# FACTORS AFFECTING AVIAN COMMUNITIES BREEDING ON GOLF COURSES AND GREEN SPACES IN MONTREAL, QUEBEC

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#### DEDICATION

"The inability of ecological theory to predict precisely future population sizes, the rates at which a fauna will collapse following insularization of its habitat, or the response of an ecosystem to a complex series of insults does not necessarily represent failure of the theory. Physicists, after all, cannot predict which of two identical radioactive nuclei will decay first or which of a series of nearly identical missiles launched from the same silo will come closest to the target. The systems modeled by ecological theorists are much more complex – composed of diverse, often little-known organisms interacting with each other and with diverse, complex, usually barely studied physical environments. The problem of balancing precision against generality is much more difficult for ecological theoreticians than it is for theoretical physicists." -- P. R. Ehrlich, p. 315.

# Roughly translated: "Ecology isn't rocket science... It's harder." -- S. R. Carpenter

Thank you to R. Hilborn and D. Ludwig for unearthing P. R. Ehrlich's statement and articulating how challenging ecology can sometimes be, and to S. R. Carpenter for making it into a great sound-bite.

Carpenter, S. 2002. Ecological futures: building an ecology of the long now. Ecology 83: 2069-2083.

Ehrlich, P. R. 1989. Discussion: Ecology and resource management – Is ecological theory any good in practice? Pages 306-318 *in* Perspectives in Ecological Theory. (J. Roughgarden, R.M. May, and S. Levin, Eds.). Princeton University Press, Princeton, NJ.

Hilborn, R. and D. Ludwig. 2002. The limits of applied ecological research. Ecological Applications 3: 550-552.

#### ABSTRACT

Recently, factors influencing avian community composition and nest survival in an increasingly urbanized landscape have received much attention. Despite this research, much is still unknown regarding the capacity of privately owned and managed golf courses to support diverse breeding bird communities and provide safe nesting grounds. My overall objective was to compare avian communities and nest success rates of open-cup nesting passerines on golf courses and green spaces to determine whether golf courses support breeding bird communities as effectively as other types of suburban green space. Breeding bird communities were assessed on six golf courses and six green spaces in Montreal, Quebec, Canada. There was no difference in the mean number of bird species between golf courses and green spaces. However, species composition differed and was most correlated with site size, housing density surrounding the site, and the extent of coniferous tree, grass, and vegetated water cover. Nesting density, survival rates, productivity data and nest-site characteristics of several common open-cup nesting passerines were also compared between a subset of these sites (four golf courses and two green spaces) from 2003-2005. Species-specific differences in nest success rates were found between golf courses and green spaces. Nest survival of upland-nesting species was influenced by year, the type of nesting substrate and its arrangement, and the interaction of these two variables. American Robin (Turdus migratorius) nest success was not influenced by any of the measured variables, whereas Red-winged Blackbird (Agelaius phoeniceus) nest survival showed variation based on nest age and nest concealment. Non-viable eggs of five species were analysed for organochlorine and polychlorinated biphenyl compounds to determine if they contained traces of these historically used chemicals. Thirty-two of the 191 compounds measured were detected in all egg samples, though most compounds were found at trace levels or were below detection limits. P,p'-DDE and the sum of all PCB concentrations dominated the samples. Though PCB levels were higher in Redwinged Blackbird eggs collected on golf courses, reproductive parameters did not differ between golf courses and green spaces. Thus, results suggest that suburban

green spaces, including golf courses, can play a role in maintaining healthy bird populations by following simple management recommendations outlined in these pages.

#### Résumé

Récemment, plusieurs projets de recherche ont étudié les facteurs influençant la composition des communautés aviaires et leur succès de nidification dans le milieu urbain. Malgré ces études, peu est connu sur la diversité des espèces d'oiseaux nichant sur les terrains de golf et sur leur succès de nidification. Mon objectif principal était de comparer les communautés aviaires et les succès de nidification de plusieurs espèces passériformes à nids ouverts sur des terrains de golf et des espaces verts. Le but était de déterminer si les terrains de golf peuvent supporter des communautés d'oiseaux nicheurs comme les autres types d'espaces verts en banlieue. Les communautés d'oiseaux nicheurs et les caractéristiques d'habitats ont été étudiées sur six terrains de golf et six espaces verts à Montréal, Québec, Canada. Il n'y avait pas de différence significative entre le nombre moyen d'espèces se trouvant sur les terrains de golf et sur les espaces verts. Cependant, la composition en espèces était différente, les facteurs expliquant le mieux cette différence étant la grandeur du site, le nombre de maisons autour du site, et les superficies occupées par les conifères, le gazon et les étendues d'eau couvertes de végétation. Les densités de nids, les taux de survie, les données sur la productivité et les caractéristiques des sites de nidification de plusieurs espèces passériformes à nids ouverts ont aussi été comparés entre quelques sites (quatre terrains de golf et deux espaces verts) entre 2003 et 2005. Des différences au niveau du succès de nidification de plusieurs espèces ont été observées entre les terrains de golf et les espaces verts. Le succès de nidification des espèces des terres émergées était influencé par l'année, le type de substrat utilisé pour le site de nidification, son agencement et l'interaction entre ces deux variables. Le succès de nidification du merle d'Amérique (Turdus migratorius) n'était influencé par aucune mesure. Par contre, le succès de nidification du carouge à épaulettes (Agelaius phoeniceus) était influencé par l'âge du nid et son camouflage. La présence de composés organochlorés et de biphényles polychlorés (BPC) a été analysée dans des oeufs non-viables de cinq espèces pour déterminer s'ils contenaient des traces de ces substances chimiques. Trente-deux des 191 composés testés ont été détectés dans tous les échantillons, mais la plupart des composés étaient présents en quantité trace ou en-dessous du seuil de détection. P,p'-DDE et la somme des concentrations de BPC étaient les composés les plus communs. Les variables de reproduction n'étaient pas différentes entre les terrains de golf et les espaces verts, même si les œufs de carouge à épaulettes provenant des terrains de golf avaient des niveaux de BPC plus élevés. Donc, les résultats démontrent que les espaces verts de banlieue, incluant les terrains de golf, peuvent jouer un rôle important dans le maintien des populations d'oiseaux, en suivant les recommandations d'aménagement simples suggérées dans ce document.

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#### PREFACE AND STATEMENT OF ORIGINALITY

This thesis contains six chapters. This is a manuscript-based thesis, so all scientific names, bibliographies and acknowledgements are included for each chapter. The entire manuscript conforms to the style requested by the American Ornithologists' Union publication, The Auk. The first chapter is an extensive literature review and overview of this project's objectives (Chapter 1: General introduction). The second chapter compares breeding bird assemblages between golf courses and green spaces, and links these communities with the habitat features on and surrounding these suburban sites (Chapter 2: Recommendations for design and management of golf courses and green spaces based on surveys of breeding bird communities in Montreal). The third chapter describes a novel nestmonitoring method developed for this project; a low-impact, portable web camera with the capability to archive images and video (Chapter 3: An affordable computerized camera technique for monitoring bird nests). The fourth chapter quantifies nest success rates of nine common open-cup nesting passerines on golf courses and on other suburban green spaces, and compares nest-site characteristics of five of these species between site types. Factors that affect nest survival are identified for all upland-nesting species using these sites to breed, as well as for American Robin (Turdus migratorius) and Red-winged Blackbird (Agelaius phoeniceus). Management recommendations are provided based on these findings (Chapter 4: Factors influencing songbird nest survival on Montreal golf courses and green spaces). The fifth chapter provides evidence that birds breeding in suburban green spaces are contaminated with organochlorine and polychlorinated biphenyl (PCB) chemicals that are no longer in use in North America, but at low levels (Chapter 5: Organochlorine and polychlorinated biphenyl contamination of passerine eggs collected on suburban golf courses and green spaces in Montreal, Quebec). The final chapter consists of my overall conclusions and suggestions concerning topics in need of further research (Chapter 6: Summary, conclusions and final recommendations for management and future studies).

Additional information is provided in three appendices. However, several tables and figures that are specific to a given chapter are presented at the end of that chapter, and are listed under the List of Tables or Figures, wherever appropriate. Appendix I contains copies of the approvals from McGill University's animal use protocols for studies involving animals. Appendix II contains copies of the federal and provincial collection permits required for the collection and analysis of non-viable bird eggs. Appendix III contains copies of the written consent provided by all co-authors and publishers, when required.

In fulfilling the requirements for the degree, *Doctor of Philosophy*, the research must show originality. Though the use of golf courses by wildlife has been investigated in the past, no study has investigated the importance of golf courses for common, open-cup nesting passerines, or has set up the framework to address the potential causes of variations in reproductive rates. This research has already made, through the previous publication of two chapters, and will make, through the anticipated publication of at least two other chapters, the following original contributions to the scientific literature:

- This study is the first to detect differences in Canadian bird communities present on golf courses and other suburban green spaces and to relate their presence to habitat characteristics specific to each land use type.
- This study introduces a novel nest monitoring technology specifically designed for this study, which allows image and video archival as voucher specimens and produces minimal disturbance to open-cup nesting birds due to quick image capture.
- 3. This study is the first to quantify success and predation rates of open-cup nesting passerine species breeding on golf courses and neighbouring green spaces and to identify the factors driving these rates.

- 4. This study is the first to show differences in nest-site characteristics between birds nesting on golf courses and green spaces, and to relate how these characteristics affect nest density and productivity.
- 5. This study is the first to quantify organochlorine and polychlorinated biphenyl contaminant loads on golf courses and other suburban green spaces, and one of the very few to quantify these chemical burdens in passerines in Quebec.

#### **CONTRIBUTIONS OF COAUTHORS**

This thesis contains four manuscripts with co-authors, which have been or will be submitted for publication. I am listed as the first author for all papers as I was responsible for data collection, data preparation and analysis, and writing. D. M. Bird is listed as second author on all papers, as he assisted with the development of this project in his capacity as my thesis supervisor, and this manuscript.

Except for the first chapter (*General introduction*), the first three chapters were co-authored by M.-A. R. Hudson and D. M. Bird. The second chapter, *Recommendations for design and management of golf courses and green spaces based on surveys of breeding bird communities in Montreal*, was published in Landscape and Urban Planning. The third chapter, *An affordable computerized camera technique for monitoring birds' nests*, was published in The Wildlife Society Bulletin. The fourth chapter, *Factors influencing songbird nest survival on Montreal golf courses and green spaces*, will be submitted for publication to Avian Conservation and Ecology. The fifth chapter, *Organochlorine and polychlorinated biphenyl contamination of passerine eggs collected on suburban golf courses and green spaces in Montreal, Quebec*, was co-authored by M.-A. R. Hudson, K. J. Fernie, D. Burniston, C. Marvin and D. M. Bird, and will be submitted for publication to Environmental Pollution. K. J. Fernie assisted with analysis and editing, while C. Marvin and D. Burniston prepared samples for analysis and assisted with logistics.

## CHAPTER 1

### GENERAL INTRODUCTION

#### URBANIZATION

Defined as a concentrated human presence and its associated effects (Chace and Walsh 2005), urbanization is one of the driving forces of habitat loss and fragmentation. Increasing urbanization leads to an increase in avian biomass and density coupled with a decrease in species richness (Marzluff 1997, Chace and Walsh 2005). These avian communities, seldom representative of native avifauna, are dominated by adaptable species with low area requirements and a high tolerance for human activity. The diversity of birds in urban areas is affected by the level of disturbance, the age of the development, the type of housing, and the degree of urbanization (Chace and Walsh 2005). However, urbanization tends to favour omnivores, granivores, aerial insectivores, and ground-foragers while hindering shrub and mid- to upper-level foliage gleaners, bark gleaners and drillers, and this is consistent over most regions examined (Lancaster and Rees, 1979, Beissinger and Osborne 1982, Bezzel 1985).

These changes in community are partly due to changes in vegetation. Vegetation in an urbanized environment is generally more fragmented, has a greater amount of ground-level cover, and a reduced amount of mid- and upper-level cover than a natural environment (Beissinger and Osborne 1982). Native plant species are often replaced by exotics, and dead or downed trees are removed, thereby increasing nest-site competition with cavity nesting, non-native species such as European Starlings (*Sturnus vulgaris*) and House Sparrows (*Passer domesticus*; Marzluff 1997). These changes may negatively influence food availability by removing foraging sites and reducing the number of insects present in the environment, as exotic plant species support fewer insects than native species (Beissinger and Osborne 1982). Urbanization can also increase the amount of food available to birds, either directly through bird feeders, or indirectly through spilled waste grain, landfills and productive urban lawns (Lancaster and Rees 1979).

This increased food supply may also benefit brood parasites such as the Brown-headed Cowbird (*Molothrus ater*) as well as omnivorous nest predators such as Blue Jays (*Cyanocitta cristata*), American Crows (*Corvus* 

brachyrhynchos), and Eastern Gray Squirrels (Sciurus carolinensis). Large predators are generally absent from urbanized areas, reducing predation pressure on medium-sized predators such as domestic dogs and cats as well as Striped Skunks (Mephitis mephitis) and Raccoons (Procyon lotor), leading to increased predation pressure on prey species (Soulé et al. 1988, Woods et al. 2003). Also, avian predators such as Sharp-shinned Hawks (Accipiter striatus) and Cooper's Hawks (A. cooperii) may concentrate their activities at bird feeders due to an artificially enhanced abundance of prey (Dunn and Tessaglia 1994, Marzluff 1997). Though results vary geographically and with larger landscape context (e.g. agricultural vs. largely forested regions), it appears that the number of predators increases slowly along a gradient of increasing housing density, but predation rates do not change (Haskell et al. 2001). However, predator species are not all affected the same way, some increasing in abundance with urbanization, some decreasing, and some not showing any trend at all (Haskell et al. 2001, Sorace and Gustin 2009). Despite the explosion of research into the effects of urbanization over the past two decades, there is very little information regarding the survival or predation rates on urban green spaces embedded in different landscapes (e.g. industrial complexes, residential developments, agriculture, or forest). Research focusing on how wildlife (both predator and prey species) at individual and population levels is affected by top-down or bottom-up processes driving urban avoidance or attraction is needed as well (Rodewald and Shustack 2008).

Instead of focusing on collecting data related to these specific processes and how they affect different species, recent efforts have attempted to summarize the effects of urbanization on avian productivity (Chamberlain et al. 2009) and richness (Pidgeon et al. 2007). However, these efforts have been hampered by the incredible variation in study results due to differences in geography and landscape. In addition, urban environments are far from stable. Morneau et al. (1999) showed how avian assemblages using urban parks in Montreal changed substantially over a period of only 15 years, likely due to an increased availability of bird feeders and the effects of southern species increasing their ranges northward. Generalizations concerning the effects of increased development will likely also be difficult, as avian guilds appear to be affected differently. For example, cavity-nesting species and short-distance migrants are more susceptible to regional changes in landscape than other guilds (Pidgeon et al. 2007).

Despite all of these sources of variation, researchers have found that birds in urban areas tend to have earlier lay dates, smaller clutches and lighter nestlings (reviewed in Chamberlain et al. 2009). They also produce fewer nestlings per nesting attempt (Chamberlain et al. 2009). Anthropogenic food availability may increase adults' pre-laying fitness, allowing for earlier laying. However, the food generally offered in urban environments (e.g. bird seed and garbage) is generally not suitable for nestlings, leading to decreased nestling weights despite high food availability. However, these generalities may not hold for other species. Chamberlain et al. (2009) warn that extrapolating these trends to other species that are not generally found in urban areas is unwise, as species studied in urban areas are clearly those that are capable of tolerating urban conditions. For example, Rodewald and Shustack (2008) explored the effects of urbanization on the Acadian Flycatcher (*Empidonax virescens*), a forest-breeding Neotropical migrant. They found that though birds nesting in areas surrounded by more development did not differ in adult or daily nest survival rates, they initiated fewer nests throughout the season and exhibited later initiation dates, higher turnover rates and higher levels of brood parasitism than in rural areas. Thus, depending on which parameter or level is chosen (i.e. nest success, site occupancy, or seasonal reproduction; individual or population), the effects of urbanization may not always be immediately apparent.

Overall, it is clear that urbanization does exert effects and they can be direct or indirect: increased loss and degradation of habitat, elevated proportion of exotic flora and fauna, modified predator communities, warmed climate (in urban centres), increased human disturbance, increased access to supplemental food, increased possibility of disease transmission, and increased competition (Chace and Walsh 2005). These effects are amplified as urbanization expands, increasing habitat loss and fragmentation.

#### FRAGMENTATION

Fragmentation of the landscape generally produces a series of remnant vegetation patches surrounded by a matrix of differing vegetation or land use (Saunders et al. 1991), and is more complex than the simple loss of habitat. Landscapes with highly fragmented habitat not only experience a reduction of species or individuals within a population through habitat loss, but also through the additive effects of fragment size (Tilghman 1987) and degree of fragment isolation (Andrén 1994). The effect on flora and fauna is also influenced by a myriad of other factors (reviewed by Saunders et al. 1991): type of fragmentation (i.e. induced or inherent; Paton 1994), degree of fragmentation (Hartley and Hunter Jr. 1998), shape (Saunders et al. 1991), spatial arrangement and surrounding landscape (Friesen et al. 1995, Tewksbury et al. 1998, Doherty Jr. and Grubb Jr. 2002). Time may also play a large role in how species respond, possibly accounting for largely inconsistent results of recent studies on fragmentation (reviewed by Debinski and Holt 2000).

Recent work has shown that the effects of fragmentation are more complex than initially described (Dunford and Freemark 2004, Tewksbury et al. 2006). It was once thought that increasing the degree of fragmentation on a landscape scale, decreasing patch size and increasing the amount of edge habitat all led to increased nest predation and decreased species richness, but we now understand that not only do the effects depend on the type of habitat in which sites are embedded (Tewksbury et al. 2006), but our ability to detect them is influenced by the scale at which fragmentation is examined and the duration of the study (Stephens et al. 2003). For example, Dunford and Freemark (2004) found that agriculture and urbanization did not consistently affect forest bird diversity in either a positive or negative way across study sites. Rather, avian communities responded differently depending on the spatial scale, and groups of species were affected differently. Indeed, fragmentation research on a landscape scale has provided one of the most compelling examples of how the landscape matrix -- the type of land use in which a study site is embedded at differing levels -- can modify avian communities and their reproductive success (Rodewald 2003).

*Sources and sinks.* -- Studies focusing on fragmentation have often relied on Pulliam's (1988) source-sink theory as a research framework. The theory suggests that, for many populations, a large proportion of individuals reside in unstable 'sink' habitats, where mortality is greater than reproduction. These sinks are continuously and actively supplied with immigrants from a source habitat, where reproduction exceeds mortality (Pulliam 1988).

In recent years, researchers have used the theory to quantify the amount of forest necessary to maintain stable populations of forest-interior species in increasingly fragmented habitats (Temple and Cary 1988, Donovan et al. 1995, Burke and Nol 2000). However, understanding source-sink population dynamics requires the following data from marked populations to create modified BIDE (Birth, Immigration, Death, Emigration) models: season-long female productivity, immigration and emigration rates, adult and juvenile mortality, and dispersal distances (Howe and Davis 1991, Robinson et al. 1995). Representing a trade-off between intensively studying local populations and extensively studying large, widespread populations, many authors have had to base their calculations on assumptions (e.g. fecundity is equal across age classes, across years, and is constant throughout the breeding season) and have used adult and juvenile survival values obtained from the literature (Donovan et al. 1995, Burke and Nol 2000). Despite this problem, the persistence of non-reproducing populations in small fragments provides some of the best evidence to date that source-sink dynamics operate on a regional scale, as opposed to a local scale (Robinson et al. 1997).

Though emphasis has been on source habitat conservation (Temple and Cary 1988, Burke and Nol 2000), sinks are also important, as they may produce individuals that will enter the breeding population at a later date, ensuring a more diverse gene pool and positively affecting the size and longevity of the overall population (Pulliam 1988, Howe and Davis 1991, Villard 1998, Burke and Nol 2000). However, sinks can easily be confused with ecological traps (i.e. habitats where attractiveness is uncoupled from quality; Kristan III 2003). For example,

Gates and Gysel (1978) postulated that edge vegetation may concentrate both nest predators and breeding birds, leading to higher nest predation rates. The difference lies only in the ability of the individual to accurately assess habitat quality, and while reproduction may be high, the number of juveniles able to join the breeding population later on may be greatly reduced.

*Edge effects.* -- Fragmentation leads to the creation of an edge, or the abrupt or transitioned junction of two different landscapes (Yahner 1988). The concept of the edge effect was created, in part, around Leopold's (1933) "law of interspersion", which predicts that increasing edge habitat results in higher population densities (Paton 1994, Lahti 2001). Several authors have since pointed out that this concept only pertains to a small number of generalist species (Yahner 1988), that suspected nest predators and brood parasites also increase at edges (Gates and Gysel 1978), and that excessive edge creation harms species that require large areas of intact forest (Yahner 1988, Paton 1994, Lahti 2001).

Though edge effects have been studied extensively over the years, the results are inconsistent due to incompatible methodologies, differences in predator communities, poor study designs (including, but not limited to pseudoreplication, insufficient sample size, and arbitrary designation of an edge), and confusion surrounding terms such as edge-avoidance and area-sensitive (Yahner 1988, Paton 1994, Murcia 1995, Villard 1998). Lahti's (2001) most recent review of 55 studies suggests that edge effects on nest predation are not a valid generalization. However, the type of edge involved (nature of adjoining communities) and the extent of fragmentation of the surrounding landscape may account for the variation in the detection of edge effects.

