Effects of Timber Harvest on Aquatic and Terrestial Communities within Wetland-Riparian Ecosystems

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BACKGROUND

Elders of the Teme-Augama Anishnabai (TAA-Deep Water People) in the Lake Temagami area were alarmed at the perceived reduction in wildlife on n'Dake Menan, their traditional land use area. The TAA believe in holistic land stewardship to achieve a harmonious relationship between land and people, and that it is necessary to sustain all lifeforms for current and future generations.

A Memorandum of Understanding signed April 23, 1990 by the respective governments of the TAA and the province of Ontario (Ontario Ministry of Natural Resources, 1990a) outlined fundamental principles including "Sustained Life wherein the natural integrity of the land and of all life forms therein and thereon are maintained." A Statement of Intent (September 21, 1991) directed that "in areas of concern, planning will be applied to all wetlands, lakes and streams and appropriate prescription applied to protect the biological integrity and diversity of the area.

The current Ontario Ministry of Natural Resources (OMNR) Timber Management Planning process does not specifically address 'other wetland areas' in the timber harvesting guidelines. Buffer guidelines for some wetlands are in place if moose aquatic feeding areas are present or if a wetland is a component of a lake or stream system. In these situations, the provincial Moose Habitat or Fish Habitat guidelines apply. In January 1992, recommendations were made by the TAA to MNR for 30m no-cut riparian buffers to be applies to all wetlands within active timber harvest operations on their traditional territory.

The Teme-Augama Anishnabai logic embraces the interconnectedness of all things in the Earth's sphere. Moose Habitat and Fish Habitat guidelines, not withstanding, most wetland and riparian areas are ignored in timber management planning because they are not valued by existing management criteria. The Teme-Augama believe that these unvalued and unevaluated areas are integral to a rich, healthy, and natural environment.

The objectives of the Teme-Augama Anishnabai recommendations concerning wetland and riparian areas were to provide increased protection to water ecosystems; to conserve more remnant habitats in cutting operations; and to provide an extensive network of contiguous habitat instead of small islands of habitat in contiguous cuts.

The Teme-Augama Anishnabai were particularly encouraged by Dr. Dan Welsh of the Canadian Wildlife Service to undertake this study as no measurement of the effects of logging on songbird communities ever had been done. This study is not conclusive in proving or disproving that all wetlands and riparian areas should be buffered from harvesting operations. The Teme-Augama Anishnabai recognize the value of this study as baseline information to a longer-term study which would produce more conclusive results given that the true effects of buffering or not buffering these areas may not be realized in two or three years. In fact, there has been some criticism from the scientific community in this regard. The TAA strongly recommend that this study be revisited and revived for a second phase or as a section of a more holistic study at some appropriate future time.

ABSTRACT

Thirty wetland sites were studied south of Temagami to determine the short term effects of timber harvesting on abiotic and biotic components of wetland edge ecosystems. Riparian buffers were established to determine whether 30m buffers mitigate timber harvesting effects. Pre- and post-harvest studies examined the riparian forest, songbirds, owls, water chemistry, fur-bearers and fish during a two year period. Treatment groups included reference wetlands with no timber harvesting, those with 30m no-harvest buffers and those without buffers. Coniferous forests surrounding wetlands were clear-cut while mixedwoods were harvested on a selective basis.

After clear-cut logging, tree basal area of the conifer forest was significantly reduced ($P < 0.01$) for jack pine (*Pinus banksiana* Lamb.), black spruce (*Picea mariana* [Mill.] B.S.P.), snags and total basal area of combined tree species. Tree species richness also decreased ($P < 0.01$) in unbuffered areas. Selectively-cut mixedwood forests were not significantly different after cutting ($P > 0.05$).

Bird breeding pair density was higher in the mixedwood forest than in conifer ($P < 0.01$). Bird territory density during the breeding season in both conifer and mixedwood forests was not different ($P > 0.05$) after logging regardless of 30m buffers in terms of: (1) total bird density, (2) species richness, or (3) territory density within the wetland-riparian habitat guild. Changes in species composition did occur as brown creepers (*Carthia americana*) decreased ($P <0.05$) in conifer unbuffered areas but not where buffers were in place. Use of buffers had little effect on Winter Wren populations as densities increased ($P<0.05$) in the conifer clear-cut in both buffered and unbuffered areas. In mixedwood forests, Swainson's thrushes (*Catmarus ustulatus*) decreased ($P <0.05$) in buffered relative to unbuffered and reference, while gray jays (*Perisoreus canadensis*) were significantly reduced ($P < 0.05$) in the second year in both buffered and unbuffered treatment groups. Wetland size was found to be highly significant ($P<0.01$) for bird community composition.

Barred owl (*Strix varia*) density decreased in mixedwood forests but boreal owl (*Aegolius funereus*) increased in the conifer clear-cut. Great horned owls (*Bubo virginiamug*) showed no discernible trend in either forest type. In all treatment groups and forest types, no differences in northern saw-whet (*Aegolius aeadicug*) and great gray owl (*Strix mebulosa*) densities were observed because the numbers were too low.

Total density of fur-bearers decreased ($P <0.05$) in winter after clear-cutting without buffers although there was no difference ($P>0.05$) in species richness after cutting in conifer forests. Significant decreases ($P < 0.05$) occurred in Red Squirrel and Snowshoe Hare densities after clear-cutting. Densities were maintained in buffered but not unbuffered areas. Squirrel and hare densities were not different ($P > 0.05$) after selective cutting in mixedwood forests. No other fur-bearers were significantly affected ($P > 0.05$) by logging or use of buffers in either conifer or mixedwoods.

In clearcut conifer forests, significant differences ($P < 0.05$) occurred in pH, aluminum, manganese and dissolved organic carbon in wetlands in spring, and pH, aluminum, manganese, oxygen and alkalinity in fall. Significant decreases ($P < 0.05$) occurred in manganese in buffered relative o unbuffered areas while significant decreases ($P < 0.05$) occurred in dissolved oxygen in unbuffered relative to buffered conifer wetlands. There were no differences in chemical composition in wetlands in mixedwood forests in spring or fall ($P > 0.05$) due to selective harvesting.

For aquatic biota, no difference ($P > 0.05$) was observed in total abundance or species richness for 12 species of minnows due to treatment effects in selectively harvested mixedwoods. In general, biodiversity of aquatic and terrestrial communities was similar before and after logging. However, some shifts in species did occur as a result of habitat modification.

Key words: wetlands, Temagami, riparian buffers, timber harvest, birds, owls, fur-bearers, water quality, aquatic biota, forest vegetation

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INTRODUCTION

Background

Wetlands are an integral part of the overall landscape and are of obvious importance in maintaining biological diversity. This wetland study was funded through the Northern Ontario Development Agreement, Northern Forestry Program from 1992 to 1996. Robin Koistinen and Mary Laronde were the Principal Investigators for the Teme-Augama Anishnabai (TAA), Thomas Whitfield and Dr. Ronald Hall were the authors and Dr. Ken Abraham (OMNR) was the scientific authority. An advisory Steering Committee consisted of members from TAA, Canadian Forest Service (CFS), OMNR, Ministry of the Environment and Energy (MOEE), Canadian Wildlife Service (CWS), Mid-North Forest Industry Alliance (MFIA), West Nipissing Trapper's Council (WNTC) and T.W.'s Ecological.

Objectives

The wetland studies described in this report were designed to quantify major abiotic and biotic components influenced by timber harvesting practices on wetland ecosystems in landscapes near Temagami, Ontario, and to quantify the effectiveness of 30 m forested buffers around wetland ecotones on wildlife (e.g., songbirds and owls, fur-bearers, fish) and water quality.

For this study, the authors collected qualitative and quantitative information on the structure and function of abiotic and biotic parameters in forested-wetland ecosystems at different temporal and spatial scales. Two dominant forest ecosystems were studied, namely mixedwood and conifer forests. Instead of concentrating on dominant single species changes due to timber management, a holistic approach on biotic communities was used at the interface between terrestrial and aquatic ecosystems with an emphasis on biogeochemical linkages.

Temporal observations before and after harvest covered all seasons and consisted of studies on: forest structure and composition in summer, songbird census during the day in summer, owl vocalization at night during winter, fur-bearer movement in winter, monitoring of water quality in spring and fall and assessing aquatic biotic communities during summer. Spatial scales consisted of wetlands in selected northern townships (Sisk, Fell, McWilliams and Maclaren) between Temagami and North Bay.

This report is organized into six sections, each describing a separate component of the study and containing an introduction, methods, results, discussion and conclusions. An overall introduction, literature review concerning studies related to riparian buffers, acknowledgments, and appendices are included.

Study Area

Sampling was conducted in the vicinity of Marten River and River Valley, between Temagami and North Bay in north-central Ontario (Fig. 1). Within this area an initial 45 wetlands were selected. After the first year, the selection was reduced to 30 wetlands with a combined wetland and riparian edge area of 358 ha (Table 1). The study region is characterized by rugged topography with numerous lakes, rivers and wetlands at an average elevation of 294 m above sea level. In this report, the term wetland/edge is used to describe each wetland along with associated aquatic and semi-aquatic vegetation, in addition to an upland riparian zone extending a distance of 50 m perpendicular from each wetland perimeter.

Table 1. Area (ha) of wetland/edge ecosystems in wetlands and surrounding riparian forests 50 m from aquatic-terrestrial interface.

AREA of WETLAND/EDGES					
MIXEDWOOD FORESTS			CONIFEROUS FORESTS		
Reference	30 m Buffer	No Buffer	Reference	30 m Buffer	No Buffer
8.9	23.4	8.3	8.9	18.2	7.8
12.4	9.7	13.3	16.7	17.4	31.6
13.9	8.9	19.1	8.9	7.8	11.5
6.8	12.1	9.0	7.4	7.6	12.4
11.9	10.1	12.4	8.9	4.8	7.5
53.9 ha	64.2 ha	62.1 ha	50.8 ha	55.8 ha	70.8 ha

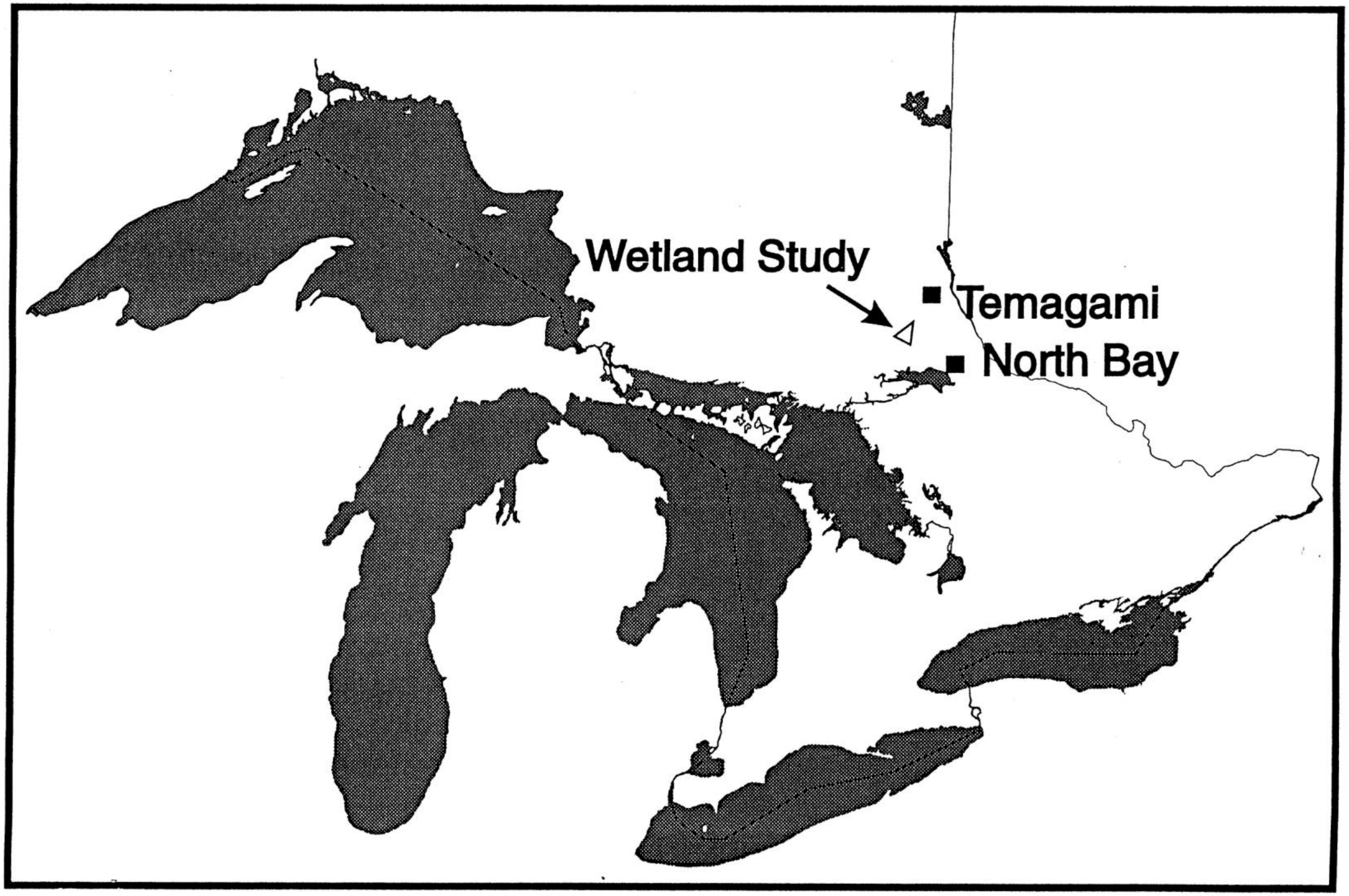
Geology

The study area is underlain by the Precambrian Shield with the majority of the area covered by varying depths of ground moraine. Soil depths vary with shallow soils and exposed bedrock in upland areas due to glacial scouring. Valley bottoms are often dominated by medium- to fine-grained sands and gravel of glaciofluvial origin (Johnson 1988). Glaciofluvial deposits, silty glaciolacustrine deposits and eskers are all found within the region. The undulating and often rocky topography creates variable site conditions over a small area. The surface geology is controlled mainly by the results of Wisconsin glacial activity and to a lesser extent by the underlying bedrock.

Climate

Lying at the interface of Site Regions 4E and 5E (Hills 1952), the study region has a humid continental climate. Wide seasonal variations in temperature occur, ranging from -13°C in January to 19°C in July (Brown et al. 1980). Mean annual air temperature is 2.5°C with a mean air temperature of 10°C in May and September, 15°C in August and June and 17.5°C in July (Environment Canada 1982). Mean annual precipitation is 862.9 mm per year with summer months (May to September) averaging 75 mm per month (Environment Canada 1982). An average spring extends from April 15 to May 21, summer from May 21 to

Figure 1. Location of the study area between Temagami and North Bay, Ontario.



September 15, autumn from September 15 to November 11, and winter from November 11 to April 15 (Crowe et al. 1977). Spring lake ice break-up usually occurs between April 24 and 27 with the average snowmelt completed by April 29. Mean length of growing season is 170 to 175 days from April 30 to October 15. The mean length of frost-free period is 80 to 100 days (Ontario Ministry of Natural Resources 1990).

Vegetation

The forested landscape of the Temagami-North Bay region lies in the transition zone between the Great Lakes-St. Lawrence forest and the more northern Boreal forest (Rowe 1972). The mixed forests as characterized by Rowe (1972)....

> ....contain eastern white pine (*Pinus strobus* L.) with scattered white birch (*Betula papyrifera* Marsh.), and white spruce (*Picea glauca* [Moench] Voss), although spruce frequently rivals pine in abundance. Another common though variable forest type is a mixture of the birch, pine and spruce, with balsam fir (*Abies balsamea* [L.] Mill.), trembling (*Populus tremuloides* Michx.) and largetooth aspen (*Populus grandidentata* Michx.). Both red pine (*Pinus resinosa* Ait.) and jack pine (*Pinus banksiana* Lamb.) occur, the former often predominant in bluffs along ridges and the latter is generally restricted to the driest sandy or rocky sites. The tolerant hardwoods, yellow birch (*Betula alleghaniensis* Britton) and sugar maple (*Acer saccharum* Marsh.), have only a scattered occurrence. The prevalent forest cover on the uplands is clearly a reflection of periodic past fires, and the sandy soils have provided conditions especially favourable for the propagation of eastern white pine, red pine and jack pine. On the lowlands, in poorly drained depressions and in swamps, black spruce (*Picea mariana* [Mill] B.S.P.) with tamarack (*Larix laricina* [Du Roi] K. Koch) or eastern white cedar (*Thuja occidentalis* L.), form well-marked communities.

A diverse understorey vegetation is dominated by saplings and seedlings of overstory species including red maple (*Acer rubrum* L.) along with woody shrubs locally dominated by speckled alder (*Alnus rugosa* [DuRoi] Spreng.), beaked hazel (*Corylus cornuta* Marsh.), mountain maple (*Acer spicatum* Lam.), chokecherry (*Prunus virginiana* L.fil.), wild raisin (*Viburnum cassinoides* L.), serviceberry (*Amelanchier sanguinea* [Pursh] DC.), fly honeysuckle (*Lonicera canadensis* Bartr.), willow (*Salix* sp.), mountain-ash (*Sorbus decora* Marsh.) and low sweet blueberry (*Vaccinium angustifolium* Ait.).

Study design

To quantify the effect of timber harvest and buffers versus no buffers on the wetland ecosystem, the optimal impact study design proposed by Green (1979) was used. An optimal impact study design has four prerequisites: before-impact baseline data are collected to establish a temporal control and to contrast with after-impact data; a sampling design based on type of impact, time and place is formulated to test against the null hypothesis of no change due to impact; measurements of relevant biological and environmental variables associated with the samples are obtained; and a reference area that will not receive the impact must be established.

In this study, the impact was the cutting of timber in selected wetlands which were equally allocated to 30 m buffer, no buffer and reference groups in both coniferous and mixedwood forest types (Table 2). For each of the components of the study (forest, songbirds and owls, fur-bearers, chemistry, and fish), the authors collected data in the three treatment areas before and after timber harvest. One treatment was the harvest of trees in the upland forest without disturbing those within 30 m of wetland edges (30 m buffer), the other was the removal of trees in both the upland and adjacent to the wetland edge (no buffer) and the third was no timber removal (reference). The null hypothesis tested was that any change in the impact area over a time period (including the impact) does not differ from the reference.

Table 2. Experimental design of wetland/edges assigned to treatments.

WETLAND			
EDGES	REFERENCE	30 m BUFFER	NO BUFFER
Conifer Forest	5	5	5
Mixedwood Forest	5	5	5

For all components of the study, a two-way (areas x times) factorial Analysis Variance (ANOVA) or multi-factorial Analysis of Variance (MANOVA) design was used to test the null hypothesis of no change in relative abundance, species richness, chemical composition in the water, etc., due to timber cutting with 30 m buffers or without buffers against the hypothesis of no areas x times interaction.

A significant interaction between space and time indicated a relative difference in parameters after harvesting (i.e., the parameter may have changed on all sites but relatively more on one site than another) and was detected using either a univariate or a Wilk's Lambda multivariate F-statistic (see results). If a significant interaction was demonstrated, we then used a one-way (M)ANOVA for each time period (before and after harvest) to determine differences in variables in the three study treatments (reference, 30 m buffers and no buffers). For those parameters that displayed significant ($p<0.05$) treatment effects, a Scheffe's pairwise comparison was used to determine if differences occurred in 30 m or no buffer treatments.

Initial work took place during the fall of 1992 when wetlands were identified, access trails created, and final study design and sampling protocols were established. Data collection took place during one year prior to the cutting (1993) and after cutting (1994) (Table 3).

Table 3. Timeline for study components before and after timber harvesting.

	TIMELINE FOR WETLAND STUDY		
	BEFORE	AFTER	AFTER
COMPONENTS	1993	1994	1995
Preliminary Field Layout	Sept-Dec		
Timber Harvest	Sept-Dec	Jan-March	
Winter Tracks of Fur-bearers	Jan-March	Jan-March	Jan-March
Owl Vocalization Survey	March	March	March
Bird Census	May-June	May-June	
Chemical Sampling and Analysis	May	May	
Minnow Trap Survey	July	early July	
Logging Hauling Impact		late July	
Inventory	Aug-Sept	Aug-Sept	
Taxonomy of Aquatic Biota	Sept-Dec	Sept-Dec	

Timber Harvesting Methods

Winter logging was carried out in accordance with standard operating procedures by local forest industries in September, 1993 and March, 1994. Clear-cutting in the conifer-dominated forests was carried out. Trees were felled and full-length logs were skidded out to tertiary winter roads on rubber-tired skidders. The trees were felled directionally away from wetlands and designated 30 m buffers with snags left standing inside riparian buffer zones. For safety reasons (e.g. potential over-head hazard), the decision as to whether snags should be felled was left up to the harvesting operator. Mixedwood forests were selectively harvested by the removal of primarily large diameter trees (with the exception of white pine which were managed under the shelterwood silvicultural system). A feller-buncher with a telescopic boom was used to harvest much of the mixedwood area. The use of this type of machinery left wider undisturbed spaces between access trails, somewhat reducing the number of trees harvested. However, rubber-tired skidders were also used for conventional cut and skid logging.

Due to a local oversupply of aspen, birch and balsam poplar during the fall of 1993, there was a concern that the study wetlands might not be harvested. Regional mills were contacted regarding the purchase of timber from the study area and logging was carried out by Fryer Forest Products Limited, Meadowside Lumber Limited and Goulard Lumber Limited. Harvesting progress was strongly influenced by other developments as well as mill strikes at two locations which occurred in the early part of the winter of 1993/94 and slowed the harvesting of all study areas. In another development, one of the major harvest operators fell into receivership during the summer of 1993. Selective cutting in the mixedwood forest was started in September of 1993 but initial clear-cutting of the conifer forests was delayed until late December. At the end of the winter harvesting season, many log piles still remained at the roadside in the conifer clear-cut. Hauling then took place in late July, once the songbird census and aquatic biota trapping was completed. Because the study was not designed to accommodate summer hauling, concern was raised about the confounding effect of July hauling near some study wetlands and the impact on water samples collected in August. For this reason, additional water samples were taken before, during and after road hauling in July to detect any effect on wetland water chemistry.

The original intent was to initiate the study using 45 wetlands for pre-harvest study and to reduce the sample size to the 30 most similar wetlands for the post-harvest phase. This contingency was in place due to concern over whether forests near wetlands would actually be harvested within the time-frame of the project. As it turned out, the only wetlands dropped were in the mixedwood area where logging did not occur for a variety of reasons including low timber quality. Preliminary data analysis was carried out on pre-harvest data on songbird and water chemistry data to help in deciding which of the wetlands to delete from the study and whether those selected for the post-harvest phase were suitable.

Literature Review

Wetlands

Wetlands are lands seasonally or permanently covered by shallow water (<2m depth) where the water table is at, or close to the surface (Ontario Ministry of Natural Resources 1992). A wetland is defined as land that is saturated long enough to promote wetland or aquatic processes as indicated by poorly drained soils, hydrophytic or water tolerant vegetation, and various kinds of biological activity which are adapted to a wet environment (National Wetlands Working Group 1987). This definition recognizes that biological processes as well as hydrology, vegetation, and soils, distinguish wetlands from other habitats. The Government of Canada (1991) recognizes that there are five general classes including marsh, fen, bog, swamp and shallow water. Wetlands are further divided on the basis of form using surface morphology and pattern, landscape setting, water type, and morphology of the underlying soil. Wetland types are also classified according to vegetation physiognomy (i.e. treed, low shrub [National Wetlands Working Group 1987]).

The study was set within an operational framework, including a logging schedule for the 1993/94 winter season. Harvest allocations to be cut over the single winter contained 75 wetlands which were then evaluated as to their suitability in terms of: (1) at least 2 ha in size, (2) mostly within (e.g. >50 percent) the block of forest stands to be harvested as often wetlands and streams form natural boundaries for cutting operations, and (3) somewhat similar wetland characteristics. Of the wetlands evaluated, marshes were by far the predominant type locally, followed by fens and some forested swamps with very few bogs. Forty-five wetlands (33 marshes and 12 fens) were selected for pre-harvest scheduling of study activities. For post-harvest data collection, these were later pared to 22 marshes and 8 fens giving a total of 30 wetlands. Final selection was determined on the basis of whether adequate harvesting was completed adjacent to study wetland/edges. Marsh and fen wetland types were grouped for data analysis as there were not enough fens to warrant separate analysis in both coniferous and mixedwood forests.

Describing wetland types requires consideration of several basic characteristics. In order to offer an overview of marsh and fen site features, a simplified list of basic wetland features used in characterizing these types was drawn from wetland classification literature as shown in Table 4. The most obvious feature that distinguishes marshes from fens is the mineral soil bottom in the case of marshes compared to an organic buildup (>40 cm peat) over time creating peatlands in fens. Marshes also generally have much higher vegetation species richness/diversity than fens, yet both types are minerotrophic in that they receive input of water from streams and neither type is classed as 'forested' (e.g. swamps have >10 percent area in live or dead trees [Riley 1987]). Data were collected concerning aquatic and semi-aquatic vegetation communities.

Table 4. General features of wetland types used in classification.

Wetland Type	Peatland	Forested	Water	Vegetation Diversity	Dominant Vegetation
Marsh	no	no	minerotrophic	high	emergents
Fen	yes	no	minerotrophic	intermediate	sedge/low shrub
Bog	yes	no	rainwater	poor	moss/low shrub
Swamp	no	yes	minerotrophic	high	trees/tall shrub

Wetland/Edge

Wetlands are among the most valuable and complex ecosystems on earth. Use of a buffer strip of intact riparian vegetation can be effective in mitigating changes to aquatic environments brought about by land disturbances (Moring et al. 1985). In general, as the intensity of land use activity increases, the width of the buffer needed to minimize adverse effects increases proportionally (Brown and Schaeffer 1987).

Terrestrial areas surrounding wetlands are very important to the ecology of wetlands. Wetland riparian areas include a transition zone with surface water on one side and an upland terrestrial portion on the other side of the transitional zone. Soils grade from poorly drained, often organic soils of the wetland into less poorly drained and well drained soils of the upland. Often the area of wetland edge is the site of ground water emergence which may show abrupt changes in slope. Within this relatively narrow wetland edge there exists a great variety of terrain, soils, and hydrology. Due to this diversity in conditions, a range of plants and animals utilize this unique zone linking wetlands with upland forests.

Various authors have reported on the diversity of plants, animals and high productivity at the edge of wetlands. Wildlife use riparian zones disproportionately more than any other type of habitat (Thomas et al. 1979a). Increased diversity has been shown at the edge of marshes along with increased ecological values of marshes with higher upland edge/marsh ratios

(Gucinski 1978). Anderson et al. (1980) determined that the greatest diversity of tree species, terrain, and soil moisture is in the transition zone. Errington (1957) notes the diversity of wildlife, including owls, rabbits, foxes, nesting ducks and herons, small birds, mammals, skunks, minks, and muskrats in riparian areas of Iowa and South Dakota marshes. Not only does the wetland edge provide a diverse habitat for plants and wildlife, but conditions along the wetland edge also affect the ecology of the wetland similar to conditions adjacent to a stream or lake. Others (Richardson et al. 1978) believe that the chemical composition of water entering and flowing across a peatland has significant effect on peatland development and vegetation. Jaworski and Raphael (1978) found that sediment from run-off accumulated along the Great Lakes littoral zones may be associated with a reduction of preferred waterfowl foods and fish spawning habitat. Odum (1978) stated that riparian zones have their greatest value to us as buffers and natural filters between the effects of human disturbance on the land, and water, our most vital life-support resource.

Riparian Ecotones

Odum (1971) defined an ecotone as a transition stage between two or more diverse communities. He suggested that an ecotonal community commonly contains many of the organisms of each of the overlapping communities as well as organisms which are characteristic of and often restricted to the ecotone. Odum described this tendency to increased variety and density at community junctions as the edge effect. Ecotones contain rich assemblages of flora and fauna, and serve as linkages for the movement of water and materials throughout the landscape (Holland and Risser 1991).

Abiotic and biotic components must move across heterogeneous landscapes and boundaries between areas take on important ecological functions. For example, ecotones are important in satisfying life-cycle needs of many organisms, and are characterized by high biological diversity. Ecotones are sometimes populated by more kinds and larger numbers of birds and mammals than can be found in either of the adjoining, more homogenous communities (Odum 1993). Odum further states that wetlands are actually ecotones, lying between land and water. Across the landscape, marsh and fen wetlands link land to rivers and lakes while isolated wetlands such as fens and bogs act as ecotones between land and groundwater.

