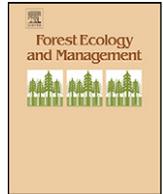




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Landscape-scale disturbance and boreal forest birds: Can large single-pass harvest approximate fires?

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ABSTRACT

Boreal forest birds have adapted to changes caused by natural disturbances such as fire and this adaptation forms the basis for the Natural Disturbance Paradigm (NDP) underlying recent proposed changes in forest harvesting practices in western Canada. To date, this paradigm has been evaluated primarily at the stand level and within conventional harvesting systems. The potential for improvements in avian conservation at the landscape scale by adopting the NDP approach is largely unknown. We examined the effects of landscape-scale disturbances on forest bird communities by contrasting richness and abundance of birds in (1) 16 single-pass harvest sites with residual forest patches, (2) 29 multi-pass harvest sites with residuals; and (3) 15 salvage-logged post-fire sites with variable harvest intensity. We contrasted bird communities in these treatments with those in unsalvaged post-fire sites of similar age. Post-fire sites were used to provide a metric of the Natural Range of Variation (NRV) to be expected in bird communities. Sites were surveyed for avian community composition and abundance 1–5 years post-disturbance. Redundancy analysis indicated that bird communities differed from the NRV in all of the harvest treatments. However, single-pass harvests provided a somewhat better fit to NRV than did multi-pass harvesting. Avian community similarity was influenced by non-linear responses to area harvested, amount of residual retention, residual composition and pre-disturbance forest composition. An optimization routine created from a General Linear Model, suggests that community similarity to NRV can be maximized by using single-pass harvests over multi-pass harvests, harvesting 66–88% of the timber in the planning unit, and retaining 5–19% of the disturbance area as live residual patches, with 50% of harvests having at least 9% of the area in residuals.

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1. Introduction

The forest industry of western Canada is moving toward ecosystem-based management approaches in an attempt to better conserve biodiversity (Boutin and Hebert, 2002). This approach is based on the concept that wildlife in the boreal forest has adapted to the frequency of large, stand-replacing, fires and other natural disturbances (Hunter, 1993). Such approximation of natural disturbance patterns provides intuitive appeal for the management of boreal forest wildlife (Bergeron et al., 2002; Hobson and Schieck, 1999; McRae et al., 2001). This approach has involved reducing the two or three-pass, clear-cut logging with cut- and leave-blocks toward partial-cut or variable-retention harvesting

involving leaving residual trees and patches within cutblocks (Schieck et al., 2000; Tittler et al., 2001). Comparisons made at the stand or patch scale have suggested that leaving more complex vegetation structure in cutblocks can maintain characteristics of old-growth forest in later generating stands used by boreal forest birds (Hobson and Schieck, 1999; Schieck and Hobson, 2000). However, maintaining components of vegetation and bird communities in harvests that are associated with those occurring in natural disturbances remains a challenge (Hobson and Schieck, 1999; Schieck and Hobson, 2000; Simon et al., 2002).

Another approach to approximating natural disturbance during harvest involves single-pass harvesting, whereby forest harvesting is done in one (shorter) time period and operations do not reoccur until the next rotation. This generally involves larger disturbances similar to those due to fire. This approach incorporates the variability of natural disturbance sizes currently not achieved in conventional harvesting practices. Most jurisdictions limit cut-block size to 150 ha or less, whereas fires may burn areas up to

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100s or 1000s km² (Bergeron et al., 2002; McRae et al., 2001). While creation of larger cutblocks might allow the forestry industry to decrease the “footprint” of roads and fragmentation (DeLong, 2002), other benefits have not been tested. Greater ranges in variation of remaining structure are generally seen in larger natural disturbances (Bergeron et al., 2002; Schulte and Niemi, 1999), and so larger cutblocks with residual patches may overlap with post-fire forests at least in terms of such structural attributes. Recent simulation studies suggest that harvest plans including retained linear strips and cutblock size restrictions generally lead to the weakest predictions in models of forest bird communities compared to other harvesting scenarios (Loehle et al., 2006; Rempel et al., 2007). In contrast, Rempel et al. (2007) found that harvesting plans that use larger harvest blocks and defer harvesting both spatially and temporally to produce large areas of similar aged forest show a lower degree of divergence from simulated NRV. To our knowledge, no field study has directly tested the landscape-scale consequences of single-pass style harvests for boreal forest bird communities.

Research supporting the use of natural disturbance emulation approaches to forest harvesting in the boreal as a means of better conserving birds and other wildlife is still lacking. In particular, large landscape-scale studies that compare multiple disturbance types have rarely been conducted within industrial forests in general (Song, 2002). To address these knowledge gaps, we focused on the effects of landscape-scale disturbance pattern on forest bird communities. Avian richness and abundance were contrasted between replicate landscapes of fire origin and single-pass (aggregated) harvesting with residuals; multi-pass harvesting with residuals; and salvage-logged post-fire landscapes with variable harvest rates. Based on a prior analysis of spatial

autocorrelation for several species within our study area, we considered “landscape” to encompass processes at the scale of >2.56 km², as this is approximately the scale over which most species showed autocorrelation in abundance (maximum extent of autocorrelation averaged 1.6 km; range 0.4–9.5 km). Our specific objectives were to assess how bird communities are influenced by each harvest type relative to fire and how amount and composition of residuals mediate forest bird community response to harvesting. We predicted that songbird community richness and abundance in treatments would be most similar to those in fire-generated landscapes in the following order: post-fire sites with salvage logging > single-pass post-harvest sites with residual standing structure left on sites > conventional multi-pass post-harvest sites. We hypothesized that single-pass harvesting and salvage logging would be more similar in avian composition and relative abundance to that in post-fire sites than in multi-pass harvesting due to greater variability in amount and composition of residual patches. Furthermore, we predicted that salvage-logged sites would be most similar to NRV since in addition to having harvest, the chemical disturbance caused by fire was also present, and therefore post-fire specialists should respond to any influx of deadwood-associated arthropods (Imbeau and Desrochers, 2002; Koivula and Schmiegelow, 2007).

2. Methods

2.1. Site description

The study area extended from Candle Lake, Saskatchewan in the east (53°50'N; 105°50'W) to the House River Fire in north-eastern Alberta (approximate location 56°44'N; 111°23'W; Fig. 1). This

