore. The angle of the photograph may have concealed the effects of tree removal, sparse foliage as well as skidder trails (which were occasionally visible from the water). Although participants were not informed of harvesting activities, comments suggested that human caused disturbance was suspected for harvest images A and E in series 2 photographs. Two individuals specifically identified trails in image A, while tree stumps were seen in both A and E. It is not likely that the suspicion of logging significantly 'colored' the impression of the image to the observer because the remarks that were provided suggested that relatively few of the participants appeared to notice signs of logging. It is worthy of noting, however, that these two images ranked among the lowest scenic values given to any photograph. Any obvious signs of logging such as tree

stumps, trails and slash may, however, disappear or become blurred as ground vegetation continues to re-colonize these areas.

Overall, the initial impact of harvesting on streamside scenery can be expected to fade as the canopy replenishes itself either naturally, or with silvicultural assistance. The increase in *Populous tremuloides* and *Betula papyrifera* at the ground stratum (identified through vegetation sampling) suggests that these species will be an important component of the regenerating understory and over-story. In addition, advanced softwood regeneration similarly implies that the recovering canopy will have a substantial softwood component. Improved light conditions will also provide the necessary environment required for thicker foliage to develop as tree species continue to mature. Considering the results discussed here, this may be of particular importance to conifer foliage.

The mix of softwood and hardwood foliage in the recovering canopy has important implications for the future streamside aesthetic. Newton (1985), stresses the scenic value of a mixed-wood forest. This is also reflected, albeit not conclusively, through some of the comments provided by study participants. Series 1 control photo-station C, for instance was noted by one individual as being 'too uniform'. Conversely, the contrast of deciduous and coniferous foliage in Series 1 control photo-station B, as well as the pre-harvest image for photo-station D, drew favorable remarks pertaining to diversity. Two post-harvest images (Series 1; F & B) also received favorable remarks pertaining to floristic diversity. This is particularly significant for image B because the angle at which this photograph was taken, and the slope of the terrain suggested that little of the visual impact of harvesting was likely to have been concealed.

5.6 Management Perspectives

Riparian management is an undertaking that presents challenges that are both unique and familiar to forest managers. The shortcomings associated with conventional 'zero disturbance' buffer management has been a perennial issue in Canada's forestry sector, raising questions pertaining to the more efficient utilization of the riparian timber

resource. The riparian zone has played a multi-faceted role in the boreal landscape, due in part to its social and recreational value, as well as its role in boreal ecology. Against this backdrop, the riparian zone represents an area where the potential for conflict between timber and non-timber values, can be substantial. The issues faced by management objectives aimed at integrating diverse forest resource values can be equally substantial.

The integration of different resource values in landscapes subject to commercial logging can be approached at both the 'landscape' level as well as at a more local 'stand' level (Kimmins 1997). While the cutting design used for the current study clearly subscribed to the latter approach, the suggestion that the impacts among the various resources considered may not be distributed evenly, is worth considering. Data appeared to suggest, for example, that the streamside aesthetic may have been the most obviously impacted. The potential for forest managers to experience difficulties in being able to strike a balance among various forest users, may become increasingly evident where resource values have proliferated. Priorities may shift on a site specific basis. Sandy River is not heavily traveled by water recreationists. Subsequently any impact to the scenic value of its streamside environment may go largely unnoticed, especially if the impact is short lived. This, however, may not be the case for more frequently traveled rivers within FML 01.

In Manitoba, integrated resource management is nested within the province's recent move towards 'ecosystems based management' (EBM) (KPMG 1996) in which it is stated that 'all uses and users of forest resources' are to be considered. The plan is similar to existing literature dealing with IRM in that wildlife and biodiversity appear to be the main non-timber values underlying the approach (Kimmins 1997, Thompson and Welsh 1993). One of the major impediments to the implementation of integrated resources management, according to Thompson and Welsh (1993), is the lack of a sufficient understanding of species interrelations and habitat requirements. In this respect the contribution of data emanating from the current study may be of only marginal significance for two reasons made reference to in a previous chapter; 1) not all

biophysical resources of the riparian environment were examined, and 2) not all potential impacts to resources considered were fully examined.

While selection cutting may intuitively be the harvesting system best suited to the integration of non-timber values, there are unique operational considerations that are associated with this approach. According to Wedeles and VanDamme (1995) such considerations include minimizing residual stand damage as well as the on-site regulation of the cut. Damage to residual trees, mature or otherwise, is nearly impossible to entirely avoid. This is particularly true where tree density is high. The use of 'rub trees' in the delimbing process is problematic in that damage to the outer bark surface may facilitate the entry of infectious agents. This can compromise the potential volume of harvestable timber throughout the rotation (Wedeles and VanDamme 1995), and perhaps the quality of habitat for certain wildlife species utilizing the post-harvest stand.

In spite of careful planning, the regulation of a selection harvest can be problematic simply because the process tends to be inherently more complex and time consuming than other harvesting systems. Selection cutting, which requires the careful removal of trees in different size/age classes, may be more easily accomplished by the felling of trees in groups rather than single tree selection. The ease with which single tree selection is operationalized may ultimately depend on the targeted percentage of mature stems to be incorporated into residual cover. The process of selection harvesting may further be complicated, as may have been the case in the current study, where certain merchantable species are preferred over others. As a merchantable species, *Abies balsamea* appeared to have assumed a disproportionately high importance in the post harvest canopy as opposed to *Picea* sp. which may have been slightly over-utilized. This is largely due to the notion that harvesting and processing equipment are not well suited for resinous species such as *A. balsamea*.

Selection harvesting in the boreal forest, while perhaps the least disruptive to the pre-harvest stand in comparison to other harvesting systems, may, in certain cases, produce unintended side effects. The thinning of the stand may potentially increase the

susceptibility of residual stems to wind-throw. This would presumably be a large factor in stands growing on thin soil. *Pinus banksiana* stands growing on granitic ridges, for instance, are a common feature of the boreal region. It is interesting, at this point, to note that during the field season, one possible wind-throw was noted on one occasion within a post-harvest cut block. In this case an *Abies balsamea* stem (approx. 8m in height) growing on a granite outcrop had been toppled. Over-story stems, an inventory of which are expected to remain following selection harvesting, might also be more susceptible to wind-throw due largely to the wind resistance of foliage combined with a high center of gravity. Older, larger trees may also be weakened as a symptom of fungal infection. Blow-down may not be as large a factor where forest stands are unaffected by disease and where soils are deep, allowing mature trees considerable anchorage.

While the intent of this study was not to compare the impact of natural disturbances, such as fire, to those of harvesting, such comparisons are difficult to avoid. Fire, being perhaps the most pervasive natural disturbance in the boreal regions of central Canada (Wedeles and VanDamme 1995), does not tend to leave the quantity of residual cover characteristic of a selection harvest. Fire skips, though common, are not the prevailing feature of a post-fire boreal stand. The initial impact of a fire, in relation to the resources addressed in the present study would, in all likelihood, be of a greater magnitude. Post-fire bird communities tend to be dominated by early successional species with perhaps a few individuals typical of mature forests finding habitat in fire skips (Cody 1981, 1985, Wedeles *et al.* 1995, Welsh 1987). Soils tend to be more susceptible to erosion immediately following fire due to reductions in the organic layer and, occasionally, the deposition of hydrophillic substances in the 'B' horizon (Barbour *et al.* 1999). Further, the work of Boxall and others (1995) suggested that burned landscapes may be among the least aesthetically attractive to backcountry recreationists in Nopoming Provincial Park. In attempting to reduce the impact of harvesting to resources related to riparian areas, selection harvesting creates somewhat of an artificial stand. The deeper issue at hand perhaps, is not the extent of a disturbance, but rather the ability of the stand to recover.

As previously noted, recent investigations suggest that harvested boreal stands may be somewhat less resilient than burned stands (Ehnes 1998). Tree species which rely on some attribute of the post-fire environment may not regenerate as well following logging (Ehnes 1998, Wedeles and VanDamme 1995, Yang and Fry 1981). Wedeles and VanDamme (1995) note that selection harvesting in boreal mixedwoods tends to regenerate *Abies balsamea* to the possible detriment of hardwood species. They further add that an increase in *A. balsamea* may reduce site availability for *Picea* sp. regeneration. The implication is such that certain attributes tied to the forest that was harvested, may be at stake in the long term if the forest cannot self-perpetuate. Subsequently, steps relating to site preparation will need to be taken to promote the regeneration of desirable species.

Post-harvest stands with high *Abies balsamea* cover may, however, undermine the potential benefits of riparian management. The forest industry in Manitoba has raised concerns over the potential damage that decadent buffers may have upon adjacent stands, or softwood regeneration in the post-harvest upland community. Much of this concern is centered on the potential spread of Budworm infestations out of aging buffers. The risk of infestation may be higher, or more severe in stands that feature a high frequency of *A. balsamea* due to the affinity of Budworm for this tree species. Harvesting may result in a significant thinning of *A. balsamea* within the forest over-story, and would likely reduce the severity of a future infestation. Advanced *A. balsamea* regeneration, however, may continue to pose a substantial threat to regenerating softwoods. Density measurments within the harvested blocks of the current study suggest that *A. balsamea* may emerge as a major component of future mid-canopy and over-story vegetation.

Chapter 6: Conclusions and Recommendations

6.1 Summary

Management in riparian buffers present formidable and, in many ways, unique challenges to forest management initiatives. A number of social and ecological resource issues converge in the riparian zone. As a result, the challenges associated with integrated resource management may arguably be most acutely felt when forestry activities are conducted in these areas. The intent of the current study was to address the effects of a selection harvest on various resource attributes of a riparian buffer situated within Manitoba Forest Management License 01. Information provided through this research may, by extension, provide forest managers with a more informed approach to future management initiatives within riparian areas. After the analysis of two seasons of primary data, the following conclusions and recommendations were made;

6.2 Conclusion

Dominant tree species within the upland reach of the study area consisted of merchantable softwood species, suggesting that much of the area had been in an advanced seral state at the time of harvesting. While an inventory of merchantable species had been retained in the post-harvest stand, data suggested that *Abies balsamea* had been underutilized. As a community, post-harvest vegetation may have been more influenced by the pre-harvest plant community rather than harvesting. Shifts in species relative abundance were noted in the bird community in the post-harvest year. The post-harvest abundance of certain species were attributed to logging activities while factors unrelated to logging may also have been at play. Bird diversity was highest in combined post-harvest experimental blocks. Upland erosion was not detected where the soil organic layer remained intact, however soil material had accumulated in sedimentation traps. Soil accumulations in sediment traps most likely originated from the streambank environment rather than from upland locations. Differences observed in the amount of accumulated sediment between control and harvest blocks were rendered unreliable, however, due to factors that emerged during the sampling period. Harvesting activities were revealed to have had a negative effect on the scenic value of experimental blocks.

128

6.2.1 Specific Conclusions
Specifically, the following conclusions were made:

6.2.1.1 The Timber Resource
The Over-story in the study area was dominated by the merchantable species *Picea* sp. and *Abies balsamea* with average relative dominance values of 36.95% and 41.94% respectively. The relative dominance of over-story stems combined with Forest Ecosystem Classification data suggested that forest in the sampled regions of the buffer was in an advanced state of forest succession. The combined volumes of merchantable species exceeded suggested minimum requirements in all study blocks. A selection harvest was designed to extract roughly two thirds of merchantable timber from designated experimental harvest blocks. Post-harvest density and relative dominance values suggested that *Picea* sp. may have been disproportionately utilized during the harvest. Targeted volumes of residual material of merchantable size were achieved in all harvest blocks

6.2.1.2 Vegetation
Correspondence analysis suggested that harvesting initially imparted less influence on the floristics of the post-harvest plant community than the spatial variability inherent to the study area prior to harvesting. Shade tolerant species and most mosses occurred at lower frequencies and reduced coverage but were maintained on harvested sites likely due to the presence of residual trees and shrubs. Several shade-suppressed species present in the pre-harvest plant community exhibited a substantial increase in frequency and cover following harvesting due to improved light conditions associated with canopy openings. Better light conditions and localized disturbances also facilitated the presence of certain native and exotic species which were not encountered in the pre-harvest community. The substantial increase in weedy exotics was attributed to the presence of a nearby seed source as well as the propensity of these species to colonize newly available sites. The inference was made that the presence of exotics may potentially be prohibitive to community resilience.

6.2.1.3 The Bird Community

Harvested blocks retained all species of which 2 or more individuals had been encountered experimental blocks during the pre-harvest year. Several of these species, including Cape May Warblers and Bay-breasted Warblers which depend on conifers, however, exhibited lower abundance in the post-harvest community potentially due to the reduction of preferred habitat. The continued presence of these species, as well as the Golden-crowned Kinglet, was attibuted to the presence of mature residual coniferous foliage in the post-harvest environment. An increase in the abundance of Dark-eyed Juncos, Mourning Warblers and White-throated Sparrows, in post harvest experimental blocks was attributed to the known affinity of these species for early successional habitat. Species including the Canada Warbler, Pine Siskin, Tennessee Warbler, White-winged Crossbill and Winter Wren exhibited changes in abundance with post-harvest experimental blocks similar to changes observed in control blocks in the post-harvest year. This suggested that factors unrelated to harvesting activities, such as season population fluctuation, had played role in the post-harvest bird community. Although a net loss of 5 species occurred in pre-harvest experimental blocks, bird diversity was highest in combined post-harvest experimental blocks. This may have been due to greater species evenness in the bird community within combined experimental blocks following harvest. Observed differences in diversity were, however, not statistically significant.