Brown (1985) states that wetland/edges consist of vegetation that requires large amounts of free or unbound water. Riparian buffers help to prevent non-point source pollutants from reaching the aquatic environment by acting as sinks, sources, filters, and transformers for substances affecting water quality. They also serve to modify temperature and flow characteristics of water (Tims 1994). These habitats also provide essential wildlife needs such as food, water and cover (Stocek 1994).

Buffers

One method of reducing the impact of land use (i.e. timber harvest, road and housing construction, agricultural practices) upon adjacent wetlands is to provide an area of uncut buffer surrounding wetlands. Vegetated buffers adjacent to wetlands are considered to be one of the richest zones for aquatic organisms, mammals, and birds (Clark 1977). Buffers provide essential habitats for wetland-associated species (Brown 1985; Thomas 1979a, b). For instance in Washington state, Thomas (1979a) found that 320 (or 85 percent) of 378 terrestrial vertebrates of the region use wetlands and/or their adjacent riparian areas.

Birds and other animals may have their life requirements met either wholly or in part by wetlands and the adjacent edge area (Naiman et al. 1988). For instance, wetland-associated mammals such as mink and river otter feed in wetlands but breed and raise their young in the riparian zone. Buffers provide essential habitats for feeding, roosting, breeding, rearing of young, cover for safety, mobility, and thermal protection (Castelle et al. 1992) of wetland-riparian associated species. Wildlife also use riparian buffers as travel corridors to move to new habitats. Depending on wildlife species, terrestrial environments must be of varying sizes to meet 'home range' requirements in maintaining viable populations. Wetland-dependent wildlife may be affected by land use changes adjacent to wetlands which fragment habitats below acceptable limits for some species (Brown and Schaefer 1987). Riparian buffers maintain fish habitat by providing shade and keeping water temperature low enough in summer to maintain dissolved oxygen levels to support fish. Buffers reduce the transport of silt from the upland disturbed forest sites to essential spawning grounds and prevent the destruction of aquatic invertebrates that are essential to fish as food (Riparian Habitat Technical Committee 1985).

Riparian buffers are sometimes referred to as vegetated filter strips (VFSs) or streamside management zones (SMZs). The emphasis on the filtering function of vegetation is derived from its ability to remove sediments and other waterborne pollutants from surface runoff. Wetland buffers act as physical barriers to flowing water as vegetation traps sediment and other insoluble pollutants (Broderson 1973; Shisler et al. 1987). Throughout the United States, best management Practices (BMPs) focus on water quality protection through the use of vegetated buffers, strategies for stream crossings and careful logging in sensitive sites (Dissmeyer 1994).

Research on wetlands has focused on one specific component, e.g., changes in surface water chemistry in relation to the use of buffers in resource management. Emphasis on managing biological components stressing a species by species conservation approach has been inefficient, biased and expensive (Lautenschlager 1995). Researchers have argued, as a result, for a more holistic approach to the study of structure and function of wetland ecosystems within a landscape context.

According to Wigley and Melchoirs (1993), shortcomings in wetland buffer research includes an overall lack of research on forest structure (basal area, species composition, diameter distribution, and various age classes) in relation to the value of buffer strips and a focus on streams and lakes but not on wetlands. Other problems relate to the fact that often wetland research is set in an agricultural setting, sample size is too small and rigorous statistical analyses have not been adequately performed. By recommending buffer characteristics such as width (Wigley and Melchiors 1993), jurisdictions are increasingly striving to protect biological diversity.

Ice et al.[1] identified the need to determine optimal buffer widths and also to determine if some cutting practices can be carried out within riparian buffers while still meeting terrestrial and aquatic ecosystem protection requirements. Buffers less than 15 m wide are generally ineffective in protecting wetland functions, but 30 m buffers seem to be adequate (Castelle et al. 1992). It is recognized that a generic 30 m reserve may not be the most appropriate width in every case, and the general rule of thumb seems to be the bigger, the better. In Iowa, Stauffer and Best (1980) showed that bird species increased with the width of wooded riparian habitats. In Virginia, Tassone (1981) recommended a minimum width of 60 m for hardwood riparian buffer strips to accommodate neotropical migrant birds.

In terms of logging impact and literature pertaining to streams, many authors recommend buffer widths of 30 m for water quality concerns such as sedimentation, nutrient loading and water temperature (Castelle et al. 1992). In this study, no attempt was made to define or measure the riparian zone of each wetland. Instead, a standardized width of 30 m was set to test riparian buffer effectiveness.

FOREST

Introduction

The two types included in the study were a conifer-dominated forest which was clear-cut and a deciduous-coniferous mixedwood forest which was selectively logged. Few studies dealing with riparian buffers have examined relationships between wildlife communities and buffer characteristics (Wigley and Melchiors 1992). In this study, forest structure was examined in terms of basal area/ha for individual tree species, overall combined basal area/ha, species richness, diameter classes of both live trees and snags, as well as total basal area/ha of snags.

Since normal cutting practices were anticipated in the forests adjacent to the study wetlands and because a mosaic of harvesting intensities seemed likely, numerous plots were established in order to accurately assess changes in the forest due to cutting.

Methods

Forest inventory plots were established near each of the five bird observation stations. Since bird censusing was completed up to a maximum distance of 50 m from the wetland perimeter, inventory of the forest was also taken beyond the 30 m buffers. At each bird observation

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[1]Ice, G.G.; Megan, W.F.; McGree, D.J.; Belt, G.H. Streamside management in northwest forests. Presentation at Society of American Foresters National Convention 18–22 September 1994, Anchorage, AK.

station, sample points were established along compass bearings extending perpendicular from perimeters. Plots at 15 and 50 m from the wetland/edge were established far enough apart to protect against overlapping in tree tallying. Pre- and post-harvest inventories took place in August to September of 1993 and 1994, respectively.

A method common to timber cruising was used to establish 'plotless' point samples (Husch et al. 1982) using a wedge prism with a basal area factor (BAF) of 2. The large forested area adjacent to all study wetlands was assessed in this way using a measure of tree basal area (m^2/ha) to assess the pre- and post-harvest differences in tree cover. Point sample centers were marked with flagging to establish a standardized location in pre- and post-harvest sampling. All trees greater than 9.5 cm diameter at breast height (DBH) were recorded to the nearest 2 cm. Standing snags were measured and recorded in the same way and according to five stages of decay (Maser et al. 1979). Appendix 6 contains a description of snag decay classes.

In order to quantify changes in forest structure, a number of features were used in the forest inventory. Overall tree density as expressed by basal area and total abundance was derived for each tree species. Live tree and snag densities were grouped using four commonly-used diameter classes. Size classes used were: (1) 10–24 cm DBH for polewood, (2) 26–38 cm DBH for small logs, (3) 40–48 cm DBH for medium sawlogs, and (4) over 50 cm DBH for large sawlogs.

Statistical Analysis

Basal area for 15 m and 50 m forested areas were analyzed for each tree species and for all species combined in both conifer and mixedwood forests. Snag data were analyzed in the same way with the addition of decay classes. Individual plot data was kept separate in the analysis in order to obtain a better estimate of variability about the mean. This method increased the power of the tests used in data analysis. In determining the extent of forest cutting both within the 30 m and beyond wetland/edges, 15 m and 30 m inventory data were analyzed separately according to forest type and treatment group.

All data were transformed using log (x+1) to more closely approximate the assumptions of linearity, normality and homogeneity of variance (Green 1979). All statistical procedures were conducted using SYSTAT for Windows, PC version 5 (1992). For a detailed explanation of the hypothesis and statistical procedures refer to section on Study Design.

Results

Refer to Appendix 2 for treatment means before and after logging within 30 m buffers (e.g. 15 m) and Appendix 3 for treatment means at 50 m. Mixedwood forest tree summaries are given in Appendices 4 and 5 for 15 m plots and 50 m inventory data respectively. A summary of reduction in basal area after logging in no buffer forests is given in Table 5.

Changes due to timber cutting of both live trees and snags according to diameter classes as well as reductions in snag decay classes are given in Table 6.

Table 5. Mean basal area (m^2/ha) reduction by tree species for 15 m and 50 m forest inventory plots combined in conifer and mixedwood no buffer wetland/edges before and after clear-cut timber harvesting.

	BEFORE	AFTER	BASAL AREA REDUCTION
CONIFER			
Jack Pine	10.6	3.8	64%
Black Spruce	6.6	1.6	76%
Balsam Fir	3.0	1.2	60%
White Spruce	2.4	1.2	50%
White Birch	2.1	0.4	81%
White Pine	0.8	0.4	50%
Poplar	0.5.	0.	100%
Maple	0.1	0.	100%
Total Basal Area	26.0	8.6	67%
MIXEDWOOD			
Cedar	4.6	4.6	0%
Balsam Fir	2.8	2.1	25%
Black Spruce	4.6	3.2	30%
White Spruce	2.1	1.1	48%
White Pine	3.7	3.1	16%
White Birch	1.6	1.6	0%
Poplar	0.8	0.3	62%
Red Pine	1.8	1.1	39%
Maple	0.1	0.1	0%
Total Basal Area	21.0	16.9	20%

Table 6. Changes in forest structure after clear-cut timber harvest in terms of coniferous tree diameter class distribution of live trees and snags including decay classes at 50 m plots beyond 30 m buffer wetland/edges.

	LIVE TREES			SNAGS			SNAG DIAMETER	
Classes (cm dbh)	Before (m^3/ha)	After (m^3/ha)	Reduction (Percent)	Before (m^3/ha)	After (m^3/ha)	Reduction	Decay (Percent)	Reduction (Percent)
10-24	8.1	8.1	nil	2.2	1.2	45	Class 1	100
26-38	6.4	4.0	38	1.4	0	100	Class 2	78
40-48	8.0	0	100	3.2	0	100	Class 3	78
50+	4.4	0	100	2.4	0	100	Class 4	95
							Class 5	50
Total	26.9	12.1	55	9.2	1.2	87		

Twelve tree species were recorded including white pine (*Pinus strobus* L.), red pine (*Pinus resinosa* Ait.), jack pine (*Pinus banksiana* Lamb.), black spruce (*Picea mariana* [Mill.] B.S.P.), white spruce (*Picea glauca* [Moench] Voss), eastern white cedar (*Thusa occidentalis* L.), balsam fir (*Abies balsamea* [L.] Mill), poplar (*Populus*, spp.), white birch (*Betula papyrifera* Marsh.), yellow birch (*Betula alleghaniensis* Britton), maple (*Acer* spp.), and ash (*Fraxinus* spp.). In the mixedwood, no significant multivariate interaction (p>0.05) occurred with all species combined (Wilk's Lambda, F-statistic = 0.868, DF = 14, 15, P = 1.0). Forest basal area was not significantly different before or after selective logging. For each individual tree species, there was no significant univariate interaction (all p's>0.05, 2-way ANOVA) before and after cutting. Also, there were no differences between unbuffered or buffered riparian forests relative to reference sites (p's>0.05, Scheffe's test). Tree species richness did not change significantly (p>0.05) after timber cutting in mixedwood forests (p>0.05, 2-way ANOVA, Appendices 4 and 5).

Conversely, in conifer forest wetland/edges with no buffers, large numbers of trees were cut to wetland shores (see Appendices 2 and 3). Forest basal area/ha for species combined in 15 m and 50 m treatments was reduced by 67 percent in the clear-cut (Table 5). Significant reductions occurred (all p's<0.05, 2-way ANOVA) for jack pine, black spruce and combined total basal area in both 15 m and 50 m conifer forests (Figs. 2, 3 and 4). No buffer 15 m forests showed significant reduction (all p's<0.05, Scheffe's test) in total basal area as did the 50 m forest for jack pine, black spruce, white birch, total basal area and snags (Figs. 3 and 4). Total basal area (m^2/ha) for live trees and snags in forests beyond 30 m in buffered wetland/edges were significantly reduced after logging (p<0.05, Scheffe's test, Fig. 4). Tree species richness decreased significantly (p<0.01, 2-way ANOVA, Scheffe's test) in no buffer wetland/edges (Fig. 5) within 30 m and at 50 m of wetland perimeters.

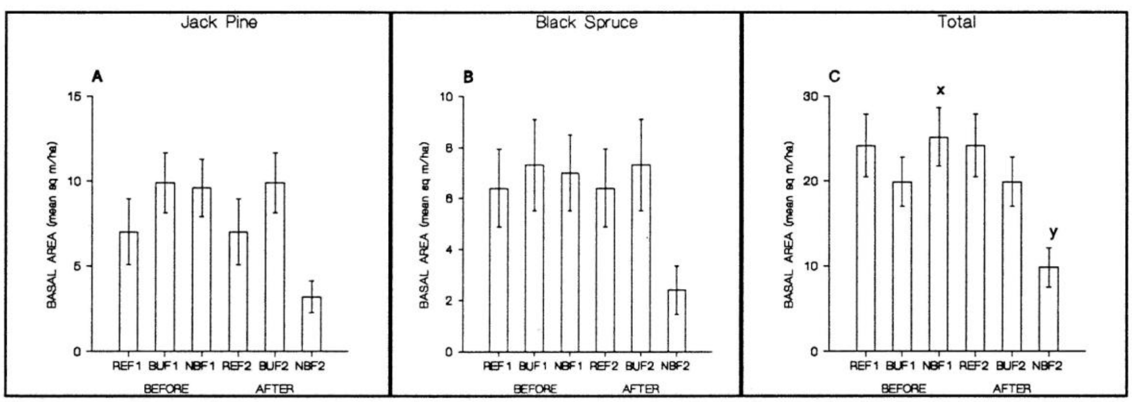


Figure 2. Tree basal area in plots at 15 m from terrestrial aquatic interface in conifer forests before (Time = 1) and after (Time = 2) timber harvest. Bars represent means, vertical lines ± S.E., n = 30. REF = reference, BUF = 30 m buffer, NBF = no buffer. For jack pine (A), black spruce (B) and total basal area (C), significant interaction (P<0.05, 2-way ANOVA) was detected. No buffer treatments in total basal area (C) were different before (x) and after (y) disturbance (Scheffe's, P<0.05).

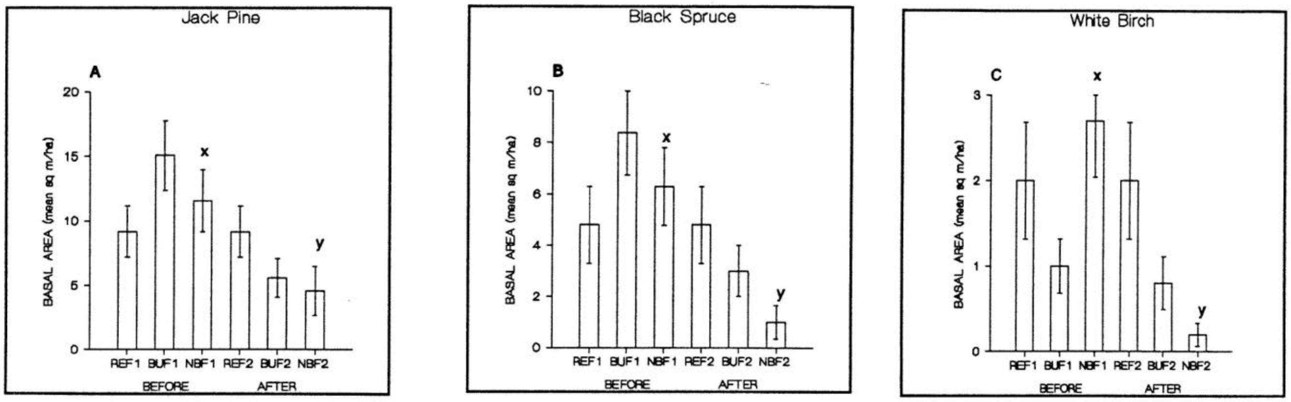


Figure 3. Tree basal area in plots at 50 m from terrestrial aquatic interface in conifer forests before (Time = 1) and after (Time = 2) timber harvest. For jack pine (A), black spruce (B), and white birch (C), significant interaction (P<0.05, 2-way ANOVA) was detected. No buffer treatments were different before (x) and after (y) disturbance (Scheffe's, P<0.05). See Figure 2 for legend details.

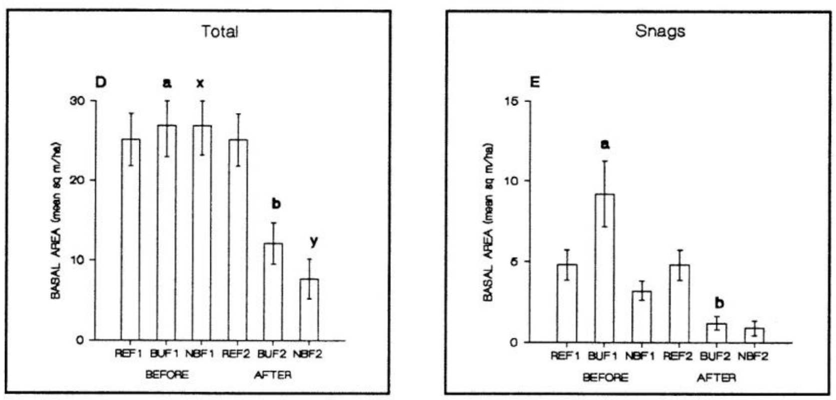
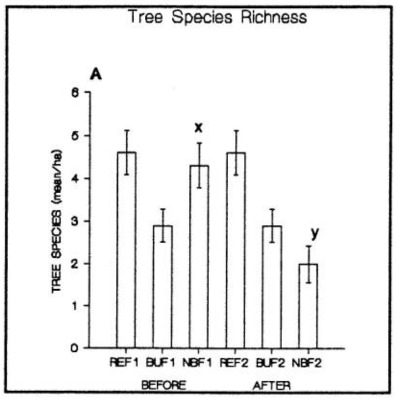


Figure 4. Tree basal area in plots at 50 m from terrestrial aquatic interface in conifer forests before (Time = 1) and after (Time = 2) timber harvest. For total basal area (D) and snag basal area (E), significant interaction was detected (P<0.05, 2-way ANOVA). No buffer treatments in D were different before (x) and after (y) disturbance (Scheffe's, P<0.05). Buffer treatments in D and E were different before (a) and after (b) disturbance (Scheffe's, P<0.05). See Figure 2 for legend details



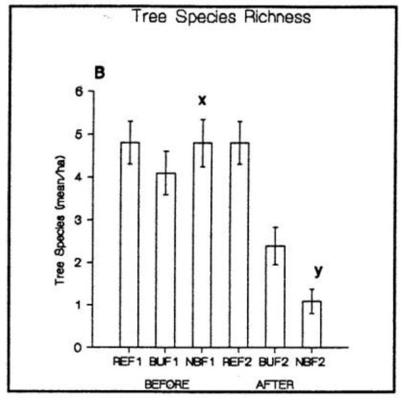


Figure 5. Tree species richness in plots at 15 m (A) and 50 m (B) from terrestrial aquatic interface in conifer forests before (Time = 1) and after (Time = 2) timber harvest. For A and B, significant interaction was detected (P<0.05, 2-way ANOVA). No buffer treatments in A and B were different before (x) and after (y) disturbance (Scheffe's, P<0.05). See Figure 2 for legend details.

Snag basal area was reduced by 61 percent from 2.8 m^2/ha prior to cutting to 1.1 m^2/ha after harvest. Upland forests adjacent to wetland/edges with 30 m buffers were clear-cut beyond the buffers and similar reductions in tree basal area occurred in no buffers (Table 5, Appendices 2 and 3). In 50 m forests, jack pine was reduced by 63 percent, black spruce by 64 percent, balsam fir by 30 percent, white spruce by 64 percent, and white birch by 23 percent. White pine and poplar were reduced by 29 percent and 43 percent respectively.

Forest structure at 50 m from the perimeter of wetlands in the conifer clear-cut are shown in Table 6. All medium and large sawlogs were removed with small polewood remaining after clear-cut timber harvesting. Snags of all diameter and decay classes were removed during harvest operations in the clear-cut with some snags remaining in the smallest size class (10–24 cm DBH). Snags in the youngest (e.g. class 1) category were all removed during logging (Table 6) and extent of reduction in other decay classes of snags varied. Where there were no buffers adjacent to wetlands, similar changes to forest structure resulted.

Discussion

Since no differences in forest structure could be detected after logging in either 30 m buffers or no buffers, it can be concluded that logging had little impact on mixedwood forest structure. Although differences could not be detected statistically, a considerable amount of timber, primarily larger diameter pine and maple with good quality birch and poplar was selectively harvested.

In conifer forests, considerable changes to forest structure resulted from clear-cut timber cutting. Basal area for jack pine, black spruce, total basal area and snags were significantly reduced after logging. Beyond 30 m, white birch basal area was also reduced significantly due to logging. Some polewood (10–24 cm DBH) trees remained uncut in pockets within inaccessible sites due to rough terrain and difficult access in areas of no buffers and beyond the 30 m buffers to 50 m. Many remaining trees were small diameter polewood (e.g.<15 cm dbh) of low merchantable value. Most snags were removed likely due to safety concerns regarding overhead hazard. Removal of these snags may have affected cavity-nesting bird species (see below). Overall tree species richness was reduced significantly after clear-cutting in the conifer forest.

Conclusions

Selective logging in mixedwood forests did not significantly alter the structure of the forest. Conversely, clear-cutting in conifer forests did significantly alter the structure in terms of a reduction in total tree basal area, tree species richness, total snag basal area and in terms of a significant reduction in the basal area of predominant tree species including jack pine, black spruce and white birch. Changes in diameter class distribution also occurred through removal of all medium (40–48 cm DBH) to large (over 50 cm DBH) diameter trees with mostly polewood (10–24 cm DBH) remaining in pockets along some wetland perimeters. The null hypothesis that timber harvesting did not significantly alter the composition of the forest is rejected for clear-cut coniferous forest wetland/edges but is accepted in selectively cut mixedwood forests. As a result of large differences in both extent of timber harvesting and changes to forest structure for the two forest types, databases for other components of this study were separated on the basis of forest type.

SONGBIRD COMMUNITIES

Introduction

In this study, a complete bird census of territories was determined rather than using bird count indices as an estimate of density as is the norm. We addressed changes in bird territory density in relation to wetland and forest stand characteristics. Few studies have examined relationships between wildlife communities and riparian buffer characteristics (Wigley and Melchiors 1993).

Methods

Before the bird census, five equally-spaced observation stations were marked with flagging along wetland perimeters. Walking trails were established to all wetlands to enable direct access during the pre-dawn period. A modified territory mapping technique was used to census the avifauna (Williams 1936, see Bibby et al. 1992 for reviews). Territory mapping was based on the fact that many breeding bird species mark their territories by song, displays or disputes with neighbours. When singing males are mapped, the registrations can all be grouped into clusters representing territories. Each territory assumes a breeding pair of one male and one female. Appendix 6 has additional detail concerning bird census methods and scores assigned to territories.

Bird censusing of only bird pairs that had all or part of their territory in the wetland and/or riparian edge was recorded over the breeding season between May 8 and June 29,1993 and 1994. Generally, two wetlands were censused on each day. Counts were made between 0530–1100 hr (Eastern Standard time), but occasionally, when weather conditions permitted (e.g. days that were relatively cool) and birds were still singing, counting continued until 1200 hr. Standardized symbols were tallied to map bird locations. All map data were transcribed onto coding sheets which were compiled into territories in databases for analysis. Counts interrupted by rain or high winds were abandoned and censuses were repeated on the following day. Three visits were made to all wetland sites over the May/June breeding period. Two birders, Chris Michener and David Kirk or Jill Jensen, each with 5–10 years of experience, worked independently on different wetlands, always visiting the same wetlands over the three visits.

On each visit the authors mapped singing males, calling individuals and sightings from the five listening stations used in the point counts (10 minute period), as well as all observations made while walking along a wetland perimeter from station to station. Observers did not leave a wetland until absolutely certain all birds were tallied. Because of the large number of wetlands (45 in 1993 and 30 in 1994), the territory mapping procedure used was modified from the 8–10 visit standard protocol (International Bird Census Committee 1969) to 3 visits per wetland.

To accurately map territories for each wetland, maps (scale 1:2 500) were prepared using enlarged photocopies of aerial photographs. A 50 m grid was superimposed to chart bird locations in the field and 30 m and 50 m bands were drawn on the maps. A digital planimeter was used to determine how much area was contained in each of the wetland/edge habitats to a distance of 50 m perpendicular from each wetland perimeter (Table 1). Observations were standardized between wetland/edge communities of various sizes by calculating the number of individuals per square kilometer.

Statistical Analysis

Bird data were grouped and analyzed in several ways: 1) total bird density (98 species, Appendix 1) and species richness, (2) density (63 species) and species richness grouped into a wetland-riparian guild, and, 3) breeding bird density for each of the 30 dominant species comprising 90 percent of the total bird abundance. Bird species used in the guild were selected on the basis of similar or overlapping general habitat needs associated with wetland and riparian habitats. Upland forest birds were included only in total bird density and overall species richness of wetland/edge communities. To group bird species according to habitat affinity, categories proposed by DeGraaf and Rudis (1986) on New England wildlife habitat requirements were used. Appendix 1 contains a complete listing of common and scientific names for birds encountered in the census.

Bird territories were based on absolute numbers derived from a thorough bird census during the May-June breeding period. The statistical analysis was patterned after the optimal impact study design by Green (1979; see also section on Study Design). Normal probability plots and histograms were examined for each of the 30 species and for total abundance on raw and log-transformed numbers to determine if parametric or non-parametric statistical procedures could be used. Testing for normality was also undertaken using the Wilk's Shapiro test. For species with normal distributions and homogeneous variances, we used univariate and multivariate ANOVA's. In cases where wetland size appeared to be an important covariate, we used ANCOVA. For species with non-normal distributions or heterogeneous variances, the nonparametric Kruskal-Wallis test was used. Multivariate statistical analysis pertaining to details of study design for testing interactions before and after timber harvest has been described earlier in the section on Study Design.

The influence of differences in wetland size (Table 1) on bird territories as a result of habitat modification was evaluated using residual analysis on both raw and log-transformed bird abundance data. Transformed data were then used in subsequent analysis with wetland size as a covariate (ANCOVA).

Results

For the two forest types in which bird species territory densities were summed over all treatments and years (Table 7), a total of 18 species were greater (seven significant) in abundance in the mixedwood forest with six species (two significant) being more abundant in the conifer forest. An additional six species were equally abundant in both forest types.

Total bird abundance (mixedwood forest: means = 675.6 vs coniferous forest = 557), total species richness (35.1 vs. 32.8), guild species abundance (388.8 vs. 324), guild species richness (19.6 vs. 18.1) were all significantly ($p<.05$, 1-way ANCOVA) higher in mixedwood forest than in coniferous wetland/edge communities (Table 7, see Appendix 1 for names of total bird species). For the dominant species with adequate data for statistical testing (total of 30), 9 bird taxa showed significant differences in density (pairs of birds/km^2) between the two forest types. white-throated sparrow (*Zonotrichia albicollis*), swamp sparrow (*Melospiza georgiana*), black-and-white warbler (*Mniotilta varia*), Canada warbler (*Wilsonia canadensis*), red-eyed vireo (*Vireo olivaceus*), black-throated blue warbler (*Dendroica caerulescens*) and black-throated green warbler (*Dendroica virens*) were greater ($p<.05$, 1-way ANCOVA) in the mixedwood forest. In contrast, the brown creeper (*Certhia americana*) and the dark-eyed junco (*Junco hyemalis*) were significantly ($p<.05$, 1-way ANCOVA; Kruskal-Wallis test) greater in the conifer forest (Table 7).

Table 7. Number of bird territories (pairs/km², mean ± S.E.) in wetland/edge ecosystems (n=30) over all years (1993 and 1994) and all treatments combined (reference, 30 m buffer, no buffer), in order of most abundant to least abundant species. The 30 species listed represent 90 percent of the total observed (98 species, see Appendix 1 for names). ANOVA and Kruskal-Wallis (k.w.) tests were used to test whether abundance within species differed between the two forest types.