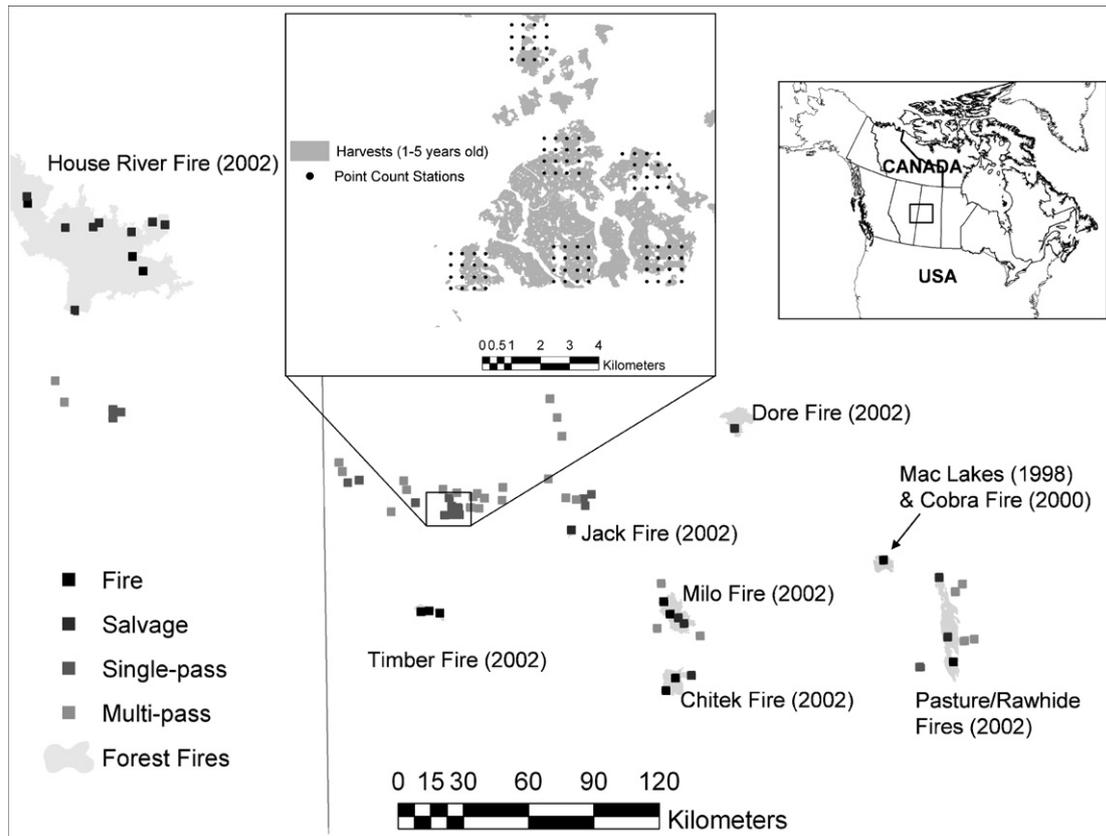


Fig. 1. Study design in the boreal forest of western Saskatchewan and eastern Alberta, Canada. Upper right, location of the study area in Canada. Large map: spatial arrangement of 72 plots across the study area by type of treatment; names and year burned are given for sampled forest fires. Inset (centre): example layout of point-count stations within plots at a single disturbance event; white background is undisturbed (mature) forest.

area was located within the Boreal Plain Ecozone (Acton et al., 1998) a gently rolling plain, covered by boreal mixedwood forest including Trembling Aspen (*Populus tremuloides*), Balsam Poplar (*P. balsamifera*), and White Birch (*Betula papyrifera*), and to a lesser extent Jack Pine (*Pinus banksiana*), White Spruce (*Picea glauca*), Black Spruce (*P. mariana*) and Balsam Fir (*Abies balsamea*) (Acton et al., 1998; Rowe, 1972). Successional patterns within the Boreal Plain are complex, with a heterogeneous mosaic of habitats created through multiple-stand disturbance patterns (Weir et al., 2000) which have led to some of the richest and most diverse bird communities in North America (Kirk et al., 1996).

2.2. Design

Site selection was limited by the number of existing aggregated harvests in our study area. We attempted to select disturbances in the other treatments that were as similar as possible in pre-disturbance vegetation to single-pass harvests. To avoid bias associated with succession, we first selected areas containing recent (<5 years) disturbances. A series of 400 ha GIS overlays were then used to select areas within recent disturbances having approximately 50% of the stands being rotation age (≥ 80 years old) and classified as hardwood- and/or hardwood- or softwood-dominated mixedwood stands. Since many of the fires we used had lower proportions of stands meeting our selection criteria, we selected a sub-sample 1.6 km \times 1.6 km (256 ha) plots that had composition closest to our selection criteria. The 256 ha plots were then treated as our sample units. Based on these plots, proportions meeting our selection criteria were reasonably similar between disturbance types (fires = 0.43 ± 0.22 S.D.; salvage = 0.41 ± 0.19 S.D.; single-pass = 0.51 ± 0.22 S.D.; multi-pass = 0.49 ± 0.29 S.D.). Plots in the salvage logged, single-pass and multi-pass harvesting treatments were selected so that similar levels of harvesting occurred between treatments. For multi-pass harvests, we restricted sampling to include areas that had recent harvest; however, these included 17 plots in which the second pass was completed (i.e. had at least one cutblock greater than 5 years older than the most recent harvest). No multi-pass harvest had the final pass completed. Attempts were also made to cover similar spatial gradients across all treatments (Fig. 1).

Seventy-two plots were surveyed over the summer breeding seasons (31 May–3 July) of 2003–2005 (Fig. 1). These plots consisted of 12 post-fire, 15 post-salvage harvest, 16 single-pass, and 29 multi-pass harvest plots. Twelve post-fire control plots were located in areas burned in 1998 (Cobra Fire), 2002 (Chitek Fire, House River Fire, Timber Fire, Milo Fire) and 2003 (Pasture/Rawhide Fire). Fifteen post-fire salvage plots were surveyed in the eight separate fires (Chitek Fire, Dore Fire, House River Fire, Timber Fire, Milo Fire, Pasture/Rawhide Fire). Single-pass plots were distributed among three large-scale (1200–2700 ha) and five small-scale (250–400 ha) aggregated harvests.

2.3. Avian sampling

Within each plot, a systematic grid of 16 sampling stations, each separated by 400 m in a four by four station configuration, was established. In cases where there was a physical barrier or a station landed in water, a replacement station was selected outside of the square plot boundary (e.g., see inset of Fig. 1). Systematic grids included both the disturbed and “green” island portions of the landscape as well as adjacent leave areas, and therefore covered any habitat type available within the plots.

Avian community composition and abundance was sampled at each station. Surveys were conducted using early morning point counts (~04:00–09:00 h) at each station based on the Indice Ponctual D'Abondance technique of Blondel et al. (1970). At each

station, all birds heard or seen during a 10-min count period were recorded and coded to indicate whether they were detected visually or by song. To augment sample size, surveys were conducted using digital recordings made with a stereo configuration bio-acoustic monitoring kit (Hobson et al., 2002). This technique allows sufficient spatial information for skilled observers to estimate abundance (Hobson et al., 2002). Digital recordings were later transcribed by four highly skilled observers. To allow comparison of field counts and digital recordings, counts were of unlimited distance. Ten-min counts were coded into three subset time periods (first 3 min, 3–5 min, and the last 5 min), to allow modelling and correction for detection probabilities (Farnsworth et al., 2002).

2.4. Vegetation variables

In order to determine vegetation patterns at the plot scale, we collated vegetation variables using GIS analysis of digital cover maps. We standardized Forest Inventory data into 12 cover types based on proportions of hardwood and softwood cover, as well as soil moisture (Table 1). Estimated stand ages were reclassified into seven age-classes. Estimates of pre-disturbance area covered by hardwood, softwood, hardwood-dominated mixedwood, softwood-dominated mixedwood and non-merchantable habitats were generated for each plot using ArcGIS v.9.1 (ESRI, Redlands, CA).

Remnant forest islands smaller than the resolution of forest inventory data (~2–4 ha depending on FMA) were mapped by using Garmin 12XL handheld GPS receivers and position-averaging of all apex coordinates. To ensure acceptable precision, data collection was limited to times when 3D navigation was available and the satellite constellation provided a (GPS) estimated horizontal accuracy of ≤ 7 m (Holden et al., 2002). Residual patches in six disturbances were digitized from IRS satellite imagery (5 m panchromatic). For a subset of nine patches mapped by both IRS image and GPS survey, there was no difference in patch area, perimeter or centroid coordinates (unpublished data). In the case of the House River Fire, however, a classified Landsat 7 image of burn intensity was used to categorize residual patches. The Landsat 7 image was classified by Ducks Unlimited Canada (Edmonton)

Table 1

Standardized habitat classes across different forest inventories within our study area

Habitat description	Criteria	Dominant species
Hardwoods	$\geq 75\%$ Hardwood	Aspen, Birch or Balsam Poplar
Mixed hardwood	50–75% Hardwood	Aspen, Birch or Balsam Poplar
Mixed softwood	50–75% Softwood	White spruce or Balsam Fir
Softwoods	$\geq 75\%$ Softwood	White spruce or Balsam Fir
Other HS	50–75% Hardwood	Black spruce, Jack pine or Tamarack
Other mixed	50–75% Softwood	Black spruce, Jack pine or Tamarack
Softwood	$\geq 75\%$ Softwood	Black spruce, Jack pine or Tamarack
Other conifer	$\geq 75\%$ Softwood	Black spruce, Jack pine or Tamarack
Bog/muskeg	Wet or flooded soils	Black spruce
Fen	Wet or flooded soils	Tamarack
Shrubby swamp	Tall shrubs on moist soils	
Open water		
Meadow	Grass or Sedge dominant	
Age class 1	Forest <2 years	
Age class 2	Forest 2–5 years	
Age class 3	Forest 5–30 years	
Age class 4	Forest 30–60 years	
Age class 5	Forest 60–80 years	
Age class 6	Forest 80–100 years	
Age class 7	Forest >100 years	

into three burn intensity scores (high, medium, and low) based on ground survey data; however, we reclassified this into a binary variable, categorizing the low burn intensity cells (defined as >95% canopy survival) as residual, and all other cells were classified as destroyed by fire. Overlay procedures in ArcGIS were used to generate total area of residuals per plot, and extract forest inventory variables associated with the residual patches (Table 1).

