

**SHORT-TERM RESPONSES OF SONGBIRDS TO ALTERNATE
HARVESTING METHODS IN A HIGH ELEVATION FOREST**

by

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ABSTRACT

Silvicultural alternatives to clearcutting have been promoted in forests of British Columbia to attempt to simulate short-term natural disturbances typical of certain forest types as these cuts are hypothesized to mitigate negative impacts on wildlife dependent on forests. However, the potential effects of these harvesting activities have not been studied enough to evaluate their success in mitigating wildlife impacts. I examined the response of songbirds breeding in high elevation, Engelmann spruce-subalpine fir forests (Sicamous Creek Research Forest) to alternative forms of forest harvesting using the variable circular method of point counting to determine relative abundance. The experimental harvesting treatments at Sicamous created openings (perforations) in the original forest that ranged in size from 10ha clearcuts to small gaps (0.01 ha) resulting from the harvest of selected trees. The community of songbirds that breed in the Sicamous Creek Research Forest was monitored over a five-year period that included both pre- and post-harvest conditions. The original songbird community remained relatively unchanged after harvest and among the various treatments. Few new species colonized the newly created habitats (openings) and did so in very small numbers. Golden-crowned kinglet declined significantly post-harvest in harvested areas with the most pronounced declines in selection and 10 ha treatments. Conversely, dark-eyed junco responded positively to the harvesting and increased in abundance in all harvested treatments. In general, the creation of a variety of habitats through alternative harvesting methods appears to lessen impacts and allow much of the songbird community to persist in high elevation forests. This persistence may be related to an evolutionary adaptation

of songbirds to small-scale disturbances typical of high elevation forests that alternative harvesting methods simulate. Future research should focus on long-term monitoring to determine reproductive success in the various harvesting techniques.

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I dedicate this thesis to my parents, who never stopped believing in me.

INTRODUCTION

Declines of Neotropical songbird species have been attributed, for the most part, to destruction and degradation of natural habitats (Fahrig 1997, Finch and Stangel 1994). In forested areas, these anthropogenic activities usually result in fragmented landscapes that lower abundances of forest-dependent songbird species (Robinson 1992). Factors associated with the declines include songbird responses to the insularization phenomenon as related to the theory of island biogeography (Mac Arthur and Wilson 1967, Hunter 1990) and other factors that result from the concomitant creation of edges, including increased predation rates (Yahner and Scott 1988, Hartley and Hunter 1998), competition (Faaborg *et al* 1995), and brood parasitism (Robinson *et al* 1995, Brittingham and Temple 1983).

Evidence of Landscape effects

Nearly all of the evidence that support landscape explanations of bird declines stem from studies carried out in large-scale fragmented landscapes surrounded by matrices of agricultural, suburban, or urban landscapes and, in most cases, in areas where fragmentation had taken place decades prior to the study (Sallabanks *et al* 2000; e.g. Askins *et al* 1987, Robbins *et al* 1989). In contrast, similar studies in forest-dominated landscapes have reported mixed responses by various taxa (including songbirds) to the hypothesized effects (Debinski and Holt 2000). Schmiegelow (1997) reported a resiliency of the boreal songbird community to various levels of fragmentation in a boreal forest in Alberta. Similarly, Schieck *et al* (1995) found only weak evidence to support relationships between songbird species richness/abundance and patch size in coastal montane forest fragments of British

Columbia. Others however, have found supporting evidence of the fragmentation paradigm in primarily forested landscapes. Enoksson *et al* (1995) found that certain songbird species responded negatively to isolated patches of deciduous forests embedded in a coniferous forest matrix in south central Sweden. Similarly, Norton and Hannon (1997) found that 39% of species declined after logging in a boreal mixedwood forest in Alberta. Martin and Eadie (1999) found a strong and positive response of cavity nesting bird communities to forest edges and degree of natural fragmentation in interior Douglas fir forest ecosystems in BC. Thus, there is no clear agreement in terms of how fragmentation affects songbird species in forest-dominated landscapes.

Issues of Incipient Fragmentation

Despite the number of habitat fragmentation studies conducted, few have documented the responses of songbirds in extensively forested landscapes where fragmentation is incipient (Robinson and Robinson 1999, Hagan *et al* 1996, Buford and Capen 1999). The process of habitat fragmentation typically begins with the creation of small openings into otherwise continuous forest tracts through a process termed perforation (Hunter 1996). Timber harvesting, and in particular selective and group selection logging are two harvesting systems that create small openings in otherwise continuous forest tracts (Chambers *et al* 1999).

Multiple openings potentially have negative consequences for the original songbird community. Although fragmentation per se has not occurred, the perforation process can result in negative impacts resulting from habitat loss and the creation of edges. The creation

of multiple openings maximizes the ratio of edge to opening area and may represent a worse case scenario for forest-interior bird populations (Robinson and Robinson 1999, Thompson 1993). Furthermore, these openings represent a net loss in forest habitat available. Recent studies have concluded that habitat loss may play a much more important role in songbird declines than spatial configuration of habitats (e.g. Schmiegelow and Mönkkönen 2002). A model examining the relative impacts of fragmentation and habitat loss on organisms also suggests that habitat loss has a much larger impact on population extinctions than habitat fragmentation (Fahrig 1997). Thus both the configuration and magnitude of forest retention may be important in determining the impacts of these harvesting methods in predominantly intact forest ecosystems (Drolet *et al.* 1999).

In contrast, multiple openings (gaps) may have potentially little effect on forest-dependent songbirds if they are created in such a way that they simulate natural disturbance processes (Robinson and Robinson 1999). For example, if the openings created resemble those typically created by treefall or localized insect outbreaks, then some species, or the entire assemblage of species inhabiting that area may be evolutionarily adapted to accommodate such changes (Steventon *et al.* 1998, Rudickny and Hunter 1993). In addition, nest predation, brood parasitism, and competition by colonizing species in forested landscapes may be mitigated because of the remoteness of an area; species responsible for these impacts, which are usually associated with human developments (e.g. corvids, brown-headed cowbirds) may be unable to colonize the newly created habitats and thus interact with the original songbird community.

Recent studies investigating the responses of the creation of small openings via these silvicultural methods have concluded that this practice has little effect on the songbird community. In continuous hardwood forests, Robinson and Robinson (1999) reported that the removal of 15-50% of the wood volume using a combination of selective harvest and group selection harvest (0.02-0.4 ha) in a hardwood forest had little effect on most bird species that typically occupy mature closed-canopy forests. Germaine *et al* (1997) also reported similar results, where 20-36% of the forested area was removed using 0.04 ha openings. In conifer-dominated landscapes, Chambers *et al* (1999), found that species composition was similar in uncut and small patch cut (0.02 ha) treatment stands where 30% of the wood volume was removed. Similarly, Steventon *et al* (1998) concluded that combinations of selection cut and small opening cuts 0.1 to 0.5 ha in size at two wood removal intensities (30% and 60%) supported songbird communities similar to those found in undisturbed mature forests in the first two years after harvest, although species composition in the heavier removal treatment showed some shift towards species typical of clearcuts.

Forestry Impacts on High Elevation Forest Ecosystems and Songbird Communities

High elevation forests are characterized by short growing seasons, extreme temperatures, and a comparatively simpler resource spectrum (Sabo 1980). As a result, songbird communities occupying high elevation forests are relatively simple and comprised of a few species that are able to tolerate extreme conditions (Sabo 1980, Sabo and Holmes 1983). High elevation forests are among the most undisturbed habitats remaining on the Pacific Northwest and as such are of important conservation concern (Martin 2001). As

lower elevation forests are degraded, mature forest-dependent songbird species that occur in a broad range of elevations may ultimately become restricted to these high elevation forests. However, in recent years high elevation forests have been subjected to increased harvesting pressures, which may be changing the overall characteristics of these forest, and the songbird communities that inhabit them. Songbird communities inhabiting high elevation forests are faced with severe environmental constraints that result in high energetic costs for living and breeding, delayed breeding schedules, reproductive stochasticity, and fewer broods per season (Martin 2001). Therefore, harvesting activities may present an additive effect to the existing environmental constraints, which may impact songbird communities of high elevation forests negatively. Despite the increased focus on high elevation forest harvesting, our knowledge of the impacts of these activities on the ecology of these forests is limited (Martin 2001, Thomas 1987).

In British Columbia, high-elevation forests comprise 14% of the landbase (Lloyd *et al* 1990). Since the early 1970's, an increasing proportion of the annual allowable cut (AAC) in B.C. has come from this biogeoclimatic zone (Yano 1991). In 1991, the harvesting rate was 10,000 ha/yr., approximately one third of the total area harvested annually in the region (Yano 1991) and has without a doubt increased since (D. Lloyd, *pers. comm.*). Clearcutting has been the preferred harvesting method. However, this large-scale disturbance is not consistent with natural disturbances typical of these forests. The factors mostly responsible for the structural make-up of these forests are guided by small-scale disturbances such as insect outbreaks and windfall (Parish *et al* 1999).

