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Influences of Adjacent Land-Use on Avian Community Structure in Sub-Boreal Peatlands of the Midwest

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Influences of adjacent land-use on avian community structure in sub-boreal peatlands of the Midwest.

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By

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ABSTRACT

Surrounding land use influences avian community structure; moreover, anthropogenic manipulation of habitat can alter bird species richness and composition. In the first chapter of my thesis work, I conducted avian surveys at 20 sub-boreal peatlands in eastern Minnesota and western Wisconsin to detect correlations between land development and bird community structure. Peatlands are wetlands that accumulate decaying organic plant material (peat) and provide valuable and diverse habitats to a variety of flora and fauna. I measured urban and cropland development at three spatial scales (500 m, 1000 m, and 2500 m radii). Effects of development on avian communities also were assessed with respect to distance from a major metropolis using three zones: counties inside the city (metro), counties immediately adjacent (collar), and rural counties adjacent to the collar (fringe). I predicted that the zone with intermediate levels of disturbance would have greater species richness (intermediate disturbance hypothesis) and that there would be a correlation between differences in community species composition (i.e., species turnover) and increased levels of development. Urban development influenced avian communities more than agricultural development and avian communities were different among the zones in species composition and partly in species richness. Statistical analysis identified negative relationships between urban cover and the proportion of human intolerant species at all three spatial scales. In the second chapter of my thesis work, I created a rapid-assessment model of habitat quality designed to detect relationships of landscape variables such as peatland area, degree of isolation from other wetlands, extent of adjacent economic development, and vegetative structure on bird community composition. I used rapid-assessment model, known as a hydrogeomorphic model (HGM), that numerically combines landscape features to predict the value of habitat to bird species diversity (functional capacity) Following HGM procedures, I predicted functional capacities at 20 sub-boreal peatlands When compared to observed species diversity, however, no relationship was found with predicted diversity levels. Of the model variables only vegetation structure was related to bird diversity. I suggest that a new model needs to be developed in order to assess the value of peatlands to support peatland bird communities.

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Urban and cropland area were measured at three spatial scales (500, 1000, and 2500 m) for each peatland. Figure adapted from Whited et al. (2000).

Mean urban area (gray bars; +/- 1 SE) at the 500-m scale was less in the fringe zone compared to the collar (*P* = 0.05) and metro (*P* = 0.02). Mean urban cover did not differ between the metro and collar zones. Mean cropland area (white bars, +/- 1 SE) did not differ between the zones at the 500-m spatial scale.

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The functional capacity index (FCI) for the support of birds in sub-boreal peatlands is modeled as being dependent on peatland area (V_{Area}), degree of isolation (V_{Isol}), the proportion of economic development in the buffer ($V_{LandDev}$), and the dominant vegetation of the peatland (V_{NatVeg}). Peatlands that are large ($\geq$ 100 ha), are in close proximity to other wetlands, have a nearly pristine buffer and have a natural peatland plant community received a 1. The functional capacity index is scaled from 0 to 1; a peatland that receives an *FCI* score of 1 was presumed to support a bird community typical of an undisturbed peatland in an undisturbed landscape.

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Chapter 1

Influences of adjacent land-use on avian community structure in sub-boreal peatlands of the Midwest.

ABSTRACT

Surrounding land use often influences avian community structure; moreover, anthropogenic manipulation of habitat can alter species richness and composition. I surveyed 20 sub-boreal peatlands in eastern Minnesota and western Wisconsin to examine the effects that land use has on avian community assemblages. Developed land was categorized as cropland and urban cover and total development. More specifically, I asked how economic development influences avian community structure. Land use variables were calculated at three spatial scales 500-m radius (79-ha circle), 1000 m (314 ha) and 2500 m (1963 ha). Land use effects on avian communities were assessed at both site and regional levels; regions were categorized into three zones based on varying degrees of development and included the highly urbanized seven-county Twin Cities metropolitan region (metro), the 12 immediately adjacent suburban counties surrounding the metropolitan region (collar), and a rural region (fringe). I predicted that increased land development would not result in decreased species richness due to species-turnover which I defined as an avian community composition comprised of human-tolerant species with limited human-intolerant species. Urban development influenced avian communities more than agricultural and avian communities were different between the zones in species composition and partly in species richness. Statistical analysis identified negative relationships between urban cover and the proportion of human intolerant species at all three spatial scales.

INTRODUCTION

Habitat loss presents a significant threat to biodiversity worldwide (Brooks et al. 2002) and is often the result of anthropogenic land development for agricultural use and urban expansion. With the human population of the United States projected to reach 438 million by 2050 (Passel and Cohn 2008) it is expected that there will be increased pressure to develop land for agricultural and urban uses. In the Midwestern United States, the loss of wetland, grassland, and forest habitats to agriculture and urban expansion and the fragmentation of remaining natural land are already substantial (Sampson and Knopf 1994, Gonzalez-Abraham et al. 2007, Dahl 2011). As natural habitats are lost and fragmented, biotic communities are significantly affected (Whited et al. 2000, Houlahan and Findlay 2003, Akasaka et al. 2010).

Because of the close relationship between organisms and habitat, biologists use composition of communities to evaluate and monitor habitat quality. The idea of a biotic index is such that community assemblages in high quality habitat will have proportionally more sensitive species or certain indicator species (i.e., those that are present only in high quality habitats) than communities in low quality habitats. Of the taxonomic groups used as indicators of habitat quality, Galatowitsch et al. (1999) suggest that bird communities reflect land use more accurately than other organismal groups (i.e., plants, fish, amphibians and invertebrates) and other studies have successfully used birds as indicators of habitat quality (O'Connell et al. 2000, Browder et al. 2002, DeLuca et al. 2004); therefore we selected birds for our model taxon.

The declining trend of many bird populations in developed landscapes has raised concerns among conservationists and for management purposes it is necessary to identify species-habitat relationships at the local and landscape-scale to maintain viable populations. Local habitat cues (e.g., foliage height diversity, vegetation density and proximity to edge) have been identified as important predictors of bird species presence and diversity (MacArthur and MacArthur 1961, Rotenberry 1985, Mills et al. 1989, Cunningham and Johnson 2006). Changes in vegetation structure at the local scale, (e.g., the reduction of dead trees and understory plants) and diversity (e.g., the introduction of non-native vegetation in place of native vegetation) can have profound effects on bird communities (Blair 1996, Schlesinger et al. 2008). However, even if local habitats are left unaltered, land alteration and development for agricultural and urban development at farther distances and greater spatial scales can influence bird communities (Whited et al. 2000). As such, the focus of our study was to identify the affects, if any that land development at the landscape scale (≥ 500 m radius) has on bird communities.

Land development for recreational use, agriculture and urban expansion can negatively affect avian density, abundance, species richness and increase the likelihood of nest predation (Best et al. 1995, Findlay and Houlahan 1997, Miller et al. 1998, Fernández-Juricic 2000, Miller et al. 2003, Thorington and Bowman 2003, Mallord et al. 2007). Additionally, land development can result in species turnover within avian communities and lead to declines or local extinctions of species that are sensitive to human activity and increases in widely-distributed opportunistic species (Beissinger and

Osborne 1982, Väisänen and Rauhala 1983, Dowd 1992, Blair 1996, Blair 2004), thereby homogenizing the communities (Devictor et al. 2007). Typically, habitat disturbance negatively affects specialist species and positively affects generalist and opportunistic species (Väisänen and Rauhala 1983, Dowd 1992, Blair 1996, Miller et al. 1998, May et al. 2002, Husté and Boulinier 2007, Banning et al. 2009) making it difficult to use species richness as an indicator of habitat quality as bird species richness may be similar between sites in urban landscapes and those in rural landscapes (Smith and Chow-Fraser 2010). Furthermore, moderate levels of development can result in increased species diversity; however, native species are typically replaced by ubiquitous species (Blair 1996). Additionally, urban environments tend to support fewer long-distance migratory species and more non-migratory species than rural environments (Husté and Boulinier 2007, Minor and Urban 2010).

Although both agricultural and urban development alters landscapes from their natural state, research suggests that the negative impact on avian communities is more drastic in urban environments than in agricultural environments. When natural habitat is converted for agricultural purposes it typically results in vegetative monocultures interspersed by marginal or poor quality habitat (e.g., fencerows and small patches of trees planted as wind breaks). When compared to forested and wetland habitats, row-tilled and small grain crops are used by few bird species and usually only for foraging (Best et al. 1995). Though some non-cultivated habitats associated with agricultural landscapes, such as wooded fence rows, may support high avian abundance they likely function as ecological traps where bird populations suffer low reproductive rates due to increased nest predation, human disturbance and brood parasitism (Best 1986). Other factors associated with agricultural practices, such as the application of pesticides, can alter bird diversity and result in the loss of foraging guilds (Genghini et al. 2006).

Avian communities are influenced by measures of urban cover including road density, building density and percent urban development (Findlay and Houlahan 1997, Whited et al. 2000, Miller et al. 2003, Minor and Urban 2010). Increased levels of human activity, noise pollution, nest predation and brood parasitism associated with urban environments affect bird species presence and reproductive success even in habitats that would be suitable for breeding purposes (Francis et al. 2011). Even relatively minor levels of disturbance (e.g., a recreational trail or a long pier) that increase human activity can influence bird communities (Miller et al. 1998, Banning et al. 2009). Pedestrian activity can lead to decreased bird species richness, abundance and population density (Fernández-Juricic 2000, Mallord et al. 2007); furthermore, species-turnover has been positively correlated with increased pedestrian rates (Fernández-Juricic 2000).

Peatlands are wetlands that accumulate decaying organic material (peat); the majority of peatlands are found in the boreal region of the northern hemisphere (Glaser 1987). Sub-boreal peatlands are found between the boreal and temperate ecoregions. Peatlands occur along a bog-fen continuum and can potentially support diverse habitats, including open bogs, sedge and forested fens (Glaser 1987). However, though peatlands provide valuable habitat for many plants and animals, they face the threat of development as they also possess valuable resources such as peat and timber and can be used in agriculture (e.g., cranberry farming). Because of the potential conflict between the economic and ecological values of peatlands, there has been contention among conservationists and industrialists as to whether peatlands can be exploited sustainably and without causing irreconcilable damage to the flora and fauna using them (Väisänen and Rauhala 1983, Desrochers et al. 1998, Chapman et al. 2003, Lachance and Lavoie 2004). The diversity of habitats found in peatlands provides breeding and foraging habitat for bird species typically found in boreal, temperate-deciduous and prairie ecoregions (Niemi and Hanowski 1992); many of which are suffering population declines due primarily to habitat loss (North American Bird Conservation Initiative 2011). In peatlands, severe levels of disturbance such as peat mining and logging result in bird species turnover in addition to reductions in species richness and abundance (Väisänen and Rauhala 1983, Niemi and Hanowski 1984, Desrochers et al. 1998). Changes in bird species composition and abundance resulting from commercial development of peatlands are largely due to the alterations in plant community structure and composition. Furthermore, development in the adjacent landscape can change aspects of peatlands such as hydrology and pH, influencing plant community structure (Girard et al. 2002, Churchill 2011, Miller 2011), potentially leading to changes in avian community composition. We sought to identify the effects that landscape-scale agricultural and urban development has on bird species richness and community assemblages in sub-boreal peatlands of eastern Minnesota and western Wisconsin.

In eastern Minnesota and western Wisconsin, urban development in the seven-county metropolis including Minneapolis and St. Paul (Metro) increased from 24 to 33% from 1986 and 2002, resulting in subsequent losses of rural land types, namely agriculture, wetland, and forest (Yuan et al. 2005). The increase in urban development led us to believe that bird communities in the Metro are noticeably different from those outside of the Metro where urban sprawl is not as prominent. Additionally, agricultural development is a common land use practice in this region; however, I expected that agricultural development would be greater outside the Metro than within and that agricultural development would negatively affect bird communities at the spatial scales that I assessed. The degree of land development was assessed at individual sites and within zones at three spatial scales in areas of 500, 1000, and 2500 m radius (Fig. 2). Regions were categorized based on varying degrees of development and included the Metro, the 12 immediately adjacent suburban counties surrounding the metropolitan region (collar), and a rural region (fringe; Fig. 1). I assessed how anthropogenic land-use practices affect avian communities in terms of species richness and the proportion of human-intolerant bird species (i.e., species that lack an affinity to human activity). Human-intolerant species are typically area-sensitive, neotropical migrants, or specialists, whereas human-tolerant species tend to be area-insensitive, sedentary, or generalists.

I expected that species richness would be greatest in the zone with intermediate levels of disturbance (here cropland and urban development). As agricultural environments tend to affect avian reproductive output more so than they do community assemblages, I did not anticipate a relationship between agricultural development and bird species richness or with the proportion of human-intolerant species. I predicted that there would be a shift in bird community assemblages from those primarily composed of human-tolerant species in sub-boreal peatlands located in the metropolitan area to bird communities composed primarily of human-intolerant species in fringe zone with intermediate levels found in the collar zone. Furthermore, the proportion of human-intolerant species within bird communities was expected to decrease as urban land cover increased within the adjacent landscape at all spatial scales.

METHODS

Bird surveys

Bird communities were sampled in five peatlands in the Metro, seven in the collar and eight in the fringe. Within each zone (i.e., metro, collar, and fringe), the sites (i.e., a peatland) surveyed included those dominated by herbaceous, shrub, or forested habitats. I conducted 121 point counts at 41 circular point-count stations (points). Count duration was 10 min and points were surveyed between 1 May and 24 June 2011. The number of points per site varied from one to four and was determined by a combination of wetland size and accessibility. I established points at least 200 m apart to avoid double counting individual birds. Points were located within the peatland at least 50 m from the peatland edge, unless I was not able to do so due to the difficulty of the terrain (e.g., deep water or floating bog).

The use of combined call-response and passive surveys for secretive marshbirds is the suggested methodology to increase detection probability of marsh bird species to monitor populations (Conway and Nadeau 2010). I played 30-sec call-response recordings of sora (*Porzana carolina)*, Virginia rail (*Rallus limicola*), yellow rail (*Coturnicops noveboracensis*), American bittern (*Botaurus lentiginosus*), least bittern (*Ixobrychus exilis*) and pied-billed grebe (*Podilymbus podiceps*) during the first session (1 May to 16 May).

I recorded all species and number of individuals seen or heard within a 100-m radius, except for birds flying at heights > 20 m. Digital copies of calls at most points (n = 116; 96 %) were collected using a handheld recorder (H4n, Zoom Corporation, Tokyo, Japan) and compared to written data in the laboratory; recordings are useful for validating vocal identification of species especially when species richness is high (Haselmayer and Quinn 2000). All points were surveyed between 30 min before sunrise and 0900 CDT with wind speeds < 20 km/hr and, at most a light rain. Each point was surveyed three times, except for those in St. Croix State Park, because it was added during the second sampling session. I measured species richness and determined a species' affinity to humans (i.e., tolerant or intolerant) based on personal observation and species-human relationships identified in other studies (Dowd 1992, Bryce et al. 2002; Appendix 1).

Land use characterizations

Land use data was developed using 2008 1-m spatial resolution remote-sensing data from the United States Department of Agriculture (USDA), Farm Services Agency (FSA) and the National Agriculture Imagery Program (NAIP) in ArcMap 10 (ESRI 2011) and classified into five categories including water bodies, forest (both deciduous and coniferous), herbaceous (grassland, emergent wetland and pasture), cropland (cultivated) and urban (roads, residential housing and industrial) within each 500, 1000 and 2500-m radius circle of the study site (Whited et al. 2000). Only developed land was used in analysis to avoid auto-correlation among land-use variables, namely the proportion of undeveloped land to developed land within each radius. Additionally I combined the area of cropland and urban cover within each circle to create another category, total development.