2.5. Statistical analysis

We performed Principal Components Analysis (PCA) to create indices of habitat composition to use as covariates to control for differences in pre-disturbance vegetation in our analyses of bird abundance (Lichstein et al., 2002). A separate PCA was conducted on post-disturbance vegetation to summarize the composition of residual patches. PCA was conducted using PC-ORD v.4.0 (MjM Software, Gleneden Beach, Oregon) and axes were selected based on the broken-stick criterion.

Prior to analysis, point-count data were corrected for detection probabilities (Farnsworth et al., 2002). We used Huggins' closed-capture models in program MARK (White and Burnham, 1999) via the RMark interface (Laake and Rexstad, 2005) to estimate detection probabilities as a linear function of $\ln(\text{effort})$, time of day, day of the month, proportion of the point-count station disturbed (within 100 m), shrub cover (high vs. low), whether the count was done by point count or recording, and observer experience (Van Wilgenburg and Hobson, unpublished).

After correcting the avian data for differences in detection probabilities among species, we derived a data matrix consisting of 27,061 individuals representing 141 species (Appendix A). A subset of 24 plots had repeat (between-year) visits. In order to treat all sites similarly (i.e. as single visit), we calculated maximum abundance across years for sites with repeat visit data, since maximum abundance tends to provide a better reflection of true abundance (Toms et al., 2006).

Prior to multivariate ordination of the avian data, all species occurring on fewer than 4 plots were removed from the data, leaving 26,939 birds representing 104 species. Plot-level bird abundance (i.e. sum of counts across 16 stations) was calculated for each species and used as the response variable. A preliminary ordination suggested a linear model was appropriate (ter Braak and Smilauer, 2002); so we used Redundancy Analysis (RDA) to model species composition. Avian data were square-root transformed to decrease the influence of extremely abundant species and standardized to zero mean and unit variance. Symmetric scaling was used since this method provides a good portrayal of both species and samples (Gabriel, 2002).

A preliminary RDA model with all explanatory variables was created and forward selection (tested using Monte Carlo permutations) was used to select explanatory variables and check for multicollinearity. Non-significant or collinear variables were removed from the candidate set for subsequent models. Spatial trends in abundance of species across the study area, were accounted for by including latitude (Y) and longitude (X) coordinates in RDA analyses, with coordinates standardized to a mean of zero and unit variance (Legendre and Legendre, 1998). Based on this analysis, we reduced the variables to include area harvested, area harvested squared, area harvested cubed, area disturbed, area disturbed squared, area of residuals, RComp1, area of residuals * RComp1, as well as X and Y coordinates and an interaction between X and Y.

To separate effects due to disturbance versus those due to geographic trends, we created five separate models using the reduced variable suite (above), and used variance partitioning to examine relative contribution of spatial- versus disturbance-related effects on community composition and abundance

(Borcard et al., 1992). First, a model was created (hereafter the Full model) using all explanatory variables and having none of the variables treated as covariates. Four subsequent models were created, varying which factors were included or treated as explanatory versus co-variables. In this fashion, it was possible to assign proportions of the variance uniquely attributable to a particular suite of variables (Borcard et al., 1992).

Statistical comparison of community composition between pairwise combinations of disturbance type were made via Monte Carlo permutations of ordination scores from partial RDAs, using dummy variable coding for disturbance type, and treating pre-disturbance forest composition, X and Y coordinates and an $X * Y$ interaction as covariates. Ordinations were conducted using Canoco for Windows v.4.5 (ter Braak and Smilauer, 2002). Ordinations were tested using Monte Carlo tests of significance (499 permutations each). Statistical significance was assessed at $\alpha = 0.1$ and corrections for multiple comparisons were made using the False Discovery Rate control (Benjamini and Hochberg, 1995). In order to display a common suite of species in resulting ordination diagrams, we displayed only those species differing significantly between treatments on the basis of Indicator Species Analysis (see below).

2.6. Optimization of community similarity

To assess the range of habitat attributes that could minimize community dissimilarity of harvest (excluding salvage) sites relative to NRV, we used Monte Carlo simulations to create an optimization routine using the YASAI add-in for Microsoft Excel™ (Eckstein and Riedmueller, 2002). We calculated Sorensen's Dissimilarity Index between all pairwise comparisons of post-fire plots (unsalvaged) and the single-pass and multi-pass sites using PC-ORD (McCune and Mefford, 2006), and averaged these for each plot. We then used General Linear Models to create a regression function to predict average community dissimilarity relative to post-fire plots. Prior to analysis, the data were logit transformed to keep predicted values within a 0–1 range. Sixteen models to explain community dissimilarity were considered, and we selected between competing models using AICc (Burnham and Anderson, 1998). Monte Carlo simulations were then created to seek the harvest type, and values for the proportion of landscape harvested, proportion of harvest left as residuals, and proportion of hardwoods left in the post-harvest landscape that minimized predicted community dissimilarity. We created 100,000 simulated landscapes to match the means and approximate distributions of habitat variables determined from a GIS query of the entire forest inventory for a 1,237,090 ha Forest Management Agreement Area (FMA) in our study area. Based on these queries, proportion harvested was simulated with a normal distribution (mean = 0.41 and S.D. = 0.17); proportion of the landscape composed of Hardwoods was normally distributed (mean = 0.47 and S.D. = 0.23), proportion of the harvest left as residuals was distributed as a combined Poisson/negative exponential random variable with mean = 0.0699, and area of hardwoods post-harvest was simulated as a negative-exponential variable with a mean of 9.92 ha.

From the 100,000 randomly simulated landscapes, we selected landscapes in the 5th percentile of simulated community dissimilarity values. For these landscapes, we then calculated percentiles for the habitat attributes associated with the sites to report as targets for minimizing community dissimilarity relative to NRV.

2.7. Species and guild responses

Differences in species frequency of occurrence and abundance were compared for all 104 species using Indicator Species Analysis (Dufrene and Legendre, 1997). Indicator values combine both the

species abundance and frequency of occurrence in each habitat and is a “distribution-free” method relying on Monte Carlo permutations to determine if the indicator value for a species is greater than expected by random occurrence between habitats. We conducted pairwise comparisons between fire versus salvage, fire versus single-pass and fire versus multi-pass harvests. Statistical significance was assessed using 1000 permutations, and results were considered significant at $\alpha = 0.05$. Indicator Species Analysis was conducted using PC-ORD v.4.0.

We examined factors influencing the abundance of cavity, canopy, ground and shrub nesters using General Linear Models (GLMs). Examination of correlations and scatterplots suggested no relationship between abundance of the canopy nesting or cavity nesting guilds and spatial variables or pre-disturbance vegetation and so those variables were not included in candidate models. GLMs incorporated treatment, area harvested, area of residuals, area harvested squared to account for potential non-linear responses to area harvested, and interactions between treatment and area harvested as well as area of residuals and area harvested. Six candidate models were considered and top models were selected using AICc (Burnham and Anderson, 1998). Scatterplots suggested both ground- and shrub-nester abundance was influenced by pre-disturbance habitat; therefore all models included PC1, and an additional model that included a treatment by PC1 interaction. We only considered the models within four AIC units of the top model as potentially useful (Burnham and Anderson, 1998).