#### VALUE OF SMALL RESERVES

Much debate has revolved around the application of the theory of island biogeography to fragmented habitat (Simberloff and Abele 1976, Higgs and Usher 1980, Tilghman 1987). The theory predicts that the smaller and more isolated the island, the fewer species it supports (MacArthur and Wilson 1963). The theory of metapopulation dynamics, which states that a species is more likely to persist in an environment if the distance between neighbouring patches is small enough for movement to occur between them, has overtaken MacArthur and Wilson's theory as the framework for conservation work (Shafer 1995). Although these relationships may be useful when considering species diversity and future trends in species number, their applications to 'habitat fragments' as opposed to true islands may be misplaced. One of the concepts behind the theory of island biogeography -- the species-area relationship -- may be more relevant. The number of studies indicating that avian species richness increases as a function of woodlot size is substantial (Shafer 1997), but not universal (Friesen et al. 1995, Debinski and Holt 2000). Friesen et al. (1995) measured Neotropical migrant species diversity within forest fragments (3-50 ha) and found that while diversity consistently increased as forest size increased, it consistently decreased as the amount of adjacent development (number of houses) increased, regardless of forest size. Thus, a 4-ha forest without any surrounding housing (i.e. surrounded by forest) had more diverse Neotropical communities than a 25-ha forest surrounded by development.

Conclusions derived from the theory of island biogeography have led many to advocate the establishment of large reserves. However, in a rapidly urbanizing environment, the choice between a large or small reserve rarely exists. Though larger sites may support a richer number of avian species, this is due to the larger number of habitats contained within and not simply a matter of size (Donnelly and Marzluff 2004). The academic SLOSS (Single Large Or Several Small) debate (Diamond 1976, Simberloff and Abele 1976, 1982, Terborgh 1976, Whitcomb et al. 1976) revolves around two observations: larger sites support larger and more stable populations, but partially isolated populations on networks of small sites buffer species from extinction (Schwartz and van Mantgem 1997). Thus, while larger habitat fragments tend to have greater habitat diversity than smaller ones, a collection of smaller fragments may have a greater number of habitat types, and thus species, simply because they cover a larger total area than one large fragment (Saunders et al. 1991, Schwartz and van Mantgem 1997). Blake and Karr (1984) tested this prediction in Illinois using 12 forest patches ranging in size from 1.8-600 ha. They found that two smaller sites were more likely to contain a greater number of species than one large site. However, while short-distance migrants and residents were well supported with this configuration, Neotropical migrants and forest-specialists were not. Thus, small sites can fill several important roles, especially in urban landscapes: i) preservation of taxa that can survive in small areas; ii) preservation of unique microhabitats; iii) use of sites as stepping-stones between larger reserves; iv) improvement of the aesthetics of surrounding landscape; and v) provision of educational and recreational opportunities (Diamond 1976). It is generally recognized however, that large sites provide habitat for species with area requirements that cannot be met in small sites, and that the relevance of the SLOSS concept depends on the organism(s) under consideration (Blake and Karr 1984, Robinson et al. 1997, Schwartz and van Mantgem 1997).

#### GOLF COURSES AND THE ENVIRONMENT

Semi-natural areas can contribute to the conservation of biological diversity (Shafer 1997, Tanner and Gange 2005). Golf courses are economically self-sufficient with well-equipped and well-organized maintenance personnel, eliminating reliance on monies slated for conservation work (Terman 1997). Given their ubiquity in the suburban environment, it is surprising that until recently, there were few studies examining the potential importance of golf courses to wildlife (Maffei 1978, Green and Marshall 1987, Brennan 1992, Moul and Elliott 1992, Terman 1997). There are now more than 31,500 existing golf courses worldwide (Tanner and Gange 2005), with out-of-play areas generally covering 60% of the average golf course (Green and Marshall 1987). This represents a vast amount of potential wildlife habitat available for management and improvement. On a smaller scale, given that golf courses are fragmented by design, each fragment within the course can be viewed as a habitat island, and concepts derived from the theory of metapopulation dynamics such as the

importance of connectivity and inter-patch distance are relevant and can be applied to meet the needs of resident species.

Most golf courses are no longer the "green deserts" of the past. A growing number of new golf courses are now being constructed over abandoned landfills, parking lots and otherwise unproductive sites. As of 1997, over 60 golf courses had been built on landfills, reclaiming and enhancing native habitat (Devine 1999). In Chiba, Japan, a stricter regulation now specifies that forest must cover 50% of a new development's area, making new golf courses much greener than those built prior to the regulation (Yasuda and Koike 2006). It has converted the typical golf course design of a wide fairway with isolated trees into one more conducive to wildlife conservation: a narrower fairway bordered by forest (Yasuda and Koike 2006). Indeed, a number of programs now exist to promote golf course naturalization, and the ever-increasing use of integrated pest management is reducing pesticide use. Examples include the Audubon Cooperative Sanctuary Program for golf courses (ACSP) and the Royal Canadian Golf Association's (RCGA) set of environmental guidelines for Canadian golf clubs (Yamada 1995). The ACSP encourages naturalization by issuing management certificates. However, only 13.5% of all the clubs in the U.S. belong to the program and less than 2% have been fully certified (Dotti 2002). While the number of participating courses is small, this movement has succeeded in fostering a gradual change in thinking that has led a growing number of golf courses to begin naturalizing their surroundings whether they are registered in the program or not. This allows for largely undisturbed areas to flourish and thus fosters healthier, more ecologically sound courses (Gillihan 2000). In fact, several authors have documented the use of golf courses by endangered or threatened flora and fauna, citing their importance as stable habitats necessary for the continued existence of these species (Green and Marshall 1987, Jodice and Humphrey 1992, Sadlon 1994, Carter III and Kocher 1995, Lavis 1995).

*Birds and golf courses.* -- A growing number of studies have suggested that golf courses have the potential to serve as wildlife habitat (Green and Marshall 1987,
Moul and Elliott 1992, Blair 1996, Terman 1997, Hostetler and Knowles-Yanez 2003, Yasuda and Koike 2006, Hodgkison et al. 2007a). More specifically, golf courses support variable numbers of native and non-native species, as well as limited numbers of species of conservation concern (LeClerc and Cristol 2005, Porter et al. 2005, Rodewald et al. 2005, Hodgkison et al. 2007b).

Over the years, researchers have been able to provide solid and logical management recommendations targeting urban areas in an effort to make them more attractive and supportive to wildlife (e.g. Savard et al. 2000). As the drivers of urban avian assemblages are now beginning to be fairly well understood (Jokimäki 1999, Savard et al. 2000, Rodewald 2003), the focus has shifted to examining whether birds on golf courses respond in the same way as birds in other urban/suburban landscapes, and how these avian communities respond to different habitat characteristics (e.g. local, patch and landscape; LeClerc and Cristol 2005, Porter et al. 2005, Hodgkison et al. 2007a).

It appears that land use itself is of little importance (Hostetler and Knowles-Yanez 2003). Rather, specific habitat features such as vegetation structure and complexity on-site are important for preserving avian species diversity in forested landscapes. In Virginia, LeClerc and Cristol (2005) found that the proportion of forest within 1.5 km of a golf course's center was the best predictor of a site's ability to support species of conservation. The importance of on-site forest cover has received support from all over the world (Jones et al. 2005, Yasuda and Koike 2006, Sorace and Visentin 2007). However, Porter et al. (2005) concluded that on-site variables were less important than the surrounding landscape (e.g. natural vs. residential buffers) as predictors of bird diversity in Ohio. Yasuda and Koike's (2006) work in Japan also highlighted the importance of surrounding land use to golf course flora and fauna; golf courses in urbanized areas had avian species and non-native plant species more common to urban areas, whereas those situated in rural landscapes had many native forest plant species. A study by Hodgkison et al. (2007a) in Australia found support for both on and off-site habitat characteristics. However, though it is becoming increasingly clear what management measures must be taken for a suburban golf

course to support diverse breeding bird communities, it is still unknown whether we should be encouraging birds to nest on golf courses at all.

Dale (2004) was one of the first to explore the potential impacts of golf courses on productivity by examining the population dynamics of endangered Ortolan Buntings (*Emberiza hortulana*). Males singing in the interior of a golf course remained unpaired and eventually emigrated off the course, where they then paired successfully (Dale 2004). No differences were found between the males' pairing success at the periphery of the course and off the golf course. Dale's study, despite its very small sample size (n = 4 males in the interior of the course), provided an important first step in understanding the dynamics of wildlife on golf courses. To my knowledge, only two studies focusing on passerine reproduction on golf courses have been published, both of them concentrating on the Eastern Bluebird (*Sialia sialis*), a cavity-nester (LeClerc et al. 2005, Stanback and Seifert 2005). There is a real need for data pertaining to the ecology and productivity of common, open-cup nesting birds inhabiting golf courses on both the species and the community level.

## FACTORS AFFECTING NEST SUCCESS IN URBAN AREAS

Reproductive performance is a critical index of population health. Simple measures such as abundance or density are not sufficient to determine the quality of a given habitat, since the presence of a species is no guarantee that it is not being forced to use a lesser quality site due to competition from others (Van Horne 1983). Individuals must be using the site for successful reproduction and subsequent recruitment into the population (Saunders et al. 1991). However, assessing the reproductive performance of wild breeding birds is not a simple task, as many variables may influence the outcome of each nesting attempt. For example, strong weather and temperature fluctuations can cause direct nest mortality, reduce egg hatchability and indirectly affect nest success by influencing food abundance (Martin 1987). Whether birds nest early or late in the breeding season affects the outcome as well. Early nesters tend to have better success and

larger clutches than later nesters, although there appears to be some variation (Gates and Gysel 1978, Grant et al. 2005).

*Nest-site selection.* -- Evolutionary theory posits that birds should select nest sites that confer the highest probability of success (Martin 1993b). Numerous studies have tested for differences between nest sites and random unused sites, and between successful and unsuccessful nests in an effort to show that the species in question is choosing its nest site based on certain characteristics that increase reproductive success (e.g. Easton and Martin 2002, Smith et al. 2007). However, it is much more informative to compare used nest sites to available but unused nest sites within the bird's territory, as sites that are unused but unavailable to the bird (e.g. random sites) may not pinpoint the differences that drive nest-site selection (Jones 2001). Once this is taken into consideration, two major factors can affect our ability to detect nest-site selection: the characteristics of the species in question (e.g. habitat generalist or specialist), and the scale at which selection is examined (e.g. nest site, nest patch, home range or territory; Jones and Robertson 2001).

The concept of sinks or ecological traps (*sensu* Gates and Gysel 1978) ties into nest-site selection if birds choose sites based on vegetative characteristics that satisfy their nest-site selection requirements, but fail to confer increased nest success. This maladaptive behaviour was demonstrated with Rose-breasted Grosbeaks (*Pheucticus ludovicianus*) in Ontario (Smith et al. 2007), as birds actively chose vegetation characteristics that varied according to silvicultural treatment but had negative effects on reproduction.

*Predation.* -- Nest predation has consistently been identified as the largest source of mortality among nesting birds (Ricklefs 1969, Wilcove 1985, Martin 1993a, b). Predation is thought to increase with predator abundance, as it seems largely based on chance encounters with a rare but profitable prey item (e.g. nest with eggs or young; Angelstam 1986, Cooper et al. 1999, Driscoll and Donovan 2004).

However, some have found indications of density-dependent predation based on higher numbers of nests near edges (Roos 2002, Driscoll and Donovan 2004).

One of the leading cited causes of avian species decline in fragmented landscapes is a higher predation rate due to increases in generalist predators near edges (Gates and Gysel 1978, Yahner 1988, Paton 1994). However, in 76% of the studies reviewed by Chalfoun et al. (2002), predator abundance was not significantly higher at edges. Gates and Gysel's (1978) ecological trap hypothesis initiated much research: 50 empirical tests have been published in the 20 years following their publication (Lahti 2001). However, findings have been largely contradictory, indicating increased predation rates near edges (Gates and Gysel 1978, Andrén and Angelstam 1988, Paton 1994, Major et al. 1999), unaffected predation rates near edges (Rogers 1994, Tewksbury et al. 1998), lowered predation rates near edges (Rudnicky et al. 1993), or edge effects present for some species but not for others (Burke and Nol 2000). These conflicting results have made generalizations almost impossible, with reviewers drawing different conclusions (Paton 1994, Lahti 2001, Chalfoun et al. 2002). This may change, however, as studies have begun emphasizing the importance of structure and composition of the surrounding landscape on predation rates (Heske et al. 2001, Tewksbury et al. 2006). General patterns are beginning to emerge, with increased predator abundance and activity at edges more prevalent in: i) studies operating on a landscape level rather than on a small scale; ii) in predominantly agricultural rather than forested landscapes; iii) in the eastern and midwestern U.S. rather than the western U.S. (regional differences in predator communities); and iv) for avian predators rather than for mammalian predators (Chalfoun et al. 2002). Very few studies have examined snake predation though, and most of the work to date has documented nest predation by one species and then extrapolated it to an entire group. Thus, the last point is perhaps misleading, as the effect of fragmentation on predation appears to be taxon-specific (Chalfoun et al. 2002).

The major issue in the study of nest predation is that much of today's knowledge is based on work using artificial nests (Wilcove 1985, Paton 1994, Major and Kendal 1996). While the use of artificial nests allows for strong

statistical designs and may provide a measure of relative predation pressure by indicating the proportion of nests taken (Rangen et al. 2000), their results must be interpreted with caution. Their use is based on the assumption that the effects of predation are similar on both artificial and natural nests, although this has not always been supported (Major and Kendal 1996, Zanette 2002, Berry and Lill 2003). Problems with artificial nests include the following: the use of galliform eggs restricts the ability of smaller predators such as rodents to depredate nests (Haskell 1995, Rangen et al. 2000); plasticine eggs have a distinctive smell that may attract small rodents to nests that they would not ordinarily depredate (but see Bayne and Hobson 1999, Rangen et al. 2000); human scent transferred to the nests may attract some predators while deterring others (Berry and Lill 2003); artificial nesting material may not be representative of natural nests (Berry and Lill 2003); some nest predators prefer nestlings and will not depredate artificial nests (Pärt and Wretenberg 2002); artificial nests are generally equally spaced at higher than natural densities, and may be placed more conspicuously than natural nests (Zanette 2002); and artificial nests suffer from an absence of attending adults or nestlings, making them vulnerable to attack from predators that may not be true nest predators (Cresswell 1997). Many have acknowledged these problems, but have continued to use this method with untested mitigation measures in place (e.g. wearing rubber boots and gloves to reduce human scent, and using various sizes of artificial eggs). Some authors have argued that this is not sufficient, suggesting that "artificial nest experiments remain an untested technique despite being used in over 70 studies, many of which are documented in high-ranking ecological journals" (Major and Kendal 1996) and that "researchers should refrain from drawing conclusions about the importance of nest predation based on artificial nest studies. Studying natural nests is more time-consuming and labour-intensive, but information will be more reliable" (Zanette 2002). The reliability of artificial nests in tandem with real nests has since been examined (Blair 2004). Artificial nests appear to suffer much higher predation rates than real nests, but they still reveal the same trend that real nests do (e.g. reduced predation pressure with increased urbanization). This suggests that, despite the inflated rates, they can be used as surrogates to compare between treatments or along gradients in lieu of real nests (Blair 2004).

Until recently, many studies examined the impact of edges on predation rates on a small scale, without examining the characteristics of the surrounding landscape (e.g. Rogers 1994, Hanski et al. 1996). The abundance of predators or brood parasites may be more influenced by large-scale landscape features rather than by smaller site-based characteristics (Tewksbury et al. 1998, Huhta et al. 2004). These large-scale patterns may then, in turn, influence predator assemblages on a smaller scale. Not only is scale important, but the definition of an edge (e.g. does the "edge" represent a break in continuous vegetation or a change in vegetation type; agriculture vs. forest) and how it is measured are critical when determining the possible effects on avian reproduction (Yahner 1988). Warnings concerning the effect of artificial nests and of scale appear to be slowly taking effect, as edge effects are now being examined using natural nests and nest boxes, and the landscape within which the study site is embedded has been incorporated into analyses. For example, Driscoll and Donovan (2004) tested and found support for the hypothesis that edge effects on nesting Wood Thrush (Hylocichla mustelina) are stronger in fragmented than contiguous landscapes. Huhta et al. (2004) found indications of higher nest predation rates on Eurasian Treecreeper (Certhia familiaris) nest boxes in higher fragmented forest mosaics than in less fragmented mosaics.

Though it has been suggested that avian species are important nest predators in fragmented and urban habitats (Nour et al. 1993), this cannot be confirmed without either direct observation from blinds or alternatively, by the use of video cameras. The former is time-consuming, costly and difficult to achieve nocturnally. As for video cameras, some problems have been documented with non-camouflaged artificial structures near nests (Herranz et al. 2002, but see Thompson III and Burhans 2003). Some have identified nest predators based on nest remains, but this method is not sound for a variety of reasons (reviewed in Larivière 1999, Williams and Wood 2002, Thompson III and Burhans 2003). An alternative is to place a plasticine or clay egg into the nest in order to identify a potential predator by tooth or beak imprint. This has allowed the identification of artificial nest predators (Major 1991, Nour et al. 1993, Miller and Hobbs 2000). However, placing plasticine eggs in active nests may attract predators (see above) or may induce abandonment or rejection of the foreign egg, as recognition of foreign objects is known for several species (Payne 1977). Thus, without continuous video monitoring, which can possibly bias results, the only measure that can be reliably calculated is the net effect of all predators, the predation rate.

*Nest concealment.* -- Many studies have examined the relationship between nest concealment, brood parasitism and nest predation, with varying results (e.g. Martin 1993b, Filliater et al. 1994, Cresswell 1997, Burhans and Thompson III 2001). Less than half of 26 studies examining the effects of concealment found a significant influence on predation, parasitism or nest-site selection, and only 7 of 23 nest predation studies found that concealment was an important variable (Burhans and Thompson III 1998). These relationships may be difficult to fully comprehend as predation rates may vary according to habitat type, composition of the predator community and its density, composition of the prey community and its density, and various defensive behaviours exhibited by prey species (Cresswell 1997, Burhans and Thompson III 1998, Flaspohler et al. 2000). For example, nest concealment may be important for prey species mainly depredated by avian species, which hunt by sight (Sugden and Beyersbergen 1986, Clark and Nudds 1991, Colwell 1992). If snakes or mammals are the main nest predators however, concealment may matter little as they tend to use olfaction and/or parental or nestling activity to detect nests (Rangen et al. 1999). For sites with diverse predator communities, nest-site selection based on increased nest concealment may have little effect, as predictably safe nest sites may not exist (Filliater et al. 1994).

*Food limitation.* -- Food limitation can affect breeding birds at all stages of the nesting cycle. Without adequate nourishment, a bird will not likely be able to physically enter into breeding condition. If parents do breed during a food

shortage, they must spend more time foraging, thus decreasing nest attendance, which may negatively affect nestling survival (Martin 1987). Decreased food abundance first results in reduced egg size and quality and then in reduced clutch size, since the latter represents a larger change in reproductive potential than the former (Martin 1987). Limited food abundance within a breeding habitat can also restrict nestling growth rates (Quinney et al. 1986). Finally, shortage of food may lengthen the interval between nesting attempts, reducing the success of a second brood. This is based on the assumption that a second nest attempt is even possible, as the parent may no longer be in breeding condition (Martin 1987).

*Parasitism.* -- The Brown-headed Cowbird (*Molothrus ater*) is an obligate, generalist brood parasite, and its population, as well as its rate of parasitism of forest songbirds in eastern North America, has increased substantially since the early 1900s (Brittingham and Temple 1983, but see Hahn and Hatfield 1995). Largely tied to open habitats (Payne 1977, Brittingham and Temple 1983), it can reduce the reproductive success of its host by: i) removing a host egg before laying its own; ii) puncturing a host's egg with either its bill or claws; iii) cracking a host's egg by laying one of its thicker-shelled eggs onto a thinner-shelled host egg; iv) lowering hatching success by inducing host abandonment; v) lowering host nestling survival by crowding, trampling and competing for parental care, especially in smaller hosts; and vi) reducing the host's seasonal reproductive output, as some hosts may build a new nest on top of the existing nest, suffocating both the parasitic egg as well as their own eggs (Payne 1977).

Better nest concealment may have little effect on parasitism rates, as cowbirds, like some nest predators, appear to use host behaviour to locate nests (Payne 1977, Hahn and Hatfield 1995, Clotfelter 1998). Brood parasites usually find host nests by cryptically watching nest-building activities, but may also actively search habitats where hosts often nest, and may cue in on the intensity of the host's alarm calls as it draws nearer to the nest (Norman and Robertson 1975, Payne 1977, Strausberger 1998). While there is some evidence that cowbirds exhibit increased nest-searching behaviour near edges (Gates and Gysel 1978), this has not been supported elsewhere (Hahn and Hatfield 1995). Parasitism rates vary substantially across regions, landscapes and urbanization gradients, suggesting that cowbirds exhibit regional differences in host and habitat use (Hahn and Hatfield 1995, Tewksbury et al. 2006).

Disturbance. -- Urbanization is intrinsically linked to increased levels of disturbance, potentially modifying avian foraging and nesting behaviour (Marzluff 1997, Chace and Walsh 2005). Disturbance related to urbanization comes in many forms, which modifies its effect on bird abundance, diversity and breeding. Industrial operations and roads (Forman and Alexander 1998, Canaday and Rivadeneyra 2001), as well as dog-walking with on- and off-trail pedestrian traffic (Miller et al. 2001, Banks and Bryant 2007), are all considered sources of disturbance, and can all exert different effects based on how they are perceived by birds (reviewed in Boyle and Samson 1995). There is much appeal in the notion that urban or suburban green spaces can simultaneously provide recreational and conservation opportunities (see VALUE OF SMALL RESERVES). However, whether this is actually of benefit to the avian species using these fragments remains uncertain (Miller and Hobbs 2000). Recent work with artificial nests placed along recreational trails suggests that in some cases, birds nesting in close proximity to trails or areas of human disturbance may benefit from reduced predation rates (Miller and Hobbs 2000). This pattern was also seen in Blackbird (*Turdus merula*) nests: those placed close to busy paths suffered from greater abandonment rates but showed reduced predation rates (Osbourne and Osbourne 1980). Avian tolerance to human disturbance varies between and within species, and differences are often context-specific. Time during the breeding season, group size, body size, conspicuousness, height of activity (ground vs. canopy), and surrounding vegetation may affect the level of avian tolerance (Gutzwiller et al. 1998), leading to large variations in results (e.g. van der Zande et al. 1984, Fernández and Azkona 1993, Gutzwiller et al. 1994, Riffell et al. 1996).

