COMPOSITION AND CONSERVATION OF RIPARIAN BIRD ASSEMBLAGES IN A BALSAM FIR ECOSYSTEM

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Composition and conservation of riparian bird assemblages in a balsam fir ecosystem

by

© Darroch M. Whitaker

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the requirements for the degree of Master of Science

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Abstract

Undisturbed riparian zones are typically viewed as the highest quality habitat available to wildlife in forested landscapes and, in keeping with this, are seen as having the highest biodiversity. Riparian buffer strips are retained during clearcutting throughout most of the boreal forest, a practice promoted as a means to reduce the impact of harvesting on terrestrial fauna. Current perceptions of the importance of riparian zones to wildlife originated from studies in southwestern North America, yet their generalization to boreal forests remains relatively untested. Furthermore, little research has been conducted to quantify the extent to which buffer strips are used by wildlife. This study was designed to evaluate the relative importance of riparian habitat for breeding birds in a boreal forest ecosystem and to assess the conservation potential of riparian buffer strips in areas of extensive clearcutting.

Breeding birds were surveyed in riparian edge, non-riparian edge (clearcut or access road), interior forest and buffer strip habitats in balsam fir (*Abies balsamea*) forests in western insular Newfoundland. Observations from riparian edge, non-riparian edge and interior forest transects were compared in order to describe the assemblages associated with each of these habitat types and to group species into habitat selection guilds. Several distinguishing species were associated with each of these habitat types, and five habitat guilds were differentiated. Total abundance and species richness did not differ between riparian and interior forest transects, but were significantly higher on non-riparian edge than riparian transects. Different habitat features lead to the development of distinct bird assemblages along the two edge types. Based on this and other recent studies, it is apparent that relatively high riparian biodiversity may be the exception for bird assemblages in coniferous and coniferous-deciduous mixed forests, where interior forest species form an important component of the avifauna.

Comparisons were made between bird assemblages (grouped by habitat guild) observed along undisturbed shorelines and buffer strips. Total avian abundance was higher in buffer strips than riparian controls, largely due to significantly higher counts of birds from the ubiquitous and open/edge guilds. Abundance of forest generalist, interior forest and riparian species were similar between the two shoreline types. Counts of riparian species did not increase in wider buffers, likely due to the association of these birds with habitat adjacent to the water, which does not increase in proportion to strip width. Riparian buffer strips did, however, provide habitat for a diverse avian assemblage, and retained many riparian and woodland species in areas of intensive clearcutting. Interior forest species, many of which are declining in northeastern North America, were more abundant in wider buffers. However, even in the widest strips (40-50 m) they were rare when compared to local interior forest habitat, and three of six species in the guild were not observed in any buffer strip. It is clear that separate (but complementary) conservation strategies are required to protect riparian and interior species. Interior species are likely not afforded adequate protection in boreal forests, where conservation efforts focus largely on preserving riparian habitat.

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Foreword

This research project was carried out as part of the Copper Lake Buffer Zone Study, which was initiated by the Newfoundland Forest Service in 1993. A description of this study, which also included projects considering water quality, brook trout (*Salvelinus fontinalis*), and mammals, has been provided in Scruton et al. (1995). A paper detailing results presented in Chapter 3 of this dissertation has been accepted for publication in *Canadian Journal of Forest Research* (Whitaker and Montevecchi 1997).

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BUF	Transect placed in a riparian buffer strip.
INT	Transect placed in interior forest habitat.
NFE	Transect placed along a non-riparian forest edge.
RIP	Transect placed along an undisturbed riparian forest edge.

Chapter 1. Introduction

1.1 Riparian ecology

Riparian zones are portions of terrestrial ecosystems associated with and influenced by the shorelines of lakes and streams. This influence is expressed most visibly in vegetation communities, which differ from those on uplands (caused by increased soil moisture, periodic flooding, etc.), but also is reflected in the associated wildlife assemblages. The study of riparian ecology has developed primarily in southwestern North America since the 1950s (see Johnson and Lowe 1985, Hooper 1989). Here wooded, mesic riparian zones contrast sharply with more open and arid uplands and consequently support a relatively high abundance and diversity of plants and wildlife (Hubbard 1977, Johnson and Lowe 1985, Szaro and Jakle 1985, Knopf 1986). It is clear that many species would be excluded from this region in the absence of intact riparian habitat (Hubbard 1977, Knopf 1986). Because of this ecological importance and limited extent (< 1% of land area; Knopf et al. 1988), as well as high social and economic value, riparian habitat has become a focal topic of research and management activity in southwestern North America.

Subsequently, the importance of riparian habitat in arid regions has been generalized to represent conditions throughout North America (Hooper 1989). Unmanaged riparian zones are typically associated with enhanced water quality and aquatic habitat, productive vegetation communities, and terrestrial wildlife habitat of high quality (e.g., Thomas et al. 1979, Barton et al. 1985, Naiman et al. 1993, Stocek 1994).

Currently regulations aimed at protecting riparian habitat are in place throughout most of the continent (Knopf et al. 1988). In the Canadian boreal forest, where 300 000 - 500 000 ha of forest are clearcut annually, provinces typically have implemented legislation restricting disturbance in riparian zones (Canadian Forest Service 1993). Considering the extent of this protection, surprisingly little has been done to assess the use of riparian habitat by wildlife in northern and eastern North America, where forests are generally continuous between shorelines and uplands. Hooper (1989) reviewed published studies of riparian wildlife and found that 80 % were conducted west of the Mississippi between 1978 - 88. Further, the majority of studies conducted in the East focused on fish and water quality. As a consequence, riparian management decisions (and expected benefits to terrestrial wildlife) in northeastern North America have been based on "best available information" (Hooper 1989, Scruton et al. 1995). This shortcoming has been broadly recognized, and since Hooper's (1989) review several relevant studies have been conducted or are ongoing in the Northeast (e.g., Small and Hunter 1989, Johnson and Brown 1990, Triquet et al. 1990, Gates and Giffen 1991, Hooper 1991, Darveau et al. 1994, Darveau et al. 1995, LaRue et al. 1995, Murray and Stauffer 1995, Spackman and Hughes 1995, Scruton et al. 1995, Parker et al. 1996). However, due to the concurrency of these studies and the recent presentation of results, there has been little synthesis or application of new information.

1.2 Conservation of riparian habitat

Conservation of riparian habitat typically involves either the restriction of activity (e.g., application of chemicals, cultivation, road building etc.) along shorelines or, in cases where riparian zones are disturbed already (e.g., after grazing), promoting the development of riparian vegetation. Legislation usually requires that "buffer strips" of uncut trees be left along shorelines during forest harvesting. In some jurisdictions thinning of portions of these strips is either allowed to increase timber yield, or required to reduce the incidence of windthrow among residual trees. Consequently decisions involved in the development of forest management legislation for riparian habitat typically involve determining the appropriate width of buffer strip and, in some cases, patterns of harvesting within these strips. Undoubtedly this relative simplicity, combined with the expected multiple benefits (e.g., maintenance of water quality, conservation of terrestrial and aquatic wildlife, aesthetics), has facilitated the widespread protection of riparian habitat. Further, riparian vegetation in the boreal forest often is characterized by forbs, shrub thickets and (on wet soils) stunted trees, in which case it is of little commercial value.

While wildlife associations with riparian and upslope habitats are becoming better understood, studies of the use of buffer strips are still few. In total, four papers have been published documenting the use of no-harvest buffer strips by birds in North America (Johnson and Brown 1990, Triquet et al. 1990, Darveau et al. 1994, Darveau et al. 1995). Only the two most recent, which present different analyses of the same data set, incorporate a replicated study design. Consequently, the anticipated benefits to terrestrial wildlife of leaving no-harvest riparian buffer strips, which are based largely on studies of the use of undisturbed habitat in other regions, are highly speculative. No wildlife studies have gathered concurrent data on both the relative importance of riparian and interior forest habitats and the use of buffer strips, and then considered the resulting information in concert to evaluate riparian management practices from a landscape perspective.

1.3 The Copper Lake Buffer Zone Study

Intensive commercial forest harvesting has been ongoing on insular Newfoundland since the early 1900s, and at present approximately 2.4 million m^3 of timber are harvested annually. Current management objectives set by the Newfoundland Forest Service are to increase this annual yield to 5.0 million m^3 by the year 2035 (Flight and Peters 1992). This is to be achieved through increased forest protection (e.g., fire suppression, insect control), silviculture (precommercial thinning and tree planting), development of technology to utilize timber that previously was considered non-merchantable, and road-building to open lands which currently are inaccessible. The Newfoundland Forest Service has outlined this objective, while simultaneously pointing to a need for greater awareness that forests are a multi-user resource (i.e. not solely for timber production) which must be managed in an ecologically sustainable manner (see Flight and Peters 1992). Clearly there will be conflict in trying to reach these seemingly incompatible goals, thus necessitating the restructuring of forest management practices.

Environmental protection guidelines for riparian habitat recently were adopted for the province. These require that, during clearcutting, 20 m wide buffer strips be left around all water bodies appearing on 1:50,000 scale topographic maps (Scruton et al. 1995). Similar regulations are in place in most other Canadian provinces.

As a result of the general lack of locally relevant information on riparian zone management, stakeholders from the private, public and academic sectors initiated the Copper Lake Buffer Zone Study in 1993. This collaborative research initiative was intended to provide the information necessary to assess the suitability of current riparian management in Newfoundland and Labrador, and suggest improvements where shortcomings were identified. A general description of the rationale, study area, methods and participating agencies for this multi-disciplinary research initiative is presented in Scruton et al. (1995). Various components of the project consider water quality, brook trout, terrestrial mammals, and the research presented in this dissertation, forest bird assemblages.

1.4 Birds as environmental indicators

Studies of avian assemblages are useful for assessment of both the distributional patterns of wildlife within forests and the effects of forest management practices on wildlife communities. Birds, which are the most diverse class of vertebrates in the boreal forest, occupy a broad range of niches and feed at several trophic levels (e.g., insectivores, granivores, frugivores, piscivores, carnivores). They are generally the most detectable vertebrates in woodland ecosystems, and are easily counted with minimal disturbance. Further, the distributional patterns of species occupying different niches are often distinct (Montevecchi 1993), and anthropogenic and natural habitat perturbations often have measurable effects on avian assemblages.