In recent years, the Forest Practices Code of British Columbia has encouraged the use of alternative silviculture practices to clearcutting including selection and group harvest methods. The underlying assumption was that by implementing these harvesting methods, it would promote the development, retention, or creation of late successional features, and mimic presettlement disturbance regimes and thus mitigate impacts on forest interior dependent species. However, literature pertaining to impacts of these alternative silviculture methods in these forests is limited (Raphael 1987, Scott *et al* 1982). Therefore, if objectives are to manage for forest dependent species, we need a clearer understanding of how well various alternative silviculture methods perform in maintaining them.

OBJECTIVES

In 1991 the Forest Science section of the Kamloops Forest Region began a study of high-elevation silviculture systems. The study's broad objective has been to provide forest managers and interested groups with ecological information regarding high elevation forests in BC's Southern Interior and their responses to various levels and styles of harvesting. More specifically, the goal was to examine the manner in which different silvicultural methods affect a variety of forest ecosystem components, including microclimate, hydrology, wildlife, soil, nutrient cycling, and tree regeneration. This larger initiative involved the experimental harvesting of one 500 ha forest tract using clearcut, group selection, and selection harvest methods and provided the opportunity to determine how these alternative silviculture methods affect songbird populations.

In this thesis I examine the responses of the breeding songbird community in a high elevation forest to various sizes of canopy opening size with a constant (30%) wood volume removal at two levels of resolution. First I examine the responses of songbirds to the resulting harvesting matrix at the study area level and here I include available data I collected for the entirety of the study area and compare the two years pre-harvest with the two years post harvest. Secondly, I examine the short-term responses (two and three years after harvest) of the original songbird community to the five treatments.

I predicted that the songbird community would not be affected by the harvesting activities. Recent studies of forest perforation have shown that the removal of approximately 30% of the wood volume via selective and group harvesting methods had little or no effect on the original songbird community for a period of 1 and 2 years post harvest (e.g. Moorman and Guynn 2001, Robinson and Robinson 1999, Chambers *et al* 1999).

Changes in species richness and abundance are usually the result of a reorganization of the entire species assemblage in the newly harvested areas following clearcut harvest (Spies *et al* 1990). However, selection and group harvesting methods may not be detrimental to the original songbird community for a number of reasons. First, the creation of small openings may mimic natural disturbance patterns typical of the area and the songbird community could be evolutionarily adapted to cope with these changes (Schmiegelow *et al* 1997, Schieck *et al* 1995). Second, the openings created may not be large enough and could be unsuitable for colonizing species (Robinson and Robinson 1999). Finally, population densities of songbirds in high elevation forests are typically lower than those found in low

elevation forests (Martin 2001). Therefore, if actual densities are below the carrying capacity (low density), it implies large spatial scale for territories. Therefore small cuts (loss of habitat) would represent only a small proportion of the territory and more likely to represent a natural gap to high elevation songbird species.

At the individual species level, I predicted that species associated with mature forests would either exhibit declines proportional to the habitat lost, or decrease at levels inversely proportional to increasing canopy opening size and that the changes observed would be related to the amount of edge created by the various harvesting treatments. That is, abundance would be lowest in those treatments with the most edge. Conversely, I predicted the opposite for species that have higher habitat plasticity (e.g. species that can be found in a variety of habitats and are commonly associated with disturbances).

Although the sampling layout was different than that of the individual treatments, I opted to examine the responses of songbirds to the resulting forest matrix created by all treatments to take advantage of available pre-treatment data and to validate the results observed at the treatment-level. A discrepancy of results between both levels of analysis would in some cases suggest that small effects not evident at the forest stand level may become significant if examined at a larger scale (McGarigal and McComb 1995; Boulinier *et al* 2001).

METHODS

Study Site

The Sicamous Creek Research Forest is located approximately 15 km south of the town of Sicamous in the southern interior of British Columbia (Fig 1). The area is estimated to receive 1,000 mm of precipitation annually (Feller 1997). The mean annual temperature is 1°C, and the continuous frost-free period is no more than 40 days, mainly accumulated in the months of July and August (Lloyd and Inselberg 1997). The study area, prior to treatments, consisted of approximately 500 ha of continuous mature forest in the Engelmann spruce-subalpine fir wet-cold (ESSFwc2) biogeoclimatic variant. The site was classified as a homogeneous old-growth stand, age classes 8 and 9 (250-400yrs), with a northeastern aspect and average slope gradients ranging between 20-40% (Puttonen *et al.* 1997). Subalpine fir (*Abies lasiocarpa*) (90%) and Engelmann spruce (*Picea engelmannii*) (10%) were the two codominant tree species (Parish 1997). The understorey was dominated by white rhododendron (*Rhododendron albiflorum*), oval-leaved blueberry (*Vaccinium ovalifolium*), and black huckleberry (*Vaccinium membranaceum*). False azalea (*Menziesia ferruginea*), and black twinberry (*Lonicera involucrata*) occurred scattered throughout the site. The herb layer consisted mainly of sitka valerian (*Valeriana sitkensis*), mountain arnica (*Arnica latifolia*), oak fern (*Gymnocarpium dryopteris*), one-leaved foamflower (*Tiarella unifoliata*), and rosy twisted stalk (*Streptopus amplexifolius*; Lloyd and Inselberg 1997).

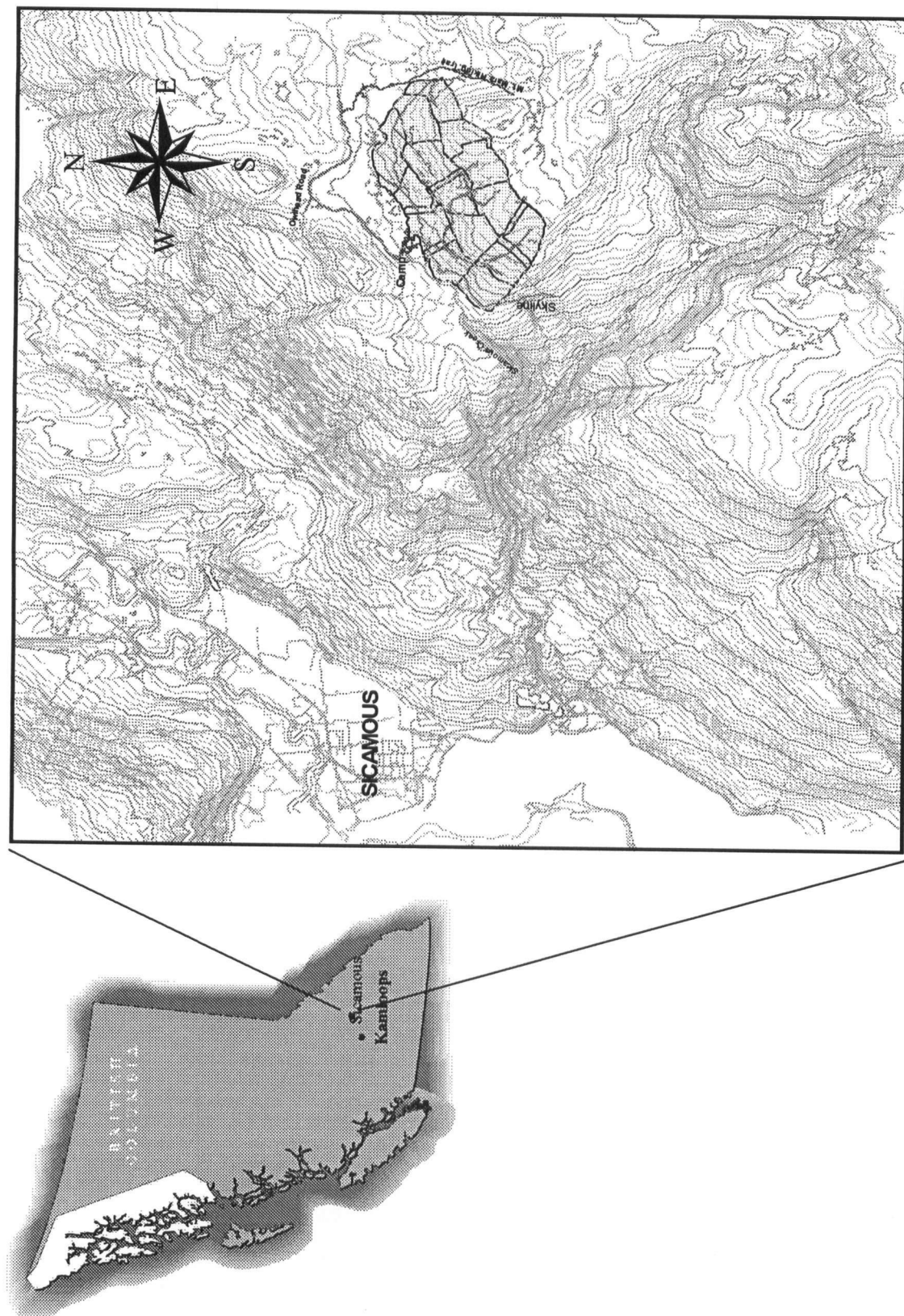


Figure 1. Relative Location of the Sicamous Creek Research Forest

Experimental treatments at the Sicamous Creek Research Forest.