The effect of investigator disturbance has been studied to a large extent (reviewed in Götmark 1992, and Mayer-Gross et al. 1997), with results again varying with species. Götmark's (1992) review produced the following: 33 (49%) of 68 papers reported observer effects, while 35 (51%) reported none. In general, species in different orders tend to respond differently to disturbance, with Charadriiformes more susceptible to human disturbance than Passeriformes (Götmark 1992) and colonial nesters more affected than solitary nesters (Mayer-Gross et al. 1997). Reasons for reduced reproductive success have been studied to a lesser extent (Mayer-Gross et al. 1997), with most studies speculating as to the cause (e.g. increased predation, increased desertion, exposure to extreme temperatures, trampling and reduced parental care).

*Pesticides.* -- If not killed outright by a lethal amount of pesticide (e.g. Andrews and Glowa 1985, Kendall et al. 1992), a bird's reproductive success can be affected at all stages of the nesting cycle through sub-lethal pesticide exposure. However, the effects vary depending on the type of pesticide (e.g. insecticides, polychlorinated biphenyls, herbicides or fungicides) and the species involved. Though resulting effects on avian reproductive success are not always evident (Powell 1984, Martin et al. 2000, Gill et al. 2003), the following have been documented: i) increased nest desertion (Busby et al. 1990); ii) deficiencies in eggshell formation (Cooke 1973); iii) reduced clutch size (Fry 1995); iv) reduced hatching success (Fry 1995); v) reduced parental care of nestlings leading to impaired nestling growth rates (Grue et al. 1982); vi) increased rates of deformities in young (Fry 1995); and vii) reduced number of fledged young (Busby et al. 1990).

Indirect effects of pesticides include the reduction of arthropod abundance, which represents a reduction in avian food supply. This may increase parental foraging time and distance (Décarie et al. 1993, Martin et al. 2000), or increase intra-pair competition which may lead to reproductive failure of lesser-quality pairs (Bouvier et al. 2005). However, evidence concerning the effects of reduced food supply as a direct result of pesticide use is lacking, with studies generally unable to detect differences in nest success and nestling growth between passerines nesting on treated and untreated fields and forests (e.g. Powell 1984, Martin et al. 1998 and 2000, Marshall et al. 2002). There is some evidence of reduced seasonal productivity, as Marshall et al. (2002) found that nests were initiated 3-5 days later in treated areas, effectively shortening the breeding season for those pairs.

When the varied and negative effects of pesticides on wildlife became apparent, concern mounted regarding high pesticide use on golf courses, which until recently was much higher than in agricultural areas (Knopper and Lean 2004). Indeed, reports of dead or dying birds found on golf courses after the application of organophosphates are a cause for concern (e.g. Zinkl et al. 1978, Stone and Koch 1982, Kendall et al. 1992). However, aside from these reports and a handful of other studies examining a variety of issues revolving around pesticide use on golf courses and its effect on water and wildlife (e.g. Wong et al. 1998, Cohen et al. 1999, Knopper and Mineau 2004, Metcalfe et al. 2008), and measures of avian response to organophosphate spraying on golf courses (e.g. Brewer et al. 1988, Frank et al. 1991, Kendall et al. 1993, Rainwater et al. 1995), very little work has been published on the impact of golf course chemicals on wildlife and the environment.

# CONCLUSION AND RESEARCH OBJECTIVES

The multitude of studies cited here make it clear that wildlife living and breeding in urbanized or otherwise modified environments are influenced by many factors, be it negatively or positively. The majority of these studies have also indicated that we have the capacity to modify these factors for the benefit of both wildlife and humans, and they have provided recommendations that should allow for improvement (e.g. increase vegetation volume and diversity, establish extensive greenbelts at small and large scales within city limits; Savard et al. 2000, Marzluff et al. 2001 and references therein). The crucial difficulty is to bridge the gap between publication and implementation, and extend these recommendations to as many areas and land uses as possible by not restricting their application to city-owned and -managed parks.

One of the most compelling results emerging from past decades of research focusing on the factors that drive urban ecology is that they operate at nested scales, with smaller communities embedded within larger ones (Hostetler 2001). Thus, management must be planned for each of these scales, which requires the cooperation of private landowners and municipal, provincial and federal managerial bodies. The task of retaining natural habitat in an urban environment should not fall to the city-planner alone, but to all private landowners. However, information is lacking on exactly how well privately owned green spaces such as golf courses in a suburban/urban environment can support wildlife, compared to other types of green spaces (e.g. nature reserves, recreational parks). Though comparing golf courses to other types of land use such as business districts, residential housing developments or agricultural landscapes (e.g. Blair 1996, 2004) would perhaps allow for more stark contrasts in results, I chose to focus this thesis on the potential differences between golf courses and suburban green spaces because they are, ecologically and structurally speaking, one of the more similar land use types to golf courses in this system. Finding potential differences between sites of relatively similar community composition and landscape coverage may then point towards simple management techniques that would allow for improvement in habitat provided by golf courses and/or green spaces.

Thus, my overall objectives were: i) to compare breeding bird communities on Montreal-area golf courses and other suburban green spaces and to link these communities to land cover characteristics representative of each site type so that management recommendations can be proposed to maximize avian richness; ii) to compare nest survival and predation rates of open-cup nesting passerines breeding on golf courses and on green spaces; iii) to identify the factors affecting the nest survival rates of these species; iv) to quantify the levels of organochlorines and polychlorinated biphenyls present in non-viable bird eggs collected on these sites; and v) to provide management recommendations targeting the planning, design and landscaping of both golf courses and green spaces to improve nest survival rates in this suburban landscape.

## LITERATURE CITED

- Andrén, H. 1994. Effects of habitat fragmentation on birds and mammals in landscapes with different proportions of suitable habitat: a review. Oikos 71: 355-366.
- Andrén, H. and P. Angelstam. 1988. Elevated predation rates as an edge effect in habitat islands: experimental evidence. Ecology 69: 544-547.
- Andrews, J. F. and W. Glowa. 1985. Diazinon poisoning of Brown-headed Cowbirds. Journal of Field Ornithology 56: 407-408.
- Angelstam, P. 1986. Predation on ground-nesting birds' nests in relation to predator densities and habitat edge. Oikos 47: 365-373.
- Banks, P. B. and J. V. Bryant. 2007. Four-legged friend or foe? Dog walking displaces native birds from natural areas. Biology Letters 3: 611–613.
- Bayne, E. M. and K. A. Hobson. 1999. Do clay eggs attract predators to artificial nests? Journal of Field Ornithology 70: 1-7.
- Beissinger, S. R. and D. R. Osborne. 1982. Effects of urbanization on avian community organization. Condor 84: 75-83.
- Berry, L. and A. Lill. 2003. Do predation rates on artificial nests accurately predict predation rates on natural nests? The effects of nest type, egg type and nest-site characteristics. Emu 103: 207-214.
- Bezzel, E. 1985. Birdlife in intensively used rural and urban environments. Ornis Fennica 62: 90-95.
- Blair, R. B. 1996. Land use and avian species diversity along an urban gradient. Ecological Applications 6: 506-519.
- Blair, R. 2004. The effects of urban sprawl on birds at multiple levels of biological organization. Ecology and Society 9. [Online.] Available at www.ecologyandsociety.org/vol9/iss5/art2/
- Blake, J. G. and J. R. Karr. 1984. Species composition of bird communities and the conservation benefit of large versus small forests. Biological Conservation 30: 173-187.

- Bouvier, J.-C., J.-F. Toubon, T. Boivin, and B. Sauphanor. 2005. Effects of apple orchard management strategies on the Great Tit (*Parus major*) in southeastern France. Environmental Toxicology and Chemistry 24: 2846-2852.
- Boyle, S. A. and F. B. Samson. 1985. Effects of nonconsumptive recreation on wildlife: a review. Wildlife Society Bulletin 13: 110-116.
- Brennan, A.-M. 1992. The management of golf courses as potential nature reserves. Aspects of Applied Biology 29: 241-248.
- Brewer, L. W., C. J. Driver, R. J. Kendall, T. E. Lacher Jr., and J. C. Galindo. 1988. Avian response to a turf application of Triumph® 4E. Environmental Toxicology and Chemistry 7: 391-401.
- Brittingham, M. C. and S. A. Temple. 1983. Have cowbirds cause forest songbirds to decline? Science 33: 31-35.
- Burhans, D. E. and F. R. Thompson III. 1998. Effects of time and nest-site characteristics on concealment of songbird nests. Condor 100: 663-672.
- Burhans, D. E. and F. R. Thompson III. 2001. Relationship of songbird nest concealment to nest fate and flushing behaviour of adults. Auk 118: 237-242.
- Burke, D. M. and E. Nol. 2000. Landscape and fragment size effects on reproductive success of forest-breeding birds in Ontario. Ecological Applications 10: 1749-1761.
- Busby, D. G., L. M. White, and P. A. Pearce. 1990. Effects of aerial spraying of fenitrothion on breeding White-throated Sparrows. Journal of Applied Ecology 27: 743-755.
- Canaday, C. and J. Rivadeneyra. 2001. Initial effects of a petroleum operation on Amazonian birds: terrestrial insectivores retreat. Biodiversity and Conservation 10: 567-595.
- Carter III, J. H. and B. G. Kocher. 1995. The Red-cockaded Woodpecker: an endangered species in golf country. USGA Green Section Record July/August 33: 8-9.

- Chace, J. F. and J. J. Walsh. 2006. Urban effects on native avifauna: a review. Landscape and Urban Planning 74: 46-69.
- Chalfoun, A. D., F. R. Thompson III, and M. J. Ratnaswamy. 2002. Nest predators and fragmentation: a review and meta-analysis. Conservation Biology 16: 306-318.
- Chamberlain, D. E., A. R. Cannon, M. P. Toms, D. I. Leech, B. J. Hatchwell, and K. J. Gaston. 2009. Avian productivity in urban landscapes: a review and meta-analysis. Ibis 151: 1-18.
- Clark, R. G. and T. D. Nudds. 1991. Habitat patch size and duck nesting success: the crucial experiments have not been performed. Wildlife Society Bulletin 19: 534-543.
- Clotfelter, E. D. 1998. What cues do Brown-headed Cowbirds use to locate Redwinged Blackbird host nests? Animal Behaviour 55: 1181-1189.
- Cohen, S., A. Svrjcek, T. Durborow, and N. LaJan Barnes. 1999. Water quality impacts by golf courses. Journal of Environmental Quality 28: 798-809.
- Colwell, M. A. 1992. Wilson's Phalarope nest success is not influenced by vegetation concealment. Condor 94: 767-772.
- Cooke, A. S. 1973. Shell thinning in avian eggs by environmental pollutants. Environmental Pollution 4: 85-152.
- Cooper, R. J., R. R. Wilson, G. D. Zenitsky, S. J. Mullin, J. A. DeCecco, M. R. Marshall, D. J. Wolf, and L. Y. Pomara. 1999. Does nonrandom nest placement imply nonrandom nest predation? -- a reply. Condor 101: 920-923.
- Cresswell, W. 1997. Nest predation: the relative effects of nest characteristics, clutch size and parental behaviour. Animal Behaviour 53: 93-103.
- Dale, S. 2004. Effects of a golf course on population dynamics of the endangered Ortolan Bunting. Journal of Wildlife Management 68: 719-724.
- Debinski, D. M. and R. D. Holt. 2000. A survey and overview of habitat fragmentation experiments. Conservation Biology 14: 342-355.

- Décarie, R., J.-L. DesGranges, C. Lépine, and F. Morneau. 1993. Impact of insecticides on the American Robin (*Turdus migratorius*) in a suburban environment. Environmental Pollution 80: 231-238.
- Devine, B. E. 1999. The Golf Course as a Nature Reserve: An Evaluation of Land Use and Diversity Applied to Ecosystem Design. PhD dissertation, University of Rhode Island, Rhode Island, USA.
- Diamond, J. M. 1976. Island biogeography and conservation: strategy and limitations. Science 193: 1027-1029.
- Doherty Jr., P. F. and T. C. Grubb Jr. 2002. Survivorship of permanent-resident birds in a fragmented forested landscape. Ecology 83: 844-857.
- Donnelly, R. and J. M. Marzluff. 2004. Importance of reserve size and landscape context to urban bird conservation. Conservation Biology 18: 733-745.
- Donovan, T. M., F. R. Thompson III, J. Faaborg, and J. R. Probst. 1995. Reproductive success of migratory birds in habitat sources and sinks. Conservation Biology 9: 1380-1395.
- Dotti, P. 2002. Is your course certified? USGA Green Section Record January/February 40: 17-20.
- Driscoll, M. J. L. and T. M. Donovan. 2004. Landscape context moderates edge effects: nesting success of Wood Thrushes in Central New York. Conservation Biology 18: 1330-1338.
- Dunford, W. and K. Freemark. 2004. Matrix matters: effects of surrounding land uses on forest birds near Ottawa, Canada. Landscape Ecology 20: 497-511.
- Dunn, E. H. and D. L. Tessaglia. 1994. Predation of birds at feeders in winter. Journal of Field Ornithology 65: 8-16.
- Easton, W. E. and K. Martin. 2002. Effects of thinning and herbicide treatments on nest-site selection by songbirds in young managed forests. Auk 119: 685-694.
- Fernández, C. and P. Azkona. 1993. Human disturbance affects parental care of Marsh Harriers and nutritional status of nestlings. Journal of Wildlife Management 57: 602-608.

- Filliater, T. S., R. Breitwisch, and P. M. Nealen. 1994. Predation on Northern Cardinal nests: does choice of nest site matter? Condor 96: 761-768.
- Flaspohler, D. J., S. A. Temple, and R. N. Rosenfield. 2000. Relationship between nest success and concealment in two ground-nesting passerines. Journal of Field Ornithology 71: 736-747.
- Forman, R. T. and L. E. Alexander. 1998. Roads and their major ecological effects. Annual Review of Ecology and Systematics 29: 207-231.
- Frank, R., P. Mineau, H. E. Braun, I. K. Barker, S. W. Kennedy, and S. Trudeau.1991. Deaths of Canada geese following spraying of turf with diazinon.Bulletin of Environmental Contamination and Toxicology 46: 852-858.
- Friesen, L. E., P. F. J. Eagles, and R. J. MacKay. 1995. Effects of residential development on forest-dwelling Neotropical migrant songbirds. Conservation Biology 9: 1408-1414.
- Fry, D. M. 1995. Reproductive effects in birds exposed to pesticides and industrial chemicals. Environmental Health Perspectives 103 (Suppl.): 165-171.
- Gates, J. E. and L. W. Gysel. 1978. Avian nest dispersion and fledging success in field-forest ecotones. Ecology 59: 871-883.
- Gill, H., L. K. Wilson, K. M. Cheng, and J. E. Elliot. 2003. An assessment of DDT and other chlorinated compounds and the reproductive success of American Robins (*Turdus migratorius*) breeding in fruit orchards. Ecotoxicology 12: 113-123.
- Gillihan, S. W. 2000. Bird Conservation on Golf Courses: A Design and Management Manual. Ann Arbor Press, Chelsea, MI.
- Götmark, F. 1992. The effects of investigator disturbance on nesting birds. Current Ornithology 9: 63-104.
- Grant, T. A., T. L. Shaffer, E. M. Madden, and P. J. Pietz. 2005. Time-specific variation in passerine nest survival: new insights into old questions. Auk 122: 661-672.
- Green, B. H. and I. C. Marshall. 1987. An assessment of the role of golf courses in Kent, England, in protecting wildlife and landscapes. Landscape and Urban Planning 14: 143-154.

- Grue, C. E., G. V. N. Powell, and M. J. McChesney. 1982. Care of nestlings by wild female starlings exposed to an organophosphate pesticide. Journal of Applied Ecology 19: 327-335.
- Gutzwiller, K. J., H. A. Marcum, H. B. Harvey, J. D. Roth, and S. H. Anderson. 1998. Bird tolerance to human intrusion in Wyoming montane forests. Condor 100: 519-527.
- Gutzwiller, K. J., R. T. Wiendenmann, K. L. Clements, and S. H. Anderson. 1994. Effects of human intrusion on song occurrence and singing consistency in subalpine birds. Auk 111: 28-37.
- Hahn, D. C. and J. S. Hatfield. 1995. Parasitism at the landscape scale: cowbirds prefer forests. Conservation Biology 9: 1415-1424.
- Hanski, I. K., T. J. Fenske, and G. J. Niemi. 1996. Lack of edge effects in nesting success of breeding birds in managed forest landscapes. Auk 113: 578-585.
- Hartley, M. J. and M. L. Hunter Jr. 1998. A meta-analysis of forest cover, edge effects, and artificial nest predation rates. Conservation Biology 12: 465-469.
- Haskell, D. G. 1995. A reevaluation of the effects of forest fragmentation on rates of bird-nest predation. Conservation Biology 9: 1316-1318.
- Haskell, D. G., A. M. Knupp, and M. C. Schneider. 2001. Nest predator abundance and urbanization. Pages 241-258 *in* Avian Ecology and Conservation in an Urbanizing World. (J. M. Marzluff, R. Bowman, and R. Donnelly, Eds.). Kluwer Academic Publishers, Norwell, MA.
- Herranz, J., M. Yanes and F. Suárez. 2002. Does photo-monitoring affect nest predation? Journal of Field Ornithology 73: 97-101.
- Heske, E. J., S. K. Robinson, and J. D. Brawn. 2001. Nest predation and neotropical migrant songbirds: piecing together the fragments. Wildlife Society Bulletin 29: 52-61.
- Higgs, A. J. and M. B. Usher. 1980. Should nature reserves be large or small? Nature 285: 568-569.

- Hodgkison, S., J.-M. Hero, and J. Warnken. 2007a. The efficacy of small-scale conservation efforts, as assessed on Australian golf courses. Biological Conservation 136: 576-586.
- Hodgkison, S. C., J.-M. Hero, and J. Warnken. 2007b. The conservation value of suburban golf courses in a rapidly urbanizing region of Australia. Landscape and Urban Planning 79: 323-337.
- Hostetler, M. 2001. The importance of multi-scale analyses in avian habitat selection in urban environments. Pages 139-154 *in* Avian Ecology and Conservation in an Urbanizing World. (J. M. Marzluff, R. Bowman, and R. Donnelly, Eds.). Kluwer Academic Publishers, Norwell, MA.
- Hostetler, M. and K. Knowles-Yanez. 2003. Land use, scale, and bird distributions in the Phoenix metropolitan area. Landscape and Urban Planning 62: 55–68.
- Howe, R. W. and G. J. Davis. 1991. The demographic significance of 'sink' populations. Biological Conservation 57: 239-255.
- Huhta, E., T. Aho, A. Jäntti, P. Suorsa, M. Kuitunen, A. Nikula, and H. Hakkarainen. 2004. Forest fragmentation increases nest predation in the Eurasian Treecreeper. Conservation Biology 18: 148-155.
- Jodice, P. G. R. and S. R. Humphrey. 1992. Activity and diet of an urban population of Big Cypress fox squirrels. Journal of Wildlife Management 56: 685-692.
- Jokimäki, J. 1999. Occurrence of breeding bird species in urban parks: effects of park structure and broad-scale variables. Urban Ecosystems 3: 21-34.
- Jones, J. 2001. Habitat selection studies in avian ecology: a critical review. Auk 118: 557-562.
- Jones, J. and R. J. Robertson. 2001. Territory and nest-site selection of Cerulean Warblers in eastern Ontario. Auk 118: 727-735.
- Jones, S. G., D. H. Gordon, G. M. Philips, and B. R. D. Richardson. 2005. Avian community response to a golf-course landscape unit gradient. Wildlife Society Bulletin 33: 422-434.

- Kendall, R. J., L. W. Brewer, R. R. Hitchcock, and J. R. Mayer. 1992. American widgeon mortality associated with turf application of diazinon AG500. Journal of Wildlife Diseases 28: 263-267.
- Kendall, R. J., L. W. Brewer, and R. R. Hitchcock. 1993. Response of Canada Geese to a turf application of diazinon AG500. Journal of Wildlife Diseases 29: 458-464.
- Knopper, L. D. and D. R. S. Lean. 2004. Carcinogenic and genotoxic potential of turf pesticides commonly used on golf courses. Journal of Toxicology and Environmental Health, Part B 7: 267-279.
- Kristan III, W. B. 2003. The role of habitat selection behavior in population dynamics: source-sink systems and ecological traps. Oikos 103: 457-468.
- Lahti, D. C. 2001. The edge effect on nest predation hypothesis after 20 years. Biological Conservation 99: 365-374.
- Lancaster, R. K. and W. E. Rees. 1979. Bird communities and the structure of urban habitats. Canadian Journal of Zoology 57: 2358-2368.
- Larivière, S. 1999. Reasons why predators cannot be inferred from nest remains. Condor 101: 718-721.
- Lavis, S. A. 1995. Pinehurst provides habitat for cavity-nesting species. USGA Green Section Record September/October 33: 11-12.
- LeClerc, J. E., J. P. K. Che, J. P. Swaddle, and D. A. Cristol. 2005. Reproductive success and developmental stability of eastern bluebirds on golf courses: evidence that golf courses can be productive. Wildlife Society Bulletin 33: 483-493.
- LeClerc, J. E. and D. A. Cristol. 2005. Are golf courses providing habitat for birds of conservation concern in Virginia? Wildlife Society Bulletin 33: 463-470.
- Leopold, A. 1933. Game Management. Charles Scribner's Sons, New York, USA.
- MacArthur, R. H. and E. Wilson. 1963. An equilibrium theory of insular zoogeography. Evolution 17: 373-387.
- Maffei, E. J. 1978. Golf courses as wildlife habitat. Transactions of the Northeast Section of the Wildlife Society 35: 120-129.