3. Results

3.1. Habitat composition

Habitat composition of pre-disturbance vegetation characteristics was reduced to a set of simplified metrics using PCA. The first three axes explained 39.8% of the variation in the vegetation variables, with axes 1–3 explaining 15.6, 13.2 and 11% of the variance, respectively. Axis 2 did not differ between treatments and was therefore not retained. PC1 was negatively associated with proportion of the plot dominated by hardwoods (eigenvalue = -0.487), and positively associated with proportion of the plot dominated by other HS stands (eigenvalue = 0.300), other SH stands (eigenvalue = 0.312), and other Conifer stands (eigenvalue = 0.430). PC3 was negatively associated with HS stands (eigenvalue = -0.312), other HS stand area (eigenvalue = -0.301) and positively associated with the shrubby swamps (eigenvalue = 0.433), water (eigenvalue = 0.401), and meadows (eigenvalue = 0.169).

PCA was also used to summarize composition of residual patches. Based on the broken-stick criterion, the first two axes were selected to represent residual-patch composition. These two axes explained 44.6% of the variance in residual patch composition, with 26.9% and 17.7% of the variation being captured by PC1 and PC2, respectively. Residual Composition PC1 (hereafter RComp1) was negatively correlated with area of hardwoods (eigenvalue = -0.417), other conifers (eigenvalue = -0.454), other HS mixedwoods (eigenvalue = -0.343) and other SH mixedwoods (eigenvalue = -0.343). Residual composition PC2 (hereafter RComp2) was positively associated with non-forested habitats such as meadow (eigenvalue = 0.323) and water (eigenvalue = 0.355) and negatively associated with HS mixedwoods (eigenvalue = -0.437), SH mixedwoods (eigenvalue = -0.511) and softwood (eigenvalue = -0.455).

3.2. Avian community composition

An initial RDA model created using all explanatory variables accounted for 45% of the variance in the species data. After

removing non-significant and collinear variables, the variation in the species matrix accounted for by the Full model was 32.7% ($F = 2.166$, $p = 0.002$). The first four axes accounted for 21.7% of the variance in the species data, and 66.4% of the species–environment relationship. Pre-disturbance vegetation, spatial variables and treatment variables all influenced community structure (Fig. 2). Green harvests split from post-fire plots primarily on Axis 1, which was strongly correlated with lowland habitats (PC3) and total area disturbed, and negatively associated with area harvested (Fig. 2). Along Axis 2, samples split primarily based on spatial variables and amount of coniferous or mixedwood (i.e. PC1) habitat (Fig. 2). Single-pass harvests showed slightly more convergence with NRV on Axes 3 and 4, which were positively correlated with area harvested.

Variance partitioning showed that the greatest proportion (21.5%) of the variation uniquely associated with the explanatory variables was associated with the disturbance variables. A significant proportion (7.1%) was also attributable to broad-scale spatial trends. Some variance (4.0%), was shared variance, not separable between spatial and disturbance variables.

Of the 21.5% of variation attributable to disturbance-related environmental factors, 4.4% was related to the pre-disturbance vegetation. The remaining 17.1% was attributable to a quadratic response to total area disturbed (8.1%), a non-linear response to amount of area harvested (6.0%), and area and composition of residuals (3.1%). Increasing area of residual patches tended to increase similarity to NRV; however, interaction with RComp1 suggests residual patches composition influences the value of patch area. Re-running the analysis as a partial RDA, using spatial and pre-disturbance vegetation as covariates, suggested a slightly closer match of single-pass harvests than multi-pass harvest areas to post-fire sites, and greatest similarity between post-fire and post-fire salvage sites (Fig. 3). Community composition differed between all treatments except fire versus salvage (Table 2). This

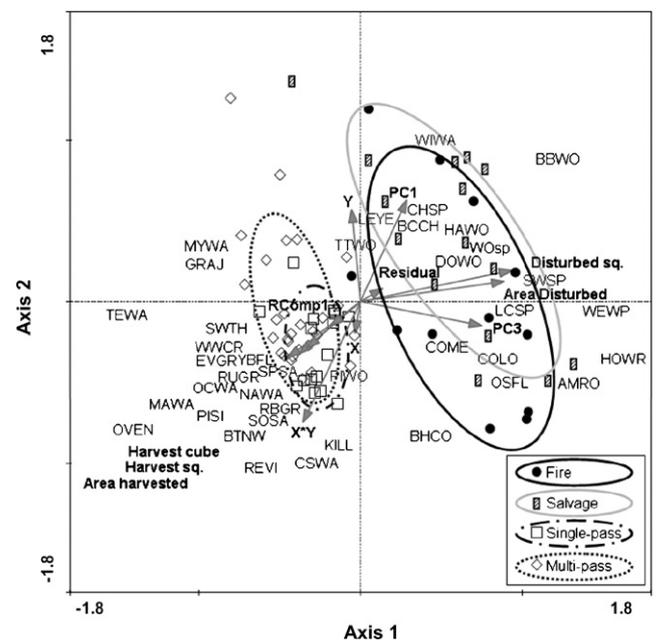


Fig. 2. Site level community composition (sample scores) in fire, salvage logged, single-pass harvest and multi-pass harvest sites inferred by RDA Axes 1 and 2 from the Full model. 67% confidence ellipses are portrayed for each treatment. PC1 and PC3 are multivariate metrics of pre-disturbance forest composition within the plots, RComp1 is a multivariate metric of residual patch composition, Y is latitude, and X is longitude, and interaction terms are given by an asterisk, Res * RComp1 is an interaction between residual patch area and composition; see text for details. Four letter codes represent species (see Appendix A).

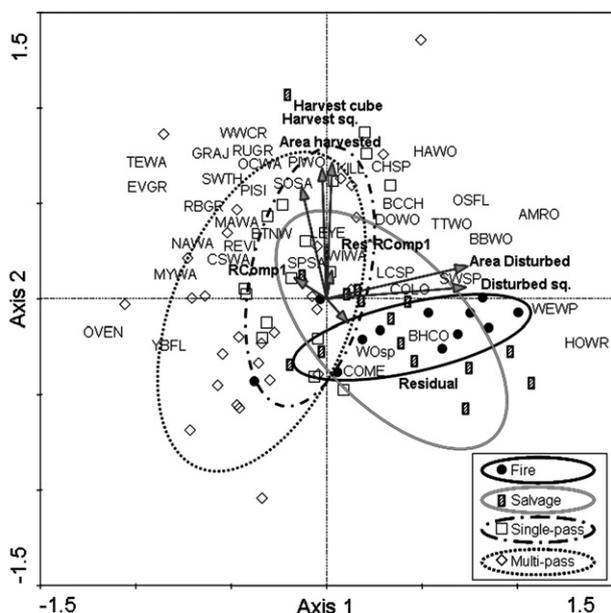


Fig. 3. Site level community composition (sample scores) in fire, salvage logged, single-pass harvest and multi-pass harvest sites inferred by RDA Axes 1 and 2 from partial Redundancy Analysis controlling for pre-disturbance vegetation and spatial trends. 67% confidence ellipses are portrayed for each treatment. RComp1 is a multivariate metric of residual patch composition, Res * RComp1 is an interaction between residual patch area and composition, see text for details. Four letter codes represent species (see Appendix A).

same pattern is true after correcting for multiple comparisons (Table 2).