The experimental design of the Sicamous Creek Research Forest involved five treatments. The main treatment effect was canopy opening size (Table 1, Fig. 2). By design, one-third of the wood volume was removed across all treatments except, of course, in the uncut reserves.

Table 1. Characteristics of experimental treatments. For each treatment approximately one third of the wood volume was removed.

Treatment	Description
<i>Uncut reserve:</i>	no removal
<i>single tree selection:</i>	uniform removal across the block
<i>0.1 ha:</i>	sixty 0.1 ha openings evenly spaced across the blocks
<i>1.0 ha:</i>	ten 100 x 100m openings evenly spaced;
<i>10 ha:</i>	single 320 x 320m opening within the block

Each treatment unit was approximately 30 ha. in size. One hundred metre wide buffer strips were left at adjoining borders between single-tree selection treatment units and 0.1 ha treatment units to minimize confounding effects. Each treatment was replicated three times in a complete randomized block design; however, one stream crossed through the center of one 10 ha treatment the logical location (center of the treatment), therefore, the ten hectare opening had to be placed on one side of the treatment so that it encroached on one uncut reserve, as a result this uncut reserve was smaller than expected. To overcome this

logistical problem and retain a third uncut unit, I sampled beyond the boundaries of this unit outside of the boundaries of the existing design (see Fig 2).

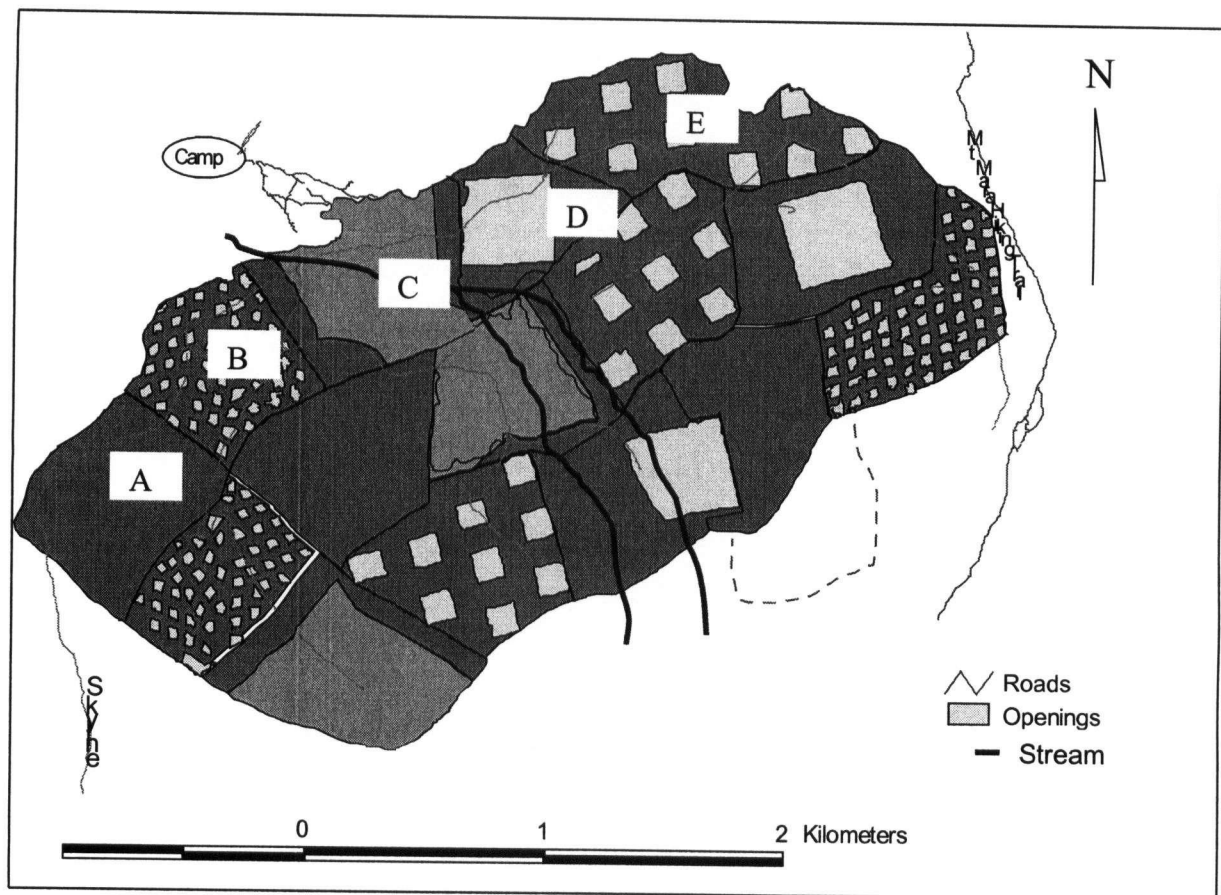


Figure 2. Experimental layout and treatments at the Sicamous Creek Experimental Forest. A- Uncut Reserves, B- 0.1ha openings, C- Selection Cut, D- 10 ha opening, E- 1ha openings . Each Treatment was replicated three times. Dashed line indicates the sampling extension of one uncut reserve.

The study area was harvested in the winter of 1994/95 according to the experimental design (Fig 3). To comply with Workers Compensation Board (WCB) regulations standing dead trees were removed from all treatment units except in the uncut reserves in June and April of 1995 and 1996 respectively. Because removal activities were conducted after 0900 hours the noise created by chain saws did not affect songbird data collection.

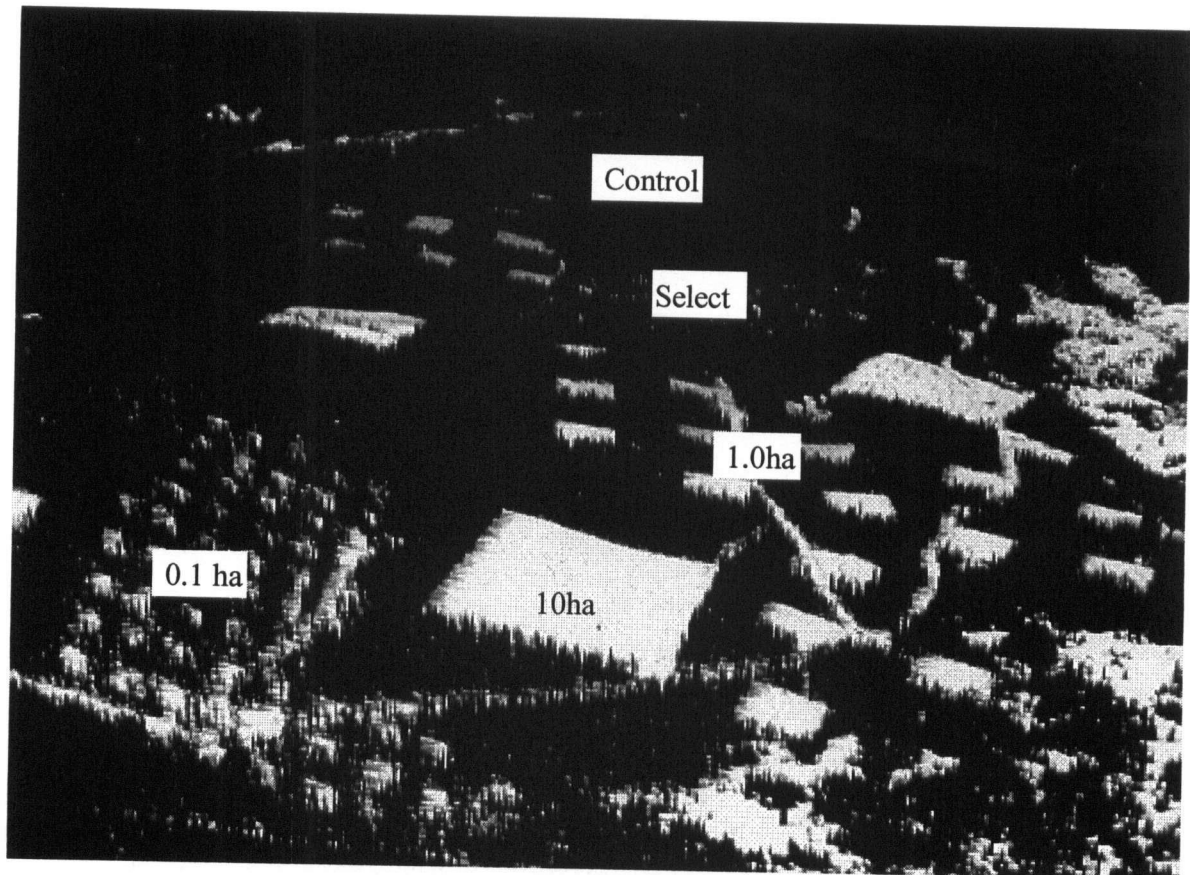


Figure 3. Aerial view (south-west) of the resulting forest matrix after harvest.

