EFFECT OF BOREAL FOREST DISTURBANCE DUE TO LOGGING AT DIFFERENT SPATIAL SCALES ON MIGRATORY SONGBIRDS.

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by

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ABSTRACT

Boreal birds have experienced population declines that may be related to alteration of the forest at a range of scales. Understanding how resource extraction may affect the distribution and abundance of species is critical to address conservation policy in the boreal forest region. This study aims to understand how habitat alteration by logging influences the abundance and habitat choices of a migratory songbird, the Canada Warbler (CAWA; *Cardellina canadensis*) in its Canadian breeding range and more specifically within the northwestern region of Ontario, where there is little information about this species at risk. I assess whether there exists a different response in the abundance of upland migratory songbirds to logging disturbance at different scales. Also, I assessed the "habitat compensation hypothesis," which states that some species can substitute their primary habitat for other alternative and less preferred habitats on the landscape. I conduct a meta-analysis of 21 studies to identify the effects of habitat alteration on a relative abundance index (RAI) of 21 upland songbird species, comparing logged to unlogged sites along the southern border of Canadian boreal forest. Using generalized linear mixed models (GLMM), I model the RAI incorporating two scales (local- and landscape-scale effects), time since logging, and forest type. Several species, including CAWA, are reported in decline in Canada. They occasionally have a higher mean RAI comparing logged areas at landscape scale than comparing at the finer local scale, suggesting that they occupy lower quality habitats in disturbed areas. The results are consistent with other findings: birds associated with old-growth forests are most sensitive to logging, as well as birds that nest on trees and those more associated with a coniferous forest. I then assess how time since logging affects CAWA occurrence and distribution in Northwestern Ontario. I use Maxent software to develop a predictive high-resolution (30 m) field-validated species distribution model (SDM). The SDM is built on occurrences (2001-2020) from diverse datasets, supplemented with field-collected data from 2021. The SDM's environmental covariates include years since last disturbance (usually by logging) among other ecological layers. The prediction of the final SDM indicates that CAWA has high association with riparian zones, areas with high shrub cover, and mixedwood forest. The probability of CAWA occurrence is high (>0.7) in undisturbed forest, as well as in areas where the forest was disturbed >6 years ago, indicating that CAWA may take advantage of regenerated forest depending on shrub density and retention of old-growth forest structure (tree canopy height >10 m). Finally, I test the influence of conspecific attraction and habitat disturbance on the occurrence and habitat choices of CAWA in logged landscapes. I use a survey of a gradient of disturbed logged landscapes and vegetation cues (shrub and canopy cover, canopy height, and forest type) conducted during three breeding seasons in Northwestern Ontario. The survey included naturally occurring aggregations of CAWA, as well as the results of simulated conspecific attraction using playback of CAWA. I select among general linear models that examine the influence of vegetation structure, level of disturbance, and conspecific song cues on the occurrence and habitat choices of CAWA. Shrub cover (>55%) is one of the main vegetation cues that influence habitat selection patterns. Canopy cover and canopy tree height also play a role, as well as disturbance at the local scale. Natural aggregations of CAWA are clustered in areas with higher rates of disturbance, and conspecific songs played during the pre-breeding season are apparently cues for males to settle while searching for breeding territories. Together, these results support the hypothesis that the CAWA selects habitat using vegetation cues, as well as conspecific cues, and that conspecific aggregation mainly occurs in logged areas. Because conspecific cues strongly influence the CAWA settlement, more research is needed to determine whether how different habitats where conspecifics aggregate relate to fitness. Conspecific cue could be used as a conservation tool to encourage this declining species to settle in areas corresponding to highest fitness.

CHAPTER 1. INTRODUCTION

Forest disturbance due to logging is a significant driver of habitat loss and degradation worldwide (Hansen et al., 2013). It can have major impacts on upland forest bird populations, including migratory songbirds (Lee et al., 2022). The decline of neotropical migratory bird populations has been a conservation concern for decades (Rappole and McDonald, 1994; Peterjohn et al., 1995, NABCI Canada, 2019). Numerous studies suggest that habitat loss, habitat degradation, and fragmentation in the breeding range are among the leading causes of the declines for long-distance migratory forest birds (Rappole and McDonald, 1994, Hutto and Young, 1999; Newton, 2004; Betts et al., 2020).

Forest birds, and more particularly long-distance migrants, are one of the groups declining most rapidly in North America (NABCI Canada, 2019, 2022; Lee et al., 2022). In the case of forest songbirds, a decline of approximately 31% occurred between 1970 and 2016 (NABCI Canada, 2019). Given that avifauna constitutes one of the best-monitored ecological indicators, declines in forest songbirds may represent just the beginning of similar or more significant losses in biodiversity generally (Rosenberg et al., 2019). Therefore, an integrated and holistic conservation approach (e.g., combining scientific research with policy development, community engagement and participation, etc.) is necessary due to extensive changes to habitat resulting from industrial activity and climate warming; these processes have affected and will continue to affect the future of the avian breeding ranges by shrinking them and/or shifting them to occupy areas of lower quality habitat (Cumming et al., 2010; Wells et al., 2018; Wells et al., 2020).

Logging activities vary in intensity, e.g., from partial harvest to salvage logging, from small-scale selective cuts to large clearcuts. Also, patterns of habitat use by different bird species differ depending on the spatial scale at which they are measured, and the habitat features that are important at one scale may not be important at another (Taylor and Krawchuk, 2005; Dalley et al., 2009). The spatial extent and intensity of logging can have diverse effects on songbird populations: on a broader scale, landscape level factors such as forest fragmentation, forest degradation, and climate patterns can significantly impact upland bird population distributions. Conservation efforts need to consider these larger spatial scales to address issues such as habitat loss or degradation and connectivity among landscapes (Cumming et al., 2010).

At the local scale (e.g., within forest stands), management will potentially improve the habitat for some species and degrade it for others (Hagan et al., 1997; Leston et al., 2018; Mahon et al., 2019). Local habitat features, and microclimates play a crucial role in the survival and reproductive success of upland forest birds. Nesting sites, food availability, and the presence of predators can vary at smaller spatial scales, determining breeding success of many bird populations (Ibarzabal and Desrochers, 2004; Tremblay et al., 2015). Thus, recognizing the significance of scale allows us to tailor conservation strategies that account for the specific needs and responses of different bird species (Dalley et al., 2009; Cumming et al., 2010).

The boreal forest is naturally patchy, and its configuration with respect to forest type and age changes continuously due to natural disturbances (e.g., wildfires, wind, pest outbreaks); as a result, forest loss is considered ephemeral (Wedeles and Sleep, 2008; Thompson et al., 2013). In addition, “habitat compensation hypothesis” states that species may “compensate” for loss or alteration of their primary habitat by using alternative ones as substitute (Norton et. al., 2000) However, changes to habitat because of anthropogenic disturbance or other causes can lead to habitat conditions with which birds cannot be productive (Betts et al., 2020). Changes in abundance at local and landscape scales can be contrasted to inform on whether a given species is able to occupy less preferred areas in alternative of its primary habitat (Norton et al., 2000).

Social interactions can also affect a species' density and spatial distribution (Campomizzi et al., 2008). A species does not always choose its habitat based on spatial and ecological patterns; individuals may be attracted by the presence of other individuals from the same species (conspecifics), and suitable habitats may be underutilized as a result of conspecific attraction (Hunt et al., 2017). Conspecific cues have been hypothesized to be surrogate indicators of habitat quality (Doligez et al., 2002), but reproductive success at some sites is low, and in some birds site fidelity is possible even among unsuccessful males (Campomizzi et al., 2012). Thus, conspecific cues are potentially unreliable indicators of habitat quality, especially early in the breeding season (Pulliam and Danielson, 1991).

Conspecific attraction has been observed to influence the habitat selection of migratory songbirds during their breeding season, e.g., in the house wren/*Troglodytes aedon* (Muller et al., 1997), least flycatcher/*Empidonax minimus* (Fletcher, 2009), and black-throated blue warbler/*Dendroica caerulescens* (Betts et al., 2008). However, whether these patterns extend to other migratory songbird species and whether the response to conspecifics is different between logged and unlogged landscapes remains poorly understood. In sum, the species-specific relationships of birds to their habitat, in both managed and unmanaged lands, need to be known at different spatial scales to develop comprehensive approaches to conservation (Westwood, 2016). Understanding how accelerating resource extraction development may affect the distribution and abundance of species is also critical to addressing conservation policy in the boreal forest region (Rudd et al., 2011).

1.1 Study species

The Canada Warbler (*Cardellina canadensis*, hereafter CAWA) is a neotropical migrant with a long migratory route, among the last warblers to arrive and the earliest to depart (Conway, 1999). The time CAWA spend each year in Canada is shorter than documented for any other of the wood warblers (CAWA stays average 62-72 days, whereas for many other warblers it is >80 days, Flockhart, 2007). Because of the limited time to select breeding territories, it has been hypothesized that male CAWA are influenced by the presence of other males (conspecific cues) as a shortcut to finding good habitat (Hunt et al., 2017).

CAWA has been declared a species at risk in Canada designated “threatened” in 2008, and the status was re-examined and designated as “special concern” in 2020 (Environment Canada, 2016; COSEWIC, 2020). The status owes to a substantial decline in abundance over the last half-century (Sauer et al., 2017; Wells et al., 2020, NABCI Canada, 2022). Causes of its decline are thought to include habitat loss, habitat alteration, and changes to forest successional patterns on the breeding grounds and wintering grounds (ESRD, 2010; Reitsma et al., 2010; COSEWIC, 2020). Wilson et al. (2018) report that the steepest declines have occurred in the southeastern portion of the species’ range, which includes areas adjacent to the Great Lakes in central Ontario and the Appalachian Mountains in the United States.

CAWA occurrence is commonly associated with old-growth forests (Bayne et al., 2016), but, depending on the region, breeding pairs consistently occupy forested areas with high densities of shrubs and small stems (Chace et al., 2009; Ball et al., 2016). Also, it can be associated with post-harvested areas of regenerating forest with dense, shrubby understory

(Becker et al., 2012; Flockhart et al., 2016; Hunt et al., 2017). Because CAWA is a ground nester, a dense understory is an important component of its breeding habitat (Goodnow and Reitsma 2011). Indeed, some have recommended timber harvest as a method for creating CAWA habitat (Hagan et al., 1997; Becker et al., 2012). However, Flockhart et al. (2016) found CAWA breeding success is low in recently logged forest. Thus, it is possible that CAWA is choosing recently logged forest due to its appearance as suitable breeding habitat (having high shrub cover), in part because nearby old-growth forest is limited in availability, making the logged areas a potential ‘ecological trap’ (Schlaepfer et al., 2002). An ecological trap occurs when a rapid change in the environment, often due to human disturbances, disrupts the signals that individuals rely on to evaluate habitat quality resulting in their misinterpretation of the true quality of the environment (Robertson and Hutto, 2006). An individual’s preferences prevail, but the outcome associated with a misleading cue to these preferences has negative fitness consequences (Misenhelter and Rotenberry, 2000).

CAWA breeding season and habitats are well described, but they vary regionally, particularly with different land use types and forest disturbances (Ball et al., 2016; Environment Canada, 2016; Westwood et al., 2019a). In Northwestern Ontario (NWO), little information about CAWA is available. Crosby et al. (2019) describe regional variation of this species generalizing differences among eastern, western, and central regions in Canada, using NWO as the central region. Due to lack of accessibility to many areas, there are also relatively few Breeding Bird Survey routes in NWO, and many are no longer in operation. The Ontario Breeding Bird Atlas also has few CAWA data for NWO. In sum, there are knowledge gaps on the response of CAWA to logging disturbance, related scale effects, and the social cues that

CAWA exhibits in its breeding territories in parts of the Canadian boreal forest (Flockhart et al., 2016; Hunt et al., 2017; Westwood et al., 2019a).

The aim of this dissertation is to describe how habitat alteration by logging influences the abundance and habitat choices of a migratory songbird species in its breeding range. The main goals of the associated research were: (1) to determine whether there exists a scale-dependent response in the abundance of migratory songbirds (including CAWA) to logging in the southern border of the Canadian boreal forest; (2) to assess whether forest disturbance influences the occurrence and distribution of Canada Warbler in Northwestern Ontario; and (3) to assess whether CAWA habitat choices and aggregation during the breeding season are influenced by landscape changes and playbacks of conspecific cues. In the next three chapters, different approaches were used to achieve the research objectives:

Chapter 2. Effects of habitat alteration due to logging on migratory songbirds' abundance in the Canadian boreal forest.

I assessed the variation in the relative abundance of 21 upland forest songbirds in relation to logging as a forest alteration. I reviewed studies across the southern border of the Canadian boreal forest, and I systematically analyzed its reports on upland migratory songbirds, where time since logging disturbance, comparisons to reference areas (unharvested or pre-harvested areas), and the scale of the study were available. I modeled a relative abundance index (RAI) comparing disturbed and reference sites and incorporating local- and landscape-scale effects, time since logging, forest type, and three bird traits (nest guild, habitat preference, and migratory strategy). I also built models for 15 species with sufficient data to assess which are the main predictors of RAI for those species.

Chapter 3. Field-validated species distribution model of Canada Warbler (*Cardellina canadensis*) in Northwestern Ontario.

I built an occurrence distribution model to predict the environmental factors influencing CAWA patterns in an Ecoregion of Northwestern Ontario. The SDM was built on occurrences (2001-2020) from diverse datasets supplemented with field-collected data. The majority of covariates were derived from high resolution (30 m) Landsat images. I assessed the accuracy of the model by doing a field-validation study collecting new observations in the field during the breeding season of 2022 guided by the preliminary model. I present the first-known field-validated SDM for CAWA.

Chapter 4. Influence of boreal forest disturbance and conspecific attraction on Canada Warbler (*Cardellina canadensis*) habitat choices during the breeding season.

I assessed the influence of vegetation and social cues on CAWA habitat choices and aggregation patterns in logged and unlogged landscapes. I surveyed a gradient of disturbed landscapes during three breeding seasons in Northwestern Ontario. I surveyed the occurrence of natural conspecific aggregations of CAWA, as well as conspecific attraction using an experiment with song playbacks in vacant sites during the pre-breeding season.

Through these chapters, I aim to provide and contribute to a comprehensive understanding of the repercussions of logging on CAWA and other migratory songbirds, shedding light on the broader implications for conservation and ecosystem management in the Canadian boreal forest.

CHAPTER 2. EFFECTS OF HABITAT ALTERATION DUE TO LOGGING ON MIGRATORY SONGBIRD ABUNDANCE IN THE CANADIAN BOREAL FOREST

2.1 ABSTRACT

It has been hypothesized boreal forest birds are able to adapt to logging disturbance due to their adaptation to the historical natural disturbance dynamic of this forest. However, boreal birds have experienced population declines that may be related to alteration of the forest and less old-growth forest availability at a range of scales. The southern portion of Canada's boreal forest has experienced increasing industrial development since 1950. I conducted a meta-analysis of 21 studies to identify the effects of habitat alteration on a relative abundance index (RAI) of 21 migratory songbird species, comparing logged to unlogged sites along the southern border of Canadian boreal forest. I modeled the RAI incorporating two scales (local- and landscape-scale effects), time since logging, forest type and three bird traits (nest guild: ground, shrub, canopy; habitat preference: young forest associated species, mature forest associated species, and generalists; and migratory strategy: long- and short to medium-distance migrants) using generalized linear mixed models. Overall, bird species traits and years since logging best explained the RAI. I also built models for 15 species, and for five of them, scale was the variable that best explained the RAI; for four, it was time since logging, and for three it was forest type. Several species, including CAWA, are reported in decline in Canada; they occasionally have a higher mean RAI comparing logged areas at landscape scale than comparing at the finer local scale, suggesting that they occupy lower quality habitats in disturbed areas. The results are consistent with other findings in relation to birds associated with old-growth forest; this group is most sensitive to logging, as well as the group of birds that nest in the tree canopy and those associated with a coniferous forest type.

2.2 INTRODUCTION

Approximately 70% of bird species that regularly breed in Canada's boreal forest biome experience impacts from one or several anthropogenic disturbances (road building, forestry, mining, etc.; Wells et al., 2020) in both breeding and wintering grounds (Rappole and McDonald, 1994; Hutto and Young 1999; Newton, 2004). Rosenberg et al. (2019) reported that 419 native migratory species experienced a net loss of 2.5 billion individuals in North America between 1970 and 2018, with boreal forest birds and long-distance migrators being two of the groups that experienced the highest decline. According to Breeding Bird Survey (BBS) data, several migratory songbird species experience a declining trend in their Canadian range (Ziolkowski et al., 2022; see examples in Appendix A, Figure A1, A2, A3). Some authors have found that negative effects of climate change on boreal bird species on the breeding grounds are accentuated by anthropogenic disturbance, with logging having the most influence on species abundance and distributions (e.g., Wells et al., 2018; Mahon et al., 2019; Cadieux et al., 2020). Given that the forestry industry creates a dominant disturbance in the Canadian boreal forest, more insight into the impact of logging on the abundance of migratory songbird species is necessary to inform management and conservation decisions.

The southern portion of Canada's boreal forest has experienced increasing industrial development in the last half-century (Figure 2.1). Over the previous decade, logging has played the main role after forest fires as the dominant disturbance in this region of the boreal forest, and the fastest changing landscapes are now dominated by smaller forest patch sizes, increased forest edge density, and decreased forest cover overall (Pasher et al., 2013; Pickell et al., 2016). Nontheless, NRCAN (2022) states that less than 1% of forest in Canada is logged each year, but the report does not consider the predominance of young forest and reduced availability of old-growth forest due to the activities of past logging (Betts et al., 2020). State of the Forest reports also include large expanses of northern forest that are inaccessible to logging. Concern about the impacts of logging on breeding birds in the southern parts of the boreal forest has increased in parallel with increase in the extent of logging (Niemi et al., 1998; Imbeau et al., 2001; Wells et al., 2020). Forestry practices can influence breeding songbirds in many ways: increased depredation of ground nests (Manolis et al., 2002; Ibarzabal and Desrochers, 2004), reducing foraging success in clear-cuts leading to reduced provisioning, and reduced chick survival (Tremblay et al., 2005).

Hobson et al. (2013) estimated a loss of migratory bird abundance in Canada (0.62 to 2.09 million individuals annually) as part of logging activity during the breeding season that directly destroys nests by tree removal. Clearcutting is the most common silvicultural practice in the boreal forest; regeneration following clearcutting, like that following forest fires, commonly results in forest with an even-aged structure and less structure complexity within stands (Le Blanc et al., 2010). However, at a landscape scale, clearcutting creates patch size distributions and proportions of young forest versus old forest stages which are different from those that arise following forest fires, with the major difference being smaller areas of older seral stages (Kuuluvainen and Gauthier, 2018). The changes to habitat associated with anthropogenic disturbance come with sufficient modifications to both the composition and configuration of landscapes that they likely intensify the decline or shift the distribution of forest birds due to climate warming (Wells et al., 2018; Betts et al., 2017, 2020; Bouderbala et al., 2023).

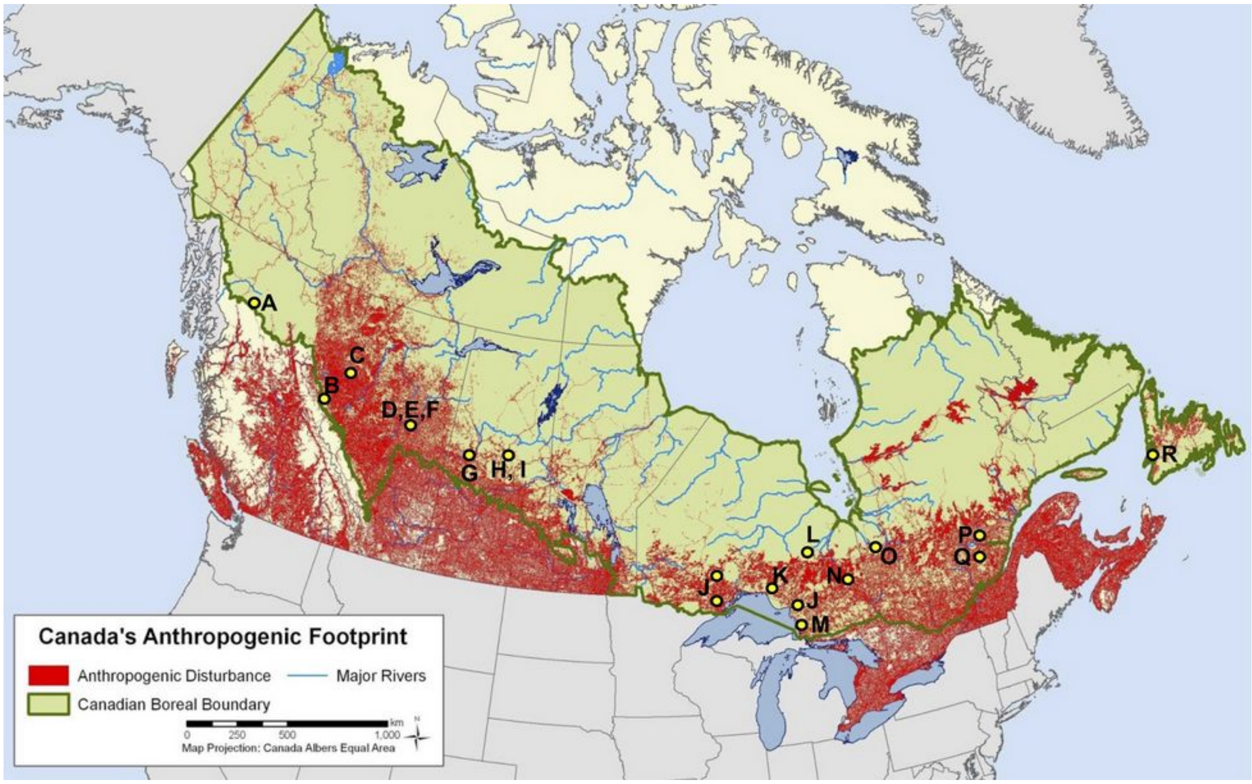

Figure 2.1. Canadian boreal forest boundary and anthropogenic disturbance distribution in Canada. It can be appreciated the main impact of disturbance has been made on the southern border of the boreal forest (map source https://www.borealbirds.org/). Sites of study within the retained items are shown atop the boreal forest's Canadian range. Individual site names are given by letters: A) Date Creek; B) Central Interior; C) EMEND; D) Athabasca E) Calling Lake; F) Quinn Creek; G) Meadow Lake; H) Prince Albert National Park; I) Prince Albert Model Forest; J) BCR12 and BCR8; K) White River and Pukaskwa Park; L) Kapuskasing; M) Stokeley forest; N) Timmins, O) Abitibi; P) Lac-Saint Jean and Faunique des Laurentides; Q) Lac Saint Jean; R) Humber River Valley to Little Grand Lake. See Table 2.1 for the list of studies.