3.3. Optimization of community similarity

The General Linear Models of (logit transformed) community dissimilarity had substantial model uncertainty, with no clear top model (Table 3); therefore, we used model averaging based on AIC model weights (from our AICc based model selection) to generate parameter estimates for our predictive equations, thus accounting for uncertainty in the parameter estimates due to uncertainty associated with model selection (Burnham and Anderson, 1998). The predictive equation based on the model averaged estimates explained 47% of the variance in the observed community similarity metrics. The simulations suggested that community dissimilarity to NRV could best be minimized using single-pass harvests versus multi-pass harvests, as 67% of the lowest dissimilarity simulations were classified as single-pass harvests. Furthermore, dissimilarity was minimized when 50% of simulated harvests removed 74% of the timber in the planning unit (range of 66–88% based on 5th–95th percentiles). In addition, leaving 5–19%

Table 2
Pairwise comparison of community composition between disturbance types

Comparison	n	Trace	F	p	FDR*
Fire vs. salvage	27	0.036	1.130	0.274	0.100
Fire vs. single-pass	28	0.060	1.873	0.006	0.050
Fire vs. multi-pass	41	0.051	2.376	0.002	0.017
Salvage vs. single-pass	31	0.049	1.777	0.006	0.067
Salvage vs. multi-pass	44	0.044	2.246	0.002	0.033
Single-pass vs. multi-pass	45	0.061	1.468	0.012	0.083

Comparisons were made using 499 Monte Carlo permutations of ordination scores from partial Redundancy Analysis controlling for pre-disturbance vegetation, latitude (Y), longitude (X) and an interaction between X and Y.
FDR = False Discovery Rate correction for multiple comparisons, statistical significance was assessed at the FDR value.

Table 3

Candidate models and model selection statistics from General Linear Models predicting logit transformed Sorensen's Community Dissimilarity to NRV for single-pass and multi-pass harvest plots

Model ^a	k	AICc	ΔAIC	ωi
Harv	3	-245.646	0.000	0.167
Harv + Resid + Harv * Resid	5	-245.301	0.345	0.140
Treat + PC1 + Harv + Resid + Treat * Harv	7	-245.288	0.359	0.139
Treat + Harv + Resid + Harv * Resid + PC1	7	-244.535	1.111	0.096
Treat + Harv + Resid + PC1	6	-244.000	1.647	0.073
Treat + Harv	4	-243.966	1.680	0.072
Treat + Harv + Resid + Treat * Harv	6	-243.649	1.997	0.061
Treat + Harv + Resid + Harv * Resid	6	-243.649	1.997	0.061
Treat + Harv + PC1	6	-243.301	2.345	0.052
Treat + PC1 + H + Harv + Resid + Treat * Harv	8	-243.288	2.359	0.051
Treat + Harv + Resid + Treat * Harv + Harv * Resid	7	-242.353	3.293	0.032
Treat + Harv + H + PC1	7	-241.649	3.997	0.023
Harv + Resid + H + H * Resid	6	-241.602	4.044	0.022
Treat + Harv + H	6	-239.966	5.680	0.010
Treat	3	-227.806	17.840	0.000
Treat + Resid	4	-225.806	19.840	0.000

k = number of parameters, AICc = Akaike's Information Criterion corrected for small sample size, and ωi = AIC weight. Sample size = 44.

^a Treat = treatment (single-pass or multi-pass), Harv = area harvested, Resid = area of residuals, H = area of hardwoods post-harvest.

of the disturbance area in live residual patches, with 50% having at least 9% of the area in residuals, and while 61–98% (median of 88%) of the pre-disturbance area of hardwoods were harvested also minimized community dissimilarity to NRV (Table 4).

3.4. Species and guild responses

Indicator Species Analysis showed that 39 out of 104 species had greater abundance and frequency of occurrence in at least one of the disturbance types (p < 0.05). Ten species had maximum indicator values in post-fire sites, eight in salvage-logged sites, 13 in single-pass sites, and eight in multi-pass sites. The majority of species with maximum indicator values in post-fire sites also had high indicator values in post-salvage sites, with seven being cavity nesters (Black-capped Chickadee, Common Merganser, Downy Woodpecker, Hairy Woodpecker, House Wren, Three-toed Woodpecker, and woodpecker spp.) known to be associated with fire-disturbed habitats (Fig. 4A). Similarly, three species (American Robin, Black-backed Woodpecker and Swamp Sparrow) had their highest indicator values in salvage-logged sites and had similar indicator values in post-fire sites and are known to be associated with fires; the other (Lesser Yellow legs) had an indicator value of zero for fires, and is likely a statistical artefact (Fig. 4B).

With the exception of Solitary Sandpiper, Spotted Sandpiper and Killdeer, the ten species with highest indicator values in single-pass harvest sites are typically associated with mature to late-seral aspen and white spruce stands (Fig. 4C). Species showing maximum abundance within multi-pass harvests were mostly associated with mature to late-seral mixedwood forests (Fig. 4D). Two species associated with shrubby growth, Nashville Warbler and Magnolia warbler, also had their highest indicator values in multi-pass harvests (Fig. 4D).

Cavity-nester abundance differed between treatments with post-fire plots (both salvaged and unsalvaged) having roughly 1.5 times the abundance of cavity nesters compared to the green harvest treatments. Mean cavity-nester abundance was 71.6 ± 11.7 (S.E.) birds/plot in post-fire, 68.8 ± 5.9 birds/plot in salvage logged, 44.8 ± 5.6 birds/plot in single-pass, and 41.7 ± 5.3 birds/plot in multi-pass sites. The top two models received 99.8% of the support, with the top model having ~58.6% of the support and the second-best model having ~41.1% of the support. The top model included treatment, area harvested and the interaction between treatment

Table 4

Habitat attributes of simulated landscapes with minimal (5th percentile of 100,000 simulations) predicted community dissimilarity relative to the Natural Range of Variation (NRV)

Harvest variable	Percentile						
	5th	10th	25th	50th	75th	90th	95th
Proportion of planning unit harvested	0.66	0.67	0.70	0.74	0.79	0.84	0.88
Proportion of harvest left as residuals	0.05	0.05	0.07	0.09	0.12	0.16	0.19
Proportion of hardwood harvested	0.98	0.97	0.94	0.88	0.79	0.70	0.61

and area harvested, while the second-best model was identical except for the addition of area of residuals. Model parameters and examination of scatter plots suggest that the interaction between treatment and area harvested captured the tendency of primary cavity nesters to increase in abundance with area harvested in single-pass and multi-pass harvests while showing no trend in the post-fire treatments. Composition of this guild differed between treatments, with 5.5–17.3% of cavity-nester abundance in post-fire and salvage-logged sites consisting of Black-backed and Three-toed woodpeckers, while these species made up 4.5–8.4% of cavity-nester abundance in single-pass sites and only 0.8–4.3% of the cavity-nester abundance in multi-pass harvests (see also Fig. 4). Mean combined abundance for Black-backed and Three-toed woodpeckers was 7.9 ± 1.6 (S.E.) birds in post-fire sites, 6.4 ± 1.3 in salvage-logged sites, 2.8 ± 1.4 in single-pass sites, and 1.4 ± 0.5 in multi-pass sites.

Canopy-nester abundance was highest in green-harvest sites, with 175.7 ± 16.3 birds/plot in single-pass sites, and 147.3 ± 10.0 birds/plot in multi-pass sites, whereas there were

121.9 ± 12.8 and 118.1 ± 8.6 birds/plot in post-fire and salvage-logged sites, respectively. All GLMs were within four AIC units.

All GLM models for ground-nester abundance were within four AIC units. To avoid vegetation bias influencing inference, we used model averaging (based on AIC weights) to generate a predictive equation to estimate treatment effects while holding vegetation PC1 constant. The estimates suggest that ground-nester abundance was highest in single-pass harvests (172.1 ± 3.0 birds/plot), then multi-pass harvests (154.7 ± 2.0 birds/plot), salvage-logged sites (146.4 ± 1.1 birds/plot) and lowest in post-fire sites (139.8 ± 2.1 birds/plot).