Statistical analysis

Statistical analyses were performed in R statistical software (2012). I used one-way analysis of variance (ANOVA) tests to compare mean land development measures between zones. Developed land use was assessed as urban, cropland and total development (i.e., urban and cropland combined) for each spatial scale. One-way ANOVA tests were also used to compare mean species richness and mean proportion of human-intolerant species between the defined zones. Alpha was set at 0.05 for all comparative analyses.

All sites were included (i.e., not grouped into zones) for regression analysis that compared measures of land-development (cropland and urban) to the composition of human-intolerant species within communities. After linear and curvilinear regression analyses were calculated for urban development, we used step-wise regression analysis in the MASS package for R (Venables and Ripley 2002) to determine goodness-of-fit. Models with lower Akaike Information Criterion (AIC) values were selected as the best-fit.

RESULTS

Bird surveys

A total of 107 bird species were recorded, species richness ranged from 19 to 41 ($\bar{x}$ = 29) per site. Three species (bald eagle (*Haliaeetus leucocephalus*), common grackle (*Quiscalus quiscula*) and common loon (*Gavia immer*)) occurred only as flyovers and were excluded from all analyses. The five most abundant species across all study sites in descending rank order were: red-winged blackbird (*Agelaius phoeniceus*), common yellowthroat (*Geothlypis trichas*), swamp sparrow (*Melospiza georgiana*), yellow warbler (*Dendroica petechia*) and American goldfinch (*Carduelis tristas*); only common yellowthroat were detected at all sites.

Land development

Cultivated land comprised a greater area of land cover in the surrounding landscape than did urban cover at all three spatial scales (Table 1). The metro had the greatest amount of urban land cover, however coverage was not different from the collar at any spatial scale (Figures 3, 4, and 5). The amount of urban area in the metro was greater than the fringe at all spatial scales (all $P < 0.02$). Additionally, the collar zone had more urban cover than the fringe zone at all three spatial scales (all $P \leq 0.03$). Cropland coverage did not differ between any of the areas at the 500 or 1000-m scale (Figures 3 and 4) or the fringe at 2500 m; however, the collar zone had more cropland cover at the 2500-m scale than the metro ($P = 0.02$). The metro and fringe zones did not differ in cropland coverage at any scale.

Bird community composition

Mean proportion of human-intolerant species did not differ between the metropolitan and collar zones, however the proportion of human-intolerant species in the fringe zone was greater than the metro ($P < 0.00$) and the collar ($P << 0.00$) zones (Fig. 6). Mean species richness was not different between the metropolitan zone and the collar or fringe zones (Fig. 7). However, mean species richness in the collar zone was greater than in the fringe zone ($P = 0.03$; Fig. 7).

Negative trends were found with regression analysis between land development measures (area of urban Figures 8, 9, and 10 and cropland Figures 11, 12, and 13) and the proportion of human-intolerant bird species at all three spatial scales; however, significant relationships were found only for urban land use. The relationship between urban development at all scales and the proportion of human-intolerant species appeared to be curvilinear so we also calculated curvilinear regressions at each scale. Step-wise regression analysis indicated that curvilinear regression models for urban development provided a better fit to human-intolerant bird data than linear models; furthermore, based on AIC values the best-fit model was identified as urban development at the 2500-m scale (Fig. 10 ; AIC = -94.71).

DISCUSSION

My results provide evidence that avian community assemblages in sub-boreal peatlands are influenced by landscape characteristics in the surrounding environment. Urban development in the buffer zone negatively affected bird community composition of human-intolerant species at all spatial scales, with the most pronounced relationship at the 2500-m scale. Urban cover did not exceed 20 % of the surrounding landscape for any site at any spatial scale; suggesting that human-intolerant species in sub-boreal peatlands are sensitive to low levels of urban development. DeLuca et al. (2004) found that when urban development in the 500 and 1000-m scale exceeded 14 and 25 %, respectively that marsh bird community integrity was significantly reduced; the results from our study suggest that for avian communities of sub-boreal peatlands urban development threshold levels may be even lower. For sites with few human-intolerant species present, common yellowthroat, swamp sparrow and yellow warbler were those commonly observed; these species are known to select breeding habitat based on local-scale habitat cues such as the presence/absence of water, patterns of vegetation distribution and availability of edge habitat (Stauffer and Best 1986, Greenberg 1988, Knopf and Sedgwick 1992) and not on landscape-scale habitat cues. Compared to urban cover, agricultural cover was much more prominent at all spatial scales. However, levels of agricultural land did not correlate with the proportion of human-intolerant species, which may be the related to human activity levels approaching but not meeting avoidance thresholds for sensitive species.

Concerning species richness, I did not observe a difference in avian species richness between the metro zone and either of the other zones. Similarly, Smith and Chow-Fraser (2010) found that bird species richness was similar between environments in an urban landscape and those in a rural landscape; yet other studies found that urbanization results in decreased bird species richness (Blair 1996, Stratford and Robinson 2005). Mean species richness was greater in the collar zone than in the fringe zone; however, the additional species were mostly ubiquitous, as evident by the bird communities in the fringe zone having a greater mean proportion of human-intolerant species. Though the fringe zone had the lowest mean species richness of the three zones, bird communities in the fringe zone were composed of more human-intolerant species than either one; suggesting that species-turnover results from increased urban cover. Similarly, when comparing avian communities in an urban-forest patch to a natural-forest patch Dowd (1992) found that the natural patch had significantly more forest-interior and human-intolerant species. Another possible explanation for our findings may be that urban development imposes more influence on species richness than agricultural development does and that intermediate levels of urban disturbance facilitate greater species richness following the intermediate disturbance hypothesis (see Grime 1973, Connell 1978, Huston 1979). The intermediate disturbance hypothesis suggests that a moderate level of habitat disturbance enables the coexistence of species that prefer low and high levels of habitat disturbance.

Overall, my data suggest that urban development near peatlands has more influence on avian species composition and richness in sub-boreal peatlands than agricultural development. Furthermore, compared to habitat patches in non-urban landscapes, urban sites are used as temporary stop-over sites to a much-lesser degree by migratory species (Stratford and Robinson 2005) and support fewer neotropical migrants (Husté and Boulinier 2007, Minor and Urban 2010), trends that are further supported by our data. That is not to suggest that peatland habitat in urban landscapes are without value, as they still provide breeding and foraging habitat for many bird species in addition to offering potential stop-over sites for migrating species. Rodewald and Matthews (2005) found that neotropical migrants use forested habitat patches in developed landscapes, though some species (e.g., Swainson's thrush [*Catharus ustulatus*]) avoid using small habitat-patches (Matthews and Rodewald 2010). I feel that efforts to conserve avian diversity in peatlands of the Midwest should be focused on areas that have minimal urban development in the surrounding landscape at the largest spatial scale possible so as to attract species that are sensitive to human activity.

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FIGURES and TABLES

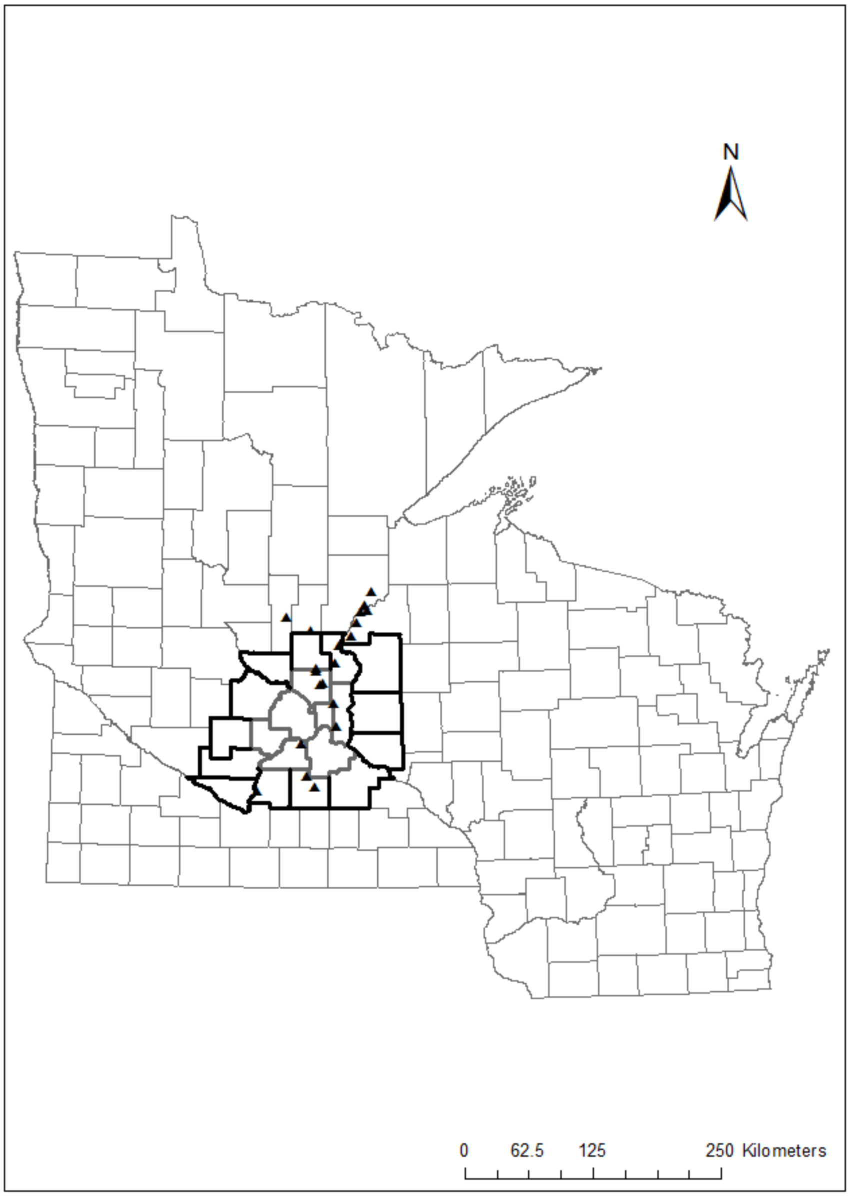


Figure 1. Study area and locations of twenty sites (black triangles), the metropolitan zone includes the seven-county area outlined in grey, the collar zone includes the 12 counties adjacent to the metropolitan zone (outlined in black) and the fringe zone includes counties outside of the metropolitan and collar zones. Five sites were located in the metropolitan zone, seven in the collar zone and eight in the fringe zone.

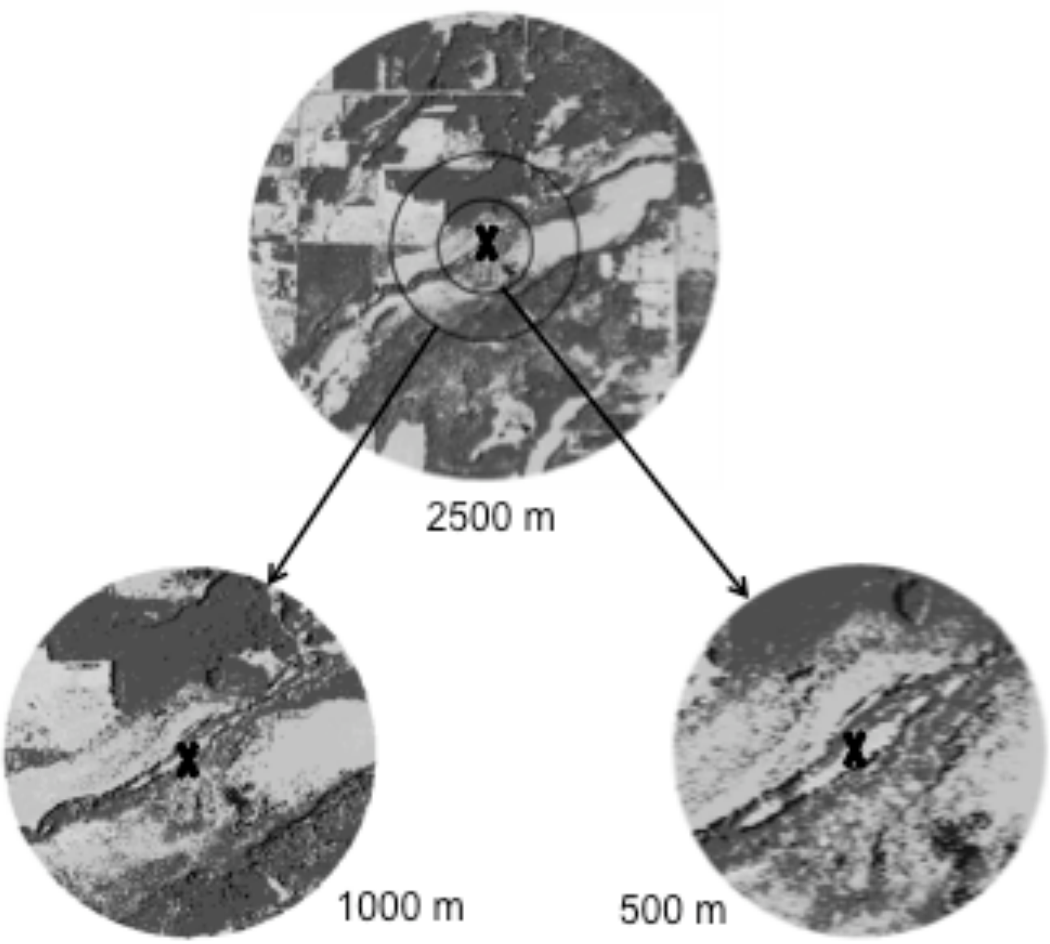


Figure 2. Urban and cropland area were measured at three spatial scales (500, 1000, and 2500 m) for each peatland. Figure adapted from Whited et al. (2000).

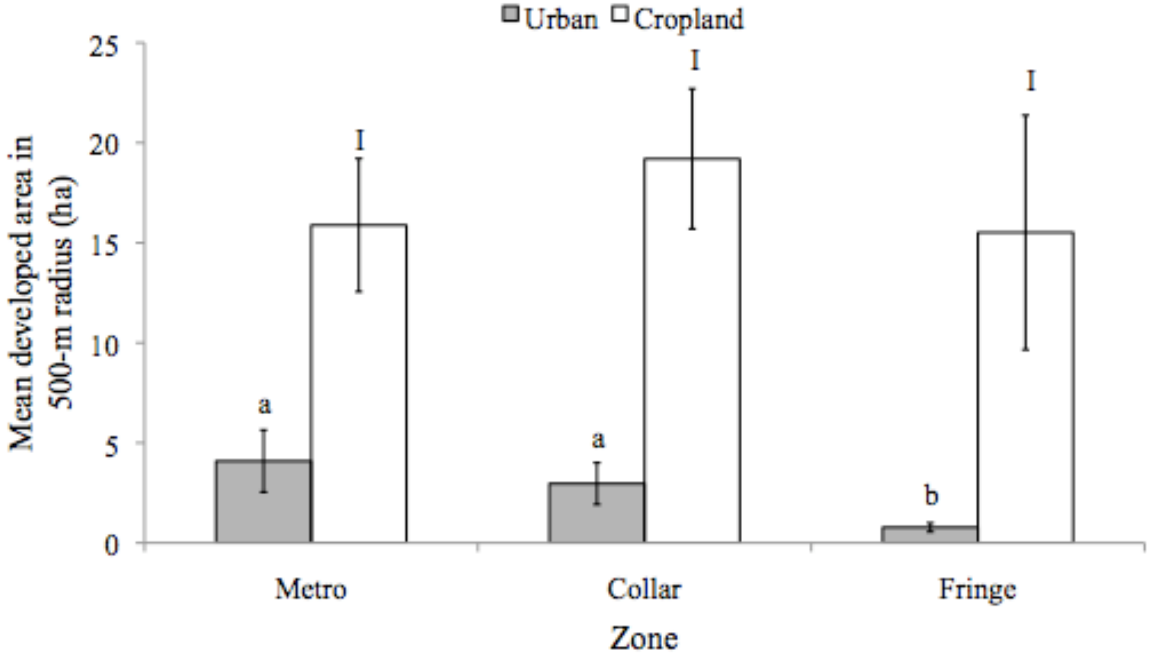


Figure 3. Mean urban area (gray bars; +/- 1 SE) at the 500-m scale was less in the fringe zone compared to the collar (*P* = 0.05) and metro (*P* = 0.02). Mean urban cover did not differ between the metro and collar zones. Mean cropland area (white bars, +/- 1 SE) did not differ between the zones at the 500-m spatial scale.