The spatial scale at which logging influences songbird populations must be considered carefully in conservation planning, particularly in a system such as the boreal forest, where small- and large-scale heterogeneities (gaps and stands of diverse ages), are intrinsic not only to anthropogenic but also to natural disturbances (Dalley et al., 2009). Patterns of habitat use by birds differ depending on the spatial scales at which they are measured, and the significant habitat features at one scale may not be apparent at another (Freemark et al., 2002). Only by considering multiple scales can we determine which might be more important in revealing effects useful in conservation (Norton et al., 2000; Richmond et al., 2012; Bosco et al., 2020). Similarly, the boreal forest has been considered naturally fragmented due to frequent historical natural disturbances (Wedeles and Sleep, 2008), and some authors contend that birds are already adapted and resilient to some degree habitat alteration (Erskine, 1977; Mönkkönen and Welsh, 1994; Venier et al., 2014). According to Norton et al. (2000), some species, again dependent on the scale of a disturbance, could use lower quality habitats within the matrix of habitats in a disturbed landscape ("habitat compensation hypothesis"). Pearson and Niemi (1998) found some breeding bird species to be prevalent in unsuitable habitat if it was bordered by favorable habitats. Thus, a mismatch in abundance responses to habitat disturbance across different scales could suggest that the use of alternative areas is occurring (Norton et al., 2000). In particular, an expectation of most of analysis of declines in songbird abundance is that logging in the boreal forest has less negative effect at larger scales, but the reality may be that the declines occur in lower quality habitat.

Large-scale and long-term studies can powerful answers to ecological questions, but they are challenging to conduct because of the time, effort, and financial support required; a meta-analysis is a practical approach to achieving goals across national or biome scales (Bender et al., 1998). A meta-analysis combines results from many studies to assess an effect across one or more common and independent variables (Arnqvist and Wooster, 1995; Gurevitch et al., 2001). In the following meta-analysis, I synthesize the results from a set of studies conducted across the southern border of the Canadian boreal forest biome to assess the effect of logging on migratory songbird abundance at different spatial scales, times since logging, and forest types (conifer, deciduous, and mixedwood forests). I assess this effect by comparing abundance in logged forests to abundance in undisturbed reference forests for 21 selected migratory species with a wide breeding range in the Canadian boreal forest. I also classify them in guilds related to their habitat association (mature or young forest, generalists), migratory strategy (long- and short to medium-distance migrants), and nest behavior (canopy, ground, and shrub nesters). The majority of the species assessed have been declining in population as evaluated by the BBS, especially those commonly associated with old-growth forests or those nesting in trees (Appendix A; Figures A1, A2, A3). The goal to assess whether abundance at different scales may reflect species are able to occupy alternative areas. In addition, I assess under what circumstances, and to what extent, logging might explain the BBS trends.

2.3 METHODS

2.3.1 Literature search

I conducted a literature search (March 2022) for peer-reviewed papers on the Canadian boreal forest using the Web of Science database. I used the following search expression to filter the references: [("boreal forest" AND "Canada") AND ("avian" OR "passerines" OR "birds") AND ("logging" OR "harvest" OR "clearcut" OR "cutblock" OR "forest management" OR "forestry" OR "habitat loss" OR "forest loss" OR "anthropogenic disturbance") AND ("scale" OR "stand" OR "landscape" OR "patch" OR "local") AND ("abundance" OR "density" OR "occurrence" OR "occupancy")]. I searched for these terms in the titles, abstracts, and keywords of papers. To be included in this review, the studies must have met the following criteria (Figure 2.2): 1) data were gathered in the Canadian portion of the boreal forest; 2) they were field-based studies (not literature reviews, systematic reviews, meta-analyses, or simulation or modeling studies); 3) they surveyed multiple species of migratory songbirds (excluding single-species studies); 4) they reported abundance, density, occurrence or occupancy; 5); they compared treatment (logging) and reference sites (generally mature or old-growth forest undisturbed by logging or pre-disturbed).

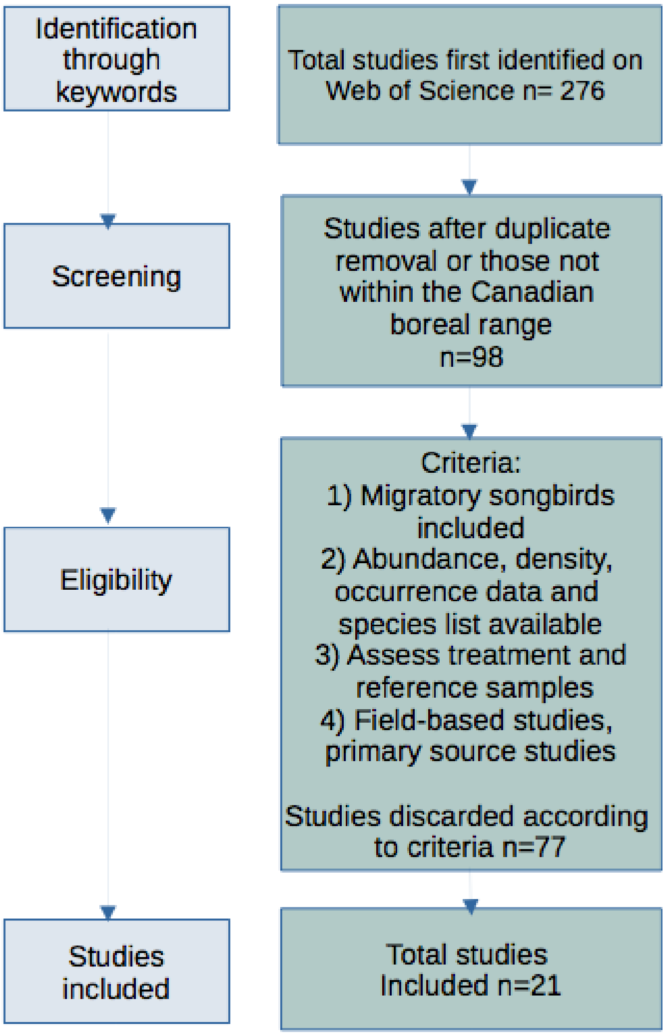

Figure 2.2. Process for searching and selecting studies for their inclusion in the analysis. Blue boxes are the general steps; green boxes show the number of studies filtered in each step and the criteria for selecting the final set of papers.

I excluded studies on areas outside the Canadian boreal forest and on experiments with artificial nests, papers that addressed the basic biology of the birds, and papers that focused on non-forest birds (wetland or grassland birds). No limits were imposed to when the studies were undertaken or published. The result was 21 papers retained for review (Table 2.1); the papers were published from 1997 to 2019, with data collected from 1991 to 2017. I sourced information on changes in abundance from the text, tables, appendix, and supplementary materials, or I extracted the values from graphs using Data Thief III (Tummers, 2006) and Web Plot Digitizer 4.5 (Rohatgi, 2021).

From the 21 articles, I collected 1,202 individual species responses from studies across Canada's boreal forest (Figure 2.1, Table 2.1). For each study, I identified the time since logging and classified them as follows: ≤ 5 years (recently logged), 6-10 years, 14-28 years, and 29-80 years since logging. I was unable to use more precise age categories because this information was lacking for most studies reviewed. I considered studies where abundance was determined based on point counts using males as the individuals registered. I classified the scales of each study at two different levels: local and landscape, based on the description given by the authors within each study. Generally, studies assessing territorial area, point counts or occurring at stand range of 6 to ≤70 ha were considered to be at a local scale; in contrast, landscape-scale studies assessed scales > 70 ha. Adjacent, mature, continuous forest (unharvested mature and old-growth forest or protected areas) was the normal reference used to compare the effect of logging (clearcut, partial retention harvest, single tree selection, or salvage logging) in the studies. Even though I recognize that different logging practices may have different effect on songbirds, due the number of papers gathered for each type, I did not attempt to test it and I assumed all harvest has a level of degradation. In some cases, the effects of logging were compared with pre-treatment conditions or with cumulative changes over time. Finally, I classified each study area as predominantly conifer forest, deciduous forest, or mixedwood.

Table 2.1. List of studies retained for meta-analysis.

Citation (author, year of publication)	Data collection period	Location of the study	# of species extracted	Area surveyed (ha)	Scale considered in the present study
Drapeau et al., 2000	1994-1995	Abitibi, QC	21	Landscape (321.67)	Landscape
Hannon and Schmiegelow, 2002	1994-1998**	Calling Lake, AB	8	Stand (10-40)	Local
Harrison, 2002*	1998-2000**	EMEND, AB	13	Stand (8-10)	Local
Hobson and Kardynal, 2019	2006-2017**	Prince Albert National Park, SK	12	Station (3.18)	Local
Hobson and Bayne, 2000	1993-1994	Prince Albert Model Forest, SK	19	Stand (average 25)	Local
Holmes et al., 2012	1997, 1998, 1999, 2004**	Stokeley forest, ON	7	Stand (not specified)	Local
Imbeau, 1999	1995	Lac-Saint Jean, Faunique des Laurentides, QC	12	Stand (not specified)	Local
Lance and Phinney, 2001*	1996-1998	Central interior, BC	10	Stand (not specified)	Local
Morissette et al., 2002.	1998	Meadow Lake, SK	14	Stand (6-70)	Local
Norton and Hannon, 1997*	1994-1995**	Calling Lake, AB	15	Stand (11, 22, 29)	Local
Norton et al., 2000	1994-1995**	Quinn Creek and Calling Lake, AB	11	Landscape (7,000 and 10,000)	Landscape
Schmiegelow et al., 1997	1993-1995**	Calling Lake, AB	14	Stand (10-40)	Local
Schiek and Hobson, 2000	1997-1998	Athabasca, AB	18	Stand (not	Local

Citation (author, year of publication)	Data collection period	Location of the study	# of species extracted	Area surveyed (ha)	Scale considered in the present study
				specified)	
Steventon, 1998*	1993-1994	Date Creek, BC	8	Stand (not specified)	Local
St-Laurent et al., 2007	2004-2008	Lac-Saint Jean, QC	10	Landscape (85 to 300)	Landscape
Thompson et al., 1999	1991-1992	Humber River valley to Little Grand Lake, NL	12	Stand (25)	Local
Thompson et al., 2013	2001-2002	Kapuskasing, ON	20	Stand (18 and 31.7)	Local
Tittler et al., 2001*	1994,1995, 1997**	Calling Lake, AB	12	Stand (22)	Local
Venier and Pearce, 2007	2001-2003	White River and Pukaskwa Park, ON	18	Landscape (79)	Landscape
Venier et al., 2015	2007-2009	Timmins, ON	19	Landscape	Landscape
Zimmerling et al., 2017	2002-2004	BCR12 and BCR8, ON	21	Stand (> 6)	Local

QC = Quebec, AB = Alberta, SK = Saskatchewan, ON = Ontario, BC= British Colombia, NL = Newfoundland.
*partial retention harvest, **Studies that provided data for each year.

I summarized the number of studies and abundance responses from 21 selected migratory songbird species within the studies (Table 2.2). I only included species with a wide breeding range in the Canadian boreal forest (distribution encompassing most of the boreal region), with a bias toward those in decline and those more associated with forested habitats to test their sensitivity to habitat alteration due to logging in the boreal forest; then I excluded birds that are already known to occupy disturbed areas. I classified the following traits of each species: a) migratory behavior (short to medium- and long-distance migrants; Leston et al., 2018), b) nesting guild (canopy, shrub, and ground nesters; Leston et al., 2018; Hobson and Bayne, 2000; Norton and Hannon, 1997), and c) habitat association (mature-forest specialists, young-forest specialists, and generalists; Leston et al., 2018; Bayne et al., 2016). Using the Breeding Bird Survey data (BBS, Ziolkowski et al., 2022), I plotted trends in their Canadian range for the species included in the study (Appendix A; Figures A1, A2, A3).

The majority of the studies used only one scale of assessment and used standard point counts (a tally of birds detected by sight and sound by an observer located at a fixed position during a specified period of time). There was no standard plot size at either scale. For inferences at the local scale (point count and stand), the size of plots in the studies varied from 3.18 (a point count station) to 70 ha, and at the landscape scale, inferences varied from 79 to 10,000 ha (Table 2.1). Sixteen studies were limited to a stand scale only (considered as a local scale). The majority of studies defined “stand” as an area with a fixed forest age class and the same dominant type of trees (i.e., conifer, deciduous, or a mixed). Only five studies collected data at the landscape scale.

Table 2.2. Species included in the analysis. Migration: long- and short to medium-distance migrants (Leston et al., 2018). Nest guilds: shrub, canopy, and ground (Leston et al., 2018; Hobson and Bayne, 2000; Norton and Hannon, 1997). Habitat association: young forest, generalist (multiple forest age classes), and mature and old-growth forest associated (Leston et al., 2018; Bayne et al., 2016). The numbers between the parentheses are the number of papers that reported the species and the number of responses (sample size).

Species	*Scientific name*	Code	Papers reported/ Detections	Nest guild	Habitat specialization
Long distance migrants					

Species	*Scientific name*	Code	Papers reported/ Detections	Nest guild	Habitat specialization
American Redstart	*Setophaga ruticilla*	AMRE	(16/63)	Shrub	Young
Black-and-white Warbler	*Mniotilta varia*	BAWW	(9/19)	Ground	Young
Bay-breasted Warbler	*Setophaga castanea*	BBWA	(11/25)	Canopy	Mature
Blackburnian Warbler	*Setophaga fusca*	BLBW	(7/17)	Canopy	Mature
Black-throated Green Warbler	*Setophaga virens*	BTNW	(17/84)	Canopy	Mature
Canada Warbler *	*Cardellina canadensis*	CAWA	(11/40)	Ground	Mature
Least Flycatcher	*Empidonax minimus*	LEFL	(15/55)	Canopy	Generalist
Magnolia Warbler	*Setphaga magnolia*	MAWA	(17/50)	Shrub	Young
Mourning Warbler	*Geothlypis philadelphia*	MOWA	(18/77)	Ground	Young
Olive-sided Flycatcher *	*Contopus cooperi*	OSFL	(8/15)	Canopy	Young
Ovenbird	*Seiurus aurocapillus*	OVEN	(17/96)	Ground	Mature
Red-eyed Vireo	*Vireo olivaceus*	REVI	(17/62)	Shrub	Generalist
Swainson's Thrush	*Catharus ustulatus*	SWTH	(18/92)	Shrub	Mature
Tennessee Warbler	*Oreothlypis peregrina*	TEWA	(15/78)	Ground	Generalist
Short to medium distance migrants					
Chipping Sparrow	*Spizella passerina*	CHSP	(17/85)	Shrub	Generalist
Cape May Warbler	*Setophaga tigrina*	CMWA	(7/17)	Canopy	Mature
Golden-crowned Kinglet	*Regulus satrapa*	GCKI	(11/59)	Shrub	Mature
Nashville Warbler	*Oreothlypis ruficapilla*	NAWA	(8/18)	Ground	Young

Species	*Scientific name*	Code	Papers reported/ Detections	Nest guild	Habitat specialization
Ruby-crowned Kinglet	*Regulus calendula*	RCKI	(15/37)	Canopy	Mature
White-throated Sparrow	*Zonotrichia albicollis*	WTSP	(20/109)	Ground/	Generalist
Yellow-rumped Warbler	*Setophaga coronata*	YRWA	(20/107)	Shrub	Mature

*Listed as species at risk (SAR) in Canada (Environment Canada, 2015, 2016)

2.3.2 Data analysis

Most studies did not list standard errors of means (SEM) or standard deviations (SD) when reporting differences, nor did they report R^2 or other measures useful to estimate the effect size of relative abundance. Therefore, I used a relative abundance index (RAI) to measure bird response (Vanderwel et al., 2007). I used RAI to compare species abundance, density, occurrence, or occupancy reported for the treatment or experimental groups (disturbance due to logging) and the reference or control group (continuous forest, unharvested, mature or old-growth forest) within the studies, as:

$$RAI = \frac{Ncut - Nref}{Ncut\ e\ \ Nref}$$

where N_{cut} and N_{ref} represent the respective abundance of species in logged and reference sites. The index ranges from -1 when species abundance is lower in logged sites (negative response to logging) to +1 when species abundance is higher in logged sites (positive response to logging). RAI is 0.0 when a species is equally abundant in logged and reference sites (null response to logging). I calculated the index for each species and estimated the mean of the index and standard error for each group of species, at each scale, and for each of the four time-since-logging classes.

I modeled the relative abundance index (RAI) against the explanatory variables (fixed effects) of scale, time since logging, and three bird traits (nest guild, habitat preference, and migratory strategy) using GLMM (generalized linear mixed models) with a Gaussian error structure, I built the models using the package lme4 in the program R, version 3.6.3 (R Core Team, 2021). I used the study identifier and the province where the study took place as two random variables in all models. I fitted 58 models using different possible combinations of the variables, and I used an information-theoretic approach to select the most parsimonious by contrasting values of Akaike's Information Criterion corrected for small sample sizes (AICc; Burnham and Anderson, 2002). I fitted another GLMM separately for species with >30 samples (abundance responses; Table 2.2). I ran seven models for each species, three models using a single variable, and another four models using the possible different combinations among variables; I used again the AIC to identify the best and most parsimonious models for both the species-level models and the models with bird traits combined into the three sets of guilds (Appendix A; Table A1). Finally, I calculated relative abundance index (RAI) values from top models (lower AIC) by summing the intercept + $ß_x$ and the mean RAI ± SE values from the rest of species with not enough data available for modelling (Table 2.4).

2.4 RESULTS

2.4.1 Bird traits predict abundance response to logging

Of the 58 models compared in explaining the response of boreal forest birds to logging, two received support: (1) the model that predicted Relative Abundance Index (RAI) with only the traits chosen to classify birds into guilds ($\omega_i = 0.52$), and (2) the trait-based model that also included time since logging as a predictor ($\omega_i = 0.27$; Table 2.3). Scale of the disturbance and forest type were not among the top models for the overall RAI or various RAI among guilds. Mature forest specialists, canopy nesters, and long-distance migrants showed a stronger and more negative response to logging (Figure 2.3). In relation to time since logging, mature forest specialists and canopy nesters also have stronger negative responses than generalists, young forest specialists, ground and shrub nesters, and short distance migrants (Figure 2.4).

2.4.2 Bird species response to logging

Fifteen species had sufficient data for building individual species models ($N > 30$ across studies, Tables 2.2 and 2.3). The full set of models for each species can be found in the Appendix A, Table A1, and those with $\Delta AIC \leq 2$ were considered the best models, for which Table 2.4 shows the RAI values (intercept+β_x) for each species. For some species, the best model includes only one of the variables. For five species, the top model included scale as a single predictor: Ovenbird, Swainson's Thrush, and Tennessee Warbler have a negative response to logging at a local scale and an increase in relative abundance with some logging at the landscape scale. Meanwhile, Red-eyed Vireo presented an opposite pattern, having a positive response to logging at a local scale and a negative response at the landscape scale. Finally, White-throated Sparrow had a positive response to logging at both local and landscape scales.

For four species, the top model had “time since logging” as the main predictor: American Redstart, Black-throated Green Warbler, Canada Warbler and Magnolia Warbler. All these species had a strong negative response to logging during the first few years (Table 2.5). Black-throated Green Warbler continued to have a negative response in later years after logging. For another three species, the top model was “forest type” as a single predictor: Chipping Sparrow, Ruby-crowned Kinglet, and Yellow-rumped Warbler. These three species experienced a negative response to logging in the mixed forest type.

Table 2.3. The set of plausible models describing the relationship of bird traits, time since logging and scale with the relative abundance index (RAI) comparing logged areas to undisturbed references. Random effects were “Study” and “Province” for all models. Models are shown with their AIC values and Akaike weight (ωi). The best models are marked in bold and with ΔAIC ≤ 2 (Burnham and Anderson, 2002). ΔAIC = $AIC_i - AIC_{min}$ values.

Model	AIC	ΔAIC	Akaike weight (ωi)
Habitat + Nest + Migration	**2044.49**	**0.00**	**0.52**
Time_logg + Habitat + Nest + Migration	**2045.79**	**1.30**	**0.27**
Habitat + Migration	2048.94	4.45	0.05
Scale + Habitat + Nest + Migration	2048.95	4.46	0.05
Full model	2058.00	13.51	0.00
Null model	2176.91	132.42	0.00

Habitat = habitat preference, Migration = migratory strategy, Nest = nest guild, Time_logg = Time since logging, Scale = scales of the studies.

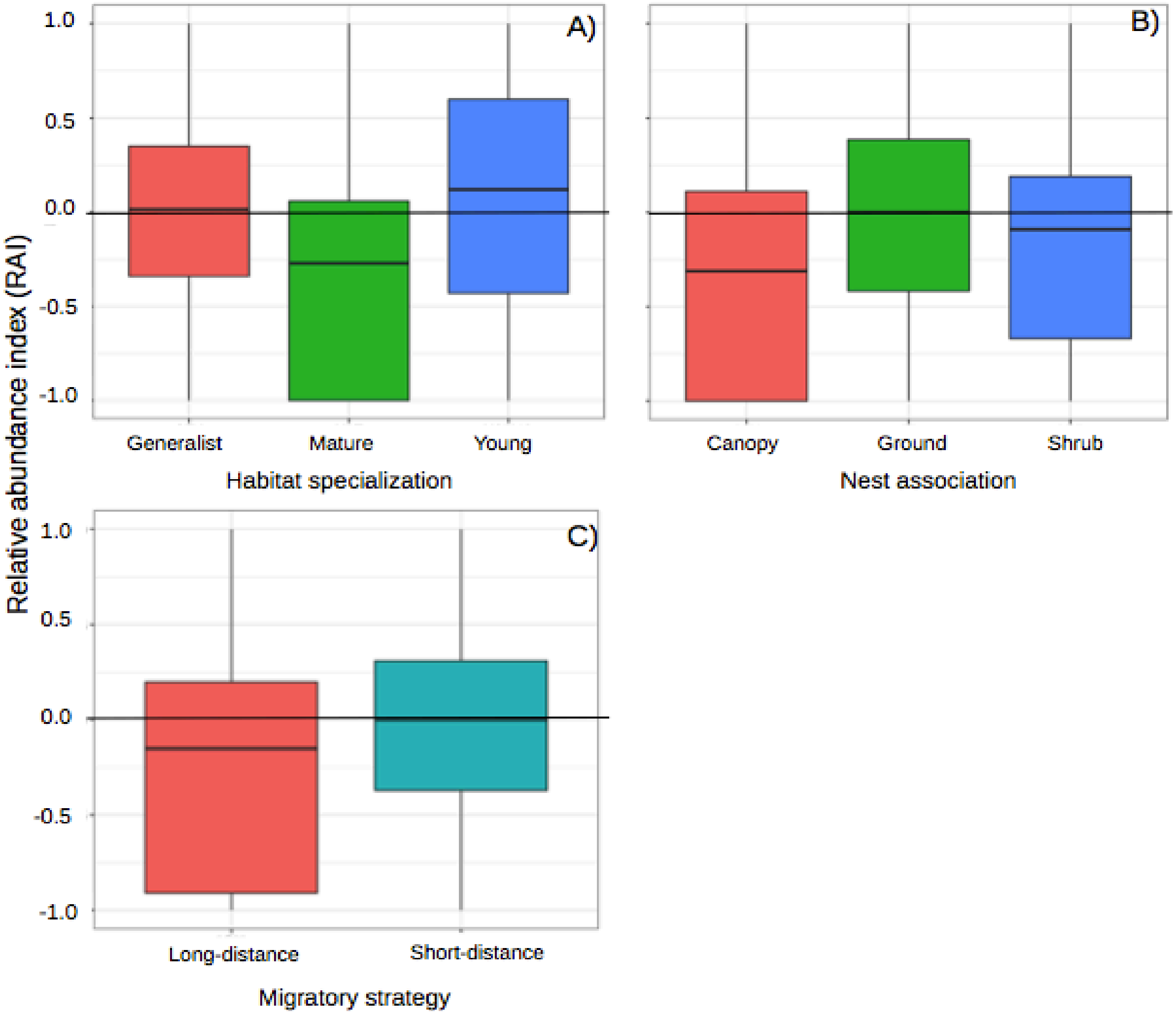

Figure 2.3. Boxplots of the mean Relative Abundance Index (RAI) for the songbird groups, comparing logged forest to reference forest and classified by trait. When RAI is close to 0, the effect is weak or neutral (the abundance in logging areas is similar to reference areas), and values closer to 1 indicate a strongest effect of logging; negative values indicate a negative effect of logging and positive values indicate abundance is higher in logging areas than in reference areas.