6.2.4 Erosion and Sedimentation

Surface erosion was not detected where soil organic layers remained intact. Observed changes in ground height were attributed to the intermittent flexing of the organic layer between rain events rather than to the lateral movement of surface material. Possible soil erosion had occurred on two isolated occasions where terrain was steep and where the activity of skidders had gradually worn away the soil organic layer. The accumulation of soil material in sediment traps was largely attributable to natural processes associated with the streambank environment. The relatively high sediment accumulations observed in blocks E2 and C2, likely reflected spatial differences in erosion rates within the study area. The perturbation of shoreline material due to potentially higher wind-driven wave action in block E2, may have contributed to the significant difference observed in

accumulations between this block and the adjacent C2. Sediment traps were occasionally exposed due to decreasing water levels. Subsequently, the difference that was observed between overall accumulations in blocks E2 and C2 were rendered unreliable. This and the lack of sufficient baseline data had pre-empted the opportunity to draw conclusions regarding sedimentation.

6.2.5 Streamside Aesthetics

Harvesting activities had produced a negative impact on the quality of scenery in harvested sites. Comments linked to images of harvest sites provided insight into the scenic preferences of volunteers participating in the study. Most comments referred to the qualitative attributes of residual trees. Of these, most comments referred to poorly foliated trees or trees that appeared unhealthy. Many residual trees exhibited sparse or uneven foliage, giving the impression of poor health. The 'sickly' appearance of residual material was likely attributable to poor, mid-canopy light conditions in the pre-harvest stand, further compounded by the presence of snags. *Abies balsamea* may have had a disproportionate role in this respect due to its position in the pre-harvest canopy as well as its relative dominance in post-harvest residual cover. Results indicated that the presence of hardwoods may have moderated the visual impact of harvesting, and may have imparted a positive scenic influence when featured with a mix of conifers. Visual evidence of logging or other human activity was conveyed through comments and may have also played a role in scenic valuation.

6.2 Recommendations

Based on the finding of this study, the following recommendations are made in the context of operational considerations and areas of future study:

6.2.1 Operational Considerations

Pertaining to operational considerations, it is recommended:

- *that regeneration of Spruce (Picea sp.) be supplemented by planting.*

Rationale:

Literature indicates that certain *Picea* sp. may exhibit poor regeneration and productivity on logged sites. To help ensure that the post-harvest canopy is able to re-approximate the relative composition of the pre-harvest community, this species may require silvicultural assistance. Post-harvest silviculture may be of particular importance where weedy or otherwise ecologically aggressive species have established in the post-harvest community.

- *that efforts be made to avoid the under-utilization of tree species most susceptible to Spruce Budworm infestation.*

Rationale:

The under utilization of *Abies balsamea* may continue to promote conditions favorable to large scale budworm infestation, and may play a disproportionate role in the degradation of the streamside aesthetic.

- *that the primary timber interest explore a more restrictive size limitation for selection harvesting purposes.*

Rationale:

The use of a harvest limitation that is more restrictive than stems with a minimum dbh of 10 cm may allow managers to better capture residual trees that are both mature and relatively wind-firm.

- *that efforts be made to avoid damage to mature residual trees.*

Rationale:

Collateral damage is virtually unavoidable in selectively harvested forests. The intentional use of 'rub trees' in the delimbing process, however, should be avoided. The breakage of bark tissue can expose underlying vascular tissue thereby acting as an entry point for plant pathogens. This may subsequently compromise the future value of residual trees, as wildlife habitat and scenic amenity. Damage of this nature can be avoided by using a 'feller-buncher' which may also allow the tree to be processed more efficiently.

6.2.2 Future Study

Pertaining to areas of future research, it is recommended:

- *that research be aimed at addressing the effects of selection harvesting in riparian buffers conducted at the time of the upland harvest.*

Rationale:

Harvesting within riparian areas may be most practical when conducted simultaneously with the adjacent upland harvest. Visiting the site only once may reduce the economic overhead of mobilizing equipment and operators associated with a second visit to the area. The costs subsequently averted may translate into a stronger incentive to leave a richer inventory of high quality and wind-firm timber on site.

- *that an area be designated for future research to better acquire baseline data for possible impacts that may require longer periods of time to be adequately addressed.*

Rationale:

Inadequate baseline data may subsequently confound attempts at identifying the impact of harvesting on resources that exhibit temporal and or spatial variability. Adequate baseline data can provide a more adequate frame of reference when analyzing the effect of harvesting on bird populations and non-timber values associated with the riparian environment.

133

- *that studies be initiated to better address the effects of harvesting on riparian bird communities including the effect of selective harvesting on forest songbird productivity.*

Rationale:

The data presented in this study that pertains to the effect of cutting on the bird community, although suggestive, is incomplete. In order to facilitate better statistical power, a larger study area will be required. Information obtained from comparable and existing upland sites subject to selective harvesting may also provide a valuable contribution to this kind of information. Further, the mere presence of certain species in a managed area reveals little about their reproductive success. Edge habitat created by selective harvesting may result in interior bird species being more vulnerable to nest predation and brood parasitism. Considering the importance of riparian areas to bird populations in the boreal forest, this information will be of value to future management considerations

- *that work be initiated to address riparian resources not subject to investigation in the present study.*

Rationale:

A more comprehensive analysis of the effect of selective harvesting on the riparian resource will better inform the decisions of resource managers in future initiatives pertaining to riparian management decisions. Areas of future study that would benefit future management decisions include the effect of selective harvesting on small mammal populations, amphibians and arthropod populations. These components of riparian ecosystems are important areas of study due to their supportive role in boreal food chains. Areas of study pertaining to the aquatic environment itself should include an investigation of the effect of harvesting on water turbidity and chemistry.

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141

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Appendices

Appendix A: Description of vegetation types based on Forest Ecosystem Classification for Manitoba (Zoladeski *et al.* 1995), which were encountered during vegetation sampling within the study area.

Vegetation Type	FEC Description	Comments
V2	Black ash, White Elm Hardwood	Occurs in isolated patches along periodically flooded creeks and small rivers
V5	Aspen Hardwood	Fire origin, successionaly young, may show signs of invasion by softwoods, will gradually succeed into mixed hardwoods
V6	Trembling Aspen-Balsam Fir/Mountain maple/Herb Rich	Somewhat older than V5, Balsam fir may gradually replace aspen as a dominant species
V7	Trembling Aspen-Balsam Fir/Shrub and Herb-Poor	Lack dense cover of broadleaf shrubs, crown closure of Balsam fir greater than V6
V8	Trembling Aspen Mixedwood/Tall Shrub	Change toward conifer dominated stands featuring an admixture White spruce, Balsam Fir and Black Spruce
V9	Trembling Aspen Mixedwood/Low Shrub	Transitional between young aspen dominated communities and older spruce dominated communities
V10	Trembling Aspen Mixedwood/Feather Moss	Fire origin and successionally young, less predictable successional pathways
V13	White Spruce Mixedwood	Successionally mature, often characterized by an Uneven-age structure
V21	White Spruce/Balsam Fir Shrub	White Spruce dominated stands with variety of species in the tree canopy, many possible successional pathways leading up to this type

153

Appendix B: Species list of plants encountered within the study area.

Abies balsamia

Acer negundo

Acer spicatum

Achillea millefolium

Achillea multiflora

Actea rubra

Agropyron trachycaulum

Agrostis scabra

Amelanchier alnifolia

Amelanchier sanguinium

Anemone canadensis

Anemone quinquifolia

Apocynum androsaimifolium

Aqualegia canadensis

Aralia nudicaulus

Asclepias sp.

Aster umbellatus

Aster hesperius

Aster ciliolatus

Atriplex patula

Betula papyrifera

Botrichium virginianum

Brachythecium sp./Eurhynchium sp.

Bromus ciliatus

Calamagrostis canadensis

Caltha palustris

Carex sp.

Cerastium vulgatum

Chimaphilla umbellata

Cinnia latifolia

Circea alpina

Circium arvense

Cladina mitis

Cladionia pyxidata

Cladonia sp.

Cladonia sulphurina

Climacium dendroides

Clintonia borealis

Coptis trifolium

Cornus canadensis

Cornus stolonifera

Corydalis aureus

Corylus cornuta

Dicranum fuscellum

Dicranum polysetum

Dicranum scoparium

Diervilla lonicera

Disporum trachycarpum

Drepanocladus sp.

Dryopteris sp.

Elymus sp.

Epilobium angustifolium

Epilobium glandulosum

Equisetum arvense

Equisetum pratense

Equisetum sylvanicum

Erigeron canadensis

Fragaria virginiana

Fraxinus nigra

Gallium borealis

Gallium triflorum

Geranium bicknellii

Geum aleplcum

Goodyera repens

Gymnocarpium sp.

Heiracium sp.

Heuchera richardsonii

Hylocomium splendens

Impatiens capensis

Iris versicolor

Lactuca pulchellum

Lathyrus ochroleucus

Lathyrus venosum

Linnaea borealis

Lycopodium annotinum

Lysmachia ciliata

Malanthemum canadensis

Matteuccia sp.

Medicago sativa

Mentha arvensis

Mertenzia paniculata

Mitella nuda

Moneses uniflora

Monotropa uniflora

Muhlenbergia racemosa

Neckera sp.

Onoclea sensibilis

Orchis rotundifolia

Oryzopsis asperifolia

Peltigera sp.

Petasites palmatus

Petasites vitifolia

Picea glauca

Picea mariana

Plagiommium sp./Mnium sp.

Plantago major

Pleurosium schreberi

Polygonam cilinode

Populus tremuloides

Potentilla norvegica

Prunus virginiana

Ptillium crista-castrensis

Pyrola asarifolia

Pyrola chlorantha

Pyrola grandiflora

Pyrola minor

Pyrola secunda

Rhamnus alnifolia

Ribes hudsonianum

Ribes glandulosum

Ribes lacustre

Ribes oxyacanthoides

Ribes triste

Rosa acicularis

Rubus idaeus

Rubus pubescens

Rhytadiadelphus sp.

Salix sp.

Sambucus sp.

Sanicula marlandica

Smilacina stellata

Solidago canadensis

Solidago sp.

Sonchus arvensis

Stachys palustris

Streptopus roseus

Symphorocarpos albus

Thalictrum dasycarpum

Thalictrum venulosum

Thuidium sp.

Thuidium recognitum

Trientalis borealis

Trillium cernuum

Urtica procera

Ulmus americana

Viburnum edule

Viburnum rafinesquianum

Viburnum trilobum

Viola adunca

Viola canadensis

Viola nephrophylla

Viola renifolia

Vicia americana

154

Appendix C: Pre – harvest frequency and average % cover of vascular and non-vascular species which occurred in at least 3 of either control (C) or experimental (E) blocks, among four separate forest vegetation strata.