	CONIFER FOREST		MIXEDWOOD FOREST		
BIRD SPECIES	Mean	S.E.	Mean	S.E.	P value
White-throated Sparrow	51.1	4.6	65.0	5.0	*
Magnolia Warbler	52.8	5.2	62.3	4.9	n.s.
Nashville Warbler	53.9	4.4	45.9	4.3	n.s.
Yellow-rumped Warbler	42.1	4.7	45.5	6.6	n.s.
Ruby-crowned Kinglet	34.9	3.8	35.2	5.2	n.s. k.w.
Common Yellowthroat	30.5	2.9	32.0	3.0	n.s.
Ovenbird	23.0	3.7	34.4	4.4	n.s.
Winter Wren	24.9	3.5	30.5	3.4	n.s.
Swamp Sparrow	17.5	2.2	38.1	3.3	*
Yellow-bellied Flycatcher	16.5	2.8	23.3	3.2	n.s.
Golden-crowned Kinglet	18.3	2.4	18.9	4.1	n.s. k.w.
Swainson's Thrush	14.6	2.1	14.7	1.8	n.s.
Black-and-white Warbler	8.0	1.9	16.4	2.4	**
Hermit Thrush	10.8	1.4	12.8	2.2	n.s.
Red-breasted Nuthatch	8.9	1.6	13.9	1.9	n.s.
Northern Flicker	7.0	1.1	12.9	2.4	n.s.
Chestnut-sided Warbler	7.1	1.9	11.9	2.5	n.s.
Black-capped Chickadee	7.9	1.4	10.9	1.5	n.s.
Canada Warbler	4.0	1.5	14.3	3.2	**
Yellow-bellied Sapsucker	6.9	1.3	10.7	1.8	n.s.
Solitary Vireo	11.0	2.1	6.3	1.7	n.s.
Blackburnian Warbler	10.0	2.0	6.6	1.4	n.s.
Red-eyed Vireo	5.4	1.5	11.0	2.0	*
Brown Creeper	10.2	1.9	5.8	1.3	* k.w.
Black-throated Blue Warbler	3.8	0.9	10.5	2.1	*
Purple Finch	6.0	1.2	7.1	1.3	n.s.
Black-throated Green Warbler	2.4	1.2	8.7	2.3	**
Dark-eyed Junco	10.3	2.6	0.5	0.4	**
Gray Jay	5.9	1.1	3.7	1.0	n.s.
Alder Flycatcher	3.3	1.4	5.8	1.9	n.s.
Total Abundance (98 species)	557.0	30.2	675.6	33.5	**
Species Richness	32.8	1.3	35.1	1.0	*
Guild Abundance (63 species)	324.0	18.6	388.8	18.8	**
Species Richness	18.1	0.8	19.6	0.7	*

* P < 0.05
** P < 0.01
n.s. = not significantly different

Species found to be strongly influenced (p<0.05, 1-way ANOVA) by wetland/edge size were white-throated sparrow, magnolia warbler (*Dendroica magnolia*), nashville warbler (*Vermivora ruficapilla)*, yellow-rumped warbler (*Dendroica coronata*), ovenbird (*Seiurus aurocapillus*), red-breasted nuthatch (*Sitta canadensis*), black-capped chickadee (*Parus atricapillus*), and alder flycatcher (*Empidonax alnorum*). Other variables were also influenced by wetland size including total density, total species richness, guild density and guild species richness. All subsequent analyses were done using wetland size as a covariate (ANCOVA).

Total bird abundance (Fig.s 6 A and B, Tables 8 and 9), species richness (Fig.s 6 C and D), and bird territory density of the wetland guild species (Figs. 7 A and B) were not changed significantly (p>.05, 2-way ANCOVA) within each treatment group of wetlands before and after timber harvest. In addition, total density for the 30 species combined (90 percent of total bird density, data not shown) was also unchanged (p>0.05, 2-way ANCOVA) after cutting of either forest type.

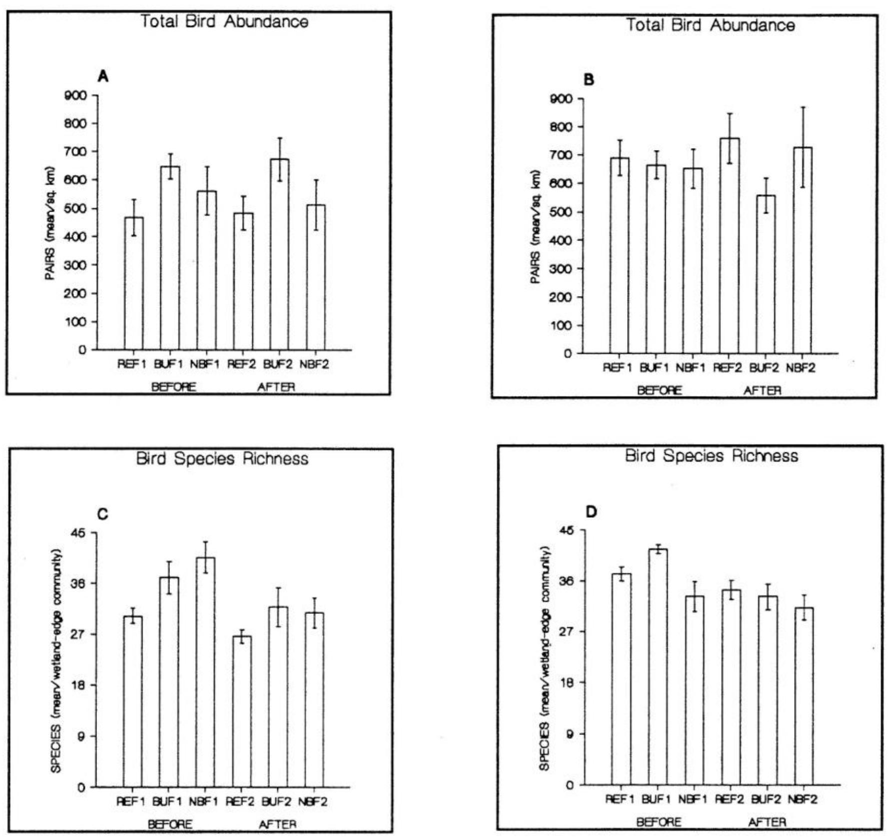


Figure 6. Total bird abundance in conifer (A) and mixedwood (B) wetland/edges, and bird species richness in conifer (C) and mixedwood (D) wetland/edges before (Time = 1) and after (Time = 2) timber harvest. Bars represent means, vertical lines ± S.E., n = 30. REF = reference, BUF = 30 m buffer, NBF = no buffer. For A, B, C, and D no significant interaction (P>0.05, 2-way ANCOVA) was detected.

Table 8. Bird territories (mean pairs/sq km ± S.E.,) of wetland/edge ecosystems by treatment (n=5) and year for conifer forests. Asterisks indicate significant differences between (2-way ANCOVA before and after harvest) and within (Scheffe's test mean reference vs. 3.0 m buffer or no buffer after harvest only) treatments.

CONIFER FOREST	BEFORE HARVEST (1993)						AFTER HARVEST (1994)					
BIRD SPECIES	Reference		30 m Buffer		No Buffer		Reference		30 m Buffer		No Buffer	
	mean	S.E.	mean	S.E.	mean	S.E.	mean	S.E.	mean	S.E.	mean	S.E.
White-throated Sparrow	42.7	7.7	59.7	5.4	55.0	9.3	36.0	14.0	55.7	14.4	57.5	15.8
Magnolia Warbler	46.4	8.3	65.6	13.0	61.2	14.7	48.0	7.7	49.5	11.9	46.3	21.0
Nashville Warbler	45.1	11.6	54.6	11.7	58.3	12.3	49.7	4.7	57.0	6.5	58.9	18.0
Yellow-rumped Warbler	28.3	7.0	31.2	6.8	25.7	3.3	49.6	8.9	66.6	17.1	51.0	12.0
Ruby-crowned Kinglet	27.1	6.7	31.0	5.7	27.4	10.7	46.0	14.1	48.7	8.2	29.4	8.1
Common Yellowthroat	20.5	7.2	43.7	7.6	30.0	5.9	25.0	6.8	39.4	7.3	24.4	4.5
Ovenbird	38.7	7.2	23.7	11.3	32.6	8.6	26.0	10.7	11.1	5.7	5.8	2.3
Winter Wren	13.7	2.4	32.9	10.5	16.7	4.6	9.1	4.2	45.5 *	10.4	31.5 *	5.5
Swamp Sparrow	16.5	3.1	21.9	6.9	15.7	3.7	9.8	2.9	27.3	6.0	13.9	6.9
Yellow-bellied Flycatcher	9.3	6.6	35.7	4.3	17.2	2.2	13.3	7.5	8.2	5.1	15.2	7.7
Golden-crowned Kinglet	10.3	3.7	21.9	1.4	15.6	3.1	27.0	3.5	15.7	2.6	19.2	12.8
Swainson's Thrush	18.8	4.6	16.8	2.1	17.9	4.2	8.1	5.7	16.9	7.6	9.3	5.6
Black-and-white Warbler	2.2	2.2	6.7	2.7	14.3	5.4	1.4	1.0	14.2	5.9	8.8	6.0
Hermit Thrush	13.3	2.1	15.7	2.2	7.7	2.6	9.2	3.1	11.6	6.3	7.1	2.8
Red-breasted Nuthatch	10.2	3.6	14.5	4.4	13.9	3.5	0.0	0.0	11.0	4.0	4.0	2.4
Northern Flicker	6.8	3.2	10.1	1.3	7.7	1.5	1.9	1.2	7.9	4.2	7.8	2.7
Chestnut-sided Warbler	7.1	5.8	9.2	7.9	7.2	2.9	6.9	4.6	4.4	2.5	7.5	4.6
Black-capped Chickadee	13.8	2.3	9.1	4.5	11.0	4.6	4.6	2.5	2.5	1.6	6.4	3.7
Canada Warbler	2.0	2.0	1.1	1.1	2.5	1.6	9.0	6.7	4.5	4.5	4.8	3.4
Yellow-bellied Sapsucker	7.3	4.2	9.1	3.8	5.9	2.3	1.2	0.8	12.0	4.2	6.2	2.1
Solitary Vireo	7.7	2.0	16.0	4.6	5.9	2.4	7.4	4.6	21.5	8.2	7.8	4.5
Blackburnian Warbler	13.1	2.4	4.0	2.1	10.1	4.5	13.6	5.4	16.0	8.9	3.5	1.5
Red-eyed Vireo	7.4	4.6	1.7	1.1	7.6	4.8	2.5	2.0	6.0	4.3	7.1	4.7
Brown Creeper	12.5	3.7	18.7	6.9	13.2	4.1	4.1	4.1	13.0	2.7	0.0 **	0.0
Black-throated Blue Warbler	3.9	1.9	3.4	2.2	8.3	3.5	4.8	1.3	1.1	1.1	1.4	0.9
Purple Finch	1.3	0.8	6.7	2.5	11.6	3.6	4.9	2.3	7.6	3.7	3.9	2.8
Black-throated Green Warbler	3.0	2.0	0.0	0.0	2.5	1.7	6.7	6.7	1.1	1.1	0.9	0.9
Dark-eyed Junco	8.0	5.1	12.3	2.6	3.0	1.3	6.2	6.2	18.5	10.3	13.7	8.1
Gray Jay	4.6	2.1	8.2	2.8	7.4	1.6	2.1	2.1	10.1	3.3	3.2	2.5
Alder Flycatcher	1.4	1.0	8.5	7.0	0.0	0.0	5.3	3.5	4.3	3.0	0.0	0.0
Total Abundance (98 species)	466.9	63.5	647.0	43.5	560.4	85.1	482.4	59.4	672.7	76.5	512.4	88.4
Species Richness "	30.2	1.4	37.0	2.9	40.6	2.8	26.6	1.2	31.8	3.5	30.8	2.6
Guild Abundance (63 species)	248.4	46.7	388.9	31.6	311.1	38.3	248.5	36.7	412.1	39.0	334.8	40.3
Species Richness "	16.4	1.1	20.0	1.9	21.4	2.2	14.2	0.7	18.0	2.9	18.6	1.9

* $P < 0.05$

** $P < 0.01$

Table 9. Bird territories (mean pairs/sq km ± S.E., of wetland/edge ecosystems by treatment (n=5) and year for mixedwood forests. Asterisk indicates significant differences between (2-way ANCOVA) and within (Scheffe's test) treatments.

MIXEDWOOD FOREST	BEFORE HARVEST (1993)						AFTER HARVEST (1994)					
BIRD SPECIES	Reference		30 m Buffer		No Buffer		Reference		30 m Buffer		No Buffer	
	mean	S.E.	mean	S.E.	mean	S.E.	mean	S.E.	mean	S.E.	mean	S.E.
White-throated Sparrow	53.4	13.1	62.5	7.6	70.9	11.5	74.2	4.0	47.3	9.6	81.6	20.1
Magnolia Warbler	73.0	10.5	74.5	11.6	63.9	14.2	74.2	12.2	39.0	10.1	49.0	7.6
Nashville Warbler	37.9	6.6	37.3	4.4	48.7	11.1	62.7	13.7	31.4	12.3	57.1	8.8
Yellow-rumped Warbler	21.8	2.4	27.1	3.3	39.7	11.0	49.8	5.1	50.1	6.4	84.7	32.4
Ruby-crowned Kinglet	27.3	6.3	10.4	5.4	29.7	14.1	42.1	3.9	40.5	2.6	61.0	23.3
Common Yellowthroat	45.4	5.0	30.7	6.1	46.8	8.0	32.6	6.7	14.5	4.6	22.1	4.1
Ovenbird	45.0	10.7	30.6	4.1	26.5	10.8	45.3	10.0	33.4	15.1	25.5	13.2
Winter Wren	20.7	3.2	24.7	5.3	21.6	8.0	32.2	4.0	37.8	4.9	46.1	15.3
Swamp Sparrow	31.7	7.3	33.6	9.3	23.7	8.0	30.7	11.2	17.6	3.7	31.4	8.7
Yellow-bellied Flycatcher	21.7	2.5	20.7	5.2	32.1	10.9	27.4	7.8	7.1	2.6	30.9	10.8
Golden-crowned Kinglet	6.3	3.3	10.7	3.8	15.2	2.3	11.1	5.0	29.9	8.3	41.0	19.4
Swainson's Thrush	11.4	5.4	18.8	2.5	9.6	1.6	21.6	5.4	5.4 *	2.1	21.3	4.0
Black-and-white Warbler	20.2	3.6	16.1	3.7	8.9	4.0	30.0	9.7	15.0	4.8	8.1	3.7
Hermit Thrush	12.3	1.5	11.0	6.7	19.1	9.1	9.4	3.9	15.5	5.4	9.4	2.8
Red-breasted Nuthatch	17.8	2.9	18.1	6.7	22.3	3.6	10.0	3.0	4.3	2.5	10.8	3.6
Northern Flicker	7.0	1.9	16.0	3.1	11.6	2.8	9.5	5.2	9.0	4.4	24.4	11.5
Chestnut-sided Warbler	14.9	8.3	10.5	5.3	12.6	5.9	14.1	8.6	5.4	2.6	3.9	2.4
Black-capped Chickadee	12.3	5.0	12.2	3.6	11.7	4.9	10.8	3.1	9.3	1.0	0.0	4.8
Canada Warbler	15.7	4.9	21.4	4.8	2.2	1.4	18.5	12.0	24.7	12.0	3.4	2.8
Yellow-bellied Sapsucker	12.0	5.5	14.9	2.8	5.9	2.7	13.8	7.1	9.5	2.6	7.9	4.0
Solitary Vireo	2.0	1.2	5.2	3.1	14.0	8.4	2.3	1.5	7.9	4.2	6.2	2.1
Blackburnian Warbler	13.0	2.7	9.4	4.1	4.2	2.4	8.1	4.3	4.7	2.2	0.0	0.0
Red-eyed Vireo	19.1	4.0	11.3	4.2	3.7	2.5	15.8	7.5	10.4	4.6	5.8	3.7
Brown Creeper	12.4	3.1	7.1	3.2	7.4	3.0	6.2	4.1	1.6	1.6	0.1	0.1
Black-throated Blue Warbler	19.5	6.5	10.8	2.7	6.4	5.1	13.4	3.7	3.7	3.1	9.2	7.7
Purple Finch	7.4	2.0	6.8	3.2	12.1	4.9	8.9	2.6	3.7	3.1	3.6	2.3
Black-throated Green Warbler	12.8	8.0	4.6	2.8	2.6	2.6	17.7	7.2	6.3	5.4	8.1	5.0
Dark-eyed Junco	0.0	0.0	0.0	0.0	0.5	0.5	0.2	0.2	0.0	0.0	2.1	2.1
Gray Jay	1.4	1.4	6.1	2.6	6.4	3.8	6.8	1.8	0.0	0.0	1.2	1.1
Alder Flycatcher	0.0	0.0	5.1	3.2	8.2	8.2	0.8	0.8	8.6	4.2	11.9	5.1
Total Abundance (98 species)	690.1	63.1	665.2	49.1	652.0	68.1	759.7	87.4	558.7	61.2	728.0	142.2
Species Richness "	37.2	1.3	41.6	0.8	33.2	2.6	34.4	1.7	33.2	2.3	31.2	2.2
Guild Abundance (63 species)	352.7	19.3	390.2	12.5	413.8	43.7	422.3	52.4	301.3	24.8	452.5	76.3
Species Richness "	18.8	1.0	23.8	1.7	19.2	1.3	18.0	1.3	20.2	1.4	17.4	1.6

* $P < 0.05$

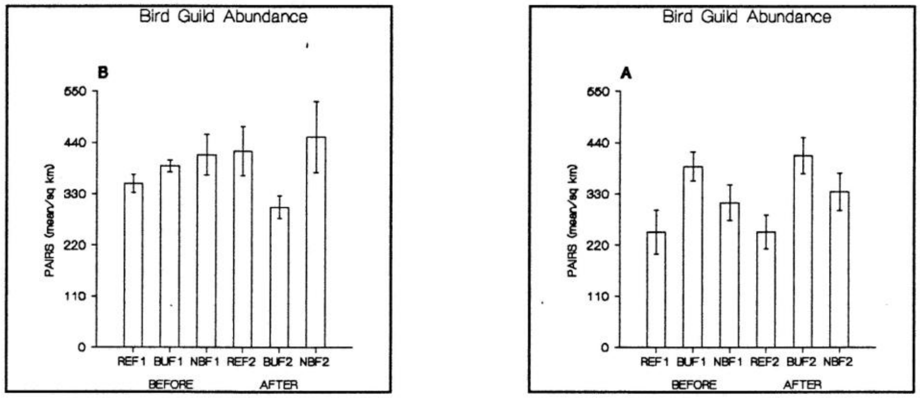


Figure 7. Bird guild abundance in conifer (A) and mixedwood (B) wetland/edges before (Time = 1) and after (Time = 2) timber harvest. For A and B no significant interaction (P>0.05, 2-way ANCOVA) was detected. See Figure 6 for legend details.

Changes in densities for some species of birds occurred in conifer and mixedwood wetland/edges after logging. In this conifer forest, a significant interaction ($p<0.05$, 2-way ANCOVA) between years and treatments was found for the winter wren (*Troglodytes troglodytes* (Fig. 8A) suggesting an effect due to timber harvesting. Significant increases in density ($p<0.05$, Scheffe's test) were evident with greater bird pair densities in 30 m buffer (BUF2) and no buffer (NBF2) treatments compared to reference areas. However, this influence of buffer relative to unbuffered wetlands on Winter wren densities was not evident ($p>.05$). Brown creeper densities also significantly decreased ($p<.05$, Kruskal-Wallis test) after harvesting (Fig. 8B) in the conifer forests. Although densities were different between buffered and unbuffered areas after harvesting for the Brown creeper, they were not significant.

In the mixedwood forest, there was a significant interaction between years (time 1 = 1993, time 2 = 1994) for the Gray jay (Fig. 8C) and Swainson's thrush (Fig. 8D) densities (breeding pairs/km^2) in wetland edges after selective cutting. No difference ($p>.05$, Scheffe's test) in densities were observed between buffered and unbuffered treatments for the former species, likely because numbers were low. However, significant decreases ($p<.05$) were found for the latter bird species (Fig. 8D).

White-throated sparrows were the most abundant species overall the two-year census (Tables 8 and 9), yet there were no significant differences ($p>0.05$) after timber cutting for this riparian/edge species in either the mixedwood or coniferous forest groups. The same occurred ($p>0.05$) for frequently recorded wetland-dependent species such as common yellowthroat (*Geothlypis trichas*) and swamp sparrow (*Melospiza georgiana*). Density of these wetland/edge species remained unaffected ($p>0.05$) by the impact of timber harvesting and the use of treatments. Other trends were observed in the data such as golden-crowned kinglet (*Regulus* *satrapa*), an upland forest species, which was also frequently observed within wetland/edges but did not change significantly (p>0.05) as a result of logging disturbance or use of buffers. Golden-crowned kinglet pairs per km^2 did increase in the mixedwoods after logging but remained about the same in the conifer wetland/edges after both no buffer and 30 m buffer treatments (Tables 8 and 9).

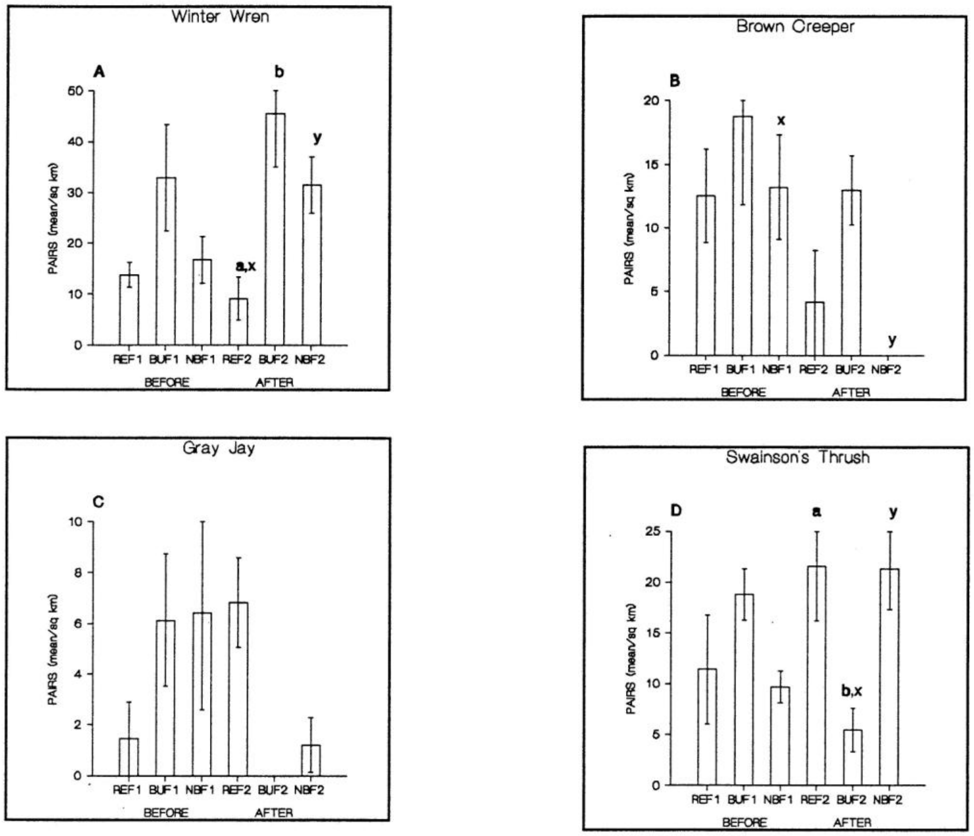


Figure 8. Bird density for winter wren (A), brown creeper (B) in conifer wetland/edges, and gray jay (C) and swainson's thrush (D) in mixedwood wetland/edges before (Time = 1) and after (Time = 2) timber harvest. For A, B, C, and D significant interaction (P<0.05, 2-way ANOVA for D, ANCOVA for A, C and Kruskal-Wallis for B) was detected. No buffer treatments in A were different after harvest (Scheffe's, P<0.05) compared to references (x,y) and in D (Scheffe's, P<0.05) in relation to 30 m buffers (x,y). No buffer treatments in B were different before (x) and after (y) disturbance (Kruskal-Wallis, P<0.05). Buffer treatments in A and D were different after harvest (Scheffe's, P<0.05) in relation to references (a,b). See Figure 6 for legend details.

Discussion

Bird Density and Species Richness

Total bird density and species richness in both coniferous and mixedwood wetland/edges did not change significantly due to clear-cutting or selective harvesting between years and within treatments. Neither logging nor the use of 30 m buffers had a significant effect on overall richness and density. Species richness has been shown by others (Johnson and Brown 1990, DeGraff and Chadwick 1987, Welsh 1981, 1987) not to change after logging. Also, in boreal mixedwood forests of Ontario, Welsh and Fillman (1980) and Welsh (1981, 1987) found that species diversity and density did not change significantly over time after timber cutting in selectively harvested stands older than one year.

This study showed higher species richness in forests prior to cutting than DeGraff and Chadwick (1987) found in New England hardwoods and mixedwoods. They reported that species richness was similar in all forest ages including uneven-aged hardwood stands at about 28-30 species/km^2. In 1993, an average for combined treatments of 36 species in mature conifer wetland/edges were recorded and an average of 37 species in mature mixedwood wetland/edges near Temagami. Conversely, an average of 31 species in conifers and 32 species in mixedwoods for combined buffers and no buffers were recorded after cutting. Although species richness tended to decline after cutting, the changes were not enough for a significant times-by-space interaction (2-way ANCOVA).

Guild Density

In the first season after cutting there were no significant changes in densities of species in the guild dependent on wetlands and riparian edges for their habitat needs. Some changes did occur in terms of species composition with shifts in species among treatments. Site fidelity, whereby birds simply returned to the same area in which they bred the year before, may have affected these results. The authors did not attempt to address reproductive success and it may be that while individuals returned after timber cutting, they may not have been successful breeders within the wetland-riparian guild.

Fluctuation in bird densities of individual species may have been "noise" associated with annual fluctuations in abundance. Also, in coniferous forests as well as mixedwoods, features of cut areas may have produced some variation in results between treatments in our study. Trees in the smallest diameter class remained the same before and after logging. Pockets of timber were left uncut adjacent to some wetlands (i.e. where no buffers were designated) due to small tree diameter size (e.g. <15 cm) and unmerchantable species composition (i.e. cedar and balsam fir). Crawford and Titterington (1979) found that balsam fir stands had lower bird densities averaging only 128 pairs/40 ha and a total of 20 species. Perhaps pockets of uncut forest were used as temporary refugia during the post-harvest census, and birds may in fact move on to more preferable habitat in the second year.

Another reason for no significant differences in guild density, species richness and individual species abundance may have been related to sample size. The power of statistical tests was reduced as a result of separating mixedwood wetland/edges from conifers. A larger sample size may have given somewhat different results.

Species Composition

Shifts in use of treatment habitats due to logging occurred for four species. Two (winter wren and brown creeper) were associated with clear-cut jack pine/black spruce coniferous wetland/edges and two (gray jay and Swainson's thrush) in selectively harvested deciduous-coniferous mixedwood wetland/edges. Welsh (1987) also found significant differences in bird species composition in mixedwood stands of various age classes and Crawford and Titterington (1979) found changes in species composition in spruce-fir stands in Maine.

Winter wren

Timber disturbance had a positive effect on winter wrens in this study in that they increased after logging in coniferous clear-cut in both buffers and no buffers relative to references (Fig. 7). The same trend occurred in selectively cut mixedwood forests but results were not significant. This is likely because of changes in forest structure and type. Winter wrens breed primarily in coniferous forests, cedar swamps and spruce bogs (Cadman et al. 1988) but our data did not show this preference for conifer over mixedwood forests (Table 9). This species often nests in a natural cavity or under stumps, amid roots of an upturned tree, and occasionally in an old woodpecker hole (Ehrlich et al. 1988). Winter wrens nest and forage on or near the ground and have been associated with slash and brush cover and cool, moist streamside habitat (Barrows 1986). Johnson and Brown (1990) found that winter wrens favoured disturbance, and nested in the selectively harvested buffer strip but not in the undisturbed lakeshore. This species has been shown by others to be positively associated with logged areas (Webb et al. 1977, Titterington et al. 1979, Freedman et al. 1981).

Brown creeper

Brown creepers decreased significantly in the clear-cut conifer forest without buffers. Declines also occurred in the no buffered mixedwood wetland/edges but differences were not statistically significant. Preferred breeding habitat for this species is mature woodland, especially in wet areas, and often with large snags. Coniferous, deciduous, mixedwoods, and bogs are used most frequently (Cadman et al. 1988).

The results of this study show that the brown creeper is more strongly associated with conifer rather than mixedwood forests. The brown creeper can build its nest under loose bark (Ehrlich et al. 1988). Significant decreases in brown creepers were likely attributed to the removal of snags in clear-cut logging of conifer forests. In our study, snags in different categories of decomposition were significantly reduced beyond 30 m buffers.