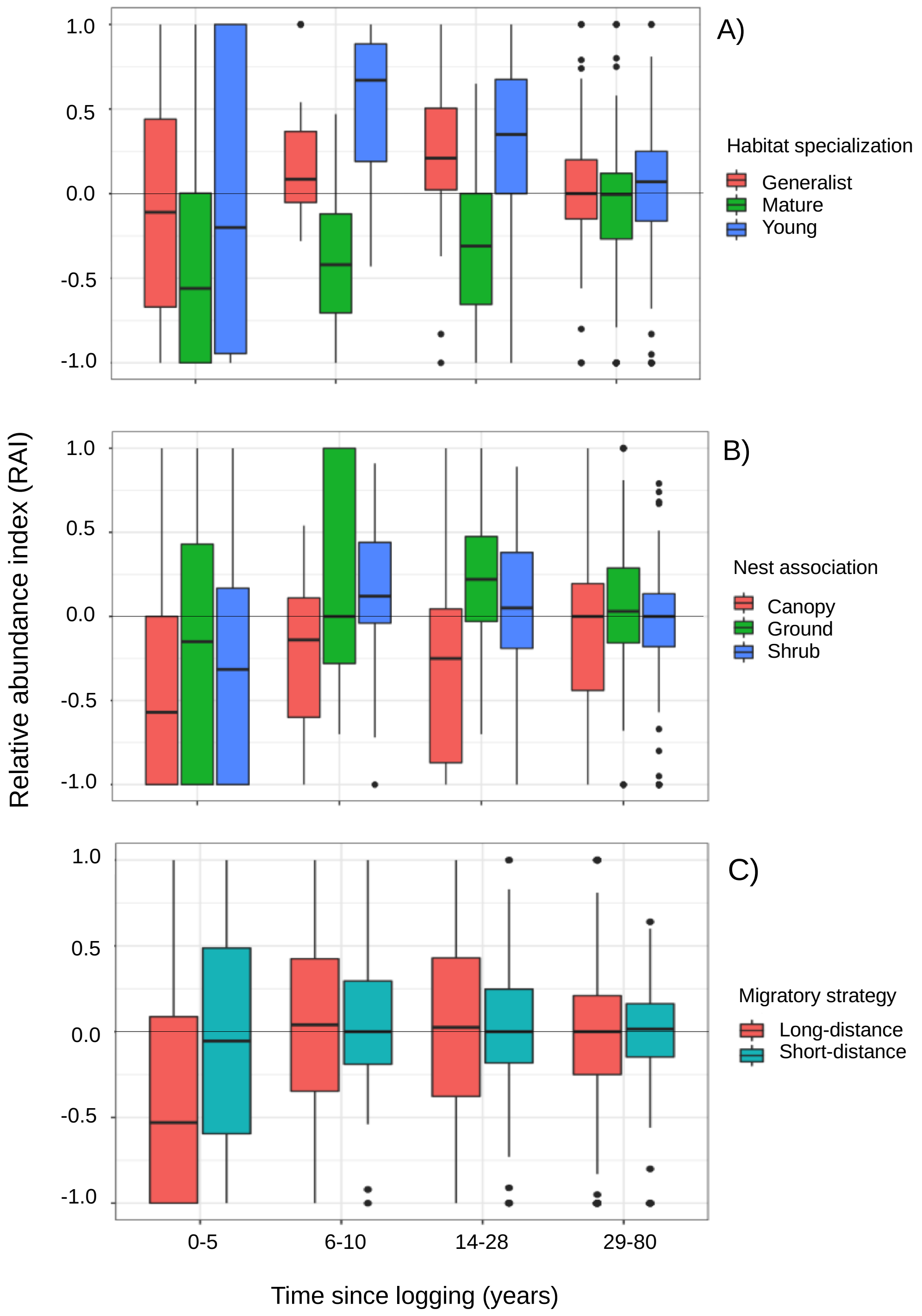

Figure 2.4. Boxplots of the mean Relative Abundance Index (RAI) for the songbird groups, comparing logged forest to reference forest and classified by time since logging. When RAI is close to 0, the effect is neutral (the abundance in logging areas is similar to reference areas), and values closer to 1 indicate a strongest effect of logging; negative values indicate a negative effect of logging and positive values indicate abundance is higher in logging areas than in reference areas.

For three species, the best model was the combination of “time since logging” and “forest type” (Appendix A, Table A1). Least Flycatcher, and Magnolia Warbler experienced a positive response in all years after logging and a negative response to logging in deciduous and mixed forest. Golden-crowned Kinglet experienced a negative response >14 years after logging and also in coniferous and mixed forest (Table 2.5). The Least Flycatcher had a second-best competing model that included all the predictors, where only in deciduous and mixed forest did a negative response to logging occur.

There were at least 10 species with an increase in RAI at the landscape scale comparing to the local scale: Black-throated Green Warbler, Canada Warbler, Cape May Warbler, Least Flycatcher, Olive Flycatcher, Swainson’s Thrush, Tennessee Warbler, Golden-crowned Kinglet, Ruby-crowned Kinglet and Yellow-rumped Warbler (Table 2.5). However, the opposite pattern occurred between scales for Ovenbird, Red-eyed Vireo, and Nashville Warbler.

Logging in coniferous and mixed forest types created a negative effect on abundance for the majority of species commonly related to these forest types and to mature or old-growth forests (Blackburnian Warbler, Black-throated Green Warbler, Canada Warbler, Cape May Warbler, Golden-crowned Kinglet, Ovenbird, Swainson’s Thrush, Yellow-rumped Warbler, Table 2.4). Meanwhile, generalists or young-forest specialists (American Redstart, Chipping Sparrow, Least Flycatcher, Magnolia Warbler, Mourning Warbler, Olive-sided Flycatcher, Red-eyed Vireo, Nashville Warbler and White-throated Sparrow) had a positive association with logging in coniferous forest (Table 2.5). The species with the strongest negative association with logging in deciduous forest were Mourning Warbler and Yellow-rumped Warbler. Also, some species had a positive response limited to the first years after logging.

Table 2.4. Calculated relative abundance index (RAI) values from top models (15 species with >30 samples): intercept+ß$_x$ ±SE at the different scales of logging disturbance, time since logging, and forest type. Species' names, migratory status, nest guild, and habitat specialization are in Table 2. LDM= Long distance migrants, SD= Short to medium distance migrants.

	Scale (ha)		Time since logging (years)				Forest type		
Species code	Local 6-70	Landsc >71	0-5	6-10	14-28	29-80	Conifer	Deciduous	Mixedwood
LDM									
AMRE	---	---	-0.34±0.50*	0.69±0.49	0.42±0.55*	0.09±0.63	---	---	---
BTNW	---	---	-0.42±0.44*	0.24±0.48	-0.35±0.50	-0.08±0.57	---	---	---
CAWA	---	---	-0.59±0.36*	---	-0.05±0.49	0.07±0.53*	---	---	---
LEFL	---	---	0.92±0.71	1.29±0.28*	1.27±0.33*	1.29±0.45	1.01±0.21	2.16±1.11*	1.97±0.91*
	0.79±0.74	1.42±0.29	1.29±0.65*	1.64±0.19*	1.62±0.22*	1.66±0.33*	1.42±0.29	-0.23±0.69*	-0.18±0.58*
MAWA	---	---	-0.14±0.93	0.46±0.45	0.38±0.57	0.24±0.81	---	---	---
	---	---	0.04±1.28	0.86±0.41*	0.78±0.51	0.62±0.73	0.16±0.33	-0.47±2.12	-0.79±1.73*
MOWA	0.58±1.99	0.84±0.42	---	---	---	---	0.85±0.43*	-0.95±0.62*	0.28±1.23*
OVEN	-0.19±2.36	-0.09±0.25	---	---	---	---	---	---	---
REVI	0.06±1.51	-0.03±0.21	---	---	---	---	---	---	---
SWTH	-0.39±1.80	-0.12±0.21	---	---	---	---	---	---	---
TEWA	-0.06±2.20*	0.69±0.28	---	---	---	---	---	---	---

	Scale (ha)		Time since logging (years)				Forest type		
Species code	Local 6-70	Landsc >71	0-5	6-10	14-28	29-80	Conifer	Deciduous	Mixedwood
SD									
CHSP	---	---	----	---	---	---	0.27±0.14	1.14±0.51*	-0.85±0.96
GCKI	---	---	0.44±0.55	---	-0.35±0.41	-0.09±0.44*	-0.05±0.09	0.95±0.39*	-0.75±0.48
RCKI	---	---	---	---	---	---	0.26±0.16	---	-0.22±0.95*
WTSP	0.28±1.90	0.37±0.22	---	---	---	---	---	---	---
YRWA	---	---	---	---	---	---	-0.08±0.09*	-0.54±0.46*	-0.18±0.69

--- no data available or variables not included in the model. * significant values p<0.05

Table 2.5. Bird species Relative Abundance Index (mean±SE) of the total 21 species at the different scales of logging disturbance, time since logging, and forest type. Species' common names, scientific names, migratory status, nest guild, and habitat specialization are in Table 2.2. LDM= Long distance migrants, SD= Short to medium distance migrants. sh=Shrub, can= Canopy, gnd= Ground, mat=Mature, yng= Young, gen= Generalist. RAI close to 0 the effect is neutral (the abundance in logging areas is similar to reference areas); negative values indicate a negative effect of logging; values closer to 1 indicate a strongest effect of logging.

	Scale (ha)		Time since logging (years)				Forest type		
Species code	Local 6-70	Landscape >71	0-5	6-10	14-28	29-80	Conifer	Deciduous	Mixedwood
LDM									
AMRE $^{\text{sh, yng}}$	-0.09±0.09	-0.01±0.05	-0.45±0.13	0.7±0.18	0.42±0.14	0.08±0.09	0.36±0.26	-0.16±0.19	-0.13±0.08
BAWW $^{\text{gnd, shr}}$	0.06±0.16	0.05±0.13	-0.02±0.33	---	0.24±0.11	-0.03±0.23	0.11±0.20	---	0.03±0.17
BBWA $^{\text{can, mat}}$	-0.62±0.11	-0.70±0.20	-0.94±0.05	-0.49±0.17	-0.70±0.11	-0.24±0.26	-0.37±0.23	---	-0.73±0.08
BLWA $^{\text{can, mat}}$	-0.46±0.15	-0.60±0.17	-0.59±0.28	-0.38±0.32	-0.40±0.23	-0.67±0.19	-1.00±0.00	---	-0.33±0.14
BTNW$^{\text{can, mat}}$	-0.33±0.06	-0.21±0.23	-0.49±0.09	0.23±0.18	-0.35±0.14	-0.07±0.08	-0.42±0.14	0.06±0.10	-0.34±0.07
CAWA$^{\text{gnd, mat}}$	-0.42±0.10	-0.24±0.43	-0.70±0.09	---	-0.05±0.18	0.07±0.23	-0.05±0.29	-0.75±0.09	-0.23±0.15
LEFL$^{\text{can, gen}}$	-0.26±0.09	-0.02±0.12	-0.49±0.09	0.19±0.18	0.30±0.18	0.04±0.06	1.00±0.00	-0.50±0.11	-0.16±0.14
MAWA$^{\text{sh, yng}}$	0.06±0.10	-0.04±0.14	-0.01±0.21	0.44±0.13	0.21±0.13	-0.17±0.13	0.32±0.15	0.17±0.25	-0.17±0.12
MOWA$^{\text{gnd, yng}}$	0.26±0.7	0.18±0.22	0.07±0.12	0.50±0.25	0.60±0.13	0.12±0.08	0.90±0.09	0.09±0.22	0.16±0.06

	Scale (ha)		Time since logging (years)				Forest type		
Species code	Local 6-70	Landscape >71	0-5	6-10	14-28	29-80	Conifer	Deciduous	Mixedwood
OSFL$^{can, yng}$	-0.23±0.25	0.52±0.29	0.18±0.38	---	0.00±0.58	-0.46±0.26	0.46±0.34	---	-0.42±0.24
OVEN$^{gnd, mat}$	-0.25±0.06	-0.32±0.12	-0.37±0.09	-0.47±0.09	-0.15±0.11	0.00±0.06	-0.19±0.19	-0.34±0.11	-0.24±0.07
REVI$^{sh, gen}$	0.01±0.07	-0.21±0.16	-0.18±0.13	0.06±0.02	0.25±0.11	0.22±0.08	0.19±0.30	-0.13±0.34	-0.03±0.04
SWTH$^{sh, mat}$	-0.49±0.05	-0.24±0.14	-0.49±0.06	-0.71±0.10	-0.51±0.14	-0.25±0.10	-0.51±0.08	-0.39±0.16	-0.56±0.07
TEWA$^{gnd, gen}$	-0.31±0.06	0.34±0.25	-0.45±0.08	-0.06±0.08	0.03±0.09	0.22±0.14	-0.10±0.16	-0.59±0.08	-0.21±0.08
SD									
CHSP$^{sh, gen}$	0.08±0.07	0.14±0.20	0.16±0.09	0.10±0.16	0.10±0.16	-0.10±0.07	0.02±0.15	1.00±0.00	0.02±0.06
CMWA$^{can, mat}$	-0.57±0.12	-0.17±0.09	-1.00±0.00	-0.60±0.30	-0.34±0.14	-0.36±0.28	-0.85±0.11	---	0.02±0.06
GCKI$^{can, mat}$	-0.42±0.08	-0.16±0.15	-0.45±0.10	---	-0.45±0.15	-0.15±0.11	-0.45±0.08	1.00±0.00	-0.64±0.09
NAWA$^{gnd, shr}$	0.40±0.14	0.11±0.24	0.00±0.08	--	0.16±0.22	0.11±0.17	0.42±0.19	---	0.08±0.06
RCKI$^{can, mat}$	-0.16±0.11	0.04±0.21	-0.22±0.18	---	-0.27±0.19	0.01±0.12	0.05±0.11	---	-0.28±0.14
WTSP$^{gnd, gen}$	0.27±0.05	0.21±0.08	0.31±0.08	0.47±0.24	0.45±0.09	0.05±0.04	0.24±0.13	0.56±0.07	0.16±0.05
YRWA$^{sh, mat}$	-0.18±0.04	0.03±0.04	-0.24±0.06	-0.15±0.20	-0.03±0.08	-0.01±0.04	-0.22±0.07	0.04±0.17	-0.18±0.04

– no data is available.

2.5 DISCUSSION

Certain traits shared by groups of boreal forest birds and the number of years that passed since logging are important predictors of their relative abundance, comparing logged areas to unlogged reference sites in this meta-analysis. The characteristics of the bird species, such as their nesting habits, migratory behavior, and habitat specialization, play a crucial role in determining their response to logging. The results suggest that logging in the southern parts of the Canadian boreal forest is contributing to the decline of many of the migratory songbird species. Even though logging cannot be considered as habitat loss or deforestation due to the forest eventually regenerating, the results here suggest that logging is being depleting and suitable habitats are less available to many migratory songbirds. In other areas of Canada, such as the Wabanaki-Acadian forest, habitat degradation due to logging has been found more detrimental than habitat loss, leading to other observed avian declines and recommendations that conservation of old-growth forest might be the best buffer against negative effects of climate warming on sensitive avian populations (Betts et al., 2020).

Logging was associated with lower relative abundance of long-distance migrants, birds associated with mature forests, and canopy nesters, consistent with patterns observed by Mahon et al. (2019) and Cadieux et al. (2020). The eventual recovery of the mature forest condition is consistent with lower variability in the Relative Abundance Index (RAI) used in this meta-analysis, which occurs with increasing time since logging. Logging, in addition to natural disturbances like wildfire, is used to explain observed increases in species that prefer early successional habitats (Imbeau et al., 1999; LaManna and Martin, 2017; Cadieux et al., 2020). Mature-forest and old-growth specialists are replaced by young-forest specialists and generalists when old-growth forest becomes low in availability. The results of this meta-analysis are also consistent with the idea that strong habitat specialization, especially specialization in late-successional forests, leads to a lower abundance of boreal forest birds following logging (Lampila et al. 2005; Vanderwel et al., 2007; Cadieux et al., 2020). The loss and degradation of mature forests are considered more detrimental than the configuration of the remnant forest in harvested landscapes, particularly when considering long-distance migrants that are associated with late successional forest stages (Drolet et al., 1999; Drapeau et al., 2000; Schmiegelow and Mönkkönen, 2002).

The spatial scale and the forest type before disturbance are less associated with any change in RAI. However, these factors are important predictors of relative abundance for certain species either as a single variable or in combination. Species such as the Golden-crowned Kinglet, Ovenbird and Swainson’s Thrush had forest type or scale as the main predictor of their response to logging. The Swainson’s Thrush and Ovenbird are commonly known as habitat area sensitive (Freemark et al., 1995). Drapeau et al. (2016) found Swainson’s Thrush among the most sensitive to the loss of old-growth forest cover at both landscape and local scales. Rempel (2007) concluded that the Golden-crowned Kinglet and Ovenbird are associated with conditions of lower disturbance and greater areas of intact mature forest. I observed in the present meta-analysis similar patterns for these species.

I expected to document a greater number of species increasing in RAI or experiencing a positive effect at landscape scales due to the hypothesized adaptation of birds to boreal forest disturbance that historically occurred at landscape scales (Mönkkönen and Welsh, 1994; Venier et al., 2014). Finding differences in abundance responses between scales suggests that either by species shifting to less preferred areas or finding other suitable habitats within the landscape

(Norton et al., 2000). However, I found just half of the species in this meta-analysis showing an increase and/or positive effect in their relative abundance at landscape scales contrasting with their mean RAI obtained at the local scale, suggesting that the ability of some species to use alternative areas does not happen consistently. Six of these ten species (Canada Warbler, Cape May Warbler, Golden-crowned Kinglet, Least Flycatcher, Swainson's Thrush, and Tennessee Warbler) have a declining trend within their Canadian range according to BBS (Appendix A; Figures A1, A2, A3). Though it is currently unknown to what extent population declines are a result of habitat alterations on the breeding grounds or the wintering grounds for these migratory species, logging may be contributing to the decline in the boreal range.

Canopy nesters were one of the most sensitive to habitat disturbance by logging, likely due to the direct removal of the trees where they normally nest. In a review across North America, tropical and temperate habitats had a low abundance of canopy nesters in clear-cut harvest areas (LaManna and Martin, 2017). I found that species considered canopy or shrub nesters that experienced a negative effect due to logging in coniferous or mixed forests, were also species commonly associated with those type of forest (Golden-crowned Kinglet, Swainson's Thrush, Yellow-rumped Warbler, Blackburnian Warbler, Olive-sided Flycatcher, and Cape May Warbler, see RAI means for all species in Appendix A. Figures A1, A2, A3). The majority of these species have been identified as declining according to BBS data (Appendix A; Figures A1, A2, A3).

The decline in species associated with coniferous and mixedwood forest can be an indicator of old-growth forest degradation due to logging. Degradation of forests happens when regeneration following clearcutting commonly results in forest stands with an even-aged structure (Le Blanc et al., 2010); clearcutting also creates greater proportions of young forest versus old forest stages on the landscape (Kuuluvainen and Gauthier, 2018). In addition, Hobson et al. (2013) estimated that the highest average annual logging activity across Canada is happening in coniferous forests (81% of the total average). Moreover, logging seems not to be emulating historical natural disturbance patterns in the boreal forest due to shorter harvesting schedules (Van Wilgenburg and Hobson, 2008; Kuuluvainen and Gauthier, 2018), a higher proportion and disturbance of logging in coniferous areas (Hobson et al., 2013; Betts et al., 2020), and degradation by the simplification of their structure and reduction of tree species diversity (Drapeau et al., 2000; Betts et al., 2020). Therefore, not only is the loss of older forests a threat to songbird abundance, but also the degradation and disturbance of specific habitats such as coniferous and mixed forests that the species rely on are likely causes for declines.

On the other hand, generalists, young-forest specialists, and shrub and ground nesters are the groups that recover easiest through forest regeneration or do not experience an important negative impact of logging on their relative abundance. Rempel (2007) observed that the American Redstart, Least Flycatcher, Mourning Warbler, Red-eyed vireo, and White-throated Sparrow were associated with high disturbance intensity and less mature forest. I observed similar patterns for the same species. At the same time, according to the BBS trends, Least Flycatcher, Mourning Warbler, and White-throated Sparrow are species in decline (Appendix A; Figures A1, A2, A3). The results do not support that logging in the breeding ground is one of the drivers of declines for these three species, and further research is required to determine the causes that may be associated with other type of disturbance in the boreal forest or other threats in the wintering grounds.

Finally, it is important to do research at the multiscale level and develop long-term studies to track long-term effects to increase the knowledge and understanding of the effects of habitat alteration on bird populations. I found that there were relatively few studies capturing abundance responses to logging at a variety of scales. Importantly, I identified that there is no homogeneous definition of scale among studies. This lack of consistency in the scale and types of logging across studies complicates the task of ensuring an accurate systematic review or meta-analysis. I encourage researchers studying the impacts of logging on birds to employ more than one scale in their assessment and identify the type of forest logged and its configuration to improve data availability and interoperability for future reviews.