Pre-harvest: Treatment: Block: Number of plots (N):	Frequency — Harvest Blocks							Frequency — Control Blocks							Average (%) Cover — Harvest Blocks							Average (%) Cover — Control Blocks						
	E1	E2S	E2N	E3S	E3N	E4S	E4N	C1	C2S	C2N	C3S	C3N	C4S	C4N	E1	E2S	E2N	E3S	E3N	E4S	E4N	C1	C2S	C2N	C3S	C3N	C4S	C4N
	10	33	38	21	20	21	23	17	26	31	17	19	15	15	10	33	38	21	20	21	23	17	26	31	17	19	15	15
Ground-1meter																												
Forbs																												
Aquelegia canadaensis	.	1	1	.	1	1	1	1	1	.	.	.	.	.	.	0.08	0.07	.	0.75	0.12	0.11	0.15	0.1	.	.	.	.	.
Arallia nudicaulus	2	8	5	8	4	7	9	2	4	10	9	12	7	6	1.75	3.94	1.32	2.74	2.38	9.4	6.74	1.03	0.87	4.68	6.32	13	6.83	5
Aster ciliolatus	4	1	3	.	2	2	2	.	.	1	5	5	2	1	6	0.08	0.2	.	0.25	0.24	0.76	.	.	0.08	0.74	0.66	1.17	0.17
Circea alpina	1	2	2	.	4	.	.	2	.	5	2	.	.	2	1.5	1.59	0.13	.	0.5	.	.	0.29	.	1.53	4.41	.	.	1.17
Cornus canadensis	4	5	16	4	11	11	14	5	7	14	13	7	6	9	6	0.76	3.95	1.07	5.63	6.07	12.9	1.47	2.12	6.13	7.65	7.24	5	4
Corylus cornuta	.	.	.	1	3	1	.	.	1	1	2	.	.	.	.	.	.	0.12	6.25	1.79	.	.	0.58	0.08	1.03	.	.	.
Diervilla lonicera	.	1	1	.	1	.	.	1	.	.	1	.	3	2	.	0.45	0.99	.	0.13	.	.	0.15	.	.	0.88	.	4.5	6.67
Dryopteris sp.	.	1	1	2	.	1	1	1	1	2	1	.	1	.	.	0.45	0.99	1.9	.	0.71	0.65	0.88	1.44	0.56	0.15	.	0.17	.
Equisetum arvense	.	.	.	.	1	2	1	.	1	2	.	.	2	.	.	.	.	.	0.13	0.24	0.11	.	0.1	0.16	.	.	0.33	.
Equisetum sylvanicum	.	.	.	.	.	1	3	2	.	.	1	1	.	2	.	.	.	.	.	0.12	0.33	0.29	.	.	0.15	0.13	.	2
Fragaria virginiana	5	7	18	7	11	12	13	3	7	9	10	12	5	5	5.05	0.91	1.84	1.43	2.63	4.88	3.59	0.44	1.15	1.13	2.21	2.89	0.83	0.83
Gallium borealis	.	.	5	.	6	5	3	.	2	2	3	10	2	1	.	.	0.33	.	0.75	1.19	0.33	.	0.19	0.16	0.44	1.32	0.33	0.17
Gallium triflorum	.	3	5	4	2	6	7	1	1	5	3	1	.	2	.	0.23	0.33	0.48	0.88	0.71	0.76	0.15	0.1	0.81	0.44	0.13	.	0.33
Goodyera repens	2	.	.	.	.	.	.	.	2	1	1	.	.	1	3	.	.	.	.	.	.	.	0.19	0.08	0.15	.	.	0.17
Gymnocarpium sp.	.	.	1	.	.	.	1	1	.	1	.	.	3	1	.	.	0.07	.	.	.	0.11	0.15	.	0.08	.	.	0.5	0.17
Lathyrus ochroleucus	.	.	.	.	.	.	3	1	.	.	1	1	1	.	.	.	.	.	.	.	0.33	0.15	.	.	0.88	0.79	0.17	.
Linnaea borealis	2	5	10	5	5	3	3	4	5	3	4	.	6	3	3	0.76	1.32	0.6	1.25	2.62	1.85	1.32	0.48	0.65	1.32	.	2.67	2.83
Malanthemum canadense	3	8	12	7	5	12	8	3	6	10	7	10	7	5	4.5	0.61	0.79	0.83	0.63	2.02	0.87	0.44	0.58	1.21	1.03	1.97	2	0.83
Mertenzia paniculata	.	4	16	3	3	4	6	1	2	9	4	5	5	2	.	1.36	4.54	1	4.5	3.93	5.33	0.15	1.15	2.74	3.38	1.97	1.67	1.17
Mitella nuda	7	15	17	9	6	4	7	10	19	16	7	2	8	6	8	2.27	3.75	3.93	4.88	1.67	3.37	5.59	8.85	2.17	6.62	0.26	6.83	5
Petasites palmatus	2	1	6	1	.	3	6	.	1	4	4	6	3	1	3	0.45	0.72	0.12	.	1.55	1.74	.	0.1	0.73	0.59	1.45	1.33	0.17
Pyrola asarifolia	5	6	6	5	6	5	4	5	2	5	8	3	4	4	7.5	0.45	0.72	1.19	1.38	1.19	0.43	0.74	0.19	0.81	2.65	0.39	0.67	1.5
Pyrola chlorantha	.	.	.	2	1	.	.	1	1	2	.	.	.	1	.	.	.	0.24	0.13	.	.	0.15	0.58	1.29	.	.	.	0.17
Pyrola secunda	3	3	5	4	4	1	.	.	2	5	2	.	.	2	4.5	0.23	0.33	0.48	0.5	0.12	.	.	0.19	0.4	1.03	.	.	1.17
Rubus pubescens	4	11	23	6	12	15	14	8	10	17	13	10	10	6	4.75	2.35	8.42	2.98	6.38	8.81	6.3	5.78	3.85	4.6	11	5.79	5.83	3.5
Streptopus roseus	.	1	5	.	2	3	1	.	8	6	2	2	1	.	.	0.08	0.66	.	0.88	0.95	0.65	.	0.77	1.29	0.29	0.92	0.17	.
Thalictrum dasycarpum	.	.	1	.	2	1	1	.	.	.	2	3	1	.	.	.	0.07	.	0.25	0.12	0.11	.	.	.	1.03	2.24	0.17	.
Trientalis borealis	3	4	9	3	1	4	6	2	4	6	2	1	2	1	4.5	0.3	0.59	0.36	0.13	0.48	0.65	0.29	0.38	0.89	1.03	0.13	1.17	0.17
Viola canadensis	.	1	3	.	2	.	3	.	.	2	.	2	.	.	.	0.08	1.12	.	0.25	.	2.39	.	.	0.16	.	0.26	.	.
Viola nephrophylla	.	.	.	.	.	.	.	1	.	1	1	.	2	2	.	.	.	.	.	.	.	0.15	.	0.08	0.15	.	0.33	0.33
Viola renifolia	2	8	3	5	1	1	4	1	10	9	3	2	2	1	3	0.98	0.2	0.6	0.13	0.12	0.98	0.15	1.92	1.13	0.44	0.26	0.33	0.17
Bryophytes																												
Brachythecium/Eurhynchium	7	30	24	18	10	9	9	4	18	23	10	5	6	11	14.8	22.7	14.3	29.2	13.5	17.2	15.8	17.5	15.5	19.2	19.7	3.95	2.67	11.3
Dicranum sp.	.	.	1	.	.	1	.	.	1	1	1	.	2	.	.	.	0.07	.	.	0.12	.	.	0.1	0.08	0.88	.	0.33	.
Hylocomium splendens	2	3	5	8	3	1	5	1	1	.	3	1	5	4	1.75	0.23	6.84	8.33	5.75	1.79	10.4	0.15	3.27	.	5.29	0.79	3.33	6.33
Mnium\plagiomnium	8	20	15	10	6	6	5	6	9	13	7	2	2	4	17.5	5.61	2.3	8.1	6.75	6.9	1.09	12.1	5.1	7.26	11.6	0.92	1.17	1.5
Pleurozium schreberi	.	4	8	9	3	3	2	5	3	6	5	1	4	4	.	2.12	2.11	4.52	3.38	0.36	0.76	5	2.12	4.68	5	0.79	8.67	0.67

Appendix C: continued...

Frequency columns (Harvest Blocks: E1, E2S, E2N, E3S, E3N, E4S, E4N, CI; Control Blocks: C2S, C2N, C3S, C3N, C4S, C4N); Average (%) Cover columns (same block order).

Pre-harvest Treatment Block — Number of plots (N)	| Frequency →	E1	E2S	E2N	E3S	E3N	E4S	E4N	CI	C2S	C2N	C3S	C3N	C4S	C4N	| Cover →	E1	E2S	E2N	E3S	E3N	E4S	E4N	CI	C2S	C2N	C3S	C3N	C4S	C4N
Number of plots (N)		10	33	38	21	20	21	23	17	26	31	17	19	15	15		10	33	38	21	20	21	23	17	26	31	17	19	15	15
Bryophytes continued...																														
Ptilium crista-castrensis		.	.	1	3	.	1	.	1	.	.	2	.	.	.		.	.	0.99	0.95	.	0.12	.	0.15	.	.	0.29	.	.	.
Rhytadiadelphus sp.		1	3	4	6	.	1	.	5	3	3	1	.	1	1		1.5	4.85	1.84	4.64	.	0.71	.	5.94	0.29	0.24	0.15	.	0.17	0.17
Thuidium recognitum		.	.	1	2	1	2	2	.	1	2	.	1	1	.		.	.	0.07	0.83	0.13	0.24	0.76	.	0.1	0.97	.	0.79	0.17	.
Graminoids																														
Oryzopsis asperifolia		.	.	1	.	2	5	9	.	.	.	.	4	2	4		.	.	0.07	.	0.25	2.26	1.52	.	.	.	.	0.53	0.33	0.67
Schizachne purpurescens		.	.	6	2	.	3	1	.	.	1	6	.	1	.		.	.	0.39	0.24	.	0.36	0.11	.	.	0.08	0.88	.	0.17	.
Tree species																														
Abies Balsamea		4	5	13	7	9	4	9	3	5	11	4	4	7	10		4.75	0.38	1.51	0.83	1.13	1.07	6.2	0.44	0.48	2.02	0.59	0.53	2	5.67
Betula papyrifera		.	1	.	.	1	1	1	.	.	.	2	.	.	1		.	0.08	.	.	0.13	0.71	0.11	.	.	.	1.03	.	.	1
Fraxinus nigra		.	1	6	.	3	9	7	5	3	4	7	3	1	4		.	0.08	2.96	.	2.13	1.67	2.83	1.47	0.29	0.73	1.76	0.39	1	2.33
Populus tremuloides		.	.	1	.	1	1	.	.	.	1	.	.	2	.		.	.	0.07	.	0.13	0.12	.	.	.	0.08	.	.	1.17	.
Shrub Species																														
Acer spicatum		3	14	15	3	2	.	.	10	19	12	1	.	2	3		4.5	3.94	5.72	0.95	0.88	.	.	7.79	13.8	6.37	0.15	.	1.17	0.5
Apocynum androsaimifolium*		.	.	.	.	.	1	1	.	.	.	2	.	.	.		.	.	.	.	.	0.12	0.65	.	.	.	2.11	.	.	.
Corylus cornuta		.	.	.	1	3	1	.	.	1	1	2	.	.	.		.	.	.	0.12	6.25	1.79	.	.	0.58	0.08	1.03	.	.	.
Diervilla lonicera		.	1	1	.	1	.	.	1	.	.	1	.	3	2		.	0.45	0.99	.	0.13	.	.	0.15	.	.	0.88	.	4.5	6.67
Prunus virginiana		.	.	1	.	.	1	.	.	1	1	1	3	1	.		.	.	0.39	.	.	0.12	.	.	0.1	1.21	0.15	1.05	0.17	.
Rhamnus alnifolia		.	.	1	1	.	.	1	.	.	2	2	.	.	.		.	.	0.39	0.12	.	.	1.63	.	.	0.29	3.95	.	.	.
Ribes oxyacanthoides		.	.	.	.	.	2	.	.	1	.	1	2	1	.		.	.	.	.	.	0.24	.	.	0.1	.	0.15	0.26	0.17	.
Ribes triste		.	2	5	1	2	4	.	4	3	7	2	.	2	3		.	0.15	1.32	0.12	0.25	0.48	.	1.32	0.29	1.37	1.03	.	1.17	1.33
Rosa acicularis		.	4	3	.	1	1	4	.	.	.	3	7	1	1		.	0.3	0.2	.	0.75	0.71	2.07	.	.	.	0.44	3.55	1	0.17
Rubus idaeus		.	.	.	2	3	4	.	.	2	2	1	.	2	.		.	.	.	2.63	0.95	3.04	.	.	.	0.56	2.35	0.79	.	2.67
Symphorocarpos albus		1	1	8	7	6	1	7	.	2	3	9	1	1	1		1.5	0.45	1.18	0.83	0.75	0.71	2.39	.	.	0.56	0.44	2.5	0.17	0.17
Viburnum edule		.	1	4	.	2	1	1	1	3	.	1	.	.	1		.	0.08	1.25	.	1.5	0.71	0.65	0.15	1.25	.	0.15	.	.	1
1-2 Meter Strata / Shrub Species																														
Acer spicatum		2	8	8	1	.	1	.	4	5	7	1	1	1	.		63.4	18.2	24.9	8.95	.	19	.	34.2	34.3	26.6	3.18	1.53	5.67	.
Corylus cornuta		.	.	1	.	4	1	1	.	.	2	3	9	.	.		.	.	0.53	.	13.6	2.29	1.3	.	.	5.21	5.94	47.4	.	.
Prunus virginiana		.	.	1	.	2	1	.	.	.	1	.	2	.	.		.	.	2.63	.	4.95	2.76	.	.	.	2.63	.	3.11	.	.
Rosa acicularis*		.	.	.	.	1	.	1	.	.	.	.	2	.	.		.	.	.	.	.	.	.	1	3.43	.	.	4.58	.	.
Viburnum rafinesquianum*		.	.	.	.	.	.	1	.	.	.	.	3	.	.		.	.	.	.	.	.	.	.	2.43	.	.	18.3	.	.
Tree Species																														
Abies balsamia		3	4	5	2	1	1	1	.	3	4	.	1	1	.		53.8	12.9	17.3	9.95	3.1	7.14	4.91	.	4.42	8.13	.	4.79	6.33	.
Fraxinus nigra		.	.	2	.	1	2	2	.	.	1	2	2	1	.		.	.	1.79	.	1.75	11.6	2.87	.	.	0.55	3.53	5.95	2	.

156

Appendix C: Continued ...