1.5 Study objectives

This component of the Copper Lakes study initially was assigned the broad mandate of assessing the effectiveness of leaving riparian buffer strips as a means of reducing negative impacts of clearcutting on forest birds. Upon consideration of this objective, as well as a review of the literature (available in 1994) and preliminary fieldwork (June - August 1994), it became apparent that two lines of investigation would have to be pursued.

First, an understanding of the distributional relationships of local bird species relative to riparian habitat would be needed. There are a number of reasons for this: (1) species dependent on riparian habitat would have to be identified, as they would be presumably the most vulnerable to poor riparian conservation practices, and a priority for preservation in buffer strips; (2) those species not encountered in riparian habitat may not benefit from shoreline buffers, and therefore should be identified so that appropriate alternative measures can be taken to ensure their well being; (3) knowledge of patterns of habitat selection by each species would be helpful in explaining differences between the assemblages observed in undisturbed riparian habitat and buffer strips; (4) knowledge of the relative importance of riparian zones as wildlife habitat would be useful in prioritizing its protection. This research, which focuses on describing patterns of habitat selection by bird species in the study area, is presented in chapter 3.

The second area of study involved a comparison of the bird assemblages inhabiting buffer strips to those found along undisturbed shorelines (chapter 4). Differences found here, combined with information from chapter 3, would allow the evaluation of the effectiveness of buffer strips for the conservation of woodland birds. It may also be useful in identifying patterns of habitat selection which influence a species' use of buffer strips. Relating differences to patterns of habitat selection should facilitate the extrapolation of results to other regions and, where necessary, point to possible alternative conservation strategies.

Chapter 2. General methods

2.1 Study area

Newfoundland lies at the southeastern limit of the North American boreal forest biome. Climate on the island is strongly influenced by the surrounding ocean, with less extreme temperatures and higher precipitation than adjacent portions of mainland North America (Robertson 1993). Most research in the Copper Lake study has been conducted in the Copper Lake watershed, located near the town of Corner Brook on the west coast of the island. However in order to increase the number of study sites and make findings more representative of the region, sites used in this project were established in several watersheds. These included lower Corner Brook Stream and two of its headwater systems, Copper Lake and Pike's Brook, as well as Cook's Brook and Grindstone Pond (Table 2.1). These watersheds are located throughout the Newfoundland Forest Service's Management Unit 15, which lies within the Corner Brook Subregion of the Western Newfoundland Ecoregion (Fig. 2.1; Damman 1983).

Landscape in the Corner Brook Subregion is characterized by heavily forested, rugged topography with bogs being common on level ground (Damman 1983); elevations of study sites ranged from 250 - 400 m. As a consequence of the topography, stream velocity is generally high, and meandering streams, oxbows and extensive flood plains are uncommon. The absence of prolonged dry periods has excluded forest fire from most

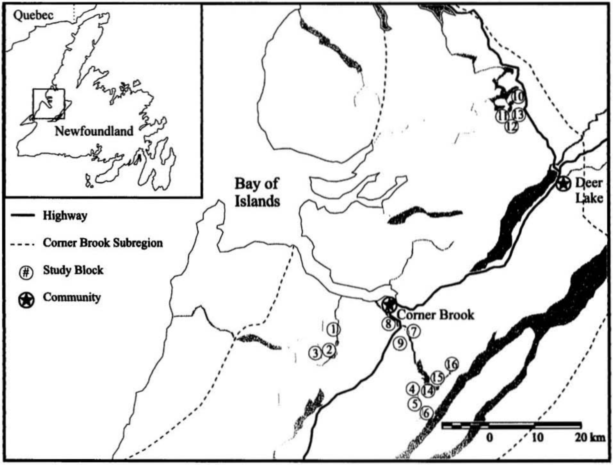

Figure 2.1. Study area in western Newfoundland (inset). Numbers indicate the locations of the 16 study blocks (see Table 2.1).

of the region allowing balsam fir[1] to dominate the forest cover (Meades and Moores 1994), although black spruce, white spruce, white birch, yellow birch, and red maple are present. Natural forest openings resulting from a hemlock looper (*Lambdina fiscellaria fiscellaria*) outbreak (1983-88), and wetlands are abundant throughout the region. Forests in the Corner Brook Subregion are some of the most productive on insular Newfoundland (Meades and Moores 1994), and are extensively clearcut for pulp and paper production, and to a lesser extent for lumber and fuel wood.

2.2 Study design

Preliminary fieldwork was carried out in 1994, with birds being surveyed in a number of habitat types including buffer strip, undisturbed riparian zone, bog, interior forest, non-riparian forest edge and clearcuts with and without a buffer strip. A more focused study design based on observations from the preceding summer was used in the 1995 breeding season. Most of the data collected in 1994 were not used in the analyses presented here, however some were included in the examination of buffer strips (chapter 4). Methods used in the collection of these data were as for 1995 (see below).

In 1995, transects 200 m in length were established in four habitat types (Fig. 2.2): undisturbed riparian controls (RIP), interior forest (INT), non-riparian forest edge (NFE) and riparian buffer strips 20 - 50 m wide (BUF). Efforts were made to place transects

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[1] Where possible, scientific species names are provided in tables 3.1 (trees), 3.2 (shrubs), 3.3 (birds), rather than in the text. Names for plants were taken from Gleason and Cronquist (1991), while the American Ornithologists Union (1983) was used as the authority for bird species names.

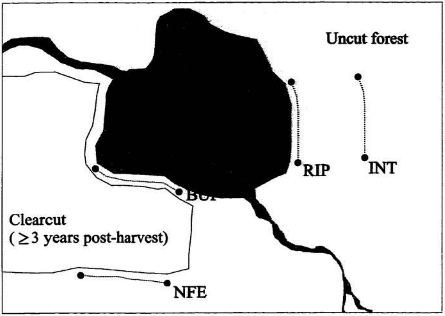

Figure 2.2. General study design. Transects (200 m) were placed in undisturbed riparian (RIP), interior forest (INT), buffer strip (BUF) and non-riparian forest edge (NFE) habitats.

along portions of buffer strips which were relatively constant in width. With the exception of those in the interior forest, all transects followed the shoreline/edge at a distance of about 20 m into the forest. Interior forest transects ran parallel to riparian control shorelines, 150 m away from the shoreline and at least 150 m from other forest edges. Non-riparian edge transects followed edges created by woodland access roads or clearcuts.

The high number of openings in the forest made it difficult to sample interior habitat at distances greater than 150 m from shoreline edges, as has been done in similar studies (e.g., Small and Hunter 1989, Gates and Giffen 1991, Hooper 1991, McGarigal and McComb 1992, LaRue et al. 1995, Murray and Stauffer 1995). This sampling, however, is representative of interior habitat on the scale at which it occurs in the region. Also, because habitat to be protected under riparian forest management legislation in Newfoundland will only extend to 20 m away from the shoreline, the riparian control transect approximates protected riparian habitat, while interior transects represent habitat vulnerable to harvesting.

Individuals of many bird species demonstrate high site fidelity across breeding seasons, and consequently when displaced by clearcutting may move into the adjacent forest (see Darveau et al. 1995). To avoid this short term "packing" of birds around clearcuts (e.g., into buffer strips), all non-riparian edges and buffer strips used had been present for at least 3 years, which should have been sufficient time for populations to stabilize (see Darveau et al. 1995).

Transects were established such that they could be grouped into blocks containing one representing each treatment (Fig. 2.2). Whenever possible the two (paired) controls and the buffer strip transect from a block were placed alongside the same stream or lake. When this was not possible (eight blocks) the buffer transect was located as close as possible to the controls, on a comparable water body within the same watershed. Thirteen complete study blocks were established in 1995, providing a total of 2600 m of transects through each habitat type. Five (1000 m) of the non-riparian edge transects followed forest access roads, while eight (1600 m) were located alongside clearcuts. Roadbeds were separated from the forest edges by deforested (but vegetated) habitat greater than 20 m in width. Five (1000 m) blocks were established along streams while eight (1600 m) were established around lakes. Streams were 4 - 15 m wide and lakes ranged in size from approximately 2 - 200 ha. Relevant features of each study block are summarized in Table 2.1.

The need to keep all transects in each block in relatively close proximity restricted the number of suitable sites in the study area. Thus it was not possible to select sites randomly from a regional "pool". This said, forest cover is relatively homogenous throughout the region, all habitat types sampled (including buffer strips) are locally common, and the sites used appeared to be quite typical. Further, harvesting on all sites was carried out by Corner Brook Pulp and Paper, the only large operator in the area. Consequently it is felt that findings of this project are generalizable throughout the region.

2.3 Habitat sampling

Vegetation sampling plots were established at the 0, 100 and 200 m points on each transect used in 1995. These 200 m^2 plots were placed such that they bordered the shoreline or edge for 10 m and extended 20 m into the forest on the riparian control, buffer strip and non-riparian edge transects. Trees on vegetation plots were tallied by species and diameter at breast height (dbh; 2 cm size classes); these data were then used to calculate basal area (m^2/ha) by species. All shrubs reaching breast height (1.3 m) were tallied by species. On buffer strip transects, the width of the strip was measured at each vegetation sampling plot. These three measurements were averaged to obtain a mean width of the buffer strip.

2.4 Bird survey techniques

Line transect sampling was used to survey birds (see Bibby et al. 1992). In an attempt to restrict sightings to the habitat being sampled counts were truncated such that only birds detected within 30 m of transects were included in data analyses. Surveys were conducted by two experienced observers from 7 June until 7 July of each year, the standardized period for counting breeding birds in the region (Robbins et al. 1986). Three rounds of surveys were completed, with all blocks being visited at the beginning, middle and end of the survey period. Surveys began within 30 min after sunrise (≈ 0500 h) and finished by 0930 h, as breeding birds are most detectable during early hours of daylight, vocalizing less often as the morning progresses (Skirvin 1981). Surveys were not conducted during rain or when winds exceeded 20 km/h, as poor weather reduces the activity and detectability of birds (Robbins 1981). Transects were traveled slowly (25 - 30 min/transect), and all birds heard or seen were recorded. A number of steps were taken to reduce variability between surveys and avoid systematic bias. The two observers were experienced birdwatchers, and practiced together in the study area for one week prior to the initiation of surveys and on afternoons thereafter. Entire blocks were surveyed by a single observer in a morning, and blocks were not sampled by the same person on consecutive visits. Also, transects within each block were visited in a random order on each of the three surveys. It was assumed that birds were equally detectable between stream, lake shore and non-riparian sites; Hooper (1991) found that stream noise did not reduce avian survey efficiency at a distance of 25 m from the shoreline of turbulent streams in Maine.