CHAPTER 3. FIELD-VALIDATED SPECIES DISTRIBUTION MODEL OF CANADA WARBLER (*CARDELLINA CANADENSIS*) IN NORTHWESTERN ONTARIO.[1]

3.1 ABSTRACT

The Canada Warbler (CAWA) is a species of conservation concern, but its ecological needs and distribution remain poorly understood. Additionally, contradictory findings exist regarding the impact of logging on CAWA abundance and habitat use. Furthermore, its habitat needs may be distorted by limitations in current habitat availability compared to historical conditions. Using Maxent, I developed a predictive high-resolution (30 m) field-validated species distribution model (SDM) in Northwestern Ontario, Canada, where little field-derived information about the species is available. I aimed to assess how time since logging affects CAWA occurrence and distribution. The SDM was built on occurrences (2001-2020) from different large datasets (including eBird) supplemented with field-collected data from 2021. The SDM's environmental covariates included digital elevation model (DEM), years since disturbance (usually by logging, LOGG), tree canopy height (CAN), distance to water bodies (WATER), distance to mature coniferous (D_CONIF) and spectral indexes such as normalized water index (NDWI) and enhanced vegetation index (EVI). I validated the final model using field-collected data in 2022. The final model showed moderate performance for both training and test data (AUC = 0.7), with, NDWI, WATER, EVI and D_CONIF being the most influential covariates indicating high association with deciduous vegetation, riparian zones, high shrub cover and importance of coniferous stands. Probability of CAWA occurrence was high (>0.7) in areas where forest has been not disturbed by logging, in addition was high (0.6) in areas that have within six years since disturbance, indicating that CAWA may take advantage of regenerated forest depending on

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[1] A previous draft of this chapter has been submitted for publication in the Journal *Avian Conservation and Ecology* and is subject to be accepted pending revisions. Co-authors are V. Cupiche-Herrera, A. Westwood, and B. McLaren.

shrub density and retention of old-growth forest structure (tree canopy height >10 m). Based on the association to CAN, I recommend that retention of tall trees be implemented for CAWA conservation, and logged areas be managed to retain favorable shrub habitat. I present the first-known ground-truth SDM for CAWA.

3.2 INTRODUCTION

Songbirds in the Parulidae family comprise one of the groups with the highest rates of decline in North America (Rosenberg et al., 2019). In particular, long-distance migratory species associated with old-growth forests experience major threats (Lampila et al., 2005). The Canada Warbler (CAWA) is a migratory songbird commonly associated with old growth forest in the boreal biome and has been declared a species at risk in Canada (Bayne et al., 2016; Environment Canada, 2016) due to a substantial decline in abundance over the last half-century (Sauer et al., 2017; Wells et al., 2020). Causes of its decline are thought to include habitat loss, habitat alteration, and changes to forest successional patterns on the breeding grounds (ESRD, 2010; Reitsma et al., 2010). Short harvest rotations and loss of older forest age classes may also be contributing to the decline (Grinde and Niemi, 2016). Studies across the species' range show mixed responses to forest harvesting and associated disturbance (Hunt et al., 2017). In Alberta, CAWA has higher density and productivity in postharvest areas (harvested areas with regeneration >5 years) than in recent clear-cuts (Ball et al., 2016; Hunt et al., 2017). There is also evidence of the detrimental effects of road networks on the density of the species (Miller, 1999; Haché et al., 2014; Westwood et al., 2019b).

Knowledge of the ecology and geographic distribution of a species is critical for prioritizing and informing conservation action, planning, and assessing threats from many anthropogenic factors (Akçakaya and Atwood, 1997; Wintle et al. 2005, Hirzel et al. 2006). However, the current understanding of the ecological needs of CAWA is biased by the selection of study sites (and associated findings), which are influenced by site accessibility (Environment Canada 2016). For example, projects such as the provincial Breeding Bird Atlases (BBAs) and the Breeding Bird Survey (BBS) are restricted mainly to roadside surveys (Kirk et al., 1997; Matsuoka et al., 2011), while most of the CAWA range is in areas with low road density or no roads at all. Additionally, the species' apparent habitat needs might be distorted by limitations in current habitat availability compared to historical conditions (Environment Canada, 2016; Wells et al., 2018). Species distribution models (SDMs) can be a useful tool for understanding habitat associations and identifying conservation and management opportunities, particularly in undersampled areas. These models are usually correlative (Guisan and Zimmermann, 2000) and quantify the relationship between field observations and a set of environmental variables that are expected to reflect some key aspects of the species-habitat association (Hirzel et al., 2006). The resulting spatial predictions of species distribution, and associated maps of these estimates, are widely used to guide conservation strategies (McShea, 2014).

For CAWA, five SDMs have already been developed; two at a national scale in Canada (Haché et al., 2014; Stralberg et al., 2015) and another three at regional scales in Alberta (Ball et al., 2016) and the Atlantic provinces; in the latter case, one at 150 m x 150 m resolution and another at 250 m x 250 m resolution (Bale et al., 2020; Westwood et al., 2019b). CAWA is known to exhibit regional variation in habitat associations, particularly between the eastern and western portions of its range (Leston et al., 2023), and regionally specific habitat associations are needed to guide local management priorities. In Northwestern Ontario, little information about CAWA is available. Quetico Provincial Park, one of the largest wilderness-protected areas in

Northwestern Ontario, remains heavily forested compared to other regions in the southeast part of Ontario. Currently, most of Northwestern Ontario either has been logged or is projected to be logged in the near future, leaving Quetico the most extensive area without timber extraction. The region of Northwestern Ontario (including the Quetico area) has one of the highest relative abundance estimates of the CAWA in Canada and especially in Ontario (>3-10 birds/route/year) from the BBS project (Sauer et al., 2017; Appendix B, Figure B1).

Most SDMs are only validated statistically by using part of the original dataset applied as a "test" dataset (Roberts et al., 2017; Westwood et al., 2019b). Although the best way to validate the accuracy of an SDM is using independent, field-collected data (Yates et al., 2018), this step is rarely taken due to a lack of time or funding to collect and process new data (Franklin, 2013). Also, notwithstanding thousands of published SDMs in the past two decades, relatively few examples are validated by an independently collected dataset (e.g., Ortega-Huerta and Vega Rivera, 2017; Westwood et al., 2019b). To my knowledge, there is no ground-truth SDM for CAWA. I built two SDMs using maximum entropy algorithm, Maxent (Phillips et al., 2006) and I ground truth the model by collecting independent field observations in 2022. The objectives were to (1) predict the distribution of CAWA in an ecoregion of Northwestern Ontario; (2) compare habitat associations and probability of occurrence between protected and logged areas to assess how habitat alteration influences distribution; and (3) use an independent field-collected dataset to validate the accuracy of the final model and measure correspondence between predicted and observed occurrences. The final model can be used to guide conservation and management action for this species at risk and inform methods for modelling CAWA or other migratory bird species in other regions.

3.3 METHODS

3.3.1 Study area

Quetico Provincial Park and its surrounding area are in the Pigeon River Ecoregion (Ecoregion 4W), which is located within the Ontario Shield Ecozone in Northwestern Ontario; it covers 2,035,903 ha (2.0% of the province; Figure 1; Crins et al., 2009). The climate of this ecoregion is cool and relatively dry; the mean annual temperature is 0.2 to 2.7 ºC, and the mean growing season length is 168 to 188 days (OMNR, 2000). Annual precipitation ranges from 674 to 838 mm, and mean summer rainfall ranges from 225 to 300 mm (Crins et al., 2009). Mixed forest is the most extensive land cover class (33.2%), sparse forest occurs at 19.3%, water at 17.5%, coniferous forest at 11.5%, deciduous forest at 10.6%, and cutovers at 3.6% of the area of this ecoregion.

Ecoregion 4W is divided into two ecodistricts, 4W-1, Quetico and 4W-2, Kakabeka. Ecodistrict 4W-1 has human settlement on <1% of its area: the largest community is Atikokan, Ontario, and the protected areas encompass 30%, including Quetico Provincial Park (4,760 km^2), the first official park in Northwestern Ontario (Wester et al., 2018). Ecodistrict 4W-2, Kakabeka represents 18.1% of the ecoregion, and settlement and associated infrastructure cover 2% of the total area, while protected areas cover only 4.3% of the ecodistrict. Predominant land uses include timber harvesting, resource-based tourism, mineral exploration, and agriculture; the city of Thunder Bay is the largest urbanized community in Northwestern Ontario. According to the last census in 2021 (Statistics Canada 2022), Thunder Bay district has a population of 146, 867, with 74.1% living in the city of Thunder Bay (population 108,849).

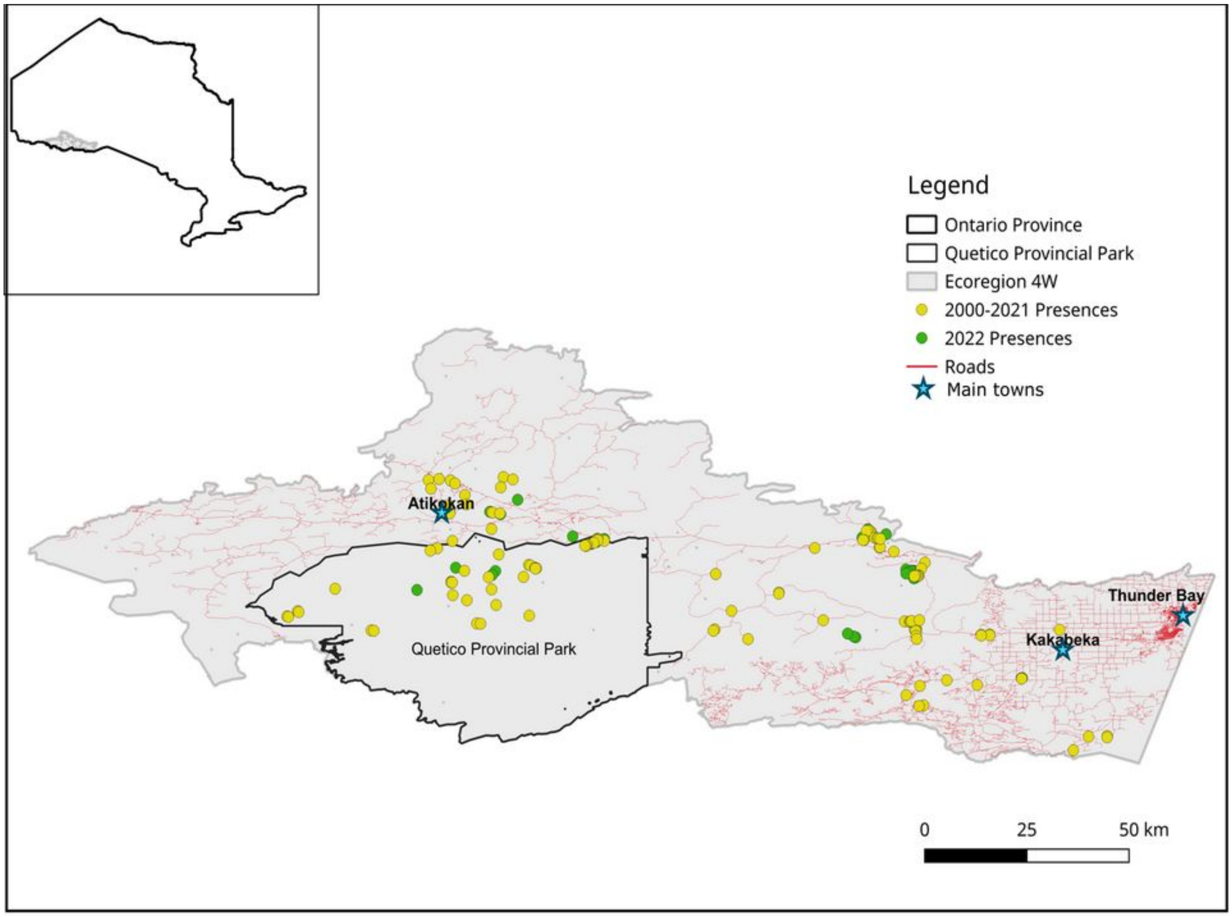

Figure 3.1. Study area. Ecoregion 4W in Northwestern Ontario. Yellow dots are occurrences (*N*=122) from 2001-2021, and green dots are field-collected observations (*N*=30) from 2022 used for validating the final model.

3.3.2 Species occurrence dataset

For the preliminary (2020) model, I used existing data sources of CAWA occurrences. I used occurrence data derived from the long-term songbird monitoring program in Quetico Provincial Park (2014-2019), BBS (Ziolkowski et al., 2022), Ontario Breeding Bird Atlas ([OBBA] Bird Studies Canada, 2001-2005), and the Cornell Lab of Ornithology's eBird project (Fink et al., 2021), which has compiled avian point count data in North America from incidental observation records collected through community science programs. The eBird project can offer an extensive coverage dataset for understudied or elusive species at no cost to the user (Sullivan et al., 2014).

Community science datasets like eBird can also provide important information outside of regularly surveyed areas (Lin et al., 2022) and support design and management of protected areas (Binley et al., 2021). The eBird dataset is recorded by a variety of people: amateurs, tourists, researchers, and volunteers, sometimes collecting data farther than from roadside.

From the three databases (eBird/OBBA, BBS, Quetico Provincial Park), I extracted 235 CAWA presence observations. I removed 66 duplicated observations, and we applied a spatial filtering of 250 m to the occurrence dataset; when the distance between points was below this threshold, the point closer to a road was removed (Bale et al., 2020). In addition, I removed localities with missing coordinates and other georeferencing errors (Boria et al., 2014; Cobos et al., 2018), as well as data falling over areas covered by clouds in the Landsat layers. I excluded 93 occurrences in total, leaving 78 observations (from 2001 to 2020) for the construction of a 2020 preliminary model (see ‘Model construction’, below). Given that 78 observations are a relatively small dataset, especially for an area of this size, I used the model 2020 to guide additional field surveys in spring of 2021. I surveyed a total of 132 point counts in 2021, obtaining 44 new observations of CAWA to add to the training dataset. I ended with a total of 122 observations available for a second model (2021) and the final model (2022), with a timespan from 2001 to 2021 (Figure 3.1; Appendix B, Table B2).

The field surveys followed the OBBA survey protocol and I recruited volunteers to assist with counts during the breeding seasons (late May to late July) of 2021 (additional data for training dataset for the final model) and 2022 (independent data for model testing). OBBA uses point counts within a 10 x 10 km square, recording birds for 5 min at each point approximately 250-300 m apart from each other. For 2022 I surveyed a total of 118 locations and obtained 30 new CAWA observations to use as validation data for the final model. I recorded the coordinates of each point count location using a GPS unit. The two observers conducted the point counts in favorable weather conditions, on days without precipitation or wind >20 km/h, both of which reduce the detectability of singing birds (Cadman et al., 2007).

After the 5-min bird point count, observers used speakers or mobile devices to play CAWA songs to increase the detectability of the species and register the number of individuals in the area. The observers recorded the total number of territorial males detected by sight or sound at each point count using the following protocol: 1) 30 s of playbacks of conspecific songs; and (2) 1 min silence, (3) 30 s of playbacks, (4) 1 min silence (protocol modified from Flockhart et al., 2016; Hunt et al., 2017). The repetition of the playback and the silent periods helped to reduce the bias of artificial calls and the possible effect of individuals approaching just due to curiosity. The observers counted only the males that responded during or after the second period of playback of conspecific songs.

3.3.3 Environmental covariates

The distribution of any species, including CAWA, is controlled by its niche (Pulliam 2000). Accordingly, I selected variables for the SDM that most closely matched the CAWA niche as described by the literature. The species inhabits many forest types but is most abundant in humid, mixed forests with a dense understory and complex ground cover; it is associated with forest disturbance that creates “suitable” understory conditions (Becker et al., 2012). Local-scale studies have suggested that the CAWA has breeding territories often on steep slopes near streams (Schieck et al., 1995; Schieck and Song, 2006; Reitsma et al., 2010). On larger scales (landscape and regional), Haché et al. (2014) found CAWA densities to generally be higher in areas with tall trees, and Westwood et al. (2019a) found the species to be sensitive to anthropogenic disturbance. Based on this information, I initially selected ten variables that captured local and landscape features (Table 3.1).

I created geospatial raster layers in QGIS 3.18 to extract the candidate variables as predictors or covariates. In order to get a high resolution SDM, I used LANDSAT 30 m resolution images. However, due to the climate conditions of the region: high humidity and condensation I could not find cloud-free (<10%) images from the whole Ecoregion to extract candidate predictor variables from different years. I was limited to use cloud-free (<10%) satellite images from LANDSAT 8 collection 2 level 1 from June and August 2018 courtesy of the U.S. Geological Survey (USGS, 2020).

I calculated spectral indices from LANDSAT images to use them as covariates Spectral indices included calculating bare soil index (BASI) as an indicator of soil uncover and open areas such as grasslands, shrublands, clear-cuts and roads. I extracted the enhanced vegetation index (EVI) and normalized difference vegetation index (NDVI) as indicators of vegetation cover. Land surface temperature (LST) is an indicator of environmental temperature. The normalized water index (NDWI) was extracted because it is associated with changes in vegetation water content and water absorption from tree canopies, serving as an indicator of deciduous vegetation (trees and shrubs). The normalized difference moisture index (NDMI) is associated with vegetation moisture and indicators of wetlands. I obtained the digital elevation model (DEM) to reflect the elevation of the terrain and I calculated the distance to water bodies (WATER) based on the known association of CAWA with riparian zones.

Another set of covariates were calculated from other data sources besides LANDSAT. I calculated the distance to mature coniferous forests (D_CONIF) using the Ontario Land Cover Compilation produced by the Ontario Ministry of Natural Resources (OMNR, 2016), publicly available through Ontario GeoHub (https://geohub.lio.gov.on.ca/). In addition, I obtained spatial layers with information on global forest canopy height (CAN; Potapov et al., 2021). To identify forest stands disturbance event (usually through logging but occasionally through wildlife; DISTURB), I used the Global Land Analysis and Discovery dataset from the University of Maryland (Hansen et al., 2013; https://glad.umd.edu /dataset/gedi/). All covariates were clipped using the Ecoregion 4W shape file freely available through the Ontario GeoHub webpage (https://geohub.lio.gov.on.ca/). All datasets, their resolution, and source information are given in Table 3.1.

Table 3.1. Potential input variables used in the Canada Warbler occurrence distribution model.

Variable	Description	Source layers	Data year	Resolution	Rights
BASI§	Bare soil index, an indicator of open areas (e.g., clear-cuts and roads, high values indicate greater open areas)	Landsat collection 2 level 1 (Landsat 8-9).	2018 (Jun & Aug)	30 m	USGS
CAN§	Canopy height, representing the tree height up to 30 m.	Landsat analysis-ready data time series.	2019	30 m	GLAD-UMD
D_CONIF§	Distance to coniferous forest	Ontario Land Cover Compilation v.2.0	2000	15 m (upscaled to 30 m)	OMNR
DEM§	Digital elevation model,	Global Multi-resolution Terrain	2010	30 m	USGS

Variable	Description	Source layers	Data year	Resolution	Rights
	representation of the bare ground topographic surface of the Earth.	Elevation Data 2010 (GMTED2010)			
EVI[§]	Enhanced vegetation index as an indicator of vegetation cover. Low values (<0.1) correspond to barren areas of rock, sand, or snow. Moderate values represent shrub and grassland (0.2 to 0.3), while high values indicate forested areas (0.6 to 0.8)	Landsat collection 2 level 1 (Landsat 8-9).	2018 (Jun & Aug)	30 m	USGS
DISTURB[§]	Year since stand-replacing disturbance, or a change from forest to non-forest state during 2000–2020, mainly due logging, but also due to wildfires (range 1-21, representing years since disturbance within the timeframe). Values of 21 represent no disturbance reported in the time series.	Landsat analysis-ready data time-series.	2021	30 m	GLAD-UMD
LST	Indicator of temperature	Landsat collection 2 level 1 (Landsat 8-9).	2018 (Jun & Aug)	30 m	USGS
NDMI	Normalized difference moisture	Landsat collection 2 level 1 (Landsat 8-	2018 (Jun &	30 m	USGS

Variable	Description	Source layers	Data year	Resolution	Rights
	index	9).	Aug)		
NDVI	Normalized difference vegetation index	Landsat collection 2 level 1 (Landsat 8-9).	2018 (Jun & Aug)	30 m	USGS
NDWI§	Normalized difference water index, >0.3 values indicate vegetation with high water content, and <0.3 values indicate low water content (deciduous or dry vegetation).	Landsat collection 2 level 1 (Landsat 8-9).	2018 (Jun & Aug)	30 m	USGS
WATER	Distance from water bodies, derived from the Canopy height layer.	Landsat analysis-ready data time series.	2019	30 m	GLAD-UMD

§Variables retained for the SDM of Canada Warbler after correlation analysis and model tests. USGS= U.S. Geological Survey, GLAD-UMD=, OMNR=Ontario Ministry of Natural Resources.

3.3.4 Model construction

I selected the algorithm Maximum Entropy (Maxent, version 3.4.1). Maxent estimates a target probability distribution by means of knowing the maximum entropy distribution and has the ability to handle observations collected using different protocols (Phillips et al., 2006). Maxent performs well compared to other modelling methods (Elith et al., 2006), even when few data are available on presence, and also Maxent does not require absence data (Hernández et al., 2006), making it useful in data-poor regions (Phillips et al., 2006).

I ran a preliminary model (2020), where I used the desktop-collected dataset from 2001-2020 (78 observations) and cross-validation with a random seed subset (25%) of the data. I ran a second model (2021) using desktop-collection dataset from 2001-2021 and the 2021 field observations (for a total of 122 CAWA observations) and cross-validation with a random seed subset (25%) of the data. For the final model (2022), I used the full dataset from 2001-2021 (N= 122), and I cross-validated with the field-collected data from 2022 (30 observations).

For both models I ran 10 cross-validation replicates and used a jackknife approach to test model fit and assess the model's prediction, expressed as the area under the curve (AUC) for both training and test data. The standard statistical method to assess model accuracy is derived from the Receiver Operating Characteristic (ROC) curves, created by plotting sensitivity (proportion of observed occurrences correctly predicted) against specificity (proportion of observed absences correctly predicted) for all possible thresholds (Pearson 2010). With the presence-only methods like Maxent, specificity cannot be calculated so the standard is to use pseudo-absences (random background points) instead of absences (Lahoz-Monfort et al., 2010). An AUC value of 0.5 shows that model predictions are not better than random; <0.5 are worse than random; 0.5–0.69 indicates poor performance; 0.7–0.9 reasonable/moderate performance; and >0.9 indicates high performance (Peterson et al., 2011).

In order to reduce the collinearity among the environmental covariates, I used QGIS 3.18 'grass' correlation analysis to select the pairs of variables with Pearson correlation coefficient values $r \geq 0.7$, and in those cases where sets of covariates were correlated, I used the covariate that better matched the geographic distribution of the species in preliminary model testing (Dormann et al., 2012). I tested all covariates for independence using a correlation matrix, which assesses relationships between variables (Appendix B, Table B1). Highly correlated covariates were NDMI with BASI and EVI, and NDVI with BASI and NDMI. I removed NDMI and NDVI from the first steps of model construction. I initially ran the models with all the remaining covariates, later identifying LST as a covariate that did not contribute to any of the models and in fact reduced its performance, so I removed it from the final modeling steps. For the preliminary model (2020) I also removed BASI as was detected as a covariate that did not contribute to the model and reduces its performance.