	Frequency														Average (%) Cover													
Pre-harvest: Treatment:	Harvest Blocks							Control Blocks							Harvest Blocks							Control Blocks						
Block:	E1	E2S	E2N	E3S	E3N	E4S	E4N	C1	C2S	C2N	C3S	C3N	C4S	C4N	E1	E2S	E2N	E3S	E3N	E4S	E4N	C1	C2S	C2N	C3S	C3N	C4S	C4N
Number of plots (N):	10	33	38	21	20	21	23	17	26	31	17	19	15	15	10	33	38	21	20	21	23	17	26	31	17	19	15	15
2-10 Meter stratum																												
Shrub species																												
Acer spicatum	1	5	8	.	.	.	.	.	9	10	7	.	1	1	32.2	42.4	62.9	.	.	.	.	.	127	122	51.8	.	11.5	8.67
*Amelanchier alnifolia**	.	.	.	.	.	.	.	.	.	.	2	.	.	.	.	.	.	.	.	.	.	.	.	.	8.63	.	.	.
*Corylus cornuta**	.	.	.	.	.	.	.	.	.	.	2	.	.	.	.	.	.	.	.	.	.	.	.	.	6.58	.	.	.
*Viburnum rafinesquianum**	.	.	.	.	.	.	.	.	.	.	1	.	.	.	.	.	.	.	.	.	.	.	.	.	11.1	.	.	.
Tree species																												
Abies balsamea	5	18	22	13	3	3	1	9	12	21	7	4	6	4	136	163	187	200	19.6	31.3	4.78	111	120	173	98.8	50.5	111	33.1
*Acer negundo**	.	.	.	.	.	.	1	.	.	.	.	1	.	.	.	.	.	.	.	.	10.9	.	.	.	.	4.11	.	.
Betula papyrifera	.	1	.	.	.	.	.	.	1	.	.	1	1	.	.	6.67	.	.	.	.	.	.	9.04	.	.	13.7	7.67	.
Fraxinus nigra	.	.	1	1	1	2	4	1	2	1	2	6	3	1	.	.	7.76	19.5	1.6	16	44	5.29	22.3	13.2	55.9	42.3	47.1	9.2
Populus tremuloides	1	.	1	.	.	2	1	3	.	.	.	2	.	1	21.1	.	0.79	.	.	6.67	1.78	29.1	.	.	.	15.4	.	6.8
*Ulmus americana**	.	.	.	.	.	.	1	.	.	.	.	.	.	.	.	.	.	.	.	.	8.22	.	.	.	.	.	.	.
> 10 meters																												
Abies balsamea	2	5	4	5	5	5	.	1	7	3	7	1	.	2	24.3	51.1	10.2	48.1	20.9	15.5	.	26.4	50.4	73	14	18.2	.	32.2
Fraxinus nigra	1	.	.	.	.	.	2	.	1	1	.	2	.	1	19.4	.	.	.	.	.	44.2	.	19.2	22.4	.	43.3	.	11.2
Picea glauca	5	8	5	5	3	8	9	2	7	6	5	3	5	4	121	160	138	166	213	106	89.5	156	165	97.5	143	72.1	102	134
Populus tremuloides	2	5	4	2	4	5	6	1	5	2	4	6	2	3	41.1	66.1	107	29.1	56.3	87.2	149	93.2	35.1	99	79.5	187	89.1	127

Appendix D: Post – harvest frequency and average % cover of vascular and non-vascular species which occurred in at least 3 of either control (C) or experimental (E) blocks, among four separate forest vegetation strata.

Post harvest: Treatment: Block: Number of plots (N):	Frequency — Harvest Blocks							Frequency — Control Blocks							Average (%) Cover — Harvest Blocks							Average (%) Cover — Control Blocks						
	E1	E2S	E2N	E3S	E3N	E4S	E4N	C1	C2S	C2N	C3S	C3N	C4S	C4N	E1	E2S	E2N	E3S	E3N	E4S	E4N	C1	C2S	C2N	C3S	C3N	C4S	C4N
(N)	10	33	38	21	20	21	23	17	26	31	17	19	14	15	10	33	38	21	20	21	23	17	26	31	17	19	14	15

Ground-1 meter

Forbs

Species	E1	E2S	E2N	E3S	E3N	E4S	E4N	C1	C2S	C2N	C3S	C3N	C4S	C4N	E1	E2S	E2N	E3S	E3N	E4S	E4N	C1	C2S	C2N	C3S	C3N	C4S	C4N
Actea rubra	·	·	·	·	·	1	1	·	2	·	1	2	1	·	·	·	·	·	·	0.11	0.71	·	0.19	·	0.15	0.26	0.18	·
Aquilegia canadensis	1	1	1	·	3	1	·	·	·	1	·	2	1	·	0.25	0.07	0.08	1	·	·	0.12	·	·	0.08	·	0.26	0.18	·
Aralia nudicaulis	·	9	11	5	2	5	6	·	7	11	9	10	11	5	·	2.04	1.44	0.88	1.19	1.2	3.45	·	1.15	5.16	9.71	8.82	11.4	4
Aster ciliolatus	·	6	8	2	4	4	6	·	2	·	3	5	1	·	·	1.51	1.21	0.5	0.83	0.65	0.48	·	0.19	·	0.44	0.66	0.18	·
Circea alpina	1	2	4	·	3	·	1	2	4	5	3	·	·	2	0.25	0.59	0.15	2.13	·	0.11	·	0.29	0.38	0.4	3.24	·	·	0.33
Cirsium arvense	·	3	4	7	2	1	1	·	·	·	·	1	·	·	·	0.92	0.61	0.25	0.83	0.11	0.12	·	·	·	·	0.13	·	·
Cornus canadensis	1	9	11	3	11	13	15	6	9	12	10	7	6	8	0.25	1.38	1.44	2.63	0.36	3.8	3.93	1.62	1.35	3.71	5	5.26	5.36	3
Corydalis aureus	·	3	2	1	·	·	·	·	·	·	·	·	·	·	·	0.13	0.23	·	0.12	·	·	·	·	·	·	·	·	·
Corylus cornuta	·	1	2	2	2	·	3	3	·	1	2	1	·	·	·	0.46	0.45	0.88	0.83	5	·	0.44	·	0.48	3.09	0.13	·	·
Dryopteris sp.	·	·	·	1	1	1	·	·	1	1	1	·	1	·	·	·	·	0.75	0.12	·	0.12	·	0.1	0.08	0.15	·	0.18	·
Epilobium angustifolium	·	·	2	1	·	1	·	·	·	·	1	2	1	·	·	0.13	·	·	0.12	·	0.12	·	·	·	0.15	0.26	1.07	·
Equisetum arvense	·	1	1	·	·	·	·	·	2	2	·	1	·	2	·	0.07	0.08	·	·	·	·	·	0.19	0.16	·	0.13	·	0.33
Equisetum pratense	1	1	·	·	1	2	·	·	3	·	3	·	1	1	0.25	·	0.08	0.13	·	·	0.24	·	0.29	·	0.44	·	0.18	0.17
Equisetum sylvanicum	·	·	1	·	·	1	·	1	1	1	·	·	·	3	·	0.07	·	0.13	·	0.11	·	0.15	0.1	·	·	·	·	0.5
Fragaria virginiana	1	7	11	7	11	11	14	4	4	9	10	6	7	·	0.25	1.71	0.53	2.63	1.43	5.33	4.76	0.59	0.38	0.32	1.32	2.63	1.07	1.17
Gallium borealis	·	2	4	3	8	6	11	·	2	4	5	5	1	3	·	0.59	0.15	1	0.36	1.25	0.71	·	0.19	0.32	0.74	0.66	0.18	0.5
Gallium triflorum	2	6	9	3	3	6	8	1	·	4	4	6	3	2	5.25	0.92	0.83	1	0.36	1.96	1.31	0.15	·	0.32	0.59	0.79	0.54	0.33
Gymnocarpium sp.	·	·	3	·	·	·	·	1	2	·	·	·	3	1	·	0.2	·	·	·	·	·	0.15	0.19	·	·	·	1.43	0.17
Lathyrus ochroleucus	·	3	3	1	6	1	2	·	·	·	1	4	·	·	·	0.2	0.23	2.5	0.12	0.22	0.71	·	·	·	0.15	1.18	·	·
Linnaea borealis	1	4	8	2	5	1	4	4	4	4	4	1	4	2	0.25	0.53	0.68	0.63	0.24	0.43	0.71	1.32	0.87	0.32	1.32	0.13	0.71	2.67
Maianthemum canadense	4	7	6	7	3	10	11	5	7	12	7	12	10	5	1	0.39	0.53	0.38	0.83	1.2	1.79	0.74	0.67	0.97	1.03	2.24	1.79	0.83
Mertensia paniculata	·	9	10	5	4	6	3	2	6	6	3	6	1	4	·	1.32	4.02	1.75	1.79	1.41	1.31	0.29	0.58	1.29	1.91	3.42	0.18	0.67
Mitella nuda	4	10	9	6	5	2	7	10	19	18	6	5	7	8	1	0.59	0.76	1.25	0.71	2.83	0.24	4.26	6.92	2.26	2.35	2.63	4.64	6.17
Petasites palmatus	·	3	4	·	·	8	4	2	·	4	4	6	5	1	·	0.26	0.61	·	·	0.43	0.95	0.29	·	0.32	1.32	2.11	0.89	0.17
Potentilla norwegica	·	3	·	1	·	2	·	·	·	·	·	·	·	·	·	·	0.23	·	0.12	·	0.24	·	·	·	·	·	·	·
Pyrola asarifolia	2	1	6	1	4	2	1	4	7	7	7	3	3	4	0.5	0.39	0.08	0.5	0.12	0.11	0.24	0.59	0.67	0.97	1.03	0.39	0.54	0.67
Pyrola chlorantha	·	·	·	2	·	·	·	·	1	1	1	·	·	1	·	·	·	·	·	·	0.24	·	·	0.58	0.48	0.15	·	0.17
Pyrola secunda	1	2	1	1	2	·	·	·	2	6	2	·	·	2	0.25	0.07	0.15	0.25	0.12	·	·	·	0.19	0.48	0.31	·	·	1.17
Rubus pubescens	6	14	19	7	13	15	15	6	14	16	12	9	9	8	7.5	5.33	4.7	4.75	2.02	12.2	11	1.62	3.75	2.1	7.5	4.34	6.96	3.83
Sonchus sp.	2	15	9	·	9	·	·	·	·	·	·	·	·	·	5.25	2.83	6.59	·	1.75	·	·	·	·	·	·	·	·	·
Streptopus roseus	·	1	1	3	3	5	6	·	8	6	1	3	2	3	·	0.07	0.08	0.38	0.36	1.2	1.79	·	0.77	0.89	0.15	0.39	0.36	1.33
Thalictrum dasycarpum	·	1	·	·	2	3	7	·	·	1	1	7	·	·	·	·	0.08	0.25	·	1.3	0.95	·	·	0.08	0.88	3.42	·	·
Trientalis borealis	2	1	2	2	3	3	3	1	6	4	5	2	5	5	0.5	0.13	0.08	0.38	0.24	0.33	0.36	0.15	0.58	0.32	0.74	0.26	0.89	0.83
Viola canadensis	·	·	1	1	2	1	2	·	·	3	·	4	·	·	·	·	0.39	·	0.25	0.12	0.76	0.12	·	0.24	·	1.18	·	·
Viola renifolia	2	4	6	4	1	3	4	6	9	10	3	2	3	4	0.5	0.39	0.3	0.13	0.48	0.43	0.36	1.62	0.87	0.81	1.18	0.26	0.54	0.67
Vicia americana	·	·	1	1	1	1	2	·	·	2	1	1	·	·	·	0.39	·	0.13	0.12	0.22	0.12	·	·	0.16	0.15	0.13	·	·

Appendix D: Continued...

Post harvest:
Treatment:

Frequency

Species	Harvest Blocks E1	E2S	E2N	E3S	E3N	E4S	E4N	Control Blocks C1	C2S	C2N	C3S	C3N	C4S	C4N
Number of plots (N):	10	33	38	21	20	21	23	17	26	31	17	19	14	15
Graminoids														
Bromus ciliatus	.	.	.	1	2	.	2	.	1	.	1	2	.	.
Carex sp.	1	6	4	.	9	.	11	4	10	10	8	7	.	7
Cinna latifolia	.	1	10	.	.	12	2	1	3	.	1	.	9	7
Oryzopsis asperifolia	.	3	3	4	6	3	8	1	2	6	7	12	2	3
Bryophytes														
Brachythecium sp. / Eurhynchium sp.	2	5	2	5	4	4	6	10	19	18	12	9	4	9
Climacium dendroides	.	1	.	1	.	3	1	1	1	.	.	1	.	1
Hylocomium splendens	1	.	3	1	.	.	.	1	2	.	2	1	5	3
Mnium sp. / Plagiomnium sp.	1	.	1	7	1	5	2	7	11	11	3	2	3	3
Pleurozium schreberi	.	.	1	3	.	1	1	3	3	2	7	.	3	3
Ptilium crista-castrensis	.	.	.	.	.	.	.	.	1	.	1	.	4	2
Rhytidiadelphus sp.	.	3	.	1	.	.	.	5	4	.	.	.	.	2
Tree Species														
Abies balsamea	1	3	.	3	4	.	2	2	1	5	5	6	5	6
Betula papyrifera	1	.	3	3	.	5	2	.	1	1	.	2	2	2
Fraxinus nigra	2	6	7	.	1	5	7	4	7	7	7	4	2	4
Populus tremuloides	3	3	3	2	5	3	2	1	1	1	1	1	.	1
Shrub Species														
Acer spicatum	7	17	15	1	1	.	3	3	14	13	2	.	2	3
Amelanchier alifolia	.	.	.	.	2	3	.	1	.	.	4	2	.	.
Apocynum androsaemifolium	.	.	.	.	.	.	.	.	4	.	4	3	.	.
Diervilla lonicera	.	2	2	2	2	4	3	.	.	.	.	.	3	2
Ribes triste	2	4	11	7	1	2	2	1	.	.	.	4	2	3
Rosa acicularis	.	1	5	2	2	6	3	1	1	.	2	8	3	2
Rubus idaeus	2	7	6	.	4	1	1	.	1	1	6	1	.	2
Symphoricarpos albus	.	2	5	1	5	5	5	1	.	1	.	9	.	2
Viburnum edule	1	1	1	.	.	1	1	1	.	2	.	.	1	2
1-2 meter stratum — Shrub species														
Acer spicatum	3	1	5	.	.	.	4	4	5	7	1	1	1	.
Corylus cornuta	.	1	1	.	4	2	.	.	.	2	3	9	.	.
Rosa acicularis	.	.	.	.	.	.	.	.	.	.	.	2	.	.
Viburnum rafinesquianum	.	.	.	1	.	1	.	.	.	.	.	3	.	.