For each species, the highest count of adults obtained along a transect over the three surveys was assumed to represent the population. This should not lead to an overestimate of abundance as, while individuals may remain undetected, it is unlikely that more birds will be counted than occupy territories within the area (Bibby et al. 1992).

Table 2.1. Summary of study blocks included in data analyses. The three 1994 blocks contained only riparian forest edge and buffer strip transects (see chapter 4). "Year logged" indicates the year when the most intensive harvesting occurred in the area of the block; some activity may have occurred prior to or since that time.

Block	Year sampled	Year logged	Watershed	Lake or Stream (RIP, BUF)[‡]	Clearcut or Road (NFE)[‡]	Buffer width (m)
1	1995	1990	Cook's Brook	Stream	Road	27.0
2	1995	1990	Cook's Brook	Lake	Road	25.3
3	1995	1990	Cook's Brook	Stream	Clearcut	30.3
4	1995	1991	Pike's Brook	Lake	Clearcut	44.7
5	1995	1991	Pike's Brook	Stream	Clearcut	24.7
6	1995	1991	Pike's Brook	Lake	Clearcut	20.0
7	1995	1991	Corner Brook Stream	Stream	Road	22.3
8	1995	1991	Corner Brook Stream	Lake	Road	38.0
9	1995	1991	Corner Brook Stream	Stream	Road	45.7
10	1995	1990	Grindstone Pond	Lake	Clearcut	51.7
11	1995	1990	Grindstone Pond	Lake	Clearcut	32.3
12	1995	1990	Grindstone Pond	Lake	Clearcut	26.3
13	1995	1990	Grindstone Pond	Lake	Clearcut	36.7
14	1994	1990	Corner Brook Stream	Lake	~	45.7
15	1994	1991	Copper Lake	Stream	~	24.7
16	1994	1990	Copper Lake	Stream	~	35.3

[‡] RIP = undisturbed riparian forest edge; BUF = buffer strip; NFE = non-riparian forest edge.

Chapter 3. Distribution of birds relative to riparian habitat

3.1 Introduction

Two properties of riparian zones generally are used to justify their protection during timber harvesting. They are important in maintaining water quality and fish habitat, and are viewed as optimal habitat for a disproportionate number of terrestrial wildlife species (LaRue et al. 1995). Relatively little research has been conducted to assess the ecological importance of riparian zones in mesic landscapes where forests are generally continuous between riparian and upland (i.e. away from shoreline) habitats (Hooper 1989, LaRue et al. 1995, Murray and Stauffer 1995). Further, most research that has been carried out has focused on water quality and fish habitat (Hooper 1989).

The habitat value of riparian zones often has been explained in part through edge effects, whereby the juxtaposition of two habitat types (aquatic and terrestrial in this case) leads to an increase in the richness and abundance of wildlife at the interface (Leopold 1933, Odum 1971, Strelke and Dickson 1980). Most studies of edge effects in forested regions typically have focused on either unnatural edges created by forest clearing (e.g., Strelke and Dickson 1980, Hansson 1983) or natural riparian edges (e.g., Gates and Giffen 1991, LaRue et al. 1995), though Small and Hunter (1989) compared these two types of edge.

In this chapter avian assemblages associated with undisturbed riparian edges, interior forests and unnatural edges created by forest harvesting are compared. In addition to identifying species associated with each habitat type, this allowed assessment of patterns of habitat selection which lead to edge effects, and of the relative importance of riparian habitat to the local avifauna. Further, the inclusion of non-riparian edge transects should allow generalist edge species to be differentiated from riparian species. Knowledge of the wildlife communities associated with these habitat types is essential in evaluating current boreal forest conservation practices, which give priority to protecting riparian habitat and maintaining biodiversity.

3.2 Methods

Comparisons presented here only use data obtained on riparian control, interior forest and non-riparian forest edge transects from the 13 blocks sampled in 1995. Analyses were conducted to test whether habitat structure differed among these three treatments. Tree basal areas (m^2/ha) were compared using analyses of variance, followed by Tukey's test for differences between pairs of treatment means (Day and Quinn 1989, Sokal and Rohlf 1995). Similar analyses were carried out comparing shrub density (number of stems/200 m^2) between treatments, however since these data were non-normally distributed, an equivalent non-parametric test was used (Kruskal-Wallis test; Sokal and Rohlf 1995). A Steel-Dwass test was then used to check for differences between pairs of treatments (Day and Quinn 1989).

Analyses were carried out to test for differences in bird assemblages between riparian transects located alongside streams and those located alongside lakes. No differences were detected in either total avian abundance (individuals/transect) or species richness (species/transect) when compared between shoreline types using a general linear model. Further, comparisons of counts of each bird species also revealed no differences between streams and lakes (Mann-Whitney U-test; Sokal and Rohlf 1995). Consequently, shoreline type was not considered in subsequent analyses of bird counts. A similar series of tests were made comparing non-riparian edge transects located beside clearcuts to those located alongside roads. Again no differences were detected, and consequently non-riparian forest edge type (road or clearcut) was not included in further analyses. Before proceeding, however, it should be noted that due to the small sample sizes used in these tests (13 data points), it is possible that subtle differences may have remained undetected.

To assess the distributional patterns of individual bird species, comparisons were made between pairs of treatments (RIP:INT, INT:NFE, RIP:NFE). Analyses were carried out using generalized linear models which included treatment and block as explanatory variables (McCullagh and Nelder 1989). Because the response variable consisted of counts (individuals/transect), a Poisson error distribution and log-link function were used (McCullagh and Nelder 1989). The link function relates the response variable to the scale of the linear predictor. In generalized linear models the probability value for each explanatory variable is obtained from the reduction in residual deviance which results from its inclusion in the model (McCullagh and Nelder 1989). This follows a Chi-squared distribution (χ^2) and is reported as such. In cases where a total of fewer than six individuals had been recorded on the two treatments being compared, models which provided a good fit to the data (as indicated by analyses of residuals) could not be found. Consequently these data were considered insufficient for statistical analyses.

A similar series of tests was conducted to evaluate the influence of habitat type on both species richness (number of species/transect) and total avian abundance (number of individuals/transect). However, the data could not be fit to a theoretical error distribution. and so for these analyses a nonparametric randomization test was used (Crowley 1992. Adams and Anthony 1996). In this test values of the response variable (counts) were randomly reassigned to the treatment levels without replacement. *F*-statistic values for each explanatory variable were then calculated from the redistributed data. Three thousand iterations of this procedure were completed. The proportion of the 3000 randomized *F*-statistic values equaling or exceeding the *F*-statistic value obtained from the original distribution of the data set was used as the probability estimate for each explanatory variable (i.e. treatment and block).

Based on both the observed distribution of sightings between treatments and published information (Godfrey 1966, Erskine 1977, Degraff et al. 1980, Welsh 1981, Hooper 1991, Parker et al. 1994, Darveau et al. 1995, Murray and Stauffer 1995), each bird species was assigned to one of five habitat association guilds: (1) Forest generalist species, found in forested habitats but showing no clear associations between forest types (i.e. riparian, interior or edge); (2) Interior forest species, found in forested habitats but not along riparian, and in some cases, anthropogenic edges; (3) Riparian species, associated with shorelines and/or riparian vegetation; (4) Open/edge species, associated with non-forested habitats (e.g., clearcuts) or interfaces between forested and non-forested habitats; (5) Ubiquitous species, those showing no clear patterns of association between the habitat types considered in this study. Totals from these guilds were used to subdivide plots of both species richness and total abundance thereby illustrating some differences in the composition of the avian assemblage associated with each habitat type.

3.3 Results

3.3.1 Habitat

Vegetation differed among treatments (Tables 3.1 and 3.2, Figs. 3.1, 3.2, and 3.3). The basal area of black spruce was significantly greater on riparian forest edge transects than on those located in interior forest or non-riparian edge habitats (Table 3.1). A significant effect of treatment on white birch was also detected, where the average basal area in riparian habitat was less than one half of that in either interior forest or non-riparian edge habitat. Standing dead wood was more abundant along interior forest transects than either of the other two treatments. Mean densities of all classes of shrubs were greater along the riparian transects than in the interior forest (Table 3.2); mountain maple was the only species for which this difference was not significant. Mean densities of all shrub classes were also greater non-riparian edge than interior forest transects. however, in this case, the difference for mountain maple was significant, whereas the differences for alder and mountain ash were not. Alder density was greater on riparian transects than on non-riparian edge transects.

3.3.2 Bird assemblages

Distinctive bird species were associated with each of the habitat types sampled. In total, 37 species were identified during the 117 transect surveys (Table 3.3). Species richness and avian abundance were significantly higher along non-riparian edges than riparian edges (Table 3.4; Fig. 3.4). Only one species, northern waterthrush, was significantly more common on riparian transects than either non-riparian edge or interior forest transects. Spotted sandpiper, belted kingfisher, black-and-white warbler, yellow warbler and rusty blackbird were found exclusively along riparian transects, though counts of these species were low. Yellow-bellied flycatcher, black-throated green warbler and ovenbird were significantly more common along interior forest than riparian transects, and intermediate in abundance along non-riparian edges. Red-breasted nuthatch and Swainson's thrush were significantly more common along interior forest transects than either riparian or non-riparian edge transects. Gray jay and dark-eyed junco were more common on non-riparian edges than on riparian edges. White-throated sparrows were more common on non-riparian edges than in the interior forest.

3.4 Discussion

3.4.1 Habitat

Although balsam fir was the dominant tree species on all transects, clear vegetation differences existed between the three treatments. Riparian habitat was characterized by relatively low basal areas of white birch and standing dead wood, a high mean basal area of black spruce and high densities of shrubs, especially alders. In some cases a narrow (< 3 m) ericaceous shrub and/or graminoid (grasses, sedges, rushes) vegetation band separated the woody vegetation from the water's edge (Fig. 3.1). These characteristics are not unexpected, given the increased soil moisture and light availability along shorelines. Although the extent of the riparian vegetation zone was not measured, the transition to upslope/interior forest was generally distinct and occurred within 5 - 50 m of the shoreline (pers. obs.).