3.3.5 Field validation

To field-validate model accuracy in Ecoregion 4W, I developed a binary prediction from the initial SDM of 2021 (Appendix B, Figure B2). I divided the prediction into two classes of Maxent probability of occurrence due to restricted accessibility to many areas and for maximizing the detection probabilities for CAWA: high predicted occurrence (index >0.60) and low predicted occurrence (≤0.59). I field collected data during the breeding season of 2022 for both prediction classes, I randomly selected locations within 100 meters of main roads and with a minimum separation of 250-300m among point counts. I identified for sampling 42 sites with high predicted occurrence and 76 sites with low predicted occurrence. Model prediction accuracy was classified using an error matrix by comparing the prediction of the first model with the reference data collected on the ground in 2022. I used Cohen's Kappa (Cohen, 1960) coefficient (*K*) to compare the correspondence between predicted and observed occurrences:

$$K = Pa - Pe/1 - Pe$$

where *Pa* is the relative observed agreement (level of agreement among each category divided by the total number of observations) and *Pe* is the expected agreement by chance. Values below 0.4 are considered poor agreement, between 0.4 and 0.5 is the median agreement level, and greater than 0.6 is considered a high accuracy.

3.4 RESULTS

3.4.1 Model performance

Across 10 replicates, the preliminary model (2020) of probability of occurrence of CAWA showed poor performance for both training data ($AUC_{training}$ average 0.68) and test data (AUC_{test} average 0.65). The second model (2021) showed moderate performance for training data ($AUC_{training}$ average 0.75) and poor to moderate performance for test data (AUC_{test} average 0.69). The final model (2022) had a similar performance to the first model; moderate performance for training data ($AUC_{training}$ average 0.75) and poor to moderate for test data (AUC_{test} average 0.70). The model's projections are shown in Figure 3.2.

For the preliminary model (2020), I retained seven covariates (Table 3.2). The covariates with high mean percent contribution (>10%) included WATER (distance to water bodies), DISTURB (years since disturbance, mainly due to logging), EVI (enhanced vegetation index), and NDWI (normalized difference water index). The covariates that have a lower percent of contribution (<10%) were, DEM (digital elevation model), D_CONIF (distance to conifer forest), and CAN (tree canopy height). For the second model (2021), I retained eight covariates (Table 2); the covariates with high mean percent contribution (>10%) included NDWI, WATER, EVI, D_CONIF, DEM. The covariates that have a lower percent of contribution (<10%) were CAN, DISTURB, and BASI (bared soil index).

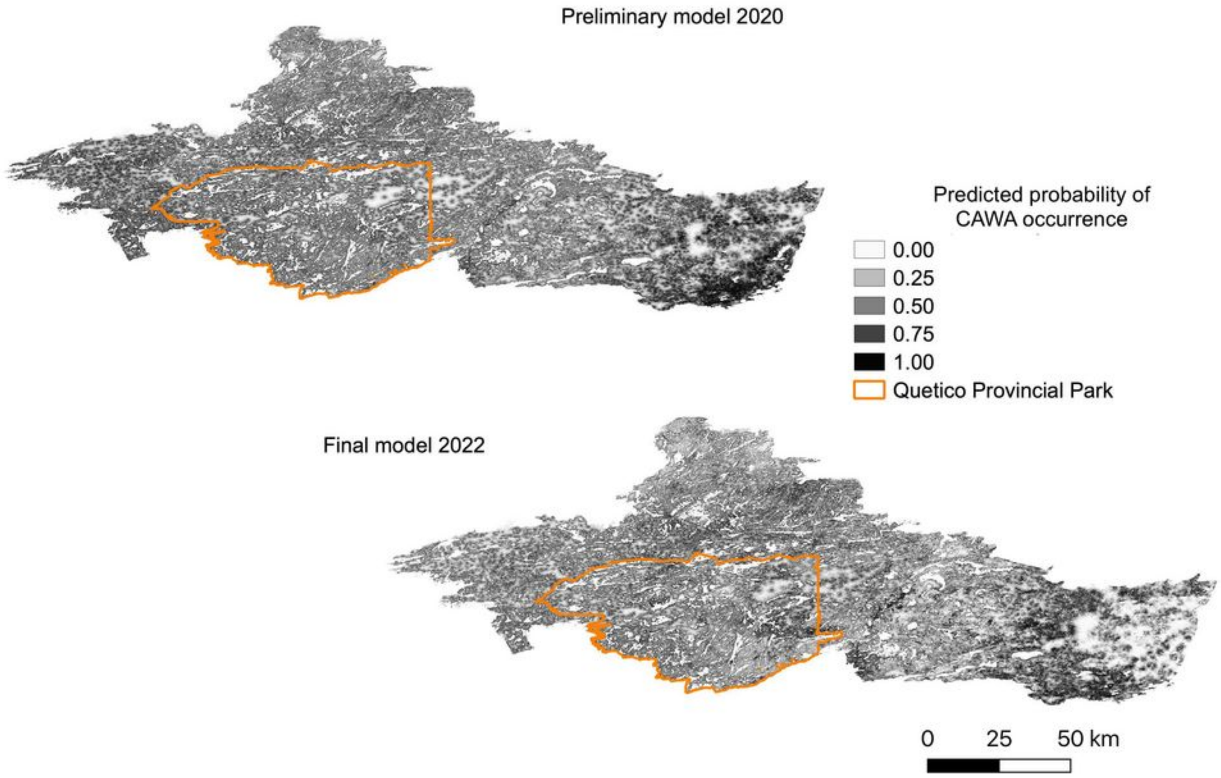

Figure 3.2. Distribution map of predicted occurrence probability of Canada Warbler (CAWA) from the preliminary (2020) and final model (2022) in Ecoregion 4W. Darker areas denote >70% probability of Canada Warbler occurrence, and gray to white areas indicate <50% probability of occurrence.

The final model (2022) used the same covariates as the 2021 model (Table 3.2). The covariates which contributed more to the model (>11%) where NDWI, WATER, EVI, and D_CONIF. The covariate that contributed most to the final model was NDWI, where the prediction of CAWA decreases while higher vegetation water content increases (Figure 3.3A). The second covariate was WATER, the CAWA probability of occurrence increases closer to the water bodies (Figure 3.3B). EVI, the third covariate that contributes most to the model, predicted increases in CAWA with increases in EVI, but decreases where the value surpasses 0.6 (Figure 3.3C), indicating a high association with shrubs and medium-density forested areas. The final covariate that contributes more to the model was D_CONIF, whereby probability of CAWA occurrence CAWA decreases with increasing distance to the coniferous forest (Figure 3.3D).

Table 3.2. The average contribution of covariates to model prediction, permutation importance, training gain, and test gain across 10 cross-validated Maxent model runs predicting the probability of occurrence of the Canada Warbler in the ecoregion 4W. SDM year refers to the preliminary model (2020, desktop-collected data only) and the second and final models (2021 and 2022, including desktop and field-collected data). Variable description and codes are in Table 3.1.

SDM year	Variable	Average % contribution	Average permutation importance	Average training gain contribution rank	Average test gain contribution rank
2020	WATER	49.3	46.0	1	1
	DISTURB	12.2	15.7	3	5
	EVI	11.3	11.7	2	2
	NDWI	10.5	10.4	7	7
	DEM	9.5	13.8	5	4
	D_CONIF	6.5	2	4	3
	CAN	0.6	1.1	6	6
2021	NDWI	23.1	29.0	1	2
	WATER	21.3	19.9	2	1
	EVI	18.9	13.7	3	3
	D_CONIF	12.7	11.9	5	3

SDM year	Variable	Average % contribution	Average permutation importance	Average training gain contribution rank	Average test gain contribution rank
	DEM	10.4	13.1	7	7
	CAN	7.9	5.0	4	4
	DISTRUB	2.9	2.0	8	6
	BASI	2.9	5.3	6	5
2022	NDWI	25.4	25.1	2	2
	WATER	22.8	19.0	1	1
	EVI	13.7	18.3	3	3
	D_CONIF	11.8	10	5	4
	CAN	11.3	7.9	4	5
	DEM	9.6	15.2	7	7
	DISTURB	2.8	0.5	8	8
	BASI	2.6	4.1	6	6

Covariates with a lower mean percentage contribution to the model (<11%, Table 2) were CAN (tree canopy height), DEM (the digital elevation model), DISTURB (years since disturbance by logging), and BASI (bare soil index). The response curve of CAN (Figure 3.4B) indicates CAWA probability of occurrence increases with the tree canopy height. According to the DEM, the highest predicted probability of occurrence of the CAWA (above 0.8) is at 400 m elevation (Figure 3.4A). The response curve of DISTURB shows CAWA had a higher predicted probability of occurrence is in areas where has happened within 6 years since disturbance and also in areas where no disturbance has happened during the last 20 years (Figure 3.4C); the observations of CAWA found in areas where the years after disturbance had occurred is reported in Appendix B, Table B3. Finally, the covariate that contributed least to the model was BASI (<0) indicates association to high shrub density (Figure 3.4D).

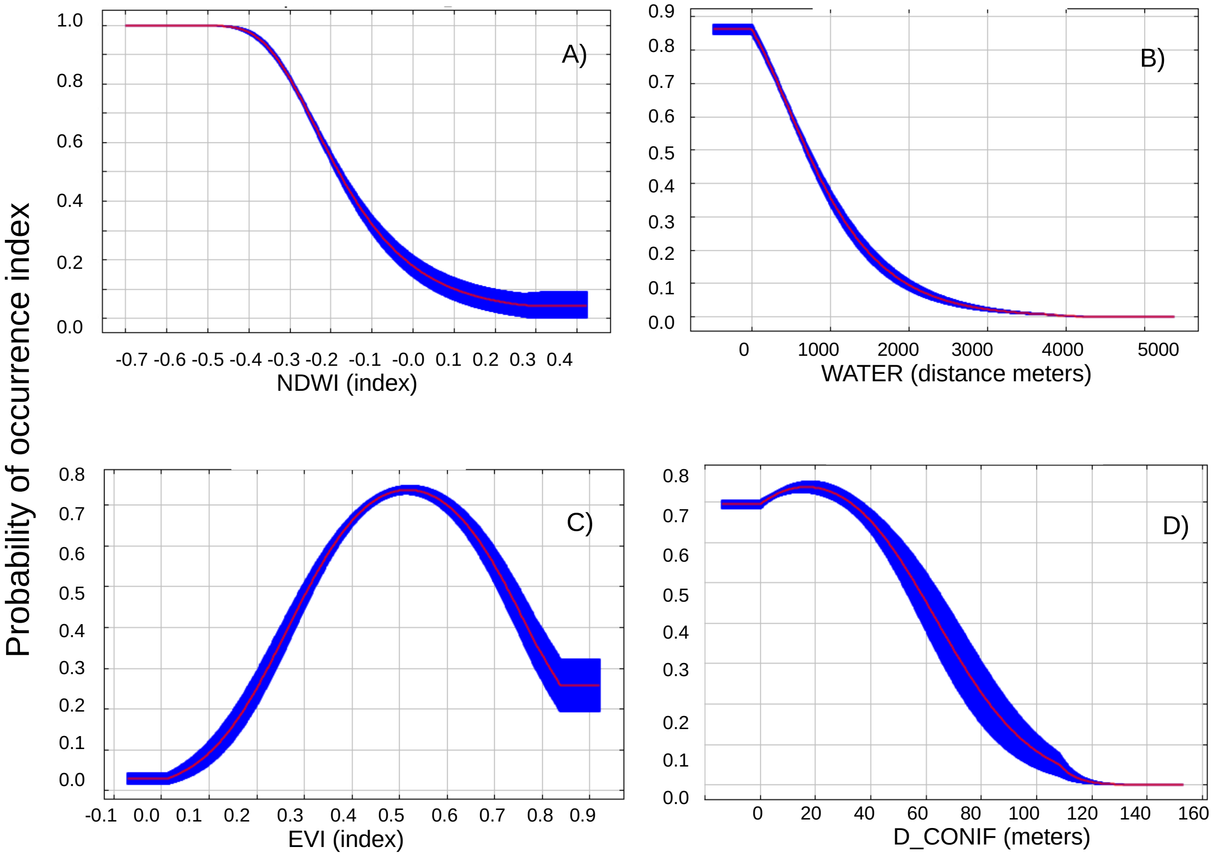

Figure 3.3. Mean response curves for the occurrence of CAWA for covariates that contributed greater than or equal to 11.5% of the final model's explanatory power. Curves show how the predicted probability of presence as each covariate varied in the full model retaining all covariates. Table 2 shows the average contribution and permutation importance for each covariate. The mean response of 10 replicates is in red, and the mean +/- one standard deviation is in blue.

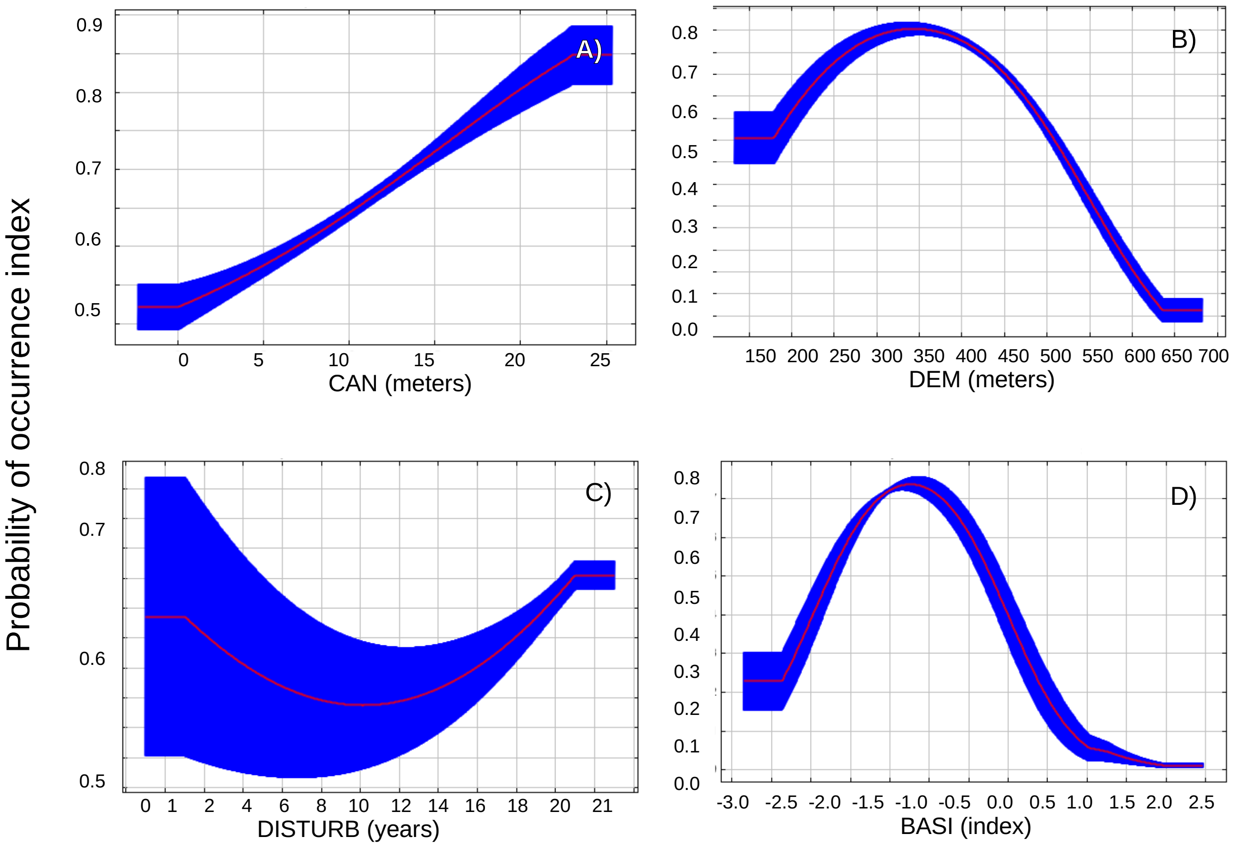

Figure 3.4. Mean response curves for the occurrence of CAWA on covariates that contributed less than 11.5% to the final model. Curves show how the predicted probability of presence as each covariate varied in the full model retaining all covariates. Table 2 shows the average contribution and permutation importance for each covariate. The mean response of 10 replicates is in red, and the mean +/- one standard deviation is in blue. DISTURB (years since last forest disturbance mainly by logging) 1-21 is the time (years) since disturbance within the period of 2000-2020; 21 value represents no disturbance reported within the time period series.

3.4.2 Field validation results

From the total 118 sites surveyed in 2022, I identified 94 matches for both categories (low and high predicted areas), achieving 66.94% total accuracy. In areas where the preliminary model showed high probability of occurrence of the CAWA (index >0.60), I observed an accuracy of 42%, and identified CAWA males in 23 of the 54 surveyed total sites. The accuracy was 87.5% in sites with a low prediction of occurrence; I observed CAWA males only in 8 sites (12.5%) from a total of 64 sites with low model prediction. The two observed class conditions received a Kappa value of 0.42, indicating agreement better than expected by chance.

3.5 DISCUSSION

This is the first study to develop a high-resolution field validated SDM for CAWA in Northwestern Ontario. The AUC value (0.68-0.75) was acceptable, and the field validation process also indicates that the model performed better than expected by chance. The AUC value in the model was similar to the Maxent model developed by Bale et al. (2020) for Nova Scotia (AUC = 0.7). Even though the dataset has a limited number of training observations ($N = 122$), the moderate performance may be due to the inclusion of systematic field-collected observations in the training dataset, the validation process through field-collected observations, and the use of more current Landsat images to develop the indexes used as covariates. The majority of CAWA models have been done solely by large datasets of observations collected mainly by volunteers, who include highly knowledgeable birders but also novices who can misidentify, overestimate or under-detect the species in the field. There was a high chance of species under-detection by the observers who registered their observations in eBird, OBBA and BBS, a reason I did not try to infer absences records from these datasets. This can be noticed through the poor performance (AUC <0.7) of the preliminary model (2020) constructed mainly with those datasets.

Previous studies on CAWA habitat in the eastern portion of this species' range have identified vegetated wetlands and moist forests as important habitat types for this species (e.g., Goodnow and Reitsma, 2011; Westwood et al., 2019a). I observed a high prediction of occurrence closer to water bodies (WATER), which may indicate that CAWA are more likely to occur in proximity wet and riparian areas. The association with the NDWI was mostly negative, indicating a strong relationship with deciduous trees and shrubs. Also, the response to bare soil (BASI) and the association with EVI suggested that relative open forested areas and denser understory are important predictors for CAWA occurrence similar to what Bale et al. (2020) found and confirming previous findings (e.g., Becker et al., 2012). Bale et al. (2020) also found that the CAWA was associated with a relatively close distance coniferous stands, being consistent with the model prediction that the occurrence of the species was higher in areas closest to coniferous forests. This may reflect a genuine association with habitats near to coniferous forests or may reflect the patchy nature of deciduous and/or mixedwood forests that occur in a heterogenous matrix among coniferous forest.

Overall, the results for canopy height (CAN) were consistent with those observed by Haché et al. (2014), where CAWA densities were generally higher in areas with tall trees (a proxy for forest age). Though canopy height was not one of the strongest predictor variables, taking this in combination with associations with mixedwood forest, I caution that CAWA may be vulnerable to short-rotation even-aged forest management, which reduces the availability of old forests on the landscape and promotes the conversion of mixed and coniferous-dominated forest landscapes to forests with broadleaf dominance (Drapeau et al., 2000; Hobson and Bayne, 2000; Imbeau et al., 2001; Schieck and Song, 2006).

The final model predicted there is a high probability of occurrence of CAWA (>0.7) in areas with no forest disturbance (Figure 3.4C). These results are consistent with patterns observed in a study in Northern Minnesota, where Canada Warbler was more common in a wilderness forest compared to a managed forest (Zlonis and Niemi, 2014). However, there are also high probabilities of occurrence (>0.6) in areas where the forest has been managed for timber extraction within six years post-harvest (Figure 3.4C, Appendix B) or post-fire. Other studies reported a high density of CAWA was found in postharvest cuts (regenerated areas >5 years since disturbance; Ball et al., 2016; Hunt et al., 2017) and light partial harvest (Becker et al., 2012), indicating that the species can take advantage of logged and post-fire regenerated areas with high shrub density, this may be caused by regional variation in habitat associations (Crosby et al., 2019; Leston et al., 2023) or differences in the types of harvesting and silvicultural techniques used in different jurisdictions.

3.5.1 Limitations in model application

This species has been reported as likely influenced by conspecific attraction (Hunt et al., 2017; Flockhart et al., 2017), and the model prediction was not able to integrate the influence of this behavior. I could have created a covariate which served as a proxy for conspecific attraction such as “distance to nearest other CAWA observation.” However, given the small number of occurrences across a large area (*N* = 122) and the strong level of spatial bias in some of the occurrence dataset, I felt the level of accuracy in such a layer was too low to warrant its inclusion. Furthermore, density alone can be misleading and not necessarily indicate habitat quality (Van Horne, 1983). I encourage managers to perform a field validation of the habitat suitability prediction (e.g., Westwood et al., 2019b) for the species, because high congregation of individuals on postharvest areas and low abundance in protected areas could not necessarily reflect the habitat quality of the CAWA.

Part of the aim of this study was to develop a high resolution SDM for what I need to use remote sensing data such as Landsat. This method also is particularly relevant in cases where other traditionally used environmental variables are difficult to collect or not relevant for the extent of the area of study (Lahoz-Monfort et al., 2010), such as bioclimatic variables which are commonly used at larger spatial extents (e.g., Stralberg et al., 2015; Cadieux et al., 2020). However, the use of remote sensing images for specific or a limited number of years may reduce the application of the model, because the predictions will be closely tight to the environment conditions of the time were the images were obtained. Another issue with Landsat is the persistent cloud cover in certain regions, more so than other sensors of its 16–18-day revisit period (Wijedasa et al., 2012); solving this problem may be possible but it is time consuming, and it is necessary specialized skills in GIS which majority of researchers and land managers don not often have. Despite its limitations, Landsat imagery has good qualities that help monitoring regional changes to forest habitat and biodiversity (Wijedasa et al., 2012).

One of the greatest limitations was that available occurrence data for this area is scarce, and with several inaccuracies (wrong coordinates, duplicated identifications, etc.), it is important to encourage volunteers and community naturalists to not only collect more observations, but also to verify their observations and locations before they upload them to eBird or OBBA datasets. Also, more systematic bird survey research will be beneficial to increase sample size and bird population knowledge. Moreover, avian surveys using autonomous recording units (ARUs) can overcome major limitations experienced by point counts methods including site access limitations associated with remote locations as well as the disruption of surveys due to inclement weather (Drake et al., 2021). ARUs can provide long-term and systematic species monitoring data for elusive species such as Canada Warbler; Quetico Provincial Park is carrying out its bird surveys through the use of ARUs, but the limited equipment, staff and accessibility still affect the number of sites at which ARUs are located. The use of ARUs in protocols such as OBBA, BBS and even for forestry managers could increase species records and improve the accuracy of SDMs developed in the region.