In Maine, Johnson and Brown (1990) found differences in species composition in an 80 m lakeshore buffer compared to uncut shoreline. Species that nested in undisturbed lakeshore but not in the buffer strip included the brown creeper. They speculated that fewer numbers of species such as the Brown creeper were probably related to the lower number of snags in the buffer strip which had been selectively harvested. Density and diversity of cavity nesters has been shown to be related to increasing availability of snags (Mannan and Meslow 1984). Brown creepers have been found to be positively associated with densities of large conifers as they are secondary cavity nesters and depend primarily on snags to provide nesting substrate (Mannan et al 1980). Brown creepers have also been reported to avoid hardwoods (Mariani 1987). Large conifer snags persist much longer than hardwood snags (Cline et al 1980). McGarigal and McComb (1992) found a greater number of large conifer snags in upslope areas within their study area and it was thought that additional snags may have contributed to higher species richness in upland as compared to streamside riparian areas. They recommended leaving large conifers and large conifer snags in riparian areas thereby making these areas more suitable to upland bird species. Leaving conifer snags standing after logging operations would increase habitat availability for dependent species such as Brown creeper.

Gray jay

There was a significant areas-by-times interaction in gray jays after logging in mixedwood forests. A large increase occurred in references after cutting coupled with decreases in both 30 m buffer and no buffer wetland/edges. Decreases in bird density which occurred relative to references were not large enough to be significant, possibly due to the low numbers of gray jays in the census. No significant treatment differences could be detected using Scheffe's pairwise comparison test. Gray jays breed in coniferous (spruce and fir), mixedwood coniferous-deciduous forest, open woodland, and bogs (Cadman et al. 1988, Ehrlich et al. 1988). An increase in density in references in mature mixedwood wetland/edges may demonstrate the extent to which annual variation occurs in some of the bird species although it is worth noting that out of the 30 dominant species, none showed a significant difference in references between years.

Swainson's thrush

Logging disturbance (selective cutting in mixedwoods) may have had a positive influence on Swainson's thrushes which breed in shrubby riparian habitat because they were significantly lower in abundance in buffered mixedwood wetland/edges and higher in no buffer mixedwood wetland/edges. Only minor changes occurred in Swainson's thrush numbers in conifers. Between years (1993 and 1994), both reference and no buffers increased in density while buffers decreased. This species is known to breed along coniferous forest edge (especially where damp) and in riparian thickets (Ehrlich et al. 1988). The Swainson's thrush prefers conifers, especially spruce and fir, but also frequents deciduous shrubs (Cadman et al. 1988) and our data showed equal densities in both forest types (Table 9). Swainson's thrushes nest on the ground and have been associated with dense thickets near streams (Bent 1949). In a study by McGarigal and McComb (1992) in Oregon, streamside and upslope areas in mature unmanaged coniferous and mixedwood forests were compared in terms of species richness, diversity, evenness, and individual species abundance and both Swainson's thrushes and Winter wrens showed significant associations with streamsides (McGarigal and McComb 1992).

Other studies have shown that breeding bird species composition changes quite rapidly over time after complete clear-cutting. Many of the earliest-arriving birds decline in just a few years as habitat conditions change (DeGraff and Chadwick 1987). In winter-logged stands larger than 10 ha in New England, DeGraff (1991) found that during the first growing season after northern hardwood stands were clear-cut, species such as white-throated sparrows and winter wrens were commonly abundant. If snags with old woodpecker cavities were present, eastern bluebirds (*Sialia sialis*) and northern flickers (*Colaptes auratus*) were abundant also. DeGraff (1991) concluded that bird species richness is enhanced by leaving standing snags in clear-cuts. In our study, snags were removed in the conifer clear-cut. Two years after clear-cutting, DeGraff (1991) showed a variety of species including common yellowthroat (*Geothlypis trichas*), chestnut-sided warbler (*Dendroica pensylvanica*), cedar waxwing (*Bombycilla cedrorum*), american goldfinch (*Carduelis tristis*), mourning warbler (*Oporornis philadelphia*), Swainson's thrush and american redstart (*Setophaga ruticilla*) to shift in species composition. Northern flicker may still be present but Eastern bluebird and Winter wren may not. During the subsequent 12 years, bird species composition changed significantly, but the number of species usually does not change much.

Wetland size

Of the thirty bird species analyzed, some were significantly influenced by wetland size (1-way ANOVA). Wetland size was used as a covariate in ANCOVA and covered each wetland area and adjacent 30 m riparian buffer zone including forest to 50 m from the aquatic terrestrial interface. Wetland/edge ecosystems ranged from 5 to 32 ha in size with an average of 12 ha (Table 1). Larger wetland/edge area offers a more diverse array of habitat niches for wildlife due to increased wetland vegetation complexity and range in percent coverage of open water, both of which vary considerably especially in marsh wetlands. In Washington, Milligan (1985) studied 23 urban wetlands and found the amount of buffered wetland edge to be moderately positively correlated with bird species diversity, relative abundance and breeding numbers. She concluded that wetland size and the amount of wetland edge were more important than buffer size. Her results showed that there was only a minor increase in bird species diversity with increased buffer widths of 15 m, 30 m, and 60 m. In addition, she found that bird species richness and breeding bird density were correlated with wetland habitat complexity (i.e. number of wetland plant communities present) which increased with wetland size.

Forest structure

Riparian forests vary in suitability as bird habitat depending on stand structure and several other characteristics. Bird communities are associated with vertical foliage layers (MacArthur and MacArthur 1961), total foliage volume (Willson 1974), habitat patchiness (Roth 1976), and stand successional stage (Shugart and James 1973). Vertical complexity of forest vegetation (i.e. the diversity of vegetation heights and the density of foliage at those heights) is positively associated with breeding forest-bird diversity (MacArthur and MacArthur 1961). Horizontal diversity or patchiness (e.g. distribution of timber size classes and openings) has been shown to be better than vertical habitat heterogeneity for predicting bird species numbers (Roth 1976 in DeGraff 1991). DeGraff (1991) reported a close relationship between habitat structure and bird species composition in predicting the effects of forest management on breeding birds. Hydroperiod also influenced bird habitat and bird community associations in riparian forests (Swift et al. 1984).

Extended flooding may create habitat for aquatic birds, protect colonial nesters such as great blue herons from predators and, in killing trees, provides woodpecker and wood duck cavity nest sites.

DeGraff and Chadwick (1987) found that breeding bird species composition varies with timber size-class. In describing bird associations within upland forests, DeGraff and Chadwick (1987) state that in New England northern hardwood forests, bird species composition varies with timber size-class, stand area, the presence or absence of softwood in the stand, within-stand features such as cavity trees, openings, and wet areas, and the presence of understorey and mid-canopy vegetation layers. DeGraff and Chadwick (1987) found that many breeding birds are closely related to forest cover types and timber size-classes. On that basis, in order to tie into forest structure attributes useful in explaining changes in birds after timber cutting, the inventory in this study collected forest data according to timber size-classes as expressed in four diameter classes, tree species, and the presence of snags according to diameter classes and stages of decay (Maser et al. 1979). As recommended by DeGraff and Chadwick (1987), forest cover types and timber size-classes were selected as an appropriate method of describing the forest before and after timber harvesting in relation to the bird census.

Of characteristics measured (DeGraff and Chadwick 1987), snag size was by far the most important in terms of bird species abundance. Vegetation strata, tree/sapling species richness, size and density have been reported (Stauffer and Best 1980) to be important in terms of habitat selection. Stauffer and Best (1980) suggested that modifications to vegetation that result in greater structural diversity would likely benefit the greatest number of species. However, they cautioned that if maintaining maximum bird species diversity is the management goal, this may be detrimental to rare species.

Effects of selective or clear-cut harvesting

Selective cutting in mixedwoods that leaves a considerable amount of hardwood and smaller diameter softwoods standing also ensures that a considerable degree of vertical structure also remains. In selectively-harvested stands, post-harvest conditions could provide suitable habitat for birds that were present prior to cutting. Welsh (1987) suggested that similar bird density and species diversity in all age classes of cut mixedwood stands may be related to the fact that they were selectively harvested. Some habitat requirements may still be met after harvest if enough forest structure remains. While maintenance of stand structure may explain lack of change in bird density and species richness in the mixedwoods, it does not explain similar findings in clear-cut conifer wetland/edges. Here, all trees were removed as in wetland/edges without buffers. Yet, total bird abundance and species richness did not change appreciably with the exception of decreases in yellow-bellied flycatchers (*Empidonax flaviventris*) and brown creepers in the buffers and Winter wren increases in the unbuffered wetland/edges.

The impact of forestry operations on populations of breeding birds in northeastern hardwoods was examined by Webb et al. (1977). The authors examined stands which had 0, 25, 50, and 100 percent of the merchantable timber removed. Webb et al. (1977) found no significant changes in breeding bird abundance, but showed changes in species composition. These bird patterns found by Webb et al. (1977) are similar to those in our wetland study. Freedman et al. (1981) found no statistically significant differences in treatments three to five years after disturbance in the total density or richness of breeding birds in a Nova Scotia hardwood forest that had been harvested by clear-cutting, strip-cutting, and thinning and compared to reference plots. They did, however, find changes in species composition with mature forest birds replaced by early successional species on the clear-cuts. In Maine, it has been shown that for coniferous forests, clear-cutting generally causes short-term decreases in both abundance and breeding diversity, followed by rapid recovery to abundance and diversity levels that frequently exceed those of uncut control forests (Crawford and Titterington 1979).

Triquet et al. (1990) compared watersheds with and without buffer strips, before and after timber harvesting. The effectiveness of state Best Management Practice (BMP) careful logging recommendations were tested against conventional clear-cut logging as to the effects of leaving a riparian buffer, 15 to 23 m along each side of a stream. Bird species richness and diversity were found to be highest on the uncut control area of mature forest and on the buffered area in the modified cutting area according to BMP guidelines. Bird diversity was lowest on the clear-cut without harvesting restrictions or riparian bufferstrip. Bird abundance increased by 21 percent and 23 percent relative to pre-harvest values on the two clear-cuts during the second growing season after cutting. Conner and Adkisson (1975) in Triquet et al. (1990) also found significantly more birds in clear-cuts (3–12 years old) than in mature forests in Virginia. In our study, bird abundance went up 8 percent in the no buffer conifer wetland/edges within the large clear-cut and by 12 percent in the deciduous/conifer mixedwoods. An increase of 4 percent also occurred in buffered wetland/edges and 3 percent in conifer references yet buffered mixedwood bird abundance decreased by 16 percent while increasing 10 percent in the references.

Scope and limitations

Several factors which may have confounded our findings: 1) The study was initially designed for clear-cut timber harvesting only. In order to keep the sample size as large as possible, it was intended to incorporate only one harvesting approach. However, mixedwood forests were selectively cut with reasonable stand structure remaining after cutting. The study design was modified and sample size was reduced from a combined total of 30 wetlands to 15 in mixedwood and 15 in coniferous forests. This reduced the power of statistical tests. (2) Size of wetland/edges were variable (comprised of wetlands and adjacent riparian area of upland forest to fifty meters). Habitat communities averaged 12 ha but ranged from 7 to 32 ha. For some species, total habitat size was highly significant and this may have complicated the results. (3) Both marsh (22) and fen (8) wetland types were represented in the study. There may have been bird species association with wetland types but in order to keep the sample size large enough for use in analysis, the two wetland types were combined. (4) Two highly competent birders completed the breeding bird census each year but a different observer was used for the post-harvest breeding season. Standardized methods were established but some degree of observational bias may have occurred.

Sampling was limited to wetlands and riparian edge communities of mature mixedwood and coniferous stands immediately before and after timber was cut. Upland forests were not sampled except within riparian ecotones up to a distance of 50 m from wetland perimeters. Marshes and fens were included but other wetland types such as bogs and swamps were not. Bird observations were limited to diurnal birds during the breeding season. Seasonal changes in wetland and riparian habitat use were not assessed. Waterfowl were not specifically addressed by the selected sampling methodology. Finally, since reproductive success was not undertaken, it is not known if birds were successful breeders in post-harvest habitats.

Conclusions

Total bird abundance and bird species richness were not significantly different due to timber harvesting or the use of 30 m buffers in either conifer or mixedwood forests. For species having specific habitat affinity for wetland and riparian environs and which were grouped into a habitat guild, total bird abundance and bird species richness were also not significantly altered. Bird abundance and species richness were found to be higher in wetlands and adjacent riparian zones associated with mixedwood forests compared to coniferous forests. Size of wetland and riparian habitat had a significant effect on some bird species, total bird abundance and species richness. Some changes in bird species composition did occur. For example, brown creeper density was significantly reduced when all trees and snags were removed by clear-cut logging in coniferous forests. Winter wren density greatly increased in the conifer clear-cut both in areas with and without buffers as this species uses over-turned stumps and other logging slash in nesting. Gray jay density increased considerably between years in uncut mixedwood forests, likely due to annual variation. Swainson's thrush density was significantly reduced in areas where 30 m buffers were applied in mixedwood forests. Thus, timber harvesting caused shifts in bird species composition and density. Some species increased while others decreased after timber harvesting.

OWL CALL SURVEY

Introduction

For this study, on the third weekend in March for the years 1993 through 1995, a survey of nocturnal owls was completed by a group of approximately 20 volunteers from the TAA, Nipissing Field Naturalists Club, Sudbury Ornithological Club, OMNR staff and other interested individuals. The study area included conifer and mixedwood wetland/edges with calls per station recorded for 26 wetland/edge habitats.

Methods

March was selected as an appropriate time to conduct the survey in relation to territorial establishment for confirmed breeders in the local area which included the great horned owl and barred owl (Cadman et al. 1988). Shepherd[2] recommends midnight to 6:00 AM as best for significantly higher numbers of calling great horned owls while Cooperrider et al (1986) stated that nesting barred owls appear to be most responsive near the middle of the night. Great horned owls and barred owls are the predominant resident species locally, but other transient species were also included on our vocalization tape including boreal, saw-whet and great gray owls.

The survey was conducted using periodic tape-recorded broadcasting of owl calls followed by periods of silence. Each crew had a set of dual auxiliary speakers connected to a pocket-sized cassette player. The volume for each cassette player had been electronically standardized for all crews. Observers snowshoed along established access trails to the middle area of each study wetland. Speakers were pointed in opposite directions in order to broadcast calls over as wide an area as possible (e.g. 200–300 m). Prior to playing the tape, an initial 5-minute listening period was held. The taped call sequence consisted of a series of five owl calls followed by varying periods of silence. The tapes played approximately 2-minute calls by each of the above-mentioned owls. Including the waiting periods before and after playing the tape, the entire survey required a total of 41 minutes at each wetland.

Crews consisted of two or three people and each crew visited three to five wetland/edges, organized in such a way that wetlands in close proximity were all done by the same crew to avoid overlapping broadcast calls. Most crews recorded owl calls from 2100 hrs, returning by 0400 hrs but some, having considerably further to travel returned by about 0700 hrs. Information collected in the field included owl species as determined from the tape, time of response and approximate location. Other weather information was recorded concerning evening air temperature, snowfall, wind speed and direction, cloud cover and moon phase.

Statistical Analysis

The number of owls for each of 26 wetlands surveyed, along with the average number of owl responses per team hour of observation were used in comparing harvested treatment (30 m and no buffer) to non-harvested reference wetland/edge habitats. Data presented in Table 10 are the total number of owls and responses per hour by treatment per wetland/edge. Statistical analysis was not undertaken on owl data as rigorous scientific design was not an objective of the owl study. Had we been able to conduct this survey three times over the course of late winter each year, data would have lent itself to this analyses. Data are presented graphically in Figures 9 and 10 to provide an indication of overall trends in owl populations. This information is not meant to provide conclusive evidence of either changes due to timber harvest or use of 30 m buffers but was obtained in order to detect generalized trends and shifts in habitat use as well as to provide baseline data on local owl species.

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[2]Shepherd, D. 1992. Monitoring Ontario's owl populations: A recommendation prepared by the Long Point Bird Observatory for Ontario Ministry of Natural Resources. Unpub. Rep. 82 p.

Table 10. Total owls surveyed in March 1993, 1994 and 1995 according to coniferous and mixedwood forest types.

	CONIFER FOREST	MIXEDWOOD FOREST
OWL SPECIES	Number of Owls	Number of Owls
Great Horned	11	13
Barred	4	14
Boreal	3	10
Saw-Whet	3	2
Great Gray	2	3
Total	23	42

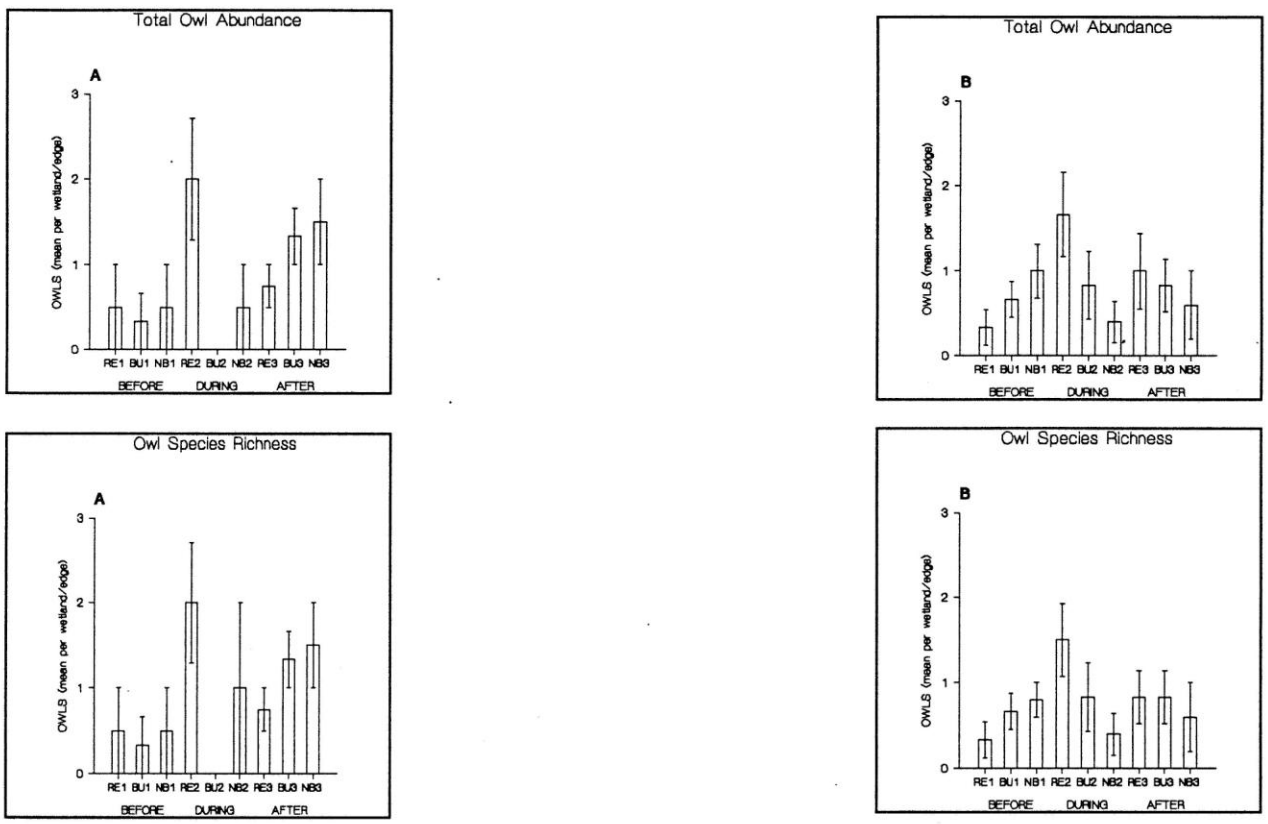

Figure 9. Total owl abundance and owl species richness for conifer (A, n=9) and mixedwood (B, n=17) forest wetland/edge habitats before (Time= 1), during (Time= 2) and after (Time= 3) timber harvest. Bars represent means, vertical lines ± S.E. Re= reference, BU= buffer, NB= no buffer.

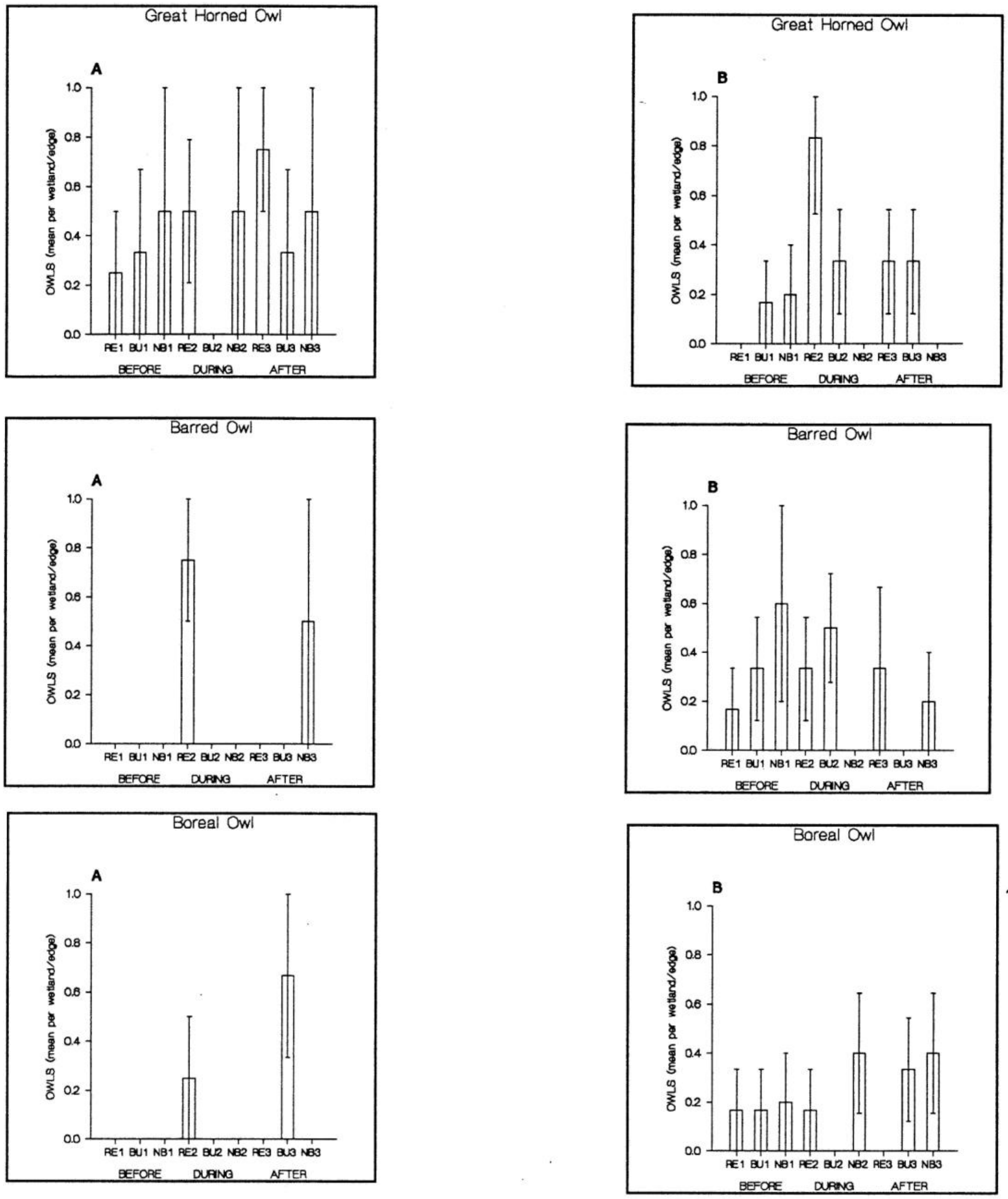

Figure 10. Great horned owl, barred owl and boreal owl abundance in conifer (A) and mixedwood (B) forest wetland/edge habitats before and after timber harvest. See Figure 9 for legend details.

Results

Snowfall may have influenced owl responses in 1993 as most calls came once snow stopped falling between 0430—0600 hrs that year. A total of 65 owls responded to broadcast calls over three surveys using the same 26 wetlands each year. Two great-horned owls actually swooped in to investigate one survey crew and in several instances, owls flew into a perch on a nearby tree. Several calls of all species came after the tape was completed and while crews were observing a period of silence before proceeding to the next wetland. All team members enjoyed listening to the sounds of the night and many heard other wildlife species such as wolves. Weather information for the night-time survey is given in Table 11.

Table 11. Evening weather summary for the owl calling survey conducted March 1993. 1994 and 1995.

WEATHER DATA	1993	1994	1995
Temperature:	- 8° C	- 7° C	0 ° C
Snowfall:	light to 4:30 AM	nil	nil
Wind:	8 km/hr, E	calm	13 km, NNW
Cloud Cover:	100% cloudy	clear	clear
Moon Phase:	waning last quarter to new moon	full moon	waning last quarter to new moon

There were differences in the numbers of owls heard in each forest type and especially between years. It was less evident as to changes within treatments. Table 10 gives a summary of owls observed according to conifer and mixedwood forest types. Table 12 presents data for 26 surveyed wetlands according to forest type, species of owl, treatment and year. Total number of owls observed are given along with calls per survey team hour.

Fig. 9 graphically presents owl abundance (mean per wetland/edge) and owl species richness for conifer and mixedwood forests. Graphs for the three dominant and resident species (in the case of great horned and barred owls) are given in Fig. 10. Trends and shifts are detected in terms of habitat use among treatments. Data are not presented graphically for northern saw-whet owls and great gray owls as both were infrequently observed. The use of volunteers afforded an opportunity to promote discussion to a wider audience as a local newspaper ran a feature story on the owl call survey.

Discussion

Great horned owls are known to prefer boreal forest but are also common in the southern portions of Ontario. This species is a generalist feeder but in the north depends on snowshoe hares (*Lepus americanus*) for prey. Calling for great horned owls in northern Ontario is best in late February to mid-March to coincide with pre-breeding vocalizations. Typical habitat for barred owls in mature mixedwood or deciduous forests preferably adjacent to wet areas in forests containing snags larger than 40 cm that have cavities which are used as nest sites (Shepherd[3]).

[3]Ibid

Table 12. Total owls surveyed in March by species and calls per hour associated with wetland ecosystems (n = 26) by treatment in conifer forests (reference n = 4, 30 m buffer n = 3, no buffer = 2), mixedwood forests (reference n = 6, 30 m buffer n = 6, no buffer = 5), and year (1993 = before, 1994 = during, 1995 = after).

	BEFORE HARVEST				DURING HARVEST			AFTER HARVEST		
CONIFER										
OWL SPECIES	Reference		30 m Buffer	No Buffer	Reference	30 m Buffer	No Buffer	Reference	30 m Buffer	No Buffer
Great Horned	1	1	1	2	0	1	3	1	1	
Barred	0	0	0	3	0	0	0	0	1	
Boreal	0	0	0	1	0	0	0	2	0	
Saw-whet	1	0	0	1	0	0	0	0	1	
Great Gray	0	0	0	1	0	0	0	1	0	
Calls per Hour	0.7	0.5	0.7	3.9	0	0.7	1.1	2.0	2.2	
MIXEDWOOD										
OWL SPECIES	Reference		30 m Buffer	No Buffer	Reference	30 m Buffer	No Buffer	Reference	30 m Buffer	No Buffer
Great Horned	0	1	1	5	2	0	2	2	0	
Barred	1	2	3	2	3	0	2	0	1	
Boreal	1	1	1	1	0	2	0	2	2	
Saw-whet	0	0	0	2	0	0	0	0	0	
Great Gray	0	0	0	0	0	0	2	1	0	
Calls per Hour	0.5	1.0	1.5	2.4	1.2	0.6	1.5	1.2	0.9	

Boreal owls are known to breed outside the study area (e.g. west of Sudbury) but were included as possible transient species. This species is a cavity nester as well, and feeds almost exclusively on red-backed and meadow voles. Damp, mature woodland with deep litter provides suitable habitat to red-backed voles while cutovers, burns and forest adjacent to bogs and drier marshes are best for meadow voles. Old pileated woodpecker and northern flicker nest holes are preferred nest sites for boreal owls. Calling should begin early in March (Shepherd[4]).

Saw-whet owls are common in south-central Ontario on the southern portion of the Canadian Shield but have been known to nest as far north as Cochrane. This species favours coniferous or northern mixedwoods with converted nest sites from the cavities occupied by pileated woodpeckers in deciduous snags such as trembling aspen. Drier, upland areas are preferred habitat for mice (Family: *Cricetidae* or *Zapodidae*) which comprise the majority of saw-whet owl diet so direr uplands should also be well-suited to these owls. Calling surveys should coincide with the breeding period extending from mid- to late-February to the end of March or early April (Shepherd[5]).

Great gray owls were also included on the calling tape even though these owls are a more northerly resident of boreal forests extending to the tree-line. These owls prefer damp habitat and black spruce/tamarack bogs having some bare snags available for use as lookout posts during hunting. Territorial calling occurs a little earlier, in January or February (Shepherd[6]).