Interior forests typically had the highest balsam fir and standing dead wood basal areas, and lowest shrub densities (Fig. 3.2). Others also have found that standing dead wood increased away from shorelines (McGarigal and McComb 1992, Murray and Stauffer 1995). One might expect the distribution of tree basal areas to be similar between interior forest and non-riparian edge transects, as these edges were located in the forest interior prior to the adjacent harvesting. Total basal area was, however, somewhat lower along non-riparian edge transects, largely due to lower conifer basal area. Also, standing dead wood basal area was low along non-riparian edges. It is likely that these reductions reflect disturbance related to harvesting, primarily windthrow, which causes high losses along unnatural forest edges in Newfoundland (Robertson 1993). Well developed shrub communities were characteristic of deforested areas, and often graded a short distance into the forests (Fig. 3.3).

Figure 3.1. Typical riparian habitat in the study area.

Figure 3.2. Interior forest habitat in the study area. Note the open understory and standing dead wood.

Figure 3.3. A non-riparian forest edge in the study area.

3.4.2 Bird assemblages

The bird assemblages observed along riparian edges included several distinguishing species. Northern waterthrush were associated with riparian transects; spotted sandpiper, belted kingfisher, yellow warbler and rusty blackbird were found exclusively (in low numbers) along riparian transects. All are generally associated with riparian habitat in boreal forests (Godfrey 1966, Erskine 1977, LaRue et al. 1995). The few sightings of black-and-white warbler were also limited to riparian transects, although this species is not generally associated with shorelines (Erskine 1977, Murray and Stauffer 1995). In addition to these terrestrial riparian birds, several species of water birds, which typically nest in riparian habitat, were seen in the study area. These species were typically seen on lakes, and included common loon (*Gavia immer*), Canada goose (*Branta canadensis*), American black duck (*Anas rubripes*), ring-necked duck (*Aythya collaris*), greater scaup (*A. marila*), common goldeneye (*Bucephala clangula*) and merganser (*Mergus* sp.). Though beyond the scope of this study, these species are important components of regional riparian bird assemblages.

Several species were more common along interior forest transects than along riparian edges. Within this group two distributional patterns were evident: (1) Red-breasted nuthatch and Swainson's thrush were significantly more common along interior transects than either riparian or non-riparian edge transects, and so are considered true "interior" (i.e. edge avoiding) species in the context of this study. (2) Yellow-bellied flycatcher, black-throated green warbler and ovenbird were also significantly more common along interior than riparian transects, but were intermediate in abundance along non-riparian edges. With the exception of Swainson's thrush, which was associated with riparian habitat in a study in Oregon (McGarigal and McComb 1992), all of these species have been associated with interior forest habitat in other regions (Derleth et al. 1987, Hooper 1991, LaRue et al. 1995, Murray and Stauffer 1995). Hermit thrush, a species often associated with interior forest habitat (Hooper 1991, LaRue et al. 1995), was most frequently observed along interior forest transects, but was uncommon.

In addition to the occurrence of several of these "interior" species, other aspects of the avian assemblages observed along non-riparian forest edges were distinctive. Observations of white-throated sparrow were significantly more frequent here than in the interior forest, and their abundance along non-riparian edges was approximately twice that on riparian edges. Both dark-eyed junco and gray jay were significantly more common along non-riparian forest edges than along riparian edges, and rare in interior forests. Other species associated with early successional openings and edges were most abundant along non-riparian forest edges (e.g., magnolia warbler, mourning warbler, Lincoln's sparrow).

Riparian habitat supports the most species-rich and dense bird assemblages in arid regions (e.g., Hubbard 1977, Johnson and Haight 1985, Szaro and Jakle 1985, Knopf 1986) and in mesic broadleaf forests in eastern North America (Hair et al. 1978, Hooper 1991; see also Gates and Giffen 1991). Zones of riparian vegetation are typically narrow in boreal forests, and the present study found neither avian species richness nor total abundance to differ between riparian and interior forest habitats. These findings are, however, consistent with those of most other studies conducted in coniferous and mixed coniferous-deciduous forests (Knopf 1985, Small and Hunter 1989, Hooper 1991, McGarigal and McComb 1992, Murray and Stauffer 1995, Haché 1996).

Unlike findings from other coniferous forests, LaRue et al. (1995) found species richness and abundance to be greatest in riparian portions of balsam fir-white cedar stands in Quebec. However, they categorized birds occurring within 200 m of the shoreline as being associated with riparian habitat (LaRue et al. 1995). Other studies (Hooper 1991, Murray and Stauffer 1995, Haché 1996), including the present one, have found that the shift from riparian to interior forest bird assemblages occurred well within 200 m of the shoreline. It seems likely that the lower resolution "riparian" plots of LaRue et al. (1995), through the inclusion of both riparian and interior species, overestimated the species richness of riparian assemblages. Indeed, all bird species identified here as being associated with interior forest habitat were common on both riparian and non-riparian forest plots sampled by LaRue et al. (1995).

3.4.3 Edge effects

Bird species richness and total abundance along anthropogenic forest edges (NFEs) were significantly greater than along riparian forest edges (Table 3.4, Fig. 3.4). Two factors seem to be responsible for this: (1) Many "interior" species, which were rare along riparian edges, were common along non-riparian edges (e.g., yellow-bellied flycatcher, hermit thrush, black-throated green warbler, ovenbird). The mechanism

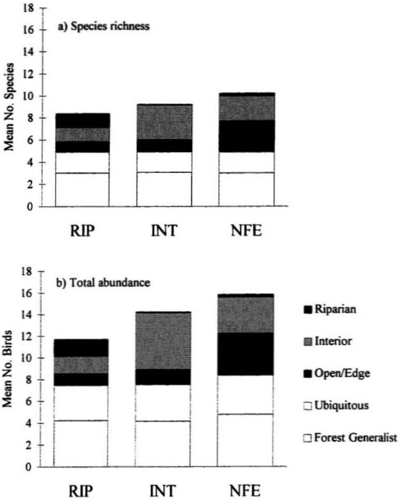

Figure 3.4. Mean bird species richness (a) and mean abundance of birds (b) on undisturbed riparian edge (RIP), interior forest (INT) and non-riparian forest edge (NFE) transects. Subdivisions of each column indicate the mean for each of five habitat guilds.

responsible for this difference was not tested but is likely linked to structural and vegetational differences between the two edge types. For example, the zone of increased black spruce which was typical of riparian edges was not present along non-riparian edges, and may act as a "barrier" to the movements of species associated with interior forest vegetation types. Black-throated green warbler, ovenbird and red-breasted nuthatch are characteristically associated with fir, but not spruce stands (Erskine 1977). (2) Avian assemblages along non-riparian edges include more open, edge and mixed-habitat species than those along shorelines (Small and Hunter 1989, Gates and Giffen 1991). Present findings support this pattern, with the highest counts of these species (gray jay, magnolia warbler, mourning warbler, dark-eyed junco, white-throated sparrow, Lincoln's sparrow) occurring along non-riparian edges (Fig. 3.4). In contrast, aquatic habitat adjacent to riparian edges contributed relatively little to the terrestrial riparian bird assemblage with the exception of aquatic foragers (belted kingfisher, spotted sandpiper, northern waterthrush). Indeed most riparian species were relatively rare and may restrict themselves to the narrow riparian vegetation zone (see chapter 4; Manuwal 1986).

The inclusion of non-riparian edge transects in the study should have allowed generalist edge species to be distinguished from riparian edge species; however, no species appeared to select both edge types over interior forest habitat. Consequently, none of the species selecting riparian edges over interior forest was provided with alternate habitat along edges created by forest clearing (see also Small and Hunter 1989). Distinct bird assemblages are associated with intrinsic riparian edges and anthropogenic non-riparian forest edges.

Table 3.1. Summary of ANOVAs comparing tree basal area between habitat types. Significant probability-values (i.e. $P <$ 0.05) are printed in boldface type. Pairs of means were compared using Tukey's test; values followed by the same letter or no letter are not statistically different. Eastern larch (*Larix laricina*), yellow birch (*Betula lutea*) and pin cherry (*Prunus pensylvanica*) were also occasionally found on vegetation plots and are included in appropriate totals.

	Mean basal area (m^2/ha) ± SE			ANOVA Summary		
	RIP[‡]	INT	NFE	MS	$F_{2,114}$	P
Coniferous species						
Balsam fir (*Abies balsamea*)	21.7 ± 1.7	26.6 ± 2.1	22.6 ± 1.8	256.75	1.87	0.159
Black spruce (*Picea mariana*)	7.7 ± 1.0^{a}	3.6 ± 1.0^{b}	2.4 ± 0.7^{b}	296.00	9.16	**< 0.001**
White spruce (*P. glauca*)	1.3 ± 0.4	1.3 ± 0.4	1.7 ± 0.4	2.13	0.34	0.715
Subtotal	30.7 ± 1.5	31.6 ± 1.8	26.7 ± 1.9	258.00	2.19	0.116
Deciduous species						
White birch (*Betula papyrifera*)	1.6 ± 0.3^{a}	3.4 ± 0.7^{a}	3.5 ± 0.7^{a}	42.03	3.14	**0.047**
Red maple (*Acer rubrum*)	0.4 ± 0.2	0.2 ± 0.1	0.2 ± 0.1	0.62	0.49	0.612
Subtotal	2.2 ± 0.4	3.9 ± 0.8	3.7 ± 0.7	33.00	2.07	0.132
Total	32.9 ± 1.6	35.4 ± 1.7	30.4 ± 1.9	241.50	2.10	0.127
Standing dead wood	9.0 ± 1.1^{a}	14.3 ± 1.3^{b}	7.8 ± 1.0^{a}	461.25	9.24	**< 0.001**

[‡] RIP, INT and NFE are riparian forest edge, interior forest and non-riparian forest edge habitat, respectively.

Table 3.2. Comparison of shrub densities across three habitat types. The effect of treatment was assessed using a Kruskal-Wallis test, and the corresponding probability is reported (*P*); significant values (i.e. $P < 0.05$) are printed in boldface type. Analyses for differences between pairs of treatments were made using a Steel-Dwass test; values followed by the same letter are not significantly different.