- Major, R. E. 1991. Identification of nest predators by photography, dummy eggs and adhesive tape. Auk 108: 190-195.
- Major, R. E., F. J. Christie, G. Growing, and T. J. Ivison. 1999. Elevated rates of predation on artificial nests in linear strips of habitat. Journal of Field Ornithology 70: 351-364.
- Major, R. E. and C. E. Kendal. 1996. The contribution of artificial nest experiments to understanding avian reproductive success: a review of methods and conclusions. Ibis 138: 298-307.
- Marshall, M. R., R. J. Cooper, J. A. DeCecco, J. Strazanac, and L. Butler. 2002. Effects of the experimentally reduced prey abundance on the breeding ecology of the Red-eyed Vireo. Ecological Applications 12: 261-280.
- Martin, P. A., D. L. Johnson, D. J. Forsyth, and B. D. Hill. 1998. Indirect effects of the pyrethroid insecticide deltamethrin on reproductive success of chestnut-collared longspurs. Ecotoxicology 7: 89-97.
- Martin, P. A., D. L. Johnson, D. J. Forsyth, and B. D. Hill. 2000. Effects of two grasshopper control insecticides on food resources and reproductive success of two species of grassland songbirds. Environmental Toxicology and Chemistry 19: 2987-2996.
- Martin, T. E. 1987. Food as a limit on breeding birds: a life-history perspective. Annual Review of Ecology and Systematics 18: 453-487.
- Martin, T. E. 1993a. Nest predation among vegetation layers and habitat types: revising the dogmas. American Naturalist 141: 897-913.
- Martin, T. E. 1993b. Nest predation and nest sites. BioScience 43: 523-532.
- Marzluff, J. M. 1997. Effects of urbanization and recreation on songbirds. Pages 89-102 in Songbird Ecology in Southwestern Ponderosa Pine Forests: A Literature Review. (W. M. Block and D. M. Finch, Eds.). Technical Report RM-GTR-292, U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO.
- Marzluff, J. M., R. Bowman, and R. Donnelly (Eds). 2001. Avian Ecology and Conservation in an Urbanizing World. Kluwer Academic Publisher, Norwell, MA.

- Mayer-Gross, H., H. Q. P. Crick, and J. J. D. Greenwood. 1997. The effect of observers visiting the nests of passerines: an experimental study. Bird Study 44: 53-65.
- Metcalfe, T. L., P. J. Dillon, and C. D. Metcalfe. 2008. Detecting the transport of toxic pesticides from golf courses into watersheds in the Precambrian Shield region of Ontario, Canada. Environmental Toxicology and Chemistry 27: 811–818.
- Miller, J. R. and N. T. Hobbs. 2000. Recreational trails, human activity, and nest predation in lowland riparian areas. Landscape and Urban Planning 50: 227-236.
- Miller, S. G., R. L. Knight, and C. K. Miller. 2001. Wildlife responses to pedestrians and dogs. Wildlife Society Bulletin 29: 124-132.
- Morneau, F., R. Décarie, R. Pelletier, D. Lambert, J.-L. DesGranges, and J.-P. Savard. 1999. Changes in breeding bird richness and abundance in Montreal parks over a period of 15 years. Landscape and Urban Planning 44: 111-121.
- Moul, I. E. and J. E. Elliott. 1992. A Survey of Pesticide Use and Bird Activity on Selected Golf Courses in British Columbia. Technical Report Series No. 163. Canadian Wildlife Service, Pacific and Yukon Region, BC.
- Murcia, C. 1995. Edge effects in fragmented forests: implications for conservation. Trends in Ecology and Evolution 10: 58-62.
- Norman, R. F. and R. J. Robertson. 1975. Nest-searching behavior in the Brownheaded Cowbird. Auk 92: 610-611.
- Nour, N., E. Matthysen, and A. A. Dhondt. 1993. Artificial nest predation and habitat fragmentation: different trends in bird and mammal predators. Ecography 16: 111-116.
- Osbourne, P. and L. Osbourne. 1980. The contribution of nest site characteristics to breeding-success among Blackbirds *Turdus merula*. Ibis 122: 512-517.
- Pärt, T. and J. Wretenberg. 2002. Do artificial nests reveal relative nest predation risk for real nests? Journal of Avian Biology 33: 39-46.

- Paton, P. W. 1994. The effect of edge on avian nest success: how strong is the evidence? Conservation Biology 8: 17-26.
- Payne, R. B. 1977. The ecology of brood parasitism in birds. Annual Review of Ecology and Systematics 8: 1-28.
- Pidgeon, A. M., V. C. Radeloff, C. H. Flather, C. A. Lepczyk, M. K. Clayton, T. J. Hawbaker, and R. B. Hammer. 2007. Associations of forest bird species richness with housing and landscape patterns across the USA. Ecological Applications 17: 1989-2010.
- Porter, E. E., J. Bulluck, and R. B. Blair. 2005. Multiple spatial-scale assessment of the conservation value of golf courses for breeding birds in southwestern Ohio. Wildlife Society Bulletin 33: 494-506.
- Powell, G. V. N. 1984. Reproduction by an altricial songbird, the Red-winged Blackbird, in fields treated with the organophosphate insecticide fenthion. Journal of Applied Ecology 21: 83-95.
- Pulliam, H. R. 1988. Sources, sinks and population regulation. American Naturalist 132: 652-661.
- Quinney, T. E., D. J. T. Hussell, and C. D. Ankney. 1986. Sources of variation in growth of Tree Swallows. Auk 103: 389-400.
- Rainwater, T. R., V. A. Leopold, M. J. Hooper, and R. J. Kendall. 1995. Avian exposure to organophosphorus and carbamate pesticides on a coastal South Carolina golf course. Environmental Toxicology and Chemistry 14: 2155-2161.
- Rangen, S. A., R. G. Clark, and K. A. Hobson. 1999. Influence of nest-site vegetation and predator community on the success of artificial songbird nests. Canadian Journal of Zoology 77: 1676-1681.
- Rangen, S. A., R. G. Clark, and K. A. Hobson. 2000. Visual and olfactory attributes of artificial nests. Auk 117: 136-146.
- Ricklefs, R. E. 1969. An analysis of nesting mortality in birds. Smithsonian Contributions to Zoology 9: 1-48.

- Riffell, S. K., K. J. Gutzwiller, and S. H. Anderson. 1996. Does repeated human intrusion cause cumulative declines in avian richness and abundance? Ecological Applications 6: 492-505.
- Robinson, S. K., F. R. Thompson III, T. M. Donovan, D. R. Whitehead, and J. Faaborg. 1995. Regional forest fragmentation and the nesting success of migratory birds. Science 267: 1987-1990.
- Robinson, S. K., J. D. Brawn, and J. P. Hoover. 1997. Effectiveness of small nature preserves for breeding birds. Pages 154-188 *in* Conservation in Highly Fragmented Landscapes. (M. W. Schwartz Ed.). Chapman & Hall, NY.
- Rodewald, A. D. 2003. The importance of land uses within the landscape matrix. Wildlife Society Bulletin 31: 586-592.
- Rodewald, A. D. and D. P. Shustack. 2008. Urban flight: understanding individual and population-level responses of Nearctic–Neotropical migratory birds to urbanization. Journal of Animal Ecology 77: 83-91.
- Rodewald, P. G., M. J. Santiago, and A. D. Rodewald. 2005. Habitat use of breeding red-headed woodpeckers on golf courses in Ohio. Wildlife Society Bulletin 33: 448-453.
- Rogers, C. M. 1994. Avian nest success, brood parasitism and edge-independent reproduction in an Alaskan wetland. Journal of Field Ornithology 65: 433-440.
- Roos, S. 2002. Functional response, seasonal decline and landscape differences in nest predation risk. Oecologia 133: 608-615.
- Rudnicky, T. C. and M. L. Hunter Jr. 1993. Avian nest predation in clearcuts, forests, and edges in a forest-dominated landscape. Journal of Wildlife Management 57: 358-364.
- Sadlon, N. P. 1994. Helping the Osprey on the road to recovery. USGA Green Section Record July/August 32: 16-17.
- Saunders, D. A., R. J. Hobbs, and C. R. Margules. 1991. Biological consequences of ecosystem fragmentation: a review. Conservation Biology 5: 18-32.

- Savard, J.-P., P. Clergeau, and G. Mennechez. 2000. Biodiversity concepts and urban ecosystems. Landscape and Urban Planning 48: 131-142.
- Schwartz, M. W. and P. J. van Mantgem. 1997. The value of small preserves in chronically fragmented landscapes. Pages 379-394 *in* Conservation in Highly Fragmented Landscapes. (M. W. Schwartz Ed.). Chapman & Hall, NY.
- Shafer, C. L. 1995. Values and shortcomings of small reserves. BioScience 45: 80-88.
- Shafer, C. L. 1997. Terrestrial nature reserve design at the urban/rural interface. Pages 345-378 in Conservation in Highly Fragmented Landscapes. (M. W. Schwartz Ed.). Chapman & Hall, NY.
- Simberloff, D. S. and L. G. Abele. 1976. Island biogeography and conservation: strategy and limitations. Science 193: 1032.
- Simberloff, D. S. and L. G. Abele. 1982. Refuge design and island biogeographic theory: effects of fragmentation. American Naturalist 120: 41-50.
- Smith, L. A., E. Nol, D. M. Burke, and K. E. Elliott. 2007. Nest-site selection of Rose-breasted Grosbeaks in southern Ontario. Wilson Journal of Ornithology 119: 151-161.
- Sorace, A. and M. Gustin. 2009. Distribution of generalist and specialist predators along urban gradients. Landscape and Urban Planning 90: 111-118.
- Sorace, A. and M. Visentin. 2007. Avian diversity on golf courses and surrounding landscapes in Italy. Landscape and Urban Planning 81: 81-90.
- Soulé, M. E., D. T. Bolger, A. C. Alberts, J. Wright, M. Sorice, and S. Hill. 1988. Reconstructed dynamics of rapid extinctions of chaparral-requiring birds in urban habitat islands. Conservation Biology 2: 75-92.
- Stanback, M. T. and M. L. Seifert. 2005. A comparison of eastern bluebird reproductive parameters in golf and rural habitats. Wildlife Society Bulletin 33: 471-482.
- Stephens, S. E., D. N. Koons, J. J. Rotella, and D. W. Willey. 2003. Effects of habitat fragmentation on avian nesting success: a review of the evidence at multiple spatial scales. Biological Conservation 115: 101-110.

- Stone, W. B. and H. Koch. 1982. American Brant killed on golf courses by diazinon. New York Fish and Game Journal 29: 95-96.
- Strausberger, B. M. 1998. Evident nest-searching behavior of female Brownheaded Cowbirds while attended by males. Wilson Bulletin 110: 133-136.
- Sugden, L. G. and G. W. Beyersbergen. 1986. Effect of density and concealment on American Crow predation of simulated duck nests. Journal of Wildlife Management 50: 9-14.
- Tanner, R. A. and A. C. Gange. 2005. Effects of golf courses on local biodiversity. Landscape and Urban Planning 71: 137-146.
- Temple, S. A. and J. R. Cary. 1988. Modeling dynamics of habitat-interior bird populations in fragmented landscapes. Conservation Biology 2: 340-347.
- Terborgh, J. 1976. Island biogeography and conservation: strategy and limitations. Science 193: 1029-1030.
- Terman, M. R. 1997. Natural links: naturalistic golf courses as wildlife habitat. Landscape and Urban Planning 38: 183-197.
- Tewksbury, J. J., L. Garner, S. Garner, J. D. Lloyd, V. Saab, and T. E. Martin. 2006. Tests of landscape influence: Nest predation and brood parasitism in fragmented ecosystems. Ecology 87: 759-768.
- Tewksbury, J. J., S. J. Hejl, and T. E. Martin. 1998. Breeding productivity does not decline with increasing fragmentation in a western landscape. Ecology 79: 2890-2903.
- Thompson III, F. R. and D. E. Burhans. 2003. Predation of songbird nests differs by predator and between field and forest habitats. Journal of Wildlife Management 67: 408-416.
- Tilghman, N. G. 1987. Characteristics of urban woodlands affecting breeding bird diversity and abundance. Landscape and Urban Planning 14: 481-495.
- van der Zande, A. N., J. C. Berkhuizen, H. C. van Latesteijn, W. J. ter Keurs, and A. J. Poppelaars. 1984. Impact of outdoor recreation on the density of a number of breeding bird species in woods adjacent to urban residential areas. Biological Conservation 30: 1-39.

- Van Horne, B. 1983. Density as a misleading indicator of habitat quality. Journal of Wildlife Management 47: 893-901.
- Villard, M.-A. 1998. On forest-interior species, edge avoidance, area sensitivity, and dogmas in avian conservation. Auk 115: 801-805.
- Whitcomb, R. F., J. F. Lynch, and P. A. Opler. 1976. Island biogeography and conservation: strategy and limitations. Science 193: 1030-1032.
- Wilcove, D. S. 1985. Nest predation in forest tracts and the decline of migratory songbirds. Ecology 66: 1211-1214.
- Williams, G. E. and P. B. Wood. 2002. Are traditional methods of determining nest predators and nest fates reliable? An experiment with Wood Thrushes (*Hylocichla mustelina*) using miniature video cameras. Auk 119: 1126-1132.
- Wong, J. W. C., C. W. Y. Chan, and K. C. Cheung. 1998. Nitrogen and phosphorus leaching from fertilizer applied on golf course: lysimeter study. Water, Air and Soil Pollution 107: 335-345.
- Woods, M., R. A. McDonald, and S. Harris. 2003. Predation of wildlife by domestic cats *Felis catus* in Great Britain. Mammal Review 33: 174-188.
- Yahner, R. H. 1988. Changes in wildlife communities near edges. Conservation Biology 2: 333-339.
- Yamada, T. 1995. Developing environmental guidelines for Canadian golf courses. USGA Green Section Record March/April 33: 8-9.
- Yasuda, M. and F. Koike. 2006. Do golf courses provide a refuge for flora and fauna in Japanese urban landscapes? Landscape and Urban Planning 75: 58-68.
- Zanette, L. 2002. What do artificial nests tell us about nest predation? Biological Conservation 103: 323-329.
- Zinkl, J. G., J. Rathert, and R. R. Hudson. 1978. Diazinon poisoning in wild Canada Geese. Journal of Wildlife Management 42: 406-408.

## CONNECTING STATEMENT 1

As highlighted in the previous chapter, many factors may affect birds living in fragmented, urbanized areas. These range from habitat-wide changes in vegetation, to smaller-scale changes such as shifts in nest-site selection. Any study focusing on potential differences in reproductive parameters between two types of site (e.g. golf courses and green spaces) should first determine how similar the breeding communities are between and within these site types, and how landscape-level variables shape these communities. As a first step towards understanding the factors affecting the breeding bird communities in this particular system, I compared the breeding bird assemblages among 12 Montrealarea sites (i.e. six golf courses and six green spaces). I identified the habitat factors that affect their relative incidence at a given site, and provide management recommendations that are designed to close the gap between the habitat provided on low-quality, depauperate golf courses and high-quality, species-rich green spaces in the Montreal area.

# $CHAPTER \ 2$

RECOMMENDATIONS FOR DESIGN AND MANAGEMENT OF GOLF COURSES AND GREEN SPACES BASED ON SURVEYS OF BREEDING BIRD COMMUNITIES IN MONTREAL

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#### Abstract

It has been suggested that golf courses can serve as wildlife habitat, provided key landscape features are present. Breeding bird communities were characterized on six golf courses and six green spaces in the Montreal area, Quebec, Canada. Landscape variables were measured to see which were the best correlates of avian species richness within and across sites. While there was no difference in the mean number of bird species (golf:  $31 \pm 2.4$  SE; green:  $33 \pm 4.5$ SE), golf courses supported fewer total species (golf: 55; green: 66). However, most of the species contributing to the green spaces' higher cumulative total were found at one site. Species composition differed between golf courses and green spaces, and was most correlated with site size, housing density surrounding the site, and the extent of coniferous tree, grass, and vegetated water cover. Our results support the work of others, and suggest that golf course architects can improve habitat quality by: 1) increasing deciduous and coniferous tree cover; 2) increasing native vegetation within and surrounding all water bodies; 3) reducing the amount of highly managed grass area; 4) ensuring that large, undeveloped buffers are maintained between golf courses and any development to prevent a shift in avian community composition with increasing urban sprawl; and 5) ensuring that both golf courses and green spaces are planned as large as possible.

The notion that relatively small, semi-natural areas can serve as wildlife habitat has garnered support in recent years, especially in urban areas (Shafer 1995, 1997, Cornelis and Hermy 2004). Until recently however, most studies have ignored an entire subset of urban reserve: privately owned, managed, and funded golf courses. As there are now well over 31,500 existing golf courses worldwide (Tanner and Gange 2005) and the size of an average 18-hole golf course is approximately 54 ha (Balogh et al. 1992), the total area occupied by golf courses is roughly two million hectares. Given that out-of-play areas represent 60% of the total area (Green and Marshall 1987), this represents a vast amount of potential wildlife habitat.

Considering the rate at which new golf courses are being created, existing golf courses are being re-landscaped, and urban green spaces are being degraded or lost, management plans are required if these sites are to provide wildlife habitat. A growing number of studies have suggested that golf courses have the potential to serve as wildlife habitat (Blair 1996, Terman 1997, Hostetler and Knowles-Yanez 2003, Yasuda and Koike 2006, Hodgkison et al. 2007a). The extent of this potential, however, is the source of recent work focusing on the role of golf courses in supporting avian species of conservation concern (LeClerc and Cristol 2005, Porter et al. 2005, Sorace and Visentin 2007), the impacts of golf course construction on breeding birds (Jones et al. 2005), and their conservation value for urban-threatened vertebrates (Hodgkison et al. 2007a, b).

Specific habitat features are important for preserving avian species diversity on golf courses in forested landscapes. In Virginia, LeClerc and Cristol (2005) found that the proportion of forest within 1.5 km of a golf course's center was the best predictor of a site's ability to support species of conservation. The importance of on-site forest cover has received extensive support from all over the world (Jones et al. 2005, Yasuda and Koike 2006, Sorace and Visentin 2007). However, Porter et al. (2005) concluded that on-site variables were less important than the surrounding landscape (e.g. natural vs. residential buffers) as predictors of bird diversity in Ohio. Hodgkison et al.'s (2007a) work found support for both on and off-site habitat characteristics.

Despite the fact that the drivers of urban avian assemblages are now beginning to be fairly well understood (Jokimäki 1999, Savard et al. 2000, Rodewald 2003), questions remain concerning the role that golf courses play in an urban system, and how they compare to other urban green spaces. Our objective was to extend the work of Terman (1997) and compare avian assemblages on Montreal-area golf courses and green spaces to see whether they are explained by key landscape elements. We predicted that, due to their design, golf courses would be more similar to each other in both landscape metrics and breeding bird communities than green spaces would be to each other. We also predicted that certain landscape elements would differ between golf courses and green spaces, with more extensive grass, conifer, non-vegetated water and unusable area on golf courses than green spaces, which would then accommodate different breeding bird communities.

#### MATERIALS AND METHODS

*Study area.* -- Twelve study sites (six golf courses and six green spaces) were studied in the greater Montreal area of Quebec, Canada ( $45^{\circ} 27^{\circ}N$ ,  $73^{\circ}51^{\circ}W$ ; Fig. 1). Each site is briefly described in Table 1, and land cover variables are listed in Table 6. All sites are representative of the St. Lawrence Lowland vegetation community (i.e. dominated by broadleaf forest and fertile agricultural soils), though broadleaf forests were largely restricted to unmanaged areas on golf courses. Sites were selected based on size and location to ensure comparable dominant vegetation types. Both site types include small (< 100 ha) and large (> 100 ha) sites. We selected green spaces with roughly 50% tree and < 10% grass cover, and each site is known as a good birding spot by local birdwatchers.

Golf courses were dominated by both highly and moderately maintained sections of turfgrass, and mixes of deciduous forest tracts (e.g. *Acer*, *Ulmus*, *Populus*, and *Tilia* spp.) with brushy understory at edges (e.g. *Crataegus*, *Rhamnus* and *Lonicera* spp.), small deciduous stands without understory, and single trees. Single or small stands of coniferous trees, both native and introduced (e.g. *Picea pungens*, *P. glauca*, *Pinus strobus*, *P. resinosa*, *Larix laricina*, *Thuja* 

*occidentalis*), and artificial, vegetated water bodies (e.g. *Butomus umbellatus, Typha, Phragmites* and *Carex* spp.) and non-vegetated water bodies were also on the courses. Green spaces were dominated by tracts of deciduous forest, as well as field and pasture species (e.g. *Solidago, Crataegus, Rhamnus* spp.). Green spaces had both natural and artificial vegetated water bodies.

*Breeding bird surveys.* -- Each site was surveyed twice from June 8 2006 to July 10 2006. Sampling was restricted to the core of the breeding season to avoid migratory and non-breeding birds. We separated the two visits by 15-20 days to maximize species detection and reduce temporal bias (Drapeau et al. 1999). Observations began at sunrise and generally continued for four hours. To avoid introducing a time-of-day bias, the second survey began from the end point of the first survey and rotated around the site in the opposite direction. For golf courses, the observer followed the course of play to facilitate covering the entire area, including out-of-play areas, deviating as necessary to identify or locate birds. This search pattern was approximated in green spaces, however the observer was free to move throughout all study areas as necessary.