Three top models for shrub-nester abundance received 95.1% of the support, with 55.1, 28.6 and 11.3% of the support coming from the first through third models, respectively. The top model included treatment, area harvested, pre-disturbance vegetation composition (PC1), and a treatment by area harvested interaction, while the second-best model was identical except for also including area of residuals. The third-best model included

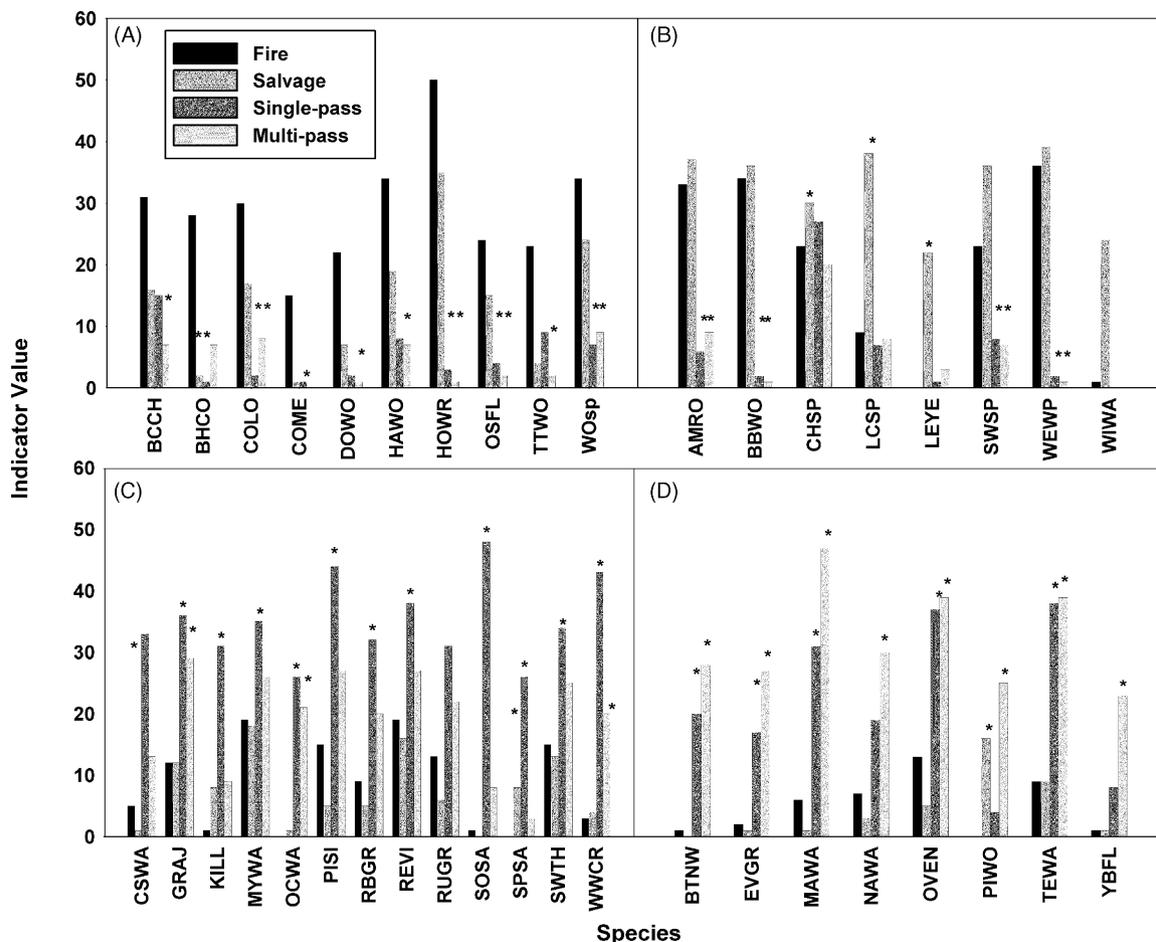


Fig. 4. Indicator values by treatment for species with maximum indicator values within (A) post-fire habitats, (B) salvage-logged habitats, (C) single-pass harvests, and (D) multi-pass harvests. *Significant ($p < 0.05$) pairwise planned contrast with post-fire habitats. See Appendix A for species codes.

treatment, area harvested, area of residuals, pre-disturbance vegetation composition (PC1), and an interaction between area of residuals and area harvested. Estimated abundance from model averaged estimates holding PC1 constant suggest that Shrub-nester abundance was highest in salvage-logged sites (20.8 ± 1.2 birds/plot), followed by post-fire sites (19.0 ± 1.2 birds/plot), multi-pass sites (16.5 ± 1.3 birds/plot), and lowest in single-pass harvests (16.1 ± 0.6 birds/plot).

4. Discussion

The occurrence of fire was the main driver of differences in avian community composition across the several disturbance types we examined. Avian community composition was most similar between single-pass and multi-pass sites (i.e. green sites) and differed considerably from salvaged and un-salvaged (post-fire) sites. Post-fire plots were primarily distinguished by cavity-nesting species, while single- and multi-pass harvests were more associated with open-habitat and shrub-nesting species. At a landscape level, single and multi-pass harvests tended to have more species associated with mixedwood and mature conifer habitats, due in part to leave patches adjacent to cutblocks. Furthermore, single-pass harvests had greater area of HS and SH mixedwoods in residual patches compared to fire (unpublished data). These general patterns are similar to those found at stand scales by previous studies that suggest post-fire bird communities are dominated by cavity nesters while post-harvest communities are dominated by ground- and shrub-nesting species and habitat generalists (Hobson and Schieck, 1999; Morissette et al., 2002; Schieck and Hobson, 2000; Schieck and Song, 2006; Simon et al., 2002) at least within 30 years of disturbance.

Previous research on avian communities in the boreal forest of western Canada suggests that species associated with late-seral and early post-fire habitats face the greatest conservation challenges related to forestry (Hobson and Schieck, 1999; Hannon and Drapeau, 2005; Morissette et al., 2002). This is due, in part, to targeted harvesting of post-rotation aged forest (Cumming and Diamond, 2002; Hobson and Bayne, 2000). However, several species inhabiting older age class stands also make use of early post-fire habitats, to take advantage of the pulse of deadwood-associated arthropods (Imbeau and Desrochers, 2002; Koivula and Schmiegelow, 2007). In addition, several species take advantage of the structural characteristics associated with early post-fire habitats (Hobson and Schieck, 1999; Schieck and Hobson, 2000; Schieck and Song, 2006; Simon et al., 2002). Given that it may take at least 30–60 years for harvested boreal forests to converge ecologically with post-fire forests (Hobson and Schieck, 1999; Schieck and Hobson, 2000; Schieck and Song, 2006), any improvements to approximating NRV earlier in succession is expected to have significant conservation value. In our study, we found evidence that avian community composition can be brought closer to that occurring in natural disturbance by the spatial and temporal aggregation of larger cutblocks with residuals than can be achieved by multi-pass harvesting with residuals. Multivariate ordinations showed differences in community composition between fire and single-pass harvests were less than between fire and multi-pass harvests, and pairwise comparison suggested that community composition of single-pass harvests were not a simple subset of multi-pass harvests. Similarly, results from our optimization favoured single-pass harvest over multi-pass harvests in 67% of the top simulations. Furthermore, cavity-nester abundance was closer to NRV in single-pass than multi-pass harvests. The cavity nester guild is of conservation concern because several of these species are considered post-fire specialists and so are sensitive to post-disturbance characteristics of sites.

Since primary cavity nesters serve a role in structuring subsequent bird communities by creating nesting opportunities (Martin and Eadie, 1999; Martin et al., 2004), the greater similarity in abundance of cavity nesters between fire and single-pass harvests may result in earlier ecological convergence of these disturbances. It remains unclear if the observed increase in similarity to NRV in single-pass harvests is sufficiently large to translate into earlier convergence with NRV. We expected greater effect sizes than observed, suggesting that perhaps convergence may not be any sooner than for other site treatments. Future research following these or other disturbances through time would be beneficial to determine if single-pass harvests converge with NRV sooner.