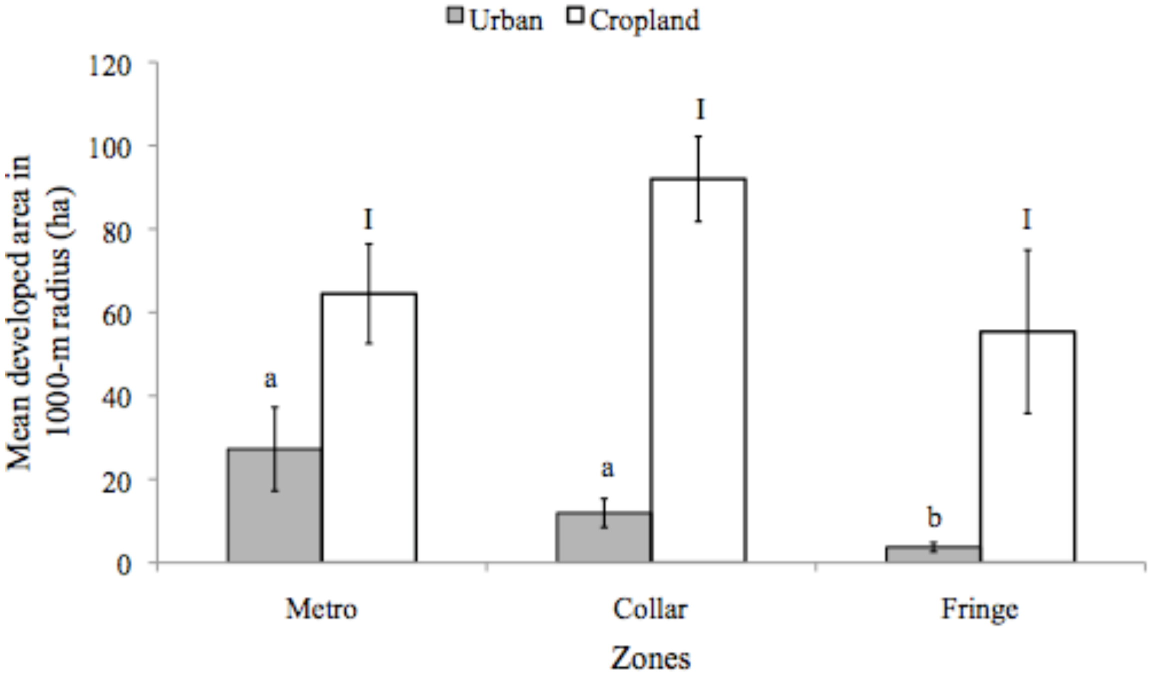


Figure 4. Mean urban area (gray bars; +/- 1 SE) at the 1000-m scale was less in the fringe than the collar (P = 0.03) and metro (P = 0.01) zones. Mean urban cover did not differ between the metro and collar zones. Mean cropland area (white bars, +/- 1 SE) did not differ between the zones at the 1000-m spatial scale.

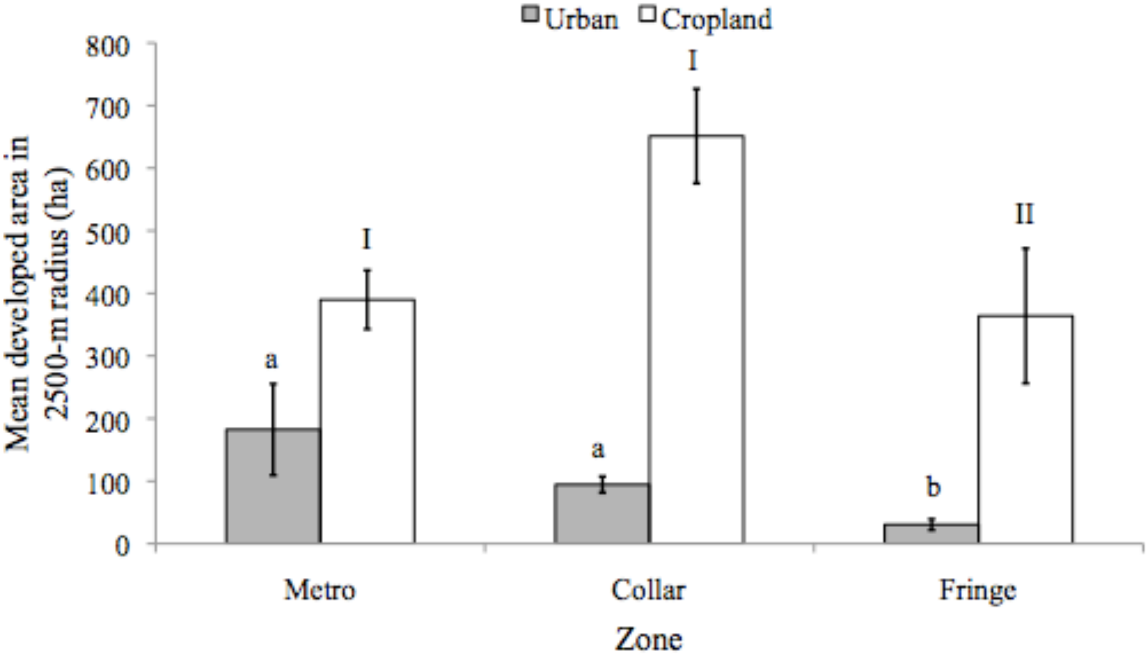


Figure 5. Mean urban area (gray bars; +/- 1 SE) at the 2500-m scale did not differ between the metro and collar zones. Mean urban area was less in the fringe zone compared to the metro ($P = 0.02$) and collar ($P < 0.00$) zones. Mean cropland cover (white bars, +/- 1 SE) was not different between the metro and fringe zones. The collar zone had greater cropland area than the metro ($P = 0.02$) and fringe ($P = 0.05$) zones.

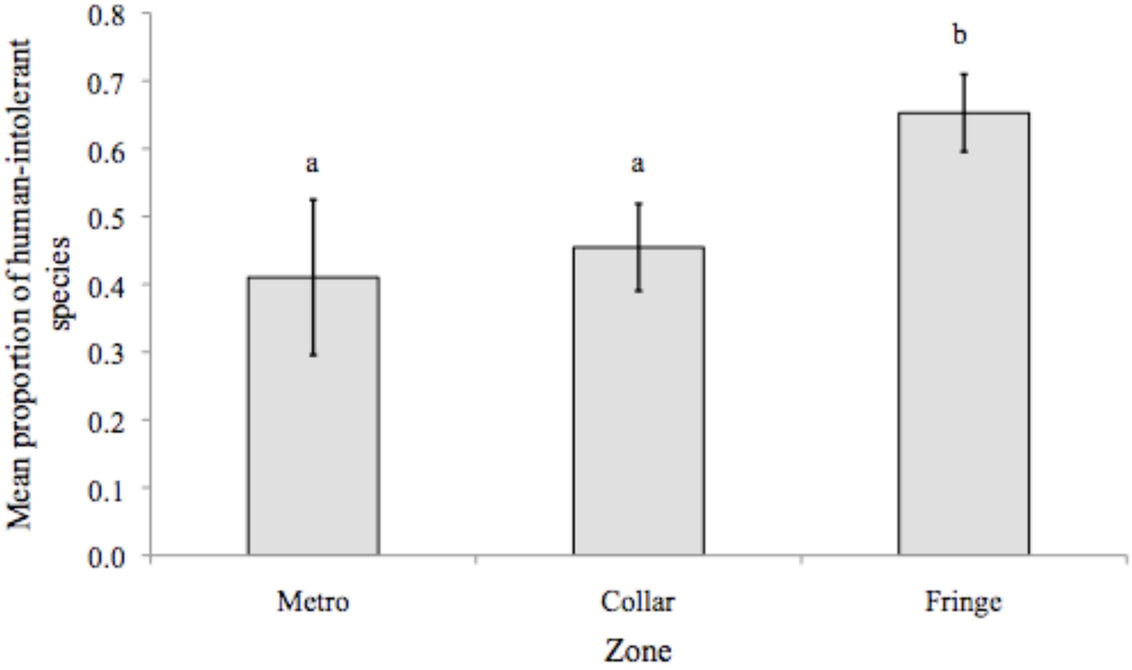


Figure 6. The mean proportion of human-intolerant bird species (+/- 1 SE) was greater in the fringe zone than it was in the collar (*P* << 0.00) and metro (*P* = 0.00) zones. There was no difference in the mean proportion of human-intolerant species between the metro and collar zones.

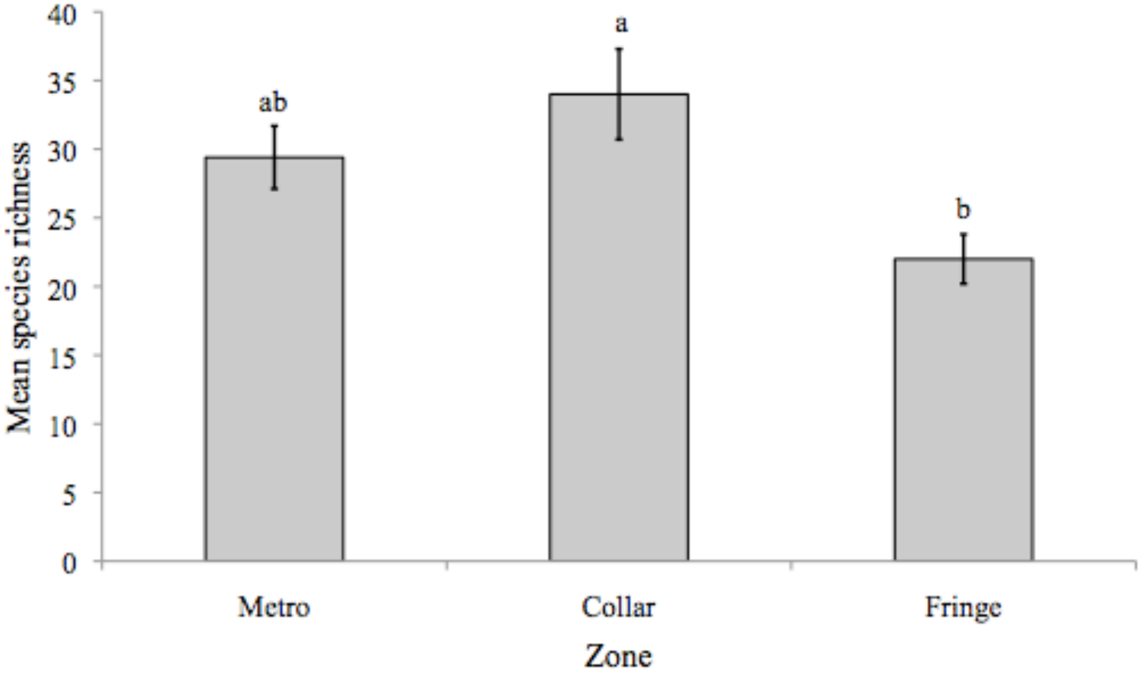


Figure 7. Mean species richness (+/- 1 SE) was higher in the collar zone than in the fringe ($P = 0.03$). Mean species richness in the metro zone was not different from the collar or fringe zones.

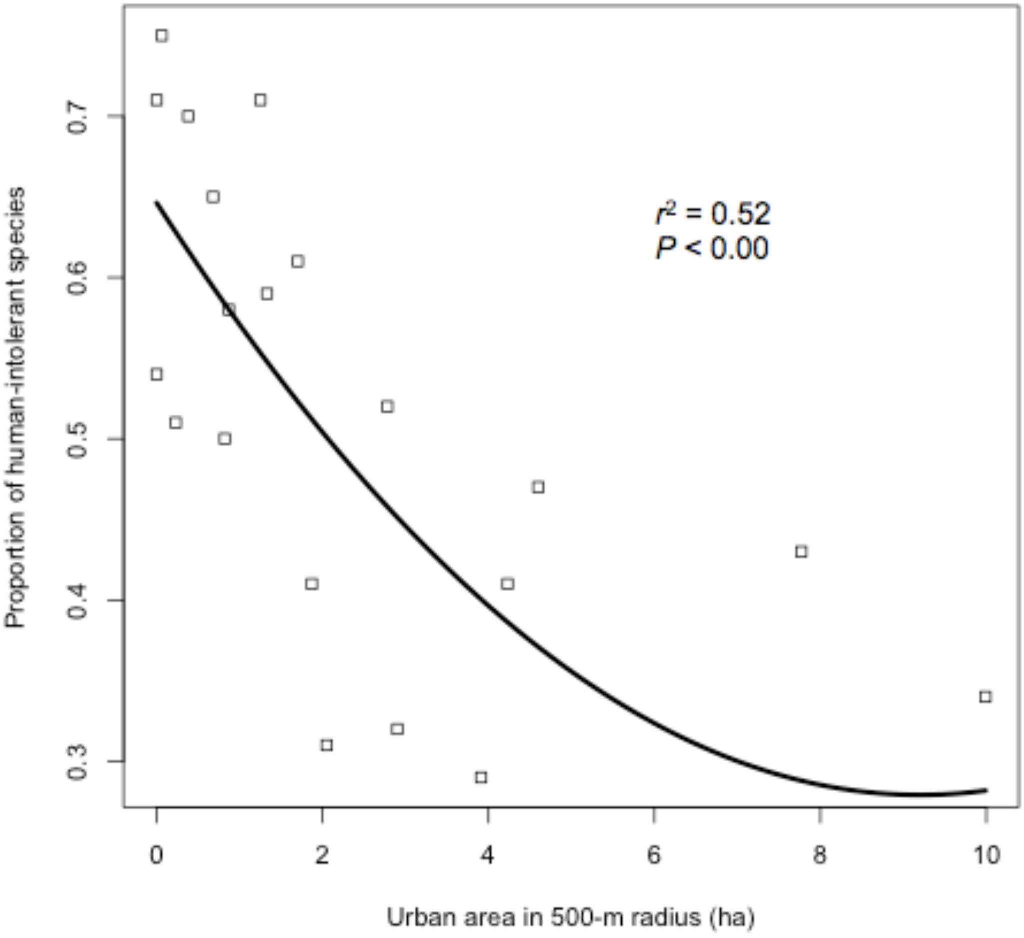


Figure 8. A negative curvilinear relationship is observed on the proportion of human-intolerant species in peatland bird communities as urban cover increases in a 500-m radius ($r^2 = 0.52$, $P < 0.00$).