Some broad trends in usage of habitats emerged. Table 10 shows that many more owls were heard in mixedwood than coniferous forests. Great horned, barred and boreal owls were the most frequently observed species over the three winters, listed in order of occurrence. Great horned owls were slightly more prevalent in mixedwood forest wetland/edges. Barred and boreal owls were heard most often in mixedwoods while the few saw-whet and great gray were in about equal numbers in conifer and mixedwood forest types.

During the winter of 1994, timber harvesting was active with total owl abundance being much higher in reference wetland/edges compared to 30 m buffer and no buffers. Owl calls per hour (Table 10) increased dramatically as well in references during harvest operations. This trend was observed in both conifer and mixedwood forests (refer to Fig. 9). This 'compression effect' may have been caused by owls moving out of actively logged areas where machinery was being used during the day. Owls may prefer undisturbed areas situated further away and in areas containing study references. Owl species richness also increased along with total owl abundance in references during the winter of 1994 during active logging (see Fig. 9). After timber cutting in 1995, total owl abundance increased in 30 m buffers and no buffers in conifer forest

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[4]Ibid

[5]Ibid

[6]Ibid

wetland/edges and remained about the same in mixedwood forests. Species richness was also higher in 30 m buffer and no buffer in conifer forests after logging but not in mixedwood forests.

The number of owls observed and calls per survey team hour remained about the same in selectively harvested mixedwood forests. A shift in use of both 30 m buffer and no buffer wetland/edges occurred after logging in conifer forests. Only one owl was observed in treatments prior to and during harvesting conifer forests but six were heard in 30 m buffer and no buffers after cutting. This large increase in the number of predatory owls in the clear-cut is likely due to an increase in available food supply in terms of voles (*Clethriomomys*, spp.).

Great horned owls remained at about the same frequency of occurrence in both conifer and mixedwood forests after logging whether or not buffers were used. However, some shifts in habitat use according to individual species after logging did occur. Barred owls were less common in treatments in mixedwood forests after logging. This species is known to prefer mature mixedwoods and may have moved out of these areas after timber cutting. Boreal owls increased in both 30 m buffer and no buffers after logging in both conifer and mixedwood wetland/edges. This species is known to move into an area after timber harvesting or a forest fire, following population increases in voles. Due to small sampling size, no trends could be detected in saw-whet or great gray owls.

No obvious preference was detected in the use of 30 m buffers versus no buffer wetland/edges. During logging disturbance, owls leave an active harvesting site in favour of undisturbed forest and wetlands. After timber harvesting, a trend towards an increase in total owl abundance and species richness occurred, likely due to an increased availability of food for predatory owls.

Conclusions

As with other bird species, a trend towards greater owl abundance and species richness in mixedwoods as compared to coniferous forests emerged. During winter timber harvesting, owl density in uncut references increased in both mixedwood and conifer forests, perhaps due to a 'compression' of owls into outlying uncut forest as a result of logging disturbance. In the breeding season following harvest, a trend was observed whereby total owl abundance was greater in clear-cut conifer forests both with and without buffers. This was likely a result of a large increase in voles after clear-cutting. Total owl abundance remained at about the same level in mixedwood forests in reference to pre-cut levels.

Barred owls prefer mature mixedwoods with large snags for nesting and our study showed a trend towards a decrease in barred owls in mixedwood forests after logging, even though significant changes to forest structure did not occur. Boreal owls increased in both clear-cut conifer and selectively harvested mixedwood forests, likely as a result of increased vole populations after logging.

FUR-BEARER WINTER TRACKING

Introduction

This study examined the effects of clear-cutting and partial timber harvesting in relation to the density of fur-bearers within riparian zones of wetlands. In a study of relative abundance of selected mammals in boreal mixedwood and coniferous forests of northern Ontario, Thompson et al. (1989) showed that track counts were a reliable index of habitat preferences and population trends. Many wildlife species are secretive, and their presence and distribution may be best detected by snow tracking. Wildlife monitoring through use of winter track counts was selected to make comparisons between treatment wetland/edge communities and uncut reference sites in relation to timber harvesting disturbance.

The time since last snowfall is an important consideration in winter track surveying (Thompson et al. 1989). If the interval is too short, animals may not have had an opportunity to move about and leave tracks. If it is too long, the resulting accumulations and overlapping of tracks will be unrepresentative in quantitative data comparisons. Snow is an inconsistent tracking medium that varies in registering the tracks with differences in depth, density, and surface hardness. Also, the appearance of tracks changes over time, depending on air temperature and wind conditions (Hatler 1991). Snow tracking must be conducted when conditions are most suitable and, since good data can be strongly influenced by the weather, flexibility in scheduling is needed. Successful snow tracking depends on a specialized knowledge of animal track identification. In this study, over the winters of 1993 through 1995, two trained trappers conducted the snow tracking surveys to quantify the horizontal movement of different wildlife species relative to timber harvesting.

Methods

Winter track counts of fur-bearing mammals were taken along transects situated within the riparian zone at approximately 15 m from the aquatic terrestrial interface and parallel along each wetland perimeter. Line transects were divided into consecutively numbered 100 m segments and distances were measured using a hip chain. Tracks were counted and recorded only if they crossed a transect. Each track was recorded individually by species and counts were later expressed as tracks per kilometer. Wetland riparian ecotones were assessed this way for 10 days and repeated each month from January to March in 1993, 1994 and 1995. Track counts were conducted 24–96 hours after fresh snowfall. Fifteen wetland/edges were allocated equally to 30 m buffer, no buffer and uncut reference groups. Nine wetlands were in mixedwood forests and six were situated in coniferous forests.

Track count frequency was higher during late winter (e.g. March). As weather became warmer, animal activity increased. Tracking was done in January, February and March from 1993 to 1995. Track counts during the month of March, 1995, were not analyzed because an early spring thaw prevented fresh snow fall for tracking. Although observations were recorded for three winters, 1994 data were not analyzed because logging occurred during the tracking period. In order to achieve consistency in comparing pre- (1993) to post-harvest (1995), data were pooled for the months of January and February.

Statistical Analysis

For the months of January and February, mean counts expressed as tracks per kilometer were analyzed and an average was used for 1993 (pre-harvest) and 1995 (post-harvest) survey years. Because of numerous zero or near zero counts for many of the fur-bearing mammals, 8 species were grouped together (e.g. weasel [*Mustela erminea*], marten [*Martes americana*], red fox [*Vulpes fulva*], fisher [*Martes pennanti*], river otter [*Lutra canadensis*], grey wolf [*Canis lupus*], mink [*Mustela vison*], and lynx [*Lynx canadensis*]) while hare and red squirrels (*Tamiasciurus hudsonicus*) were both analyzed separately. Variance heterogeneity required that all data be log-transformed. Due to the small sample size, data on conifer (n=6) and mixedwood (n=9) forests were pooled for analysis. Variances were within acceptable limits according to the Fmax-test of homogeneity for combined mixedwood and coniferous forest data. Species richness was analyzed using non-transformed data. A 3-way ANOVA for combined fur-bearers (8 species) showed a significant interaction with wetland/edge size (Table 1). Therefore, a 3-way ANCOVA was used. Where significant interaction between years (Time 1 = 1993, Time 2 = 1995) occurred, Scheffe's paired comparisons test was used to examine differences within treatments for each species (e.g. hare, squirrel, combined fur-bearers and total abundance).

As sample sizes were small, data were not separated into mixedwood and conifer forest groups for analysis. Track counts (tracks/km) of fur-bearing mammals within the riparian zones of wetlands were used for total abundance of all species (e.g. 10 species), species richness, and track density for individual species (e.g. hare, squirrel and combined fur-bearers). The null hypothesis tested was that no significant change in fur-bearer use of wetland/edges with or without buffer zones occurred due to timber harvesting.

Results

Total fur-bearer abundance

There was a significant difference after timber cutting ($p<0.05$, 3-way ANOVA) in total fur-bearer abundance for combined conifer and mixedwood wetland/edges (Table 13; Fig. 11). There were no significant differences in mixedwood treatments compared to references after logging ($p>0.05$, Scheffe's test, Fig. 11B). Increases occurred in total fur-bearer abundance among references and 30 m buffers in 1995 compared to 1993 for both conifer and mixedwood wetland/edges (Figs. 11A, B; Table 13). Further, after timber was harvested in conifer forests, total fur-bearer abundance was quite similar in 30 m buffer and uncut references (Fig. 11A). However, a significant reduction was found (p's<0.10, Scheffe's test) in no buffer sites compared to both 30 m buffer and reference wetland/edges.

Species richness

No significant interaction was found for fur-bearer species richness ($p>0.05$, 3-way ANOVA) between years (Time 1 = 1993, Time 2 = 1995) or among treatments (reference, 30 m buffer, no buffer) due to logging (Tables 13, 14, and 15; Figs. 11C, D). Neither clear-cut nor selective logging had a significant effect (p>0.05) on fur-bearer species richness.

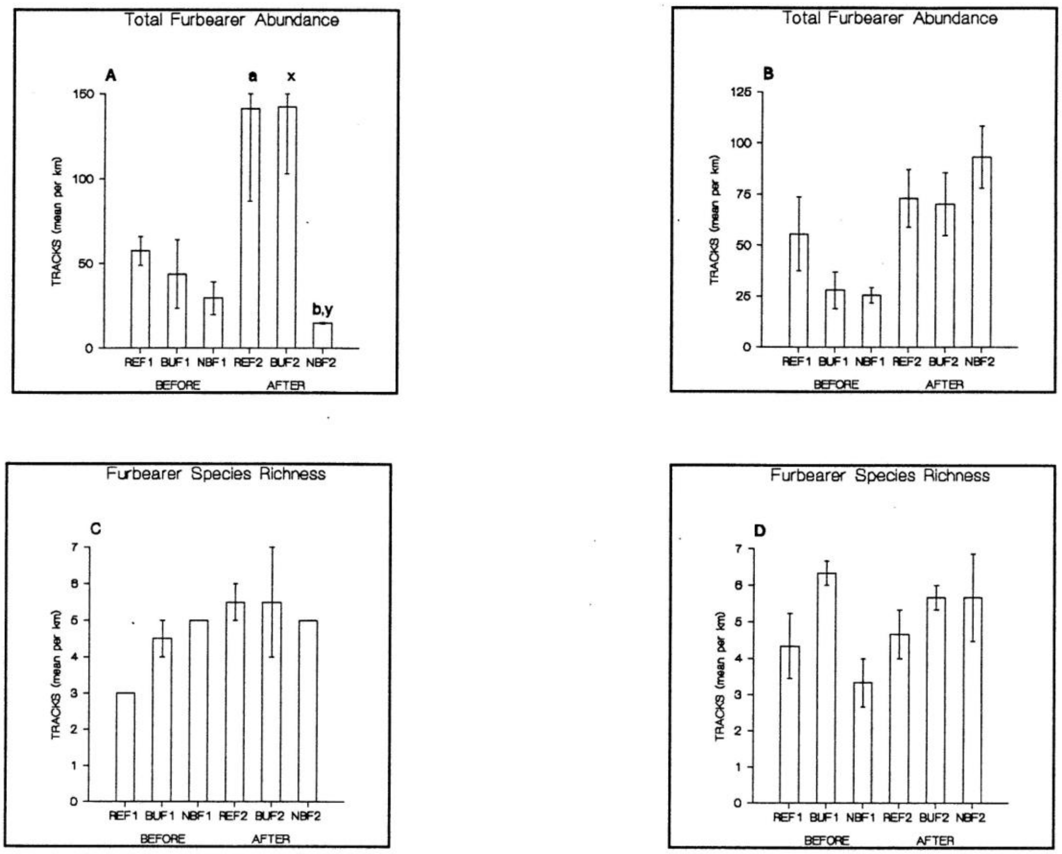

Figure 11. Total fur-bearer abundance in conifer (A, n=6) and mixedwood (B, n=9) riparian forest, and fur-bearer species richness in conifer (C) and mixedwood (D) riparian forest before (Time = 1) and after (Time = 2) timber harvest. Bars represent means, vertical lines ± S.E. REF = reference, BUF = buffer, NBF = no buffer. For combined A and B, significant interaction (P<0.05, 3-way ANOVA) was detected. No buffer treatments in A were different after harvest (P<0.10, Scheffe's) compared to both references (a,b) and 30 m buffers (x,y). For B, C and D no significant differences among treatments (P>0.05, Scheffe's) after harvest was detected.

Table 13. Winter track counts of furbearer density (mean/km ± S.E., n=5) using mean January and February counts within 30 m of wetland edge by treatment and year (1993=before, 1995=after) for combined mixedwood and coniferous wetland/edge ecosystems.

FURBEARER	BEFORE HARVEST Reference		30m Buffer		No Buffer		AFTER HARVEST Reference		30m Buffer		No Buffer	
SPECIES	mean	S.E	mean	S.E	mean	S.E	mean	S.E	mean	S.E	mean	S.E
Snowshoe Hare	40.5	8.3	17.9	10.4	10.6	3.9	39.3	16.5	51.4	17.3	17.9	10.4
Red Squirrel	11.2	3.0	12.5	5.5	12.6	1.5	44.3	8.9	40.2	7.1	35.4	10.7
Other Furbearers	4.7	1.1	3.9	1.6	3.9	1.2	16.8	2.5	7.5	2.2	8.5	2.6
Total Abundance	56.4	10.3	34.3	9.0	27.1	3.8	100.4	25.2	99.1	23.2	61.8	20.9
Species Richness	3.8	0.6	5.6	0.5	4.0	0.5	5.0	0.4	5.6	0.5	5.4	0.7

Other = weasel, marten, red fox, fisher, river otter, grey wolf, mink, and lynx.

Table 14. Winter track counts of furbearer density (mean/km ± S.E., n=2) using mean January and February counts within 30 m of wetland edge area by treatment and year (1993=before, 1995=after) for coniferous wetland/edge ecosystems. Asterisks indicate signifiacnt differences between (2-way ANOVA) and within (Scheffe's test) treatments.

FURBEARER	BEFORE HARVEST Reference		30m Buffer		No Buffer		AFTER HARVEST Reference		30m Buffer		No Buffer		
SPECIES	mean	S.E	mean	S.E	mean	S.E	mean	S.E		mean	S.E	mean	ES
Snowshoe Hare	41.3	1.3	36.7	22.2	16.8	6.5	64.6	40.2	85.0	24.2	0.9	0.7	
Red Squirrel	13.1	4.8	5.0	2.5	9.8	2.1	58.8	8.8	51.9	12.2	9.4	1.8	
Other Furbearers	3.2	2.5	2.1	0.7	2.8	0.9	17.8	5.3	5.5	2.9	4.4	1.5	
Total Abundance	57.5	8.6	43.9	20.4	29.4	9.6		141.3	54.4	142.4	39.3	14.8 *	40
Species Richness	3.0	0	4.5	0.5	5.0	0	5.5	0.5	5.5	1.5	5.0	0	

Other = weasel, marten, red fox, fisher, river otter, grey wolf, mink, and lynx.

* P < 0.10

Table 15. Winter track counts of furbearer density (mean/km ± S.E., n=3) using mean January and February counts within 30 m of wetland edge by treatment and year (1993=before, 1995=after) for mixedwood wetland/edge ecosystems.

FURBEARER	BEFORE HARVEST Reference		BEFORE HARVEST 30m Buffer		BEFORE HARVEST No Buffer		AFTER HARVEST Reference		AFTER HARVEST 30 m Buffer		AFTER HARVEST No Buffer	
SPECIES	mean	S.E	mean	S.E.	mean	S.E	mean	S.E	mean	S.E	mean	S.E
Snowshoe Hare	40.0	15.1	5.3	1.6	6.5	4.1	22.4	3.2	29.0	13.2	29.1	14.3
Red Squirrel	9.9	4.4	17.6	8.1	14.5	1.3	34.7	11.0	32.4	6.6	52.8	2.7
Other Furbearers	5.7	0.6	5.1	2.5	4.6	2.0	16.1	3.2	8.8	3.4	11.2	3.6
Total abundance	55.6	18.1	28.0	9.0	25.6	3.7	73.2	14.2	70.3	15.4	93.2	15.1
Species Richness	4.3	0.9	6.3	0.3	3.3	0.7	4.7	0.7	5.7	0.3	5.7	1.2

Other = weasel, marten, red fox, fisher, river otter, grey wolf, mink, and lynx.

Snowshoe hare and red squirrel

Snowshoe hare and red squirrel (*Tamiasciurus hudsonicus*) comprised approximately 90 percent of total fur-bearer abundance (Tables 13, 14 and 15). There was a significant interaction ($p<0.05$, 3-way ANOVA) between years for both snowshoe hare and red squirrel densities (tracks/km) due to logging (Figs. 12A-D). Decreases in hare and squirrel densities were found in no buffer riparian zones compared to 30 m buffer and references in clear-cutting conifer forests, although the differences within treatments were not statistically significant (p's>0.05, Scheffe's, Fig. 12A, C). A marginal difference in red squirrel density did occur in 30 m buffers (Fig. 12C) between years ($p<0.10$, Scheffe's) in the conifer forest. Snowshoe hare and red squirrel densities increased within treatments and references in mixedwood forests in 1995 (Fig. 12B, D) but these differences were not statistically significant (p's>0.05, Scheffe's).

Combined fur-bearers

A significant interaction was not detected ($p>0.05$, 3-way ANCOVA) in combined fur-bearers (weasel, marten, fox, fisher, otter, wolf, mink, lynx, Tables 13, 14 and 15) after timber harvesting (Figs. 12E, F). An increase in combined fur-bearers occurred for references during the winter of 1995 compared to 1993 in both conifer and mixedwood forest types, but these differences were not significant ($p>0.05$, Scheffe's). The no buffer and 30 m buffer sites had similar densities in 1993 and 1995.

Discussion

In our study, partial harvesting in mixedwoods did not significantly alter forest structure; therefore, it is not surprising to find that fur-bearer abundance did not change either. Conversely, significant changes did occur to forest structure in clear-cut conifer forests. Here, basal area was reduced, and large diameter classes were removed leaving only pockets of polewood with snags almost completely removed.

Many studies have reported that chipmunks (Family: *Sciuridae*) and red squirrels are uncommon in recent clear-cuts, yet they become more common as forest stands mature (Brooks and Healey 1988, Kirkland 1977). The most important habitat requirement of squirrels are cone-bearing conifer trees which are used for both food and nesting sites (Fancy 1980). Other fur-bearers such as fisher, prefer mature coniferous forest and are generally absent in recently logged or burned-over areas due to a lack of cover, food and denning sites (de Vos 1952). Allen (1983) developed a habitat suitability index model for fisher that defined good habitat as dense mixed stands with greater than 50 percent crown closure, a multi-storied canopy, and in a late successional stage of forest development. Marten have also been shown to strongly prefer uncut mature forests because harvested areas provide fewer den sites and hunting opportunities (Thompson et al. 1989). Other studies (Soutiere 1979) have shown that partial harvesting provides adequate habitat for species like marten but that clear-cuts up to 15 years of age are poor marten habitat.

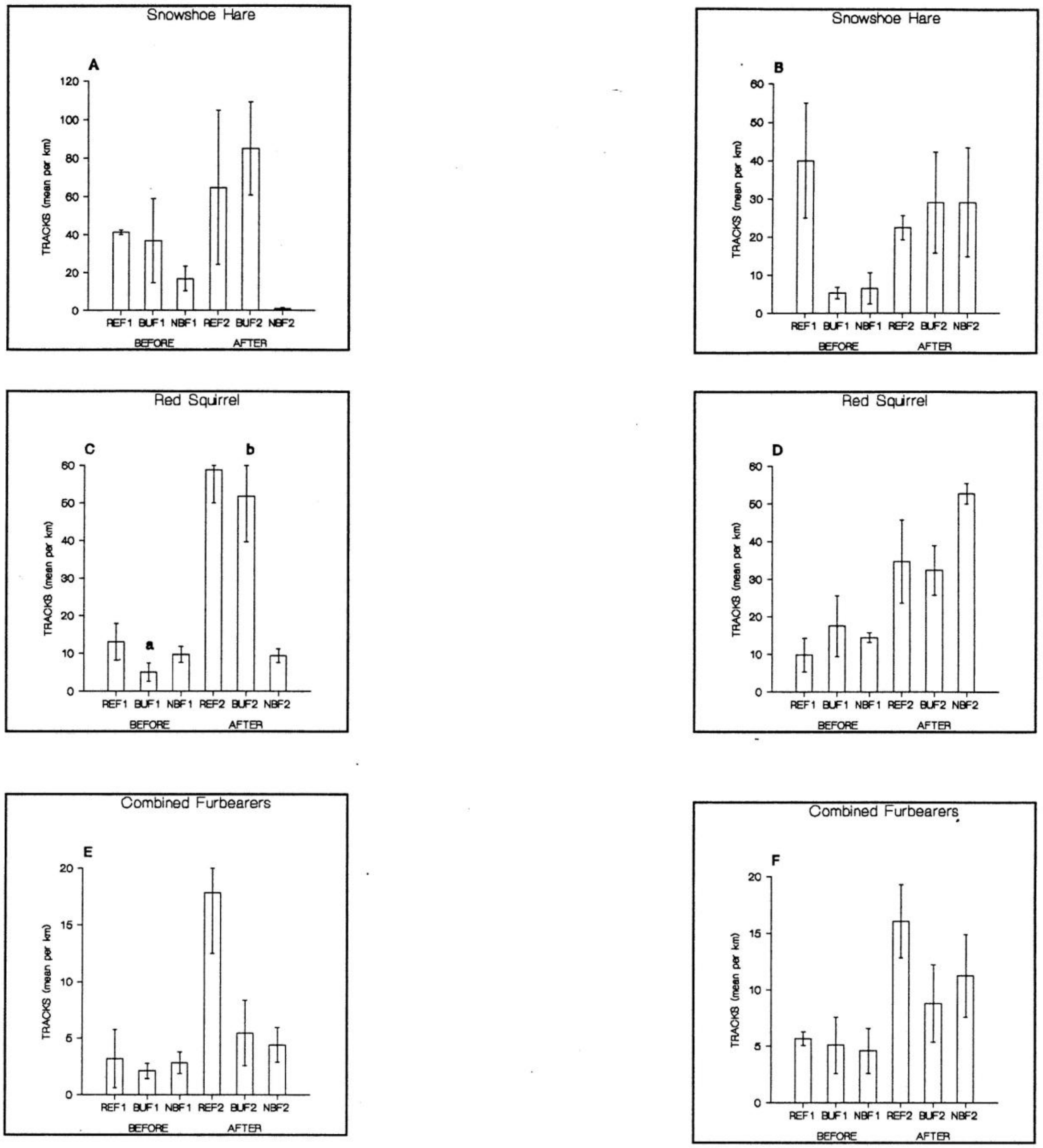

Figure 12. Snowshoe hare abundance in conifer (A) and mixedwood (B) forests, red squirrel abundance in conifer (C) and mixedwood (D) forests, and combined fur-bearers in conifer (E) and mixedwood (F) riparian forest before and after timber harvest. For A and C, significant interaction ($P<0.05$, 3-way ANOVA) was detected. For B and D, no significant interaction ($P>0.05$, 3-way ANOVA) was detected. For E and F, no significant interaction ($P>0.05$, 3-way ANCOVA) was detected. For A, B, C, D, E, and F no differences among treatments ($P>0.05$, Scheffe's) after harvest was detected. For C, 30 m buffers (a,b) were different between years ($P<0.10$, Scheffe's).

Large areas generally have more species than small areas and species richness depends on sample size (Wittaker 1970 *in* Lautenschlager 1995). In this study, fur-bearer species richness, density of snowshoe hare, red squirrel and total abundance did not interact with the size of wetland/edge habitats. The combined fur-bearer group did interact with wetland/edge size; therefore, sample size was used as a covariate (ANCOVA) to adjust for differences in size of wetland/edge habitats. This interaction implies that predatory species such as weasel, marten, fox, fisher, otter, wolf, mink and lynx are more affected by size of habitat than are hare and squirrel.

Early successional stages of forest growth were used by Thompson et al. (1989) to compare young clear-cut areas between one and 38 years old, relative to uncut forest. Hare and lynx were least abundant in logged stands less that 5 years old and most abundant in second-growth stands of 20 to 30 years of age. Snowshoe hare require low, dense woody cover for winter browse and protection from predators (Radvanyi 1987) and these conditions are not found in recently clear-cut areas. Fluctuations in lynx populations closely follow that of hare which is the main prey of lynx (Brand and Keith 1979). Thompson et al. (1989) observed that red squirrel and marten were most abundant in uncut forests, conversely red fox were at their lowest level in uncut forest.

Soutiere (1979) compared marten densities in partially cut and clear-cut forests in Maine and showed that selectively harvested mixedwood and hardwood stands (e.g. 50–80 percent basal area reduction) supported marten at levels similar to uncut mature forest but that clear-cut areas of up to 15 years old provided quite poor marten habitat. In that study, it was shown that the degree of forest structure remaining in selectively harvested stands was likely enough to provide adequate habitat for martens. Soutiere (1979) also made a recommendation for the establishment of riparian buffers in order to provide mature forest habitat and travel corridors for wildlife within clear-cuts.

Even though some aquatic fur-bearing species like mink and otter prefer riparian sites, they have been observed quite far from water to distances of more than 200 m (Thompson 1988). These movements are probably linked to searches for terrestrial species due to a lack of adequate food in preferred aquatic environments (Linscombe and Kinler 1982). A study in Quebec (Burgess 1978) located the majority of mink activity within 3 m of stream edges while in Michigan, Marshall (1936) reported all mink tracks within 30 m of water. In Ontario, Racey and Euler (1983) noted a decrease in mink activity with cottage lot development that removed riparian trees, shrubs and aquatic vegetation. In this study, 30 m buffers maintained hare and squirrel densities adjacent to wetlands at levels similar to the uncut forest yet other fur-bearers decreased in abundance in both buffer and no buffer wetlands after logging.

The winter of 1993 was exceptionally cold with the coldest February reported since 1979 (North Bay Weather Office 1993, pers. comm.). In mixedwoods, with so little changed in terms of forest structure due to selective logging, large increases in total fur-bearer abundance occurred in both treatment and reference wetlands in 1995 (refer to Figs. 11 and 12) compared to the winter of 1993. Weather greatly influences animal activity (Broom 1981) and some species restrict activity when temperatures are very low (Hatler 1991). The annual variation of fur-bearers fluctuates considerably between years (Pulliainen 1981). Hatler (1991) suggested that use of particular habitats may change as winter progresses with high wildlife movement occurring early (November-December) and late (March) in the winter season. Therefore, it is necessary to observe and record tracks from similar periods within the winter season when comparing activity between years.

Conclusions

Total fur-bearer abundance was greatly reduced in coniferous riparian forests adjacent wetlands without 30 m buffers after clear-cut logging, due to large decreases in snowshoe hare and red squirrel density. Changes to forest structure altered preferred habitats where buffers had not been established in terms of removal of dense understorey thickets used by snowshoe hare and cone-bearing tops required by red squirrels. Thirty meter buffers were effective in maintaining hare and squirrel densities at levels similar to uncut forests after timber harvesting. Fur-bearers in selectively harvested mixedwoods were not greatly affected by partial logging which is likely attributed to maintenance of forest structure. Species richness was not affected by logging in either clear-cut conifer or partially harvested mixedwood riparian forests. Other predatory fur-bearers which had been grouped together had a similar density after logging as compared to pre-cut levels. However, as these species rely heavily on hare and squirrel as a food source, predator density may be reduced in subsequent years within clear-cut wetland riparian areas without 30 m buffers.

WATER CHEMISTRY

Introduction

Streamside management zones (SMZ buffer strips) have been recommended and are often required by states and provinces throughout North America to mitigate impact of forest management practices on adjacent surface waters. Concern for retention of vegetation along water courses results from observations of loss of quality of water and biotic degradation resulting from the impact of forest cutting and road construction (Castelle et al. 1992).

Earlier studies stressed that the most commonly recognized value of buffer zones is the protection of water quality. The parameters commonly involved are sediments, nutrients and pesticides (Comerford et al. 1992). To date, most studies investigating the impact of timber harvesting have focused on the linkage of riparian strips or ecotones surrounding streams and lakes, mainly in the United States. While the use of buffer zones to protect wetlands and surface water bodies is proposed for Ontario, little quantitative information is available on the effectiveness of these strips. For policymakers to develop sound guidelines on the usefulness of SMZs in protecting water quality and wildlife, quantitative research is required to determine what criteria should be considered in the design and maintenance of riparian strips in different forest types (e.g., conifer and mixedwood forests) at the landscape level. One of the objectives of this study was to provide information on the suitability of vegetative buffer strips for maintenance of water quality and protection of aquatic wildlife.