Our data generally fit the hypothesized pattern of community similarity to NRV, with salvage-logged sites showing greatest ecological similarity, single-pass harvests being second most similar, and multi-pass harvests least similar to post-fire sites. This pattern generally fit at both the community and species levels, with more species showing greater similarity to NRV in salvage-logged sites, single-pass harvests and multi-pass harvests in that general order. This, combined with the lack of overlap in community composition, suggests that while approximating disturbance patterns may bring harvests closer to NRV, pattern alone is not sufficient to cause harvest to converge with NRV. That salvage-logged sites were close to NRV, suggests that the chemical process of fire and associated standing dead material is crucial to the conservation of early post-fire bird communities, a conclusion similarly reached in several recent studies (Hobson and Schieck, 1999; Imbeau and Desrochers, 2002; Koivula and Schmiegelow, 2007; Schieck and Hobson, 2000; Schieck and Song, 2006). In particular, the cavity nesting guild was generally most abundant and occurred most frequently in post-fire sites followed by salvage-logged sites, single-pass harvests, and were generally least abundant in multi-pass harvests, however, individual species of cavity nesters had disparate responses. Species of the genus *Picoides* (Black-backed, Three-toed, Downy and Hairy woodpeckers) were all most abundant in post-fire sites, particularly un-salvaged sites, while Yellow-bellied sapsuckers and Northern flickers were most common in green harvests, particularly in single-pass harvests. Only Pileated Woodpecker was most common in multi-pass harvests, likely due to the presence of patches of old forest to be harvested in later passes. Conservation of most cavity-nesting species will therefore require maintaining post-fire habitats protected from salvage that are representative of a broad spectrum of burn severity. This result is consistent with those of other recent studies (Koivula and Schmiegelow, 2007; Schmiegelow et al., 2006).

Community composition was strongly influenced by pre-disturbance vegetation. While we attempted to control these effects by site selection, post-fire landscapes tended to be wetter and have more softwoods than green-harvest landscapes, as would be expected given the stand types targeted for forestry operations in our study area. Similarity to NRV in green-harvest landscapes was driven, in part, by non-linear responses to area harvested. Cubic polynomial terms for area harvested suggest that intermediate to high proportions of planning units in early succession increased similarity to NRV. This non-linearity was perhaps due to the trade off with residual retention, with larger disturbances tending to also contain more residuals. In our Full model, residual retention was positively correlated with Axis 1 of the bird community ordination. This trend was supported by our optimization models, which suggested community similarity was greatest when 66–88% of the planning unit was harvested with 5–19% of the area as residuals. To a lesser extent, greater proportions of the residual patches comprised of hardwoods and other mixedwood stand types contributed to green harvest similarity with NRV, as

Axis 1 was negatively associated with RComp1 which was in turn negatively associated with those habitats. Green harvests also tended to be most similar to post-fire landscapes that had less area disturbed by fire. While area disturbed is not necessarily a measure of fire severity, it would seem reasonable that green harvests are more similar to less severely disturbed landscapes. Little is currently known about the influence of burn severity on avian community composition, however, recent work suggests abundance of woodpeckers generally increase with fire severity (Koivula and Schmiegelow, 2007; Schmiegelow et al., 2006). If area disturbed by fire can be considered a surrogate for fire severity, then our results suggest green harvests are more similar to less-severe fire.

Logging road density is related to fragmentation of the regenerating landscape and previous simulation studies have shown that the spatio-temporal aggregation of cutblocks can reduce the footprint of roads (Delong, 2002). However, this might not be the case in mountainous terrain or where older harvests may not have constrained road networks (D'Eon, 2007). Forest bird abundance has previously been shown to display threshold responses to linear disturbances (Bayne et al., 2005). Future research quantifying differences in logging road densities between single- versus multi-pass harvesting scenarios and their influence on forest biota is encouraged.

Broader scale (>2.56 km²) differences in community composition and avian abundance could not be investigated with our current study design since several of the single-pass harvests were adjacent to planning units previously harvested in a multi-pass fashion. In theory, forest harvesting within a planning unit may influence the abundance and population dynamics of forest biota in adjacent planning units, particularly for forest interior specialists. Broad-scale aggregation of harvest units might decrease fragmentation effects on forest interior specialists in adjacent leave areas and planning units (e.g., Rempel et al., 2007). Replicated, adaptive management experiments where new planning units of similar forest composition (adjacent to un-harvested planning units) are assigned to either single or multi-pass harvest would be useful to determine the influence of harvest pattern on adjacent planning units and on bird abundance at planning unit or regional scales (>50 km²).

5. Conclusions

Our comparisons of how bird communities differ from patterns expected with NRV suggest that approximating disturbance pattern is insufficient to bring about ecological convergence with NRV in the early post-disturbance phase. Salvage-logged areas were ecologically more similar to fire than single-pass harvests, highlighting the role of the chemical disturbance and standing dead material caused by fire. If the natural disturbance approximation paradigm is to be taken any further as a means of preserving representative biodiversity and ecological function where harvesting occurs, the most fruitful avenue would therefore be experimentation with prescribed burning as a method of post-harvest site treatment, or some other means of approximating the chemical disturbance caused by fire. Future research should attempt to examine this as a potential management tool from both ecological and economic perspectives. However, our results suggest that moving toward single-pass, aggregated harvest plans in the boreal forest will provide a better fit with NRV than multi-pass harvests. Furthermore, our optimization routine provides the first quantitative targets for residual retention based on data other than landscape patterns. Aggregated harvest systems are often considered more cost effective due to harvesting and transportation efficiencies (Delong, 2002; D'Eon, 2007). Given the slight

improvement in approximating NRV and increased efficiencies, we suggest forest managers in the Boreal plain preferentially use single-pass harvests instead of multi-pass harvests, and use the percentiles from our optimization to set targets for and relative proportions of harvests receiving these attributes.

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Appendix A. Species names and AOU species codes

AOU code	Common name	Latin name
RNGR	Red-necked Grebe	<i>Podiceps grisegena</i>
PBGR	Pied-billed Grebe	<i>Podilymbus podiceps</i>
COLO	Common Loon	<i>Gavia immer</i>
RBGU	Ring-billed Gull	<i>Larus delawarensis</i>
FRGU	Franklin's Gull	<i>Larus pipixcan</i>
BLTE	Black Tern	<i>Chlidonias niger</i>
COME	Common Merganser	<i>Mergus merganser</i>
MALL	Mallard	<i>Anas platyrhynchos</i>
AMWI	American Wigeon	<i>Anas americana</i>
AGWT	Green-winged Teal	<i>Anas crecca</i>
BWTE	Blue-winged Teal	<i>Anas discors</i>
REDH	Redhead	<i>Aythya americana</i>
RNDU	Ring-necked Duck	<i>Aythya collaris</i>
COGO	Common Goldeneye	<i>Bucephala clangula</i>
BUFF	Bufflehead	<i>Bucephala albeola</i>
CAGO	Canada Goose	<i>Branta canadensis</i>
AMBI	American Bittern	<i>Botaurus lentiginosus</i>
SACR	Sandhill Crane	<i>Grus canadensis</i>
SORA	Sora	<i>Porzana carolina</i>
AMCO	American Coot	<i>Fulica americana</i>
COSN	Common Snipe	<i>Gallinago gallinago</i>
GRYE	Greater Yellowlegs	<i>Tringa melanoleuca</i>
LEYE	Lesser Yellowlegs	<i>Tringa flavipes</i>
SOSA	Solitary Sandpiper	<i>Tringa solitaria</i>
SPSA	Spotted Sandpiper	<i>Actitis macularia</i>
KILL	Killdeer	<i>Charadrius vociferus</i>
SPGR	Spruce Grouse	<i>Falciptennis canadensis</i>
RUGR	Ruffed Grouse	<i>Bonasa umbellus</i>
MODO	Mourning Dove	<i>Zenaidura macroura</i>
SSHA	Sharp-shinned Hawk	<i>Accipiter striatus</i>
COHA	Cooper's Hawk	<i>Accipiter cooperii</i>
RTHA	Red-tailed Hawk	<i>Buteo jamaicensis</i>
BWHA	Broad-winged Hawk	<i>Buteo platypterus</i>
BAEA	Bald Eagle	<i>Haliaeetus leucocephalus</i>

Appendix A (Continued)