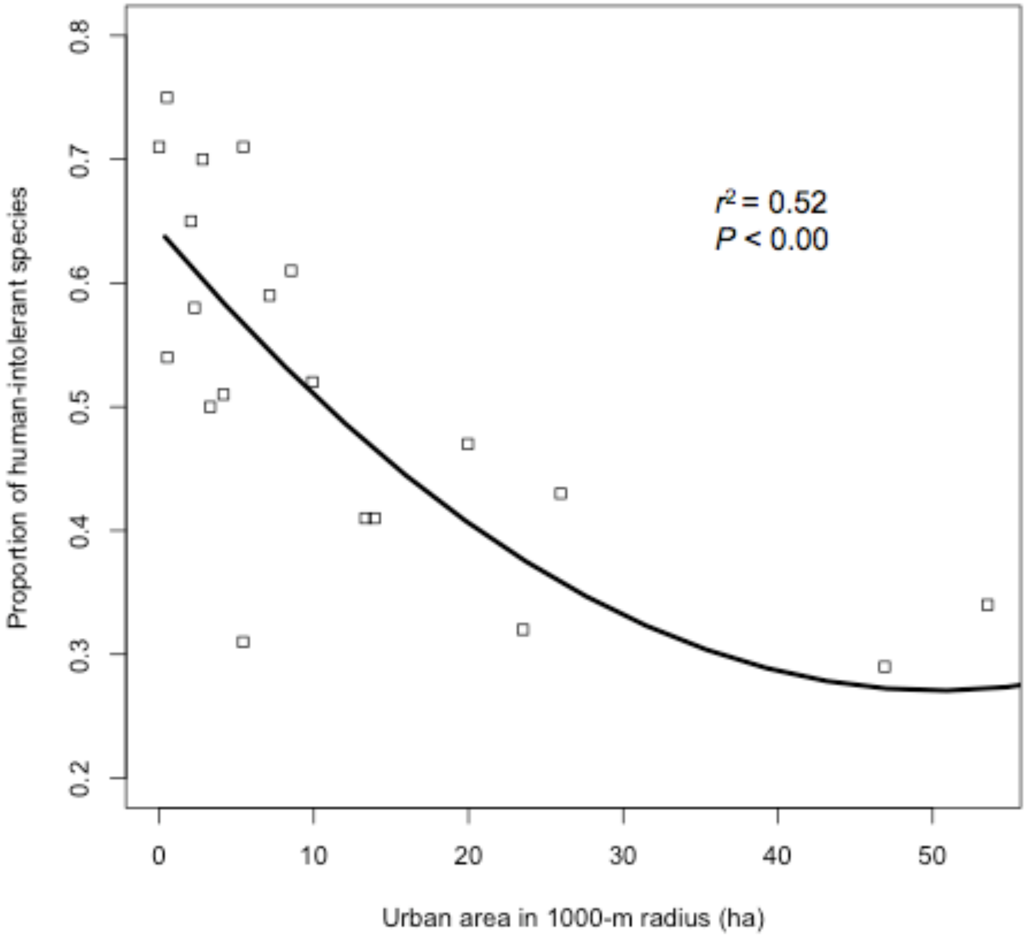


Figure 9. A negative curvilinear relationship is observed on the proportion of human-intolerant species in peatland bird communities as urban cover increases in a 1000-m radius ($r^2 = 0.58$, $P < 0.00$).

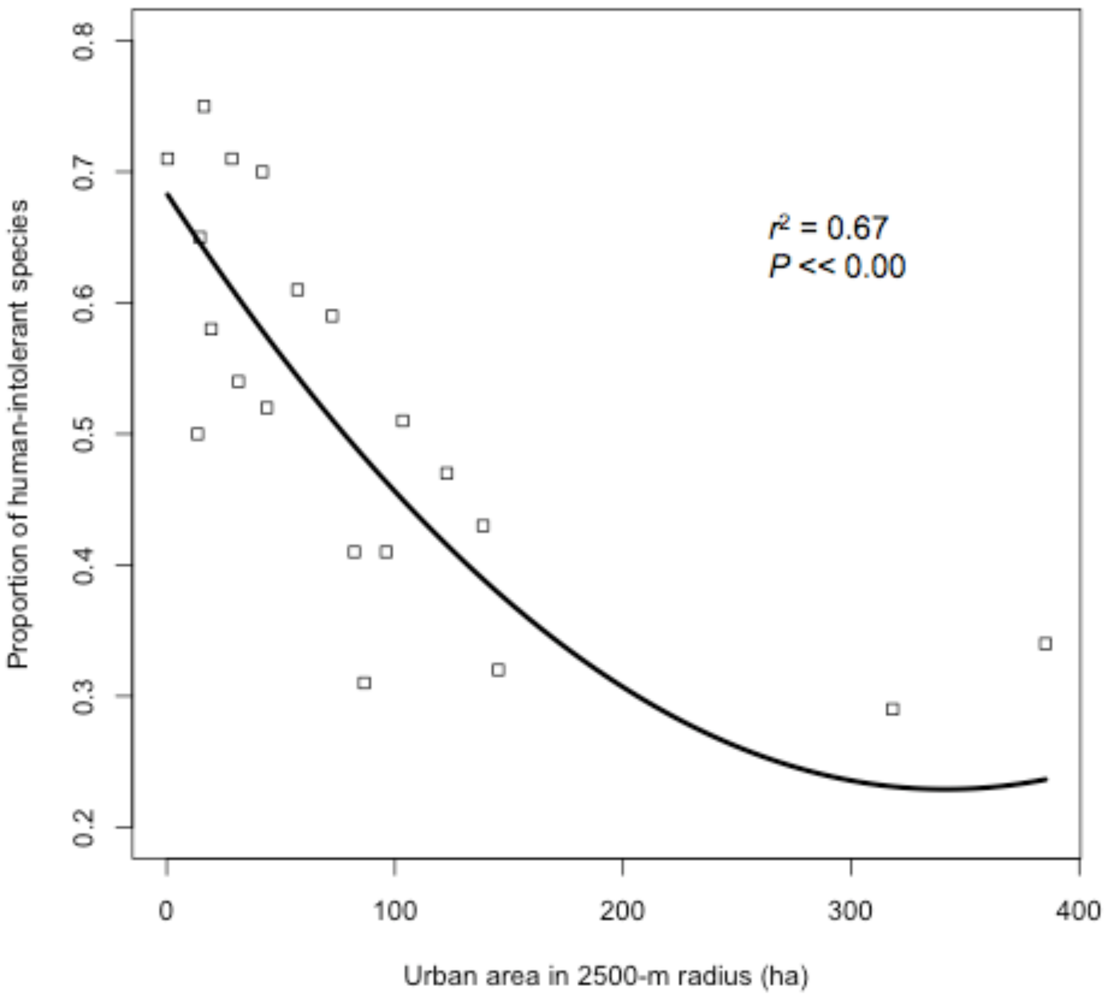


Figure 10. A negative curvilinear relationship is observed on the proportion of human-intolerant species in peatland bird communities as urban cover increases in a 2500-m radius ($r^2 = 0.67$, $P << 0.00$).

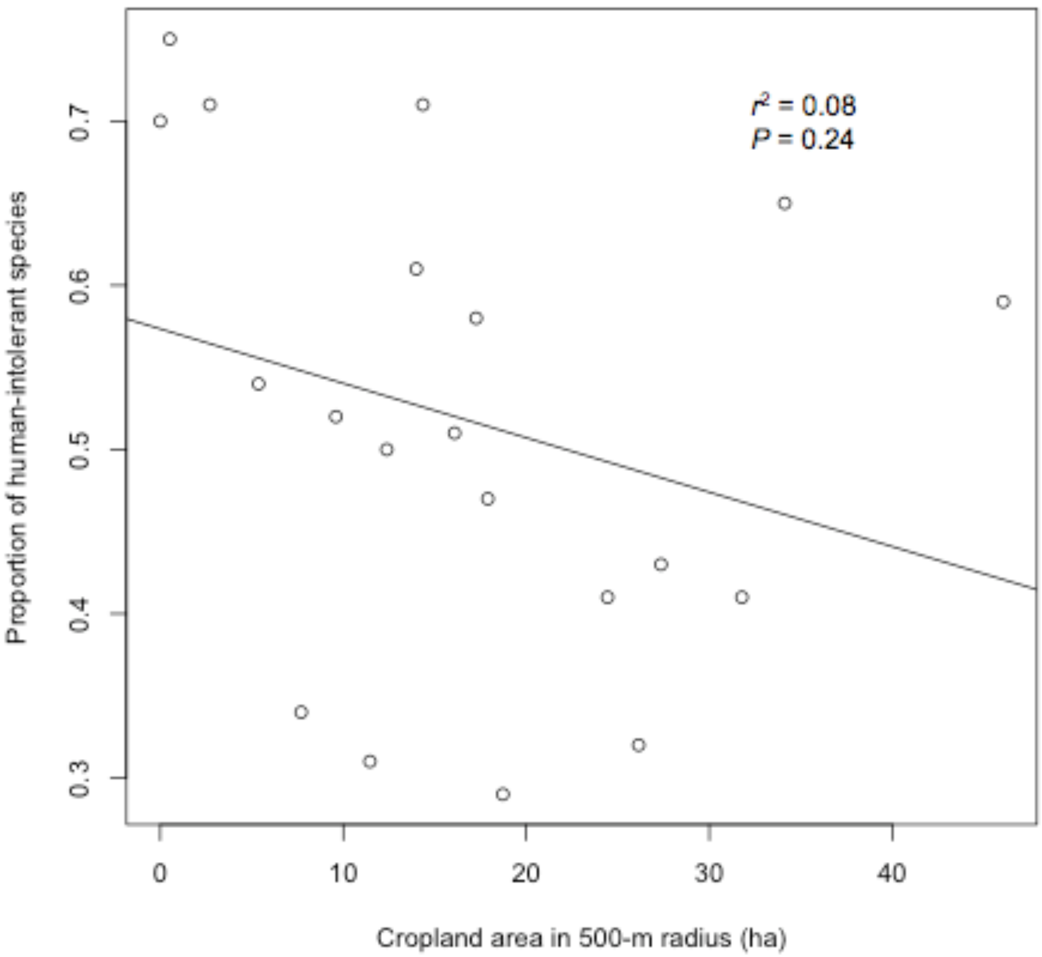


Figure 11. No relationship between the amount of cultivated land in a 500-m radius and the proportion of human-intolerant species was observed.

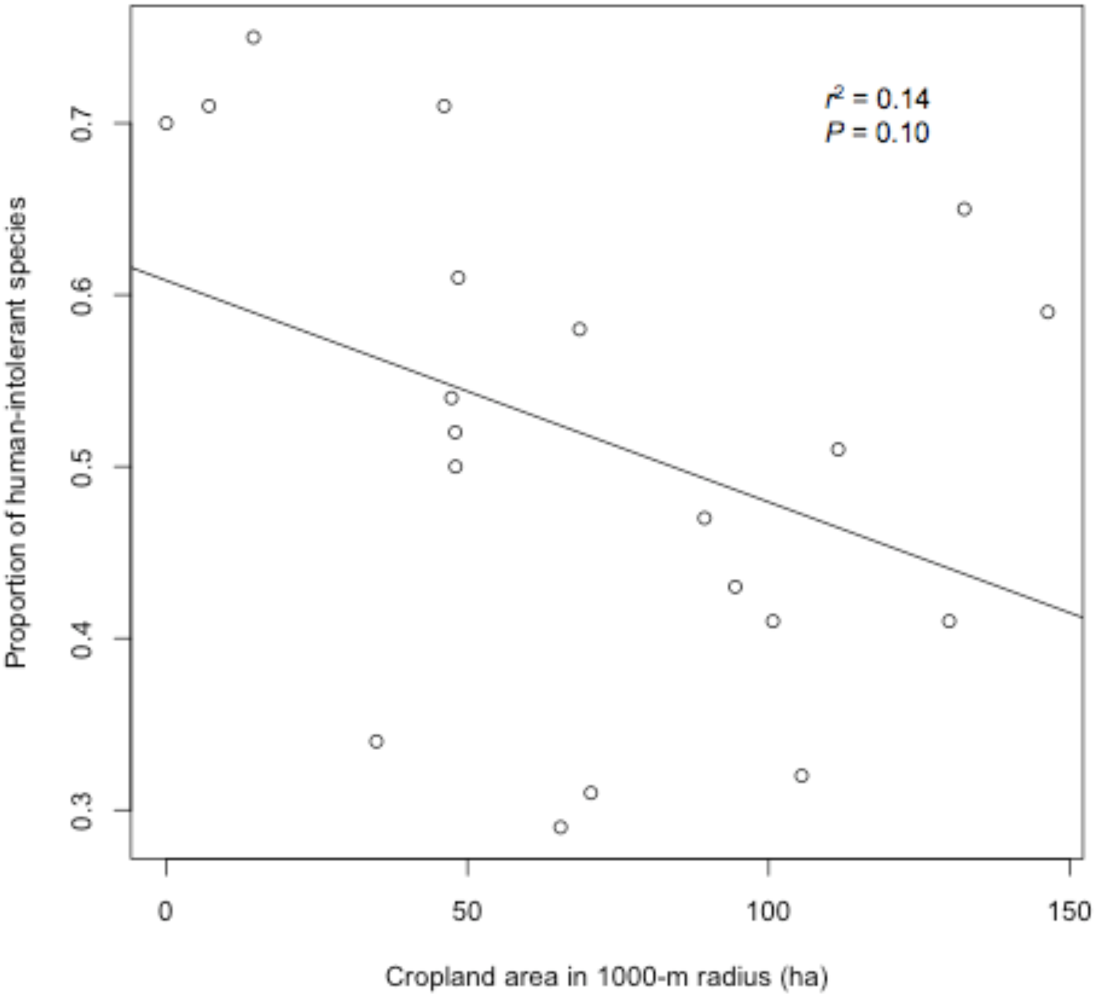


Figure 12. No relationship between the amount of cultivated land in a 1000-m radius and the proportion of human-intolerant species was observed.