Average (%) Cover

Species	Harvest Blocks E1	E2S	E2N	E3S	E3N	E4S	E4N	Control Blocks C1	C2S	C2N	C3S	C3N	C4S	C4N
Number of plots (N):	10	33	38	21	20	21	23	17	26	31	17	19	14	15
Graminoids														
Bromus ciliatus	.	0.59	.	.	0.12	0.22	.	.	0.1	.	.	0.26	.	.
Carex sp.	0.25	1.32	0.45	0.25	0.95	3.37	2.02	4.12	2.79	1.61	2.65	1.58	3.39	1.17
Cinna latifolia	.	.	0.08	.	.	0.22	.	.	0.77	.	0.15	0.15	.	.
Oryzopsis asperifolia	.	0.53	0.23	0.75	0.48	1.96	0.36	0.15	0.19	0.48	1.03	1.58	0.36	0.5
Bryophytes														
Brachythecium sp. / Eurhynchium sp.	0.5	0.46	3.26	1.75	11.3	1.74	1.07	19.7	14.7	16.9	19.3	0.97	.	11.2
Climacium dendroides	.	.	0.08	.	0.12	0.11	0.36	0.15	.	.	.	.	.	.
Hylocomium splendens	1.5	0.53	.	.	2.98	.	0.12	5	0.1	.	0.29	0.13	.	2.5
Mnium sp. / Plagiomnium sp.	.	0.07	.	0.75	2.13	2.28	6.31	1.76	1.54	12	10.1	0.79	8.57	4.5
Pleurozium schreberi	.	0.99	.	.	1.55	0.11	0.12	2.66	2.98	0.16	5.88	0.92	0.54	1.33
Ptilium crista-castrensis	.	.	.	.	.	.	.	.	3.08	.	0.88	0.13	4.29	2
Rhytidiadelphus sp.	.	.	0.98	.	0.12	.	.	5	2.69	.	0.15	.	.	0.17
Tree Species														
Abies balsamea	.	0.07	.	.	0.48	0.33	0.24	0.29	0.1	0.4	0.74	1.45	1.79	6.5
Betula papyrifera	0.25	0.2	.	0.38	0.36	0.76	.	0.15	.	0.08	.	.	.	0.33
Fraxinus nigra	0.5	1.45	1.21	0.75	.	1.85	1.19	1.32	0.67	0.56	1.03	1.18	0.36	2.33
Populus tremuloides	4.5	0.53	0.61	1.25	0.24	0.76	1.55	0.15	0.1	0.08	0.88	0.13	.	0.17
Shrub Species														
Acer spicatum	13.5	3.88	5.61	0.13	0.12	.	1.79	1.18	5.96	1.85	2.35	.	2.14	1.33
Amelanchier alifolia	.	.	.	0.88	.	0.76	.	0.15	.	.	0.15	0.26	.	.
Apocynum androsaemifolium	.	.	.	.	.	.	.	.	.	.	2.06	2.24	.	.
Diervilla lonicera	.	1.05	0.53	2.63	0.83	2.93	5.12	.	.	0.32	.	.	2.32	3.5
Ribes triste	0.5	1.05	0.31	0.13	0.83	0.76	0.83	0.15	0.87	.	.	0.53	2.86	1.33
Rosa acicularis	.	.	0.08	0.88	0.24	0.33	1.9	0.15	.	.	1.32	3.03	2.32	0.17
Rubus idaeus	0.5	1.32	0.91	1.75	.	3.59	0.12	.	.	0.08	1.03	0.13	.	1.17
Symphoricarpos albus	0.25	1.64	0.15	0.63	0.12	1.63	0.6	0.15	.	0.08	0.88	1.84	.	0.33
Viburnum edule	0.25	0.33	0.08	.	.	.	1.79	0.13	0.38	0.16	.	.	0.18	1.17
1-2 meter stratum — Shrub species														
Acer spicatum	15.2	.	0.97	.	.	.	15.2	34.2	34.3	26.6	3.18	1.53	5.67	.
Corylus cornuta	.	3.55	0.55	6.9	6.19	8.22	4.86	.	.	5.21	5.94	47.4	.	.
Rosa acicularis	.	.	.	.	.	.	.	.	.	.	.	4.58	.	.
Viburnum rafinesquianum	.	.	.	.	.	.	2.22	.	.	.	.	18.3	.	.

Appendix D: Continued...

Post harvest:
Treatment:

Frequency

	Harvest Blocks							Control Blocks						
Block:	E1	E2S	E2N	E3S	E3N	E4S	E4N	C1	C2S	C2N	C3S	C3N	C4S	C4N
Number of plots (N):	10	33	38	21	20	21	23	17	26	31	17	19	14	15
Tree species														
Abies balsamea	.	.	1	1	.	.	.	.	3	4	.	1	1	1
Fraxinus nigra	.	1	2	1	.	.	3	.	.	1	2	2	1	.
Populus tremuloides	.	.	4	.	3	2	.	.	.	.	.	.	.	1
2-10 Meter stratum														
Abies balsamea	.	6	4	8	5	5	3	9	12	21	7	4	6	4
Acer negundo	.	.	.	.	.	.	1	.	.	.	.	2	.	.
Acer spicatum	1	2	2	.	.	1	2	9	10	7	.	.	1	1
Amelanchier alnifolia	.	.	.	1	.	.	.	.	1	.	.	2	1	.
Betula papyrifera	.	.	.	1	.	.	.	.	1	.	.	1	.	.
Corylus cornuta	.	.	.	.	.	.	.	.	.	.	.	2	.	.
Fraxinus nigra	.	.	1	1	.	2	3	1	2	1	2	6	3	1
Populus tremuloides	.	.	.	.	.	1	1	3	.	.	.	2	.	1
Ulmus americana	.	.	.	.	.	1	1	.	.	.	.	.	.	.
Viburnum rafinesquianum	.	.	.	.	.	.	.	.	.	.	.	1	.	.
>10m stratum														
Abies balsamea	1	2	2	3	2	1	1	.	7	3	7	1	.	2
Fraxinus nigra	1	2	2	.	.	.	.	1	1	.	2	2	.	1
Picea glauca	1	1	2	2	.	1	.	2	7	6	5	3	5	4
Populus tremuloides	2	3	3	4	.	4	7	1	5	2	4	6	2	3

Average (%) Cover

	Harvest Blocks							Control Blocks						
Block:	E1	E2S	E2N	E3S	E3N	E4S	E4N	C1	C2S	C2N	C3S	C3N	C4S	C4N
Number of plots (N):	10	33	38	21	20	21	23	17	26	31	17	19	14	15
Tree species														
Abies balsamea	.	.	.	.	.	.	.	.	.	.	.	.	.	.
Fraxinus nigra	.	2.16	.	.	3.48	.	.	.	4.42	8.13	.	4.79	6.33	.
Populus tremuloides	.	2.79	0.58	1.95	1.81	4.3	.	.	.	0.55	3.53	5.95	2	.
		2.47	.	2.1	.	.	4.05	.	.	.	.	.	.	0.27
2-10 Meter stratum														
Abies balsamea	.	69.1	17.2	84.1	56.2	66.7	11	111	120	173	98.8	50.5	111	33.1
Acer negundo	.	.	.	.	.	.	13	.	.	.	.	4.11	.	.
Acer spicatum	4.1	2.82	9.39	.	.	4.81	7.04	127	122	51.8	.	.	11.5	8.67
Amelanchier alnifolia	.	.	.	.	.	.	.	.	.	.	.	8.63	.	.
Betula papyrifera	.	.	.	2.86	.	.	.	.	9.04	.	.	13.7	.	.
Corylus cornuta	.	.	.	.	.	.	.	.	.	.	.	6.58	.	.
Fraxinus nigra	.	.	.	1.19	4.24	12.9	12.5	5.29	22.3	13.2	55.9	42.3	47.1	9.2
Populus tremuloides	.	.	.	.	.	0.48	2.17	29.1	.	.	.	15.4	.	6.8
Ulmus americana	.	.	.	.	.	.	6.96	.	.	.	.	.	.	.
Viburnum rafinesquianum	.	.	.	.	.	.	.	.	.	.	.	11.1	.	.
>10m stratum														
Abies balsamea	1.9	8.2	14.1	12.3	16.1	0.9	3.3	26.4	50.4	73	14	18.2	.	32.2
Fraxinus nigra	12.3	28.0	32.1	.	.	.	.	.	19.2	22.4	.	43.3	.	11.2
Picea glauca	1.31	25.8	23.6	11.3	7.5	13.2	.	156	165	97.5	143	72.1	102	134
Populus tremuloides	42.5	20.0	42.43	.	56.4	76.2	178.1	93.2	35.1	99	79.5	187	89.1	127

Appendix E: Pre- and post-harvest frequency and average % cover of species that demonstrated either an increase or a decrease in frequency in at least three harvest blocks for an overall change of no less than 25%. Species marked with '*' showed smaller changes but may be of interest.

	Frequency (pre-harvest)							Frequency (post-harvest)							% Cover (pre-harvest)							% Cover (post-harvest)							
Block:	El	E2S	E2N	E3S	E3N	E4S	E4N	El	E2S	E2N	E3S	E3N	E4S	E4N	El	E2S	E2N	E3S	E3N	E4S	E4N	El	E2S	E2N	E3S	E3N	E4S	E4N	Performance
Number of Plots (N):	10	33	38	21	20	21	23	10	33	38	21	20	21	23	10	33	38	21	20	21	23	10	33	38	21	20	21	23	
Ground-1 m Stratum																													
Forbs																													
Aster ciliolatus	4	1	3	.	2	2	2	.	6	8	2	4	4	6	6	0.1	0.2	.	0.3	0.2	0.8	.	1.2	1.5	0.8	0.5	0.5	0.7	I
Achillea millefolium*	.	.	.	.	.	.	.	.	.	.	.	.	1	.	.	.	.	.	.	.	.	.	.	.	.	.	0.1	.	I*
Atriplex patula*	.	.	.	.	.	.	.	.	.	.	.	.	.	1	.	.	.	.	.	.	.	.	.	.	.	.	.	0.1	I*
Cerastium sp. *	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	0.1	I*
Circium arvense	.	.	.	.	.	.	.	.	3	4	7	2	1	1	.	.	.	.	.	.	.	.	0.9	0.6	0.3	0.8	0.1	0.1	I
Corydalis aureus	.	.	.	.	.	.	.	.	3	2	1	.	.	.	.	.	.	.	.	.	.	.	0.1	0.2	.	0.1	.	.	I
Epilobium Angustifolium	.	.	.	.	.	.	.	.	.	2	1	.	1	.	.	.	.	.	.	.	.	.	0.1	.	.	0.1	.	0.1	I
Equisetum pratense	.	.	.	.	.	.	.	1	1	.	.	1	2	.	.	.	.	.	.	.	.	0.3	.	0.1	0.1	.	.	0.2	I
Erigeron canadensis*	.	.	.	.	.	.	.	.	.	.	.	1	.	.	.	.	.	.	.	.	.	.	.	.	.	0.1	.	.	I*
Gallium borealis	.	.	5	.	6	5	3	.	2	4	3	8	6	11	.	.	0.3	.	0.8	1.2	0.3	.	0.6	0.2	1	0.4	1.3	0.7	I
Geranium bicknellii*	.	.	.	.	.	.	.	.	.	1	.	.	.	.	.	.	.	.	.	.	.	.	.	0.4	.	.	.	.	I*
Heuchera richardsonii*	.	.	.	.	.	.	.	.	.	.	.	1	.	.	.	.	.	.	.	.	.	.	.	.	.	.	0.1	.	I*
Hieracium sp.*	.	.	.	.	.	.	.	.	.	1	1	.	.	.	.	.	.	.	.	.	.	.	.	0.1	0.1	.	.	.	I*
Impatiens capensis	.	.	.	.	.	.	.	.	1	.	.	1	.	3	.	.	.	.	.	.	.	.	0.1	.	.	0.1	.	0.3	I
Lathyrus ochroleucus	.	.	.	.	.	.	3	.	3	3	1	6	1	2	.	.	.	.	.	.	0.3	.	0.2	0.2	2.5	0.1	0.2	0.7	I
Linnaea borealis	2	5	10	5	5	3	3	1	4	8	2	5	1	4	3	0.8	1.3	0.6	1.3	2.6	1.8	0.3	0.7	0.5	0.2	0.6	0.7	0.4	D
Mitella nuda	7	15	17	9	6	4	7	4	10	9	6	5	2	7	8	2.3	3.8	3.9	4.9	1.7	3.4	1	0.8	0.6	0.7	1.3	0.2	2.8	D
Plantago major*	.	.	.	.	.	.	.	.	.	.	.	.	1	1	.	.	.	.	.	.	.	.	.	.	.	.	0.1	0.1	I*
Polygonum cilinode	.	.	.	.	.	.	.	.	.	.	1	.	.	.	.	.	.	.	.	.	.	.	.	.	0.1	.	.	.	I*
Pyrola asarifolia	5	6	6	5	6	5	4	2	1	6	1	4	2	1	7.5	0.5	0.7	1.2	1.4	1.2	0.4	0.5	0.4	0.1	0.5	0.1	0.1	0.2	D
Pyrola secunda	3	3	5	4	4	1	.	1	2	1	1	2	.	.	4.5	0.2	0.3	0.5	0.5	0.1	.	0.3	0.1	0.2	0.3	0.1	.	.	D
Solidago sp.	.	.	.	.	.	.	.	1	1	1	.	.	.	2	.	.	.	.	.	.	.	0.3	0.1	0.1	.	.	.	0.2	I
Sonchus arvensis	.	.	.	.	.	1	.	2	15	9	.	9	.	.	.	.	.	.	.	.	0.1	5.3	2.8	6.6	1.8	.	.	.	I
Thalictrum dasycarpum	.	.	1	.	2	1	1	.	1	.	.	2	3	7	.	.	0.1	.	0.3	0.1	0.1	.	.	0.1	0.3	.	1.3	1	I
Thalictrum venulosum*	.	.	.	.	.	.	.	.	.	.	.	1	.	.	.	.	.	.	.	.	.	.	.	.	.	0.8	.	.	I*
Trientalis borealis	3	4	9	3	1	4	6	2	1	2	2	3	3	3	4.5	0.3	0.6	0.4	0.1	0.5	0.7	0.5	0.1	0.1	0.4	0.2	0.3	0.4	D
Viola adunca	-	-	-	-	-	-	-	-	2	-	-	-	-	-	-	-	-	-	-	-	-	-	0.2	-	-	-	-	-	I*
Bryophytes/lichens																													
Brachythecium sp.\	7	30	24	18	10	9	9	2	5	2	5	4	4	6	8	23	15	29	14	17	16	0.5	0.5	3.3	1.8	11	1.7	1.1	D
Eurhynchium sp.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	
Cladonia sp.	.	.	1	.	1	.	2	.	.	.	.	.	.	1	.	.	0.1	.	0.1	.	0.2	.	.	.	.	.	.	0.1	D
Climacium dendroides	.	2	.	.	.	.	.	.	1	.	1	.	3	1	.	0.5	.	.	.	.	.	.	0.1	.	0.1	0.1	.	0.4	I

161

Appendix F: Species list of birds encountered in the study area.