All species encountered by sight and/or sound were recorded in 20-min intervals. Each individual was assigned a breeding code (Gauthier and Aubry 1996). This system infers a hierarchy based on breeding-activity evidence, the two most important categories for this study being: 1) probable breeding (e.g. pair observed in suitable nesting habitat, territoriality, courtship display) and 2) confirmed breeding (e.g. nest-building, nest with young or eggshells found, adult carrying food or nesting material, recently fledged young). Possible breeding was assigned when a species was observed in suitable nesting habitat.

We used an intensive search method with a results-based stopping rule to standardize results over sites very different in area (range: 22–218 ha; Watson 2003). Surveys ended when two consecutive 20-min intervals elapsed without any additional species being recorded. We used this method instead of the more traditional point-count system because a trial in 2005 indicated that point-counts failed to accurately represent the sites' avifauna for the following reasons: 1)

point-to-point travel was too long and difficult on certain sites, making the number of point-counts executed in one morning too low; 2) sites were all fairly heterogeneous, so the assumption of no bias in spatial sampling due to randomly distributed points was violated; and 3) considering that only five point count stations distanced 250 m apart (the suggested minimum in open areas) fit on the smaller sites, the method fell short of the suggested minimum of 20-30 stations per site for statistical validity (Ralph et al. 1995).

We did not survey when winds were over 20 km/h or under rainy or foggy conditions. Surveys were occasionally paused until noise levels were reduced to acceptable levels. The same observer was used for all counts. Birds flying over the site and birds whose positions (on or off-site) could not be clearly established were listed separately.

Landscape measurements. -- Landscape-level habitat variables were collected using a combination of GIS-based methods, Google Earth (v.3.0.072, Google, Inc., Mountain View, CA) imagery, and site visits. Distances were measured using Google Earth and all orthophotos (scale 1:40 000) were digitized using Map Maker Pro software (version 3.5, Map Maker Ltd, Kintyre, UK) to estimate habitat composition of each site. Overall area (ha) was measured (SIZE), sites were categorized based on land use (golf course or green space), and the number of buildings within a 200-m buffer surrounding each site was counted and divided by site area (HOUSE). We calculated proportional surface area (ha) of vegetated water bodies (VGWATER), non-vegetated water bodies (NVWATER), deciduous canopy cover (TREE), coniferous canopy cover (CONIFER), unusable surfaces (e.g. paved paths, buildings, parking lots; UNUSABLE), mowed grass (e.g. daily; GRASS), and fallow field, pasture or shrubs (PASTURE). The shortest distance from the centre of the site to the nearest forest fragment larger than 100 ha (FOREST) and, since the greater Montreal area is mostly made up of islands, the distance to the nearest river or lake shoreline (WATER; e.g. Lac St-Louis, Lac des Deux-Montagnes), were also calculated.

Data analysis. -- Relative species incidence was used in species assemblage analyses. To calculate species incidence, results from the second survey were combined with the first. We only considered breeding species (confirmed or probable). A scale of relative species incidence (0 indicating absence to 10 indicating high abundance) was calculated as the number of intervals during which a species was encountered divided by the total number of intervals for that site, then multiplied by 10 to obtain an integer. Species were classified as either significantly increasing or decreasing at P < 0.05 (Breeding Bird Survey (BBS), Quebec region 1986-2006; Downes and Collins 2007, Sauer et al. 2007), and categorized as either urban or suburban, successional forest or scrubland, grassland, wetland, or woodland breeders (Gauthier and Aubry 1996, Downes and Collins 2007, Sauer et al. 2007). Generalist species that nest in more than one category of habitat were assigned the habitat in which they were found in this study. Species of Regional Importance (SRI) were also identified based on Partners in Flight (PIF) species assessment of species present in Bird Conservation Region (BCR) 12 (PIF 2002). SRIs are defined as species which meet the following three criteria: Regional Combined Scores > 13 (which are calculated using global population size, regional population trend, global breeding distribution, regional threats to breeding and relative density scores); high or moderate regional threats paired with significant population declines; and significant populations within the BCR (PIF 2002).

Following McCune and Grace (2002), we used Whittaker's (1972) three definitions of diversity: alpha ( $\alpha$ , species richness per study area); beta ( $\beta$ , the total number of species divided by the average number of species,  $\gamma/\alpha$ -1); and gamma ( $\gamma$ , the total number of species across all study plots). Two levels of gamma were used: the total number of species over all sites ( $\gamma_{total}$ ), as well as a lower-level gamma representing the total number of species over golf courses ( $\gamma_{golf}$ ) and green spaces ( $\gamma_{green}$ ). Though Whittaker's  $\beta$  does not have units, it is useful for gauging the amount of heterogeneity in the data set, with 0 indicating identical species over all study plots, values < 1 considered low and values > 5 considered high (McCune and Grace 2002).

Survey completeness was assessed by calculating seven non-parametric species richness estimators (ACE, ICE, CHAO1, CHAO2, JACK1, JACK2, BOOTSTRAP) using EstimateS (version 7.5) and averaging them to produce the expected species richness for each site and type of site (Colwell 2005). Pairwise Sørensen, Jaccard, Morisita-Horn and Bray-Curtis similarity coefficients were calculated to assess each site's community similarity to all other sites (EstimateS; Colwell 2005). Richness estimates were then used as the response variable in simple linear regressions using STATISTICA (StatsSoft, Inc., Tulsa, OK) to examine the effects of each landscape variable on species richness. These regressions were done after verifying the degree of correlation between each variable pair and each variable was transformed to improve normality (log for SIZE, square root for HOUSE, FOREST, WATER, and arcsine square root for all cover variables).

To explore the associations between sites, species, and landscape habitat variables, we first used indirect gradient analysis with non-metric multidimensional scaling (NMS) to ordinate sample units (12 sites) in species space (72 breeding species), using the ranked distances between the sample units (Legendre and Legendre 1998, McCune and Grace 2002, Beals 2006). Using PC-ORD (version 4.17; McCune and Mefford 1999), a detrended correspondence analysis (DCA) was first run on the matrix, its coordinates used as the starting configuration for the NMS ordination. We then ran a first NMS with a step-down in dimensionality from six to one axes to identify the number of axes required to best represent this community. This was run using the Sørensen (Bray-Curtis) distance measure with 400 iterations, 50 runs with real data, 50 runs with randomized data, an instability criterion of 0.0005, and a step length (rate of movement towards minimum stress) of 0.20. A Monte Carlo simulation using randomized data was run (50 iterations) to test the probability that a similar stress value would be found by chance. A second NMS was then calculated using the same configuration but with only one run with real data. The coordinates from the DCA were used as the starting point for both NMSs. To illustrate the correlation of all measured landscape variables with each resulting NMS axis, joint plots were created depicting the variables with  $r^2 > 0.2$  as vectors plotted from the
centroid of the ordination scores. Multiple  $r^2$  was also calculated as the overall measure of correlation of each variable with the entire ordination. This value is constant, as opposed to the axes-specific  $r^2$  values that can vary if the ordination is rotated (McCune and Grace 2002).

Multi-response permutation procedures (MRPP) were performed on the rank-transformed Sørensen (Bray-Curtis) distance matrix after weighting groups with  $C_i = n_i/\Sigma n_i$  (PC-ORD, version 4.17; McCune and Mefford 1999) to test the hypothesis that there are differences in species composition between golf courses and green spaces (i.e. there are differences in average within-group ranked distances; McCune and Grace 2002). Indicator species analysis (ISA; Dufrêne and Legendre 1997) was also used to test how faithful each species was to either site type, by combining information on species incidence and faithfulness of occurrence within a given site type. An indicator value was created for each species, which was then tested for significance using a Monte Carlo randomization procedure (1000 iterations). ISA complements MRPP well, as it supplements the test of differences between site types with an indication of how well each species differentiates between types (McCune and Grace 2002). Pairwise Mann-Whitney U tests were performed between incidence rates on golf courses and green spaces (significance levels set at *P* < 0.05; STATISTICA).

#### RESULTS

Breeding bird communities. -- Ninety species were detected over all 12 study sites. Non-breeders were eliminated, leaving 72 species for ensuing analyses ( $\gamma_{total}$ ). The number of 20-min intervals ranged between 8 and 13 for all surveys, representing 160-260 min surveys. Surveys captured 83% of the expected total number of breeding species on golf courses and green spaces (Table 2), indicating that the surveys were relatively complete and equal between habitats. Alpha diversity ranged from 22.5-39.0 on golf courses and 23.0-54.5 on green spaces, with a cumulative total of 55 species found on golf courses and 66 species on green spaces (Table 3). However, the mean number of species breeding on golf courses was similar to green spaces (Mann-Whitney U = 18, *P* = 1.00) since most of the species contributing to the green spaces' higher cumulative total were found at one site (IB). Beta diversity (0.986 for all sites) at golf courses (0.610) was also similar to green spaces (0.722).

Despite similar beta diversity and mean number of species, MRPP found significant differences in species composition between site types (T = -3.02, A = 0.16, P = 0.01). Thirteen species were found on all sites, seven species were only detected on golf courses, and 17 were only detected on green spaces (Table 4; see table for all scientific names). Seven species were found with higher consistency and frequency on golf courses than on green spaces (Mourning Dove, Barn Swallow, American Crow, European Starling, Chipping Sparrow, Song Sparrow, and Brown-headed Cowbird), while only one (Pileated Woodpecker) was found at a higher incidence on green spaces (Table 4). Fourteen species were only detected at one site and were removed before the analysis. Six of these species are marsh specialists (American Bittern, Least Bittern, Sora, Common Moorhen, Black Tern, and Marsh Wren), and were exclusively found at IB. The other seven species were spread between four golf courses (AG, BG, RM, IP; Cliff Swallow, Carolina Wren, Eastern Bluebird, and Black-throated Blue Warbler, respectively) and two green spaces (MR and BL; Least Flycatcher and Black-throated Green Warbler, Willow Flycatcher and Chestnut-sided Warbler, respectively).

Urban-breeding species were more common on golf courses than green spaces (Table 5). Also, there were more wetland- and woodland-breeding species on green spaces than on golf courses (woodland: 24 vs. 18; wetland: 14 vs. 8), but the mean number was not significantly different. The number of significantly increasing and decreasing species was similar between site types, with both having more species on the decrease than on the increase (Table 5). The mean number of PIF Species of Regional Importance on green spaces was similar to golf courses, though the cumulative total was marginally higher on green spaces (Table 5).

Golf courses had more homogeneous avian communities than green spaces (Appendix 1). Of the top 10 most similar sites for each of the four similarity coefficients, between five and seven pairs were golf courses, and only one was a green space pair (BL-TC; Morisita-Horn). Conversely, not one golf course pair was listed in the 10 least similar sites, while three green space pairs were listed (IB-CP, IB-TC and CP-MR).

Landscape variables and breeding bird communities. -- TREE, GRASS, NVWATER, and UNUSABLE differed between types of sites, whereas SIZE, HOUSE, FOREST, WATER, VGWATER, CONIFER, and PASTURE did not differ significantly (Table 6). Golf courses had three times less deciduous tree cover, over 12 times more grass cover, 17 times more non-vegetated water (which was virtually nil on green spaces), and twice as much unusable surface cover than green spaces. Green spaces had roughly four times more vegetated water surface cover than golf courses, but within-group variability reduced significance.

Avian communities on golf courses differed from those on green spaces (Fig. 2; ordination represents 93.5% of the variation in the data, stress = 5.091), which is supported by MMRP (P = 0.01) and Monte Carlo tests (50 runs, P =0.02). Golf courses were more tightly clustered than green spaces, reflecting their homogeneity. Pairwise ordinations with environmental variable joint plots also indicated separation between site types and location. In the first plot (Fig. 3A), golf courses and green spaces group together by site type. The three environmental variables most correlated to axis 1, the most important axis representing 49.8% of variation in the data, were GRASS, NVWATER and TREE (Table 7). When the environmental variables most correlated with axis 2, which represented 22.0% of the variation in the data, were also taken into account (SIZE, HOUSE and CONIFER; Table 7), it became clear that these breeding bird communities were different, and were correlated with higher deciduous tree cover and reduced grass cover on green spaces, and higher non-vegetated water, grass and coniferous tree cover on golf courses, as well as an overall increase in site size associated with golf courses. Axis 3, which represented 21.6% of the variation in the data, did not separate golf courses from green spaces when paired with axis 1 (Fig. 3B). The differences in community composition differentiated between larger, more natural sites situated at the periphery of the island, from smaller, highly managed sites closer to the city, regardless of site type (see Fig. 1 for locations). SIZE, VGWATER, CONIFER, FOREST, and HOUSE were all strongly correlated to axis 3 (Figs. 3B and C). Overall, these results suggest that site size, degree of urbanization surrounding the site, and degree of conifer, grass, and vegetated water cover on-site are important drivers of species composition in this system (Table 7). Linear regressions support these results, indicating that SIZE and CONIFER were positively related to species richness ( $r^2 = 0.43$ , P = 0.02 and  $r^2 = 0.44$ , P = 0.02), while HOUSE was negatively related ( $r^2 = 0.45$ , P = 0.02). However, both SIZE and HOUSE were significantly correlated to CONIFER, and SIZE and HOUSE were correlated, though not significantly (Appendix 2).

#### DISCUSSION

*Breeding bird communities.* -- Golf course assemblages were more alike than green space assemblages. Golf courses are built and managed for the same overall purpose, creating a more homogeneous vegetation community. Green spaces, however, are generally parcels of land that are simply set aside, making them inherently more variable in habitat. Previous work supports these results (Blair 1996, 2004, Merola-Zwartjes and DeLong 2005), with avian communities from golf courses in different states being more alike than the same pairwise comparisons between biological preserves (Blair 2001). It follows that if habitat complexity increases on golf courses (i.e. they become more naturalized), avian communities should become increasingly dissimilar. Our data do not support this prediction: the two most natural sites in this study, defined as those with the highest percent cover of TREE and VGWATER and lowest percent GRASS cover (RM and IP), were among the most similar. This question deserves further study, as our study did not have sufficient sample size to fully address this issue.

Golf courses house more urban-breeding species and less wetland- and woodland-breeding species than other more natural urban green spaces. This is likely because green spaces generally offer more high-quality wetlands (increased vegetation resulting in increased aquatic invertebrate production; Krull 1970), larger tracts of forest, fewer highly modified areas, and fewer conifers than golf courses. Thus, golf courses may be a source of dispersing suburban-adaptable bird species that may outcompete native species (LeClerc and Cristol 2005). However, while there were significantly more European Starlings on golf courses than on green spaces, there were also more Barn Swallows, a significantly declining species and a Species of Regional Importance (PIF 2002). While most studies have found fewer species of conservation concern on golf courses than on other reference sites (Terman 1997, LeClerc and Cristol 2005, Hodgkison et al. 2007b), the fact remains that certain golf courses may be important for species of conservation concern, especially in urban areas. Golf courses also provide habitat for native, urban-avoiding species, the importance of which should not be overlooked (Hodgkison et al. 2007a).

Landscape variables and breeding bird communities. -- Several landscape cover variables differed between site types. As expected, TREE, GRASS, UNUSABLE, and NVWATER differed drastically. Some of the predicted differences however, were not significant. For example, while CONIFER values tended to be higher on golf courses than on green spaces, the difference was lessened because IB contains a large eastern white cedar grove. Also, green spaces only had a slightly higher HOUSE average than golf courses, but had a much higher median (4.04 vs. 1.42 houses/site size). Though the effect of outliers and high within-group variability would be lessened with a larger sample size, we believe that they are themselves evidence of the inherent heterogeneity, with respect to habitat matrices, prevalent among green spaces. Despite this variability though, the use of powerful multivariate tools like NMS indicated that patterns may exist where basic statistics are unable to detect them (e.g. the role that conifer cover and the number of houses surrounding the site may play in avian species composition).

Our results suggest that these avian communities are shaped by both sitespecific land cover variables and overall landscape context, which support the findings of Hodgkison et al. (2007a). It is possible that the CONIFER variable was significant because increased conifer cover provided another type of habitat or structure within a largely deciduous landscape, increasing species richness as a result. Tilghman (1987) found that the percentage of coniferous cover explained roughly 25% of the variation in the number of birds present in urban woodlands, but warned that it may be due to sampling and a resulting correlation with woodland size. The importance of the CONIFER variable might be an artefact of its correlation with SIZE in this study as well, since the three largest sites also had the highest degree of conifer cover. CONIFER was also negatively correlated with HOUSE. This is likely due to the larger number of houses around green spaces than golf courses, which in turn tended to have more conifer cover than green spaces. None of these differences were significant however, so it may be that the relationship between conifer cover and species richness is real.

Overall, site size was the most important variable in this study. A large body of literature on urban green spaces mirrors these results (e.g. Pelletier 1984, Tilghman 1987, Fernández-Juricic and Jokimäki 2001, Cornelis and Hermy 2004). Work on golf courses has also found support for site size being an important driver of species richness, though Porter et al. (2005) argued that natural land cover (e.g. forested upland and riparian areas) of various buffer sizes within 1 km of the course has a stronger influence on avian diversity than either site size or any environmental variables measured within the site. They were not, however, able to completely disentangle the effects of site size due to correlations with land cover characteristics measured within nested buffers.

Our results appear to disagree with Porter et al. (2005) in terms of the importance of land-cover characteristics within the site, however a direct comparison is difficult since we studied both golf courses and green spaces together. Our results highlight the differences in vegetation between golf courses and green spaces, suggesting that land use, as indicated for example by the amount of tree cover versus grass cover (both highly correlated, r = -0.96, Appendix 2), plays a large role in species composition and richness. However, the fact that some on-site characteristics present on both site types (e.g. VGWATER) were correlated with breeding bird communities indicates that on-site variables can be important and should not be overlooked. This supports the conclusions of Hostetler and Knowles-Yanez (2003): land use as a qualitative or categorical

designation has limited power in predicting species richness values. Rather, vegetation structure and landscape design, which can be driven and shaped by land use, are more predictive.

*Limitations.* -- The TREE variable provided equal weight to areas with closed canopies but without understory and deciduous forests with dense understories. Clearly, deciduous forests with large amounts of understory and habitat complexity are of much greater conservation value than frequently mowed areas interspersed with large trees (e.g. Tilghman 1987, Jokimäki 1999). A clear distinction between the two types could not always be made using aerial photographs, so they were treated as one habitat type. The potential importance of shrub and understory habitat on golf courses should be investigated, as its importance in terms of presence, shape and degree of connectivity has been underscored in previous work concerning urban environments (Tilghman 1987, Savard et al. 2000, Hodgkison et al. 2007a).

The importance of buildings and maintenance equipment, lumped into the UNUSABLE category, should not be overlooked. Barn Swallows, one the PIF Regionally Important Species and a species undergoing decline, were found exclusively nesting in and on buildings. Cliff Swallow, Eastern Phoebe, Carolina Wren, American Robin, European Starling, and House Sparrow were also found nesting in maintenance garages or in/on equipment. Thus, buildings are not necessarily unusable, and managers should be aware of certain species' predilection for man-made shelters. If these birds are not to be tolerated in or around the premises for fear of damage to the equipment, for example, measures such as hole-sealing must be taken well before the breeding season to prevent nesting. The best solution, however, is to encourage nesting to further support breeding bird communities by providing a sheltered nest location, and to sensitize workers and players to wildlife.

The HOUSE variable, though we found it to be very important in this study, should be replaced with percent cover of unusable surfaces within the buffer in future studies. There were instances where industries or parking lots were within buffer boundaries, but contained only one building. Thus, their impact was equated to that of a residential home, which does not reflect reality.

*Management implications.* -- Golf courses are part of our suburban-urban environment, and are increasing in number. Though they may tend to house more urban-adaptable species that have less societal and environmental value, they do play a role in the urban environment – one that can be enhanced through careful management (Hodgkison et al. 2007a). Most golf course managers now recognize the importance of good wildlife practices and are willing to make changes to their management schemes (Hammond and Hudson 2007). It is critical that we encourage the beneficial management of these sites and continue to test a variety of landscaping management techniques that will be satisfactory for players and wildlife alike. Though we did not directly test management techniques, our data show the importance of key landscape characteristics in shaping breeding bird communities in Montreal.

Management strategies can be implemented to create and manage golf courses in the image of high-quality green spaces, as well as to restore degraded green spaces. We specify quality green spaces because of the large disparity between study sites. IB, with its exceptional aquatic habitat, contributed nine species to the green spaces' cumulative total of 66. Without IB, the cumulative total drops to 57, a much closer total to the golf courses' 55 species. Other green spaces contributed at most two unique species to the total, and some none. The most species-rich golf course (RM) contributed three species to the cumulative total, tied with EG and AG, sites with lower species totals. Though Porter et al. (2005) found little variation in species richness across their six sites, LeClerc and Cristol (2005) found enormous variation within 87 Virginia golf courses, as did Hodgkison et al. (2007b) within their 20 golf courses in Australia. Though this variability may be due to the distribution of study sites across several different eco-regions and sample size, it still highlights the importance of creating new golf courses and/or maintaining already established courses in the image of sites that provide quality habitat. In order to identify these quality sites, we have attempted to quantify the landscape metrics that promote species richness and provide structure to avian breeding communities in this region.