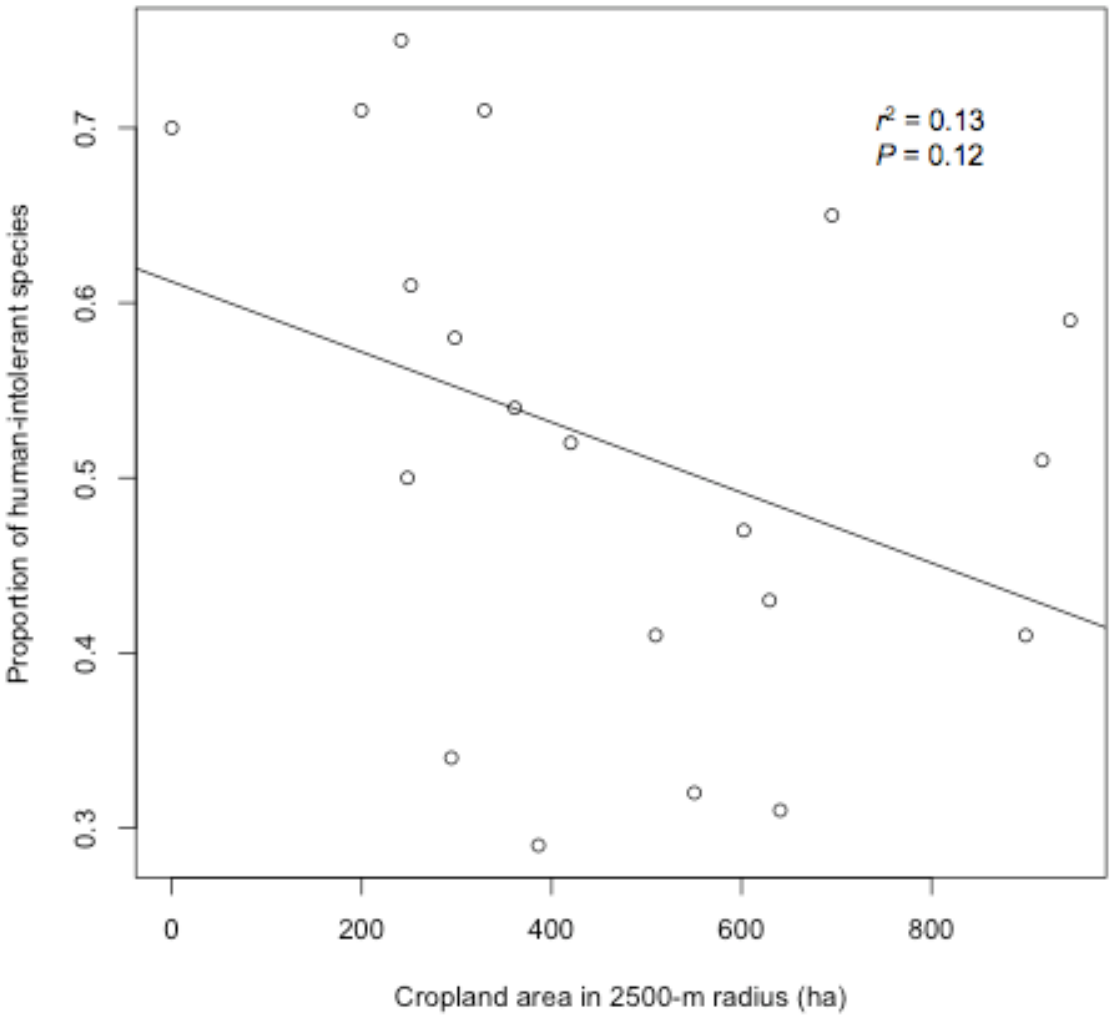


Figure 13. No relationship between the amount of cultivated land in a 2500-m radius and the proportion of human-intolerant species was observed.

Table 1. Area of developed land for the three zones.

Development type	Metropolitan		Collar		Fringe	
	Mean(SE)	Range	Mean(SE)	Range	Mean(SE)	Range
500 m						
Urban cover (ha)	4.1(1.6)	1-10	3.0(1.0)	0-8	0.8(0.2)	0-2
Cropland cover (ha)	15.9(3.3)	8-26	19.2(3.5)	5-31	15.5(5.9)	0-46
1000 m						
Urban cover (ha)	27.2(10.0)	2-54	11.9(3.5)	0-26	3.7(1.1)	0-9
Cropland cover (ha)	64.5(11.9)	35-106	92.0(10.2)	47-130	55.4(19.6)	0-146
2500 m						
Urban cover (ha)	182.4(73.0)	20-385	94.5(13.0)	31-139	30.6(8.7)	0-73
Cropland cover (ha)	389.8(47.0)	294-550	651.2(75.5)	361-916	363.9(107.8)	0-946

Chapter 2

Development and field-test of a hydrogeomorphic model of bird diversity in sub-boreal peatlands in the Upper Midwest.

ABSTRACT

Peatlands are wetlands that accumulate decaying organic plant material (peat) and provide valuable and diverse habitats to a variety of flora and fauna. Though there are no bird species that solely use or breed in natural peatlands many selectively breed or forage in peatland habitats. For example, the palm warbler (*Dendroica palmarum*) selectively nests in open bogs, a type of peatland. I created and tested a rapid-assessment model designed to detect relationships of landscape variables such as peatland area, degree of isolation from other wetlands, extent of adjacent economic development, and vegetative structure on components of bird community composition. Model variables were selected because previous studies have shown them to influence bird communities. One type of rapid-assessment model, known as a hydrogeomorphic model (HGM), numerically combines landscape features to predict what are called functional capacities, parameters such as water flow or animal abundance. Following HGM procedures, I calculated functional capacity index scores (FCIs) using standardized values for model variables. All values were standardized on a scale from 0 to 1; where 1 represents the best support for birds that use peatlands with natural plant communities. A non-significant positive relationship was found between FCI scores and peatland bird community composition

measures. It is suggested that a new model needs to be developed in order to rapidly assess the value of peatlands for the support of bird communities as well as for other faunal groups.

INTRODUCTION

Peatlands are wetlands that accumulate decaying organic material (peat), and although they possess valuable resources such as peat and timber and can be used in agriculture (e.g. cranberry farming), they also provide valuable habitat for many plants and animals (Glaser 1987). Because of the potential conflict between the economic and ecological values of peatlands, there has been contention among conservationists and developers as to whether peatlands can be developed sustainably and without causing irreconcilable damage to the flora and fauna using them (Väisänen and Rauhala 1983, Desrochers et al. 1998, Chapman et al. 2003, Lachance and Lavoie 2004). Anthropogenic development for urban expansion, agriculture and forestry increases the isolation of sub-boreal peatlands that are already insular in nature. Furthermore, development of the adjacent landscape can change aspects of peatlands such as hydrology and pH, both influence plant community composition and structure (Girard et al. 2002, Churchill 2011, Miller 2011).

The US Army Corps of Engineers (ACE) developed the hydrogeomorphic approach for wetland assessment (HGM) to estimate the gains and losses of wetland function due to human impacts and subsequent mitigation (Brinson 1995). HGM models apply functional capacity indices (FCIs) to rapidly assess the ecological value of wetlands in specified regions for different wetland types (e.g., regional subclasses), for example, prairie pothole wetlands of the Midwest or tidal fringe wetlands of the east coast. The HGM approach utilizes abiotic factors (geomorphic setting, water source and hydrodynamics) and biotic factors (such as vegetation physiognomy, presence of predators, distances from sources of immigrants) to assess the physical, chemical and biological functions of wetlands (Brinson 1993). The HGM approach uses reference sites, a set of sites used to create or test models, typically exhibiting the range of values of variable or conditions serving as predictors in the models. A particular type of reference site, the reference standard site, displays the highest function for the specified variable (e.g., water storage). Explain here what the reference sites and the reference standard sites are used for.

One important use of the HGM approach is to estimate the functionality of wetlands to support wildlife. Various animal taxa have been used in integrity assessment models (namely, Indices of Biotic Integrity (IBIs) and Habitat Evaluation Procedures (HEPs)) as indicators of habitat quality (for example see; Urich and Graham 1983, Ganasan and Hughes 1998, Welsh and Ollivier 1998, Van Dolah et al. 1999, Johnson and Swift 2000, Crozier and Gawlik 2003). Typically, HGM wildlife models have been developed to assess a wetland's support of *all* wildlife; we however, focused on the bird community (Klimas et al. 2011, Noble et al. 2011). Birds may serve as an indicator for wildlife in general in sub-boreal peatlands of the Midwest because Galatowitsch et al. (1999) suggest that bird communities have been linked to land use more frequently than other organismal groups (i.e., plants, fish, amphibians and invertebrates). Birds have often been used as indicators of habitat quality (Canterbury et al. 2000, Browder et al. 2002, Bryce et al. 2002, Crozier and Gawlik 2003); birds are ideal indicator organisms because of their sensitivity to environmental conditions, their affinity to particular habitats and their conspicuous nature that makes them easy to survey. HGM model variables incorporate abiotic and biotic factors that are important for supporting diverse wildlife communities such as wetland area, upland land use and various aspects of habitat cover (for examples see U.S. Army Corps of Engineers 2010, Klimas et al. 2011, Noble et al. 2011). Though HGM procedures have been in use since the early 1990's there has been a general lack in using field data to validate model effectiveness (Eckles et al. 2002, Franklin et al. 2009) and to our knowledge there has not been a validation of any wildlife models. I sought to develop and test a HGM model for wildlife that uses natural bird diversity as an indicator of the ability of a peatland to support a diverse natural wildlife community. To determine relative values of use of sub-boreal peatlands by birds that typically use peatlands, we developed a hydrogeomorphic model (Equation 1). The reference domain or geographic area of applicability of the model, included peatlands located between the boreal and temperate eco-regions (i.e., sub-boreal) of the Midwest (Fig. 1). Bird abundance is often dependent on habitat structure and in sub-boreal peatlands bird communities are comprised of birds that typically use grasslands, deciduous forests and boreal forests (Neimi and Hanowski 1992).

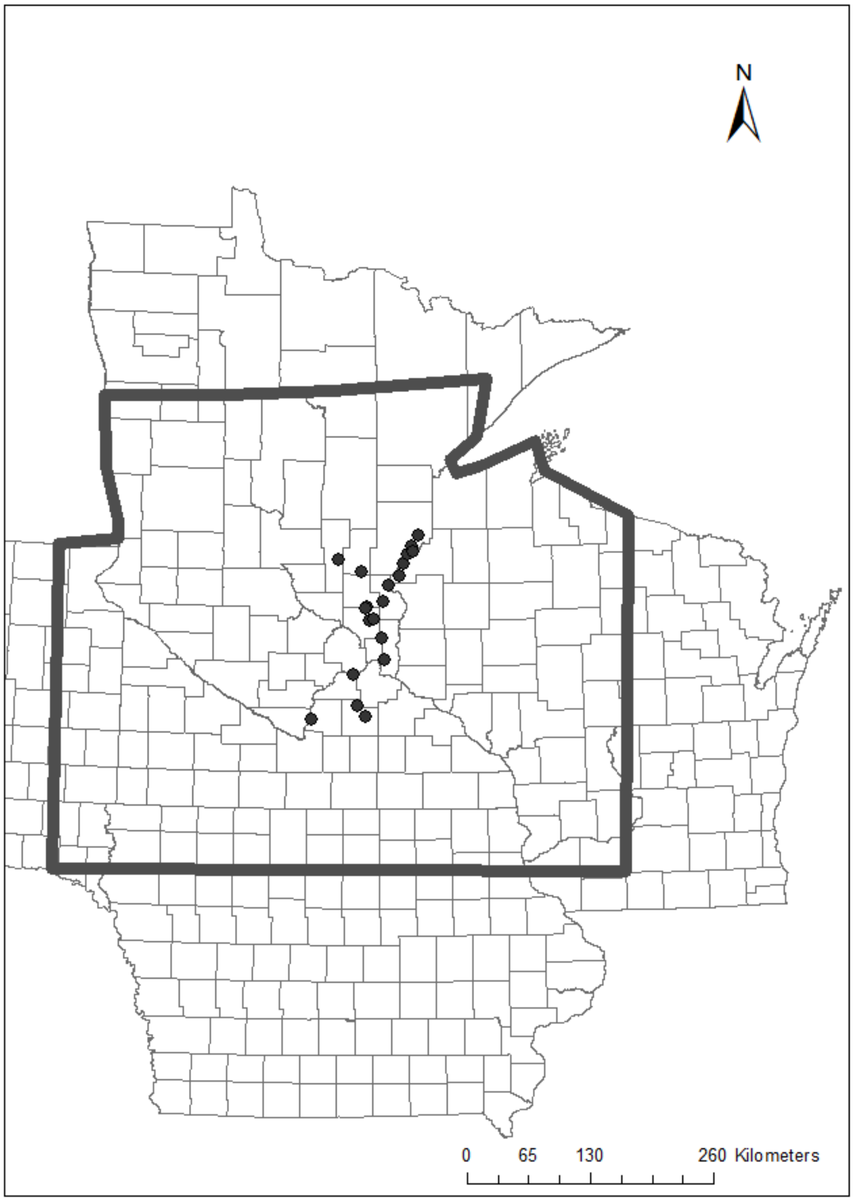


Figure 1. The reference domain (dark grey line) includes the transitional area between the boreal (northern MN and north-western WI) and temperate (northern IA and eastern SD) eco-regions. The reference domain represents the area of applicability of the model. Reference sites (*n* = 28) represented by closed circles were located in boreal, temperate and sub-boreal eco-regions; bird surveys were conducted at 20 of the 28 sites.

My model incorporated variables that were known to influence bird assemblages in prior studies and would be easily measured by field technicians. I estimated the relationship between each variable and its corresponding subindex value. I developed the subindices by standardizing the variation in each landscape variable using a 0 to 1 scale among the reference sites, such that a subindex value of 1 is representative of a reference standard site. Subindex values were entered into the model based on their assigned level of importance to calculate a functional capacity index (FCI) score; FCI scores are scaled between 0 and 1 where a site that receives a score of 1 provides the best support for peatland bird species.

METHODS

Model variables

Peatland area (V_{Area})

The species-area relationship (MacArthur and Wilson 1967) has been documented in a variety of habitats (Brown and Dinsmore 1986, Findlay and Houlahan 1997, Calmé and Desrochers 1999, Rodewald and Vitz 2005) in both pristine and altered landscapes (Shriver et al. 2004). Area is also positively correlated with breeding bird species richness (Craig and Beal 1992), abundance (Mora et al. 2011) and density (Boström and Nilsson 1983). However, the relationship between area and bird abundance, density and species richness, is typically more prominent in small habitat patches (< 50 ha) than in larger patches (Boström and Nilsson 1983, Brown and Dinsmore 1986, Mora et al. 2011). Helzer and Jelinski (1999) suggest that species richness is maximized in large habitat patches (> 50 ha) that are shaped to minimize edge effects for breeding birds in grasslands. Some bird species, e.g. bobolink (*Dolichonyx oryzivorus*) and savannah sparrow (*Passerculus sandwichensis*), are area-sensitive (Robbins et al. 1989, Herkert 1994, Helzer and Jelinski 1999) and avoid using structurally suitable habitat if the habitat patch is small (Brown and Dinsmore 1986, Naugle et al. 1999). The relationship between area and species is driven by the provision of a greater number of micro-habitat types by larger areas which allows the biota to develop a greater number of niches (MacArthur and Wilson 1967). Calmé and Desrochers (1999) found a strong positive relationship between area and micro-habitat richness in peatlands: both area and micro-habitat richness were positively related to bird species richness.

I used area as a more-easily measured surrogate of micro-habitat richness: peatland area (V_{Area}) was measured by delineating the boundary of the peatlands in ArcMap 10 using National Wetland Inventory (NWI) shapefiles within a 2000-m radius of each vegetation survey point established by Clement (2011). A positive relationship between wetland area and the subindex value was assumed up to 100 ha (Fig. 2). Peatlands ≥ 100 ha were assumed to be large enough to attract and support area-sensitive species and provide a greater diversity of micro-habitats.