Bird Sampling

I sampled the songbird community using the variable circular plot method (Reynolds et al 1980). To ensure independence between plots, all points were positioned at least 250m apart (Fig. 4). In each year, point count stations were sampled twice during the breeding season (June and mid July). All surveys were conducted between 0500-and 0900 in the morning to coincide with peak singing activity (Ralph *et al* 1993). Each census was conducted for fifteen minutes preceded by a one-minute rest period to allow for birds nearby to resume their activities after arriving at the census point. Surveys were not conducted on

rainy days because singing activity is reduced and less detectable (Robbins 1981). I conducted all surveys in 1993 and 1994 along with a second individual that acted as a recorder. In subsequent years (1995-1997) an additional experienced observer/recorder team participated in the census. In these years, observer variation was standardized across all stations by rotating observers among the point count stations. Aural and visual detections of individual birds were recorded at each point (Appendix I). Locations of birds that were recorded were plotted onto a Cardinal plane to monitor bird movements and thus avoid counting individuals more than once. Method of detection (i.e. song, call, visual) and horizontal distance from the bird to the observer was also recorded.

Sampling Design

Entire Study Area Sampling

The entire study area was sampled 2 years prior to (1993-94) and two years post-harvest (1995, 1996). Seventy permanent point count stations were established along ten parallel transects placed 250 metres apart (Fig. 4). Each transect contained seven permanent plots placed at equal intervals of 250 metres. Based on the size of territories of the songbirds sampled, this distance was considered appropriate to avoid counting individual birds more than once (see also Delasalla *et al*, 1996). The systematic arrangement of transects and plots, was chosen because at the time of placement, the actual locations of the future experimental treatments were unknown. Each point was identified using a submetric Global Positioning System (GPS) and incorporated into a digital map of the study area. After treatment

application, sixty-nine (15 in uncut reserves units; 54 in treatment units) of the seventy points fell within the experimental study area, and were used for pre and post harvest comparisons. (Fig. 4). Because the number of uncut forest point count stations was anticipated to be low, an additional permanent transect with seven points located in uncut forest had been placed 500 m away from but adjacent to the study site, which increased the number of point count stations for uncut forest to 22.

Individual Treatment sampling

Individual treatments were sampled using a different strategy following harvest in 1996 and 1997. Each replicate treatment unit contained three sampling points strategically placed in each treatment unit for a total of 45 point count stations. Each point was positioned so that its center would lay at least 100 m away from the edges of a treatment and at least 250 m away other point count stations. Point count stations were selected so that each would sample approximately an area that was two-thirds forested, which approximated the amount of forest that remained in each treatment after harvest (Fig 4).

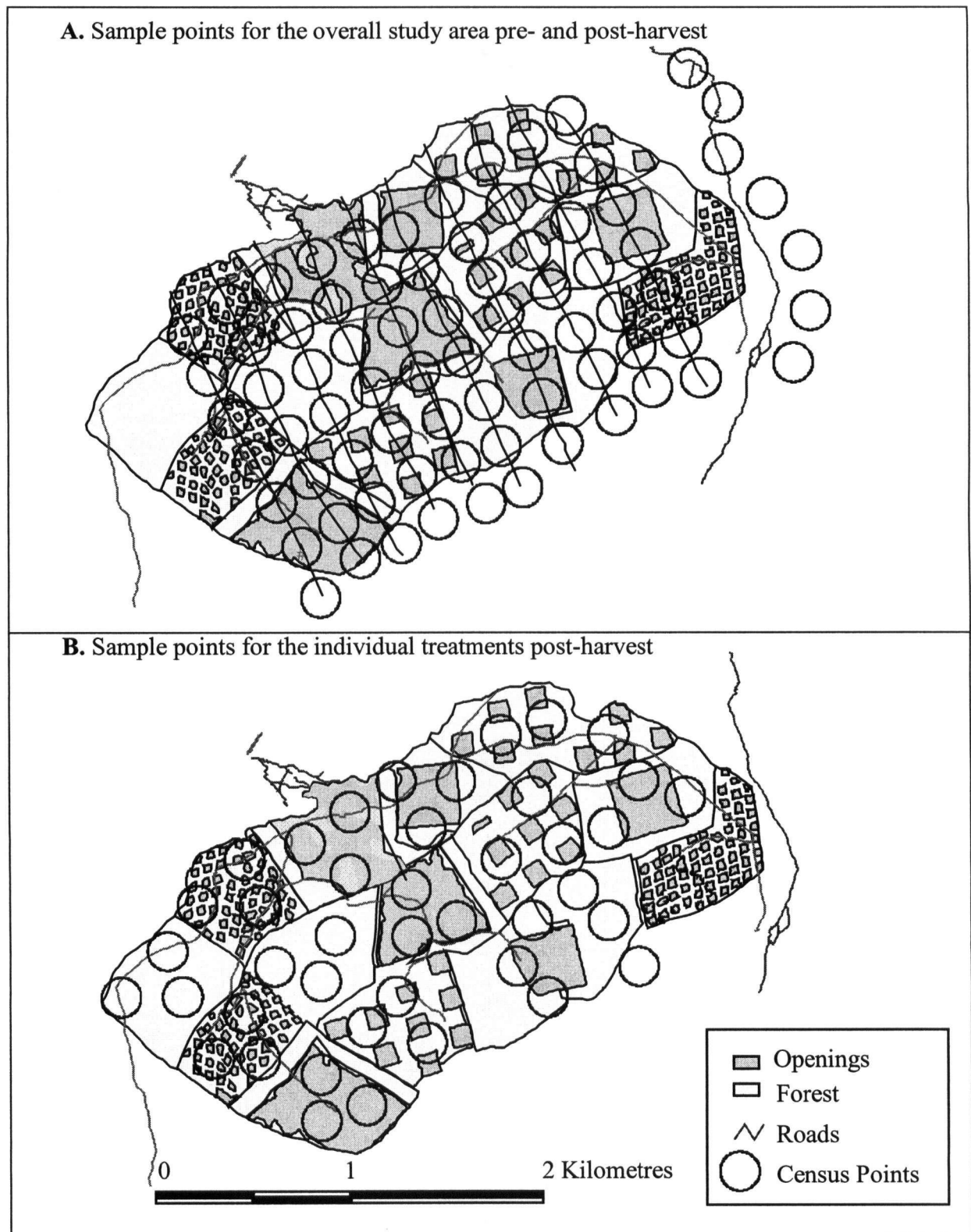


Figure 4. Sampling strategies and point count layout for sampling songbirds at the Sicamous Creek Research Forest

Data Analysis

Songbird Community Measures

Species richness (total number of species), total abundance (number of detections of all species), diversity indices, and proportional similarity, were used to describe and compare avian community structure among treatments and years. Species richness values were calculated by averaging observations of all species recorded during census for both census periods in each year. Total relative abundance was determined by averaging all detections made within a 100-meter radius of the observer for the first census period in a given year because numbers of core songbird species detected during the second census period were very low. Shannon diversity $H' = -\sum p_i \log p_i$ and evenness $\left(\frac{H'}{H_{\max}} \right)$ indices were used to describe community diversity (Hunter 1990). Community similarity indices were derived using the Renkonen Index (Renkonen 1938 in Krebs 1999), an index value that reflects the relative abundance of each species and the species difference at treatments relative to the uncut reserves. Renkonen index values can range from 100, where both communities support exactly the same species in the same relative abundance, to 0, where none of the species are the same.

For certain community measures (richness and abundance) I grouped species into mature forest associates and others. The criteria for placing a species into either category

were based primarily on consistent species-habitat associations as identified by Harris (1984), Delasalla (1996), and others.

Individual Species Abundance

Because species-specific songs have effective detectability distances that vary considerably from species to species (Wolf *et al* 1995), I determined the effective detection distances (EDD's) for common species (>5% of the total detections) following methods developed by Reynolds *et al* (1980, Table 2). The EDD's for each species were derived by plotting the number of individuals detected in 10 metre concentric rings away from the observer. The distance where calculated densities declined by more than 50% (a rough estimate of the inflection point) between concentric rings was then considered the effective detection distance (the maximum distance at which individuals of a given species could be heard reliably). The EDD for each species was then used to calculate the area effectively sampled for each species at every census point. Relative abundance was calculated by averaging the number of detections within the EDD of singing males for both census periods.