Common Name	Scientific Name	Common Name	Scientific Name
Magnolia Warbler	*Dendroica magnolia*	Mallard	*Anas platyrhynchos*
Least Flycatcher	*Empidonax minimus*	Mourning Dove	*Zenaida macroura*
Hooded Merganser	*Lophodytes cucullatus*	Mourning Warbler	*Oporornis philadelphia*
Hermit Thrush	*Catharus guttatus*	Nashville Warbler	*Vermivora ruficapilla*
Hairy Woodpecker	*Picoides villosus*	Northern Flicker	*Colaptes auratus*
Great Horned Owl	*Bubo virginianus*	Northern Oriole	*Icterus galbula*
Great-crested Flycatcher	*Myiarchus crinitus*	Northern Parula Warbler	*Parula americana*
Gray Jay	*Perisoreus canadensis*	Northern Raven	*Corvus corax*
Golden-Crowned Kinglet	*Regulus satrapa*	Three-toed Woodpecker	*Picoides tridactylus*
Evening Grosbeak	*Coccothraustes vespertinus*	Northern Waterthrush	*Seiurus noveboracensis*
Eastern Phoebe	*Sayornis phoebe*	Olive-sided Flycatcher	*Contopus borealis*
Eastern Wood-Pewee	*Contopus virens*	Ovenbird	*Seiurus aurocapillus*
Downy Woodpecker	*Picoides pubescens*	Palm Warbler	*Dendroica palmarum*
Dark-eyed Junco	*Junco hyemalis*	Philadelphia vireo	*Vireo philadelphicus*
Common Yellowthroat	*Geothlypis trichas*	Pied-Billed Grebe	*Podilymbus podiceps*
Common Snipe	*Gallinago gallinago*	Pileated Woodpecker	*Dryocopus pileatus*
Common Merganser	*Mergus merganser*	Pine Siskin	*Carduelis pinus*
Cliff Swallow	*Hirundo pyrrhonota*	Purple Finch	*Carpodacus purpureus*
Chipping Sparrow	*Spizella passerina*	Red-breasted Nuthatch	*Sitta canadensis*
Chestnut-Sided Warbler	*Dendroica pensylvanica*	Red-eyed Vireo	*Vireo olivaceous*
Cedar Waxwing	*Bombycilla cedrorum*	Rose-breasted Grosbeak	*Pheucticus ludovicianus*
Cape May Warbler	*Dendroica tigrina*	Ruby-Crowned Kinglet	*Regulus calendula*
Canada Warbler	*Wilsonia canadensis*	Ruby-throated Hummingbird	*Archilochus colubris*
Canada Goose	*Branta canadensis*	Ruffed Grouse	*Bonasa umbellus*
Brown Creeper	*Certhia americana*	Sandhill Crane	*Grus canadensis*
Boreal Chickadee	*Parus hudsonicus*	Sharp-shinned hawk	*Accipiter striatus*
Blue Jay	*Cyanocitta cristata*	Solitary Sandpiper	*Tringa solitaria*
Blue-headed Vireo	*Vireo solitarius*	Song Sparrow	*Melospiza melodia*
Black-throated Green Warbler	*Dendroica virens*	Spotted Sandpiper	*Actitis macularia*
Black-Capped Chickadee	*Parus atricapillus*	Swainson's Thrush	*Catharus ustulatus*
Blackburnian Warbler	*Dendroica fusca*	Swamp Sparrow	*Melospiza georgiana*
Black-backed Woodpecker	*Picoides arcticus*	Tennessee Warbler	*Vermivora peregrina*
Black and White Warbler	*Mniotilta varia*	Veery	*Catharus fuscescens*
Belted Kingfisher	*Ceryle alcyon*	Warbling Vireo	*Vireo gilvus*
Bay-Breasted Warbler	*Dendroica castanea*	Whip-poor-Will	*Caprimulgus vociferus*
Bald Eagle	*Haliaeetus leucocephalus*	White-throated Sparrow	*Zonotrichia albicollis*
American Robin	*Turdus migratorius*	White-winged Crossbill	*Loxia leucoptera*
American Redstart	*Setophaga ruticilla*	Winter Wren	*Troglodytes troglodytes*
American Goldfinch	*Carduelis tristis*	Wood Duck	*Aix sponsa*
American Crow	*Corvus brachyrhynchos*	Yellow-bellied Sapsucker	*Sphyrapicus varius*
Alder Flycatcher	*Empidonax alnorum*	Yellow-rumped Warbler	*Dendroica coronata*

Appendix G: Relative abundance values for 60 bird species for each year in both experimental and control blocks. Values are given as a percentage of the total abundance of all species combined within either control or experimental blocks, for either the pre- or post-harvest year. Individuals in transit or waterfowl including geese, ducks, mergansers and grebes are not included.

Treatment Year Species	Control Pre-Harv.	Exp. Pre-Harv.	Control Post-Harv.	Exp. Post-Harv.	Treatment Year Species	Control Pre-Harv.	Exp. Pre-Harv.	Control Post-Harv.	Exp. Post-Harv.
Alder Flycatcher	0.34	0.61	0.34	0.00	Mourning Warbler	2.02	2.76	2.74	3.75
American Crow	0.34	0.00	0.34	0.34	Mourning Dove	0.00	0.31	0.00	0.00
American Redstart	4.38	4.91	2.74	4.10	Nashville Warbler	6.06	4.60	5.14	4.78
American Robin	0.00	0.31	0.68	1.71	Northern flicker	0.34	0.00	0.34	0.34
Bay-breasted Warbler	3.37	6.44	6.16	2.39	Northern Oriole	0.34	0.00	0.00	0.00
Belted Kingfisher	0.67	0.31	0.00	0.00	Northern Parula Warbler	6.40	6.13	5.48	4.78
Black and White Warbler	3.37	3.68	2.40	2.73	Northern Raven	1.01	1.23	0.34	0.00
Black-backed Woodpecker	0.00	0.00	0.00	0.68	Northern Three-toed Woodpecker	0.67	0.31	0.00	0.00
Blackburnian Warbler	6.40	6.75	6.16	6.48	Northern Waterthrush	0.00	0.31	0.00	0.00
Black-capped Chickadee	0.00	0.61	0.34	0.00	Olive-sided Flycatcher	0.00	0.00	0.00	0.34
Black-throated Green Warbler	0.00	0.92	0.34	0.34	Orange-crowned Warbler	0.00	0.00	0.00	0.34
Blue-headed Vireo	0.00	0.00	0.34	0.00	Ovenbird	5.05	2.76	5.14	4.10
Blue Jay	0.67	0.00	0.68	1.02	Palm Warbler	0.34	0.00	0.00	0.34
Boreal Chickadee	1.01	1.53	2.05	0.68	Philadelphia Warbler	0.00	0.00	0.00	0.34
Brown Creeper	1.01	1.23	2.40	1.71	Pileated Woodpecker	1.01	0.61	0.00	0.00
Canada Warbler	0.34	0.61	1.71	2.05	Purple Finch	0.00	0.00	0.34	0.00
Cape May Warbler	1.35	0.92	1.71	0.68	Red-breasted Nuthatch	6.40	4.60	3.42	2.39
Chestnut-sided Warbler	1.01	1.84	1.71	2.05	Red-eyed Vireo	6.73	4.29	4.79	6.83
Chipping Sparrow	1.35	1.23	1.03	2.05	Rose-breasted Grosbeak	3.03	2.76	2.74	4.10
Common Snipe	0.00	0.31	0.00	1.71	Ruby-crowned Kinglet	0.34	0.31	1.71	0.34
Common Yellowthroat	0.67	0.92	0.34	1.02	Ruffed Grouse	1.01	0.31	1.03	0.34
Dark-eyed Junco	0.00	0.00	0.00	1.02	Song Sparrow	0.00	0.00	1.03	3.07
Downy Woodpecker	0.00	0.31	0.00	0.34	Swainson's Thrush	2.69	5.52	5.82	3.75
Eastern Pewee	1.01	0.61	1.37	1.37	Swamp Sparrow	0.00	0.61	0.00	0.00
Eastern Phoebe	0.00	0.00	0.00	0.00	Tennessee Warbler	8.08	7.67	3.42	2.05
Golden-Crowned Kinglet	1.35	0.92	1.03	1.02	Veery Thrush	0.00	0.31	0.00	0.00
Great -crested Flycatcher	0.34	0.31	0.34	0.00	Warbling Vireo	0.00	0.31	0.34	0.00
Hairy Woodpecker	0.67	0.31	0.00	0.34	White -throated sparrow	4.71	4.91	5.14	9.56
Hermit Thrush	1.01	1.23	0.68	0.34	Winter Wren	1.68	1.84	4.11	3.41
Least Flycatcher	3.70	1.23	2.40	1.71	Yellow-bellied Sapsucker	1.01	0.92	2.05	2.73
Magnolia Warbler	6.06	7.98	6.85	4.10	Yellow-rumped Warbler	0.00	0.00	0.34	0.34

Appendix H: Abundance values for 27 species of birds within combined control blocks and combined experimental blocks during pre- and post-harvest years.
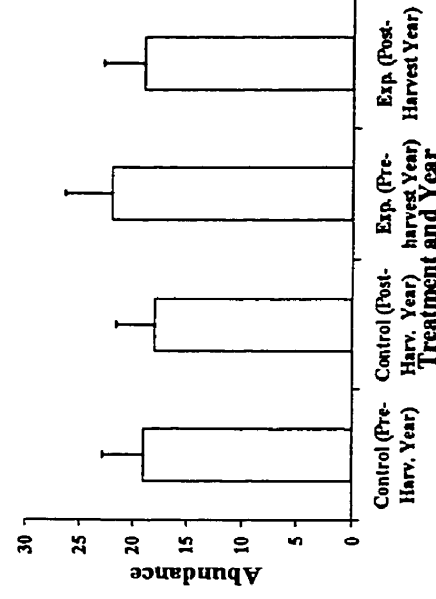
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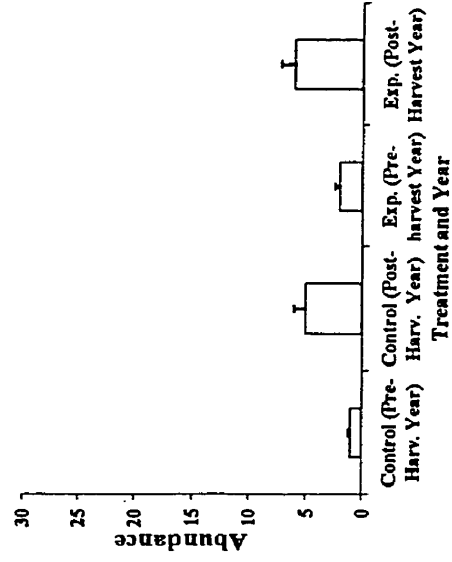