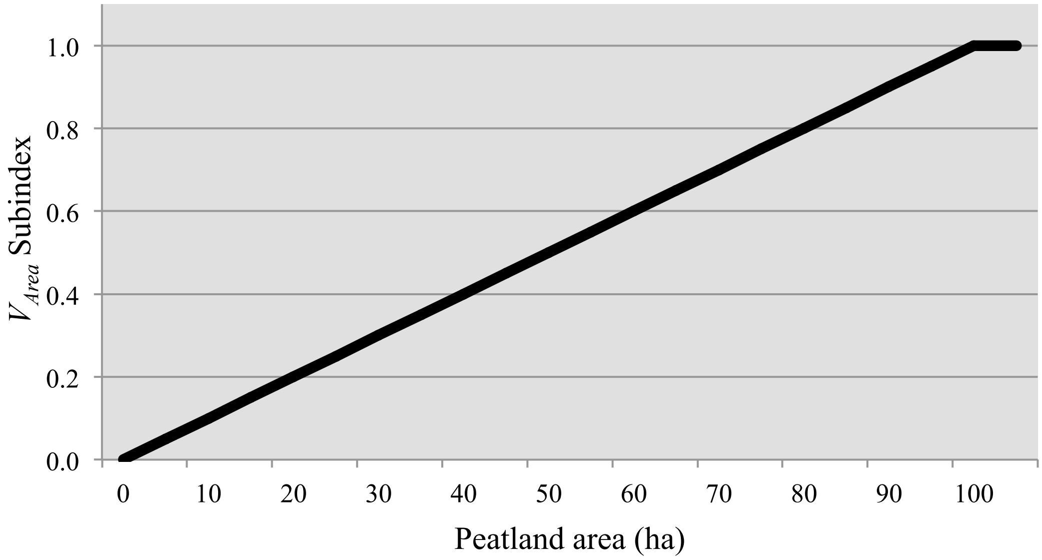


Figure 2. The relationship between the subindex value and peatland area (ha) is assumed to increase linearly up to 100 ha. It is assumed peatlands ≥100 ha will support area sensitive species and provide more heterogeneous habitats and niches leading to increased biodiversity.

Degree of Isolation (V_{Isol})

Isolated habitat patches exhibit a lower rate of colonization by new species and a higher rate of local extinction (MacArthur and Wilson 1967). However, the response of bird species to isolation can vary regionally and temporally (Brown and Dinsmore 1986, Paracuellos and Telleria 2004). Furthermore, bird species in unaltered landscapes may be more sensitive to isolation than those in altered landscapes (Shriver et al. 2004). It may be that bird species that select habitats with low levels of human activity prefer areas that are less isolated from similar habitats. The degree of isolation may affect user species (species that primarily forage in wetlands) and breeder species (species that reproduce in wetlands) differently. Craig and Beal (1992) found that user richness was positively correlated with the proximity to other wetlands whereas breeder richness was not;

however Smith and Chow-Fraser (2010) found that obligate marsh-nesting species richness was greater on less-isolated wetlands.

Although birds are reputed for their long-distance travels, non-migratory movements (e.g., daily movements of resident birds and that of nesting Neotropical migrants) are often inhibited by subtle barriers, such as deforested patches in a forested landscape. Movement of individuals between habitat patches across barriers (habitat gaps and between-patch matrix habitat) is unique for each bird species. Some bird species will move longer distances between habitat patches; for example northern flickers move farther than many other species (600 m; *Colaptes auratus*) while black-throated green warblers typically avoid crossing patches greater than 25 m (*Dendroica virens*; Harris and Reed 2002).

I measured degree of isolation as the mean distance to the three nearest wetlands ≥ 0.5 ha. A minimum area of 0.5 ha was selected because it is assumed to be large enough to support a territory for common yellowthroat (*Geothlypis trichas*) or sedge wren (*Cistothorus palustris*; Stewart 1953, Burns 1982); both species regularly use peatlands and have relatively small territories compared to other peatland using species. The relationship between the subindex value and the degree of isolation was assumed to decrease linearly between 0.05 – 0.95 km (Fig. 3). Sites with a mean isolation distance ≥ 0.95 km were assumed to always support a low level of peatland bird species. I used movement threshold distances for 12 peatland species (summarized by Harris and Reed 2002) to estimate the relationship between the subindex value and the degree of isolation. The 12 species include resident, short-and long-distance migrants and were therefore deemed representative of peatland bird communities.

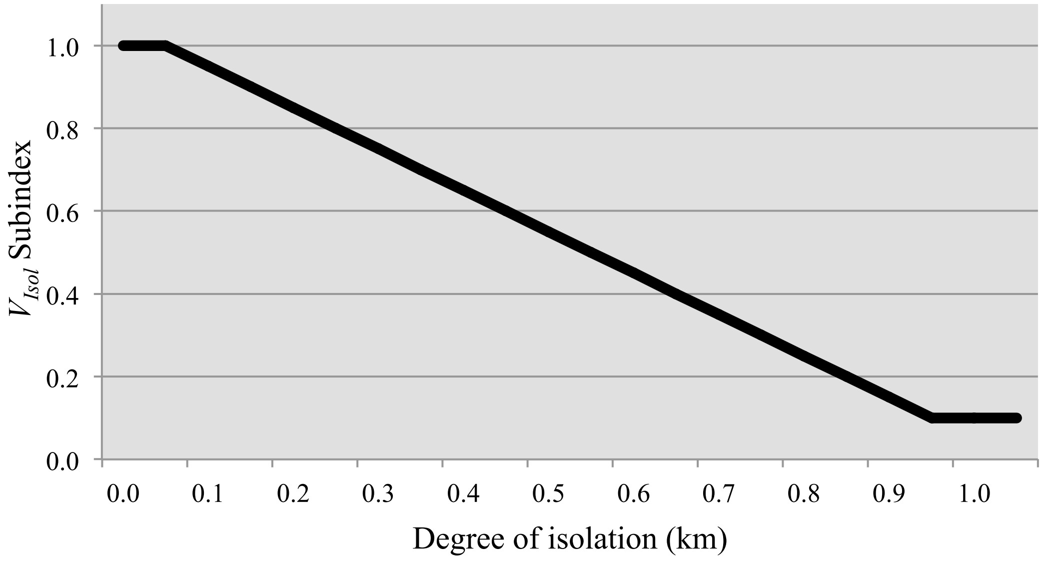


Figure 3. As the degree of isolation, measured as mean distance to the three nearest wetlands ≥ 0.5 ha, increases it is assumed that the subindex value decreases linearly until it approaches 1 km. It is assumed that the greater the degree of isolation the lower the rate of colonization by new species and that fewer migratory and user species will utilize the peatland.

Land Development ($V_{LandDev}$)

The development of land for recreational use, agriculture and urbanization can have negative effects on avian density, abundance and species richness (Findlay and Houlahan 1997, Miller et al. 1998, Fernández-Juricic 2000, Miller et al. 2003, Mallord et al. 2007). Additionally, land development can change species composition (turnover) and lead to local extinctions of species that are vulnerable to human activity (Beissinger and Osborne 1982, Väisänen and Rauhala 1983, Blair 1996, Blair 2004), thereby homogenizing the communities (Devictor et al. 2007). However, not all bird species are negatively affected by anthropogenic disturbances; some opportunistic species benefit (Väisänen and Rauhala 1983, May et al. 2002). Typically, habitat disturbance negatively affects habitat specialists and positively affects generalist and opportunistic species (Dowd 1992, Blair 1996, Miller et al. 1998, Banning et al. 2009) making it difficult to use species richness as an indicator of habitat quality as bird species richness may be similar between sites in urban landscapes and those in rural landscapes (Smith and Chow-Fraser 2010).

Even relatively minor levels of disturbance (*e.g*., a recreational trail or a long pier) that increase human activity can influence bird communities (Miller et al. 1998, Banning et al. 2009). Pedestrian activity can lead to decreases in bird species richness, abundance and population density (Fernández-Juricic 2000, Mallord et al. 2007); furthermore, species turnover has been positively correlated with increased pedestrian rates (Fernández-Juricic 2000). Moderate levels of development may increase species diversity; however, native species are often replaced by widely distributed species (Blair 1996).

Agriculture typically consists of vegetative monocultures interspersed by marginal habitat (e.g., fencerows and small patches of trees planted as wind breaks). When compared to forested and wetland habitats, row-tilled and small-grain crops are used by few bird species and then only for foraging (Best et al. 1995). Though some non-cultivated habitats associated with agricultural landscapes, such as wooded fence-rows, may support high avian abundance they may function as ecological traps where birds suffer low productivity due to nest predation, human disturbance and brood parasitism (Best 1986).

Severe levels of disturbance such as peat mining and logging can result in bird species turnover in addition to reductions in species richness and abundance of birds using peatlands (Väisänen and Rauhala 1983, Niemi and Hanowski 1984*b*, Desrochers et al. 1998). Habitat loss from commercial development in peatlands leads to changes in bird species composition and abundance due to the effects of altered vegetation structure and composition. Due to the significant negative effect that land development has on natural bird communities, specifically on species that are sensitive to human activity I identified land development with remote-sensing data in ArcMap. I did not measure undeveloped land-use types (water bodies, forested and non-agricultural herbaceous) because as the proportion of developed land increased the proportion of undeveloped land decreased. I measured the area of total development (impervious and cropland combined) within a 500-m radius from reference sites. The relationship between the subindex value and the proportion of land development was predicted to decrease linearly to a value of 0.10 at complete land development (Fig. 4).

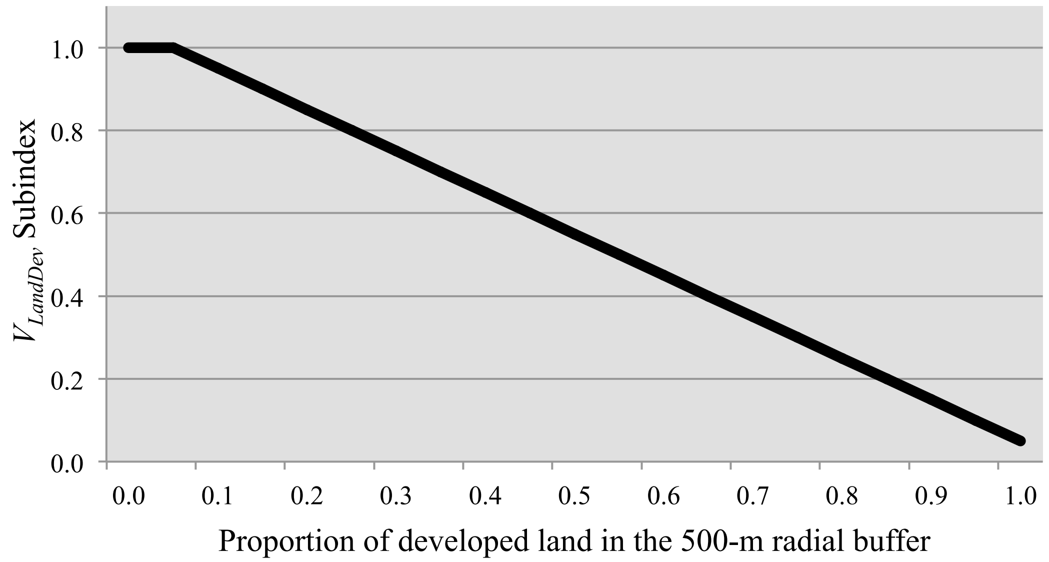


Figure 4. The relationship between the subindex value and the proportion of the 500-m buffer that is developed land is presumed to be negative to a value of 0.1. As developed land increases habitats become more susceptible to invasion by opportunistic species leading to the homogenization of bird communities.

Natural vegetation (V_{NatVeg})

Plant community composition and structure in peatlands is greatly influenced and are in part determined by hydrology, pH and peat thickness (Girard et al. 2002, Churchill 2011, Miller 2011) and is important because bird communities are sensitive to vegetation structure (Hanowski et al. 1997, Desrochers et al. 1998) in terms of foliage height diversity (MacArthur and MacArthur 1961). Plant species composition is not always related to avian support (MacArthur and MacArthur 1961); however, there is a relationship between tree species richness and bird species richness and density (James and Wamer 1982). For example, the distribution of wood warblers (Parulidae) in north-central Minnesota is determined by habitat structure (Collins et al. 1982). Species richness, abundance and density are greater in landscapes with greater habitat complexity

(Niemi and Hanowski 1984*a*, Hanowski et al. 1997, Desrochers et al. 1998). Habitat preferences of bird species can vary annually and seasonally (Brewer 1967, Wilson et al. 1998); making it difficult to model habitat support at the community level. Further complicating matters, generalist species may not display habitat preferences as strongly as specialist species (Wilson et al. 1998) because of their ability to utilize diverse habitats. Although the group of birds identified as peatland users includes both habitat specialists and generalists, the species included are known to regularly use natural peatland habitat (Neimi and Hanowski 1992).

I assumed that a site dominated by natural peatland vegetation provides the best support for peatland birds compared to a site dominated by aquatic or marsh-like vegetation. An impacted site was likely to have plant community that was not representative of a natural peatland and would therefore support an avian community that was not representative of a natural peatland. Sites were identified as having a natural or unnatural peatland plant community from dominant indicator species (Clement 2011). Natural sites were assumed to provide the best support for peatland birds and therefore received a multiplier in the subindex equation of 1.0; impacted sites were assumed to retain only marginal support for peatland birds and received a multiplier of 0.5. Multiplier scores were weighted accordingly for peatlands that had both natural and unnatural peatland plant communities. For example, a peatland with two natural vegetation communities (both score a 1.00) and one unnatural (would score a 0.50) received a multiplier score of 0.83, or {(1.00 + 1.00 + 0.50)/3.00}.

Model

After subindex values were determined using the methods aforementioned they were entered into the FCI (Equation 1) to determine the functional capacity of the peatland. Within the FCI, I assumed that the mean value of area (V_{Area}) and isolation (V_{Isol}) were equal to that of land development ($V_{LandDev}$), as area and isolation are not consistently influential on avian communities whereas development is. Furthermore, area and isolation are equilibrial in nature (MacArthur and Wilson 1967) and their influence on species richness is the result of the interaction between them; as evident in cases where species richness on small less-isolated patches have greater species richness than large isolated patches (Brown and Dinsmore 1986). The multiplier (V_{NatVeg}) insures that sites are given an FCI score that reflects a peatland bird community; for example a large less-isolated site in an undeveloped landscape that is dominated by an upland plant community would receive an FCI score of 0.5 because the bird community was expected to reflect an upland community. Linear regressions were used to determine if a relationship existed between FCI scores and the proportion of peatland birds within each community. I predicted that FCI scores would be positively correlated with the proportion of peatland birds within each community.

$$FCI = \left(\frac{\left(\left(\frac{V_{Area} + V_{Isol}}{2} \right) + V_{LandDev} \right) * \left(V_{NatVeg} \right)}{2} \right)$$

> **Equation 1.** The functional capacity index (FCI) for the support of birds in sub-boreal peatlands is modeled as being dependent on peatland area (V_{Area}), degree of isolation (V_{Isol}), the proportion of economic development in the buffer ($V_{LandDev}$), and the dominant vegetation of the peatland (V_{NatVeg}). Peatlands that are large (≥ 100 ha), are in close proximity to other wetlands, have a nearly pristine buffer and have a natural peatland plant community received a 1. The functional capacity index is scaled from 0 to 1; a peatland that receives an *FCI* score of 1 was presumed to support a bird community typical of an undisturbed peatland in an undisturbed landscape.

I conducted sensitivity analyses to determine how influential each of the additive variables were on model output. I determined model sensitivities by multiplying raw data values by 90%, 100%, and 110% for V_{Area}, V_{Isol} and $V_{LandDev}$ separately and then recalculating FCI scores for each site using the adjusted subindex values. If FCI scores were changed by more than 5% by any one variable that variable was determined to be too influential on model output and would be adjusted accordingly.