Because the data under analyses consisted of counts, and followed a Poisson distribution, square root transformations in the form of $X' = \sqrt{X + \frac{3}{8}}$ were performed on the data to stabilize the variances around the means and thus better conform to the assumptions of normality where X is the observed relative abundance mean and X' is the transformed mean (Sokal and Rohlf, 1995). All statistical tests were performed on the transformed data,

but tabulated values are presented in their original form as mean detections of singing males /census point (\pm SE).

Table 2. Effective detection distances for seven common species at Sicamous Creek.

Common Name	Effective Detection Distance (m)
Red-breasted Nuthatch	90
Winter Wren	90
Golden-crowned Kinglet	50
Hermit Thrush	70
Varied Thrush	100
Yellow-rumped Warbler	70
Dark-eyed Junco	80

Study Area Comparisons

I used a split-plot ANOVA procedure (SPSS 1992) to test for overall differences in community indices and relative abundance of common species among point count stations falling within harvested and unharvested forest patches, regardless of harvesting type applied. The data for the two conditions (harvested vs. unharvested) was pooled for the two years of pre (1993-94)- and post-harvest (1995-96) (A. Kozak *pers. comm.*). Because unharvested plots within the study site were relatively few (n=15), point count stations in uncut forest from the transect placed adjacent to the study area were included in the analyses to provide a larger sample size (n=22). I tested for significant year by treatment interaction

effects where a significant value would indicate either positive or negative responses by songbirds and changes in the songbird communities to the harvesting activities (Zar 1999).

Treatment Comparisons

Avian community indices and relative abundance for each common species (defined as one that made up at least 5% of the total songbird detections) were compared among treatments and between years using a repeated measures ANOVA (Zar 1999). I used a repeated measures analysis of variance because each census point and treatment was sampled repeatedly over the duration of the study (Kuehl 1994). Species richness was calculated by pooling the numbers of species detected for all three point count stations in a treatment unit. Likewise, relative abundance of a particular species was calculated by pooling the number of detections of singing males for all three point count stations in a treatment unit in a given year.

When significant year and year-by-treatment interactions were detected, treatment effects were evaluated using a separate two-way ANOVA for each year. Where no differences between the two years following harvest were detected, both samples (mean for the two years) were included in a single two-way ANOVA to test for treatment effects. In instances where ANOVA procedures identified significant treatment effects for community indices, abundance of mature forest associates, and individual bird abundance, Tukey's HSD was used as the *a posteriori* test to identify differences among treatments (Zar 1999).

When species showed significant year effects before and after harvest at the level of the entire study area, I consulted regional trends (estimating equations method) obtained by the Breeding Bird Survey to help determine if the trends observed were consistent with regional trends (Sauer *et al* 2001).

RESULTS

Community Structure of Songbirds Inhabiting the Sicamous Creek Research Forest

The songbird community at the study area was relatively simple. Based on the sampling conducted at the study area level, a total of 21 songbird species were detected at the study site prior to harvest. Eight species comprised 90% of the observations (n= 1,559). Varied thrush was the most abundant species followed by golden-crowned kinglet, winter wren, hermit thrush, pine siskin, dark-eyed junco, red-breasted nuthatch and yellow-rumped warbler. Twelve additional species accounted for the remaining 10% of the breeding songbird community detections. Following harvest the species richness across the entire area increased to 26 species. However, the core species remained the same, albeit at slightly different ranks, and comprised 84% of the total observations (n=1,590; Table 3). Winter wren became the most abundant species, followed by varied thrush, dark-eyed junco, hermit thrush, pine siskin, golden-crowned kinglet, yellow-rumped warbler, and red-breasted nuthatch. Seven songbird species were recorded for the first time during the 2 years post-harvest. Some however consisted of single detections and likely transients through the study area (Table 3).

Table 3. Proportional abundance of all songbird species detected at point count stations (n=77) before and after harvest in the Sicamous Creek study area.

Species	Treatment	
	Pre-Treatment	Post-treatment
Varied Thrush	0.141	0.132
Hermit Thrush	0.117	0.119
Golden-crowned Kinglet	0.116	0.079
Winter Wren	0.114	0.143
Pine Siskin	0.109	0.091
Dark-eyed Junco	0.106	0.128
Red-breasted Nuthatch	0.080	0.068
Yellow-rumped Warbler	0.073	0.078
Gray Jay	0.045	0.024
Pine Grosbeak	0.037	0.034
Boreal Chickadee	0.031	0.026
Mountain Chickadee	0.009	--
Brown Creeper	0.008	0.004
Wilson's Warbler	0.005	0.014
American Robin	0.0006	0.014
Chipping Sparrow	0.0006	0.010
Fox Sparrow	0.0006	0.007
Olive-sided Flycatcher	0.0006	0.0006
Swainson's Thrush	0.0006	0.001
Townsend's Warbler	0.0006	0.004
Vesper Sparrow	0.0006	--
Clark's Nutcracker	--	0.0006
Evening Grosbeak	--	0.001
Lincoln's Sparrow	--	0.002
Ruby-crowned Kinglet	--	0.006
Red Crossbill	--	0.001
Townsend's Solitaire	--	0.002
White-crowned Sparrow	--	0.001
Total species	21	26

Across the entire study area, the total number of species detected (species richness) was not significantly different between points in the uncut and harvested points before and after harvest (treatment by year interaction: $F=0.36$; $df=1,75$; $p=0.55$, Table 4). During the two years post harvest, at individual treatment level, there was a significant year effect ($F=15.43$; $df=1,10$; $p=0.003$) but no treatment or interaction effects (treatment $F=2.19$; $df=4,10$; $p=0.143$; treatment by year interaction: $F=1.83$; $df=4,10$; $p=0.20$).

When years were analyzed separately, year one post-harvest also showed non-significant results for treatments ($F=1.16$, $df=4,14$; $p=0.38$). However, in year two, there was a significant treatment effect ($F=3.61$; $df=4,14$; $p=0.04$; Table 5), although differences among the treatments could not be discerned by multiple comparison tests. Qualitatively, it appeared that species richness was higher in the 1.0 ha treatments followed by 10ha, 0.1 ha, selection, and uncut treatments.

The number of songbirds detected (species abundance) per census point across the study area at uncut and harvested plots prior to and after harvest was marginally significant (year by treatment interaction $F=3.85$; $df=1,75$; $p=0.053$) with numbers of individuals detected decreasing in uncut reserves and increasing in the harvested plots (Table 4). However, when the individual treatments were examined separately, total relative abundance was not significantly different between years ($F=0.74$; $df=1,10$; $p=0.41$) or among the different treatments ($F=1.92$; $df=4,10$; $p=0.18$).

Shannon diversity and evenness indices were not significantly different in the uncut and harvested treatments prior to- or after treatment application (Shannon treatment by year interaction: $F=0.75$; $df=1,75$; $p=0.39$; Evenness treatment by year interaction: $F=0.91$; $df=1,75$; $p=0.34$). Results were also similar for comparisons among the individual treatments: Shannon diversity, and evenness community indices were not significantly different among uncut and any other treatment ($F=0.36$; $df=4,10$; $p=0.83$ and $F=0.50$; $df=4,10$; $p=0.73$).

Table 4. Community parameters for uncut and treatment sites before and after harvest.

Community parameter	Uncut reserves n=22		Treatments n=55	
	Pre-treatment	Post-treatment	Pre-treatment	Post-treatment
Shannon diversity	0.91	0.90	0.91	0.93
Evenness $J=H/H_{max}$	0.95	0.93	0.94	0.94
Similarity	0.63 (0.02)		0.64 (0.01)	
Species richness	9.22 (0.29)	9.40 (0.36)	9.49 (0.23)	9.98 (0.26)
Total relative abundance	9.88 (0.39)	9.31 (0.46)	10.07 (0.23)	10.70 (0.32)

Community similarity indices between uncut and harvested plots did not differ ($F=0.04$; $df=1,75$; $p=0.84$). At the individual treatment level, similarity indices ranged between 41% and 69% although there were no significant differences among the treatments (Year 1: $F=3.83$; $df=3,6$; $p=0.07$; and Year 2: $F=1.22$; $df=3,6$; $p=0.37$). Qualitatively, in year 1 the 0.1 ha treatment was most similar to uncut reserves followed by 1.0, selection and 10 ha. In the second year, selection cut was most similar to uncut reserves followed by 1.0, 0.1, and 10 ha treatments.