Methods

Water samples for chemical analyses to determine the impact of timber harvesting were collected in inflows and outflows in spring and fall during 1993 (before harvest) and 1994 (after harvest). A total of eight bottles were filled with water at each station for 15 different chemical analyses. Care was taken not to disturb the sediment while collecting the water. In fens, during low water levels in fall, mat surface water was collected by pressing the sample bottle into wet depressions in the peat surface. Water samples were filtered with a 200 um-mesh Nitex net to remove coarse particulate matter, except for those samples that were analyzed for total suspended solids. Manganous sulphate followed by alkali-iodide-azide reagents were immediately added to water samples in the field to prevent changes in dissolved oxygen (DO). Samples were kept in coolers containing ice during transfer and final processing at the laboratory (up to 72 hours).

All chemical analyses for water were conducted at the Ontario Ministry of the Environment and Energy Research Centre, Dorset, Ontario following standard procedures (LaZerte 1984, Locke and Scott 1986, American Public Health Associaiton 1989). The pH values were measured in the laboratory with Ross electrodes standardized against appropriate buffers and a Fisher Accumet Model 750 meter. Conductivity readings from a Radiometer Type CDM2e meter were standardized to 25°C. Alkalinity was measured by electrometric titration (total inflection point) on a Metrohm E636 Titroprocessor. Total suspended solids (TSS) were determined by filtering a known volume of water through a pre-weighed Millipore (type AP40) glass fibre filter, oven dried, and the filtrate was weighed gravimetrically. The iodometric method with the azide modification was used to measure DO. Nutrients (total P, total Kieldal N, nitrate-N, ammonium-N) were measured by ion colorimetrically on a Auto-Analyzer IIC plus system. Sulphate was measured by ion chromatography to avoid problems associated with methyl thymol blue determinations in coloured waters. Calcium (Ca) was measured on an atomic absorption Spectophotometer (Varian 1275). Aluminum (Al), iron (Fe), and manganese (Mn) were measured by atomic absorption on a Perkin-Elmer 4000 Spectrophotometer equipped with a Varian 975 graphite furnace. Dissolved organic carbon (DOC) was determined on an Astro Modal 1850 Total Organic Carbon Analyzer.

Statistical Analysis

A paired t-test was used to test for differences in mean chemical concentration in inflows versus outflows within a season (spring or fall) and a two-sample t-test for differences between seasons (spring and fall) and forest types (mixedwood versus conifer). Since no difference for each wetland was detected between inflow and outflow chemical concentrations for all 15 parameters within a season (paired t-test, $p<0.003$, Bonferroni corrected), results for inflows and outflows were combined for subsequent statistical analyses.

Analyses of the water chemical variables were patterned after the optimal impact study design proposed by Green (1979). (For more detail see section 1.4.) Samples of water were collected in inflows and outflows during spring and fall before (1993) and after (1994) timber harvest. All variables, except pH, were log-transformed to normalize distributions before analyses. We used a two-way (areas-by-times) factorial ANOVA or MANOVA design to test the null hypothesis that no change in chemical composition occurred from timber harvesting with 30 m buffers or without buffers compared to reference areas against the hypothesis of no significant ($p<0.05$) areas-by-times interaction (see section on results).

Truck log-hauling study

A change occurred during the tree harvesting experiment that was not part of the original design. The intent was to remove all of the harvested trees during the winter cutting period. The timber industry chose to delay the transport of a large volume of trees cut around the study wetlands the previous winter until July of the second year. Since this disturbance by trucks could bias the chemical results of the regular sampling in August, an experiment was designed to quantify the impact of hauling truckloads of logs on water quality of a subset of the study wetlands in July and to compare these chemical concentrations to those determined for the scheduled fall sampling period (August, 1994). Ten wetlands, a subset of the 30 wetlands sampled before and after timber harvest, were chosen. Five wetlands were located upstream of the newly created logging road and designated reference sites; five wetlands were chosen downstream close to the road and considered treatment sites. Duplicate water samples were collected in both inflows and outflows during (July 11–13, 1994) and after (August 2–3, 1994) truck traffic. Again, the experimental design was patterned after the optimal impact study design proposed by Green (1979). The authors tested the null hypothesis that no change in chemical concentration occurred in treatment relative to reference wetlands due to logging-truck travel.

Results

Differences in season and forest type

Changes in chemical concentrations in all wetlands were closely associated with the seasonal differences in hydrology (Tables 16 and 17). Ammonium-N, total N, total phosphorus (TP), pH, alkalinity, conductivity, Ca, DOC, Fe, Mn, Al and TSP were significantly higher in concentration in fall (paired t-test, $p<0.003$, $N = 75$, Bonferroni corrected) relative to spring. In contrast, nitrate, sulphate, and DO were significantly greater ($p<0.003$) in spring compared to the fall season.

Differences in chemical concentration in wetlands were found for the different forest types (Tables 16 and 17). Total-N, nitrate-N, total P, pH, alkalinity, Ca, DOC, Fe, and TSS were significantly higher (paired t-test, ($p<0.003$, $n= 75$, Bonferroni corrected) in wetlands located in mixedwood forests compared to those parameters in wetlands within conifer forests. In contrast, DO, sulphate, Mn and Al were significantly higher ($p<0.003$) in conifer forested wetlands than in mixedwoods. No difference ($p>0.003$) in concentration between wetlands within forest types was detected for ammonium-N and conductivity.

Table 16. Chemical concentrations (mean ± S.E., inflows and outflows combined) for wetlands in conifer forests during spring and fall before (1993) and after (1994) timber harvest; n = 15 wetlands for each treatment (reference, 30 m buffer, no buffer) before harvest; n = 10 wetlands for each treatment after timber harvest. TIP = total inflection point; µS/cm = microSiemans per centimeter; mg/L = milligrams per litre; µ/L = micrograms per litre. Asterisks indicate significant differences in chemical concentrations between ($P<0.05$, 2-way (M)ANOVA) and within ($P<0.05$, Scheffe's test) treatments relative to reference areas

	BEFORE HARVEST						AFTER HARVEST					
CHEMICAL PARAMETERS	Reference		30m Buffer		No Buffer		Reference		30m Buffer		No Buffer	
SPRING CONIFER	Mean	S.E.	Mean	S.E.	Mean	S.E.	Mean	S.E.	Mean	S.E.	Mean	S.E.
Total Kjeldahl nitrogen (mg/L)	0.33	0.03	0.38	0.03	0.3	0.03	0.28	0.014	0.31	0.2	0.32	0.03
Ammonium nitrogen (µg/L)	6.9	0.9	15.7	1.75	10.3	1.3	19.6	3.6	14.5	1.4	15.7	2.5
Nitrate nitrogen (µg/L)	19.2	9.8	15.8	3.8	11.4	2.4	14.2	7.9	5.2	0.94	5.1	0.5
Total Phosphorus (µg/L)	7.4	0.8	7.8	0.82	6.57	0.73	9.0	1.6	9.5	1.28	8.4	1.1
Dissolved Oxygen (mg/L)	8.35	0.33	7.43	0.57	7.1	0.46	12.9	0.71	12.3	0.35	13.7	0.95
pH	5.89	0.32	5.04	0.22	5.17	0.2	5.42	0.26	4.69 *	0.17	4.83	0.16
Alkalinity (TIP, mg/L)	14.2	3.9	6.54	0.88	6.92	0.89	8.1	1.4	5.18	0.5	5.6	0.42
Conductivity (µS/cm)	43.9	7.2	31.1	1.65	34.4	1.67	35.9	2.5	33.1	1.3	36.7	2.4
Calcium (mg/L)	4.2	0.99	2.1	0.27	2.35	0.3	3.1	0.44	2.1	0.14	2.65	0.39
Sulphate (mg/L)	7.8	0.58	6.87	0.5	8.6	0.63	8.5	0.34	8.1	0.37	9.9	0.88
Dissolved Organic Carbon (mg/L)	7.47	0.84	10.1	1.2	7.05	0.66	7.2	0.52	11.7 **	0.75	10.7 *	1.4
Iron (µg/L)	248.5	39.4	429.8	38	260.5	37.2	195.4	36.6	279.9	17.6	251	46.0
Manganese (µg/L)	35.9	7.33	67.6	10.9	70.2	7.8	36.0	8.2	84.9 **	10.6	111 **	14.5
Total Aluminum (ug/L)	154.6	26.23	267.9	46.9	294.3	62.0	175.2	27.8	339.2**	64.0	354 **	61.7
Total Suspended Particulates (mg/L)	0.96	0.15	1.38	0.33	0.73	0.08	1.14	0.21	0.96	0.2	1.04	0.14
FALL CONIFER												
Total Kjeldahl nitrogen (mg/L)	0.74	0.16	0.7	0.07	0.6	0.07	0.58	0.12	0.59	0.03	0.59	0.05
Ammonium nitrogen (µg/L)	36.5	10.8	104.4	46.0	98.5	45.0	20.2	3.5	22.7	4.5	40.6	15.7
Nitrate nitrogen (µg/L)	9.1	1.5	17.2	5.4	9.8	2.1	19.4	9.3	12.4	0.89	10.3	1.1
Total Phosphorus (µg/L)	22.7	6.3	21.7	2.36	19.0	1.7	19.3	6.5	15.2	1.38	21.2	3.6
Dissolved Oxygen (mg/L)	6.96	0.93	6.6	0.7	6.73	0.67	8.0	0.62	7.0	0.47	5.4 *	1.0
pH	6.1	0.39	5.6	0.31	5.5	0.31	5.9	0.3	4.9 *	0.18	5.2	0.17
Alkalinity (TIP, mg/L)	23.6	9.1	11.7	3.9	11.4	3.8	18.7	6.9	5.9 *	0.95	8.1	1.1
Conductivity (µS/cm)	66.6	17	43.3	7.3	40.8	8.0	47.1	13.6	30.0	1.7	32.0	3.0
Calcium (mg/L)	6.5	2.1	3.3	0.89	3.1	0.95	5.65	1.9	2.7	0.47	3.4	0.45
Sulphate (mg/L)	6.38	1.16	6.9	0.63	5.9	0.7	2.3	0.81	3.5	0.63	1.8	0.74
Dissolved Organic Carbon (mg/L)	12.1	2.1	11.0	1.0	10.1	1.4	13.6	2.7	19.0	1.2	17.0	2.9
Iron (µg/L)	1544	641	917.3	174	914	191		731.3	93.6	1139	187	762
Manganese (ug/L)	96.3	35.0	62.3	4.9	61.5	5.3	47.0	6.8	65.2	9.0	116 **	26.0
Total Aluminum (µg/L)	232	60.0	202.6	29.0.	223	43.0	243	34.7	449 *	51.0	381 *	43.0
Total Suspended Particulates (mg/L)	3.57	1.0	3.0	0.61	4.2	0.9	4.1	2.0	4.6	1.66	4.1	1.2

* $P < 0.05$

** $P < 0.01$

Table 17. Chemical concentrations (mean ± S.E., inflows and outflows combined) for wetlands in mixedwood forests during spring and fall before (1993) and after (1994) timber harvest; n = 15 wetlands for each treatment (reference, 30 m buffer, no buffer) before harvest; n = 10 wetlands for each treatment after timber harvest. TIP = total inflection point; μS/cm = microSiemans per centimeter; mg/L = milligrams per litre; μ/L = micrograms per litre. No significant differences in chemical concentrations between treatments (P>0.05, 2-way (M)ANOVA) were detected.

	BEFORE HARVEST						AFTER HARVEST					
CHEMICAL PARAMETERS	Reference		30m Buffer		No Buffer		Reference		30m Buffer		No Buffer	
	Mean	S.E.	Mean	S.E.	Mean	S.E.	Mean	S.E.	Mean	S.E.	Mean	S.E.
SPRING MIXEDWOOD												
Total Kjekdahl nitrogen (mg/L)	0.48	0.04	0.51	0.02	0.5	0.04	0.48	0.02	0.45	0.025	0.5	0.03
Ammonium nitrogen (μg/L)	8.05	1.47	14.1	4.0	6.7	1.3	26.5	13.4	22.4	5.7	19.0	3.4
Nitrate nitrogen (μg/L)	14.1	3.5	25.9	5.5	17.8	4.3	11.9	3.3	17.8	3.2	19.0	5.6
Total Phosphorus (μg/L)	9.7	1.3	13.5	1.6	15.7	2.4	11.4	2.3	16.2	2.07	15.2	1.4
Dissolved Oxygen (mg/L)	6.7	0.43	7.1	0.35	6.15	0.4	9.86	0.88	10.1	0.88	9.46	0.63
pH	5.63	0.21	5.57	0.16	5.6	0.22	5.05	0.31	5.83	0.14	5.6	0.28
Alkalinity (TIP, mg/L)	15.0	3.8	11.0	1.7	12.7	2.3	10.2	3.7	11.8	1.6	9.8	1.7
Conductivity (μS/cm)	43.8	6.8	38.9	3.7	40.0	3.5	40.6	6.8	44.7	6.2	36.2	3.0
Calcium (mg/L)	4.5	1.0	3.5	0.43	3.9	0.62	3.05	0.67	4.1	0.58	3.5	0.6
Sulphate (mg/L)	5.7	0.49	6.23	0.32	5.2	0.44	6.3	0.41	6.37	0.3	5.9	0.6
Dissolved Organic Carbon (mg/L)	12.9	1.2	12.8	0.77	14.7	1.1	14.6	1.5	12.1	1.2	14.3	1.35
Iron (μg/L)	345.2	55.7	421	68.0	365	58.8	396.3	132	304.5	64.8	240.4	35.6
Manganese (μg/L)	29.6	5.3	49.7	10.0	28.0	3.4	52.8	22.0	36.0	10.0	20.5	4.1
Total Aluminum (μg/L)	163.4	21.9	203	23.5	155.8	19.0	157.7	23.3	154.7	23.5	172.8	33.0
Total Suspended Particulates (mg/L)	1.4	0.58	1.4	0.34	1.14	0.23	1.02	0.29	2.36	0.24	3.04	0.72
FALL MIXEDWOOD												
Total Kjekdahl nitrogen (mg/L)	2.4	0.88	1.7	0.64	1.3	0.31	1.77	0.08	0.81	0.09	0.77	0.074
Ammonium nitrogen (μg/L)	135	39.9	171	52.0	45.9	10	22.1	3.7	78.5	23.0	38.1	13.5
Nitrate nitrogen (μg/L)	10.6	1.1	11.0	1.3	10.9	2.36	13.7	1.6	9.0	1.0	12.3	2.4
Total Phosphorus (μg/L)	40.4	8.8	37.7	4.7	26.4	2.9	12.3	1.4	29.0	5.8	29.7	3.8
Dissolved Oxygen (mg/L)	3.5	0.55	4.14	0.78	5.6	0.82	6.5	1.3	5.9	0.46	5.86	0.89
pH	5.92	0.19	6.06	0.15	5.9	0.23	5.5	0.34	5.93	0.13	5.6	0.28
Alkalinity (TIP, mg/L)	29.6	6.6	25.6	4.6	24.0	5.6	13.3	5.9	18.1	3.2	14.9	2.9
Conductivity (μS/cm)	72.2	11.7	63.0	9.4	63.0	9.2	42.2	8.8	44.3	8.0	36.9	4.8
Calcium (mg/L)	8.4	1.7	6.8	1.3	7.08	1.3	5.02	1.5	5.7	0.93	5.15	0.97
Sulphate (mg/L)	4.64	2.0	3.8	0.7	5.16	1.49	0.56	0.2	1.75	0.22	1.38	0.33
Dissolved Organic Carbon (mg/L)	20.9	2.3	18.4	2.0	17.7	1.7	28.5	3.5	19.9	2.0	21.4	2.6
Iron (μg/L)	1996	516	2624	637	1244	381	1144	231	1662.3	315	1121	309
Manganese (μg/L)	327.3	114	370.5	133	148.5	55.0	132.8	79.4	119	17.9	159	58.4
Total Aluminum (μg/L)	207.6	28.5	276	48.0	173	23.7	290.3	47.0	261	42.5	282	62.0
Total Suspended Particulates (mg/L)	6.68	1.3	7.6	1.36	6.1	1.1	2.15	0.49	4.5	0.7	4.98	0.51

Harvesting impact

In the selectively cut mixedwood forests, no significant multivariate interaction occurred in spring (Wilk's Lambda= 0.67; approximate F= 0.89; df= 34,136; $p>0.05$) or fall (Wilk's Lambda= 0.62; approx. F= 0.97; df= 34, 120; $p>0.05$), indicating that conditions for water chemistry before the treatments (reference, 30 m buffer, no buffer) were similar to conditions after timber harvest. Similarly, no significant univariate interactions ($p>0.05$, 2-way ANOVA) were evident for each of the 15 chemical parameters (Table 16) in spring and/or fall. However, increases in concentration of DOC and TSS were evident relative to reference areas after harvest, but not significant because of large variability in these concentrations.

Many chemical parameters changed in coniferous wetlands after timber harvesting (Table 16). Significant multivariate interaction was evident during both spring (Wilk's Lambda = 0.34; approx. F= 1.6; df= 34,76; $p<0.04$) and fall (W.L.= 0.32; approx. F= 1.67; df= 34,74; $p<0.03$). However differences ($p>0.05$, one-way ANOVA, Table 16) in chemical concentration of the 15 parameters were not found before (time=1) harvest for both spring and fall. Significant differences ($p<0.05$, one-way ANOVA's) were found after (Time=2) timber harvest between the three treatments for pH, DOC, Mn, Al in spring (Fig. 13) and pH, alkalinity, Mn, Al (Fig. 14) and dissolved oxygen (Fig. 15) in fall.

Truck traffic impact

A significant interaction ($p<0.05$, 2-way MANOVA, n= 20) occurred for total-N, total phosphorus, and suspended solid concentrations in the wetlands because of logging truck traffic in July (Fig. 16). The concentrations were higher downstream (DNS-1) compared to upstream (UPS-1) during (time=1) and after truck traffic (time=2, DNS-2 versus UPS-2, $p<0.05$, one-way ANOVA's). Significant decreases in concentration were evident downstream (DNS-2) after compared to downstream during (DNS-1) truck hauling ($p<0.05$, Scheffe's) for all three chemical parameters. Thus, truck traffic did increase chemical concentration of nutrients and suspended particles in wetlands close to the road relative to those wetlands upstream and farther away from the road. Also, concentrations were lower in August (after hauling) and did not show any significant change during the regular sampling period in fall (Tables 16 and 17).

Effectiveness of riparian buffers

Mitigative effects of the 30 m buffer were detected for Mn and DO (Scheffe's pairwise comparison, $p<0.05$), but not for pH, Al, DOC and alkalinity ($p>0.05$). DO was significantly reduced in the clear-cut without a buffer (Fig. 15, Scheffe's, $p<0.05$)) relative to the 30 m buffer. In contrast, Mn was greater in concentration in the no buffer treatment relative to the 30 m buffer (Figs. 13 and 14, Scheffe's, $p<0.05$).

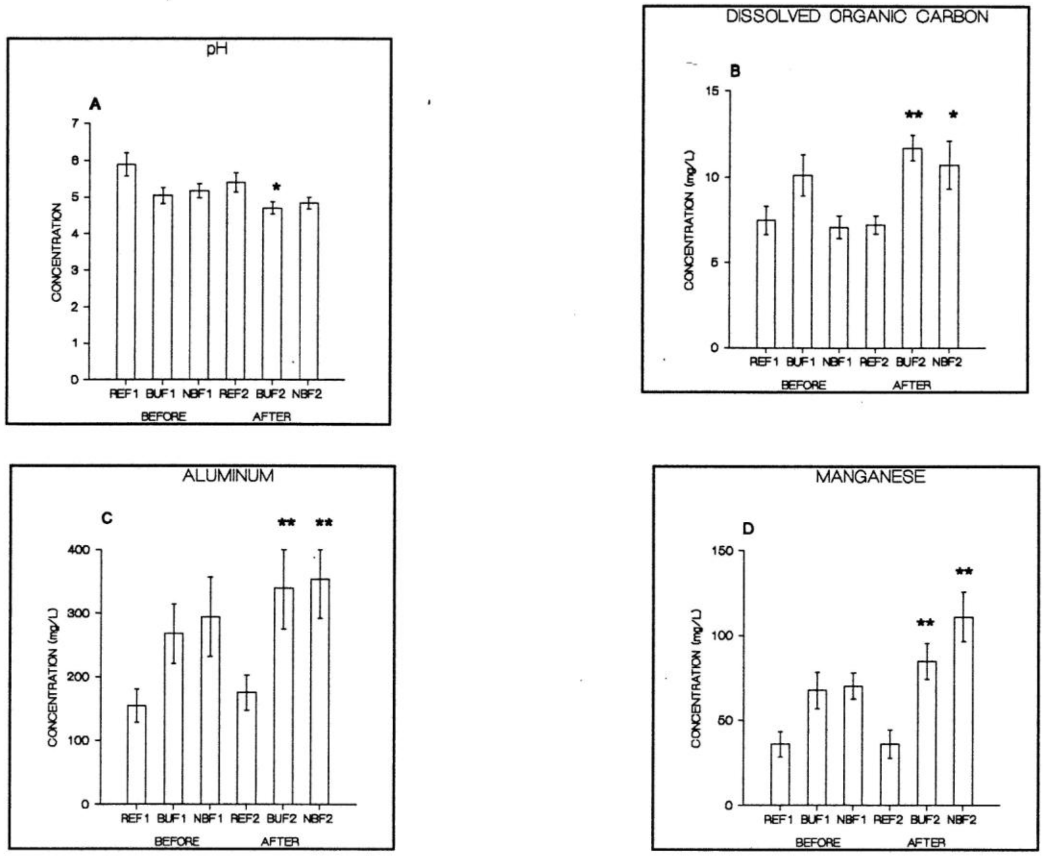

Figure 13. Bars represent concentrations (mean ± SE, inflows and outflows combined) for (A) hydrogen ion (pH), (B) dissolved organic carbon, (C) aluminum and (D) manganese in wetlands in conifer forests in spring. REF= reference; BUF= 30 m buffer; NBF= no buffer. Before = Time 1 (1993) and AFTER = Time 2 (1994) timber harvest. n= 15 wetlands for each treatment before harvest; n= 10 wetlands for each treatment after harvest. Asterisks indicate significant differences in chemical concentration between (2-way (M)ANOVA) and within (Scheffe's test) treatments.
**= p<0.05; **= p<0.01.*

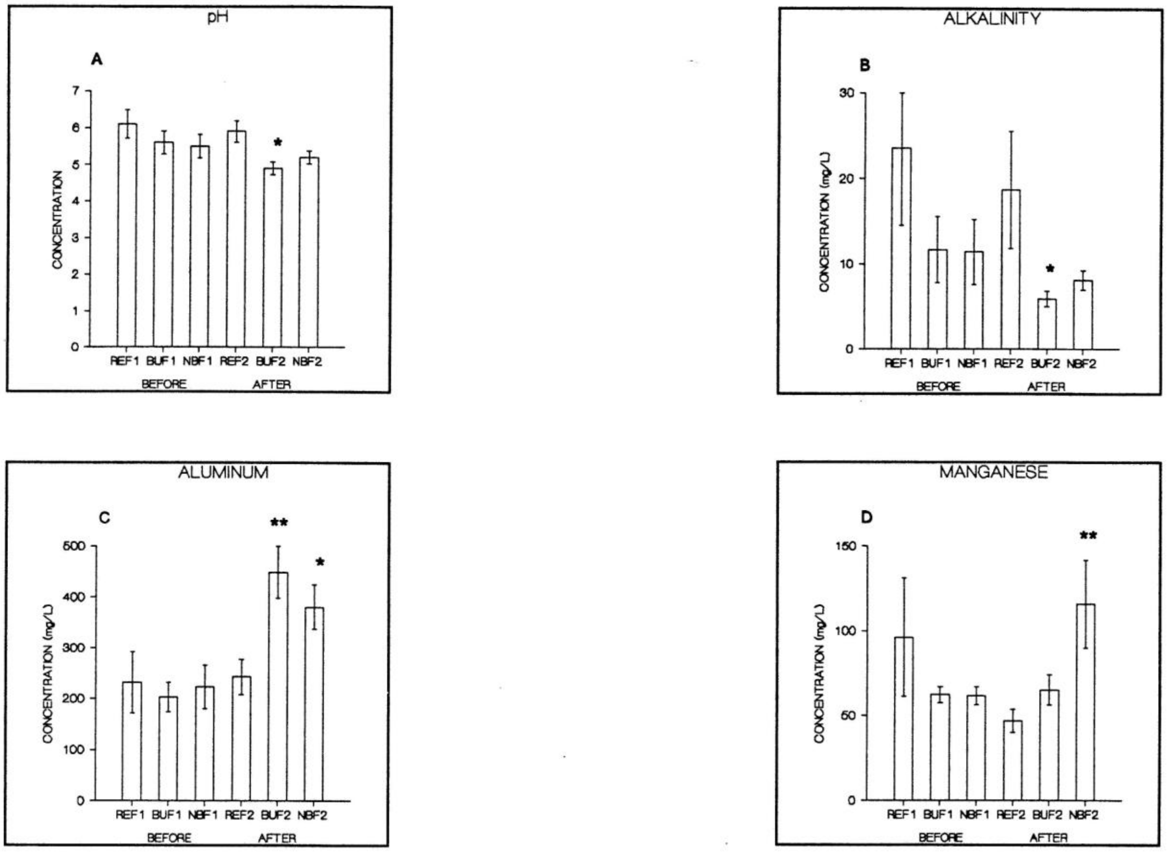

Figure 14. Bars represent concentrations (mean ± SE, inflows and outflows combined) of (A) hydrogen ion (pH), (B) alkalinity, (C) aluminum and (D) manganese in wetlands in conifer forests in fall. See Figure 13 for further details.

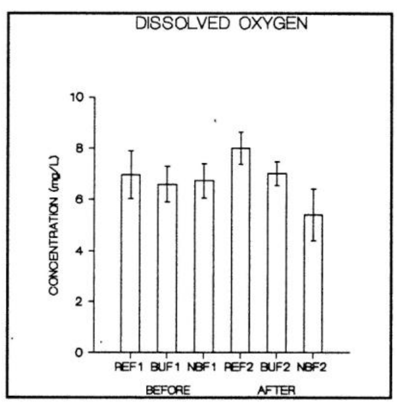

Figure 15. Bars represent concentrations (mean ± SE, inflows and outflows combined) for dissolved oxygen in wetlands in conifer forests in fall. See Figure 13 for further details.

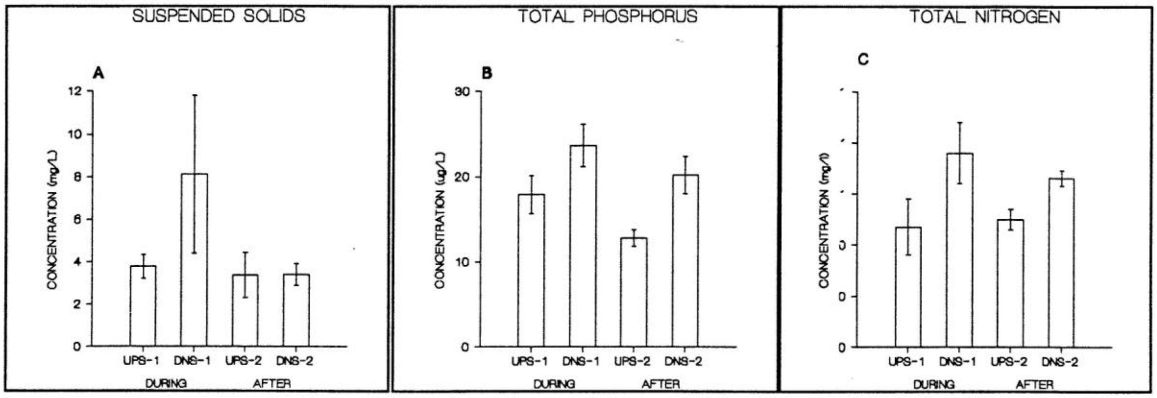

Figure 16. Bars represent concentrations (mean ± SE) in inflows for wetlands upstream (UPS, Time 1) and downstream (DNS, Time 2) of road used for hauling of logs in July, 1994. DURING= truck hauling logs; AFTER= 2 weeks after truck hauling.