Site descriptions

My sites were selected from reference sites Clement (2011) characterized based on abiotic data and plant community composition. Reference sites represented the range of peatlands along the bog-fen continuum; bogs in part are defined as ombrotrophic (rain-fed) peatlands and fens are defined as minerotrophic (ground-water fed) peatlands. Plant species typically found in bogs were *Sphagnum* spp., leatherleaf (*Chamaedaphne* *calyculata*), *Carex oligosperma* and tamarack (*Larix laricina*). In non-forested fens, herbaceous species included native species such as bulblet-bearing water-hemlock (*Cicuta bulbifera*) various *Carex* spp. and tufted loosestrife (*Lysimachia thyrsiflora*) and at some sites invasive species, namely reed-canary grass (*Phalaris arundinacea*). Plant species of forested-fens include common ladyfern (*Athyrium filix-femina*), skunk cabbage (*Symplocarpus foetidus*), *Cornus* spp., *Salix* spp., eastern white-cedar (*Thuja occidentalis*) and black spruce (*Picea mariana*). Measures of pH and electrical conductivity (µS) of reference sites ranged from 4.4(0.2) to 7.1(0.1) and 112.1(32.9) to 500.9(71.7), respectively. Total alkalinity (mg/L) ranged from 17(4) to 222(23) and available phosphorous (mg/L) ranged from 0.9(0.2) to 2.3(0.4). All values are reported as means(SE).

Birds of sub-boreal peatlands

To assess the ability of the model to provide the best support for bird communities we must first define the focal community expected to be typical of an undisturbed site. Defining an avian community that adequately represents peatlands is difficult, in part because peatlands tend to occur along a continuum from fens to bogs and support myriad plant communities from open herbaceous to closed coniferous forest. Additionally, although no bird species use peatlands solely, Neimi and Hanowski (1992) compiled a list of 110 bird species that regularly use natural peatlands for reproduction, foraging, and resting and feeding sites during migration. I adjusted their bird list appropriately for the sub-boreal peatlands of our sample area, i.e. reference domain, to produce a list of birds expected to be supported by reference standard sites. Bird species were removed from the list if their distribution or spring migration path did not extend into sample region. I retained 97 of the 110 species after filtering out those that did not fit our criteria (Appendix 1).

Bird surveys

I sampled 20 sub-boreal peatlands in eastern Minnesota and western Wisconsin that ranged in size from 0.50 to 1013.00 ha (median 90.50 ha). Sites were located between latitudes of 44.20 and 46.29 and longitudes of 93.55 and 92.29 (Fig. 1). I conducted 121 point counts at 41 circular point count stations (points). Count duration was 10 min and points were surveyed between 1 May and 24 June 2011. The number of points per site varied from one to four and was determined by a combination of wetland size and accessibility. I established points a minimum of 200 m apart to avoid double counting individual birds and > 50 m from the edge, unless we were not able to do so due to the difficulty of the terrain (e.g., deep water or floating bog). I played 30-sec call-response recordings of sora (*Porzana carolina)*, Virginia rail (*Rallus limicola*), yellow rail (*Coturnicops noveboracensis*), American bittern (*Botaurus lentiginosus*), least bittern (*Ixobrychus exilis*) and pied-billed grebe (*Podilymbus podiceps*) during the first session (1 May to 16 May). The use of a combination of call-response and passive surveys for secretive marshbirds is the commonly accepted methodology to increase detection probability of marsh bird species to monitor populations (Conway and Nadeau 2010).

I recorded all species and number of individuals seen or heard within a 100-m radius, except for flyovers >20 m height. Digital copies of calls at most points (n = 116; 96 %) were collected using a handheld recorder (H4n, Zoom Corporation, Tokyo, Japan) and compared to written data in the laboratory; recordings are useful for validating vocal identification of species especially when species richness is high (Haselmayer and Quinn 2000). All points were surveyed between 30 min before sunrise and 0900 CDT with wind speeds < 20 km/hr and, at most a light rain. Each point was surveyed three times, except for those in St. Croix State Park, because that site was added to the study during the second sampling session.

Bird community measures

I organized bird data to characterize peatland communities; therefore bird species were classified based on the affinity to natural peatland environments (i.e., either a regular peatland user or not). I assumed a reference standard site would provide better support for peatland birds than mitigated sites. In theory, a reference standard site with an FCI score of 1.0 should have a peatland bird community that is solely (a proportion of 1.0) composed of peatland species.

RESULTS

Bird community measures

I recorded a total of 2923 birds representing 107 species. Of the 107 species, the common grackle (*Quiscalus quiscula*), bald eagle (*Haliaeetus leucocephalus*) and common loon (*Gavia immer*) were excluded from analysis as they were recorded only as flyovers. The five most abundant species across all study sites in descending rank order were: red-winged blackbird (579; *Agelaius phoeniceus*), common yellowthroat (296; *Geothlypis trichas*), swamp sparrow (168; *Melospiza georgiana*), yellow warbler (157;

Dendroica petechia) and American goldfinch (133; *Carduelis tristis*). Common yellowthroat was the only species that was detected at all study sites. Sixty peatland species were observed during the study, of the remaining 47 species observed, many were woodland specialists (such as the ovenbird [*Seiurus aurocapillus*] and red-eyed vireo [*Vireo olivaceus*]) or wetland obligates (such as the marsh wren [*Cistothorus palustris*] and Virginia rail [*Rallus limicola*]). Peatland bird species richness ranged from 12.0 – 26.0 ($\bar{x}$ = 19.2) for sites and the proportion of peatland birds within each community ranged from 0.5 - 0.8.

Landscape variables

Peatland area ranged from 0.5 – 1192.0 ha (median = 95.5) for reference sites. Reference sites (*n* = 28) included eight that were not included for bird surveys. Eight of the sampled sites met reference standard requirements for area (≥ 100 ha); all reference standard sites were given a subindex value of 1. Mean isolation distance ranged from 0.02 – 1.15 (km) for reference sites; nearly half of the sampled sites (*n* = 9) were identified as reference standards (mean distance ≤ 0.1 km). The proportion of development (cropland and impervious land-use combined) within a 500-m radius ranged from 0.00 – 0.62; three sites met reference standard criteria (proportion ≤ 0.05). Eleven of the 20 sites had vegetation communities that were representative of a natural peatland community.

FCI scores ranged from 0.29 – 1.00 ($\bar{x}$ = 0.66) for sampled peatlands. No significant relationship between FCI scores and the proportion of peatland birds in the community was found using linear regression analysis (Fig. 5). Sensitivity analyses suggest that V_{Area}, V_{Isol} and $V_{LandDev}$ did not disproportionately influence FCI scores (Fig. 6). Increasing V_{Area} values did not change mean FCI scores and decreasing V_{Area} values only slightly decreased the mean. Manipulating V_{Isol} values (+/- 10 %) did not affect mean FCI scores and adjusting $V_{LandDev}$ only slightly changed mean FCI scores.

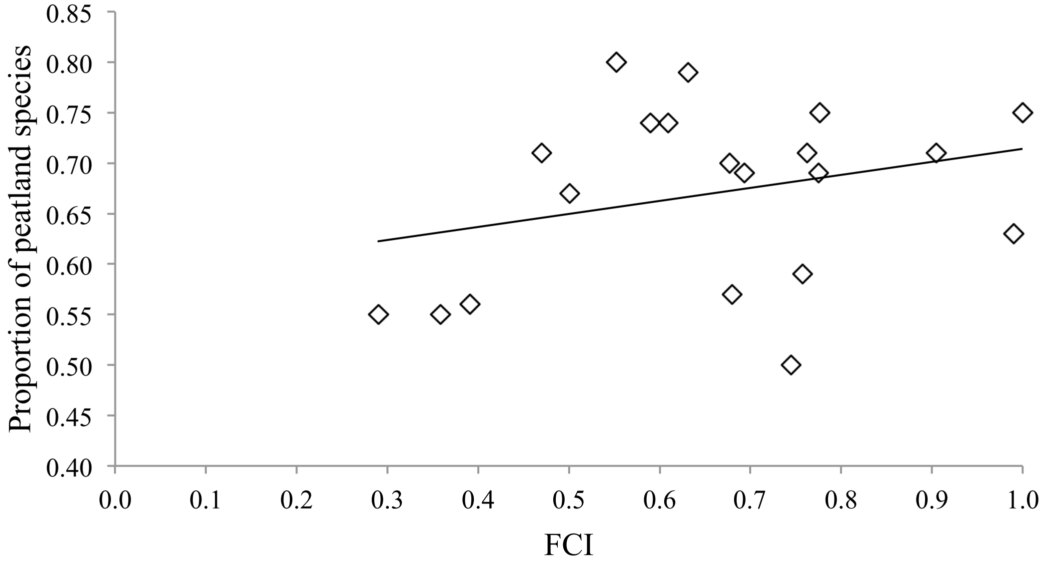
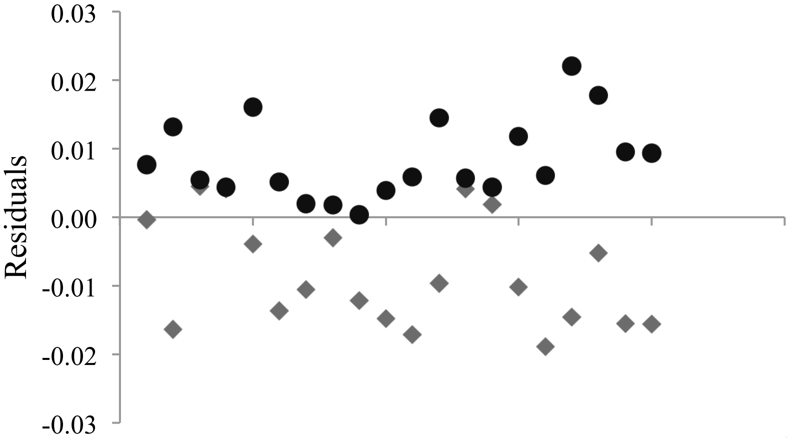


Figure 5. No relationship was found between FCI scores and the proportion of peatland bird species in the community ($r^2 = 0.08$, p-value = 0.22).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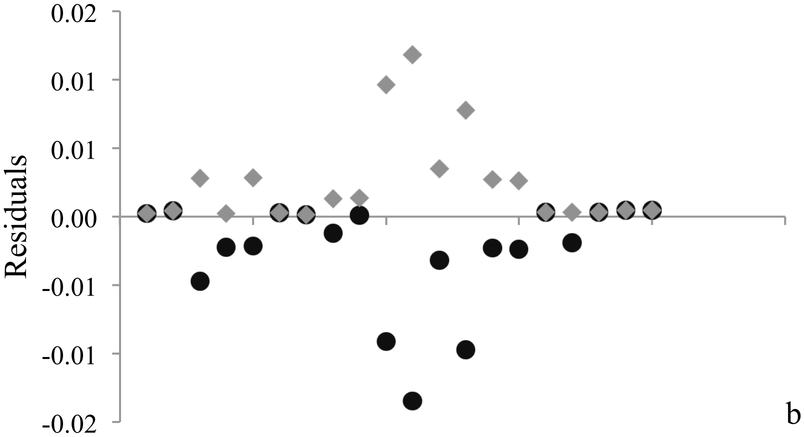
Residuals

a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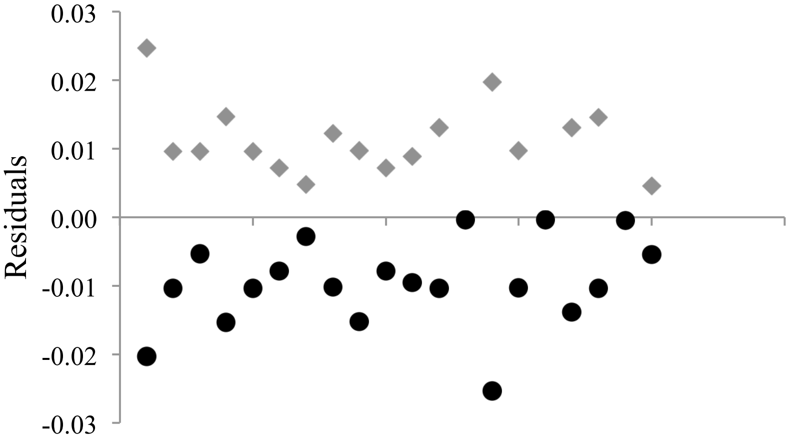
Residuals

b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Residuals

c

Figure 6. Plots of residuals were used to determine how the FCI model was influenced by subindex values. FCI scores using unaltered values for all model variables are represented by the x-axis (0). Black circles represent values that have been increased by 110 % and grey diamonds represent those values that have been decreased by 90 %. Deviation from FCI values calculated with unaltered values appears to be similar for 90 and 110 % changes in V_{Area} (a), V_{Isol} (b), and $V_{LandDev}$ (c).

DISCUSSION

Overall the model did not accurately predict the proportion of peatland birds within communities; however refining the model with step-wise regression methods or by reassigning subindex value distribution may net a better relationship. Care should be taken however, so that future modeling efforts do not fit the model to the data, rather once it is refined it could be used as a predictive tool for the ability of other sub-boreal peatlands to support peatland bird communities. The eight reference sites that were not included for bird surveys would serve well for the purpose of testing the refined model.

Though we attributed the highest subindex values to large peatlands (≥ 100 ha) the importance of small wetlands for avian communities should not be discounted as small wetlands that are part of complexes are very important for preserving biodiversity (Gibbs 1993) and may have greater species richness than large isolated wetlands (Brown and Dinsmore 1986, Craig and Beal 1992). Additionally, small patches of forest habitat are important for migrating passerines as stop-over sites, (Swanson et al. 2003). Furthermore, many bird species (e.g., red-winged blackbird [*Agelaius phoeniceus*], sora [*Porzana carolina*] and ring-necked duck [*Aythya collaris*]; Brown and Dinsmore 1986) are not area-dependent and edge species (e.g., red-winged blackbird and American goldfinch [*Carduelis tristis*]; Herkert 1994) may be negatively affected by increased area.

Furthermore, four of the six smallest sites (< 20 ha) had relatively high proportions of peatland species in the community (0.7 – 0.8) and three of the four were highly connected; the fourth site was moderately connected and had the lowest proportion of peatland species of the four. All four sites had natural peatland plant communities. Our results suggest that small less-isolated peatlands with natural peatland plant communities are important for preserving peatland bird communities and future HGM modeling efforts should account for their value. Three of the larger sites (88 - 369 ha) had low proportions of peatland bird species (0.50 – 0.56), two of these had altered vegetation communities and the other was isolated; this provides additional support for rescaling subindex values for V_{Area}.

Land development influences bird community composition, often negatively. The presence of many peatland bird species ($n = 25$) may not be influenced by land development in the adjacent landscape as they are generalists and are not sensitive to human-activity; therefore a sub-boreal peatland in a highly developed landscape could theoretically have natural peatland bird community composed of generalist peatland species. For future modeling efforts it may be of interest to assess the types of land development (urban and agricultural) separately as they influence avian communities differently. Urban development in the landscape surrounding habitat patches influences species richness, leads to species-turnover and will often result in the loss of species that are sensitive to human-activity. Road density within 500 m of study sites may have the greatest influence on bird assemblages (Findlay and Houlahan 1997, Whited et al. 2000). The negative impacts of agriculture on avian communities is much less pronounced than is that of urban development; however agricultural landscapes can function as ecological sinks for bird populations (Best 1986) so they likely negatively impact avian communities. For modeling purposes, agricultural development in the surrounding landscape may not be important for maintaining natural peatland bird communities as agricultural land use does not appear to influence species presence; therefore for the benefit of the model it may need to be removed from inclusion in $V_{LandDev}$ calculations. Support for omitting agriculture land from $V_{LandDev}$ calculations is found for three sites that had high proportions of peatland bird species (0.74 – 0.80) in their respective communities that received lower $V_{LandDev}$ scores primarily based on the amount of agricultural land in the 500-m buffer. Agricultural practices, such as the installation of drainage systems under fields lead to changes in water level and eventually to changes in nearby plant communities; changes in plant communities leads to changes in avian communities and this change will be accounted for by the multiplier variable V_{NatVeg}.