	Mean density (stems/200 m^2) ± 95% C.I.			
	RIP	INT	NFE	*P*
Alder[†] (*Alnus* spp.)	30.2 ± 14.3^{a}	0.7 ± 1.3^{b}	1.9 ± 2.8^{b}	**< 0.001**
Mountain maple (*Acer spicatum*)	13.2 ± 8.0^{a,b}	6.7 ± 5.6^{a}	29.0 ± 11.8^{b}	**0.007**
Mountain ash (*Sorbus americana*)	6.5 ± 2.7^{a}	1.1 ± 0.8^{b}	4.1 ± 2.1^{a,b}	**0.001**
Other shrubs[‡]	16.0 ± 6.6^{a}	1.4 ± 1.3^{b}	13.4 ± 7.0^{a}	**< 0.001**
Total	65.8 ± 16.7^{a}	9.9 ± 6.6^{b}	48.3 ± 16.5^{a}	**< 0.001**

[†] Mountain alder (*Alnus crispa*) and speckled alder (*A. rugosa*).

[‡] Common species included beaked hazelnut (*Corylus cornuta*), Canadian yew (*Taxus canadensis*), chuckley pear (*Amelanchier* spp.), red elderberry (*Sambucus pubens*), red-osier dogwood (*Cornus stolonifera*), wild rasin (*Viburnum cassinoides*) and squashberry (*V. edule*).

Table 3.3. Mean frequency of bird observations along 13 transects in each of three habitat types. Pairs of treatments were compared using a generalized linear model with Poisson error distribution (null model = 25 d.f.). Significant probability-values (i.e. $P < 0.05$) are printed in boldface type. Based on these observations, as well as published information (see text) species were separated into five habitat selection guilds, and are grouped as such below.

Habitat selection guild	Total count	Mean frequency (individuals/200 m transect)			Pairwise comparisons P (χ^2, 1 d.f.)‡		
Species†		RIP	INT	NFE	RIP:INT	INT:NFE	RIP:NFE
Forest Generalist							
Ruffed grouse (*Bonasa umbellus*)	2	0	0	0.15	~	~	~
Black-backed woodpecker (*Picoides arcticus*)	4	0.08	0.08	0.15	~	~	~
Downy woodpecker (*P. pubescens*)	7	0.23	0.23	0.08	0.978 (<0.01)	~	~
Hairy woodpecker (*P. villosus*)	4	0.08	0.08	0.15	~	~	~
Black-capped chickadee (*Parus atricapillus*)	12	0.31	0.23	0.39	0.701 (0.15)	0.476 (0.51)	0.737 (0.11)
Boreal chickadee (*P. hudsonicus*)	26	0.69	0.69	0.62	0.963 (<0.01)	0.798 (0.07)	0.795 (0.07)
Winter wren (*Troglodytes troglodytes*)	9	0.23	0.15	0.31	~	0.409 (0.68)	0.703 (0.15)
Golden-crowned kinglet (*Regulus satrapa*)	8	0.23	0.15	0.23	~	~	0.967 (<0.01)
Ruby-crowned kinglet (*R. calendula*)	58	1.23	1.69	1.54	0.329 (0.95)	0.758 (0.10)	0.504 (0.45)
Black-and-white warbler (*Mniotilta varia*)	2	0.15	0	0	~	~	~
Pine grosbeak (*Pinicola enucleator*)	4	0.15	0.15	0	~	~	~
Pine siskin (*Carduelis pinus*)	33	0.85	0.62	1.08	0.489 (0.48)	0.197 (1.66)	0.542 (0.37)
Purple finch (*Carpodacus purpureus*)	2	0	0.08	0.08	~	~	~
Subtotal	171	4.23	4.15	4.78			
Interior							
Yellow-bellied flycatcher (*Empidonax flaviventris*)	43	0.54	1.54	1.23	**0.011** (6.52)	0.504 (0.44)	0.057 (3.62)
Red-breasted nuthatch (*Sitta canadensis*)	8	0.08	0.46	0.08	**0.047** (3.96)	**0.047** (3.96)	~
Hermit thrush (*Catharus guttatus*)	9	0.08	0.39	0.23	0.088 (2.91)	0.477 (0.51)	~
Swainson's thrush (*C. ustulatus*)	11	0.08	0.62	0.15	**0.013** (6.20)	**0.049** (3.86)	~
Black-throated green warbler (*Dendroica virens*)	49	0.77	1.62	1.39	**0.046** (4.00)	0.631 (0.231)	0.128(2.32)
Ovenbird (*Seiurus aurocapillus*)	13	0.08	0.62	0.31	**0.013** (6.20)	0.244 (1.36)	0.165 (1.93)
Subtotal	133	1.63	5.25	3.39			

Riparian							
Spotted sandpiper (*Actitis macularia*)	4	0.31	0	0	~	~	~
Belted kingfisher (*Ceryle alcyon*)	1	0.08	0	0	~	~	~
Northern waterthrush (*Seiurus noveboracensis*)	15	0.85	0.08	0.23	**0.002** (9.75)	0.306 (1.04)	**0.027** (4.86)
Yellow warbler (*Dendroica petechia*)	2	0.15	0	0	~	~	~
Rusty blackbird (*Euphagus carolinus*)	2	0.15	0	0	~	~	~
Subtotal	24	1.54	0.08	0.23			
Open/Edge							
Gray jay (*Perisoreus canadensis*)	10	0	0.23	0.54	~	0.200 (1.65)	**0.002** (9.70)
Magnolia warbler (*Dendroica magnolia*)	10	0.15	0.15	0.46	~	0.148 (2.10)	0.148 (2.10)
Mourning warbler (*Oporornis philadelphia*)	13	0.15	0.31	0.54	0.409 (0.68)	0.362 (0.83)	0.086 (2.95)
Dark-eyed junco (*Junco hyemalis*)	10	0	0.23	0.54	~	0.200 (1.65)	**0.002** (9.71)
White-throated sparrow (*Zonotrichia albicollis*)	35	0.77	0.46	1.46	0.315 (1.01)	**0.008** (7.10)	0.092 (2.84)
Lincoln's sparrow (*Melospiza lincolnii*)	4	0	0	0.31	~	~	~
Subtotal	82	1.07	1.38	3.85			
Ubiquitous							
American robin (*Turdus migratorius*)	43	1.08	0.92	1.31	0.693 (0.16)	0.351 (0.87)	0.588 (0.29)
Blackpoll warbler (*Dendroica striata*)	9	0.23	0.23	0.23	0.977 (<0.01)	0.979 (<0.01)	0.977 (<0.01)
Yellow-rumped warbler (*D. coronata*)	78	1.92	2.15	1.92	0.680 (0.17)	0.680 (0.17)	1.00 (<0.01)
Fox sparrow (*Passerella iliaca*)	3	0	0.08	0.15	~	~	~
Subtotal	133	3.23	3.38	3.61			

[†] Common flicker (*Colaptes auratus*), olive-sided flycatcher (*Contopus borealis*) and Wilson's warbler (*Wilsonia pusilla*) were seen only along buffer strip transects (see table 4.3).

[‡] In generalized linear models, the probability value for an explanatory variable is calculated from the change in deviance resulting from its inclusion in the model, which can be approximated to a Chi-squared (χ^2) distribution.

~ Insufficient observations for statistical comparison (n < 6 individuals).

Table 3.4. Comparisons of avian assemblage parameters between habitat types. Figures for total abundance indicate the mean number of individual birds observed per transect (all species combined), while species richness indicates the mean number of species per transect. Pairs of treatments were compared using randomization tests (3000 iterations).

	Mean (n/200 m transect) ± S.E.			*P*		
	RIP	INT	NFE	RIP:INT	INT:NFE	RIP:NFE
Total abundance	11.6 ± 0.9	14.3 ± 1.1	15.9 ± 1.2	0.059	0.272	**0.004**
Species richness	8.4 ± 0.7	9.2 ± 0.7	10.2 ± 0.8	0.310	0.310	**0.037**

Chapter 4. Bird assemblages inhabiting riparian buffer strips

4.1 Introduction

Riparian buffer strip reserves are typically promoted as a means of minimizing the impacts of logging on terrestrial and aquatic wildlife, but specific conservation objectives are generally poorly defined. Presumably one of the primary goals is the protection of species preferring or dependent on riparian habitat. Riparian and interior forest bird assemblages can be distinguished in coniferous and mixedwood forests, each typically containing characteristic species (Chapter 3; see also Knopf 1985, Small and Hunter 1989, McGarigal and McComb 1992, Murray and Stauffer 1995, Haché 1996). However, managers generally assume that most species prefer or frequent riparian habitats, and will consequently use buffer strips (see Hooper 1989). These assumptions remain largely untested in boreal forest ecosystems.

The widespread acceptance of the notion that riparian habitats are preferred by most species is evident in the few studies designed to evaluate the conservation potential of buffer strips for forest birds (see Johnson and Brown 1990, Triquet et al. 1990, Darveau et al. 1994, Darveau et al. 1995). These studies compared abundances of species inhabiting undisturbed riparian forests to those observed in buffer strips, with little or no consideration being given to the relative use of riparian and interior forest habitats by each species. Consequently, these studies did not provide information on the use of buffer strips by either riparian or non-riparian (e.g., interior forest) species. Further, the authors often indicated a need to identify the minimum width of buffer strip necessary to maintain a species assemblage similar to that found on an undisturbed shoreline (Johnson and Brown, 1990, Darveau et al. 1994; see also Spackman and Hughes 1995). It is hopeful at best to suggest that even wide buffer strips (i.e. 50 - 100 m in the boreal forest) could support an unaltered and complete forest bird assemblage. A more productive approach would be to identify patterns of habitat selection by the species involved, and then use this information to explain differences in bird assemblages found along undisturbed and buffered shorelines. Appropriate conservation strategies, which likely go beyond simply altering the width of buffer strips, can then be developed for those species which are not benefiting.

The objective of research presented in this chapter was to evaluate the use of buffer strips by terrestrial birds in a boreal forest ecosystem. However, beyond simply reporting observed differences between buffer strips and undisturbed shorelines, information on the distribution of each species (as described in chapter 3) was used to explain changes. Vegetation also was compared between control and buffered shorelines, as this has been shown to change rapidly after clearcutting, and may affect the use of these strips by birds (Darveau et al. 1994). This approach should point to the mechanisms leading to many of the observed differences in bird assemblages, thus allowing changes to be better understood and, where necessary, solutions proposed.