AOU code	Common name	Latin name
MERL	Merlin	<i>Falco columbarius</i>
AMKE	American Kestrel	<i>Falco sparverius</i>
OSPR	Osprey	<i>Pandion haliaetus</i>
LEOW	Long-eared Owl	<i>Asio otus</i>
BAOW	Barred Owl	<i>Strix varia</i>
GGOW	Great Gray Owl	<i>Strix nebulosa</i>
NOHO	Northern Hawk Owl	<i>Surnia ulula</i>
BEKI	Belted Kingfisher	<i>Ceryle alcyon</i>
HAWO	Hairy Woodpecker	<i>Picoides villosus</i>
DOWO	Downy Woodpecker	<i>Picoides pubescens</i>
BBWO	Black-backed Woodpecker	<i>Picoides arcticus</i>
TTWO	Three-toed Woodpecker	<i>Picoides tridactylus</i>
YBSA	Yellow-bellied Sapsucker	<i>Sphyrapicus varius</i>
PIWO	Pileated Woodpecker	<i>Dryocopus pileatus</i>
NOFL	Northern Flicker	<i>Colaptes auratus</i>
CONI	Common Nighthawk	<i>Chordeiles minor</i>
RTHU	Ruby-throated Hummingbird	<i>Archilochus colubris</i>
EAKI	Eastern Kingbird	<i>Tyrannus tyrannus</i>
GCFL	Great Crested Flycatcher	<i>Myiarchus crinitus</i>
EAPH	Eastern Phoebe	<i>Sayornis phoebe</i>
OSFL	Olive-sided Flycatcher	<i>Contopus cooperi</i>
WEWP	Western Wood-Pewee	<i>Contopus sordidulus</i>
YBFL	Yellow-bellied Flycatcher	<i>Empidonax flaviventris</i>
WIFL	Willow Flycatcher	<i>Empidonax traillii</i>
ALFL	Alder Flycatcher	<i>Empidonax alnorum</i>
LEFL	Least Flycatcher	<i>Empidonax minimus</i>
BLJA	Blue Jay	<i>Cyanocitta cristata</i>
GRAJ	Gray Jay	<i>Perisoreus canadensis</i>
CORA	Common Raven	<i>Corvus corax</i>
AMCR	American Crow	<i>Corvus brachyrhynchus</i>
BHCO	Brown-headed Cowbird	<i>Molothrus ater</i>
YHBL	Yellow-headed Blackbird	<i>Xanthocephalus xanthocephalus</i>
RWBL	Red-winged Blackbird	<i>Agelaius phoeniceus</i>
WEME	Western Meadowlark	<i>Sturnella neglecta</i>
BAOR	Baltimore Oriole	<i>Icterus galbula</i>
RUBL	Rusty Blackbird	<i>Euphagus carolinus</i>
COGR	Common Grackle	<i>Quiscalus quiscula</i>
EVGR	Evening Grosbeak	<i>Coccothraustes vespertinus</i>
PIGR	Pine Grosbeak	<i>Pinicola enucleator</i>
PUIF	Purple Finch	<i>Carpodacus purpureus</i>
RECR	Red Crossbill	<i>Loxia curvirostra</i>
WWCR	White-winged Crossbill	<i>Loxia leucoptera</i>
AMGO	American Goldfinch	<i>Carduelis tristis</i>
PISI	Pine Siskin	<i>Carduelis pinus</i>
SAVS	Savannah Sparrow	<i>Passerculus sandwichensis</i>
LCSP	Le Conte's Sparrow	<i>Ammodramus leconteii</i>
STSP	Saltmarsh Sharp-tailed Sparrow	<i>Ammodramus caudacutus</i>
WCSP	White-crowned Sparrow	<i>Zonotrichia leucophrys</i>
WTSP	White-throated Sparrow	<i>Zonotrichia albicollis</i>
CHSP	Chipping Sparrow	<i>Spizella passerina</i>
CCSP	Clay-colored Sparrow	<i>Spizella pallida</i>
DEJU	Dark-eyed Junco	<i>Junco hyemalis</i>
SOSP	Song Sparrow	<i>Melospiza melodia</i>
LISP	Lincoln's Sparrow	<i>Melospiza lincolni</i>
SWSP	Swamp Sparrow	<i>Melospiza georgiana</i>
RBGR	Rose-breasted Grosbeak	<i>Pheucticus ludovicianus</i>
WETA	Western Tanager	<i>Piranga ludoviciana</i>
BARS	Barn Swallow	<i>Hirundo rustica</i>
TRES	Tree Swallow	<i>Tachycineta bicolor</i>
CEDW	Cedar Waxwing	<i>Bombycilla cedrorum</i>
REVI	Red-eyed Vireo	<i>Vireo olivaceus</i>
PHVI	Philadelphia Vireo	<i>Vireo philadelphicus</i>
WAVI	Warbling Vireo	<i>Vireo gilvus</i>
BHVI	Blue-headed Vireo	<i>Vireo solitarius</i>
BAWW	Black-and-white Warbler	<i>Mniotilta varia</i>
NAWA	Nashville Warbler	<i>Vermivora ruficapilla</i>
OCWA	Orange-crowned Warbler	<i>Vermivora celata</i>
TEWA	Tennessee Warbler	<i>Vermivora peregrina</i>
CMWA	Cape May Warbler	<i>Dendroica tigrina</i>
YWAR	Yellow Warbler	<i>Dendroica petechia</i>
MYWA	Yellow-rumped Warbler	<i>Dendroica coronata</i>
MAWA	Magnolia Warbler	<i>Dendroica magnolia</i>
CSWA	Chestnut-sided Warbler	<i>Dendroica pensylvanica</i>
BBWA	Bay-breasted Warbler	<i>Dendroica castanea</i>
BLBW	Blackburnian Warbler	<i>Dendroica fusca</i>
BTNW	Black-throated Green Warbler	<i>Dendroica virens</i>

Appendix A (Continued)

AOU code	Common name	Latin name
WPWA	Palm Warbler	<i>Dendroica palmarum</i>
OVEN	Ovenbird	<i>Seiurus aurocapillus</i>
NOWA	Northern Waterthrush	<i>Seiurus noveboracensis</i>
CONW	Connecticut Warbler	<i>Oporornis agilis</i>
MOWA	Mourning Warbler	<i>Oporornis philadelphia</i>
COYE	Common Yellowthroat	<i>Geothlypis trichas</i>
WIWA	Wilson's Warbler	<i>Wilsonia pusilla</i>
CAWA	Canada Warbler	<i>Wilsonia canadensis</i>
AMRE	American Redstart	<i>Setophaga ruticilla</i>
HOWR	House Wren	<i>Troglodytes aedon</i>
WIWR	Winter Wren	<i>Troglodytes troglodytes</i>
MAWR	Marsh Wren	<i>Cistothorus palustris</i>
BRCR	Brown Creeper	<i>Certhia americana</i>
WBNU	White-breasted Nuthatch	<i>Sitta carolinensis</i>
RBNU	Red-breasted Nuthatch	<i>Sitta canadensis</i>
BCCH	Black-capped Chickadee	<i>Poecile atricapillus</i>
BOCH	Boreal Chickadee	<i>Poecile hudsonicus</i>
GCKI	Golden-crowned Kinglet	<i>Regulus satrapa</i>
RCKI	Ruby-crowned Kinglet	<i>Regulus calendula</i>
VEER	Veery	<i>Catharus fuscescens</i>
SWTH	Swainson's Thrush	<i>Catharus ustulatus</i>
HETH	Hermit Thrush	<i>Catharus guttatus</i>
AMRO	American Robin	<i>Turdus migratorius</i>
MOBL	Mountain Bluebird	<i>Sialia currucoides</i>
BBsp	Un-identified blackbird	
THsp	Un-identified thrush	
Vlsp	Un-identified vireo	
Gull	Un-identified gull	
Dusp	Un-identified duck	
MERG	Un-identified merganser	
W0sp	Un-identified woodpecker	

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