3.5.2 Model accuracy and implications for management

Field validation demonstrated that the model accurately predicted areas delimited at medium-low occurrence (87.5%). Thus, when using model results to identify locations to conserve or manage habitat for CAWA, users can reliably exclude all areas with a predicted probability of occurrence ≤0.59 from the search. Though the accuracy of the model when predicting high occurrence areas (≥ 0.60) was not exceptional, these areas can be identified for further ground truthing. I reiterate past calls for developers of SDMs to field-validate their models, as this step not only provides assurances of the accuracy of the model, but also it can make it simpler for managers to implement conservation and management plans with more confidence.

The number of occurrences of the CAWA particularly in Quetico Provincial Park are low not only due to the inconspicuous characteristic of the species, but also due to low accessibility to the park, where the main interests among visitors are canoeing and fishing, not recording bird observations. Quetico Provincial Park will benefit from increasing interest in visitors making bird observations and uploading them into public datasets (e.g., iNaturalist, eBird). The present species distribution model could be used to guide both park staff and visitors to search for CAWA within the protected area in areas with higher probability of occurrence. However, it is important to encourage users of community science platform the verification of locations coordinates and species identification before submitting birds record, it will reduce bias and improve quality and accuracy of the data available to use for modelling.

Critical habitat identification for species at risk in Canada is mandated (Government of Canada, 2023). The results from the study could be important to land managers and logging companies in the ecoregion that need to consider the protection of species-at-risk in land use planning. At the site level, when conducting forestry or land-clearing activities during the breeding season, operators should search all areas with a probability of occurrence >0.60 for CAWA prior to incurring disturbance to mitigate potential impacts to this listed species at risk. In terms of landscape-level harvesting plans, it is necessary to preserve large, unharvested fragments or implement the retention of important forest features for the species during forest management, such as tall trees. The model predicted CAWA has mid to high-probability of occurrence in many post-harvested areas, the species is still relying on features associated with old-growth forest. In Ontario 86% of harvested forests are under a clearcut silviculture system; then shifting to a more uneven-aged forest management system will be beneficial to CAWA conservation.

CHAPTER 4. INFLUENCE OF BOREAL FOREST DISTURBANCE AND CONSPECIFIC ATTRACTION ON CANADA WARBLER (CARDELLINA CANADENSIS) HABITAT CHOICES DURING THE BREEDING SEASON[2].

4.1 ABSTRACT

Understanding how, when, and why species select habitats is essential to identify more accurate conservation strategies, particularly with increasing anthropogenic change. However, studies rarely disentangle the roles of environmental cues and social information when they examine habitat selection. I tested the influence of conspecific attraction and habitat disturbance on the occurrence and habitat choices of the Canada Warbler (*Cardellina canadensis*, CAWA) in logged landscapes. I surveyed a gradient of disturbed landscapes during the 2021, 2022 and 2023 breeding seasons in Northwestern Ontario. I surveyed naturally occurring social aggregations of CAWA, as well as simulated conspecific attraction by using playbacks of CAWA songs as an artificial cue during the pre-breeding season. I used generalized linear models to examine the influence of vegetation structure (shrub and canopy cover, canopy height and forest type), level of disturbance, and the song cues on the occurrence of CAWA. Shrub cover >55% appeared as one of the main cues that might influence habitat choice in CAWA, with canopy cover and canopy height also playing a role. Disturbance at the local scale influenced occurrence and conspecific aggregation, whereas landscape-scale disturbance did not have a strong influence on CAWA patterns. Natural aggregations were clustered in areas with higher disturbance.

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[2] A previous draft of this chapter has been submitted for publication in the journal *Avian Conservation and Ecology* and will be considered to be accepted pending revisions. Co-authors are Cupiche-Herrera, V., B. McLaren and A. Westwood. In addition part of the methods of this paper were published in the journal *Hardware X:* Cupiche et al., 2023, https://doi.org/10.1016/j.ohx.2023.e00418.

Conspecific songs during the pre-breeding season thus appear to be a cue to males searching for breeding territories; likely because of these cues, CAWA males are able to occupy new sites in areas with a low level of disturbance. As I observed clustered occurrence mainly in disturbed habitats, it is possible that conspecific attraction may lead to selection of habitats with lower productivity and poorer outcomes that could act as “ecological traps.”

4.2 INTRODUCTION

Knowledge of how a species selects its breeding habitat allows for identification and management of that habitat on local and landscape scales. For forest-dwelling birds, vegetation cues directly signal ultimate factors such as nest site and food availability, and risks of predation and parasitism (Hildén, 1965). Information about proximate factors is commonly used in models to predict a species’ habitat or distribution across a landscape, which are then used to inform conservation and management decisions. Whereas most studies modelling forest-dwelling birds use environmental variables representing landcover, and vegetation composition and structure to represent habitat (e.g., Zlonis et al., 2017; Westwood et al., 2019a), very few studies also incorporate social cues, such as the influence of intra- and interspecific interactions (Campomizzi et al., 2008). Nevertheless, interactions among organisms are fundamental to determining their realized niche, and thus overwhelmingly likely to influence their distribution. Disentangling the relative roles of environmental cues and social information in habitat selection is essential to identify more accurate conservation strategies for habitats and species, particularly with increasing anthropogenic global changes (Thomas et al., 2001; Campomizzi et al., 2008).

A primary mechanism driving clustered distributions in species is conspecific attraction. Researchers have suggested many reasons that individuals may select habitat based on the presence of conspecifics, including finding more potential mates, benefits of group vigilance, and use of conspecifics as indicators of habitat quality (e.g., resource type and quality, previously successful breeding; summarized by Muller et al., 1997). Although vegetation cues require direct sampling of possible sites by individuals before selecting a breeding location, locating conspecifics may be a more efficient means of site assessment (Hildén, 1965; Cornell and Donovan, 2010; Valente et al., 2021). Birds, particularly migratory birds, have short breeding seasons, so decisions related to habitat selection must be made quickly, and their breeding communities must reassemble themselves annually (Nocera and Betts, 2010). Fletcher and Sieving (2010) reconcile the use of social information (social cues) in landscape ecology studies; landscape features can change the accessibility and value of social information, and this interaction may profoundly affect the efficiency and outcome of habitat selection.

Species density is frequently used to infer habitat quality under the assumption that individuals occur at greater densities in better quality habitats, or in areas where they can achieve higher survival rates and reproduction rates (Fretwell and Lucas, 1969; Doligez and Bouliner, 2008). However, density alone can be a misleading proxy for habitat quality, and social interactions such as conspecific attraction could lead to an aggregation of individuals from the same species even when the habitat is unsuitable (Van Horne, 1983). This behavior has been postulated in the case of Canada Warbler (*Cardellina canadensis*, CAWA), a species at risk in Canada (Environment Canada, 2016), where it occurs in logged stands in the western boreal forest (Flockhart et al., 2016; Hunt et al., 2017). The CAWA breeding season and its habitats are reasonably well described, but they vary regionally, particularly with different types of land use, forest types, and forest disturbances (Haché et al., 2014; Ball et al., 2016). While there is evidence that habitat use by CAWA in harvested forests is influenced by conspecific attraction, its breeding success is also lower in areas with high conspecific density (Flockhart et al., 2016; Hunt et al., 2017). This evidence suggests that conspecific density might be higher in harvested than unharvested landscapes. However, this hypothesis has not been tested either in an undisturbed habitat or at the landscape scale.

The use of artificial conspecific cues (song playbacks) has been proposed to test the influence of conspecific attraction in habitat choices (e.g., Betts et al., 2008; Cornell and Donovan, 2010; Albrecht-Mallinger and Bulluck, 2016), and as a conservation strategy to promote settlement of a species in more suitable areas (Schlossberg and Ward, 2004). Previous studies used artificial conspecific cues to attract CAWA, but their protocol included calls of shorter duration to increase detectability when the species was already expected to be breeding (Flockhart et al., 2016; Hunt et al., 2017; Westwood et al. 2019a). I instead assess the influence of conspecific cues in habitat choices through the use of conspecific playbacks during the pre-breeding season (at least seven days before breeding). I predict that (1) where playbacks are not used, individuals will be more attracted to the sites within disturbed landscapes with clusters of other CAWA, and (2) where continuous playback is used in undisturbed habitat, CAWA will preferentially select those habitats. I evaluated three proximate cues that may be important for habitat choices made by CAWA: (1) disturbance of the territory and the surrounding landscape, (2) shrub cover, canopy tree cover, and tree height within each territory, and (3) the presence of vocal conspecific cues.

4.3 METHODS

4.3.1 Study area

We conducted the study in three Ontario Breeding Bird Atlas (OBBA) squares (10 x10 km) within Ecoregion 4W (Pigeon River), which is located in the Ontario Shield Ecozone in Northwestern Ontario (Figure 4.1). Mixed forest is the most extensive land cover class (33.2%), followed by sparse forest (19.3%), water (17.5%), coniferous forest (11.5%), deciduous forest (10.6%), and cutovers (recently logged forest, 3.6%; Crins et al., 2009). Predominant land uses include timber harvesting, resource-based tourism, mineral exploration, and agriculture; the city of Thunder Bay is the only urbanized community in the ecoregion (Wester et al., 2018).

I selected sample squares based on information from the previous OBBA 2001-2005 (Cadman et al., 2007), after pilot visits during May of 2021, and after identifying disturbed or undisturbed areas from the global forest change layer from 2000-2020 (Hansen et al., 2013), publicly available through Global Land Analysis and Discovery webpage from the University of Maryland (https://glad.umd.edu/dataset/gedi/). Disturbance was classified in QGIS 3.18 as unlogged, early (0–7 years since logging), mid (9–14 years since logging), and late (18–22 years since logging). I identified the number of areas with less than 20 years of disturbance, and also considered any activity observed in the field (e.g., current logging). With this information, I then selected three squares, one each to represent low (≤50 % logged), mid (50-60% logged), and highly disturbed landscapes (>60% logged).

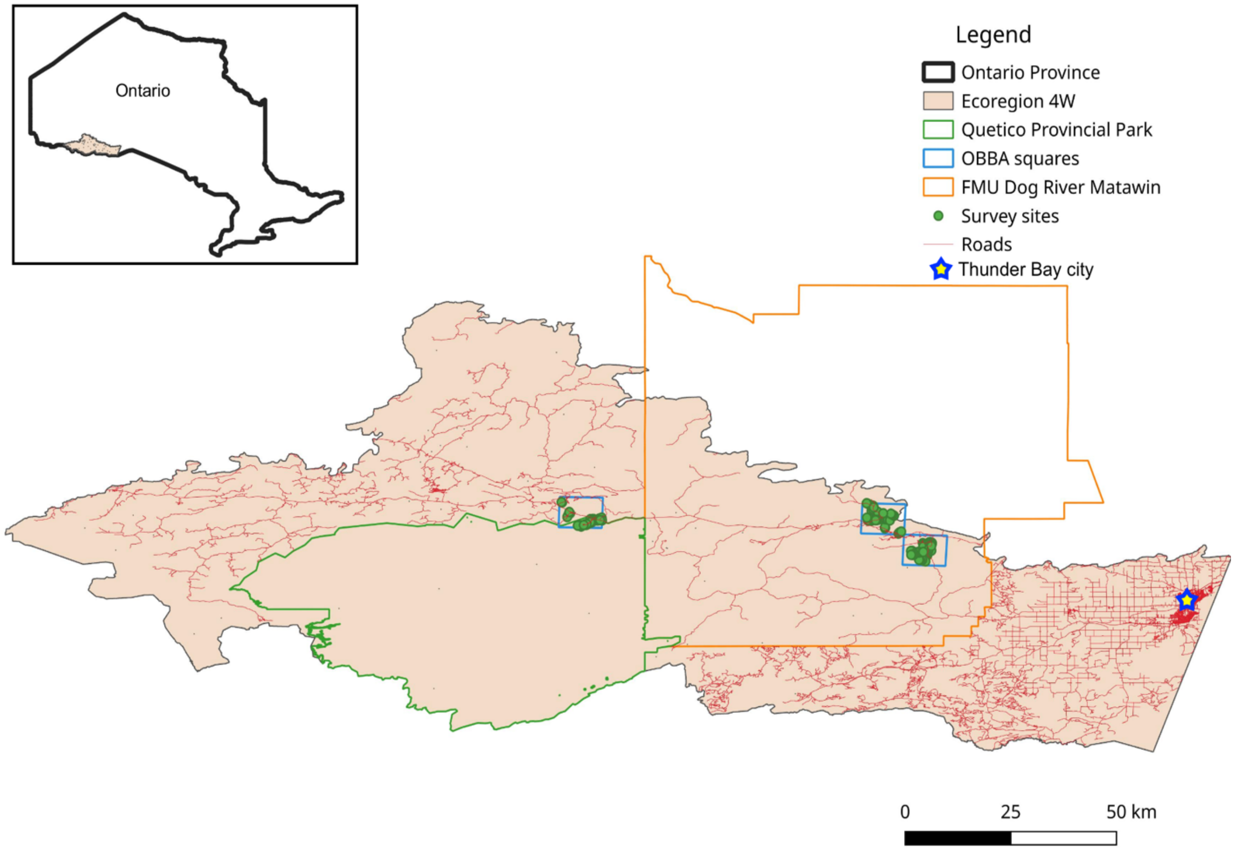

Figure 4.1. Ecoregion 4W and Ontario Breeding Bird Atlas squares selected for the present study. One square is located in Quetico Provincial Park and the other two squares in the Dog River Matawin Forest Management Unit (Kabaigon on the left and Shebandowan on the right).

The square considered to represent the "low disturbance" landscape was located in part of Quetico (Gwetaming) Provincial Park (Figure 4.1). Quetico encompasses 4,718 km^2 with numerous lakes and streams (OMNR, 2018). Quetico is located in Northwestern Ontario, south of the town of Atikokan, approximately 160 km west of Thunder Bay, and adjacent to the Canada-United States (U.S.) boundary. The park occupies a zone of transition between boreal forests to the north, mixed forests to the south and Great Plains forests to the west and southwest. The other two squares were located in the Dog River Matawin Forest Management Unit (Figure

4.1), which is managed by Resolute, Inc. in a long-term (2020-2030) lease. The mid disturbance landscape was located in Kabaigon, where forest management was done during a previous planning period (before 2020), and the "high disturbance" landscape was in Shebandowan, where there are recent post-logged areas (<20 years) and current timber extraction.

4.3.2 Bird surveys

4.3.2.1 Occurrence observations

All survey methods were approved by the Lakehead University Animal Care Committee (Animal Utilization Protocol #1468679). I conducted bird point count surveys in each square during the breeding season from late May to early July of 2021 and 2022. I located a total of 145 point counts randomly within each year since logging category: unlogged N=83, early N= 37, mid N= 11, late N= 14 points; 48 were located in Quetico, 48 in Kabaigon, and 49 in Shebandowan, and points were approximately 250-300 m apart from each other. The observations were conducted in favorable weather conditions, never during precipitation events or with wind > 20 km/h, both of which reduce the detectability of singing birds (Cadman et al., 2007). After a five-minute point count, the observers (commonly two) used speakers or mobile devices to play CAWA songs to increase the detectability of the species and to register the number of individual conspecific responses.

The observers registered the total number of territorial males detected by sight or sound at each point count using the following protocol: 1) 30 s of conspecific playback; (2) 1 min of silence, (3) 30 s of playback, (4) 1 min of silence (protocol modified from Flockhart et al., 2016;

Hunt et al., 2017). The repetition of the playback and the silent periods helped to reduce the bias of artificial calls and the possible effect of individuals approaching just out of curiosity. The observers counted only the males that responded during or after the second playback period of conspecific vocalizations.

4.3.2.2 Conspecific attraction settlement experiment

With information from 2021 and 2022 bird surveys, we selected twelve vacant sites in each square during the breeding season of 2022 and 2023 to implement an experimental playback protocol adapted from Betts et al. (2008) and Cornell and Donovan (2010) and followed the recommendations from Ahlering et al. (2010). I selected the sites using a gradient of shrub cover from high to low to assess the influence and importance of shrub cover in CAWA settlement. As a control or reference sites, 12 vacant sites were visited within each square, each located 250-300 m from the treatment sites. From late May to mid-June of 2022 and 2023, I installed on each treatment site an automated speaker system (Figure 4.2; Cupiche et al., 2023), which played CAWA vocalizations for eight hours daily (from 05.00 to 13.00) over an entire seven-day treatment period. To limit habituation to playbacks, the soundtracks contained 30 s gaps between vocalizations; I also programmed 10 min of continuous silence between soundtracks, and the songs and calls were alternated every 20 min.

I used same short playback procedure as the occurrence survey (see previous section) in each control point count to detect whether CAWA was present. Also, I applied this protocol when re-visiting the treatment locations after the seven-day period to detect whether CAWA was present at the sites. During the breeding season I re-visited treatment sites at least two times after the treatment period and I re-visited at least one more time each control site.

4.3.3 Vegetation survey

To assess potential vegetation cues for breeding site selection, the following vegetation data were collected in all point counts: forest type (conifer, mixed, deciduous), canopy tree cover, mean canopy tree height, and shrub cover. The percentage of canopy tree cover (trees >5 m) and shrub cover (1-5 m) were measured with a spherical densiometer and the app Canopy Capture (2018) in a radius of 50 m. I collected the data at the end of June and early July of each year of study (2021, 2022, 2023).

Figure 4.2. Automated speaker system (Cupiche et al., 2023) installed for a minimum of seven days on each site. Left picture: speakers installed in the field. Right image: an overview of the speaker system used (microcontrollers, speakers, and battery inside a waterproof box).

Using QGIS 3.18, I extracted canopy tree height average from each site through the canopy height 30 m resolution layer from the global forest (Potapov et al., 2021), publicly available through the Global Land Analysis and Discovery webpage from the University of Maryland (https://glad.umd.edu/dataset/gedi/). Also, to identify the forest type in each site I used observations in the field, as well as the Ontario Land Cover Compilation (15 m resolution layer) produced by the Ontario Ministry of Natural Resources (OMNR, 2016), publicly available through Ontario GeoHub (https://geohub.lio.gov.on.ca/); I recorded coniferous forest when the conifer tree species covered ≥75% of the site, deciduous forest when deciduous tree species covered ≥75% of the site, and mixedwood forest when conifer and deciduous tree species each covered 50% of the site.

4.3.4 Data analysis

I modeled CAWA occurrences, conspecific attraction (sites where I detected more than one individual), and settlement (response to the artificial conspecific cue during the pre-breeding season) against potential predictors (Table 4.1): year, site disturbance, landscape disturbance, shrub cover, canopy cover, canopy height, forest type, and treatment (the latter only used as a predictor for CAWA settlement). I used a generalized linear model with a logistic function structure using the package lme4 in the program R, version 3.6.3 (R Core Team, 2021) to do the analysis. For numerical variables I did a correlation analysis to reduce collinearity among them (Appendix C, Tables C1, C2). I fitted the models using different combinations of variables to test which variables best predicted CAWA patterns; I used an information-theoretic approach to select the most parsimonious model by contrasting values of Akaike's Information Criterion corrected for small sample sizes (AICc; Burnham and Anderson, 2002). I contrasted the outcome of the three models to determine the influence of conspecific and vegetation cues on the presumed CAWA habitat choices.

4.4 RESULTS

During 145 ten-minute point counts conducted in 2021 and 2022, I counted 54 presences and 84 total CAWA male individuals. In 2021, I counted 33 presences and 53 individuals, and in 2022, we detected 21 presences and 31 individuals. During the playback experiment, we detected a total of 13 individuals in treatment sites where the conspecific playback cues were implemented, seven individuals in 2022 and six in 2023. I only detected three individuals in control sites (silent/no playback) during the 2023 experiment period.

Table 4.1. Variables used to develop the generalized linear models predicting CAWA occurrences, conspecific attraction, and settlement.

Variable	**Unit**
Shrub cover	%
Canopy cover	%
Canopy tree height	meters
Forest type	Coniferous, Mixed, Deciduous
Year	Surveyed years 2021, 2022, 2023
Site disturbance (60 m radii)	Disturbed/undisturbed
Landscape disturbance (10 x10	Low, mid, high disturbance

Variable	Unit
km square)	
Treatment	Speakers/control
Time since logging	Years category

Table 4.2. Vegetation data collected during the bird surveys and playback experiment (mean ± SE) in sites where CAWA absences and occurrences were detected.

Vegetation cues	Absences (experiment)	Occurrences (experiment)	Absences (point counts)	Occurrences (point counts)
Shrub cover (%)	35.9± 2.1	59.7±1.3	39.8 ±1.8	58.5±1.2
Canopy cover (%)	63.9±2	57.9±1.6	60.75±9.5	55.6±7.3
Canopy tree height (m)	12.8±0.4	13.0±0.6	11.7±0.4	12.3±0.4

Occurrences in both point counts and experimental sites were higher in areas where shrub cover was >55% (Table 4.2), and the majority of occurrences and abundances were in mixedwood forest, followed by deciduous forest, but conspecific aggregation was slightly higher in deciduous than in mixedwood sites (Figure 4.3, A). Whereas I observed more occurrences in undisturbed sites (N = 32), abundances (number of individuals observed) were higher in disturbed sites at both local and landscape scales (Figure 4.3, B and C); conspecific aggregation was also higher in disturbed areas, specifically in recently logged areas (<9 years; Figure 4.3, D) From a total of 22 sites within the three squares where I found ≥2 individuals, 16 locations

(75.8%) were in logged areas or forested patches within logged areas; meanwhile, from a total of 32 sites where I registered one individual, 22 (84%) were recorded in undisturbed sites.