4.2 Methods

Data analyses presented here provide a comparison of observations along undisturbed riparian edge (RIP) and buffer strip transects (BUF). Bird surveys from three pairs of undisturbed shoreline and buffer strip transects sampled in 1994 were found to meet the criteria of the 1995 study design (i.e. 200 m transects placed approximately 20 m from the shoreline, similar sampling methodology and intensity etc.), and were pooled with these data to increase sample size. These three pairs of transects were not included in the 13 blocks sampled in 1995. Thus, the data set included bird surveys from a total of 16 transects (3200 m) along each type of shoreline. Nine pairs of transects (1800 m) were located along lake shores, while seven pairs (1400 m) followed streams (see Table 2.1).

Analyses were carried out to assess whether vegetation differed between control and buffer strip shorelines (1995 transects only). Tests for differences in tree basal areas were carried out using a paired-comparison analysis of variance (Sokal and Rohlf 1995). Wilcoxon's signed ranks test was used to compare densities of shrubs between control and buffer strip shorelines (Sokal and Rohlf 1995).

No differences in bird assemblages were found between riparian controls placed alongside rivers and lakes (Chapter 3). Similar tests were carried out comparing the frequency of bird observations between riverine and lacustrine buffer strips using a general linear model. Explanatory variables included the width of the buffer strips (as a covariate), the type of water body, and the interaction between the two. Again, there was no effect of water body type, and this was not included as a variable in further analyses.

Due to the small number of 1994 blocks, we were unable to test for any differences between years, however all trends observed on the 1995 blocks also were evident on 1994 blocks, and 1994 counts fell within the range observed in 1995. Consequently, study blocks from both years were pooled in further data analyses.

Comparisons of bird observations between undisturbed and buffered shorelines were carried out at three levels. Initially two community level parameters, species richness (i.e. number of species observed per transect) and total abundance of birds, were compared. Following this, comparisons were made at the species and habitat guild levels (following the classification of chapter 3).

As in chapter 3, bird counts were compared between control and buffer strip transects using a generalized linear model with a Poisson error distribution and log-link function (McCullagh and Nelder 1989). These models were acceptable at the species and guild levels. However, as the mean value of count data increases, its error distribution approaches normality. Consequently, total relative abundance (i.e. all bird observations combined) and species richness (number of species) were better modeled using a paired-comparison analysis of variance (Sokal and Rohlf 1995). All comparisons of bird observations between control shorelines and buffer strips were carried out twice. The first analyses included all observations obtained within 30 m of the transects. However the clearcuts adjacent to buffer strips often extended within this sampling area, likely leading to underestimation of the density of forest-dwelling species within buffer strips. Hence, the original analyses were repeated after excluding observations obtained in clearcuts and an equivalent portion of the paired riparian controls (i.e. considering only forested habitat). Both analyses are reported.

In order test the influence of buffer strip width on bird numbers, counts obtained within each buffer strip were regressed against its width. This was done for each of the five guilds and for all species combined.

4.3 Results

Habitat structure differed between control and buffer strip shorelines. Total tree basal area was significantly lower in the buffer strips than along the control shorelines (Table 4.1). This difference was reflected in lower basal areas of both conifers and hardwoods, as well as some individual species. Shrub densities were variable, and no significant differences were observed (Table 4.2).

Avian assemblages differed between the control and buffer strip shorelines. The total number of birds observed in buffer strips was significantly higher than in the riparian controls, while the difference in mean species richness between these two approached significance (Table 4.3). These differences arose largely due to significant increases in the open/edge and ubiquitous habitat selection guilds, while counts for other guilds remained relatively similar between buffer strip and control shorelines (Table 4.4). Several species from the open/edge and ubiquitous guilds were significantly more abundant along buffer strip transects, while no species from any guild was significantly less abundant in buffer strips.

Comparisons of analyses including all birds seen within 30 m of the transects to analyses of those seen within the buffer strip alone reflected differences in patterns of habitat selection between guilds (Table 4.4). As expected, the difference between buffers and controls was more pronounced for the open/edge guild when clearcut habitat was included in the analysis. Two species from this guild, magnolia warbler and white-throated sparrow, were significantly more frequently observed along the buffer strip transects only when clearcut habitat was included. The opposite was true for the ubiquitous guild, where the frequency of observation was significantly higher for two species (blackpoll warbler and yellow-rumped warbler) and the guild as a whole only when counts were restricted to forested habitat. The initial comparison (including clearcut habitat) of forest generalists between treatments seemed to indicate a possible reduction in numbers along buffer strips, however after the exclusion of deforested habitat there was no evidence of such a trend.

Predictably, regression of the total number of birds observed within buffer strips on buffer width revealed a significant positive relationship (Table 4.5). However at the guild level this trend approached significance only for the interior forest and ubiquitous guilds (Table 4.5). Counts of forest generalists, riparian and open/edge guilds showed no evidence of being influenced by buffer width.

Figure 4.1. A typical buffer strip in the study area. The paired control shoreline was located on the far shore. Note the windthrown trees along the edge of the clearcut.

4.4 Discussion

4.4.1 Habitat

While it was not measured directly, the most likely explanation for the reductions in tree basal area observed within buffer strips is through windthrow. Large numbers of blown down trees were observed in many buffers (see Fig 4.1), and comparable reductions in basal area were observed in anthropogenic non-riparian edges (Chapter 3). Similar losses of trees due to windthrow were observed in balsam fir stands in Quebec, leading to an annual decline in populations of the forest dwelling birds inhabiting 20 - 60 m wide buffer strips (Darveau et al. 1994). The authors concluded that in regions where hilly relief induces wind corridors and tree species are susceptible to windthrow (e.g., balsam fir), narrow riparian buffers may have only short-term value as habitat for breeding birds (Darveau et al. 1994). This may be the case in Newfoundland, where windthrow causes annual losses of 10 - 15 % of wood volume in buffer strips (Robertson 1993).

4.4.2 Bird assemblages

Presumably riparian forest management policies are most directly aimed at conserving species dependent on riparian habitat. Consequently, the response of these species is of primary concern in evaluating the effectiveness of buffer strips. At the guild level, counts of riparian species were similar between control shorelines and buffer strips (Table 4.4). Further, regression analyses showed no association between counts of riparian species and buffer strip width (Table 4.5). This likely stems from the association of these species with habitat provided along the shoreline edge of the buffer strip, which would not increase in proportion to its width. Thus, it seems unlikely that increasing the width of buffer strips would increase their use by riparian birds. In contrast to findings in Maine (Johnson and Brown 1990), northern waterthrush was not less abundant along buffers than undisturbed shorelines. However, the study conducted in Maine was unreplicated, making it difficult to separate the effect of harvesting from intrinsic variability between two lake shores. The five remaining riparian species were all uncommon, and no differences between control shorelines and buffer strips were detected for these (Table 4.4).

The general rarity of riparian birds in riparian habitat was unexpected. A likely explanation is that these species occupy narrow, linear shoreline territories. Manuwal (1986) described two distinctive territory shapes along streams in Montana. Some species typically occupied symmetric territories spanning riparian and upland vegetation, while others had elongated territories which fell almost entirely within riparian vegetation. While Manuwal (1986) did not assess the dependence of each species on riparian habitat, it is likely that those with linear shoreline territories would not be found elsewhere in the landscape. Such a pattern of habitat selection by riparian species would result in low encounter rates by observers using linear or point sampling techniques, as each territory would occupy a long segment of shoreline. Studies of riparian species may thus require more extensive sampling than is typically necessary in studies of species selecting other (i.e. nonlinear) habitat types. Also, comparisons of densities between riparian (and those associated with other edge types) and interior species should be made with caution. Studies incorporating territory mapping or telemetry (see Bibby et al. 1992) would be useful in identifying patterns of riparian habitat use.

Darveau et al. (1995) suggested that, since densities of forest birds observed within 80 m of shorelines having narrow buffers (20 - 40 m) were lower than on forested controls, while numbers of ubiquitous birds increased, narrow buffer strips were more favorable for ubiquitous species. Woodland species would not, however, be expected to inhabit deforested areas, and consequently analyses including sampling in clearcuts are likely biased. Analyses considering buffer area only indicated that in Newfoundland, as in Quebec (Darveau et al. 1995), densities of forest generalists remained relatively unchanged within narrow buffer strips. Thus, we suggest that in the boreal forest, riparian buffers of any width greater than 20 m are likely beneficial to forest generalists. In both this study and Darveau et al. (1995) many woodland species were found in areas that would have been unsuitable without buffer strips (see Triquet et al. 1990; Whitaker, unpubl. data). The lack of association between buffer strip width and counts of forest generalists is unexpected, given that the habitat area available to these species (as we have defined it) should increase with strip width. It is possible that the range of buffer widths sampled was not great enough to detect a response.

As with forest generalists, no difference in abundance was detected for the interior forest guild between control shorelines and buffer strips (Table 4.4). Consideration of data from shoreline habitats alone might lead to the conclusion that these species are adequately protected in buffers. However, the rarity of these species on control plots resulted from their general absence in riparian habitat, not overall rarity in the region. The mean frequency of sightings along interior forest transects (i.e. 150 m from the shoreline) was greater than three times that observed along either riparian controls or buffer strips (Chapter 3). Indeed only three of six interior forest species were observed in buffer strips. Thus both undisturbed shorelines and buffer strips may be poor or marginal habitat for these species.

Regression analyses suggested a possible positive relationship between counts of interior forest birds and buffer strip width (Table 4.5). It is likely that, were they wide enough, buffers may be able to provide adequate interior forest conditions to support some species selecting this type of habitat. However, even the widest buffers sampled (40 - 50 m) supported densities of less than 50% of that observed in interior forest habitats. In the boreal forest, where surface water is abundant, the economic impacts of setting aside riparian buffers large enough to support populations of interior forest species would be high. In addition, core interior forest (> 100 m from any edge; Temple 1986) can only be preserved by setting aside large, relatively symmetric reserves, not extensive linear buffers. Consequently, it is clear that separate conservation strategies are required to maintain populations of riparian and interior forest species in managed woodlands. There is reason to give detailed consideration to the conservation of interior forest birds. Populations of many of these species have declined throughout northeastern North

America in recent years (Robbins et al. 1989b, Sauer and Droege 1992), a trend which has been linked to anthropogenic forest fragmentation (Askins et al. 1990).