American Redstart

American Robin

Bay-breasted Warbler

165
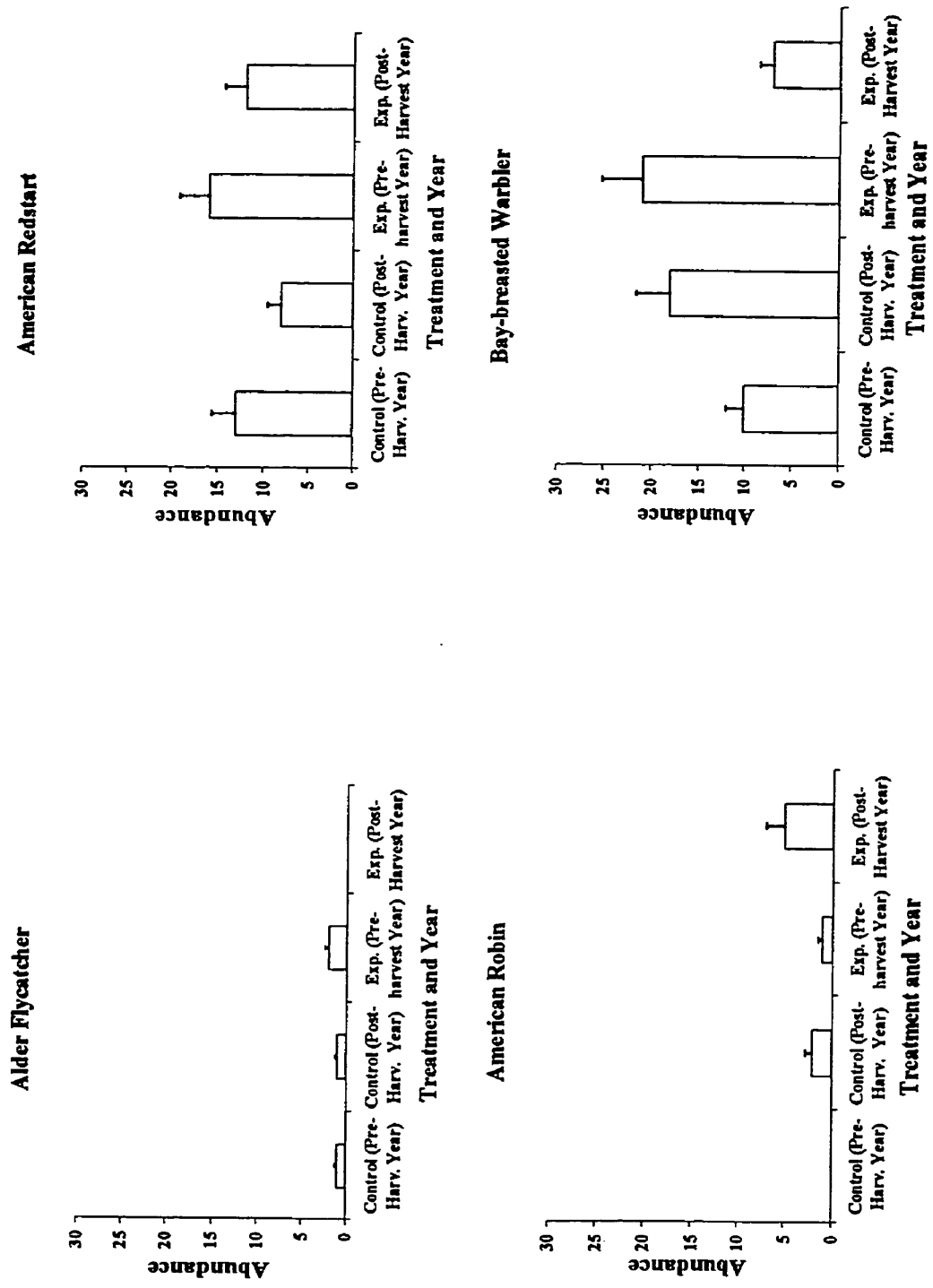
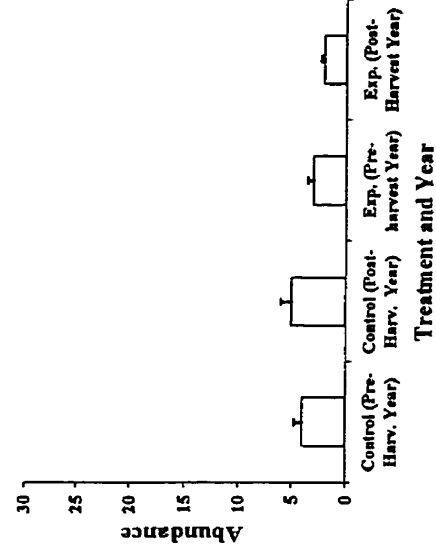# Appendix H: Continued

Black and White Warbler
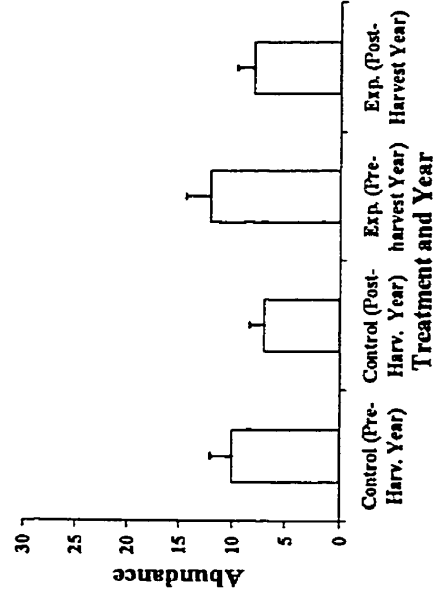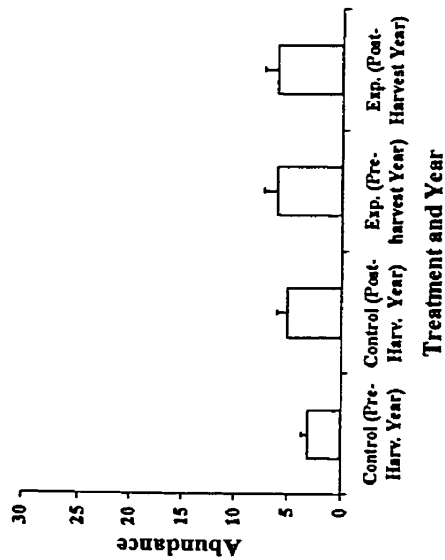

Blackburnian Warbler
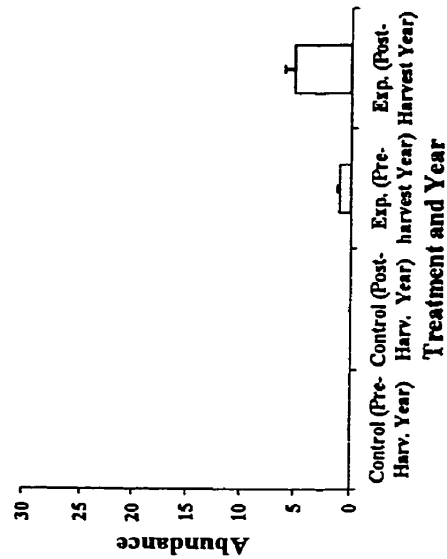

Canada Warbler
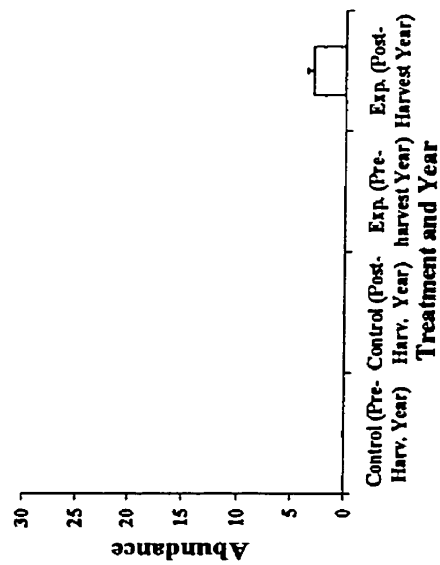

Cape May Warbler

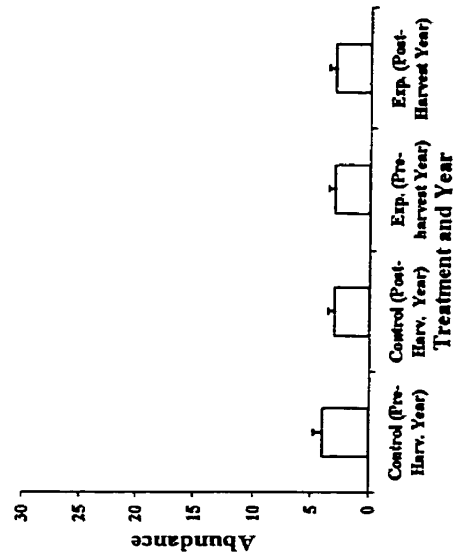# Appendix H: Continued
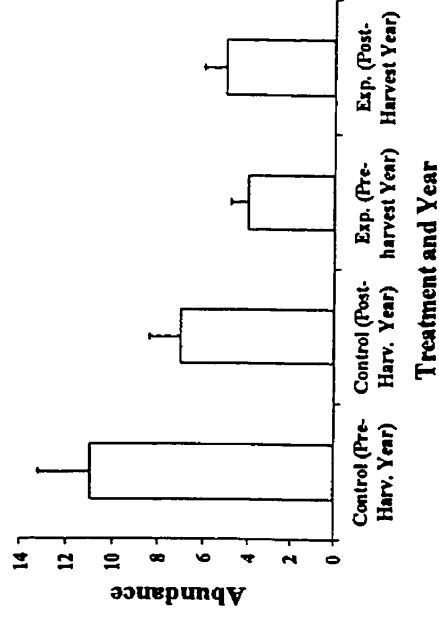
Chestnut-sided Warbler

Common Snipe

Dark-eyed Junco

Golden-crowned Kinglet

Least FLycatcher

Magnolia Warbler

Mourning Warbler

Nashville Warbler
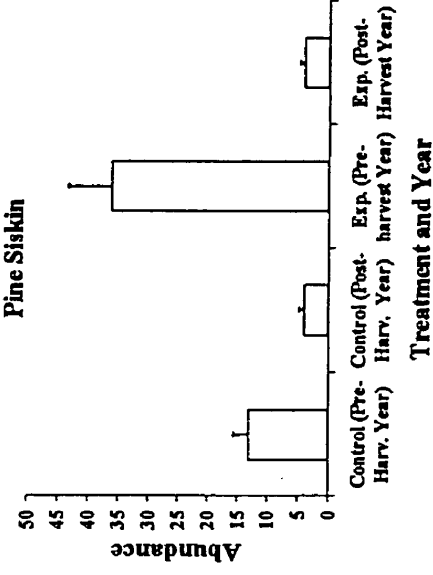
168
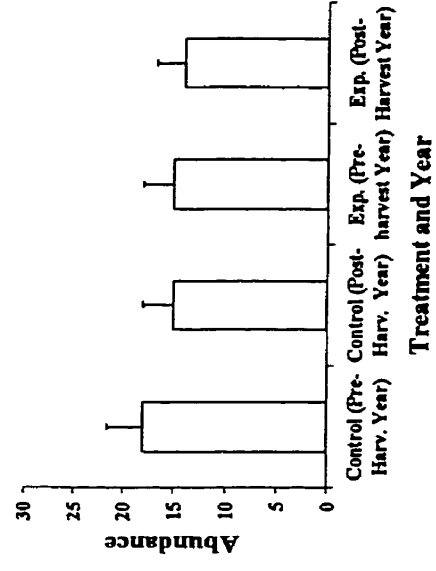
Appendix H: Continued

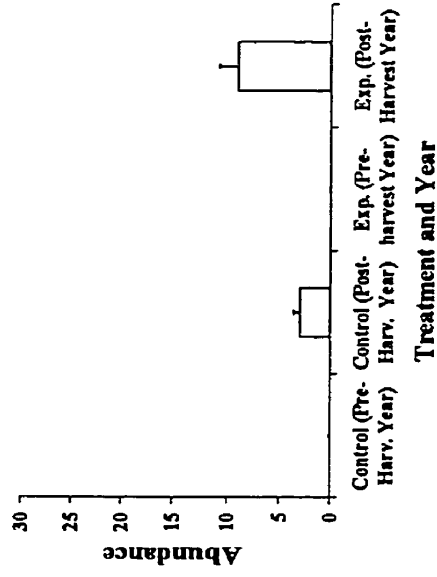## Northern Parula Warbler
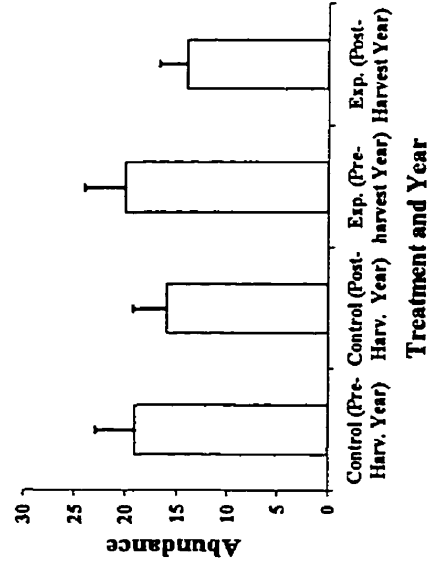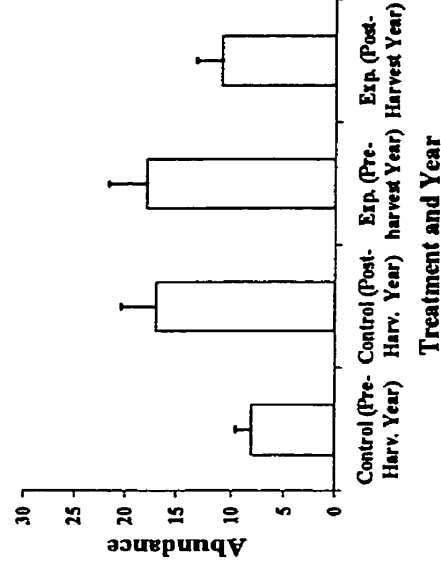