Site management in southern Quebec should involve increasing deciduous and coniferous tree cover, converting non-vegetated water bodies to vegetated water bodies, and reducing the amount of highly managed grass area to the minimal amount required for play. LeClerc and Cristol (2005) found that the golf courses with the highest value for species of conservation concern in Virginia had double the proportion of forest than the ones with the lowest value. Jones et al. (2005) reported that the total number of species, the number of migrant Neotropical species, and the number of species of conservation concern were all positively influenced by increasing proportions of forested area in South Carolina. Sorace and Visentin (2007) determined that the percentage of forested area correlated positively with species richness and with the number of species sensitive to forest fragmentation on Italian golf courses. In addition, by adopting this environmentally sound management option, golf courses would save money on staff time, effort, and equipment with fewer areas requiring high maintenance. Also, a smaller playing arena could increase the difficulty of the game by reducing easily playable surfaces, increasing the need for aim which might be a marketable challenge appealing to advanced golfers. As a compromise, even reducing highly managed areas on part of the course could increase species richness, while allowing both novice and advanced golfers to play at their respective levels. Granted, this management strategy is likely to be the least favourable with golfers, so care must be taken to weigh the economic consequences of a reduction in membership with any major changes to the landscape.

Another important consideration is the amount of vegetation surrounding and within natural or artificial water bodies. This includes not only native reeds and aquatic vegetation, but shrubs, and ideally larger buffers of native trees extending down to the water's edge, interspersed with bare ground and short grass to accommodate all foraging guilds (Cicero 1989, White and Main 2005). Increasing the structural complexity of these water bodies, even in a disturbed system, will increase habitat quality and species diversity (Cicero 1989).

The extent to which a site can be improved in terms of providing suitable wildlife habitat may ultimately be limited to its size and landscape context, as these characteristics (e.g. distance to nearest large forested area and number of houses surrounding the site) cannot be easily modified. However, site size is usually tied to species richness because larger sites tend to contain a greater variety of microhabitats, which in turn support a greater diversity of species (Donnelly and Marzluff 2004). Thus, increasing habitat complexity on small sites can likely increase species richness to a certain degree, though this measure will rarely address certain species-specific habitat restrictions (e.g. forest-interior species). More importantly though, the role that site size and degree of surrounding urbanization play in determining avian community composition, as suggested by our results and the previously cited results of others, imply that change in landscape context must be considered when planning a green space near an urbanized area, be it for conservation or recreation. Urban reserves should be designed to be as large as possible, since avian communities present on small urban sites are usually nested within those on larger sites (Fernández-Juricic and Jokimäki 2001). However, the positive relationship between site size and species richness does not always hold. Friesen et al. (1995) found that the amount of residential development adjacent to a woodland was more important to Neotropical migrant diversity than woodland size. Golf courses in particular should have large, undeveloped buffers incorporated into their designs, regardless of whether the site is located in an exurban or urban setting, in order to prevent a shift in community composition with encroaching or increasing urban sprawl, to maintain aesthetics, and to provide safety from wayward golf balls. Of the studies examining the ability of golf courses to support avian communities, all have suggested that having undeveloped, vegetated buffers surrounding golf courses is one of the most important factors in having increased species richness (LeClerc and Cristol 2005, Porter et al. 2005, Hodgkison et al. 2007a). LeClerc and Cristol (2005) specified that the courses supporting higher numbers of species of conservation concern were the ones with buffer zones that had half as much development as the more depauperate courses. Thus, the move towards golfing communities, e.g. golf courses whose vegetated borders have been replaced by residential developments, will likely be a major limiting factor in avian community composition and richness, and should therefore be avoided. This is especially true if these sites are small and have little wildlife habitat to begin with. Not all golf courses will be able to support species with special requirements, but by following these guidelines, golf courses will no doubt increase their ability to support a variety of avian communities.

Additional studies are needed to further our understanding of the role golf courses play in an urban environment. A question arising from this study is whether golf courses are part of urban green space complexes, since wide-ranging species might be able to exploit urban areas by moving from one green space to another, or young produced in one green space may disperse to another. If there is a high degree of interchange between golf courses and green spaces, this could mean that even relatively poor breeding sites might provide habitat for dispersing, feeding or hunting individuals. It is important that landscape-level effects be examined to best understand how well these urban sites support avian communities.

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- Balogh, J. C., V. A. Gibeault, W. J. Walker, M. P. Kenna, and J. T. Snow. 1992. Background and Overview of Environmental Issues. Pages 1-37 *in* Golf Course Management & Construction: Environmental Issues. (J. C. Balogh and W.J. Walker, Eds.). Lewis Publishers, Chelsea, MI.
- Beals, M. 2006. Understanding community structure: a data-driven multivariate approach. Oecologia 150: 484-495.
- Blair, R. 2004. The effects of urban sprawl on birds at multiple levels of biological organization. Ecology and Society 9. [Online.]. Available at www.ecologyandsociety.org/vol9/iss5/art2.
- Blair, R. B. 1996. Land use and avian species diversity along an urban gradient. Ecological Applications 6: 506-519.
- Blair, R. B. 2001. Birds and butterflies along urban gradients in two ecoregions of the United States: is urbanization creating a homogeneous fauna? Pages 33-56 *in* Biotic Homogenization. (J. L. Lockwood and M. L. McKinney, Eds.). Kluwer Academic/Plenum Publishers, New York, NY.
- Cicero, C. 1989. Avian community structure in a large urban park: controls of local richness and diversity. Landscape Urban Planning 17: 221-240.
- Colwell, R. 2005. EstimateS: Statistical estimation of species richness and shared species from samples. Version 7.5. [Online.]. Available at purl.oclc.org/estimates.
- Cornelis, J. and M. Hermy. 2004. Biodiversity relationships in urban and suburban parks in Flanders. Landscape Urban Planning 69: 385-401.
- Donnelly, R. and J. M. Marzluff. 2004. Importance of reserve size and landscape context to urban bird conservation. Conservation Biology 18: 733-745.
- Downes, C. M. and B. T. Collins. 2007. Canadian Bird Trends web site. Version 2.2. [Online.]. Available at www.cws-scf.ec.gc.ca/mgbc/trends/.
- Drapeau, P., A. Leduc and R. McNeil. 1999. Refining the use of point-counts at the scale of individual points in studies of bird-habitat relationships. Journal of Avian Biology 30: 367-382.

- Dufrêne, M. and P. Legendre. 1997. Species assemblages and indicator species: the need for a flexible asymmetrical approach. Ecological Monographs 67: 345-366.
- Fernández-Juricic, E. and J. Jokimäki. 2001. A habitat island approach to conserving birds in urban landscapes: case studies from southern and northern Europe. Biodiversity Conservation 10: 2023-2043.
- Friesen, L. E., P. F. J. Eagles, and R. J. MacKay. 1995. Effects of residential development on forest-dwelling neotropical migrant songbirds. Conservation Biology 9: 1408-1414.
- Gauthier, J. and Y. Aubry. 1996. The Breeding Birds of Quebec: Atlas of the Breeding Birds of Southern Quebec. Association québécoise des groupes d'ornithologues, Province of Quebec Society for the Protection of Birds, Canadian Wildlife Service, Environment Canada, Quebec Region, Montreal, QC.
- Green, B. H. and I. C. Marshall. 1987. An assessment of the role of golf courses in Kent, England, in protecting wildlife and landscapes. Landscape Urban Planning 14: 143-154.
- Hammond, R. A. and M. D. Hudson. 2007. Environmental management of UK golf courses for biodiversity - attitudes and actions. Landscape Urban Planning 83: 127-136.
- Hodgkison, S., J.-M. Hero, and J. Warnken. 2007a. The efficacy of small-scale conservation efforts, as assessed on Australian golf courses. Biological Conservation 136: 576-586.
- Hodgkison, S., J.-M. Hero, and J. Warnken. 2007b. The conservation value of suburban golf courses in a rapidly urbanising region of Australia. Landscape Urban Planning 79: 323-337.
- Hostetler, M. and K. Knowles-Yanez. 2003. Land use, scale, and bird distributions in the Phoenix metropolitan area. Landscape Urban Planning 62: 55–68.
- Jokimäki, J. 1999. Occurrence of breeding bird species in urban parks: effects of park structure and broad-scale variables. Urban Ecosystems 3: 21-34.

- Jones, S. G., D. H. Gordon, G. M. Philips, and B. R. D. Richardson. 2005. Avian community response to a golf-course landscape unit gradient. Wildlife Society Bulletin 33: 422-434.
- Krull, J. N. 1970. Aquatic plant-macroinvertebrate association and waterfowl. Journal of Wildlife Management 34: 707-718.
- LeClerc, J. E. and D. A. Cristol. 2005. Are golf courses providing habitat for birds of conservation concern in Virginia? Wildlife Society Bulletin 33: 463-470.
- Legendre, P. and L. Legendre. 1998. Numerical Ecology: Developments in Environmental Modelling 20. 2<sup>nd</sup> ed. Elsevier Scientific Publishing Company, Amsterdam, The Netherlands.
- McCune, B. and J. B. Grace. 2002. Analysis of Ecological Communities. MjM Software Design, Gleneden Beach, OR.
- McCune, B. and M. Mefford. 1999. Multivariate analysis of ecological data. Version4.17. MjM Software Design, Gleneden Beach, OR.
- Merola-Zwartjes, M. and J. P. DeLong. 2005. Avian species assemblages in New Mexico golf courses: surrogate riparian habitat for birds? Wildlife Society Bulletin 33 : 435-447.
- Pelletier, R. 1984. Relation entre la superficie des parcs et des espaces verts urbains de Montréal (Québec), la physionomie de la végétation et la structure des peuplements d'oiseaux. M.Sc. dissertation, Université du Québec à Montréal, Montréal, QC.
- Partners in Flight (PIF). 2002. Partners in Flight Species Assessment Database. Rocky Mountain Bird Observatory. [Online.] Available at www.rmbo.org/pif/pifdb.html.
- Porter, E. E., J. Bulluck, and R. B. Blair. 2005. Multiple spatial-scale assessment of the conservation value of golf courses for breeding birds in southwestern Ohio. Wildlife Society Bulletin 33: 494-506.
- Ralph, C. J., S. Droege, and J. R. Sauer. 1995. Managing and monitoring birds using point counts: standards and applications. Pages 161-168 in Monitoring Bird Populations by Point Counts. (C. J. Ralph, J. R. Sauer,

and S. Droege, Eds.). Pacific Southwest Research Station, USDA Forest Service, Albany, CA.

- Rodewald, A. D. 2003. The importance of land uses within the landscape matrix. Wildlife Society Bulletin 31: 586-592.
- Sauer, J. R., J. E. Hines, and J. Fallon. 2007. The North American Breeding Bird Survey, Results and Analysis 1966-2006. Version 10.13. [Online.]. Available at www.mbr-pwrc.usgs.gov/bbs/bbs2006.html
- Savard, J.-P., P. Clergeau, and G. Mennechez. 2000. Biodiversity concepts and urban ecosystems. Landscape Urban Planning 48: 131-142.
- Shafer, C. L. 1995. Values and shortcomings of small reserves. BioScience 45: 80-88.
- Shafer, C. L. 1997. Terrestrial nature reserve design at the urban/rural interface. Pages 345-378 in Conservation in Highly Fragmented Landscapes. (M.W. Schwartz Ed.). Chapman & Hall, New York, NY.
- Sorace, A. and M. Visentin. 2007. Avian diversity on golf courses and surrounding landscapes in Italy. Landscape Urban Planning 81: 81-90.
- Tanner, R. A. and A. C. Gange. 2005. Effects of golf courses on local biodiversity. Landscape Urban Planning 71: 137-146.
- Terman, M. R. 1997. Natural links: naturalistic golf courses as wildlife habitat. Landscape Urban Planning 38: 183-197.
- Tilghman, N. G. 1987. Characteristics of urban woodlands affecting breeding bird diversity and abundance. Landscape Urban Planning 14: 481-495.
- Watson, D. M. 2003. The 'standardized search': An improved way to conduct bird surveys. Austral Ecology 28: 515-525.
- White, C. L. and M. B. Main. 2005. Waterbird use of created wetlands in golfcourse landscapes. Wildlife Society Bulletin 33: 411-421.
- Whittaker, R. H. 1972. Evolution and measurement of species diversity. Taxon 21: 213-251.
- Yasuda, M. and F. Koike. 2006. Do golf courses provide a refuge for flora and fauna in Japanese urban landscapes? Landscape Urban Planning 75: 58-68.



Fig. 1. Map of the greater Montreal area, Quebec, Canada. Sites shaded in black are golf courses; sites highlighted in white are green spaces (research reserves and nature parks). See Table 1 for site codes and descriptions.

Table 1. Description of 12 study sites in the Montreal area, listed from smallest in area to largest.

Study Site	Landscape context and site description
Stoneycroft Wildlife Area (SW)	Exurban; research area with restricted access surrounded by forest and fields; mix of shrub, pastureland and deciduous forest, small wetland; 22 ha
Terra Cotta Park (TC)	Suburban; recreational nature park surrounded by residential development; mix of shrub, pastureland and deciduous forest; 34 ha
Centennial Park (CP)	Suburban; recreational nature park surrounded by residential development; deciduous forest with artificial lake; 50 ha
Meadowbrook Golf Club (MG)	Urban; public golf course surrounded by residential development and rail yard parking lot, very little water on site; 56 ha
Beaconsfield Golf Club (BG)	Suburban; private, Audubon-certified golf course completely surrounded by residential development, several well-vegetated ponds; 65 ha
Ile Perrot Golf & Country Club (IP)	Exurban; private golf course surrounded by residential development, deciduous forest and agriculture, several well vegetated ponds; 69 ha
Molson Reserve (MR)	Exurban/Suburban; research area partly surrounded by residential development; dominated by deciduous forest, large parts flooded into wetlands; 73 ha
Atlantide Golf Club (AG)	Exurban/Suburban; semi-private golf course surrounded by large water body and residential development; several ponds, most vegetated; 105 ha
Bois-de-Liesse Park (BL)	Suburban/Urban; nature park surrounded by residential and industrial development; mix of deciduous forest and fallow fields, some flooded areas; 129 ha
Elm Ridge Country Club (EG)	Exurban; private golf course surrounded by fields, deciduous forest and slight residential development; several non-vegetated ponds; 162 ha
Royal Montreal Golf Club (RM)	Exurban; private, Audubon-certified golf course surrounded by fields, deciduous forest and slight residential development; several ponds, most well-vegetated; 215 ha
Ile-Bizard Nature Park (IB)	Exurban; nature park surrounded by slight residential development; mix of shrub and deciduous forest, coniferous forest, several large wetlands; 218 ha

Table 2. Mean species richness estimators ( $\pm$  SE) for evaluating bird species richness on golf courses and green spaces.

Species richness estimator	Golf courses	Green spaces
Observed mean species richness	30.92 (2.4)	33.42 (4.5)
ACE	46.01 (6.64)	48.08 (3.87)
ICE	35.83 (3.55)	41.07 (3.74)
Chao1	44.30 (6.20)	45.39 (3.89)
Chao2	33.13 (2.87)	36.59 (4.21)
Jack1	32.87 (2.78)	36.09 (4.28)
Jack2	34.74 (3.20)	38.73 (4.13)
Bootstrap	32.41 (2.68)	35.43 (4.32)
Mean estimated species richness	37.04 (2.32)	40.20 (1.86)
% of total estimated species detected by surveys	83.48%	83.13%

	Golf courses	8		Green space	S	
		Cumulative			Cumulative	
Sites	Mean $\alpha$ (± SE)	species	Sites	Mean $\alpha$ (± SE)	species	
		richness			richness	
BG	26.5 (± 1.5)	28	IB	54.5 (± 1.5)	58	
RM	39 (± 3.0)	45	BL	32.5 (± 0.5)	36	
IP	34 (± 0.0)	37	SW	32.5 (± 0.5)	38	
EG	33 (± 2.0)	38	TC	23 (± 0.0)	29	
AG	30.5 (± 2.5)	32	CP	27 (± 2.0)	32	
MG	$22.5 (\pm 0.5)$	25	MR	31 (± 1.0)	37	
Total	30.92 (± 2.4)	55	Total	33.42 (± 4.5)	66	
	Rank transfe	ormed Bray-Cu	rtis dista	nce (MRPP output	)	
	0.315		0.525			

Table 3. Species richness within and between golf courses and green spaces. Mean species richness is the average over the two surveys, while cumulative species richness is the total number of species detected over both surveys.

Table 4. Variation among site types (golf courses and green spaces) in mean species incidence for individual species. Species incidence is scaled from zero to 10 (0 = absent, 10 = present during all intervals). Indicator species analysis (ISA) *P*-values are based on Monte Carlo test of significance, with values ranging from zero to 100% (100 = presence of a species points to the group in question without error). Breeding Bird Survey (BBS) trends are represented as upward-facing arrows for significantly increasing species at P < 0.05, whereas significantly decreasing species are shown as downward-facing arrows. Non-significant trends are shown as dashes. Species are arranged in ascending order of the Mann-Whitney *P*-value.

Spacing norma	Scientific nome	Mean species in	Mean species incidence ( $\pm$ SE)			BBS	Breeding	PIF
Species name	Scientific name	Golf courses	Green spaces	<i>P</i> -value	<i>P</i> -value	trend	habitata	SRI <sup>b</sup>
(i) species showing significant d	ifferences in incidence between	n golf courses an	d green spaces					
Chipping Sparrow	Spizella passerina	7.33 (± 1.23)	0.67 (± 0.49)	0.005	0.005	$\downarrow$	Urban	-
Mourning Dove	Zenaida macroura	3.67 (± 0.71)	0.5 (± 0.34)	0.008	0.010	<b>↑</b>	Urban	-
American Crow	Corvus brachyrhynchos	$8.00 (\pm 0.68)$	3.83 (± 0.70)	0.010	0.011	<b>↑</b>	Wood	-
Song Sparrow	Melospiza melodia	9.33 (± 0.33)	$7.5 (\pm 0.50)$	0.014	0.017	-	Scrub	-
Pileated Woodpecker	Dryocopus pileatus	0.17 (± 0.17)	1.67 (± 0.56)	0.031	0.048	-	Wood	-
Barn Swallow	Hirundo rustica	3.50 (± 1.43)	0.33 (± 0.21)	0.037	0.046	$\downarrow$	Urban	Yes
Brown-headed Cowbird	Molothrus ater	$7.67 (\pm 0.88)$	4.5 (± 0.76)	0.037	0.040	$\downarrow$	Scrub	-
European Starling	Sturnus vulgaris	6.67 (± 1.09)	3.0 (± 0.89)	0.045	0.043	$\downarrow$	Urban	-
(ii) species showing no significat	nt differences in incidence betw	ween site types						
Black-and-white Warbler	Mniotilta varia	0 (± 0)	1.67 (± 0.92)	0.055	0.062	-	Wood	-
Ovenbird	Seiurus aurocapillus	$0(\pm 0)$	2.0 (± 1.06)	0.055	0.062	<b>↑</b>	Wood	-
Eastern Kingbird	Tyrannus tyrannus	5.0 (± 1.48)	1.33 (± 0.67)	0.066	0.100	$\downarrow$	Scrub	-
House Sparrow	Passer domesticus	$4.0 (\pm 0.77)$	1.67 (± 0.99)	0.078	0.065	$\downarrow$	Urban	-
Indigo Bunting	Passerina cyanea	0.33 (± 0.21)	$2.0 (\pm 0.68)$	0.109	0.115	-	Scrub	-
Gray Catbird	Dumetella carolinensis	2.5 (± 1.31)	4.83 (± 0.91)	0.128	0.112	-	Scrub	-
Yellow Warbler	Dendroica petechia	5.0 (± 1.46)	$8.0 (\pm 0.82)$	0.128	0.133	-	Scrub	-
Winter Wren	Troglodytes troglodytes	$0(\pm 0)$	0.67 (± 0.33)	0.150	0.187	$\uparrow$	Wood	-