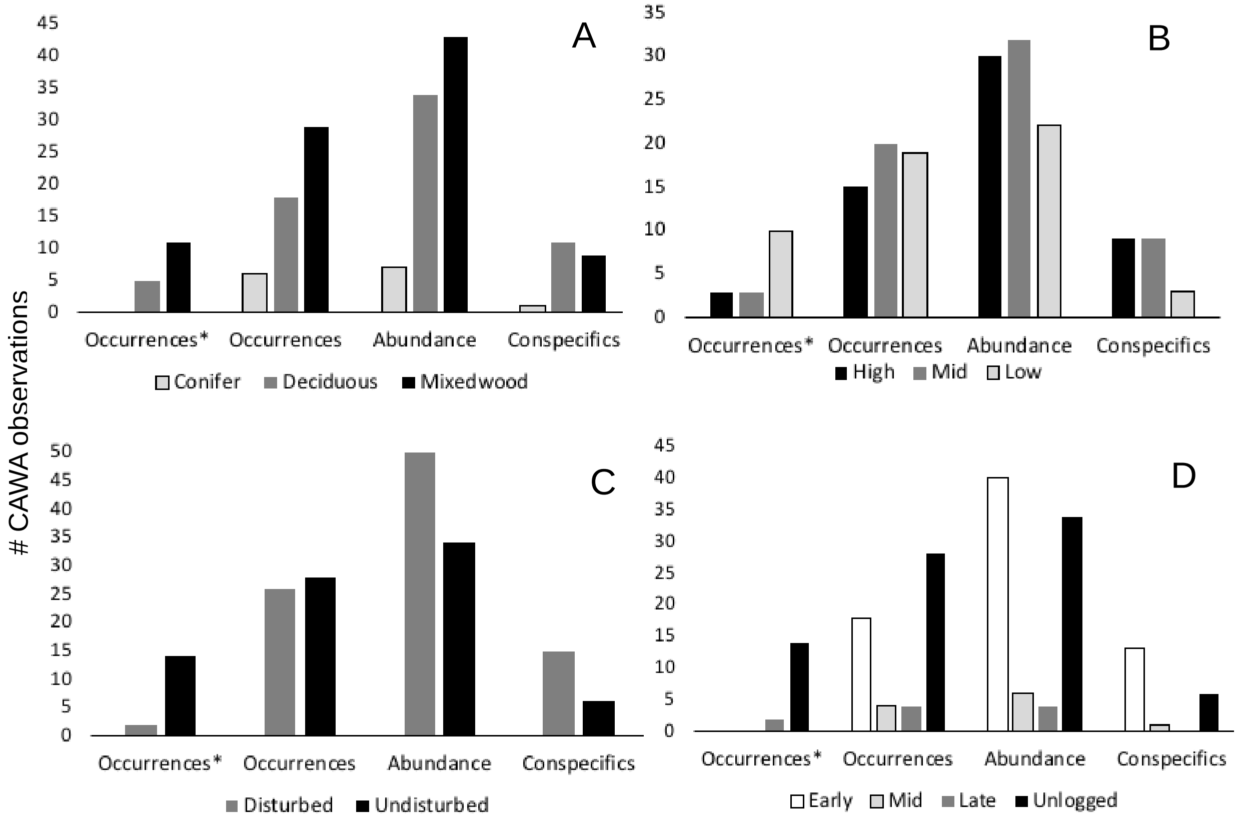

Figure 4.3. Canada Warbler (CAWA) observations, occurrences, abundance, and conspecific aggregation (≥2) detected in relation to (A) forest type, (B) disturbance level per square (landscape scale), (C) disturbance category per site (local scale), and (D) time since logging categories. Occurrences marked with * are those from the playback experiment.

In all top ranked models sets, shrub cover was a predictor (Table 4.3; Appendix C, Table C3). Canopy cover and canopy height were other important predictors for occurrence and conspecific aggregation (Table 4.3). Disturbance conditions at the local (site) scale influenced CAWA occurrence and settlement, while the same influence did not occur at the landscape (BBS square) scale. Time since logging appeared in the top models of conspecific aggregation and settlement. Conspecific vocalizations in pre-breeding season were an important predictor for CAWA settlement patterns.

Table 4.3. Best set of models to assess CAWA occurrence and conspecific attraction during the bird surveys and playback experiment in the breeding season (ΔAIC ≤2; Burnham and Anderson, 2002). Models are shown with their AIC values and Akaike weights (ωi). Time since logging was used as random effect. Shr = shrub cover, yr = year of study, cancov = canopy cover, can_h = canopy height, for_type = forest type (coniferous, deciduous, mixewood), sitedisturb = disturbance condition in surveyed site (disturbed/undisturbed), t_logg = time since logging (early, mid, late, unlogged), treat = treatment (speakers/control).

Model	**AIC**	**ωi**
A) Occurrence		
Probability of occurrence		
Shr+yr+cancov+for_type	125.9	0.37
Shr+yr+cancov+can_h+for_type	126.6	0.35
Shr+yr+cancov+can_h+for_type+sitedisturb	127.6	0.16
B) Conspecific aggregation		
Sites identified with more than one individual (<60 m radii)		
Shr+yr+cancov+can_h+t_logg	73.2	0.66
Shr+yr+cancov+can_h+for_type+t_logg	74.7	0.31
C) CAWA settlement		
Selection of sites after conspecific playback treatment		
Treat+shr+sitedisturb	29.5	0.33
Treat+shr+cancov+sitedisturb	29.6	0.32
Treat+shr+cancov+t_logg	30.3	0.22

4.5 DISCUSSION

Both social and vegetation cues influence CAWA habitat choices. CAWA responds to pre-settlement conspecific cues in areas with lower disturbance, suggesting this cue can be highly influential on settlement decisions in unlogged areas. The most influential vegetation cues are shrub cover, followed by canopy cover and canopy height. Local disturbance (site) seems to influence CAWA patterns, but landscape disturbance did not appear as an important predictor. The effect of the landscape could be diluted due to the size of the scale used (10 x 10 km square), but I could observe abundance was higher in the BBS squares with mid and high levels of disturbance, probably indicating more continuous and undisturbed fragments are less available in those landscapes. However, there is also possibility of more detectability in disturbed areas.

CAWA occurrence patterns are consistent with those observed in the predictive species distribution model developed in Ecoregion 4W (Cupiche et al., 2023, Chapter 3), where the high predicted probability of occurrence was in undisturbed areas. Also, it is consistent to what was found in a study in Northern Minnesota, where CAWA was more common in an unlogged forest compared to a managed forest (Zlonis and Niemi, 2014). However, the abundance and conspecific aggregation in the present study was higher in areas where the forest has been managed for timber extraction (Kabaigon and Shebandowan). Previous studies in the western boreal forest have reported higher density of CAWA in postharvest cuts (regenerated areas >5 years since disturbance; Ball et al., 2016; Hunt et al., 2017) and light partial harvest (Becker et al., 2012). Another study in the Canadian Maritimes found no difference in predicted density of CAWA between areas managed for forest harvesting and protected areas (Westwood et al., 2019a). Abundance or density is often used to infer habitat quality, assuming that individuals occur at greater densities in favorable areas to achieve higher survival and reproduction (Fretwell and Lucas, 1969; Harrison et al., 2005; Doligez and Bouliner, 2008). However, density alone can be misleading, due to conspecific attraction that can lead the aggregation of individuals in unsuitable habitats.

The higher abundance and conspecific aggregation of CAWA in logged areas could be due to the combination of high shrub cover availability (Reitsma et al., 2010) and the attraction to conspecifics as a shortcut given the limited time for individuals to assess habitat and search for mates (Hunt et al., 2017). It is also possible that areas disturbed by logging allow for easier detectability of conspecifics due to their calls carrying over longer distances and our ability to count them properly. Given evidence that CAWA experience lower spatial habitat connectivity in areas of higher disturbance (Hannon and Schmiegelow, 2002; Westwood et al., 2019a), it is possible that the social selection cues are causing CAWA to select less suitable habitats for breeding (following the 'ecological trap hypotheses'; Robertson and Hutto, 2006, 2007). First-time breeders (often arriving later) are more likely to respond to social cues; experiments on conspecific attraction have shown this to be the case (Ward and Schlossberg, 2004; Nocera et al., 2006; Betts et al., 2008). The use of conspecific cues to locate habitat may be especially valuable to reduce the costs of searching and settlement, especially when the time spent to search is reduced (Greene and Stamps, 2001; Fletcher, 2006; Alhering et al., 2010), further research is needed to determine the impact of logging and conspecific attraction on CAWA populations, including comparing breeding success in logged areas to unlogged habitats, primarily to determine whether areas with high conspecific aggregation are productive for CAWA or pose a risk to its survival rates.

I found that CAWA individuals are more prone to select novel sites in less disturbed sites with the influence of conspecific song cues during the pre-breeding season. Therefore, artificial conspecific cues could be a useful tool in conservation and management of CAWA, promoting their settlement and increasing their population density in undisturbed areas or areas of low disturbance (Schlossberg and Ward, 2004; Schepers and Proppe, 2016), as well as potentially expanding its geographical range or “reintroduction” in areas where extirpation is detected (Anich and Ward, 2017). Because knowledge gaps about the long-term efficacy of conspecific broadcasts as an attraction strategy remain, caution should be used when introducing conspecific broadcasts into unoccupied habitat. Care should be taken to ensure that CAWA are not drawn into unproductive sink habitats (Pulliam, 1988). Until knowledge gaps are filled, this technique should be used only where habitat is known with sufficiently high quality to allow managers to be confident that breeding will be successful.

Given the study findings of a strong influence of conspecific attraction on breeding site selection, social cues should be used in species distribution models and other models to predict and manage CAWA habitat. Acknowledging the sociality of many bird species, conspecific attraction is likely to play an important role in habitat choices for many migratory birds, and should be considered and incorporated into modelling, management, and conservation frameworks.

CHAPTER 5. CONCLUSIONS

As logging continues and degradation increases (persistence of younger forest, less old-growth forest availability), the boreal forest, more specifically along its southern border, may become less suitable for songbirds associated to old-growth forest. As I identified in the metanalysis, several bird species are sensitive to logging, especially in mixedwood forests (Schieck et al., 1995; Hobson and Bayne, 2000). The metanalysis did not support the hypothesis of adaptation by the majority of the species studied to boreal forest disturbance. Even though some species increased their relative abundance at landscape scales, reflecting they can occupy alternative areas, it is not enough adaptation to prevent declines, reflecting lower suitable habitat availability in logged landscapes. Furthermore, logging in conjunction with other natural and human disturbances both in breeding and wintering grounds could be driving the decline of numerous migratory songbird species (COSEWIC, 2020; NABCI Canada, 2020). BBS data indicate that declines in CAWA have reversed in some parts of Canada, including Quebec, northern and southern Ontario, northern and eastern Manitoba, and Saskatchewan, but also that declines continue in Alberta, central Ontario, and the Maritimes (COSEWIC, 2020).

Logging disturbance in conjunction with conspecific interactions can lead to altered habitat choices. Conspecific attraction is an important aggregation driver in many songbird species (Campomizzi et al., 2008). Particularly for CAWA, conspecific attraction may explain aggregation in post-harvested areas during the breeding season. The aggregation of conspecifics has implications for species conservation. Without knowledge explaining densities across available habitats, the study will erroneously inform managers on strategies for conservation. As intraspecific interactions influence habitat use, it is thus important to assess quantitative thresholds on abundances of territorial males, as the variation in their densities across sites of varying quality would differentially influence their habitat use.

The response of CAWA to the conspecific cue experiment and their conspecific aggregation in recently logged (or in post-fire sites) areas suggest that disturbed landscapes may pose an ecological trap for this species at risk. To confirm this hypothesis will require studying reproductive success and comparing it in undisturbed and disturbed areas. In migratory species, older birds often arrive at the breeding grounds and settle in preferred habitats earlier than younger birds (Ward and Schlossberg, 2004; Nocera et al., 2006; Betts et al., 2008). After the arrival of young birds, high-quality sites that were once accessible might no longer be available, forcing juveniles to settle in habitats of lower quality (Bayne, 2000). Inexperienced young individuals may not have the ability to recognize logged habitats as less suitable than unlogged ones. I acknowledge the lack of identification of CAWA ages and arrival times in this study. Also, I acknowledge the limitation of not measuring the distance from the harvested areas surveyed to the nearest old-growth forest to account for the probability that CAWA individuals are living in the edge of remnant old forest patches; future studies on CAWA must make the effort to incorporate these measures and to test whether younger individuals are more reliant on conspecific cues and susceptible to inhabit edges of old-growth forest. Such studies should attempt to measure proxies for fitness in old-growth forest and forest edge.

According to the results from the metanalysis and CAWA patterns, it can be inferred that logging is limiting suitable habitat availability; it may be due to the overuse of large-scale clearcuts with a rotation cycle shorter than old-growth stage (<100 years; Bergeron and Harper, 2009; Kuuluvainen and Gauthier, 2018). In North America, timber extraction has resulted in a considerable reduction in the extent of primary old-growth forest (Bergeron and Harper, 2009;

Drapeau et al., 2016). It reveals the importance of maintaining the supply of old-growth forest in the landscape, by either preserving larger tracts of unlogged fragments or increasing the length of harvest rotation cycles. Also, partial harvesting at light to moderate degrees (15-50% partial cut) may be an effective strategy to manage habitat for late successional bird species (Vanderwel et al., 2007). In Ontario, 86% of forest falls under a clearcut silviculture system, 8% under shelterwood retention, and 6% under a tree selection system (OMNRF, 2020); it is important to reduce the percentage of clearcut and give preference to either shelterwood or tree selection systems, which may be less detrimental for birds associated with late successional stages. Through the species distribution model, it was detected that CAWA seems to have high association with riparian zones, and increasing the area buffering riparian zones from logging should be beneficial in the conservation and management of this species at risk.

It has been suggested that forest management should base its practices on historical natural disturbances regimes to complement protection of primary forest for sustaining avifauna (Drapeau et al., 2016). However, it can be controversial given predictable changes in global climate. There are studies on fire regimes projections under climate change scenarios in Canada, which suggest fire activity will increase (Gauthier et al., 2014, 2015). Then, as increased burn rates move forest ecosystems ranges, extensive clear-cutting will become more detrimental, as cumulative impacts of logging and wildfires will exceed historical bounds (Drapeau et al., 2016).

Together, the results in this dissertation support the hypothesis that habitat choices made by CAWA are influenced not only by vegetation cues, but also by social cues. Additionally, the results show that conspecific aggregation is higher in logged than unlogged areas. Meanwhile, the higher occurrence of CAWA in unlogged areas and its association with canopy height suggests old-growth structures are important in the conservation of CAWA. Long-term monitoring programs designed to record trends in avian abundance and habitat quality should examine multiple habitats over the same time period, as ecological traps are likely to occur at different forest regenerated stages. In spite of several limitations in this study, the design had the advantage of documenting the accuracy of the species distribution model of CAWA through the ground truth of model predictions. In addition, this work documents the settlement choices by individuals attracted by artificial conspecific cues during the pre-breeding season, something novel for this species at risk.

Incorporating an understanding of how species aggregate in specific habitats and how it influences their distribution is crucial for improving the accuracy of predictive occupancy models that provide detailed spatial information. It helps in correctly identifying whether areas with low or no species present result from unsuitable habitat conditions or simply because the species tends to cluster in specific areas (Lichstein et al., 2002; Campomizzi et al., 2008). Furthermore, alterations in habitats due to human activities (e.g., land cover conversion, including urbanization) or natural disturbances (e.g., wildfire) can create conditions that are unfavorable for birds to thrive. If birds are unable to recognize harmful changes in their habitat and adjust their choices accordingly, their tendency to be attracted to areas where they see or hear other members of their species can lead them only to occupy habitats that are not favorable to their survival. Therefore, understanding how human-induced changes affect the quality of habitats and how these changes can create a mismatch between the cues used by birds for habitat selection and the characteristics that influence their fitness is of utmost importance.

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APPENDIX A. Supplemental material for Chapter 2.

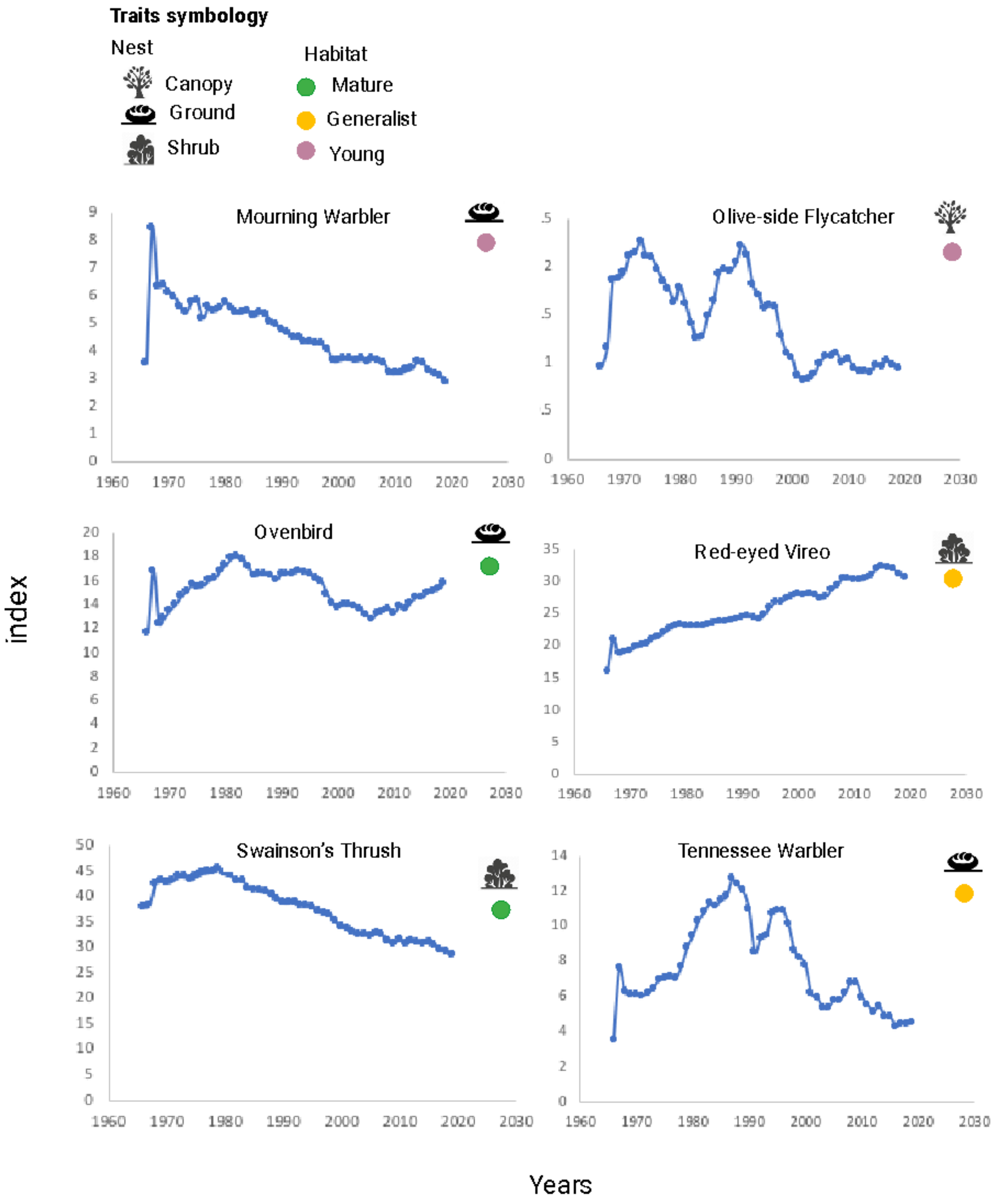

Figure A1. Breeding Bird Survey trends for Canada from 1967-2019. Long-distance migrants.

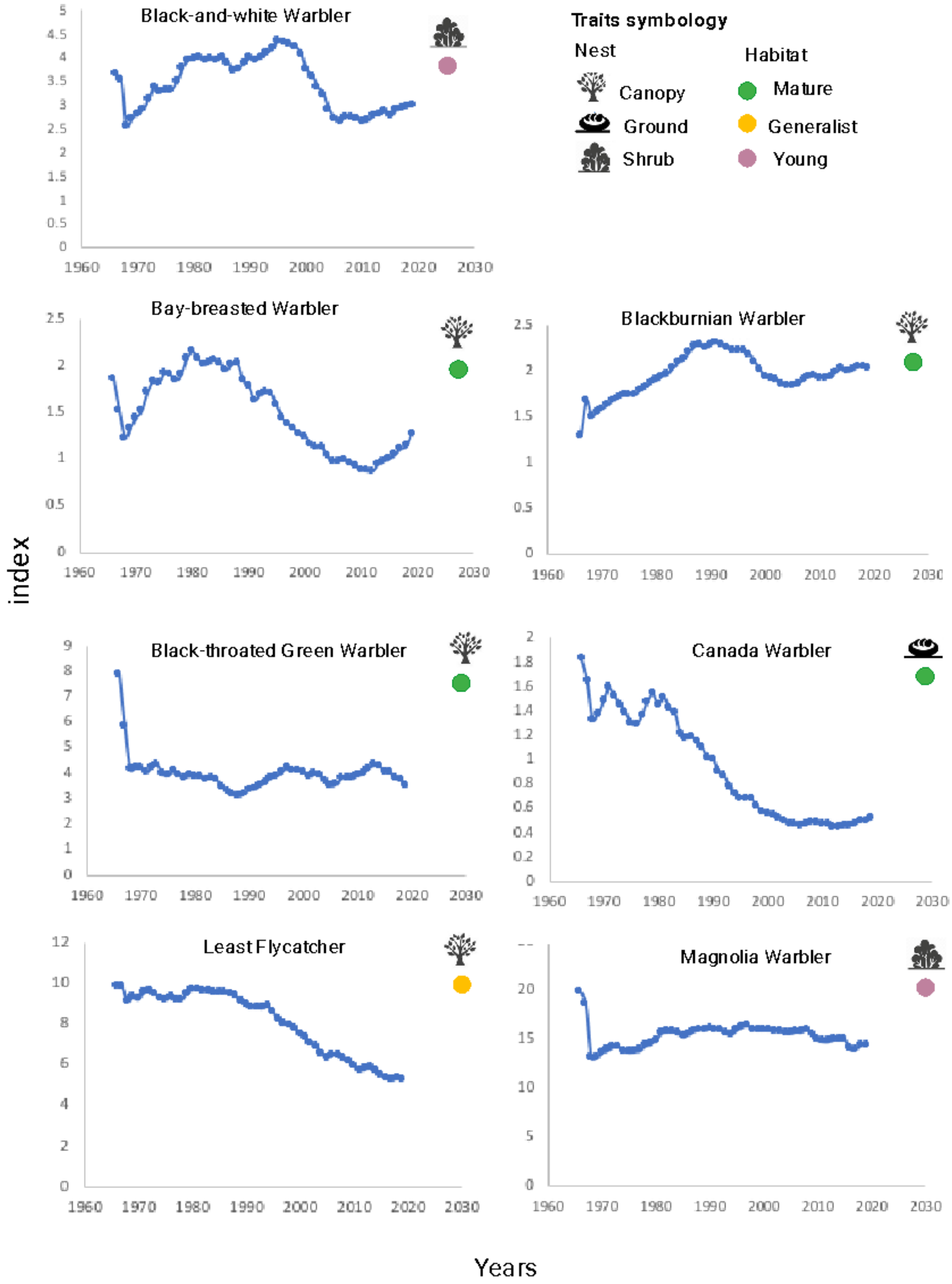

Figure A2. Breeding Bird Survey trends for Canada from 1967-2019. Long-distance migrants.