Discussion

Hydrology and season

The effects of timber harvesting on changes in chemical concentrations in the study wetlands were a function of the method of harvest (selective vs. clear-cut), forest type (mixedwoods vs. conifer), season (spring vs. fall) and proximity of disturbance to the forest/wetland interface (30 m buffer vs. no buffer). Many of the temporal and spatial gradients in water quality resulted from changes in hydrology alone, such as mixing of surface runoff (i.e., snowmelt and/or rainstorms) with deeper soil water (Johnson et al. 1969, Hall et al. 1980). The mixing of a larger amount of dilute snowmelt water with groundwater in spring resulted in lower chemical concentrations in the study wetlands. In contrast, greater ground water input compared to surface water resulted in higher chemical concentrations in fall. Similar findings have been found for other streams (Hall and Likens 1984; Hall 1990) and wetlands (Devito and Dillon 1993) on the Canadian Shield.

The three dissolved compounds that were significantly greater in spring relative to fall for all the study wetlands were nitrate, sulphate and DO. The former two are the result of dispersal and deposition of mineral acids (H_2SO_4 and HNO_3) downwind from industrial sources (e.g., Sudbury) and the subsequent accumulation of these acids in snowpacks during winter. With the onset of snowmelt, Johannessen and Henricksen (1978) reported that 50–80 percent of the acidified materials (H_2SO_4 and HNO_3) stored in the accumulated snowpack were released within the first 30 percent of the snowmelt. Schofield (1977) and Jeffries et al. (1979) reported similar results in New York and Ontario, respectively. Likewise, DO would be greater in snowpack in spring during high turbulent flow resulting from snowmelt compared to low flows in fall when increased temperature and greater organic matter decomposition would occur.

Timber harvest

Superimposed on the natural fluctuations in wetland hydrology and chemistry were chemical changes spring and fall that resulted from timber harvesting. Significant reductions in pH and increases in concentrations of Al and Mn were detected in clear-cut conifer forested wetlands during both spring and fall. DOC increased in spring and alkalinity and DO significantly decreased in fall due to timber harvesting. The results in this study support the general statement by Krause (1982a) that clear-cutting of coniferous forests with acid soils and granite geology will cause increases in dissolved organic substances and lower surface water pH. Others have also reported increases in colour (Hetherington 1976, Plamondon et al. 1982) and organic particulates (Plamondon et al. 1982) after clear-cut logging.

The decrease in alkalinity and pH in wetlands in the conifer forest may have resulted from increases in weak acids (e.g., organic acids, Al, Fe and Mn), strong acids (H_2S0_4 and $HN0_3$) and/or dilution of base cations (i.e., Ca) from mixing of surface runoff (i.e., snowmelt and/or rainstorms) with deeper soil water (Johnson et al. 1969). Mean $S0_4^{-2}$, $N0_3^{-1}$, Ca^{-2} concentrations were not different after timber harvest relative to before treatment. Thus, pH depressions were attributed to weak acids. Pelers et al. (1976) concluded that the introduction of large amounts of organic debris can indirectly influence pH by increasing the concentration of organic acids, increasing oxygen demand and increasing $C0_2$ inputs in surface waters due to bacterial respiration.

Cutting of mixedwood forests and the impact on water quality were summarized by Krause (1982a). In general, clear-cuts of mixedwoods on podsol soils underlain by crystalline bedrock caused increases in some combination in nutrients (Nitrate-N, total N, total Phosphorus), base cations (Ca, Mg, Na, and K) and reductions in pH (Martin et al. 1986, Krause 1982a, 1982b, Likens et al. 1970). Similarly, increases in particulate export were reported as a result of mixedwood clear-cutting (Bormann et al. 1974, Hornbeck 1960). The result of no significant changes in chemical concentrations in the wetlands in the present study due to harvesting compared to those cited above is likely the result of differences in the harvesting method. Chemical increases occurred in surface waters in the studies cited above because the catchments were clear-cut. However, in this study, only a selective removal of high quality timber was conducted. Partial cutting of a forest on the west coast of Canada did not elevate dissolved ion concentrations to impact surface waters (Scrivener 1982). Singh et al. (1974) reported that partial harvesting produced no difference in cation concentrations between treated and untreated catchments. In New England, differences in surface water chemistry between recently clear-cut and nearby uncut watersheds were generally small in a wide variety of soil and forest types (Martin et al 1984). The largest differences that could be attributed to harvesting occurred in entirely clear-cut watersheds. The authors concluded that clear-cutting only portions of the entire watersheds in the form of patch and strip cuts reduced the magnitude of changes in surface water chemistry.

Truck hauling impact

Although no significant increases in suspended sediments and nutrients (N and P compounds) were detected as result of timber harvesting, the parameters were significantly elevated because of logging truck traffic. Many studies by researchers have concluded that road building and road maintenance are important sources of suspended particulates to surface waters. These particulates can be eroded from sloping of soils associated with road construction and drainage (Beschta 1978; Duncan et al. 1987, Megahan and Bohn 1989) as well as from eroded road surfaces (Reid and Dunne 1984). These inputs of particulates would likely continue with heavy truck traffic. Thus, regulations should restrict the construction of roads close to surface waters because roads were the major cause of sedimentation leading to poor water quality in forested areas (Trimble and Sartz 1957). Even if traffic is restricted on these roads, storm events can continue to cause erosion and mass movements (Everest et al. 1987) unless roads are removed and vegetation allowed to regrow.

Role of riparian buffers

For all 15 chemical parameters measured, the significant difference associated with 30 m buffers compared to no buffer strips within the clear-cut conifer forest was detected for only Mn in spring and fall and for DO in fall. Mn was lower in the 30 m buffered wetlands compared to wetlands with no buffer. In contrast, DO was lower in the no buffer relative to sites with riparian buffers and reference areas. A few studies present evidence that riparian zones reduce sediment and P loads in adjacent streams (McColl 1978, Schlosser and Karr 1981). Forest buffer strips were effective in reducing the concentration of total soluble N, P, K, and N at a distance of 3.8 m (Doyle et al. 1977). Martin et al. (1984) concluded from chemical surveys of surface waters in New England in which the surrounding forests were harvested at different intensities that leaving buffer strips reduced the changes in water quality. Plamondon et al. (1982) also showed significant reductions in DO in clear-cut boreal forest catchments without buffers compared to no change in clear-cut catchments with 30 m buffers left intact.

The decrease in DO in fall in wetlands without buffers may have resulted from large inputs of fine particulates, bark and other organic debris entering the wetlands during snowmelt before the spring sampling period or during rain storms in the summer. During the fall sampling period when temperatures were high and rate of reaeration low because of impoundments created by beavers (*Castor canadensis*), the input of fresh organic debris could have caused an increase in biological oxygen demand (BOD) and thus DO was significantly decreased. Hall and Lantz (1969) and Wringler and Hall (1975) have reported depleted DO levels from heavy inputs of fresh organic material, combined with sedimentation and increased temperature in streams after harvesting. The reason for the decrease in DO in a stream in Quebec without a buffer was that slash in the stream after logging impeded the flow of water by impounding the stream (Plamondon et al. 1982).

Many studies have reported that buffer strips protect surface water quality and biotic communities from increased siltation resulting from timber harvesting (Castelle et al. 1992). However, no significant elevation in TSS was evident in the study wetlands south of Temagami due to the different timber harvesting treatments. Plamondon et al. (1982) reported an increase in suspended particles in clear-cut boreal forested streams without buffers but no change in surface waters with buffers. However, upon closer inspection of the results, the increase without buffers resulted predominantly from logging skid trails across the streams. No such disturbances were reported for the study streams with buffers. Thus, it is difficult to conclude that buffers protect the surface waters from siltation in the above study because the experimental design is flawed. Although not significant, TSS did increase in the treatment areas in the mixedwood forests (Table 13), but less so in the buffered sites, relative to the reference area. MacDonald et al (1991) reported that it is difficult to detect significant increases in suspended solids in surface waters after timber harvest even in well planned experimental studies because of the high natural variability in sediment suspension.

Conclusions

Marshes in mixedwood forests have higher alkalinities and thus greater buffer capacities for neutralization of acidic substances than marshes in conifer forests. Selective cutting did not significantly change surface water quality of wetlands. However, clear-cut logging significantly decreased the buffer capacity (alkalinity), then reduced pH and elevated metal concentrations. The mechanism for these chemical changes was, in part, from increased input of dissolved organics leached from the forest floor (ground water and surface water). Riparian buffers (30 m in width) reduced changes in some chemical parameters such as DO and Mn concentrations. Ground water inputs may override the benefit of buffers for some dissolved chemical substances (hydrogen ion, Al, and DOC concentrations) but may be beneficial for protection of surface waters from increased transport of soil derived particulates. The surface water chemical changes were related to the extent of tree extraction within the watershed (e.g., total, partial or selective cutting).

FISH

Introduction

Riparian zones are intimately associated with surface water fish and other aquatic biota (Junk et al. 1989). These habitats are ranked among the most productive and valuable ecosystems on earth (Hunt 1988). Aquatic biota depend on riparian organic input such as leaf fall. Leaf input is the food-base for production of fish food organisms. Fish also depend on the function of the aquatic-terrestrial transition zone (Junk et al 1989). Removal of vegetation around wetlands could decrease the invertebrate food-base for fish which could ultimately decrease fish communities.

Fishes are the highest order consumers in wetlands and are directly dependent upon fish-food organisms and indirectly dependent on physiochemical constraints for acceptable habitat and conditions for metabolism. With the removal of riparian vegetation, environmental conditions in the aquatic ecosystems often change, particularly for temperature, DO and sedimentation (Hall and Lantz 1969, Moring 1982). Thus, presence of an intact riparian zone is essential for mitigating wide variations in environmental conditions (Erman et al. 1977).

The objective of this study was to evaluate qualitatively and quantitatively the role of timber management in the form of riparian buffers versus no buffers in influencing aquatic biota, particularly fish, in wetland ecosystems. Details of the experimental design for forest harvesting are described in section on Study Design.

Methods

Aquatic communities were only collected from marshes because they contained an average of 10–30 percent open water, while peatland fens lacked enough surface water in July. The sampling procedures followed those outlined in the Food Chain Monitoring Program proposed by the CWS (McNicol et al. 1987). Aquatic biota were collected using cylindrical minnow traps (Gee's type dimensions 0.3 m long by 0.3 m diameter) constructed of 6 mm wire mesh, with openings 40 mm in diameter at each end. Five traps were placed (a minimum of 200 m apart) in the littoral zones of marshes (water depth <2m) in large or small pools or in inflow and outflow channels. Fourteen wetlands in the mixedwood forest were studied (7 reference, 4 with 30 m buffers and 3 with no buffers) before (1993) and after (1994) selective timber harvest. Only three wetlands (one reference, one-30 m buffer and one-no buffer) were studied after (1994) clear-cut harvesting of the conifer forests. Each trap was baited with 250 ml of Purina Puppy Chow® (9percent fat content) and 2-1 cm slices of Burns® wieners. The traps were tethered by placing a galvanized steel spike through the trap and into the substrate. Baited traps were retrieved within 24 hours. Collected specimens from each trap were sorted and placed in Zip-lock® bags according to major groups (fish, insects, leeches, amphibians and crayfish). Samples were returned to the laboratory in coolers and frozen each day. Subsequently, the frozen specimens were transported in coolers containing dry ice to the Royal Ontario Museum (ROM) in Toronto for species identification. All identified specimens were donated to the ROM because of the limited collections from the Temagami area wetlands.

Statistical Analysis

The optimal impact study design proposed by Green (1979) was used to analyze the fish data before and after timber harvest. Details of this procedure are outlined in section 1.4. Statistical analyses were done with fish assemblages captured in the mixedwood forested wetlands only. Numbers were log (x+1) transformed to reduce observed heterogeneity of variances. The number of fish samples collected were too small for statistical analyses in the conifer forested wetlands. The latter results were presented primarily for presence/absence comparisons of species from the two forested types.

Results

A total of 6,058 specimens were collected and identified to species in 1993 and 7,192 specimens in 1994 (Appendix 1). Two species of crayfish, 5 species of leeches, and 15 species of aquatic insects were collected in the minnow traps.

In the mixedwood forests, 11 species of fish were captured in 140 minnow traps retrieved from 14 marshes before (1993) and after (1994) the selective timber harvest treatment (Table 18). These species included 8 cyprinid minnows (northern redbelly dace [*Phoxinus eos*], finescale dace [*P. neogaeus*], pearl dace [*Semotilus margarita*], fathead minnow [*Pimephales promelas*], common shiner [*Notropis cornutus*], creek chub [*Semotilus atromaculatus*] and blacknose shiner [*Notropis heterolepsis*]), brook stickleback (*Culaea inconstans*), white sucker (*Catostomus commersoni*), mottled sculpin (*Cottus bairdi*). Dace hybrid crosses (finescale x northern redbelly x pearl daces) were abundant and kept separate from populations of their parent species. Five minnow species (including dace hybrids) represented approximately 90 percent of the total numbers of fish collected. The sculpin and the blacknose shiner were only found in one wetland each in the no buffer treatment.

No significant multivariate or univariate site x time interaction ($p>0.05$) was detected in fish abundance at reference and treatment sites before and after selective timber harvest of mixedwoods (Table 18), indicating that fish abundance after forest disturbance was similar to before harvesting. A shift in abundance between the two years was primarily due to the redbelly dace which represented 67 percent of the total number for the 11 species (Table 18). In addition, no significant interaction ($p>0.05$,two-way ANOVA) was found for fish species richness before or after harvesting. The results for fish abundance are consistent with those found for chemistry in that no differences due to selective harvesting were apparent (Table 18).

In marshes in the conifer forests, seven species of fish were captured in the 15 minnow traps after timber harvest only. The genus *Phoxinus* (redbelly dace and dace hybrids) represented 87.2 percent and pearl dace another 11.5 percent of the total numbers captured in the three treatments combined (Table 19). The species that represented less than 6 percent were finescale dace, central mudminnow, fathead minnow, common shiner, and brook stickleback. Only two species (pearl dace and central mudminnow [*Umbra limi*]) were captured in the wetland with the 30 m buffer strip.

Richness of species was also lower in the conifer forested wetlands after harvest compared to those species collected in wetlands in the mixedwood forests. Dace hybrids were present in greater numbers in the conifer (42.7 percent) relative to the mixedwood sites (2.2 percent). The reduced number of taxa, particularly in the 30 m buffer treatment, is correlated with changes in spring chemistry, particularly pH and Al, (Tables 16 and 19) as a result of clear-cut timber harvesting.

Discussion

Although no significant difference was found, the shift in abundance of fish assemblages in the study wetlands was due to a combination of interactions among differential timber harvesting (selective vs. clear-cut; buffer and no buffer treatments), water quality parameters (specifically pH, Al, DOC, dissolved oxygen) and wetland type (stream and/or pond).

Table 18. Mean (± SE) number, species richness and total percent of each species collected (in descending order of abundance) for 5 minnow traps combined per wetland in mixedwood forests during July before (1993) and after (1994) timber harvest. Sample size (N)=7 for reference, 4 for 30 m buffer and 3 for no buffer. See Appendix 1 for scientific names. Estimated pH thresholds are for disappearence of fish species in surface waters based on laboratory bioassays and field collections: 1 = Matuszek et al. (1990); 2 = Gunn and Belzile (1994); 3 = McCormick et al. (1989); 4 = Holtz and Hutchinson (1989); 5 = Rahel and Magnuson (1983).

	MIXEDWOOD FOREST													
	BEFORE HARVEST						AFTER HARVEST							
	Reference		30 m Buffer		No Buffer		Reference		30 m Buffer		No Buffer		Total	pH
FISH TAXA Thresholds	Mean	S.E.	Mean	S.E.	Mean	S.E.	Mean	S.E.	Mean	S.E.	Mean	S.E.	%	
Northern Redbelly Dace	127	55.2	375.3	226	170	90	349	137	110	63.2	285	243	67.1	[illegible]
5.23[1,2]														
Pearl Dace	19	11	11.8	11.4	54.3	32	58	26	16.5	15.2	31.3	22.5	9.0	[illegible]
Finescale Dace	29	11	14.8	9.4	7.3	7.3	32	9.2	21.8	15.3	28.3	28.3	6.3	[illegible]
Brook Stickleback	4.2	1.9	34.8	18.1	19.7	14	4.0	2.1	20.8	8.7	17.6	15.7	4.8	[illegible]
Common Shiner	0	0	0	0	23	17	0.4	0.4	2.0	2.0	32.3	28	2.7	[illegible]
5.6[1,2,4]														
Central Mudminnow	0.4	0.4	12.5	2.5	32	30.5	15	5.2	6.3	2.7	5.0	4.5	2.7	[illegible]
Dace Hybrids*	15	11	3.8	3.8	4.7	3.3	22	11.2	1.0	0.7	0.33	0.33	2.2	-
Creek Chub	0	0	0	0	15.7	9	0.4	0.4	1.5	1.5	22	14	1.9	[illegible]
5.35[1,2]														
Fathead Minnow	3.9	2.0	1.8	1.44	0	0	15	5.2	15	14.7	0.67	0.67	1.7	[illegible]
5.25[1,2,3]														
White Sucker	0.6	0.6	0.25	0.25	6.7	3.8	16.4	15.3	0.75	0.75	5.0	3.9	1.4	[illegible]
Mottled Sculpin	0	0	0	0	1.0	1.0	0	0	0	0	11	1.3	0.11	[illegible]
Blacknose Shiner	0	0	0	0	0	0	0	0	0	0	1.0	1.0	0.05	[illegible]
Total Abundance	199.1	79.8	454.8	272.9	334.3	207.9	497.3	154.5	195.0	124.7	430.1	363.2		
Fish Species Richness	8		8		10		10		10		12			

*Dace Hybrids = Finescale Dace x Northern Redbelly Dace x Pearl Dace

Table 19. Number, species richness and total percent of each species collected (in descending order of abundance) for 5 minnow traps combined per wetland in conifer forests during July after (1994) timber harvest only. N = 1 for reference, 1 for 30 m buffer and 1 for no buffer wetlands. See Appendix 1 for scientific names. See legend in Table 18 for additional details on pH thresholds.

CONIFER FOREST AFTER HARVEST

FISH TAXA	Reference	30 m Buffer	No Buffer	Total %
Dace Hybrids*	421	0	147	42.7
Northern Redbelly Dace	282	0	248	40
Pearl Dace	1.0	8	144	11.5
Finescale Dace	7	0	26	2.5
Central Mudminnow	0	27	2	2.0
Fathead Minnow	0	0	7	0.5
Common Shiner	0	0	7	0.5
Brook Stickleback	4.0	0	0	0.3
Total Abundance	715	35	581	
Fish Species Richness	5	2	7	

*Dace Hybrids = Finescale Dace x Northern Redbelly Dace x Pearl Dace

In general, the baseline chemistry within the two forest types could account for some of the differences in aquatic biota. A larger species pool of minnows existed in wetlands of the mixedwood forest because these marshes were less acidic, had higher alkalinities (greater buffer capacities for acidic substances), higher Ca and DOC and lower Al concentrations compared to the concentrations of these chemicals in wetlands in the conifer forests. The mixedwood selective cut was not enough to significantly change the chemical parameters to affect fish. However, it is believed that increased timber harvesting approaching a clear-cut within this forest type would result in chemical changes (see chemical discussion) that would effect aquatic biota. Thus, the extent of cut in the catchments would be important for the protection of aquatic biota.

Similar species composition was observed in the study marshes in the mixedwood forests compared to research on fish assemblages in peatland ponds and fens (<10 ha) in northern Ontario (Blancher and McNicol 1986, Bendell and McNicol 1987). Two types of habitats were sampled within the study marshes (i.e., ponds and streams) near Temagami. Bendell and McNicol (1987) reported that cyprinid taxa can be divided into two ecological groups based on these habitats. Pond-dwelling species such as *Phoxinus* spp. (redbelly dace, finescale dace), pearl dace and fathead minnow occurred more frequently in small drainage areas. Stream-dwelling species consisted of common shiner, creek chub and blacknose dace and occurred more frequently in larger drainage areas. The pond species of minnows were ubiquitous in the marshes near Temagami, Ontario and represented 93 percent in mixedwoods and 98 percent in the conifer forest wetlands. The remaining 2–7 percent in the two forested types were those representing the stream-dwelling species that were collected in inflows and outflows of ponds or in stream channels flowing through the wetlands.

In contrast to the chemical and biological results in the selectively cut mixedwood forests, the significantly reduced oxygen concentrations in wetlands resulting from clear-cut timber harvesting without buffers could have an influence on the presence/absence of fish assemblages in small ponds. Previous researchers have recognized that one important factor that distinguishes surface waters containing small fishes only (predominantly cyprinids) from those with small and large fish species is the low DO level in winter. In Wisconsin, predators such as sunfish and pike occurred in small lakes with high winter oxygen levels or in small lakes with low DO concentrations in winter if a stream or connecting lake provided refuge from these conditions in winter (Tonn and Magnuson 1982). When no outlet was present, lakes with low DO in winter lacked these predators, but contained cyprinid-mud minnow assemblages. Although the DO was high (5.4 mg/L) in fall in the present study, further reductions could occur after ice cover in winter. Similarly, McNicol et al. (1987) reported that some of the fishless ponds near Ranger Lake, Ontario may be strongly influenced by winter anoxia.

In addition to low DO in the study wetlands as a result of clear-cutting, the interaction with decreased buffer capacity and elevated acidity could further restrict the presence/absence of small fishes in the study marshes. Significant reductions in pH occurred in the clear-cut timber harvested wetlands. In spring, pH values were depressed to 4.69 in the 30 m buffers and to 4.83 in the no buffer wetlands compared to 5.42 in the reference area. In autumn, pH dropped to 4.9 in the 30 m buffer habitats, and 5.2 in the no buffer sites relative to 5.9 in the reference areas. In a survey of 43 northern Wisconsin lakes, Rahel (1984) concluded that low winter oxygen concentrations harbour cyprinid assemblages if the pH of the surface waters is above 5.2–5.4. Matusek et al. (1990) reported that many minnow species were sensitive to depressed pH levels based on presence/absence of minnows from 488 Ontario lakes. They proposed that the fathead minnow, common shiner, bluntnose minnow (*Pimephales notatus*), blacknose shiner and slimy sculpin (*Cottus coqnatus*) were potential early warning indicator species to changes in the fish community composition as a result of elevated acidity. These species disappear along a graded decline in pH from 6.0 to 5.0. For example, population declines of fathead minnows at pH values below 5.7 were reported in field and laboratory experiments by Zischke et al (1983), Mills et al (1987), Matusek et al. (1990), McCormick et al. (1989), and Gunn and Belzile (1994). The common shiner is also sensitive to pH depressions of 5.6 and below (Holtz and Hutchinson 1989, Matusek et al 1990, Gunn and Belzile 1994). The creek chub and blacknose shiner have pH thresholds ranging from 5.35-5.0 (Matusek et al. 1990 and Gunn and Belzile 1994). Matusek et al. (1990) also predicted that no minnow species would occur at pH values below 4.7. Again, a mean pH level of 4.69 was measured after clear-cut timber harvesting.

Elevated Al concentrations in addition to low pH observed in the clear-cut conifer forest could enhance the species decline of fish assemblages over the reduction of pH values alone. The toxic fraction of Al to biota is predominantly the monomeric inorganic Al form (Driscoll et al. 1980; Hall et al. 1985; 1987). Although only total Al was measured in the present study, increased concentrations of inorganic Al relative to the total occurred as the pH decreased (<5.6, Burrows

1977). Toxic effects of increased inorganic Al and decreased pH levels below 6.0 were reported for the fathead minnow (McCormick et al. 1989), common shiner (Holtz and Hutchinson 1989) and white sucker (Baker and Schofield 1982, Holtz and Hutchinson 1989).

The greatest pH reduction occurred in the clear-cut 30 m buffer sites (pH 4.69). Likewise, the greatest reduction in fish species was found after harvest within a 30 m buffer site (Table 19). Although only suggestive, the results indicated that a greater stress was evident there. The species present at pH 4.69 in the 30 m buffer resembled the mud minnow-dominated communities characteristic of low DOC and low pH (<5.2) lakes reported in Northern Wisconsin (Rahel and Magnuson 1983, Rahel 1984). Similarly, the mean pH level measured (<4.7) was the same as that predicted by Matusek et al (1990) and McNicol et al (1987) for fishless lakes in Ontario. The pearl dace was the only minnow species captured along with the mudminnow. This former species is quite tolerant to acid water. Mills et al. (1987) and Matusek et al. (1990) observed increases in populations of the pearl dace in acid waters based on whole-lake experimental acidification experiments and field surveys in acidic surface waters.

With the exception of DO and Mn concentrations, 30 m buffers did not mitigate changes of the key parameters (pH, Al, Alk., etc.) that can regulate the organization of small fishes in wetland ecosystems. Changes in chemical parameters likely resulted from factors that occurred in the upland forest due to changes in runoff of surface and/or ground water. Increased Al resulted from leaching of the soil as a result of neutralization of acid snowmelt. Increases in DOC concentrations were likely due to leaching, in part, of organic acids from freshly cut debris such as bark and fine particulates on the forest floor. However, buffers may be important in impeding particulate transport to surface waters. Extent of cutting (selective vs. clear-cut) instead of buffers versus no buffer timber management appeared to be more important in regulating chemical changes and impact on aquatic communities.

The pH thresholds for fish taxa in Wisconsin were lower than thresholds for similar fish species in Ontario. The difference in fish sensitivity to pH between those reported by Rahel (1984, <5.2) and Matusek et al (1990, <4.7) may be related to DOC in the study marshes. High DOC concentrations were reported to mitigate the impact of low pH and elevated metal levels on fish (Baker 1982). Similarly, McNicol and Wayland (1992) concluded that the major difference between peatland and fens that were either fishless or had fish (minnows and brook stickleback) was that the latter habitats had higher DOC concentrations.

Other aquatic biota

One cannot discern the direct effect of timber harvesting on insects, crustaceans, leeches, and amphibians because numbers were generally low (Appendix 1). The procedures for collection of these biota were outlined by Blancher and McNicol (1986) and McNicol et al. (1987). However, lists of taxa indicate that the biota collected from the study wetlands can tolerate acidic environments similar to those reported by Blancher and McNicol (1986). Concentrations of pH and Al concentrations measured in the mixedwood forests were at levels that can be tolerated by *Rana* spp. (Clark and Hall 1985, Clark and LaZerte 1985). Similarly, the leech species collected at levels between 5.6–6.06 (Table 16) in the mixedwood wetlands of this study (Appendix 1, Table 13) were reported to occur at pH levels >5.2 in 40 other wetlands (Bendell and McNicol 1991). The insect and crayfish species present in the wetlands are commonly found in acidic environments (Blancher and McNicol (1986). However, based on the chemical results before and after timber harvest in the conifer forest, the decrease in alkalinity and DO and increase in H^+ and Al concentrations were at levels that could be toxic to different life stages of many biotic species within the wetland ecosystems.

Conclusions

Marshes in mixedwood forests are more alkaline and provide a greater buffer capacity to neutralize acidic substances to protect fish than can marshes in conifer forests. Selective cutting protects water quality and aquatic biota. However, clear-cut logging can decrease buffer capacity, then reduce pH and elevate metals high enough to be toxic to many species of aquatic biota. With the higher DOC in spring and fall (although not significantly increased in the former season), DO may be further reduced under ice cover in the winter to cause extinction of small fish and other biota due to clear-cutting. Buffers (30 m in width) may help prevent changes in some chemical parameters, such as DO, Mn and suspended sediments, that could be toxic to different life stages of fish and other biota. However, the 30 m buffers did not mitigate the flux to wetlands of potentially toxic dissolved chemicals that were transported from the terrestrial ecosystem due to clear-cut timber harvesting.

SUMMARY

Only short-term responses of biotic communities were measured as a result of different timber harvesting methods in this study. Thus, longer-term responses can only be inferred at this time. Observations that were measured will likely change and be influenced by successional changes in regrowth of forest communities (Bormann and Likens 1979).

Natural background differences in richness of songbirds, owls, fur-bearers, fish and water chemistry existed between the two forest types. In general, the mixedwood forests had greater structure and species richness of trees, songbirds, fur-bearers and fish. Surface water chemistry in the mixedwood forest had greater buffer capacity (alkalinity), higher pH and lower concentrations of toxic metals (e.g. aluminum) than wetlands in the coniferous forest.

Selective cutting in the mixedwood forests did not result in appreciable changes in forest structure, songbirds and owls, fur-bearers, fish communities or the flux of chemical parameters between the terrestrial and aquatic interface. However, large changes in community structure and ecosystem function resulted from clear-cutting of the conifer forest with and without buffers.