Wood Thrush	Hylocichla mustelina	$0(\pm 0)$	1.5 (± 0.72)	0.150	0.187	$\downarrow$	Wood	Yes
American Robin	Turdus migratorius	9.67 (± 0.33)	8.83 (± 0.48)	0.173	0.285	-	Urban	-
Tree Swallow	Tachycineta bicolor	4.83 (± 1.9)	2.17 (± 1.38)	0.200	0.267	$\downarrow$	Wet	-
Cedar Waxwing	Bombycilla cedrorum	6.17 (± 0.75)	4.67 (± 1.12)	0.262	0.335	-	Scrub	-
Green Heron	Butorides virescens	0.17 (± 0.17)	$1.5 (\pm 0.81)$	0.262	0.304	$\downarrow$	Wet	-
House Wren	Troglodytes aedon	3.0 (± 1.67)	$0.5 (\pm 0.34)$	0.262	0.275	-	Scrub	-
Northern Flicker	Colaptes auratus	4.17 (± 0.79)	2.83 (± 0.75)	0.262	0.290	-	Scrub	Yes
Baltimore Oriole	Icterus galbula	5.50 (± 1.02)	4.17 (± 1.01)	0.298	0.435	-	Scrub	-
Scarlet Tanager	Piranga olivacea	0.17 (± 0.17)	$0.83 (\pm 0.48)$	0.298	0.410	-	Wood	-
Alder Flycatcher	Empidonax alnorum	$0(\pm 0)$	0.67 (± 0.42)	0.337	0.448	-	Scrub	-
Belted Kingfisher	Megaceryle alcyon	0.83 (± 0.65)	$0(\pm 0)$	0.337	0.466	-	Wet	Yes
Killdeer	Charadrius vociferus	0.33 (± 0.21)	$0(\pm 0)$	0.337	0.451	$\downarrow$	Scrub	-
Ruby-throated Hummingbird	Archilochus colubris	$0(\pm 0)$	0.67 (± 0.42)	0.337	0.454	-	Wood	-
Swamp Sparrow	Melospiza georgiana	$0(\pm 0)$	1.17 (± 0.75)	0.337	0.458	-	Wet	Yes
Veery	Catharus fuscescens	0.83 (± 0.54)	3.0 (± 1.51)	0.337	0.311	-	Wood	Yes
Yellow-bellied Sapsucker	Sphyrapicus varius	0.5 (± 0.22)	2.5 (± 1.28)	0.337	0.326	-	Wood	Yes
American Redstart	Setophaga ruticilla	$1.50 (\pm 0.85)$	3.50 (± 1.52)	0.378	0.369	$\downarrow$	Wood	-
Eastern Phoebe	Sayornis phoebe	0.33 (± 0.33)	0.83 (± 0.48)	0.378	0.441	-	Scrub	-
Eastern Wood-Pewee	Contopus virens	1.83 (± 0.91)	3.33 (± 1.41)	0.378	0.474	$\downarrow$	Wood	-
Warbling Vireo	Vireo gilvus	5.0 (± 1.03)	3.33 (± 0.84)	0.378	0.276	-	Wood	-
Black-capped Chickadee	Poecile atricapillus	8.33 (± 0.71)	8.33 (± 0.33)	0.423	1.000	↑	Wood	-
White-breasted Nuthatch	Sitta carolinensis	1.67 (± 0.49)	2.33 (± 0.61)	0.423	0.652	-	Wood	-
Blue Jay	Cyanocitta cristata	2.83 (± 0.87)	2.17 (± 1.08)	0.471	0.584	-	Urban	-
Common Yellowthroat	Geothlypis trichas	1.17 (± 0.60)	2.33 (± 1.12)	0.471	0.490	-	Scrub	Yes
Great Crested Flycatcher	Myiarchus crinitus	4.83 (± 1.11)	4.17 (± 0.79)	0.471	1.000	-	Wood	-
Red-eyed Vireo	Vireo olivaceus	6.33 (± 0.84)	7.17 (± 1.01)	0.471	0.583	↑	Wood	-
Brown Creeper	Certhia americana	0.17 (± 0.17)	2.0 (± 1.37)	0.522	0.448	-	Wood	-
Hairy Woodpecker	Picoides villosus	2.5 (± 0.81)	1.83 (± 0.91)	0.522	0.778	↑	Wood	-
Wood Duck	Aix sponsa	1.5 (± 0.96)	2.0 (± 0.93)	0.522	0.531	-	Wet	-
American Bittern	Botaurus lentiginosus	$0(\pm 0)$	0.17 (± 0.17)	0.631	n/a	-	Wet	-

Black Tern	Chlidonias niger	0 (± 0)	0.33 (± 0.33)	0.631	n/a	n/a	Wet	-
Black-throated Blue Warbler	Dendroica caerulescens	$0.83 (\pm 0.83)$	$0(\pm 0)$	0.631	n/a	<b>↑</b>	Wood	Yes
Black-throated Green Warbler	Dendroica virens	$0(\pm 0)$	0.17 (± 0.17)	0.631	n/a	1	Wood	Yes
Brown Thrasher	Toxostoma rufum	$1.0 (\pm 0.82)$	0.33 (± 0.33)	0.631	0.752	-	Scrub	Yes
Carolina Wren	Thryothorus ludovicianus	0.17 (± 0.17)	$0 (\pm 0)$	0.631	n/a	n/a	Scrub	-
Chestnut-sided Warbler	Dendroica pensylvanica	$0(\pm 0)$	$0.50 (\pm 0.50)$	0.631	n/a	-	Scrub	Yes
Cliff Swallow	Petrochelidon pyrrhonota	$0.50 (\pm 0.50)$	$0 (\pm 0)$	0.631	n/a	-	Urban	-
Common Grackle	Quiscalus quiscula	6.17 (± 1.47)	4.67 (± 0.76)	0.631	0.440	-	Urban	-
Common Moorhen	Gallinula chloropus	$0(\pm 0)$	$0.50 (\pm 0.50)$	0.631	n/a	n/a	Wet	-
Eastern Bluebird	Sialia sialis	0.17 (± 0.17)	$0 (\pm 0)$	0.631	n/a	-	Scrub	-
Least Bittern	Ixobrychus exilis	$0(\pm 0)$	$0.67 (\pm 0.67)$	0.631	n/a	n/a	Wet	-
Least Flycatcher	Empidonax minimus	$0(\pm 0)$	0.33 (± 0.33)	0.631	n/a	$\downarrow$	Wood	Yes
Marsh Wren	Cistothorus palustris	$0(\pm 0)$	$0.83 (\pm 0.83)$	0.631	n/a	n/a	Wet	-
Red-winged Blackbird	Agelaius phoeniceus	5.0 (± 1.75)	6.33 (± 1.48)	0.631	0.652	$\downarrow$	Wet	-
Sora	Porzana carolina	$0(\pm 0)$	$0.83 (\pm 0.83)$	0.631	n/a	n/a	Wet	-
Willow Flycatcher	Empidonax traillii	$0(\pm 0)$	0.17 (± 0.17)	0.631	n/a	-	Scrub	Yes
American Goldfinch	Carduelis tristis	8.50 (± 0.67)	$8.50 (\pm 0.48)$	0.689	1.000	↑	Scrub	-
Mallard	Anas platyrhynchos	$4.0 (\pm 1.0)$	3.17 (± 1.19)	0.749	0.616	↑	Wet	-
Rose-breasted Grosbeak	Pheucticus ludovicianus	1.67 (± 0.92)	$2.0 (\pm 0.82)$	0.749	0.752	$\downarrow$	Wood	Yes
Downy Woodpecker	Picoides pubescens	5.50 (± 0.76)	5.17 (± 0.98)	0.810	0.889	-	Wood	-
Hooded Merganser	Lophodytes cucullatus	$0.67 (\pm 0.67)$	0.33 (± 0.33)	0.936	1.000	n/a	Wet	-
House Finch	Carpodacus mexicanus	$0.83 (\pm 0.83)$	0.17 (± 0.17)	0.936	1.000	-	Urban	-
Pied-billed Grebe	Podilympus podiceps	1.17 (± 0.75)	$1.0 (\pm 0.68)$	0.936	1.000	-	Wet	-
White-throated Sparrow	Zonotrichia albicollis	$0.5 (\pm 0.5)$	0.33 (± 0.33)	0.936	1.000	-	Scrub	Yes
Northern Cardinal	Cardinalis cardinalis	5.83 (± 1.17)	5.83 (± 0.83)	1.00	1.000	-	Scrub	-
Red-breasted Nuthatch	Sitta canadensis	$1.0 (\pm 0.63)$	0.67 (± 0.33)	1.00	1.000	1	Wood	-

<sup>a</sup> Breeding habitat: scrub = successional forest/scrubland, wet = wetland, and wood = woodland.

<sup>b</sup> PIF SRI = Partners in Flight Species of Regional Importance for Bird Conservation Region 12.

Table 5. Number of species for each breeding habitat and trend classification from 12 sites in the Montreal area in 2006. Classifications are based on Breeding Bird Survey (BBS) data, with  $\uparrow$  and  $\downarrow$  categories indicating only those that were significant at *P* < 0.05. "PIF SRI" is the number Species of Regional Importance for each site, as defined by PIF. Mann-Whitney U tests were run between golf courses and green spaces for all categories. Golf course = GC; green space = GS.

Sites		Breeding habit	at		BBS trend		DIE CDI
51105	Urban	Successional /scrubland	Wetland	Woodland	1	$\downarrow$	FII' SKI
BG	8	10	3	7	6	8	2
RM	8	16	8	13	8	13	7
IP	8	10	5	14	8	10	7
EG	8	14	1	15	8	9	7
AG	10	10	4	8	6	9	2
MG	6	12	0	7	6	5	3
GC cumulative total	10	19	8	18	9	13	10
GC mean ± SE	$8\pm0.52$	$12 \pm 1.0$	$3.5 \pm 1.2$	$10.7\pm1.5$	$7\pm0.4$	$9 \pm 1.1$	$4.7 \pm 1.1$
IB	7	16	14	21	10	12	10
BL	4	14	3	15	8	7	5
SW	6	13	6	13	7	9	6
TC	3	12	0	14	6	4	4
СР	8	11	4	9	6	8	2
MR	3	10	2	22	8	7	7
GS cumulative total	9	19	14	24	11	14	14
GS mean ± SE	$5.2\pm0.9$	$12.7 \pm 0.9$	$4.8 \pm 2.0$	$15.7\pm2.0$	$7.5\pm0.6$	$7.8 \pm 1.1$	$5.7 \pm 1.1$
Cumulative total	10	22	15	25	12	15	16
Mann-Whitney U	4.5	14	16.5	7.5	15	11.5	15.5
( <i>P</i> -value) <sup>a</sup>	(0.024)	(0.512)	(0.809)	(0.090)	(0.604)	(0.293)	(0.681)

<sup>a</sup> Significant *P*-values from the Mann-Whitney U tests are bolded.

	Sites	SIZE (ha)	HOUSE (#)	VGWATER	NVWATER	UNUSABLE	Tree	CONIFER	GRASS	PASTURE	Forest (km)	WATER (km)
	MG	56.2	8.58	0.0	0.2	5.3	22.7	0.2	61.7	2.1	7.6	3.2
rses	BG	64.7	11.7	1.4	0.2	12.8	23.3	4.3	56.6	2.8	4.6	1.2
Cou	IP	68.8	1.32	1.9	0.1	4.9	29.4	0.4	51.4	0.9	0.4	0.8
olf (	AG	104.5	1.52	6.1	8.8	7.2	15.7	2.7	59.9	1.2	3.2	0.6
Ğ	EG	162.5	0.28	0.3	0.8	6.7	20.4	8.3	52.2	3.2	1.7	0.9
	RM	215.2	0.25	2.3	0.3	4.4	24.1	6.1	49.7	7.5	1.5	1.0
	Mean	111.9	3.94	2.0	1.7	6.7	22.6	3.7	55.3	3.0	3.2	1.3
	(± SE)	(± 26.1)	(± 2.01)	(± 0.9)	(± 1.4)	(± 1.3)	(± 1.8)	(± 1.3)	(± 2.0)	(± 2.4)	(± 1.1)	(± 0.4)
	SW	21.9	0.46	9.6	0.0	0.1	65.4	2.0	0.0	19.6	0.2	2.2
	TC	33.8	10.63	0.0	0.6	2.1	67.2	0.0	9.7	8.6	5.1	1.0
lces	CP	49.9	12.13	20.2	0.0	6.7	42.7	0.3	13.7	7.4	2.9	3.0
Spi	MR	73.0	3.55	16.3	0.0	2.1	71.6	1.8	0.0	0.0	1.5	0.5
een	BL	128.5	4.54	2.6	0.0	4.1	79.2	0.7	1.6	7.4	0.0	1.5
Ğ	IB	217.6	2.22	7.5	0.0	3.3	73.3	7.8	0.3	7.8	0.0	1.1
-	Mean	87.4	5.59	9.4	0.1	3.1	66.6	2.1	4.2	8.5	1.6	1.6
	(± SE)	(± 30.2)	(± 1.93)	(± 3.2)	(± 0.1)	(± 1.2)	(± 5.2)	(± 1.2)	(± 2.4)	(± 2.6)	(± 0.8)	(± 0.4)
(P	M-W -value) <sup>a</sup>	13.0 (0.423)	11.0 (0.262)	6.5 (0.066)	4.0 (0.025)	4.0 (0.025)	0.0 (0.004)	12.0 (0.337)	0.0 (0.004)	8.0 (0.109)	9.5 (0.173)	13.0 (0.423)

Table 6. Landscape-level habitat variables for each study site, plus each mean ( $\pm$  SE) and associated Mann-Whitney U test statistics (*P*-value). Unless otherwise indicated, all values are %.

<sup>a</sup> Significant *P*-values from the Mann-Whitney U tests are bolded.



Fig. 2. Three-dimensional non-metric multidimensional scaling ordination of 12 sites plotted according to their avian communities. Filled circles represent golf courses, empty circles represent green spaces.





Fig. 3. Non-metric multidimensional scaling ordination of 12 sites in species space, split into a two-dimensional pairwise plot. Joint plot vectors are overlaid for landscape variables that have an  $r^2 > 0.2$ . The angle and length of each line represents the direction and strength of the relationship. Filled circles represent golf courses, empty circles represent green spaces. A) Axis 1 vs. Axis 2; B) Axis 2 vs. Axis 3; C) Axis 1 vs. Axis 3.

Table 7. Pearson correlation coefficients (r) of landscape variables with respect to each NMS axis. Multiple  $r^2$  is also presented as the overall measure of correlation of the variable with the entire ordination. Variables are arranged in descending order of multiple  $r^2$ .

	Axis 1	Axis 2	Axis 3	Multiple $r^2$
SIZE	0.220	-0.772	-0.641	0.726
HOUSE	0.195	0.758	0.583	0.669
CONIFER	0.041	-0.750	-0.593	0.663
GRASS	-0.721	-0.489	0.193	0.573
VGWATER	0.295	0.142	-0.711	0.513
TREE	0.673	0.263	-0.410	0.488
FOREST	-0.397	0.307	0.619	0.424
NVWATER	-0.621	-0.248	0.012	0.391
UNUSABLE	-0.400	-0.481	0.118	0.281
PASTURE	-0.066	0.091	-0.335	0.112
WATER	-0.032	0.166	-0.127	0.032

Appendix 1. Pairwise comparisons of four similarity coefficients calculated between avian species found breeding on six Montreal-area golf courses and six green spaces. See Table 1 for site codes. Values range from zero, indicating no similarity, to one, indicating perfect similarity.

Site	Site	Shared	Inconrd	Saronson	Morisita-	Bray-
1	2	species	Jaccalu	Sørensen	Horn	Curtis
RM	EG	34	0.694	0.819	0.850	0.734
RM	IP	34	0.708	0.829	0.883	0.758
IP	EG	31	0.705	0.827	0.807	0.712
RM	AG	29	0.604	0.753	0.824	0.724
BG	RM	27	0.587	0.740	0.826	0.678
IP	AG	27	0.643	0.783	0.827	0.707
BG	AG	26	0.765	0.867	0.821	0.687
BG	IP	26	0.667	0.800	0.835	0.694
EG	AG	25	0.556	0.714	0.733	0.612
RM	MG	25	0.556	0.714	0.698	0.580
BG	EG	24	0.571	0.727	0.828	0.665
EG	MG	23	0.575	0.73	0.756	0.644
IP	MG	21	0.512	0.677	0.722	0.617
AG	MG	19	0.500	0.667	0.679	0.586
BG	MG	19	0.559	0.717	0.782	0.664
Golf course min - n		min mov	0.500 -	0.667 -	0.679 -	0.580 -
		mm - max	0.765	0.867	0.883	0.758
IB	SW	36	0.600	0.750	0.721	0.616
IB	MR	35	0.583	0.737	0.741	0.616
IB	BL	31	0.492	0.660	0.752	0.622
IB	CP	29	0.475	0.644	0.650	0.550
IB	TC	29	0.500	0.667	0.668	0.518
BL	MR	27	0.587	0.740	0.788	0.669
BL	SW	27	0.574	0.730	0.791	0.637
SW	CP	26	0.591	0.743	0.806	0.686
SW	MR	26	0.531	0.693	0.685	0.562
TC	MR	24	0.571	0.727	0.677	0.553
BL	CP	23	0.511	0.676	0.819	0.662
BL	TC	23	0.548	0.708	0.840	0.664
SW	TC	22	0.489	0.657	0.753	0.595
CP	MR	21	0.438	0.609	0.697	0.580
TC	CP	20	0.488	0.656	0.821	0.681
Creation			0.438 -	0.609 -	0.650 -	0.518 -
Greei	space	mm - max	0.600	0.750	0.840	0.686
RM	IB	41	0.661	0.796	0.726	0.621
EG	IB	36	0.600	0.750	0.704	0.605
IP	IB	34	0.557	0.716	0.712	0.617
RM	SW	34	0.694	0.819	0.773	0.648

EG	SW	31	0.689	0.816	0.739	0.643
IP	SW	30	0.667	0.800	0.851	0.726
RM	CP	29	0.604	0.753	0.737	0.622
AG	CP	28	0.778	0.875	0.749	0.643
AG	IB	28	0.452	0.622	0.603	0.503
RM	BL	27	0.500	0.667	0.756	0.619
BG	IB	26	0.433	0.605	0.648	0.537
EG	CP	26	0.591	0.743	0.720	0.605
IP	MR	26	0.542	0.703	0.668	0.555
RM	MR	26	0.464	0.634	0.599	0.494
AG	SW	25	0.556	0.714	0.740	0.627
BG	CP	25	0.714	0.833	0.827	0.722
BG	SW	25	0.610	0.758	0.783	0.662
EG	BL	25	0.510	0.676	0.805	0.663
EG	MR	25	0.500	0.667	0.650	0.534
IP	CP	25	0.568	0.725	0.761	0.645
RM	TC	25	0.510	0.676	0.714	0.533
EG	TC	24	0.558	0.716	0.800	0.620
MG	IB	24	0.407	0.578	0.638	0.543
IP	BL	23	0.460	0.630	0.758	0.619
IP	TC	22	0.500	0.667	0.727	0.557
MG	SW	22	0.537	0.698	0.712	0.597
AG	BL	21	0.447	0.618	0.662	0.533
MG	BL	21	0.525	0.689	0.835	0.707
AG	MR	20	0.408	0.580	0.543	0.450
BG	BL	20	0.455	0.625	0.782	0.651
BG	MR	19	0.413	0.585	0.579	0.473
MG	CP	19	0.500	0.667	0.792	0.669
AG	TC	18	0.419	0.590	0.603	0.459
MG	MR	18	0.409	0.581	0.637	0.536
MG	TC	18	0.500	0.667	0.832	0.632
BG	TC	17	0.425	0.596	0.815	0.641
т	otal mir	i - may	0.407 -	0.578 -	0.543 -	0.450 -
1		г - Шал	0.778	0.875	0.883	0.758

Appendix 2.  $R^2$  values from simple linear regressions between landscape variables and estimates of species richness, and correlation coefficients between all landscape variables used in analyses. Significant regressions and correlations at P < 0.05 are bolded.

Variables	$R^2$	SIZE	TREE	CONIFER	GRASS	NVWATER	VGWATER	HOUSE	UNUSABLE	PASTURE	FOREST
SIZE	0.43	-	-	-	-	-	-	-	-	-	-
TREE	0.08	-0.17	-	-	-	-	-	-	-	-	-
CONIFER	0.44	0.68	-0.22	-	-	-	-	-	-	-	-
GRASS	0.12 <sup>a</sup>	0.21	-0.96	0.11	-	-	-	-	-	-	-
NVWATER	0.03 <sup>a</sup>	0.15	-0.56	0.12	0.54	-	-	-	-	-	-
VGWATER	0.10	-0.09	0.34	0.05	-0.55	-0.19	-	-	-	-	-
HOUSE	<b>0.45</b> <sup>a</sup>	-0.42	0.12	-0.58	-0.02	-0.19	-0.05	-	-	-	-
UNUSABLE	$0.08^{a}$	0.39	0.66	0.20	0.72	0.33	-0.23	0.35	-	-	-
PASTURE	0.02	-0.25	0.37	0.02	-0.40	-0.30	0.03	-0.08	-0.47	-	-
FOREST	0.01	-0.13	0.14	0.04	-0.14	-0.16	-0.17	-0.02	-0.21	0.94	-
WATER	$0.00^{a}$	-0.05	-0.07	0.04	0.11	-0.01	-0.31	-0.00	0.00	0.81	0.96

<sup>a</sup> Indicates a negative slope.

# **CONNECTING STATEMENT 2**

It is clear that various avian communities are using the suburban sites (i.e. golf courses and green spaces) described in Chapter 2 during the breeding season, and that their presence on a given site is affected by the habitat characteristics inherent to each site. However, simple presence or absence does not provide any indication of site quality in terms of its ability to provide adequate breeding habitat. Intensive nest-monitoring was done on a subset of these 12 sites (four golf courses and two green spaces) to compare nest success rates between site types. In this chapter, I describe the novel nest-monitoring equipment that I used to track the development of almost 900 nests over three years, creating a permanent record of my work while minimizing disturbance to nesting birds.

Chapter 3

AN AFFORDABLE COMPUTERIZED CAMERA TECHNIQUE FOR MONITORING BIRD NESTS

Hudson, M.-A. R. and D. M. Bird. 2006. An affordable computerized camera technique for monitoring bird nests. Wildlife Society Bulletin 34: 1455-1457.

# Abstract

We describe a computerized nest monitoring unit that uses a small camera (commonly marketed as a "webcam") mounted on a telescopic pole to relay digital video and still images to a portable laptop computer carried by the operator. This system captures and archives digital colour images of open-cup nesting birds, enabling the operator to determine egg type (host or brood parasite) and number, as well as nestling developmental stages. These archived photographs represent a permanent record that may be revisited and re-used as many times as needed. This system allows researchers to easily create and maintain their own monitoring equipment using readily available materials at very low cost.

Until recently, studies focusing on avian breeding biology relied on monitoring elevated bird nests with pole-mounted mirrors (Best and Stauffer 1980, Martin and Geupel 1993, Ralph et al. 1993). A variety of systems are now available that allow researchers to video-tape or photograph the contents of an open-cup or cavity nest (Ouchley et al. 1994, Proudfoot 1996, Purcell 1997, McQuillen and Brewer 2000, King et al. 2001). These systems suffer from two drawbacks: they can be quite expensive (pre-assembled units can cost roughly US \$4,000-\$6,000); and most are only able to transmit images to a small monitor. Typically, images are recorded by a VCR, which can be difficult to operate and power in the field. Digital surveillance cameras are also available, however they can be expensive and must be left in place until monitoring is complete. The computerized camera technique described here is similar to these commercially available video-monitoring systems, but its principal components have been modified to drastically reduce cost and to increase flexibility of use in the field. Our system allows researchers to easily create and maintain their own monitoring equipment using readily available materials and to quickly produce and archive both digital photographs and video.