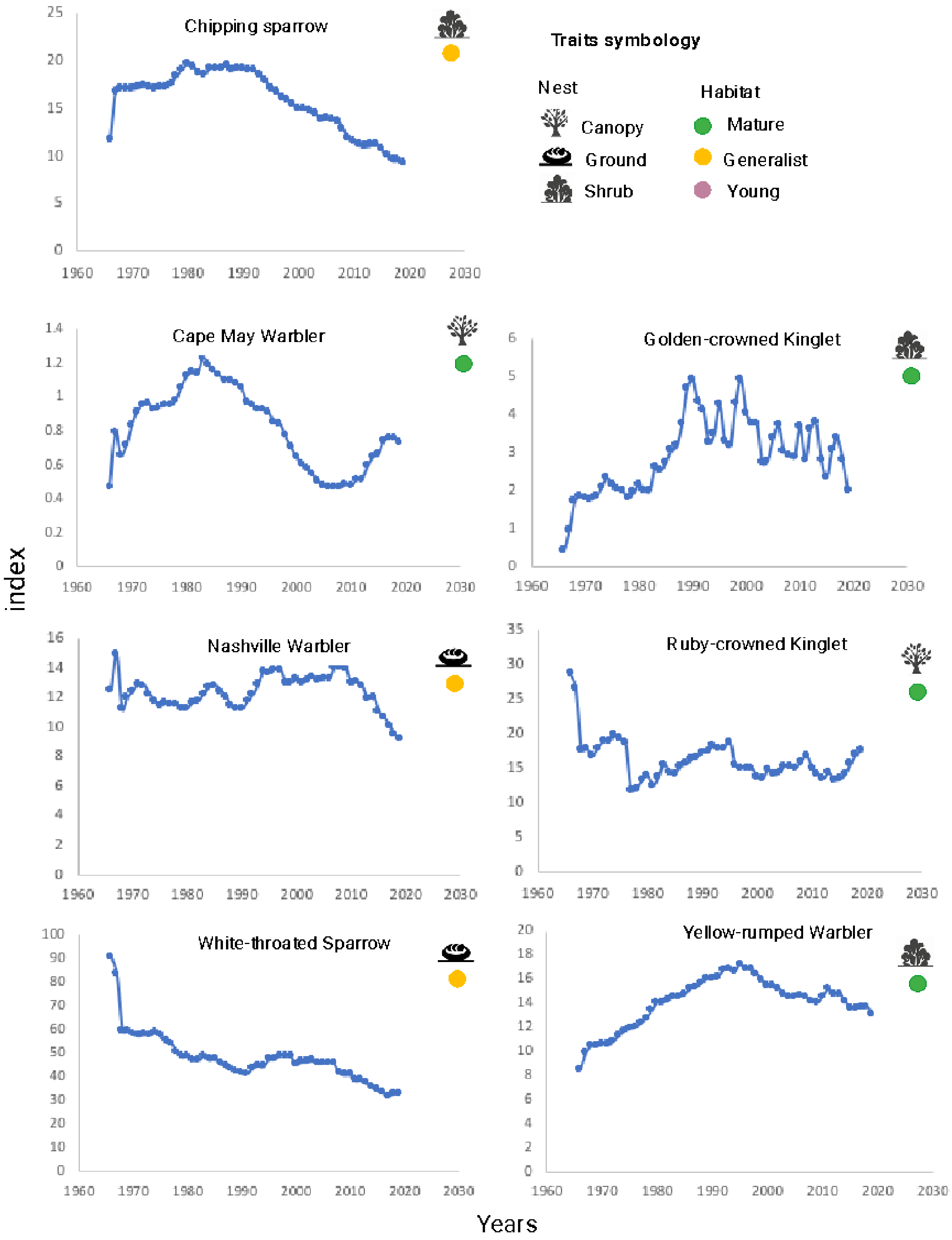

Figure A3. Breeding Bird Survey trends for Canada from 1967-2019. Short-distance migrants.

Table A1. Models by species describing the relationship of relative abundance index with scale, time since logging, and forest type. Models are shown with their AIC values and Akaike weight (ωi). We deemed and marked in bold the best set of models as those having ΔAIC ≤ 2 (Burnham and Anderson, 2002). ΔAIC= $AIC_i - AIC_{min}$ values.

Species name	**Model**	**AIC**	**ΔAIC**	**ωi**
Long distance migrants				
American Redstart	**Time_logg**	**125.27**	**0.00**	**0.49**
	Time_logg+For_type	**126.87**	**1.60**	**0.22**
	Time_logg+Scale	127.60	2.33	0.15
	Time_logg+Scale+For_type	127.86	2.59	0.13
	Scale	137.30	12.03	0.00
	For_type	137.96	12.69	0.00
	Scale+For_type	139.87	14.60	0.00
Black-throated Green Warbler	**Time_logg**	**146.23**	**0.00**	**0.59**
	Time_logg+For_type	148.98	2.75	0.12
	Scale	149.28	3.05	0.15
	For_type	150.89	4.66	0.06
	Time_logg+Scale+For_type	152.15	5.92	0.03
	Time_logg+Scale	152.68	6.45	0.02
	Scale+For_type	153.57	7.34	0.01
Canada Warbler	**Time_logg**	**77.62**	**0.00**	**0.68**
	Time_logg+Scale	80.02	2.40	0.20
	Time_logg+For_type	82.81	5.19	0.05
	For_type	84.57	6.95	0.01
	Time_logg+Scale+For_type	85.06	7.44	0.02
	Scale	85.34	7.72	0.01
	Scale+For_type	86.66	9.04	0.00
Least Flycatcher	**Time_logg+For_type**	**51.73**	**0.00**	**0.46**

Species name	**Model**	**AIC**	**ΔAIC**	**ωi**
	Time_logg+Scale+For_type	**52.33**	**0.60**	**0.34**
	For_type	53.91	2.81	0.15
	Scale+For_type	56.72	4.99	0.04
	Time_logg	62.47	10.47	0.00
	Scale	64.36	12.63	0.00
	Time_logg+Scale	65.15	13.42	0.00
Magnolia Warbler	**Time_logg**	**88.32**	**0.00**	**0.35**
	Time_logg+For_type	**88.53**	**0.21**	**0.32**
	Time_logg+Scale	**89.78**	**1.46**	**0.17**
	Time_logg+Scale+For_type	**90.13**	**1.81**	**0.14**
	For_type	101.08	12.76	0.00
	Scale	101.50	13.18	0.00
	Scale+For_type	103.17	14.85	0.00
Mourning Warbler	**For_type+Scale**	**107.68**	**0.00**	**0.85**
	Time_logg+For_type	111.97	4.29	0.10
	Time_logg+Scale+For_type	113.71	6.03	0.04
	Scale	139.40	31.72	0.00
	Time_logg	145.83	38.15	0.00
	Time_logg+Scale	148.16	40.48	0.00
	For_type	150.64	42.96	0.00
Ovenbird	**Scale**	**177.04**	**0.00**	**0.60**
	Time_logg	**178.79**	**1.75**	**0.25**
	For_type	181.50	4.46	0.06
	Time_logg+Scale	181.77	4.73	0.05
	Scale+For_type	184.05	7.01	0.02

Species name	**Model**	**AIC**	**ΔAIC**	**ωi**
	Time_logg+For_type	185.88	8.84	0.00
	Time_logg+Scale+For_type	188.80	11.76	0.00
Red-eyed Vireo	**Scale**	**107.68**	**0.00**	**0.63**
	For_type	110.14	2.46	0.18
	Time_logg	111.90	4.22	0.07
	Scale+For_type	112.06	4.38	0.07
	Time_logg+Scale	113.80	6.12	0.03
	Time_logg+For_type	116.80	9.12	0.00
	Time_logg+Scale+For_type	118.75	11.07	0.00
Swainson's Thrush	**Scale**	**141.58**	**0.00**	**0.58**
	For_type	**142.90**	**1.32**	**0.30**
	Scale+For_type	146.14	4.56	0.06
	Time_logg	146.90	5.32	0.04
	Time_logg+Scale	150.03	8.45	0.00
	Time_logg+For_type	151.33	9.75	0.00
	Time_logg+Scale+For_type	154.47	12.89	0.00
Tennessee Warbler	**Scale**	**127.55**	**0.00**	**0.79**
	Time_logg	132.62	5.07	0.06
	Scale+For_type	133.11	5.56	0.05
	For_type	133.15	5.60	0.05
	Time_logg+Scale	133.49	5.94	0.04
	Time_logg+For_type	138.00	10.45	0.00
	Time_logg+Scale+For_type	139.09	11.54	0.00
Short to medium distance migrants				
Chipping Sparrow	**For_type**	**146.49**	**0.00**	**0.75**

Species name	**Model**	**AIC**	**ΔAIC**	**ωi**
	Scale+For_type	149.05	2.56	0.20
	Time_logg+For_type	152.38	5.89	0.04
	Time_logg+Scale+For_type	154.96	8.47	0.01
	Scale	161.51	15.02	0.00
	Time_logg	164.63	18.16	0.00
	Time_logg+Scale	167.07	20.58	0.00
Golden-crowned Kinglet	**Time_logg+For_type**	**79.04**	**0.00**	**0.46**
	For_type	**80.65**	**1.62**	**0.20**
	Scale+For_type	**80.96**	**1.92**	**0.18**
	Time_logg+Scale+For_type	81.43	2.39	0.14
	Scale	120.00	40.96	0.00
	Time_logg	121.95	42.91	0.00
	Time_logg+Scale	124.14	45.10	0.00
Ruby-crowned Kinglet	**For_type**	**72.52**	**0.00**	**0.44**
	Scale	**73.56**	**1.04**	**0.26**
	Scale+For_type	74.61	2.09	0.15
	Time_logg	76.16	3.64	0.07
	Time_logg+For_type	77.35	4.83	0.04
	Time_logg+Scale	78.86	6.10	0.02
	Time_logg+Scale+For_type	79.23	6.71	0.01
White-throated Sparrow	**Scale**	**168.99**	**0.00**	**0.58**
	For_type	**170.34**	**1.35**	**0.30**
	Scale+For_type	173.46	4.47	0.06
	Time_logg	175.68	6.69	0.02
	Time_logg+Scale	175.76	6.77	0.02

Species name	**Model**	**AIC**	**ΔAIC**	**ωi**
	Time_logg+For_type	177.61	8.62	0.00
	Time_logg+Scale+For_type	180.32	11.23	0.00
Yellow-rumped Warbler	**For_type**	**126.54**	**0.00**	**0.51**
	For_type+Scale	**128.10**	**1.56**	**0.23**
	Scale	128.8	2.26	0.17
	Time_logg+For_type	131.76	5.22	0.04
	Time_logg+Scale+For_type	132.81	6.27	0.02
	Time_logg	133.23	6.69	0.02
	Time_logg+Scale	135.08	8.54	0.00

Scale = scales of the studies (local and landscape), Time_logg= years since logging (0-5, 6-10, 14-28, 29-80 years), For_type=forest type (conifer, deciduous and mixedwood)

APPENDIX B. Supplemental material for Chapter 3.

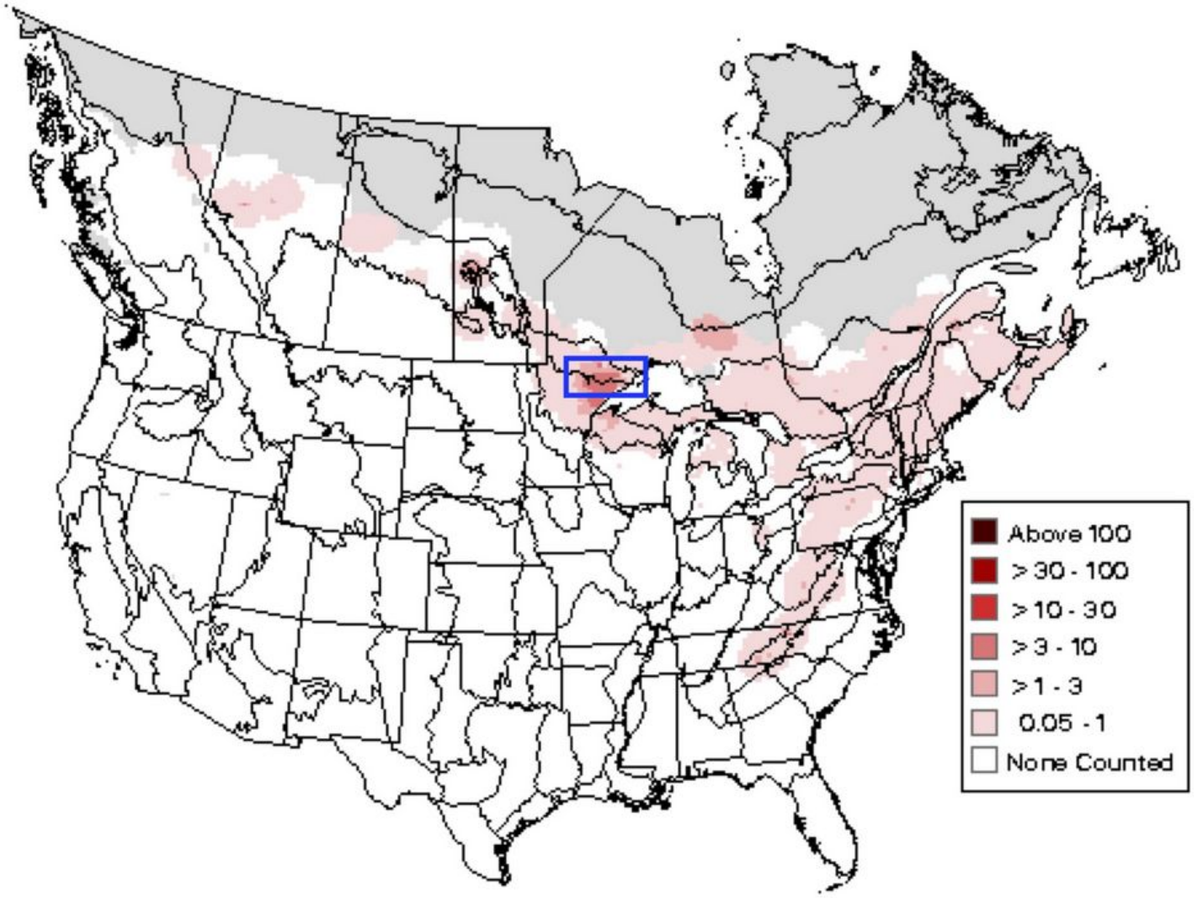

Figure B1. Breeding distribution and relative abundance of the Canada Warbler in North America based on the federal Breeding Bird Survey, 2011-2015 (Sauer et al., 2017). Ecoregion 4W is marked within Northwestern Ontario by the blue square

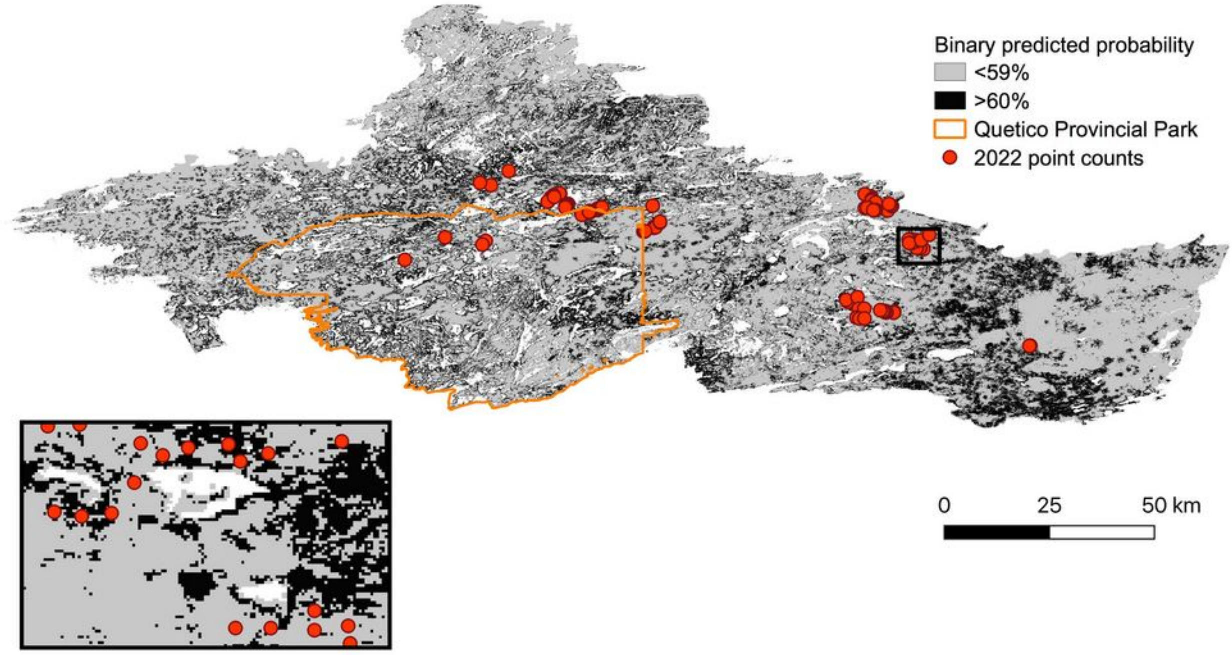

Figure B2. Binary distribution map of predicted probability of occurrence of Canada Warbler (CAWA) in Ecoregion 4W based on the model 2021. The model used a dataset of observations from 2001-2021 cross-validated with 25% random seed data. Red dots are point counts selected for survey in 2022 to field-validating the model. Black square shows a close view of an area where the point counts were collected.

Table B1. Correlations coefficients (rho) for the 11 possible predictors.

	BASI	**CAN**	**DEM**	**D_CONIF**	**EVI**	**DISTURB**	**LST**	**NDMI**	**NDVI**	**NDWI**	**WATER**
BASI	1										
CAN	-0.51	1									
DEM	-0.17	0.18	1								
DISTURB	-0.18	0.30	-0.06	1							
D_CONIF	-0.01	-0.02	-0.34	0.04	1						
EVI	-0.35	0.23	0.00	-0.05	0.20	1					
LST	-0.16	-0.04	0.26	-0.09	-0.32	0.04	1				
NDMI*	**-0.85**	0.51	0.08	0.15	0.11	**0.70**	0.03	1			
NDVI*	**-0.79**	0.38	0.28	0.04	-0.05	0.62	0.43	**0.76**	1		
NDWI	-0.27	0.22	-0.22	-0.25	-0.01	-0.18	-0.25	0.27	-0.15	1	
WATER	0.13	-0.09	-0.09	-0.04	-0.01	-0.07	0.01	-0.14	-0.09	-0.06	1

Table B2. Year distribution of 122 observations of CAWA for used as training dataset (2001-2021). Sources: eBird search from 2000-2020, OBBA (Ontario Breeding Bird Atlas 2001-2005), BBS (Breeding Bird Survey 2000-2020), Quetico Provincial Park (2024-2019), own field collected data in 2021.

Year	# observations	Source
2001	2	eBird
2003	36	eBird (4), OBBA (32)
2012	1	eBird
2013	4	eBird
2014	5	Quetico
2015	1	BBS
2016	2	eBird
2017	10	eBird (6), Quetico (4)
2018	8	eBird (5), Quetico (3)
2019	4	eBird (2), BBS (1) Quetico (1)
2020	5	eBird
2021	44	OBBA volunteers (1), own field data (43)

Table B3. Years of Canada Warbler observations (2001-2022), and years of disturbance registered for that site during the period of 2000-2021.

Occurrence year	# occurrences	Year of disturbance (DISTURB covariate)
2001	2	no disturbance reported in a period of 2000-2020
2003	35	
2012	1	
2013	4	
2014	5	
2015	1	
2016	2	
2017	10	
2018	7	
2019	3	
2020	5	
2021	38	
2022	24	
2003	1	Disturbed in 2007
2018	1	Disturbed in 2015
2019	1	Disturbed in 2006
2021	3	Disturbed in 2006
2021	2	Disturbed in 2015
2021	1	Disturbed in 2018
2022	4	Disturbed in 2015
2022	2	Disturbed in 2017

APPENDIX C. Supplemental material for Chapter 4

Table C1. Correlation matrix of numerical variables from speaker experiment.

Variables	**Shrub**	**Canopy cover**	**Canopy Height**
Shrub cover	1		
Canopy cover	-0.60	1	
Canopy tree height	-0.09	0.25	1

Table C2. Correlation matrix of numerical variables from occurrence observations

Variables	**Shrub**	**Canopy cover**	**Canopy Height**
Shrub cover	1		
Canopy cover	-0.43	1	
Canopy tree height	-0.08	0.45	1

Table C3. Set of models of Canada Warbler occurrence and conspecific attraction during the bird surveys and playback experiment in the breeding season. Models are shown with their AIC values and Akaike weight (ωi). Marked in bold are the best set of models as those having ΔAIC ≤ 2 (Burnham and Anderson 2002). ΔAIC= $AIC_i - AIC_{min}$ values. Shr=shrub cover, yr= year of study, cancov=canopy cover, can_h= canopy height, for_type= forest type (coniferous, deciduous, mixewood), landsc= landscape disturbance level (low, mid, high), sitedisturb= disturbance condition in surveyed site, (disturbed/undisturbed), t_logg=time since logging (early, mid, late, unlogged), treat= treatment (speakers/control).

Model	**AIC**	**ΔAIC**	**ωi**
A) Occurrence			
Shr+yr+cancov+for_type	**125.97**	**0.00**	**0.37**
Shr+yr+cancov+can_h+for_type	**126.08**	**0.11**	**0.35**
Shr+yr+cancov+can_h+for_type+sitedisturb	127.65	1.68	0.16
Shr+yr+cancov+can_h+for_type+landsc	128.97	3.00	0.08
Shr+yr+cancov+can_h+for_type+t_logg	131.42	5.45	0.02
Shr+yr	141.02	15.05	0.00
Shr	146.87	20.90	0.00
Null	193.47	67.5	0.00
B) Conspecific aggregation			
Shr+yr+cancov+can_h+t_logg	**73.28**	**0.00**	**0.54**
Shr+yr+cancov+can_h+forest_type+t_logg	**74.77**	**1.49**	**0.26**
Shr+yr+cancov+t_logg	75.59	2.31	0.17
Shr+yr+cancov+can_h+forest_type+sitedisturb	80.06	7.32	0.01
Shr+yr+cancov+can_h+forest_type	81.41	8.13	0.01
Shr+yr+cancov+can_h+forest_type+landsc	83.44	10.16	0.00
Shr+yr	87.43	14.15	0.00
Shr	92.96	19.68	0.00
Null	121.51	48.23	0.00

Model	**AIC**	**ΔAIC**	**ωi**
C) CAWA settlement			
Treat+shr+sitedisturb	**29.56**	**0.0**	**0.33**
Treat+shr+cancov+sitedisturb	**29.61**	**0.05**	**0.32**
Treat+shr+cancov+t_loss	**30.35**	**0.79**	**0.22**
Treat+shr+cancov	33.57	4.01	0.04
Treat+shr+cancov+can_h	33.62	4.06	0.04
Treat+shr+cancov+can_h+for_type	33.89	4.33	0.04
Treat+shr+cancov+landsc	36.96	7.40	0.00
Shr+treat	39.96	7.4	0.00
Shr+treat+yr	41.34	11.78	0.00
Shr+can_h+for_type	45.95	16.39	0.00
Shr+can_h	46.13	16.57	0.00
Shr+cancov+can_h	46.14	16.58	0.00
Shr+cancov	46.42	16.86	0.00
Shr+can_h+for_type+square	49.03	19.47	0.00
Shr+can_h+for_type+t_logg	49.07	19.51	0.00
Shr	49.32	19.76	0.00
Treat	70.65	41.09	0.00
Null	77.25	47.69	0.00