The most pronounced changes in riparian assemblages were observed in the open/edge and ubiquitous guilds, both of which were significantly more common along buffer strips than along undisturbed shorelines (Table 4.4). Similar increases in many of these species were observed in buffer strips in Maine and Quebec (Johnson and Brown 1990, Darveau et al. 1995). The change in open/edge species is not unexpected given that their preferred habitats have been created in an area that previously supported continuous forest cover. Correspondingly, the difference between controls and buffers was more pronounced with the inclusion of clearcut habitat. Counts of open/edge species showed no response to increasing buffer strip width. As with the riparian guild, this likely stems from their association with habitat provided along the edge of the buffer strips, which would not increase in proportion to buffer width. The mixed habitat provided by buffer strips appears to be favorable to species classified here as being ubiquitous. The difference for the ubiquitous guild, and two of its species (blackpoll and yellow-rumped warblers) was significant only after the exclusion of clearcut habitat, thus suggesting greater use of habitat within the buffer strip. Counts of ubiquitous species did show a positive relationship with strip width. Given that ubiquitous species were less abundant along control shorelines, where forests extended greater than 300 m upslope, there is presumably an optimum buffer width beyond which densities of these species would decline.

The higher total abundance and species richness of the bird assemblage observed in buffer strips is not surprising given that two guilds became more abundant, while no guild, or even individual species, was significantly less abundant in this habitat. These increases could be explained through packing of birds which originally inhabited the (now harvested) forest into the residual buffer strip (see Darveau et al. 1995). This explanation is, however, unlikely for two reasons. First, the time since harvesting (≥ 3 years) on buffer strip plots should have been great enough for densities of breeding birds to stabilize. Darveau et al. (1995) found that populations of boreal forest birds in buffer strips (20, 40 and 60 m wide) returned to approximately preharvest levels within three years. Second, a large portion of the observed increase can be attributed to species in the open/edge guild, which are associated with clearcut habitat that has been added to the area, not the interior forest which was removed. Non-riparian forest edges also were distinguished from undisturbed riparian control shorelines through higher species richness and bird abundance, resulting largely from high counts of open ground and edge associated birds (Chapter 3). Thus the juxtaposition of natural riparian and unnatural non-riparian forest edges along the length of narrow buffer strips resulted in the development of bird assemblages containing the distinguishing species of both edge types. In addition, ubiquitous species became more abundant than they were along either type of edge alone.

Table 4.1. Results of ANOVAs comparing tree basal area between undisturbed riparian forest edges (RIP) and riparian buffer strips (BUF). Three 200 m^2 plots were sampled along thirteen transects in each shoreline type (null d.f. = 77). Significant probability values (i.e., $P < 0.05$) are printed in boldface type.

	Mean basal area (m^2/ha) ± SE		ANOVA summary		
	RIP	BUF	MS	$F_{1,64}$	P
Coniferous species[‡]					
Balsam fir (*Abies balsamea*)	21.7 ± 1.7	16.3 ± 1.6	566.68	6.50	**0.013**
Black spruce (*Picea mariana*)	7.7 ± 1.0	8.1 ± 1.3	3.20	0.09	0.768
White spruce (*Picea glauca*)	1.3 ± 0.4	0.8 ± 0.3	2.79	0.92	0.342
Subtotal	30.7 ± 1.5	25.9 ± 1.4	456.4	5.41	**0.023**
Deciduous species[‡]					
White birch (*Betula papyrifera*)	1.6 ± 0.3	1.0 ± 0.3	8.21	4.69	**0.034**
Red maple (*Acer rubrum*)	0.4 ± 0.2	< 0.1	3.06	2.84	0.097
Total hardwoods	2.2 ± 0.4	1.0 ± 0.3	27.29	9.27	**0.003**
Total[‡]	32.9 ± 1.6	26.8 ± 1.4	718.85	8.38	**0.005**
Standing dead wood	9.0 ± 1.1	8.9 ± 1.0	0.15	0.00	0.994

[‡] Eastern larch (*Larix laricina*), pin cherry (*Prunus pensylvanica*) and yellow birch (*Betula lutea*) were also present (but rare) and are included in the appropriate totals.

Table 4.2. Shrub densities in undisturbed riparian forest edges (RIP) and riparian buffer strips (BUF). Treatments were compared using Wilcoxon's signed ranks test.

	Mean density (stems/200 m^2) ± 95% C.I.		
	RIP	BUF	*P*
Alder (*Alnus* spp.) [†]	30.2 ± 14.3	21.9 ± 10.9	0.189
Mountain maple (*Acer spicatum*)	13.2 ± 8.0	5.3 ± 3.4	0.086
Mountain ash (*Sorbus americana*)	6.5 ± 2.7	9.2 ± 4.3	0.492
Other shrubs[‡]	16.0 ± 6.6	25.5 ± 8.7	0.131
Total	65.8 ± 16.7	61.7 ± 16.9	0.503

[†] Mountain alder (*Alnus crispa*) and speckled alder (*A. rugosa*).

[‡] Common species included beaked hazelnut (*Corylus cornuta*), Canadian yew (*Taxus canadensis*), chuckley pear (*Amelanchier* spp.), red elderberry (*Sambucus pubens*), red-osier dogwood (*Cornus stolonifera*), wild rasin (*Viburnum cassinoides*) and squashberry (*V. edule*).

Table 4.3. Comparison of avian assemblage parameters between undisturbed riparian controls (RIP) and buffer strips (BUF). Comparisons were carried out using a randomized complete blocks ANOVA. Two analyses were conducted: (1) All individuals within 30 m. (2) Individuals detected in the buffer area only. Significant probability values (i.e., $P < 0.05$) are printed in boldface type.

	Mean frequency/transect ± S.E.		ANOVA summary		
	RIP	BUF	MS	$F_{1,15}$	P
All sightings					
Total relative abundance	10.7 ± 0.9	13.2 ± 1.1	50.00	8.33	**0.011**
Species richness	8.1 ± 0.6	8.6 ± 0.5	2.00	1.07	0.317
Buffer area only					
Total relative abundance	7.8 ± 0.9	10.5 ± 1.2	60.50	16.35	**0.001**
Species richness	6.2 ± 0.7	7.2 ± 0.5	8.00	4.44	0.052

Table 4.4. Mean frequency of bird observations on transects following undisturbed riparian edges (RIP) and riparian buffer strips (BUF). Treatments were compared using a generalized linear model with Poisson error distribution. Significant probability values (i.e., $P < 0.05$) are printed in boldface type. Two analyses were conducted: (1) All individuals within 30 m. (2) Individuals detected in the buffer area only.

	All sightings				Buffer area only			
		Mean frequency/transect				Mean frequency/transect		
Habitat selection guild Species‡	n_{birds}	RIP	BUF	P (χ^2; 1 df)	n_{birds}	RIP	BUF	P (χ^2; 1 df)
Forest generalist								
Black-backed woodpecker (*Picoides arcticus*)	2	0.06	0.06	~	2	0.06	0.06	~
Downy woodpecker (*P. pubescens*)	5	0.19	0.13	~	3	0.06	0.13	~
Hairy woodpecker (*P. villosus*)	2	0.13	0	~	2	0.13	-	~
Black-capped chickadee (*Parus atricapillus*)	7	0.25	0.19	0.703 (0.15)	3	0.13	0.06	~
Boreal chickadee (*P. hudsonicus*)	23	0.56	0.88	0.294 (1.10)	19	0.44	0.75	0.247 (1.34)
Winter wren (*Troglodytes troglodytes*)	5	0.19	0.13	~	2	0.06	0.06	~
Golden-crowned kinglet (*Regulus satrapa*)	3	0.19	0	~	2	0.13	-	~
Ruby-crowned kinglet (*R. calendula*)	31	1.06	0.88	0.584 (0.30)	23	0.63	0.81	0.528 (0.40)
Black-and-white warbler (*Mniotilta varia*)	2	0.13	0	~	1	0.06	-	~
Pine grosbeak (*Pinicola enucleator*)	2	0.13	0	~	2	0.13	-	~
Pine siskin (*Carduelis pinus*)	22	0.94	0.44	0.084 (2.98)	18	0.75	0.38	0.152 (2.05)
Subtotal	104	3.81	2.69	0.077 (3.13)	77	2.57	2.25	0.566 (0.33)
Interior								
Yellow-bellied flycatcher (*Empidonax flaviventris*)	20	0.56	0.69	0.652 (0.20)	14	0.31	0.56	0.281 (1.16)
Red-breasted nuthatch (*Sitta canadensis*)	1	0.06	-	~	0	-	-	~
Hermit thrush (*Catharus guttatus*)	4	0.06	0.19	~	2	-	0.13	~
Swainson's thrush (*C. ustulatus*)	2	0.13	-	~	1	0.06	-	~
Black-throated green warbler (*Dendroica virens*)	16	0.63	0.38	0.313 (1.01)	12	0.38	0.38	0.952 (< 0.01)
Ovenbird (*Seiurus aurocapillus*)	1	0.06	-	~	0	-	-	~
Subtotal	44	1.50	1.25	0.544 (0.37)	29	0.75	1.06	0.351 (0.87)

Riparian								
Spotted sandpiper (*Actitis macularia*)	8	0.31	0.19	0.470 (0.51)	8	0.31	0.19	0.476 (0.51)
Belted kingfisher (*Ceryle alcyon*)	1	0.06	-	~	1	0.06	-	~
Northern waterthrush (*Seiurus noveboracensis*)	25	0.88	0.69	0.564 (0.37)	21	0.75	0.56	0.511 (0.43)
Wilson's warbler (*Wilsonia pusilla*)	2	-	0.13	~	2	-	0.13	~
Yellow warbler (*Dendroica petechia*)	2	0.13	-	~	0	-	-	~
Rusty blackbird (*Euphagus carolinus*)	3	0.13	0.06	~	3	0.13	0.06	~
Subtotal	41	1.50	1.06	0.272 (1.20)	35	1.25	0.94	0.396 (0.72)
Open/Edge								
Olive-sided flycatcher (*Contopus borealis*)	2	-	0.13	~	2	-	0.13	~
Gray jay (*Perisoreus canadensis*)	4	-	0.25	~	4	-	0.25	~
Magnolia warbler (*Dendroica magnolia*)	15	0.13	0.81	**0.003** (9.02)	9	0.13	0.44	0.086 (2.94)
Mourning warbler (*Oporornis philadelphia*)	14	0.13	0.75	**0.005** (7.93)	8	0.06	0.44	**0.024** (5.06)
Dark-eyed junco (*Junco hyemalis*)	3	-	0.19	~	2	-	0.13	~
White-throated sparrow (*Zonotrichia albicollis*)	38	0.75	1.63	**0.022** (5.28)	31	0.75	1.19	0.206 (1.60)
Lincoln's sparrow (*Melospiza lincolnii*)	4	0.06	0.19	~	2	0.06	0.06	~
Subtotal	80	1.07	4.06	< **0.001** (33.70)	58	1.00	2.63	< **0.001** (12.09)
Ubiquitous								
Common flicker (*Colaptes auratus*)	1	-	0.06	~	1	-	0.06	~
American robin (*Turdus migratorius*)	29	0.94	0.88	0.847 (0.04)	22	0.75	0.63	0.668 (0.18)
Blackpoll warbler (*Dendroica striata*)	13	0.25	0.56	0.160 (1.98)	11	0.13	0.56	**0.028** (4.82)
Yellow-rumped warbler (*D. coronata*)	65	1.75	2.31	0.264 (1.25)	57	1.31	2.25	**0.046** (4.00)
Fox sparrow (*Passerella iliaca*)	5	-	0.31	~	2	-	0.13	~
Subtotal	113	2.94	4.12	0.073 (3.21)	93	2.19	3.63	**0.017** (5.75)

‡ Ruffed grouse (*Bonasa umbellus*) and purple finch (*Carpodacus purpureus*) were not observed along RIP or BUF transects, but were seen on INT and/or NFE transects (see table 3.3).