The multiplier, V_{NatVeg}, appears to be useful because it is related to natural peatland bird communities. Three of the four sub-boreal peatlands that did not have representative peatland plant communities had low proportions of peatland bird species (0.55 – 0.56), the remaining site had inundated water levels that influenced plant communities however, structurally the vegetation was similar to a sedge-fen and is likely why it had a higher proportion of peatland bird species (0.71). Eight of the 11 sites that had natural peatland plant communities had high proportions (0.69 – 0.80) of peatland bird species; two of the three that had lower proportions of peatland bird species (0.57 – 0.63) were structurally similar to eastern deciduous upland forests and were used by many species that are found in upland deciduous forests, the remaining site was small (2 ha) and surrounded by upland deciduous forest and was also used by many species typical of upland deciduous forest habitats.

I suggest that a new FCI model needs to be created to monitor and assess the ability of sub-boreal peatlands to support faunal communities that are representative of natural peatlands. I feel that peatland bird species serve as the best indicators of habitat quality for other peatland fauna (e.g., mammals, herptiles, and invertebrates) because of their habitat affinity, sensitivity to human activity and landscape factors, and they are more-easily surveyed. An improved model will allow mitigation efforts to be assessed for the improvement of wildlife habitat and to determine which sites or portion(s) of sites are of the least value to wildlife for future development; additionally the effects of site development can be monitored.

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APPENDICES

Appendix 1. List of all bird species included in analysis and their tolerance of human activity.

Common name	Scientific name	Affinity
Alder flycatcher	*Empidonax alnorum*	Intolerant
American bittern	*Botaurus lentiginosus*	Intolerant
American coot	*Fulica americana*	Intolerant
American crow	*Corvus brachyrhynchos*	Tolerant
American goldfinch	*Carduelis tristis*	Tolerant
American redstart	*Setophaga ruticilla*	Intolerant
American robin	*Turdus migratorius*	Tolerant
Baltimore oriole	*Icterus galbula*	Tolerant
Barn swallow	*Hirundo rustica*	Tolerant
Bay-breasted warbler	*Dendroica castanea*	Intolerant
Belted kingfisher	*Megaceryle alcyon*	Tolerant
Black and white warbler	*Mniotilta varia*	Intolerant
Blackburnian warbler	*Dendroica fusca*	Intolerant
Black-capped chickadee	*Poecile atricapilla*	Tolerant
Black-throated green warbler	*Dendroica virens*	Intolerant
Black tern	*Chlidonas niger*	Intolerant
Blue jay	*Cyanocitta cristata*	Tolerant
Blue-winged warbler	*Vermivora pinus*	Intolerant
Blue-winged teal	*Anas discors*	Intolerant
Bobolink	*Dolichonyx oryzivorus*	Intolerant
Blackburnian warbler	*Dendroica fusca*	Intolerant
Broad-winged hawk	*Buteo platypterus*	Intolerant
Brown creeper	*Certhia americana*	Intolerant
Brown thrasher	*Toxostoma rufum*	Intolerant
Brown-headed cowbird	*Molothrus ater*	Tolerant
Canada goose	*Branta canadensis*	Tolerant
Cape May warbler	*Dendroica tigrina*	Intolerant
Cedar waxwing	*Bombycilla cedrorum*	Tolerant
Chestnut-sided warbler	*Dendroica pensylvanica*	Intolerant
Chipping sparrow	*Spizella passerina*	Tolerant
Clay-colored sparrow	*Spizella pallida*	Intolerant
Common raven	*Corvus corax*	Intolerant
Common yellowthroat	*Geothlypis trichas*	Intolerant
Double-crested cormorant	*Phalacrocorax auritus*	Tolerant
Downy woodpecker	*Picoides pubescens*	Tolerant
Eastern bluebird	*Sialia sialis*	Tolerant
Eastern kingbird	*Tyrannus tyrannus*	Tolerant
Eastern meadowlark	*Strunella magna*	Intolerant
Eastern phoebe	*Sayornis phoebe*	Tolerant
Eastern wood-pewee	*Contopus virens*	Intolerant
Field sparrow	*Spizella pusilla*	Tolerant
Golden-crowned kinglet	*Regulus satrapa*	Intolerant
Golden-winged warbler	*Vermivora chrysoptera*	Intolerant
Gray catbird	*Dumetella carolinensis*	Tolerant
Great blue heron	*Ardea herondias*	Intolerant
Great crested flycatcher	*Myiarchus crinitus*	Tolerant
Green heron	*Butorides virescens*	Intolerant
Green-winged teal	*Anas crecca*	Intolerant
Hairy woodpecker	*Picoides villosus*	Tolerant
House wren	*Troglodytes aedon*	Tolerant

Indigo bunting	*Passerina cyanea*	Intolerant
Killdeer	*Charadrius vociferus*	Tolerant
Least flycatcher	*Empidonax minimus*	Intolerant
Le Conte's sparrow	*Ammodrammus leconteii*	Intolerant
Magnolia warbler	*Dendroica magnolia*	Intolerant
Mallard	*Anas platyrhynchos*	Tolerant
Marsh wren	*Cistothorus palustris*	Intolerant
Mourning dove	*Zenaida macroura*	Tolerant
Nashville warbler	*Vermivora ruficapilla*	Intolerant
Northern cardinal	*Cardinalis cardinalis*	Tolerant
Northern flicker	*Colaptes auratus*	Tolerant
Northern parula	*Parula americana*	Intolerant
Northern rough-winged swallow	*Stegidopteryx serripennis*	Intolerant
Northern shoveler	*Anas clypeata*	Intolerant
Northern waterthrush	*Seiurus noveboracensis*	Intolerant
Orange-crowned warbler	*Vermivora celata*	Intolerant
Ovenbird	*Seiurus aurocapillus*	Intolerant
Palm warbler	*Dendroica palmarum*	Intolerant
Pied-billed grebe	*Podilymbus podiceps*	Intolerant
Pileated woodpecker	*Dryocopus pileatus*	Intolerant
Red-bellied woodpecker	*Melanerpes carolinus*	Tolerant
Red-breasted nuthatch	*Sitta canadensis*	Tolerant
Red-eyed vireo	*Vireo olivaceus*	Intolerant
Red-winged blackbird	*Agelaius phoeniceus*	Tolerant
Ring-necked duck	*Aythya collaris*	Intolerant
Ring-necked pheasant	*Phasianus colchicus*	Tolerant
Rose-breasted grosbeak	*Phuecticus ludovicianus*	Tolerant
Ruby-crowned kinglet	*Regulus calendula*	Tolerant
Ruby-throated hummingbird	*Archilochus colubris*	Tolerant
Ruffed grouse	*Bonasa umbellus*	Intolerant
Sandhill crane	*Grus canadensis*	Intolerant
Savannah sparrow	*Passerculus sandwichensis*	Tolerant
Scarlet tanager	*Piranga olivacea*	Intolerant
Sedge wren	*Cistothorus platensis*	Intolerant
Sharp-shinned hawk	*Accipiter striatus*	Intolerant
Song sparrow	*Melospiza melodia*	Tolerant
Sora	*Porzana carolina*	Intolerant
Spotted sandpiper	*Actitis macularia*	Intolerant
Swainson's thrush	*Catharus ustulatus*	Intolerant
Swamp sparrow	*Melospiza georgiana*	Intolerant
Tree swallow	*Tachycineta bicolor*	Tolerant
Veery	*Catharus fuscenscens*	Intolerant
Virginia rail	*Rallus limicola*	Intolerant
White-breasted nuthatch	*Sitta carolinensis*	Intolerant
White-throated sparrow	*Zonotrichia albicollis*	Intolerant
Winter wren	*Troglodytes troglodytes*	Intolerant
Willow flycatcher	*Empidonax trailii*	Intolerant
Wilson's snipe	*Gallinago delicata*	Intolerant
Wood duck	*Aix sponsa*	Intolerant
Woodthrush	*Hylocichla mustelina*	Intolerant
Yellow warbler	*Dendroica petechia*	Intolerant
Yellow-bellied sapsucker	*Sphyrapicus varius*	Intolerant
Yellow-headed blackbird	*Xanthocephalus xanthocephalus*	Tolerant
Yellow-rumped warbler	*Dendroica coronata*	Intolerant
Yellow-throated vireo	*Vireo flavifrons*	Intolerant

Appendix 2. List of 97 bird species that use peatlands in the reference domain for reproduction, foraging, or as stop-over sites during migration.

Species	Scientific name
Alder flycatcher[a]	*Empidonax alnorum*
American bittern[a]	*Botaurus lentginosus*
American crow[a]	*Corvus brachyrhynchos*
American goldfinch[a]	*Carduelis tristis*
American redstart[a]	*Setophaga ruticilla*
American robin[a]	*Turdus migratorius*
Bay-breasted warbler[a]	*Dendroica castanea*
Black-and-white warbler[a]	*Mniotilta varia*
Black-backed woodpecker	*Picoides arcticus*
Black-billed cuckoo	*Coccyzus erythropthalmus*
Black-billed magpie	*Pica hudsonia*
Black-capped chickadee[a]	*Poecile atricapilla*
Black-throated green warbler	*Dendroica virens*
Blackburnian warbler[a]	*Dendroica fusca*
Blue jay[a]	*Cyanocitta cristata*
Blue-headed vireo	*Vireo solitarius*
Blue-winged teal[a]	*Anas discors*
Bobolink[a]	*Dolichonyx oryzivorus*
Boreal chickadee	*Poecile hudsonica*
Brewer's blackbird	*Euphagus cyanocephalus*
Brown creeper[a]	*Certhia americana*
Brown thrasher[a]	*Toxostoma rufum*
Brown-headed cowbird[a]	*Molothrus ater*
Cape May warbler[a]	*Dendroica tigrina*
Cedar waxwing[a]	*Bombycilla cedrorum*
Chipping sparrow[a]	*Spizella passerine*
Clay-colored sparrow[a]	*Spizella pallida*
Common raven[a]	*Corvus corax*
Common yellowthroat[a]	*Geothlypis trichas*
Connecticut warbler	*Oporornis agilis*
Dark-eyed junco	*Junco hyemalis*
Downy woodpecker[a]	*Picoides pubescens*
Eastern kingbird[a]	*Tyrannus tyrannus*
Evening grosbeak	*Coccothraustes vespertinus*
Golden-crowned kinglet[a]	*Regulus satrapa*
Golden-winged warbler[a]	*Vermivora chrysoptera*
Gray catbird[a]	*Dumetella carolinensis*
Gray jay	*Perisoreus canadensis*
Gray-cheeked thrush	*Catharus minimus*
Greater yellowlegs	*Tringa melanoleuca*
Hairy woodpecker[a]	*Picoides villosus*
Hermit thrush	*Catharus guttatus*
House wren[a]	*Troglodytes aedon*
LeConte's sparrow[a]	*Ammodramus leconteii*
Lesser yellowlegs	*Tringa flavipes*
Lincoln's sparrow	*Melospiza lincolnii*
Magnolia warbler[a]	*Dendroica magnolia*
Mallard[a]	*Anas platyrhynchos*
Nashville warbler[a]	*Vermivora ruficapilla*
Nelson's sharp-tailed sparrow	*Ammodramus nelsoni*
Northern flicker[a]	*Colaptes auratus*
Northern goshawk	*Accipiter gentilis*
Northern harrier	*Cirus cyaneus*

Northern parula[a]	*Parula americana*
Northern waterthrush[a]	*Seiurus noveboracensis*
Olive-sided flycatcher	*Contopus cooperi*
Orange-crowned warbler[a]	*Vermivora celata*
Palm warbler[a]	*Dendroica palmarum*
Philadelphia vireo	*Vireo philadelphicus*
Pine siskin	*Carduelis pinus*
Pine warbler	*Dendroica pinus*
Purple finch	*Carpodacus purpureus*
Red crossbill	*Loxia curvirostra*
Red-breasted nuthatch[a]	*Sitta canadensis*
Red-winged blackbird[a]	*Agelaius phoeniceus*
Ring-necked duck[a]	*Aythya collaris*
Rose-breasted grosbeak	*Pheucticus ludovicianus*
Ruby-crowned kinglet[a]	*Regulus calendula*
Ruby-throated hummingbird[a]	*Archilochus colubris*
Ruffed grouse[a]	*Bonasa umbellus*
Rusty blackbird	*Euphagus carolinus*
Sandhill crane[a]	*Grus canadensis*
Savannah sparrow[a]	*Passerculus sandwichensis*
Sedge wren[a]	*Cistothorus platensis*
Sharp-shinned hawk[a]	*Accipiter striatus*
Sharp-tailed grouse	*Tympanuchus phasianellus*
Short-eared owl	*Asio flammeus*
Solitary sandpiper	*Tringa solitaria*
Sora[a]	*Porzana carolina*
Swainson's thrush[a]	*Catharus ustulatus*
Swamp sparrow[a]	*Melospiza georiana*
Tennessee warbler	*Vermivora peregrina*
Tree swallow[a]	*Tachycineta bicolor*
Veery[a]	*Catharus fuscescens*
White-breasted nuthatch[a]	*Sitta carolinensis*
White-crowned sparrow	*Zonotrichia leucophrys*
White-throated sparrow[a]	*Zonotrichia albicollis*
Wilson's phalarope	*Phalaropus tricolor*
Wilson's snipe[a]	*Gallinago delicata*
Wilson's warbler	*Wilsonia pusilla*
Winter wren[a]	*Troglodytes troglodytes*
Yellow rail	*Coturnicops noveboracensis*
Yellow warbler[a]	*Dendroica petechia*
Yellow-bellied flycatcher[a]	*Empidonax flaviventris*
Yellow-rumped warbler[a]	*Dendroica coronata*

[a] Species was observed.

Appendix 3. Values for area, isolation, land development, multiplier, peatland bird species richness and the proportion of peatland bird species.

Site ID	Area (ha)	Isolation	Proportion of developed land	Natural vegetation multiplier	Peatland species richness	Proportion of peatland species
BL	16.0	0.04	0.45	1.00	26	0.74
CA10	98.0	0.03	0.23	1.00	22	0.71
CA5	2.0	0.13	0.16	1.00	12	0.57
CC	0.5	0.05	0.29	1.00	15	0.79
CR	40.0	0.10	0.21	1.00	22	0.59
EF	281.0	0.02	0.20	0.75	25	0.69
FL	1013.0	0.03	0.17	0.50	17	0.71
JJ	19.0	0.10	0.46	0.50	22	0.55
JN	183.0	0.07	0.60	0.50	16	0.55
NP	334.0	0.50	0.20	0.75	23	0.74
OT	369.0	0.57	0.17	0.92	13	0.50
PB	72.0	0.20	0.34	0.67	26	0.67
RR	3.0	0.37	0.01	1.00	14	0.70
SF	5.0	0.11	0.44	1.00	16	0.80
SM	88.0	0.16	0.37	0.50	19	0.56
SP	444.0	1.15	0.03	1.00	15	0.71
SV	93.0	0.05	0.29	0.90	18	0.75
TS	46.0	0.02	0.23	1.00	22	0.69
UC	407.0	0.04	0.00	1.00	15	0.75
WP	272.0	0.04	0.07	1.00	26	0.63