Table 5. Comparison of community parameters (mean and (SE) for 3 point count stations) for individual treatment units 1 and 2 yrs post-harvest

Community indices		Treatment				
		Uncut	Selection	0.1 ha.	1.0 ha	10.0ha
Shannon diversity	Year 1	1.05	0.88	1.04	1.01	0.96
	Year 2	0.97	0.96	0.98	1.05	0.97
Evenness	Year 1	0.91	0.90	0.90	0.91	0.92
	Year 2	0.93	0.92	0.91	0.91	0.88
Species richness	Year 1	16.33 (1.45)	13.00 (1.52)	15.33 (0.66)	15.33 (1.45)	14.00 (0.57)
	Year 2	11.33 (0.88)	10.66 (0.66)	11.67 (0.33)	14.33 (0.88)	13.66 (1.20)
Percent similarity	Year 1	NA	63.33 (4.68)	69.50 (2.01)	66.00(1.98)	56.66 (2.76)
	Year 2	NA	50.50 (6.94)	46.50 (1.06)	49.66 (8.90)	41.83 (7.81)
Total relative abundance	Year 1	33.60 (3.85)	32.5 (3.15)	39.00 (3.61)	34.30 (3.04)	30.00 (2.29)
	Year 2	29.33 (3.17)	27.00 (4.16)	35.66 (2.90)	40.66 (2.90)	33.66 (4.63)

Subsets of species in the community (species associated with mature forest and others) were examined separately for treatment effects. Based on Harris (1984) and Delasalla (1996) and others the following species were considered mature forest species: Varied Thrush, Townsend's Warbler, Red Crossbill, Pine Siskin, Red-breasted Nuthatch, Brown Creeper, Winter Wren, and Golden-crowned Kinglet. Other species were: Gray Jay, Olive-sided Flycatcher, Swainson's Thrush, Hermit Thrush, American Robin, Wilson's Warbler, Lincoln's Sparrow, Mountain Chickadee, Vesper Sparrow, White-crowned Sparrow, Yellow-rumped Warbler and Dark-eyed Junco. Mean mature forest species detections per census point were not significantly different between treatments and uncut plots prior to and after harvest (treatment by year interaction: $F=0.10$; $df=1,75$; $p=0.74$; Table 6). For other species there were significant year ($F=4.10$; $df=1,75$; $p=0.03$), treatment ($F=4.10$; $df=1,75$; $p=0.04$) and treatment by year interaction effects ($F=6.20$, $df=1,75$; $p=0.01$). Other species became more abundant in harvested areas after harvest.

Table 6. Mean abundance (SE) of mature forest associates and other species detected per point count stations in uncut and harvested treatments prior to- and after harvest.

	Uncut reserves n=22		Treatments n=55	
	Pre-treatment	Post-treatment	Pre-treatment	Post-treatment
Mature Forest associates	12.86 (0.67)	11.77 (0.59)	12.98 (0.36)	12.25 (0.44)
Others	7.04 (0.34)	8.77 (0.68)	7.4 (0.39)	12.54 (0.45)

At the individual treatment level, abundance of both mature forest and other species were not significantly different between harvested and uncut treatments one year post-harvest (mature forest spp. $F=2.05$; $df=4,8$; $p=0.17$, and others $F=1.09$; $df=4,8$; $p=0.42$) (Table 8). In the second year after harvest abundance of mature forest associates was also not significantly different between treatments. However, it is worth noting that both selection and the 10 ha treatment supported fewer mature forest associates than any other treatment for both years. Conversely, other species did show significant responses among treatments in the second year after harvest. Abundance was lowest in uncut (mean= 9.80), followed by selection and 0.1 ha treatments (means=15.70, and 16.10), and then 1.0 ha and 10 ha treatments (means= 24.22, and 28.07). However, significant differences were only detected between the uncut and the ten ha treatments (Tukey's HSD).

Changes in Abundance of Individual Species

Only eight species occurred at sufficiently high densities to conduct analysis on changes in relative abundance (Table 8.): Dark-eyed Junco, Golden-crowned Kinglet, Hermit Thrush, Pine Siskin, Red-breasted Nuthatch, Varied Thrush, Winter Wren and Yellow-rumped Warbler. However, I did not conduct analyses on Pine Siskin because, although common, this species was most commonly detected flying over the study area during the bird census.

Table 7. Mean (SE) detections per 3 point count stations of mature forest associates and other species at treatment units one and two years post-harvest.

	Treatments					Significance for Treatment effects
	Uncut	Selection	0.1 ha	1.0 ha	10 ha	
Mature forest Associates						
1996	20.16 (2.60)	16.16 (0.83)	19.83 (1.58)	18.00 (2.88)	12.80 (2.16)	F=2.00, p=0.17
1997	16.33 (2.40)	12.00 (1.32)	15.80 (2.45)	19.00 (1.04)	12.66 (0.72)	F=2.69; p=0.09
Others						
1996	13.67 (2.80)	16.33 (2.58)	19.33 (2.58)	16.33 (1.16)	17.16 (1.83)	F=0.95, p=0.47
1997	9.83 (0.72) A ^y	16.33 (4.17) AB	16.33 (2.84) AB	24.33 (2.33) AB	28.67 (5.60) B	F=4.53, p=0.02

^yMeans sharing the same letter across the table did not differ significantly.

Two of seven species showed significant responses to harvesting and among the individual treatments (Tables 8 and 9). Prior to harvest, relative abundance of golden-crowned kinglet appeared to be higher at treatment than in control sites although the differences were not statistically significant ($F=2.42$; $df=1,75$; $p=0.12$). However, after harvest, detections of golden-crowned kinglet declined significantly by more than two times at harvested plots (treatment by year interaction $F=10.30$; $df=1, 74$; $p=0.002$). At the level of individual treatments, golden-crowned kinglet detections were significantly different between treatments ($F=6.54$; $df=4,10$; $p=0.007$). Golden crowned kinglet were significantly more abundant in uncut plots compared to the 10ha and selection harvest treatments (Tukey's HSD $\alpha=0.05$) (Table 9, Fig. 5).

Dark-eyed junco detections were not significantly different between treatment and uncut plots prior to harvest (Table 8). After harvest the number of detections in harvested plots increased significantly (treatment by year interaction: $F=7.23$; $df=1,74$; $p=0.009$), suggesting a positive response as a result of harvest. When detections were compared among individual treatments, there were significant year ($F=24.84$; $df=1,10$; $p=0.001$) and treatment ($F=6.70$; $df=4,10$; $p=0.007$) effects. Dark-eyed junco detections were significantly lower in uncut reserves than in any treatment (Tukey's HSD $\alpha=0.05$).

Although not significant, there were qualitative trends for some of the remaining species. Red-breasted nuthatch declined throughout the study area after harvest. At the individual treatment level, abundance was highest in the 1.0 ha opening treatments and lowest in the selection cut treatment. Hermit thrush increased in both uncut and harvested

areas after harvest but was least abundant in the uncut reserves. Conversely, varied thrush declined in abundance in both uncut and harvested plots after harvest. At the individual treatment level, varied thrush was most abundant in uncut treatments than in other treatments. Winter wren showed little change in abundance in all areas after the harvest, although they were most abundant in all treatments other than the uncut reserves. Finally, yellow-rumped warbler showed a response to harvesting similar than that of dark-eyed junco. After harvest, yellow-rumped warbler abundance declined in the uncut reserves and increased in the harvested areas. Among the individual treatments, yellow-rumped warbler was most abundant in the 10 ha treatment and least abundant in the selection cut treatment.

Table 8. Mean (1 SE) number of detections per census point of "core" songbird species in uncut and treatment areas prior to and after harvest. Number of detections was averaged for 2 years pre-harvest and 2 years post-harvest.