~ Insufficient observations for statistical comparison (n < 6 individuals).

Table 4.5. Regressions of bird counts for each guild and the total count against buffer strip width. Significant probability values (i.e., $P < 0.05$) are printed in boldface type.

Habitat selection guild	Slope	Intercept	r^2 (%)	$F_{1,14}$	P
Forest generalists	0.02	1.61	2	0.23	0.641
Interior forest	0.04	0.41	19	3.21	0.095
Riparian	0.03	-0.01	7	1.01	0.332
Open/Edge	0.07	0.22	9	1.44	0.250
Ubiquitous	0.12	-0.22	24	4.40	0.055
Total count	0.28	1.20	35	7.41	**0.017**

Chapter 5. Concluding discussion

5.1 Riparian bird assemblages

The high density and species richness typically attributed to riparian wildlife assemblages is often used to justify protecting riparian habitat (e.g. Naiman et al. 1993, LaRue et al. 1995). In this study, as well as several comparable studies in coniferous and coniferous-deciduous mixed forests, riparian bird assemblages were similar in species richness and abundance to those associated with interior forests. These results suggest that, relative to interior habitat, high riparian biodiversity is the exception in coniferous forests rather than the rule. This should not, however, be taken as an argument against protecting riparian habitat. The presence of several terrestrial riparian bird species (spotted sandpiper, belted kingfisher, northern waterthrush, Wilson's warbler, yellow warbler, rusty blackbird) as well as several species of water birds in the study area indicates that riparian habitat is important to the regional avifauna. Indeed, the fact that bird assemblages along unnatural edges and buffer strips were more species-rich and dense than those in undisturbed riparian habitat demonstrates the inadequacy of such measures of biodiversity as indicators of habitat quality, a role for which they are often advocated (see Magurran 1988). The validity of this concept is dependent on the existence of a positive correlation between habitat quality and biodiversity; such an association was not found in this study.

As a conservation practice, this research suggests that leaving riparian buffer strips was successful. These 20 - 50 m wide strips were used by a relatively abundant and diverse forest bird assemblage, which included species from a variety of habitat guilds. Notably, many species associated with riparian and woodland habitats were maintained in areas of extensive clearcutting. It appears that riparian species generally restrict themselves to riparian vegetation, and thus will not increase in numbers in wider buffer strips. The creation of an anthropogenic edge along the length of the buffer strip generates an influx of ubiquitous and open/edge species, leading to an increase in biodiversity compared to undisturbed shorelines. However, given the extent of fragmentation and clearing in most managed forests, the conservation of such species is generally not a great concern (Robbins et al. 1989a, Kirk et al. 1996).

In spite of this stated success of the 3 -5 year old buffers sampled, there may still be reason to consider modifying buffering practices. Over time, high windthrow rates, which were evident on many buffer strips sampled in this study, may reduce their habitat value for breeding birds (Darveau et al. 1994). Leaving wider buffers, and possibly conducting some thinning within them, may increase their "life span" in areas where windthrow causes high annual losses of trees along unnatural edges, as occurs in Newfoundland (Robertson 1993). In this case, determination of optimum buffer width, and possibly thinning intensity, can be achieved through modeling rates of windthrow in buffers against the time required for adjacent clearcut forests to regenerate to a point where they again provide habitat for forest wildlife and shelter from the wind.

5.2 Interior forest birds

From a conservation perspective, one of the most important findings of this study is that interior forest species formed an important component of this boreal forest avifauna, accounting for 37 % of sightings along interior forest transects. Concern for the protection of interior forest species is warranted, as populations of many, including black-throated green warbler and ovenbird, have declined significantly in portions of northeastern North America in recent years (Robbins et al. 1989b, Askins et al. 1990, Sauer and Droege 1992). Typically riparian buffers, which are presumably implemented to maintain habitat for riparian species, are viewed as inadvertently benefiting non-riparian species; however this assumption is unfounded (Thompson and Welsh 1993) and is not supported by this study.

The widening of buffer strips has been recommended in situations where all species encountered in undisturbed riparian habitat are not protected. However it is unlikely that this strategy will result in the successful conservation of interior forest birds. Even the 40 - 50 m wide buffer strips sampled in this study supported low numbers of these birds when compared to interior forests, and three of six species in the guild were not observed in any buffer strip. Spatially, wider buffers preserve large areas of corridor habitat, but cannot provide interior forest conditions greater than tens of meters from edges. This could be unfavorable to interior birds as edges, small patches and fragmented forests often function as ecological traps with high nest predation rates (Gates and Gysel 1978, Wilcove 1985, Møller 1988, Small and Hunter 1988, Yahner and Scott 1988, Gibbs

1991, Rudnicky and Hunter 1993, but see Hanski et al. 1996). One of the most frequently cited groups of nest predators are corvids (crows and jays; Gates and Gysel 1978, Wilcove 1985, Yahner and Scott 1988). We found gray jays, a known nest predator (Madge and Burn 1994), to be more abundant along non-riparian edges and buffer strips than riparian controls. From an economic perspective, widening buffer strips would require setting aside a large proportion of productive woodlands from harvesting, particularly in Newfoundland where surface water is abundant. In order to compensate for this restriction there may be a displacement of some harvesting into other areas of interior forest, leading to greater fragmentation of woodlands in the region.

Limited information is available regarding appropriate conservation practices for interior species. In a study in the middle Atlantic states, Robbins et al. (1989a) found that the highest probability of detecting most forest nesting neotropical migrants during a point count occurred in patches larger than 250 ha. For ovenbird, wooded plots greater than 175 m wide were required in Iowa (Stauffer and Best 1980), and the probability of detection in the middle Atlantic states was reduced by greater than 50% in patches of less than 6 ha (Robbins et al. 1989a). Freemark and Collins (1992) found that at least 50% of area-sensitive bird species could be detected in patches of 54 - 65 ha at study sites in Ontario, Missouri and Illinois. However, it should be noted that, while patches of 50 - 250 ha may support many forest songbirds, much larger patches may be required to maintain species having larger home ranges. For example boreal owl (*Aegolius funereus*), which is associated with old age classes of balsam fir forest in the study area (Gosse, submitted), may occupy annual home ranges exceeding 1500 ha (Johnsgard 1988). Robbins et al. (1989a) suggested that 3000 ha was the minimum size of stand that might be expected to retain all forest interior bird species.

An effective forest bird conservation strategy would incorporate current riparian buffers, while ensuring that some large (>> 250 ha), relatively symmetric (i.e., low edge to area ratio) tracts of forest are maintained in areas of intensive harvesting. Valleys or watersheds may prove to be useful units within which to carry out such management, as these are typically harvested as such in the province. Given the recent concern over the status of interior forest birds, the inadequacy of buffers as habitat, and the extent of clearcutting in the boreal forest, detailed information on the impacts of current management practices on populations of these species is needed. Until this becomes available, a precautionary approach in which large tracts of interior forest habitat are protected, is necessary (see Montevecchi and Bouman 1993).

5.3 Scope and limitations

In interpreting the information presented here, consideration should be given to both the scope and limitations of the research. Findings consider habitat use by terrestrial birds in a balsam fir dominated ecosystem during the breeding season. Caution must be exercised when extending these findings to other forest types, regions or seasons. The use of relative abundance as an indicator of habitat quality may be misleading in some cases. For example, though numbers of a species may be high along edges, nest predation may be so frequent that edge habitats act as population sinks or ecological traps (see Gates and Gysel 1978, Wilcove 1985, Small and Hunter 1988, Yahner and Scott 1988, Rudnicky and Hunter 1993). The grouping of species into guilds has also been criticized in past, as species within a guild may not exhibit a common response to a treatment (Mannan et al. 1984). However this should not be interpreted as a failure of the guild based approach. Rather, it indicates that the trait used to define the guild (e.g., foraging behaviour, diet, etc.) does not predispose species to a certain response (i.e. no common effect). Also, it should not be assumed that rare species necessarily respond similarly to other guild members. There is always the possibility that a small sample size (as with many of the less commonly observed species in this study) may lead to a failure to detect an effect on a species. Indeed some woodland species known to be present in the study area (e.g., three-toed woodpecker, *Picoides tridactylus*) were not observed during any survey.

5.4 Summary

If Newfoundland forest managers set aside riparian buffer strips in order to maintain habitat for riparian species, then the current practice may be successful. However, buffer strips do not provide suitable habitat for interior forest birds, which are currently experiencing a widespread decline in much of North America. Other wildlife, such as the endangered Newfoundland subspecies of American pine marten (*Martes americana atrata*), may also suffer from forest fragmentation. Research conducted in the Corner Brook area has found that individual pine marten require intact stands of old growth balsam fir exceeding 15 ha (Bissonette et al. 1991). Consequently, there is a need to ensure that, in addition to buffer strips, large tracts of continuous forest (not bisected by access roads etc.) are maintained in watersheds where extensive harvesting is occurring. It is clear that separate complementary strategies are required for the conservation of riparian and interior forest wildlife.

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