Olive-sided Flycatcher
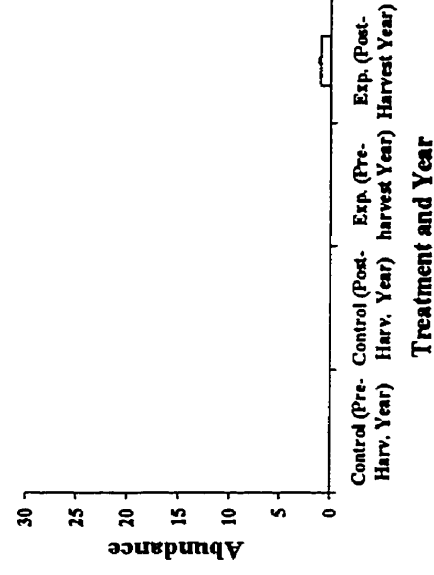
Ovenbird

Pine Siskin

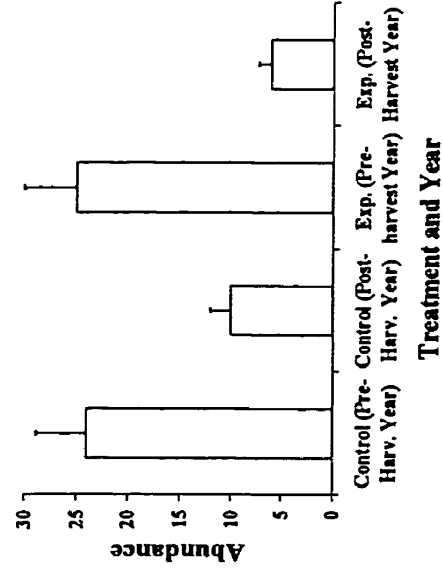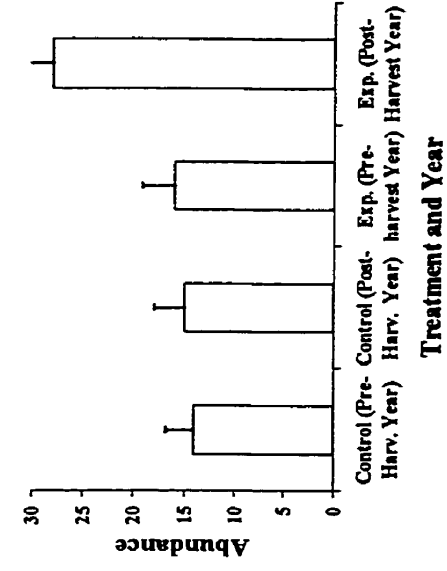
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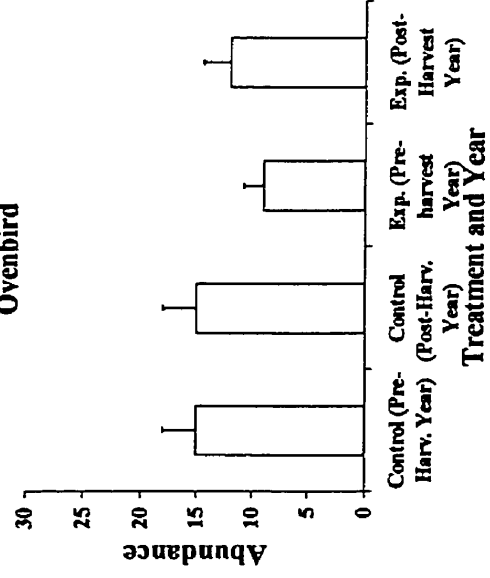
Appendix H: Continued

Song Sparrow

Swainson's Thrush

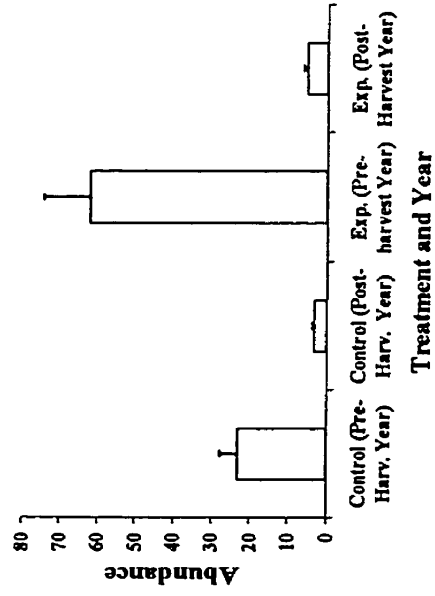
Tennessee Warbler

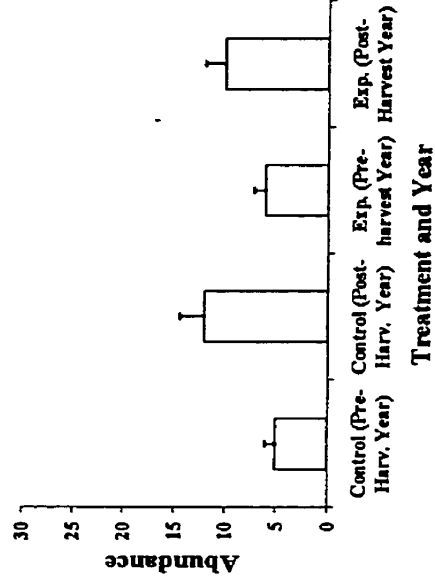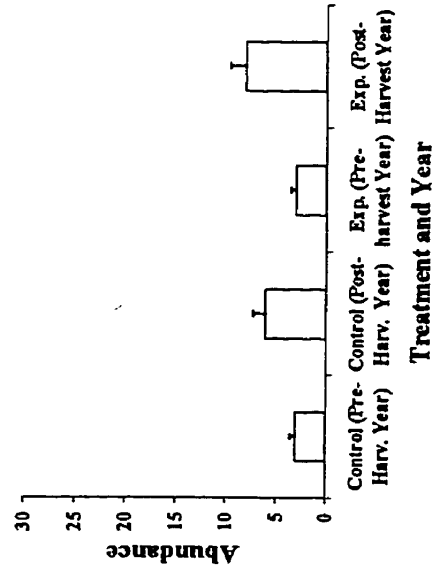**White-throated Sparrow**

Appendix H: Continued

White-winged Crossbill

Winter Wren

Yellow-bellied Sapsucker

Appendix I: Ground height measurements (cm) for ridge, center and bottom erosion pins for upland transects harvest (a) and control (b) sites as well as depth of soil organic layer (OL) for ridge pins in all transects. A negative value indicated a *decrease* in ground height relative to the measurment of ground hieght recorded at the time of pin installment. (*Targeted due to slope and disturbance attributes)

a.

Experimental	Pin:	Ridge			Center			Bottom		
Transect No.:	Observation: Depth of OL*	1st	2nd	Avg.	1st	2nd	Avg.	1st	2nd	Avg.
1	9.0	0.50	-0.40	0.05	0.75	-0.25	0.25	1.40	-0.05	0.68
2	7.7	0.30	-0.60	-0.15	1.10	0.15	0.63	-0.40	-1.20	-0.80
3	7.7	-0.55	0.15	-0.20	-0.45	0.10	-0.18	-0.70	-0.30	-0.50
4	6.4	0.50	-0.53	-0.02	0.10	0.05	0.08	0.15	-0.30	-0.08
5	7.7	0.40	-1.20	-0.40	0.35	0.20	0.28	0.10	-0.25	-0.08
6	11.5	0.05	-0.25	-0.10	0.70	0.20	0.45	-0.10	-0.45	-0.28
7	11.5	0.15	-1.35	-0.60	0.40	-1.00	-0.30	0.15	0.20	0.18
8	9.0	-0.50	0.75	0.13	-0.35	-0.10	-0.23	0.10	0.75	0.43
9	7.7	0.60	0.90	0.75	-0.30	-0.05	-0.18	-1.10	0.05	-0.53
10	6.4	-0.10	0.70	0.30	-0.40	0.30	-0.05	0.20	1.00	0.60
11	12.8	-0.35	0.90	0.28	0.30	0.10	0.20	-0.25	0.10	-0.08
12	10.2	0.20	-1.05	-0.43	0.45	-0.20	0.13	1.00	-0.35	0.33
13	9.0	-0.35	0.60	0.13	-0.20	1.50	0.65	-0.65	-0.10	-0.38
14	0.0	-0.20	-0.45	-0.33	0.20	-0.35	-0.08	0.25	-0.30	-0.03
15	6.4	0.30	-0.50	-0.10	1.05	0.45	0.75	0.20	0.05	0.13
16	7.7	-0.10	1.55	0.73	-0.25	-0.20	-0.23	-0.20	0.40	0.10
17	7.7	0.10	-0.70	-0.30	0.05	-0.30	-0.13	0.75	0.90	0.83
18	7.7	-0.50	1.10	0.30	-0.10	0.65	0.28	-0.30	0.20	-0.05
19	11.5	0.25	-0.60	-0.18	-0.50	-0.35	-0.43	0.10	-0.25	-0.08
20	7.7	1.50	-0.50	0.50	0.20	-0.30	-0.05	0.60	-1.10	-0.25
21	12.8	0.15	-0.65	-0.25	0.25	-0.95	-0.35	0.35	0.20	0.28
22	10.2	0.55	-0.75	-0.10	0.25	0.10	0.18	0.30	0.20	0.05
23	10.2	0.45	0.35	0.40	-0.35	-0.05	-0.20	0.85	0.20	0.53
24	7.7	0.45	-0.20	0.13	0.40	0.10	0.25	0.35	0.95	0.65
25	6.4	-0.15	0.80	0.33	-0.45	-0.25	-0.35	-0.15	0.30	0.08
26	7.7	0.20	0.40	0.30	0.50	0.25	0.38	0.30	0.05	0.18
27	9.0	-0.10	1.00	0.45	-0.20	0.60	0.20	-0.50	0.20	-0.15
28	10.2	0.25	1.05	0.65	-0.10	-0.05	-0.08	0.20	-0.75	-0.28
29	7.7	-0.35	-0.10	-0.23	0.30	0.10	0.20	0.20	-0.30	-0.05
30	9.0	-0.20	0.40	0.10	-0.20	-0.10	-0.15	-0.85	-0.25	-0.55
31*	0.0	-0.15	-0.45	-0.30	-0.50	-0.85	-0.68	0.20	-0.45	-0.13
32*	0.0	-0.40	-1.70	-1.05	-0.30	-0.15	-0.23	0.30	0.40	0.35
Average:				0.025			0.032			0.034

b.

Control	Pin:	Ridge			Center			Bottom		
Transect No.:	Observation: Depth of OL*	1st	2nd	Avg.	1st	2nd	Avg.	1st	2nd	Avg.
1	9.0	-0.20	0.75	0.28	0.20	0.40	0.30	-0.50	0.35	-0.15
2	14.1	0.25	0.10	0.18	0.30	0.05	0.18	0.40	0.25	0.25
3	11.5	0.40	0.20	0.30	0.60	-0.25	0.18	0.30	-0.50	-0.18
4	9.0	0.05	0.55	0.30	-0.50	-0.10	-0.30	0.25	0.50	0.30
5	11.5	-0.25	0.10	-0.08	0.60	0.15	0.38	0.10	0.35	0.15
6	10.2	0.35	-0.40	-0.03	-0.40	-0.70	-0.55	1.20	0.60	0.83
7	12.8	-1.30	0.45	-0.43	-1.15	-0.70	-0.93	0.05	0.45	0.18
8	7.7	-0.60	0.15	-0.23	0.35	0.50	0.43	-0.45	0.10	-0.25
9	10.2	0.35	-0.90	-0.28	-0.25	0.70	0.23	0.20	-0.80	-0.38
10	11.5	0.55	0.25	0.40	0.20	-0.50	-0.15	-0.10	-0.50	-0.38
11	15.4	-0.40	0.30	-0.05	0.20	0.35	0.28	0.45	-0.20	0.05
12	11.5	0.05	-0.40	-0.18	0.20	-0.60	-0.20	0.50	0.30	0.33
13	9.0	-0.30	0.25	-0.03	-0.65	0.20	-0.23	-0.35	-0.25	-0.38
14	9.0	1.30	0.10	0.70	0.45	-1.10	-0.33	0.70	-0.30	0.13
15	12.8	0.10	-0.60	-0.25	0.25	-0.70	-0.23	0.65	0.40	0.45
16	16.6	-0.45	-0.25	-0.35	0.10	0.50	0.30	-1.00	0.10	-0.53
17	14.1	0.25	-0.30	-0.03	0.30	-0.40	-0.05	0.50	0.50	0.43
18	10.2	-0.50	-0.15	-0.33	-0.05	0.60	0.28	-0.60	-0.10	-0.43
19	9.0	-0.20	0.45	0.13	-0.30	-0.85	-0.58	-0.45	0.20	-0.20
20	7.7	0.10	0.60	0.35	1.00	0.60	0.80	0.35	-0.50	-0.15
21	12.8	0.55	-0.10	0.23	0.30	-0.55	-0.13	0.60	0.40	0.43
22	11.5	0.50	-0.25	0.13	0.20	-1.20	-0.50	0.80	-0.25	0.20
23	9.0	0.20	-0.45	-0.13	-0.25	-0.35	-0.30	0.65	0.40	0.45
24	14.1	-0.75	-0.30	-0.53	0.35	0.40	0.38	-0.55	0.25	-0.23
25	11.5	0.60	0.20	0.40	-0.25	-0.35	-0.08	0.40	0.10	0.18
26	14.1	0.50	0.10	0.30	0.45	-0.15	0.15	0.05	-0.60	-0.35
27	11.5	0.35	0.55	0.45	0.20	0.45	0.33	-0.20	0.10	-0.13
28	10.2	0.45	-0.20	0.13	0.74	-0.30	0.22	-0.40	1.40	0.43
29	14.1	0.90	0.35	0.63	-0.05	-0.35	-0.20	0.20	-0.80	-0.38
30	10.2	-0.85	0.25	-0.30	0.55	0.60	0.58	-0.05	0.50	0.15
Average:				0.057			-0.014			0.022