Species	Uncut n=22		Treatments n=54		Trend	Significance ^α
	Pre-harvest	Post-harvest	Pre-harvest	Post-harvest		
Golden-crowned kinglet*	1.01 (0.10)	1.09 (0.07)	1.24 (0.06)	0.63 (0.06)	-	F=10.3; p=0.009
Dark-eyed junco*	1.00 (0.10)	0.88 (0.14)	1.16 (0.08)	1.51 (0.10)	+	F=7.23; p=0.009
Hermit thrush	0.77 (0.09)	0.88 (0.15)	0.74 (0.08)	0.84 (0.08)	+ ns	F=0.04; p=0.834
Varied thrush	1.43 (0.12)	1.02 (0.15)	1.28 (0.06)	0.90 (0.07)	- ns	F=0.06; p=0.807
Winter wren	1.00 (0.09)	1.18 (0.13)	1.14 (0.06)	1.16 (0.09)	+ ns	F=1.76; p=0.188
Yellow-rumped warbler	0.77 (0.08)	0.66 (0.10)	0.56 (0.06)	0.79 (0.08)	- ns	F=0.76; p=0.380
Red-breasted nuthatch	0.97 (0.12)	0.59 (0.09)	0.87 (0.07)	0.69 (0.06)	- ns	F=1.91; p=0.171

* significant interaction effects (years vs. treatments) reflect responses to the harvest treatment. ^α All p values are for treatment by year interaction effects

Table 9. Mean detections (SE) per 3 point count stations of core species at individual treatment units

<i>Species</i>	Treatment			
	<i>Uncut</i>	<i>Select</i>	<i>0.1 ha</i>	<i>1.0 ha</i>
Golden-crowned kinglet	2.91 (0.20) A ^b	1.25 (0.33) B	1.66 (0.21) B	2.08 (0.23) B
Dark-eyed junco	2.75 (0.30) A	5.16 (0.94) B	6.16 (0.78) B	5.58 (0.41) B
Hermit thrush	2.16 (0.35)	2.67 (0.96)	2.58 (0.70)	2.41 (0.43)
Varied thrush	5.25 (0.69)	3.50 (0.61)	4.33 (0.90)	4.25 (1.04)
Yellow-rumped warbler	2.50 (0.75)	1.33 (0.30)	2.50 (0.50)	2.75 (0.64)
Winter wren	3.16 (0.60)	4.16 (0.82)	5.50 (0.76)	4.75 (0.44)
Red-breasted nuthatch	1.58 (0.41)	0.83 (0.24)	1.25 (0.47)	1.91 (0.30)
				1.41 (0.32)

^b Means sharing the same letters along each row did not differ significantly (Repeated Measures ANOVA, Tukey HSD test; $\alpha=0.05$.)

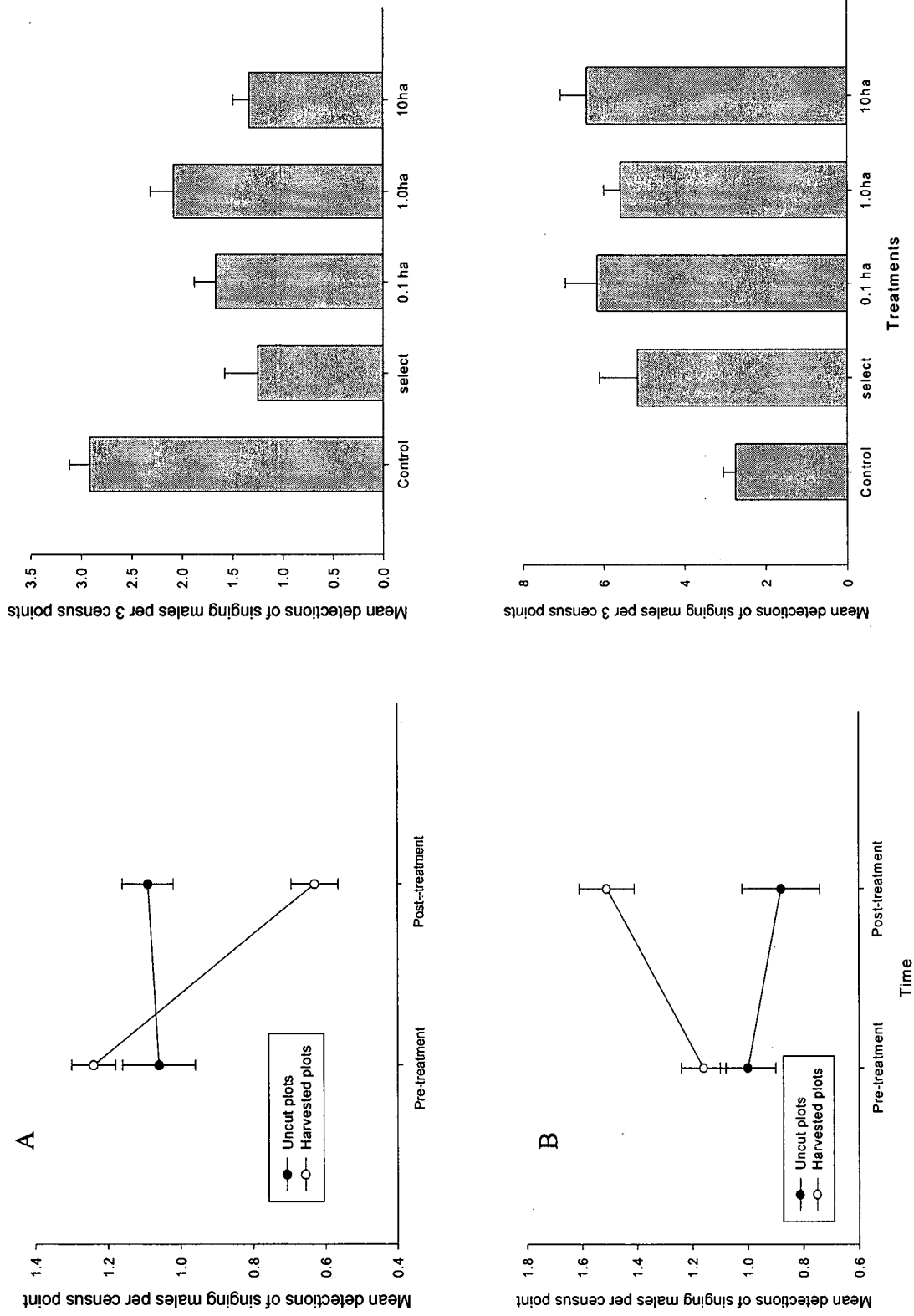


Figure 5. Example of two species A: golden-crowned Kinglet and B: Dark-eyed Junco that showed opposite responses to harvesting when examined prior to and after harvest, and their responses to the individual treatment levels following harvest.

DISCUSSION

The original songbird community inhabiting the Sicamous Creek Study area consisted of a relatively small number of dominant species and several rare ones. This pattern, and the community that inhabits these forests appears to be very similar to other conifer-dominated high elevation forests when compared to lower elevation forests (Sabo 1980). In British Columbia, Dickinson *et al* (*unpubl. report*), and Dickinson and Leupin (*unpubl. report*) reported similar community compositions in two studies within subalpine forests near Barriere, B.C. and Avola, B.C. respectively. Similarly, Davies *et al* (1999) identified similar songbird species composition within the ESSF biogeoclimatic zone in southern Cariboo region of British Columbia. Elsewhere in the Pacific Northwest, Wiens (1989) estimated that the average number of species inhabiting high elevation Rocky Mountain forests was 14. Similarly, Raphael (1987) in a review of studies conducted in high elevation forests in Colorado identified 14 species as being commonly reported in subalpine forests in the Central Rocky Mountains. In his paper, all species listed, with the exception of 1 species (Cassin's Finch) were also the species found at the Sicamous Creek Study site.

The responses of the songbird community to the various levels of perforation were subtle suggesting that the original songbird community inhabiting the Sicamous Creek research forest may have some resilience to the harvesting activities as has been observed in other experimental studies in the boreal forests of Alberta (Schmiegelow *et al* 1997). Dominant species that were originally present maintained dominance, although their representation changed in terms of rank. At Sicamous Creek, new species appeared in

relatively low numbers despite the presence of extensive clearcuts so that they did not play, at least in the two years following harvest, a large role in shaping the original songbird community. In similar studies in hardwood forests, Robinson and Robinson (1999), and Annand and Thompson (1997) attributed the lack of colonization by early seral stage-associated species in gapped forests to the small size (and consequently, unsuitability) of the new habitats created.

There are three plausible reasons for the observed results. In subalpine forests, large-scale disturbances (typically created by fires) normally are of a stand-replacing nature and result in dramatic changes to the songbird community, but these occur at infrequent (≈ 400 year) intervals (Veblen 1986). In high-elevation and subalpine forests, the factors mostly responsible for the structural make-up of these forests are guided by small-scale disturbances such as insect outbreaks and windfall (Parish *et al* 1999). It is thus possible that the original songbird community may be able to accommodate such disturbances without appreciable changes in the songbird community. Similar results have been obtained in other areas in the Pacific Northwest and boreal forests in Alberta (Chambers *et al* 1999, Steventon *et al* 1998, Schieck *et al* 1995, Schmiegelow *et al* 1997). These authors hypothesized that the lack of response by mature forest-associated songbirds in their areas was the result of the harvesting methods used which resembled the usual natural disturbance events of the area.

A second possible reason for the low level of response may be that the removal of 30% of the habitat was not sufficient to affect the original songbird community negatively

because the original forest may have been supporting populations below carrying capacity. Martin (2001) noted that high elevation habitats tend to support lower densities of organisms (including birds) than lower elevation habitats (see also Sabo and Holmes 1983). Therefore, if songbirds occurred at densities lower than the carrying capacity, the “displaced” individuals could occupy vacant territories within the remaining forest. Several studies have reported crowding effects in uncut forests after the surrounding forest had been harvested. In boreal forests of Alberta, Schmiegelow *et al* (1997) reported that bird abundance increased significantly in isolated fragments relative to controls one year after harvest suggesting such an accommodation or “crowding effect” may have occurred. However, species associated with mature forest at Sicamous Creek did not appear to show this “crowding effect”. The lack of “crowding” may imply a shortage of one or more critical resources in high elevation forests that maintain densities at lower levels than those observed at lower elevations.