The bird communities were not affected appreciably, possibly because of their aerial mobility, allowing free movement in and out of the area. The bird species most negatively impacted by timber harvest to the shore were those dependent upon snags. Perhaps no overall changes for most of the bird communities in the conifer clearcut resulted because the abundant species associated with wetland/riparian habitat are those commonly found in shrubby areas, young forests or edges. Therefore cutting the adjacent upland forest may have had little negative impact because pockets of shrubby, polewood vegetation remained along wetland edges. Certain owl species seem to take advantage of newly-created open areas by increasing in abundance likely due to more readily available prey. However, owl abundance was not strongly influenced by the use of buffers. Buffers seem to aid in maintaining furbearer communities since open areas resulting from clear-cutting reduced the abundance of hares and squirrels in wetland/riparian areas. Providing the riparian habitat for hares and squirrels could also increase the numbers of upland predators of furbearers.

Clear-cutting caused significant changes in aquatic chemistry while selective cutting did not. The chemical changes were large enough to alter fish community composition. In general, biodiversity of aquatic and terrestrial communities was similar before and after logging. However some shifts in species did occur as a result of habitat modification. Buffers mitigated only a few of the chemical changes but did not for many others, implying that the width of buffer was too small and/or that the extent of the clear-cut was too large to protect against chemical flux to the wetlands.

Even though impact on different biotic communities was not evident for many of the species, the results showed that changes in terrestrial species were greater in the clear-cut without buffer compared to 30 m buffers. Therefore, clear-cutting the forest to the wetland edge would significantly reduce the biodiversity of wildlife compared to wetlands with a minimum of 30 m of buffer. If the wetland-forested ecosystems were linked via corridors between wetlands, some of the original biodiversity would likely be maintained but at a reduced level due to loss of forest structure. From the results of the wetland study, it is clear that forest ecosystem health is linked to the extent of cutting within the watershed and within the upland forest around the wetlands.

Consideration should be given to wildlife habitat by forest managers in their timber harvesting decision-making process. The use and size of buffers, for example would be best applied on a site-specific basis. Novak (1987) showed that beavers may be negatively impacted by the use of buffers around all aquatic areas. In order to meet waterfowl and aquatic furbearer goals, beaver impoundments are an important component (Wiley 1988). In areas where beavers are active, perhaps cutting only some conifer trees might facilitate regrowth of hardwoods and would help to ensure that less desirable timber species such as balsam fir and spruce would not predominate during regeneration of riparian forests.

When designing buffers around wetlands, forest managers must also give consideration to terrestrial and aquatic resources. The size of the buffer may protect one aspect of a wetland/riparian ecosystem, but not others. Therefore, in conifer forests, the buffer width may vary to meet a specific need. Wetland buffers appear to be less beneficial in local mixedwood forests harvested in winter, likely due to the large amount of small diameter timber remaining. Further studies should examine the relationship between wetland size and wildlife use, benefits of various buffer widths, the impact of partial cutting in conifer-dominated wetland/riparian ecosystems and the longterm impact of clearcut harvesting on the adjacent coniferous wetlands used in this study.

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APPENDIX 1: SPECIES LISTS

BIRD COMMUNITIES

Breeding bird census completed in May-June of 1993 and 1994.
In order of descending abundance.

Most Common 30 Species Comprising 90 percent of Total Bird Abundance

	Common Name	Scientific Name
1	White-throated sparrow	*Zonotrichia albicollis*
2	Magnolia warbler	*Dendroica magnolia*
3	Nashville warbler	*Vermivora ruficapilla*
4	Yellow-rumped warbler	*Dendroica coronata*
5	Ruby-crowned kinglet	*Regulus calendula*
6	Common yellowthroat	*Geothlypis trichas*
7	Ovenbird	*Seiurus aurocapillus*
8	Winter wren	*Troglodytes troglodytes*
9	Swamp sparrow	*Melospiza georgiana*
10	Yellow-bellied flycatcher	*Empidonax flaviventris*
11	Golden-crowned kinglet	*Regulus satrapa*
12	Swainson's thrush	*Catharus ustulatus*
13	Black-and-white warbler	*Mniotilta varia*
14	Hermit thrush	*Catharus guttatus*
15	Red-breasted nuthatch	*Sitta canadensis*
16	Northern flicker	*Colaptes auratus*
17	Chestnut-sided warbler	*Dendroica pensylvanica*
18	Black-capped chickadee	*Parus atricapillus*
19	Canada warbler	*Wilsonia canadensis*
20	Yellow-bellied sapsucker	*Sphyrapicus varius*
21	Solitary vireo	*Vireo solitarius*
22	Blackbumian warbler	*Dendroica fusca*
23	Red-eyed vireo	*Vireo olivaceus*
24	Brown creeper	*Certhia americana*
25	Black-throated blue warbler	*Dendroica caerulescens*
26	Purple finch	*Carpodacus purpureus*
27	Black-throated green warbler	*Dendroica virens*
28	Dark-eyed junco	*Junco hyemalis*
29	Gray jay	*Perisoreus canadensis*
30	Alder flycatcher	*Empidonax alnorum*

Less Common 68 Species Comprising 10 percent of Total Bird Abundance

31	Olive-sided flycatcher	*Contopus borealis*

32	Pileated woodpecker	*Dryocopus pileatus*
33	Hairy woodpecker	*Picoides villosus*
34	Blue jay	*Cyanocitta cristata*
35	Least flycatcher	*Empidonax minimus*
36	Boreal chickadee	*Parus hudsonicus*
37	Bay-breasted warbler	*Dendroica castanea*
38	Northern waterthrush	*Seiurus noveboracensis*
39	Veery	*Catharus fuscescens*
40	American redstart	*Setophaga ruticilla*
41	Downy woodpecker	*Picoides pubescens*
42	Common grackle	*Quiscalus quiscala*
43	Scarlet tanager	*Piranga olivacea*
44	Song sparrow	*Melospiza melodia*
45	Black-backed woodpecker	*Picoides arcticus*
46	Tennessee warbler	*Vermivora peregrina*
47	Great crested flycatcher	*Myiarchus crinitus*
48	Tree swallow	*Tachycineta bicolor*
49	Pine warbler	*Dendroica pinus*
50	Yellow warbler	*Dendroica petechia*
51	American bittem	*Botaurus lentiginosus*
52	American robin	*Turdus migratorius*
53	Ruby-throated hummingbird	*Archilochus colubris*
54	Ruffed grouse	*Bonasa umbellus*
55	Rose-breasted grosbeak	*Pheucticus ludovicianus*
57	Mourning warbler	*Oporornis philadelphia*
58	Wilson's warbler	*Wilsonia pusilla*
59	Evening wrosbeak	*Coccothrautes vespertinus*
60	Red-winged blackbird	*Agelaius phoeniceus*
61	Northern raven	*Corvus corax*
62	Belted kingfisher	*Megaceryle alcyon*
63	Cape may warbler	*Dendroica tigrina*
64	Hooded merganser	*Lophodytes cucullatus*
65	Spotted sandpiper	*Actitis macularia*
66	Cedar waxwing	*Bombycilla cedrorum*
67	Mallard	*Anas platyrhynchos*
68	Eastern kingbird	*Tyrannus tyrannus*
69	Black duck	*Anas rubripes*
70	Solitary sandpiper	*Tringa solitaria*
71	Northern three-toed woodpecker	*Picoides tridactylis*
72	Lincoln's sparrow	*Melospiza lincolnii*
73	Chimney swift	*Chaetura pelagica*
74	Pine siskin	*Carduelis pinus*
75	Pine grosbeak	*Pinicola enucleator*
76	Eastern wood-pewee	*Contopus virens*

77 Palm warbler	*Dendroica palmarum*
78 Great blue heron	*Ardea herodias*
79 White-crowned sparrow	*Zonotrichia leucophrys*
80 Green-winged teal	*Anas crecca*
81 Wood duck	*Aix sponsa*
82 American goldfinch	*Carduelis tristis*
83 Ring-necked duck	*Aythya collaris*
84 Northern cardinal	*Cardinalis cardinalis*
85 Broad-winged hawk	*Buteo platypterus*
86 American crow	*Corvus brachyrhynchos*
87 Lesser yellowlegs	*Tringa flavipes*
88 Eastern bluebird	*Sialia sialis*
89 Killdeer	*Charadrius vociferus*
90 Common loon	*Gavia immer*
91 White-winged crossbill	*Loxia leucoptera*
92 Common merganser	*Mergus merganser*
93 Sharp-shinned hawk	*Accipiter striatus*
94 Red crossbill	*Loxia curvirostra*
95 Rusty blackbird	*Euphagus carolinus*
96 Turkey vulture	*Cathartes aura*
97 Sandhill crane	*Grus canadensis*
98 Northern harrier	*Circus cyaneus*

OWL COMMUNITIES

Owl vocalization survey carried out in March of 1993, 1994 and 1995.
In order of abundance.

Common Name	Scientific Name
Great horned owl	*Bubo virginianus*
Barred owl	*Strix varia*
Boreal owl	*Aegolius funereus*
Northern saw-whet owl	*Aegolius acadicus*
Great gray owl	*Strix nebulosa*

FOREST VEGETATION

Vegetation found in study area (alphabetically).

Common Name	Scientific Name
Balsam fir	*Abies balsamea* (L.) Mill.
Beaked hazel	*Corylus cornuta* Marsh.
Black spruce	*Picea mariana* (Mill.) B.S.P.

Chokecherry	*Prunus virginiana* (L.)fil.
Eastern white cedar	*Thuja occidentalis* L.
Eastern white pine	*Pinus strobus* L.
Fly honeysuckle	*Lonicera canadensis* Bartr.
Jack pine	*Pinus banksiana* Lamb.
Largetooth aspen	*Populus grandidentata* Michx.
Low sweet blueberry	*Vaccinium angustifolium* Ait.
Mountain maple	*Acer spicatum* Lam.
Mountain-ash	*Sorbus decora* Marsh.
Red maple	*Acer rubrum* L.
Red pine	*Pinus resinosa* Ait.
Serviceberry	*Amelanchier sanguinea* Pursh DC.
Speckled alder	*Alnus rugosa* (DuRoi) Spreng.
Sugar maple	*Acer saccharum* Marsh
Tamarack	*Larix laricina* (Du Roi) K. Koch
Trembling aspen	*Populus tremuloides* Michx.
White birch	*Betula papyrifera* Marsh.
White spruce	*Picea glauca* (Moench) Voss
Wild raisin	*Viburnum cassinoides* L.
Willow	*Salix* sp.
Yellow birch	*Betula alleghaniensis* Britton

FUR-BEARER COMMUNITIES

Snow tracking survey carried out in January and February of 1993 and 1995.
In order of abundance.

Common Name	Scientific Name
Snowshoe hare	*Lepus americanus*
Red squirrel	*Tamiasciurus hudsonicus*
Weasel	*Mustela erminea*
Marten	*Martes americana*
Red fox	*Vulpes fulva*
Fisher	*Martes pennanti*
River otter	*Lutra canadensis*
Gray wolf	*Canis lupus*
Mink	*Mustela vison*
Lynx	*Lynx canadensis*

WATER QUALITY

Samples taken from inflow and outflow channels during May, August of 1993 and May, July, and August, 1994. Water chemistry parameters tested at the Dorset Research Center, Ontario Ministry of the Environment.:

- Total unfiltered phosphorus
- Aluminum.
- Iron.
- Calcium.
- Manganese.
- Ammonium.
- Nitrate.
- Sulphate.
- pH.
- Alkalinity (total inflection point).
- Dissolved organic carbon.
- Conductivity.
- Total suspended solids
- Dissolved oxygen.
- Nitrogen (total kjeld).

AQUATIC BIOTA

Minnowtrapping undertaken during June 27 to July 7 of 1993 and 1994.
In order of abundance.

Common Name	Scientific Name
Fish:	
Northern redbelly dace	*Phoxinus eos*
Common shiner	*Luxilus cornutus*
Pearl dace	*Margariscus margarita*
Finescale dace	*Phoxinus noegaeus*
Creek chub	*Semotilus atromaculatus*
Brook Stickleback	*Culaea inconstans*
Central mudminnow	*Umbra limi*
White sucker	*Catostomus commersoni*
Mottled sculpin	*Cottus bairdi*
Blacknose shiner	*Notropis heterolepsis*
Fathead minnow	*Pimephales promelas*
Dace hybrids	*Phoxinus eos x Margariscus mrgarita*
	Foxinus noegaeus x M. margarita
Insects:	
Giant wter bg	*Lethocerus sp.* (immatures)
Diving betle	*Dytiscus sp.* (immatures)
Diving betle	*Acillus semisulcatus* Aube
Diving betle	*Dytiscus verticalis* Say
Dragonfly darner	*Aeshna interrupta* Walker
Giant wter bug	*Lethocerus americanus* (Leidy)
Diving beetle	*Neoscutopterus hornii* (Crotch)
Diving beetle	*Acillus sylvanus* Hilsenhoff
Diving beetle	*Colybetes paykulli*
Diving beetle	*Colybetes sculptilis*

Diving beetle	*Hydrochara obtusata* (Say)

Leeches:

(no common names)	*Percymoorensis marmoratis* *Nephelopsis obscura* *Macrobdella decora* *Mollibdella grandis*

Amphibians:

Green frog	*Rana clamitans*
Mink frog	*Rana septentrionalis*
Wood frog	*Rana sylvatica*
Red-spotted newt	*Notophalmus viridescens*
Northern leopard frog	*Rana pipiens*
Dina/mooreobdella complex	

Crayfish:

Robust crayfish	*Cambarus robustus*
Northern clearwater crayfish	*Orconectes propinquus*

<u>Total specimens collected in 1993</u>

12 species of fish	5,761 specimens
15 species of aquatic insects	147 specimens
4 species of leeches	90 specimens
4 species of amphibians	47 specimens
2 species of crayfish	13 specimens

<u>Total specimens collected in 1994</u>

13 species of fish	6,976 specimens
11 species of aquatic insects	129 specimens
5 species of amphibians	41 specimens
6 species of leeches	31 specimens
1 species of crayfish	15 specimens

Taxonomic identification to genus and species was carried out by various departments of the Royal Ontario Museum in Toronto. Leeches were also identified at the Canadian Museum of Nature in Ottawa.

APPENDIX 2. Forest basal area (mean sq m/ha ± S.E., n=5) within 30 m from wetland perimeters by treatment and year (1993=before, 1994=after) for conifer forests. Asterisks indicate significant differences between (2-way ANOVA) and within (Scheffe's test) treatments.

CONIFER FOREST 15 m	BEFORE HARVEST						AFTER HARVEST					
TREE SPECIES	Reference		30 m Buffer		No Buffer		Reference		30 m Buffer		No Buffer	
	mean	S.E.	mean	S.E.	mean	S.E.	mean	S.E.	mean	S.E.	mean	S.E.
Jack Pine	7.0	1.9	9.9	1.8	9.6	1.7	7.0	1.9	9.9	1.8	3.2	0.9
Black Spruce	6.4	1.5	7.3	1.8	7.0	1.5	6.4	1.5	7.3	1.8	2.4	0.9
White Spruce	4.0	1.1	0.9	0.5	2.3	0.6	4.0	1.1	0.9	0.5	1.6	0.7
White Birch	3.0	1.0	0.5	0.2	1.3	0.5	3.0	1.0	0.5	0.2	0.5	0.3
White Pine	2.2	0.3	0.3	0.2	0.9	0.3	2.2	0.3	0.3	0.2	0.5	0.2
Balsam Fir	0.7	0.3	1.0	0.4	4.1	1.3	0.7	0.3	1.0	0.4	1.6	0.6
Poplar	0.4	0.2	0.0	0.0	0.0	0.0	0.4	0.2	0.0	0.0	0.0	0.0
Red Pine	0.3	0.2	0.0	0.0	0.0	0.0	0.3	0.2	0.0	0.0	0.0	0.0
Cedar	0.2	0.1	0.0	0.0	0.0	0.0	0.2	0.1	0.0	0.0	0.0	0.0
Maple	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Yellow Birch	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Ash	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Total Tree Abundance	24.2	3.7	19.9	2.9	25.2	3.4	24.2	3.7	19.9	2.9	9.8 **	2.3
Tree Species Richness	4.6	0.5	2.9	0.4	4.3	0.5	4.6	0.5	2.9	0.4	2.0 **	0.4
Total Snag Abundance	2.5	0.6	3.0	0.7	2.5	0.5	2.5	0.6	3.0	0.7	1.4	0.5

* $P < 0.05$ ** $P < 0.01$

APPENDIX 3. Forest basal area (mean sq m/ha ± S.E., n=5) at 50 m from wetland perimeters by treatment and year (1993=before, 1994=after) for conifer forests. Asterisks indicate significant differences between (2-way ANOVA) and within (Scheffe's test) treatments.

CONIFER FOREST 50 m	BEFORE HARVEST						AFTER HARVEST					
TREE SPECIES	Reference		30 m Buffer		No Buffer		Reference		30 m Buffer		No Buffer	
	mean	S.E.	mean	S.E.	mean	S.E.	mean	S.E.	mean	S.E.	mean	S.E.
Jack Pine	9.2	2.0	15.1	2.7	11.6	2.4	9.2	2.0	5.6	1.5	4.6 *	1.9
Black Spruce	4.8	1.5	8.4	1.7	6.3	1.5	4.8	1.5	3.0	1.0	1.0 *	0.7
White Spruce	2.4	0.7	0.4	0.3	2.4	0.6	2.4	0.7	1.4	0.6	0.7	0.4
White Birch	2.0	0.7	1.0	0.3	2.7	0.7	2.0	0.7	0.8	0.3	0.2 *	0.1
White Pine	4.0	1.0	0.5	0.2	0.9	0.3	4.0	1.0	0.4	0.2	0.2	0.1
Balsam Fir	0.2	0.1	0.6	0.2	2.1	0.9	0.2	0.1	0.4	0.2	0.9	0.4
Poplar	1.7	0.6	0.5	0.2	0.7	0.4	1.7	0.6	0.3	0.3	0.0	0.0
Red Pine	0.6	0.3	0.0	0.0	0.0	0.0	0.6	0.3	0.0	0.0	0.0	0.0
Cedar	0.1	0.1	0.0	0.0	0.0	0.0	0.1	0.1	0.0	0.0	0.0	0.0
Maple	0.1	0.1	0.2	0.1	0.2	0.1	0.1	0.1	0.2	0.1	0.0	0.0
Yellow Birch	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Ash	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Total Tree Abundance	25.1	3.3	26.9	3.9	26.9	3.7	25.1	3.3	12.1 **	2.5	7.7 **	2.5
Tree Species Richness	4.8	0.5	4.1	0.5	4.8	0.6	4.8	0.5	2.4	0.4	1.1 **	0.3
Total Snag Abundance	4.8	0.9	9.2	2.0	3.2	0.6	4.8	0.9	1.2 **	0.4	0.9	0.5

* $P < 0.05$ ** $P < 0.01$

APPENDIX 4. Forest basal area (mean sq m/ha ± S.E., n=5) within 30 m from wetland perimeters by treatment and year (1993=before, 1994=after) for mixedwood forests.

MIXEDWOOD FOREST 15 m	BEFORE HARVEST						AFTER HARVEST					
TREE SPECIES	Reference		30 m Buffer		No Buffer		Reference		30 m Buffer		No Buffer	
	mean	S.E.	mean	S.E.	mean	S.E.	mean	S.E.	mean	S.E.	mean	S.E.
Cedar	1.1	0.4	2.4	1.1	5.5	1.9	1.1	0.4	2.4	1.1	5.5	1.9
Balsam Fir	3.8	1.0	3.4	1.0	3.9	1.1	3.8	1.0	3.4	1.0	3.0	0.9
Black Spruce	1.0	0.4	1.9	0.8	5.2	1.4	1.0	0.4	1.9	0.8	3.6	1.1
White Spruce	2.5	0.5	4.3	1.4	2.5	0.7	2.5	0.5	4.3	1.4	1.7	0.6
White Pine	0.7	0.3	1.4	0.4	2.5	0.5	0.7	0.3	1.4	0.4	2.0	0.5
White Birch	3.7	0.9	1.8	0.6	1.5	0.8	3.7	0.9	1.8	0.6	1.7	0.8
Poplar	0.6	0.5	1.0	0.5	0.1	0.1	0.6	0.5	1.0	0.5	0.0	0.0
Red Pine	0.0	0.0	0.0	0.0	0.6	0.2	0.0	0.0	0.0	0.0	0.5	0.2
Maple	1.8	0.6	0.2	0.2	0.1	0.1	1.8	0.6	0.2	0.2	0.2	0.2
Yellow Birch	0.6	0.3	0.2	0.1	0.0	0.0	0.6	0.3	0.2	0.1	0.0	0.0
Ash	0.3	0.2	0.2	0.2	0.0	0.0	0.3	0.2	0.2	0.2	0.0	0.0
Total Tree Abundance	16.0	2.0	16.9	2.5	21.9	3.0	16.0	2.0	16.9	2.5	18.1	2.8
Tree Species Richness	4.2	0.4	3.6	0.5	4.2	0.4	4.2	0.4	3.7	0.5	3.5	0.4
Total Snag Abundance	2.5	0.5	2.8	0.6	3.2	0.6	2.5	0.5	2.8	0.6	2.1	0.5

APPENDIX 5. Forest basal area (mean sq m/ha ± S.E., n=5) at 50 m from wetland perimeters by treatment and year (1993=before, 1994=after) for mixedwood forests.

MIXEDWOOD FOREST 50 m	BEFORE HARVEST						AFTER HARVEST					
TREE SPECIES	Reference		30 m Buffer		No Buffer		Reference		30 m Buffer		No Buffer	
	mean	S.E.	mean	S.E.	mean	S.E.	mean	S.E.	mean	S.E.	mean	S.E.
Cedar	1.2	0.5	2.4	1.0	3.7	1.3	1.2	0.5	2.3	1.0	3.7	1.3
Balsam Fir	3.4	0.8	1.3	0.4	1.7	0.8	3.4	0.8	1.4	0.5	1.3	0.5
Black Spruce	1.8	0.8	3.0	1.2	3.9	1.1	1.8	0.8	2.0	1.0	2.8	0.9
White Spruce	1.6	0.4	2.6	0.6	1.8	0.5	1.6	0.4	1.8	0.6	0.6	0.3
White Pine	1.4	0.7	2.6	0.6	4.9	1.1	1.4	0.7	2.2	0.5	4.2	0.8
White Birch	0.4	1.2	1.4	0.4	1.7	0.6	0.4	1.2	1.4	0.4	1.4	0.4
Poplar	0.9	0.3	4.1	1.6	1.4	0.5	0.9	0.3	2.9	1.3	0.6	0.3
Red Pine	0.0	0.0	0.2	0.2	1.2	0.5	0.0	0.0	0.2	0.2	1.1	0.5
Maple	3.6	0.8	0.3	0.1	0.0	0.0	3.6	0.8	0.6	0.2	0.0	0.0
Yellow Birch	0.7	0.3	0.0	0.0	0.0	0.0	0.7	0.3	0.0	0.0	0.0	0.0
Ash	0.3	0.2	0.1	0.1	0.0	0.0	0.3	0.2	0.0	0.0	0.0	0.0
Total Tree Abundance	19.2	2.5	17.9	2.6	20.2	2.6	19.2	2.5	14.8	2.3	15.7	2.2
Tree Species Richness	4.8	0.5	4.2	0.5	4.6	0.5	4.8	0.5	3.7	0.4	3.8	0.5
Total Snag Abundance	3.0	0.6	3.5	0.8	2.9	0.5	3.0	0.6	3.4	0.8	2.8	0.5

APPENDIX 6

STANDING COARSE WOODY DECAY DESCRIPTIONS.

Decay Classes	Deciduous and Coniferous Snags
Decay class 1.	Tree is recently dead. Top is intact. Most fine branching still present. Bark is intact.
Decay class 2.	Top is intact. Most of the fine branches have dropped. More than 50% of the coarse branches are left. Bark may begin to loosen.
Decay class 3.	Top is intact. Fewer than 50% of the coarse branches are left. Depending on the species, bark may (e.g. white pine) or may not (e.g. white birch) have sloughed off.
Decay class 4.	Top is broken. No coarse branches remain. Bark may or may not have sloughed off. Height > 6 m.
Decay class 5	(Stub). Top repeatedly broken. No coarse branches remain. Bark may or may not have sloughed off. Height < 6m.

APPENDIX 7

BIRD CENSUS METHODS and SCORING

Analysis of Modified Spot Map Data used in Territory Mapping

The objective of this exercise was to determine the number of individual territories for each species associated with each wetland. The modified spot maps were used to produce territory maps on tracing paper used in both the territory and habitat databases. Territories were color-coded on the onionskin sheets and given a subscript according to bird species. Lead pencil was used to encircle a territory value of 1.0, blue for 0.5, orange for 0.1, and purple for a value of 0.001 on onionskins (Welsh pers. comm. 1993). An explanation of counts according to field observation follows.

Simultaneous Observations

Non-overlapping rings were drawn around distinct clusters of registrations (singing males or calling individuals etc.). The most important criteria to decide whether two clusters were separate was to use simultaneous registrations (i.e. two males singing simultaneously against one another). Even when there was only one registration with a male singing simultaneously with another, it was treated as a separate cluster and given a value of 1 territory (e.g. one mated pair). If there were no simultaneous records then judgment was used to decide whether two or more distant registrations represented the same individual which had traveled between points or whether these were two separate territories. To resolve this problem required knowledge of territory size and movements of different bird species. For example, Least flycatchers (*Empidonax minimus*) are semi-colonial breeders with very small territories, whereas Ruby-crowned kinglet (*Regulus calendula*) territories are much larger; a Gray jay (*Perisoreus canadensis*) family may use the entire wetland/edge community and beyond.

Singing or Calling Birds

To begin analysis, territory boundaries were sketched using simultaneous records of singing males and other registrations. Territories were only indicated using singing males, although calling individuals were included within the cluster. For instance, several records of a calling Hermit thrush (*Catharus guttatus*) could not be counted as a territory if a singing male was never recorded as this may have been a bird passing through the area on migration or a non-breeder. Two or more registrations of singing birds on different visits was counted as 1. A single registration was counted as one pair for Cedar waxwings (*Bombycilla cedrorum*) as their breeding season was likely after the census period. Two or more registrations of calling birds was also counted as one territory for Red-breasted nuthatch (*Sitta canadensis*), Black-capped chickadee (*Parus atricapillus*), Brown creeper (*Certhia americana*), and Yellow-bellied sapsucker (*Sphyrapicus varius*) or woodpecker species which may have begun breeding prior to censuses. For most other species, one registration of a singing bird was counted as 0.5.

Birds with large territories and assumed to be possible breeders, were also counted as 0.5. These included species such as American bittern (*Botaurus lentiginosus*), Ring-necked duck (*Aythya collaris*), and Mallard (*Anas platyrhynchos*). For these species, one registration of a calling bird was given a 0.1 score, as this may have been a transient individual, however, two registrations of calling birds on different dates was recorded in the database as 0.5, as this denoted a possible breeder. Species with large territories (such as Northern Raven *Corvus corax*, Broad-winged Hawk *Buteo playpterus*, and Great Blue Heron *Ardea herodias*) that did not breed in the wetland/edge community itself, were given 'visitor' status and denoted by presence only (0.1 times the number observed). This is because it would have been impossible to determine boundaries for these territories. If species were known to have bred on the wetland/edge community then they were recorded as breeders (0.5).

Semi-colonial Species

Mapping methods work well for territorial songbird species that sing or call consistently. However, even among songbirds the mapping method works better for some species than others. For semi-colonial species (e.g. Pine Siskin *Carduelis pinus*) clusters may represent groups of territories and the highest count within a cluster was the maximum number of individuals. If all records were of single flying birds then clusters were drawn and totals calculated. We assumed that the sex ratio was 50:50, the maximum count divided by two was taken as the number of pairs breeding in or by the wetland. A fly-over of a flock of non-territorial or semi-colonial birds was counted as 0.001 times the flock size or number observed.

Spread of Dates

The spread of dates over which records occur is normally used in standard interpretations of maps (i.e. if 9 visits are made as in the IBCC (1969), a minimum of 3 records is required to from a cluster and these must be separated by a minimum of 10 days), but because only 3 visits were used, this rule was not applicable.

Edge Clusters

These were dealt with in order to eliminate overlap between adjacent wetlands. Territories were only used when at least a portion of the territory overlapped with the wetland/edge community (e.g. to 50 m). If a territory was entirely beyond 50 m of a wetland, it was not entered into the database for analysis of territories. Edges occurred where a particular study wetland was in series with an adjacent wetland lying side-by-side within harvest blocks. Here, a line was drawn between wetlands and territories were split with that fraction of territory assigned to either wetland (e.g. 50 percent or 33 percent). In this way, bird territories were not counted twice. This situation occurred with study wetlands identified as A and B as well as #21, 27, and 29.

Field data collection consistency was maintained in that one of the birders returned in the second year and observed the same wetlands as in the previous year. In the second year, a new observer was trained by the experienced person according our field data collection and modified spot mapping techniques.