Finally, the influx of species into the study area after harvest was negligible. The ten ha opening treatments supported a higher abundance of non-old growth species when compared to the uncut treatments. However there was a non-significant tendency for higher numbers of non-old growth species in larger canopy openings in the second year post-harvest. This could be accounted for by increases in species with broad habitat plasticity (i.e. species that can be found in a variety of habitat types) that were already present in the original forest. New species were detected at very low levels after harvest so that they did not play an important role in the reshaping the songbird community after harvest. In other studies, influx of new species has been documented following clearcut harvesting practices

and in forest remnants embedded in non-forest habitat (Thompson and Capen 1988, Rice *et al* 1984, Germaine *et al* 1997).

Increased abundance of predators and brood parasites has been implicated in impacting forest-dwelling songbirds negatively after harvesting (Robinson and Robinson 1999, Andren and Angelstram 1988). Gray Jay, the only predatory species in the area occurred at low densities and utilized areas that overlapped treatment units and showed no apparent response to the harvesting activities, suggesting that impacts of predation rates on the original songbird community by this species were probably similar to those that would occur at pre-harvest levels. Brown-headed cowbirds are common in lower elevations where agricultural lands make up a significant portion of the landscape. However, they were never detected within the study area or in extensive clearcuts adjacent to the site (T. Dickinson, *unpubl. data*).

Open country species, which were present in adjacent clearcuts did not colonize the newly created habitats (openings). After harvest, there was a dramatic decline in the original shrub and herb cover as a result of the harvest (Lloyd *et al* 1997). Secondary succession also did not occur within the first two years after harvest so that the openings may have been unsuitable for colonizing species. Robinson and Robinson (1999), and Annand and Thompson (1997) conducted similar studies addressing openings and their effects on forest interior birds. Both studies obtained similar results and attributed the lack of colonization of

early seral stage-associated species in forest gaps to the small size (and consequently, unsuitability) of the new habitat created.

Despite a lack of significant response by the community as a whole, the abundance of several species did respond negatively to the harvesting activities. The responses of these species can thus have serious implications in forest management practices where the objective is to retain the original species at similar abundances typical of the original forest. I predicted decreases in abundance of species commonly associated with mature- and old-growth forest interior. Four of the seven species common at the study site decreased in abundance. Golden-crowned kinglet, a species associated with high canopy closures (Galati 1991, Mannan and Meslow 1984), decreased in abundance at treatment areas after harvest. This result is consistent with studies elsewhere and in other forest types where consistent negative responses by this species to harvesting activities have been observed (i.e. Wetmore *et al* 1983; Delasalla *et al* 1996, Hutto *et al* 1993). More importantly and of potential management concern, abundance of this species decreased by a degree greater than the amount of wood removal suggesting that the remaining forest in harvested sites are unable to support densities typical of the undisturbed forest.

Other species like the dark-eyed junco responded positively to the harvesting activities. This species can be found in habitat types that range from old-growth forests to open habitats (Titterington *et al* 1979). Similar results were obtained by Wetmore *et al* (1985), where this species was more abundant in disturbed sites than in control areas

following harvest. In addition, the results also suggest that Dark-eyed junco were attracted by and crowded into the newly created habitats.

Finally, other species showed significant results at the study site level between years but not between treatments. Red-breasted nuthatch a secondary cavity nester, showed a significant decline while hermit thrush showed a increase after harvest. I consulted regional Breeding Bird Surveys for the two species that exhibited the year effects to determine if the results observed were likely the result of regional temporal declines or whether the observed responses were the result of harvesting pressures. The results of the BBS suggested that red-breasted nuthatch detections in the period of 1993-1997 declined (BC Trend -6.80 , $p=0.01$, $n=50$; Sauer *et al* 2001), suggesting that the observed declines may have been associated with regional changes rather than the harvesting activities at the study area. However, because this species is a year-round resident in British Columbia, the regional trends may in fact be a reflection of harvesting throughout the province. At Sicamous Creek, this species forages in family units over large tracts of forests and may have expanded territory sizes to accommodate the loss of foraging/nesting habitat available to them and thus excluded other families from the surrounding area. Given the fact that standing dead trees (nesting sites) were removed to comply with WCB regulations strengthens this assumption.

Hermit thrush showed a significant increase in both treatment and control areas. When compared to regional BBS trends, the species shows declining trends throughout British Columbia over the period of 1993-1997 (Trend estimate -6.57 ; $p=0.005$, $n=31$, Sauer

et al 2001), which suggests that increases observed at the study area were likely the result of changes in habitats at a local level that favored this species.

CONCLUSIONS

Overall, the removal of 30% of wood volume removal by various means of forest perforation had little short-term effects on the original songbird community at Sicamous Creek. Although we observed some significant negative and positive effects for some species and groups of species, the magnitude of these effects were comparatively small. In general, it appears that songbirds inhabiting high-elevation appeared to accommodate the openings created by alternative silvicultural treatments perhaps because these mimicked natural disturbances typically experienced in this ecosystem. However, when interpreting the generality of these results it is important to consider several points.

First, this study investigated only the short-term (1-2 yrs) effects of the first cutting cycle in this forest. The trends that lacked statistical significance in this study could accumulate for some of the species and become significant over time. In the intervening five if eight years post-treatment (1998-2002), observations on the same sites, 4 new species associated with open habitats have been observed at treatment sites (T. Dickinson, *pers. comm.*). Second, we used the number of detections of singing birds as the treatment response variable as opposed to more direct measures of bird productivity, such as pairing and nesting success. Abundance or density estimates can be misleading indicators of habitat quality

because the harvested forests may have been sub-optimal habitats supporting non-breeding males (Van Horne 1983). In such instances, presence of singing males could have been maintained through a "rescue" effect from adjacent undisturbed forests (Brawn and Robinson 1996). Easton and Martin (1998) investigated the effects of removal of deciduous vegetation in managed Interior Cedar-Hemlock forests near the Sicamous Creek study area and reported higher abundance of common species but lower nesting success. Third, the experimental treatments measured only one degree (30%) of tree volume removal. Conventional forestry practices in turn remove 80-90% of the tree volume (Easton and Martin 1998, Schmiegelow *et al* 1997). An increased amount of habitat removal may in fact incrementally reduce the value of the remaining forests and make them less suitable for forest interior species. Steventon *et al* (1998) concluded that although removal of 30% and 60% of the wood volume in forests near Hazelton, BC, using a combination of selection harvest and 0.1-0.5 ha openings provided some habitat for mature forest songbird species, the heavy removal began to show a shift to species typical of clearcuts. As a result, subsequent entries into high elevation forests harvested using alternative silvicultural methods to clearcutting may be highly detrimental to songbirds associated with mature forest and should be reconsidered.

Finally, the effects of forest perforation on the songbird community must be interpreted within the context of the surrounding landscape. Instances of increased brood parasitism and predation have been linked to the amount of regional forest cover (Robinson *et al* 1995). The Sicamous Creek Research Forest was located in a predominantly forested landscape, however, as the forest area is reduced, the creation of gaps can increase the risks

to forest interior birds to parasitism and predation. Nevertheless, if the management objectives of mature forest species are to be met, the first cutting cycle of alternative silvicultural treatments at low removal levels can provide for both timber extraction and the retention of structural characteristics important to songbirds inhabiting these forests.

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APPENDIX I. Sample data sheet for songbird sampling at Sicamous Creek

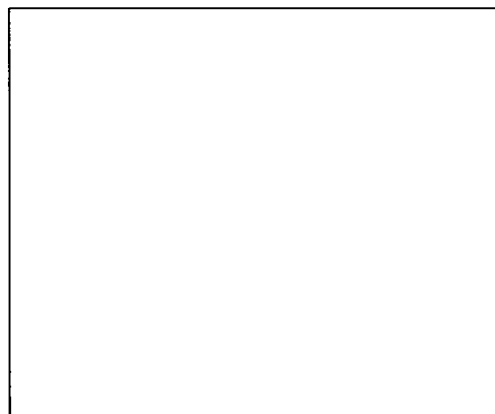
SICAMOUS CREEK SONGBIRD STUDY 1996

Date: _____ Weather: _____

Observer(s): _____ Start Time: _____

Transect #: _____ Census Point: _____ Census Period: _____

Site Description: _____



Species	Time	Distance	S/C/V	# of Indiv.	M/F	Comments

Comments: _____