This system was developed for a study examining the reproductive success of open cup-nesting passerines (e.g. American Robin *Turdus migratorius*, Redwinged Blackbird *Agelaius phoeniceus*, and Gray Catbird *Dumetella carolinensis*) on golf courses and parks around Montreal, Quebec, Canada in 2003-2005. Breeding habitats used by these species include tracts of mixed deciduous forest, mixed and single conifer stands, marshes and pond edges, and various native and ornamental shrubs.

### MATERIALS AND METHODS

To create the system, a 108-g camera, more commonly marketed as a "webcam" (i.e. Creative WebCam Pro, Creative Technology Ltd., Singapore), was bolted into plastic, weather-resistant housing (e.g. peanut butter jar with lid) slightly larger than the camera itself. Holes were cut in the housing to accommodate the camera's Universal Serial Bus (USB) cable and sealed with

weather-resistant silicone. The camera housing was then attached to a pole by tightly wrapping an aluminum sleeve around the housing, creating a flange. This flange inserted into a notch cut into the handle of a paint-roller holder (Fig. 1). The entire assembly was screwed onto the end of a 5.4-m telescopic painter's pole (Mr. LongArm International, Greenwood, MO). A 4.5-m USB extension was added to the existing 1.8-m camera cable so that nests up to 8 m above ground could be reached when the base of the pole was held at shoulder height. The connection between the USB extension and the camera was weatherproofed using shrink tubing and silicone grease. The USB cable connected to the laptop via the USB port and threaded along loops on the pole to prevent tangling when the pole was extended. When not in use, the cable was coiled and fastened onto the pole using Velcro strips.

A homemade clip-on tray attached to a backpack enabled hands-free nest monitoring (Fig. 2). Sheet aluminum was cut according to the computer's dimensions and bent upward on all four sides to create an edge. The tray was attached to the straps of the backpack using webbing and plastic clips. The clips allowed the tray to be detached from the backpack for storage when not in use. Though nest monitoring was never done in the rain, the backpack was lined with plastic sheeting to limit exposure to moisture. With the laptop cover closed, the tray could be held against the chest to facilitate passage through dense vegetation or used as a writing table.

Construction and maintenance of the camera system cost approximately US \$150 (excluding the laptop). Any laptop running the Windows 98 operating system or above, with a processor running at 266 MHz and 64 MB of RAM, can be used. The only software required was the Creative PC-CAM Center version 1.22.01, which came with the camera. The program enabled one-touch photography or videography, allowed custom archiving of both still images and video, and provided the basic components for photo editing. The latter was useful if an image's brightness, contrast or sharpness required adjustment, especially on bright days. The only maintenance performed on the system during the three-year study was the replacement of the USB extension due to wear from repeatedly
connecting and disconnecting the camera to the computer, and the reapplication of silicone to ensure continued water-resistance. For maximum durability, we suggest soldering the USB extension to the USB port of the laptop.

The laptop's rechargeable battery provided approximately 3 hours of power to both the computer and the camera, so no external batteries were required. Researchers working in remote areas could extend battery life by purchasing a second battery or by recharging the battery using a 12-V inverter plugged into a vehicle's power outlet or solar panel.

### **RESULTS AND DISCUSSION**

We used this computerized camera unit to view and archive the contents of 873 nests in 2003-2005, resulting in over 3,300 stored images. We were unable to view the contents of 17 nests, all located over 8 m in the tree canopy (n = 15) or in cattails over 8 m from shore (n = 2). The camera's adjustable focus ring (focal range of 8 cm to infinity) allowed us to obtain an overview of the entire nest and surrounding vegetation or a close-up of the nest's contents. We used colour imaging to examine the incidence of brood parasitism by comparing egg type and colour and to document gape colour and feather development of nestlings (Fig. 3).

This system has several advantages over the conventional mirrored-pole. The camera worked very well for monitoring nests under low-light conditions. Coniferous trees and shrubs, a prominent feature on our study sites, are very dense, making reflected images dark. A light-emitting diode (LED) attached to the camera increased visibility and colour detection. However, an umbrella or sunshade was required on sunny days to reduce the amount of light hitting the computer screen. Visibility was greatest when a shadow was cast over the screen.

Unlike a mirrored-pole, our nest monitoring technique allowed observers to quickly save an image or video clip for later review (visits ranged from 20 seconds to 4 minutes) and to leave with minimal site disturbance. The ability to detect nestling movement and to adjust contrast and brightness greatly facilitated the counting of huddled nestlings. The hinge that fixes the camera to the pole also proved very useful, as it allowed the camera to be used both vertically for elevated nests, and horizontally. Cattails and reeds (*Typha* and *Phragmites* spp.) are challenging for nest monitoring, as dense patches completely block images reflected by a mirrored-pole, and are then easily crushed underfoot. The camera can be tilted downwards directly over the nest, relaying the image to the computer without interference. The use of the pole horizontally, as opposed to vertically, also allows the observer to check a nest's contents from afar, reducing human disturbance to vegetation. This is an important advantage, as it has been suggested that predators may use researcher's trails to locate potential prey (Martin and Geupel 1993).

This system can be expanded to reach nests above 8 m by using either a ladder or 5-m USB active extension cables. When paired with a longer pole and connected in series, 5 USB active extension cables (the maximum recommended to ensure signal quality is maintained) can increase the range up to 25 m. Longer poles are available (e.g. the Universal Telescopic Pole System by Exel Industry can reach up to 20 m), but they are more expensive and can be difficult to maneuver safely and comfortably when fully extended.

The most valuable aspect of this technique, aside from its affordability, is the ability to store images. Properly labeled and dated images of nests, eggs, nestlings at various stages of development and adults are important to archive as voucher samples (Wheeler 2003). This reduces the potential for observer bias by allowing any number of observers to review the images (Cutler and Swann 1999) and provides lasting information on the species studied, nest structure, clutch size and date of nest initiation. While this system cannot be used in lieu of more expensive, continuous video surveillance (see McQuillen and Brewer 2000 and King et al. 2001) for research focusing on nest predation, it does provide an affordable means of digitally recording nest contents. In the future, use of a personal digital assistant (PDA) may become a viable alternative to a laptop computer, greatly reducing the size and increasing the portability of our system. To our knowledge however, the imaging software required to make the PDA and camera compatible does not exist at this time.

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### LITERATURE CITED

- Best, L. B. and D. F. Stauffer. 1980. Factors affecting nesting success in riparian bird communities. Condor 82: 149-158.
- Cutler, T. L. and D. E. Swann. 1999. Using remote photography in wildlife ecology: a review. Wildlife Society Bulletin 27: 571-581.
- King, D. I., R. M. DeGraaf, P. J. Champlin, and T. B. Champlin. 2001. A new method for wireless video monitoring of bird nests. Wildlife Society Bulletin 29: 349-353.
- Martin, T. E. and G. R. Geupel. 1993. Nest-monitoring plots: methods for locating nests and monitoring success. Journal of Field Ornithology 64: 507-519.
- McQuillen, H. L. and L. W. Brewer. 2000. Methodological considerations for monitoring wild bird nests using video technology. Journal of Field Ornithology 71: 167-172.
- Ouchley, K., R. B. Hamilton, and S. Wilson. 1994. Nest monitoring using a micro-video camera. Journal of Field Ornithology 65: 410-412.
- Proudfoot, G. A. 1996. Miniature video-board camera used to inspect natural and artificial nest cavities. Wildlife Society Bulletin 24: 528-530.
- Purcell, K. L. 1997. Use of a fiberscope for examining cavity nests. Journal of Field Ornithology 68: 283-286.
- Ralph, C. J., G. R. Geupel, P. Pyle, T. E. Martin, and D. F. DeSante. 1993.Handbook of Field Methods for Monitoring Landbirds. United StatesForest Service, Pacific Southwest Research Station, Albany, CA.

Wheeler, T. A. 2003. The Role of Voucher Specimens in Validating Faunistic and Ecological research. Document series no. 9. Biological Survey of Canada (Terrestrial Arthropods), Ottawa, ON.



Fig. 1. Creative WebCam Pro (Creative Technology, Ltd., Singapore) fastened to the inside of a weather-resistant plastic container. The aluminum flange wrapped around the housing provides a single-point attachment to a paint-roller handle, allowing the camera assembly to tilt forward or backward as needed. Photo by M.-A. R. Hudson.



Fig. 2. The camera is attached to an extendible pole (Mr. LongArm International, Greenwood, MO) and is connected to the laptop computer by a USB extension. A lightweight aluminum tray that is clipped to the backpack worn by the researcher supports the computer and enables hands-free operation. Photo by M.-A. R. Hudson.



Fig. 3. Sample images of various species' nests taken in Montreal, Quebec, Canada. Images were modified from colour to black-and-white for reproduction purposes. A) American Robin eggs during incubation, B) a Gray Catbird egg in the process of hatching, C) Red-Winged blackbird nestlings soon after hatching, and D) American Robin nestlings about to fledge (13 days old). All photos by M.-A. R. Hudson.

## **CONNECTING STATEMENT 3**

Acquiring reliable and permanent records of nests from initiation through to fledging or failure allows the calculation of nest survival rates. Comparing rates between sites is a well-supported way to detect differences in site quality. In previous chapters, I identified the species breeding on golf courses and on green spaces, and outlined a method which allowed the capture of thousands of images during the nesting period. In this chapter, I use information derived from this work to calculate and compare nest survival rates of open-cup nesting passerines breeding on golf courses and green spaces in the Montreal area. I also identify the factors that affect the nest density and predation rates of various species, and suggest management strategies designed to maximize breeding success on these sites.

# CHAPTER 4

# FACTORS INFLUENCING SONGBIRD NEST SURVIVAL ON MONTREAL GOLF COURSES AND GREEN SPACES

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### ABSTRACT

Recent studies have examined the ability of golf courses to provide breeding habitat for cavity-nesting species. However, we know very little about the factors influencing the nest survival of open-cup nesting passerines on golf courses, in comparison to other urban green spaces. We studied breeding birds on four golf courses and two green spaces in the greater Montreal area, Quebec, Canada, from 2003-2005. We compared nesting density and Mayfield survival rates for nine species nesting on these sites and examined productivity data and nest-site characteristics for five of the most common open-cup nesting passerines (American Robin Turdus migratorius, Gray Catbird Dumetella carolinensis, Cedar Waxwing Bombycilla cedrorum, Northern Cardinal Cardinalis cardinalis and Red-winged Blackbird Agelaius phoeniceus). We also identified the temporal, landscape- and micro-habitat characteristics influencing nest survival rates of the two most common species (American Robin and Red-winged Blackbird), as well as for all upland-species pooled. We found species-specific differences in nest success rates between golf courses and green spaces. Modelling indicated that nest survival for upland-species was influenced by year, the type of nesting substrate and its arrangement, and the interaction of these two variables. Our results also showed that sites embedded in a suburban matrix had lower predation rates than sites surrounded by forest or agriculture. American Robin nest success was not influenced by any of the variables we modelled, whereas Red-winged Blackbird nest success was influenced by nest age and nest concealment. Suburban green spaces, including golf courses, can play a role in maintaining healthy bird populations, if managers follow some simple recommendations to help increase nest success: avoid planting isolated trees or shrubs, increase the amount of emergent aquatic vegetation in ponds, and avoid landscaping (e.g. hedgetrimming) during the breeding season. Though further investigation is required, we also recommend that if new golf courses are to be built, they should be as natural as possible and restricted to urbanized areas. This would not only improve aesthetics, but also provide safe nesting grounds for many open-cup nesting passerines.

The most frequent cause of nest failure is nest predation (Ricklefs 1969). Researchers have studied the effects of urbanization on nest predation to understand how to manage or mitigate this pivotal component of avian survival in an increasingly urbanized world. These studies have produced mixed results, some finding support for a relaxation effect due to a reduction in predator abundance in urban areas (Gering and Blair 1999), and some reporting an increase in predation pressure due to higher avian predator abundance (Jokimäki and Huhta 2000) and the introduction of non-native or domestic predators (Wilcove 1985, Grandmaison and Niemi 2007). Others have found no effect at all (Melampy et al. 1999, Morrison and Bolger 2002).

It was once thought that increased fragmentation, decreased patch size and increased edge habitat would lead to increased nest predation through increased predator abundance due to mesopredator release (Soulé et al. 1988), a response to increased concentrations of nesting birds (Heske et al. 2001), or food subsidies available in human-modified landscapes (Schmidt 1999, Haskell et al. 2001). It is now clear that patterns cannot be generalized across all landscapes since effects depend on the type of habitat in which sites are embedded (Tewksbury et al. 1998, Melampy et al. 1999, Heske et al. 2001). These "conditional effects" (sensu Tewksbury et al. 2006) may be due to differences in the three main predator communities (birds, mammals and snakes) and differences in the way they respond to fragmentation (Heske et al. 2001, Patten and Bolger 2003). For example, Tewksbury et al. (2006) found support for an additive predation model which hypothesized that predation rates are highest at sites buffered by natural habitat but embedded within a largely agricultural area, suggesting that nest predation rates were the product of both local forest-dwelling specialists and generalist predators associated with agriculture and able to cross between different habitats.

Very few studies focusing on nest predation in an urban/suburban environment have examined the role that golf courses play in supporting healthy breeding bird populations. As of 2005, there were over 31,500 golf courses worldwide (Tanner and Gange 2005). Based on the size of an average 18-hole

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golf course (54 ha; Balogh et al. 1992), the total area occupied by golf courses represents just under two million hectares. Given that out-of-play areas cover roughly 60% of the total area (Green and Marshall 1987), they represent a vast amount of potential habitat largely lying within urban/suburban areas. Thus, they can be managed to act as a reserve if the sites are large enough, and/or movement corridors or stepping-stones for wildlife depending on the surrounding landscape (Shafer 1995).

Recent work has focused on the ability of golf courses to support avian species of conservation concern (Terman 1997, LeClerc and Cristol 2005, Smith et al. 2005, Sorace and Visentin 2007), and to provide habitat for varying levels of avian species richness and abundance in different ecoregions (Green and Marshall 1987, Moul and Elliott 1992, Blair 1996, Terman 1997, Merola-Zwartjes and DeLong 2005). Virtually all of these studies have emphasized that productivity studies are required to improve our understanding of these systems. The few studies that have directly examined golf course productivity have focused on cavity-nesting species (LeClerc et al. 2005, Stanback and Seifert 2005), which generally suffer from lower predation pressure than open-cup nesting species (Martin 1993, 1995). Though direct examination is lacking, it has been suggested that predation rates are higher on or near golf courses than on non-golf habitats (Sorace and Visentin 2007, Kus et al. 2008).

Thus, our objectives were to: i) compare nesting density and nest survival rates between golf courses and other suburban green spaces; ii) compare initiation dates, clutch sizes, egg hatchability, and overall fledgling production for open-cup nesting passerines using these sites; iii) compare nest-site micro-habitat variables at successful and depredated nests of these species; iv) identify the temporal, landscape and micro-habitat characteristics influencing nest survival rates in an information-theoretic framework; and v) identify the main nest predators at these sites.

We examined the effects of four types of factors that influence nest survival using an information-theoretic framework (Burnham and Anderson 2002): temporal, site, nest-patch and nest-site levels. First, we hypothesized that temporal factors affect nest survival through annual variation (Dinsmore et al. 2002), seasonal effects (Vierling 2000, Grant et al. 2005, but see Filliater et al. 1994), and nest age (i.e. from nest initiation through incubation, hatching and fledging; Grant et al. 2005). Second, we examined site-level factors to ascertain whether nest survival varies according to site and landscape context. We hypothesized that golf courses have lower survival rates due to increased predation (Sorace and Visentin 2007, Kus et al. 2008), and that sites embedded in an urban matrix have lower survival rates due to higher predation rates (Jokimäki et al. 2005). Third, we examined the effects of nest-patch and nest-site factors. Many studies have examined the effects of nest-site vegetation on nest survival, but few have consistently found what effect, if any, these characteristics have on survival (Filliater et al. 1994, Smith et al. 2007). We hypothesized that substrate types and arrangements affect survival, with larger, more complex vegetation allowing nests to escape detection by predators by camouflaging auditory, visual or chemical cues, or by increasing the number of potential nest sites (Martin and Roper 1988). We predicted that the nest-site variables we measured (e.g. disturbance levels, nest height, nest cover, distance to foliage edge) would affect survival, but that the direction of the influence would depend on the composition of the predator community (avian vs. mammalian; Best and Stauffer 1980, Wilcove 1985, Mayer-Gross et al. 1997, Rangen et al. 1999, Miller and Hobbs 2000). For example, nests placed high in trees with little cover have higher chances of being depredated by avian predators, but lower chances of being depredated by mammals.

### MATERIALS AND METHODS

*Study area.* -- This study was carried out on six sites in the greater Montreal area (hereafter Montreal), Quebec, Canada, in 2003-2005 (45° 25'N, 73°56'W; Fig. 1). Two Audubon-certified golf courses, two non-certified courses and two green spaces were searched intensively for nests from early April to late August each year. The Audubon certification program consists of six certificates of merit (chemical use reduction and safety, water conservation, water quality

management, outreach and education, wildlife and habitat management and environmental planning), which are awarded to golf courses that meet the requirements of each certificate. The golf course is awarded the status of "sanctuary" once all six are obtained. Though we had initially intended to look for differences between Audubon-certified and non-certified courses, we discovered during field work that golf course managers on the non-certified courses we studied were following the same principles as Audubon-certified courses without applying to the program. Since evaluating certified vs. non-certified would provide little insight into potential benefits provided by the program, we compared golf courses to green spaces, regardless of certification. Each site's landscape context is briefly described in Table 1.

All four golf courses (Royal Montreal Golf Club, Elm Ridge Country Club, Ile Perrot Golf and Country Club and Beaconsfield Golf Club; Table 1) were characterized by similar vegetative communities. They were dominated by highly maintained turfgrasses and lesser-maintained rough areas (e.g. Kentucky and annual bluegrasses, Poa pratensis and P. annua, and bentgrass, Agrostis spp.). Tree cover was variable: mixed deciduous forest tracts composed mainly of maple (Acer spp.), elm (Ulmus spp.) and oak species (Quercus spp.) with brushy understory at edges, small deciduous stands without understory, and single trees. There were variable numbers of coniferous trees, both native and introduced (e.g. spruce [Picea spp.], pine [Pinus spp.], cedar [Thuja spp.], and junipers [Juniperus spp.]). Shrub species such as buckthorn (*Rhamnus* spp.), honeysuckle (*Lonicera* spp.), hawthorn (*Crataegus* spp.), sumac (*Rhus* spp.) and dogwood (*Cornus* spp.) were found mainly along forest edges or in the rough areas bordering them. There were also variable numbers of apple and crab-apple trees (*Malus* spp.) on all sites, ranging from individual trees to small orchards. Each golf course had a number of artificial vegetated and/or non-vegetated ponds (dominated by Typha spp. and Phragmites australis). Beaconsfield Golf Club differed from the others as it was bisected by a major highway and railway track, so the north and south portions of the course were separated, linked only by a tunnel. Stoneycroft Wildlife Area, a private research area, had a large deciduous forest tract mainly composed of maple bordering the site to the west and north. The rest of the site was covered by a well-vegetated pond and wetland (dominated by both *Typha* and *Salix* spp., with some *Phragmites australis* and *Iris pseudacorus*), a pasture dominated by hawthorn, buckthorn, and goldenrod (*Solidago* spp.), and two small coniferous tree patches of mainly cedar, spruce and firs (*Abies* spp.). Terra Cotta Park, a recreational nature park, was dominated by a maple forest, with open areas of pasture with small hawthorn and buckthorn trees leading into dense hawthorn stands. This was the only site with very little water, the lone source being a small artificial ravine fed by a storm drain that remained dry for most of the season.

Efforts were made to ensure equal coverage of each site in terms of observer hours, though some non-significant differences did occur (mean 628.0 hrs  $\pm$  35.0 SE, range 462.8–688.3 hrs;  $F_{(5, 12)} = 1.32$ , P = 0.32). Observer hours do not necessarily reflect search hours, but overall searching and monitoring effort. Thus, the overall bias potentially introduced is considered negligible, since the differences were due to slower movement through thick brush stands on certain sites, and large colonies of nesting Red-winged Blackbirds (*Agelaius phoeniceus*) on certain sites, which required additional time to monitor.

*Nest searching and monitoring.* -- Nest searching and monitoring was conducted by three or four observers between approximately 0500 and 1600 EST. Nests were located both by systematically searching the area and cueing in on adult behaviour such as carrying food or nesting-material, or vocalization (Martin and Geupel 1993, Rodewald 2004). Two study sites were visited each day, with visit sequence alternating to ensure a morning visit once per week for each site. Since birds tend to be more active in the morning, intensive searches were often conducted in the afternoon while behaviour-based searches were more productive in the morning. Nest sites were never flagged. They were entered into a GPS unit (Garmin Ltd., Software Version 2.09) and plotted on an aerial photograph. This was to avoid both possible predator attraction (Picozzi 1975) as well as human curiosity on these heavily frequented sites. We monitored nests every 2-5 days until they either fledged at least one young or failed. Nest monitoring was done with a small colour camera attached to an extendible pole (described in Hudson and Bird 2006; Chapter 3). The camera instantly relayed images or video to a laptop computer carried by the operator, allowing photos or video to be easily captured and archived with minimal disturbance.

Nest-site measurements. -- Landscape-level variables associated with each site were measured following the methods outlined in Hudson and Bird (2009; Chapter 2). Sites were categorized based on land use (LOCATION; golf course or green space), and the number of buildings within a 200-m buffer surrounding each site was counted and divided by site area (HOUSE) to provide an index of urbanization. Habitat structure at each nest site was measured once the nest fledged or failed. Nest-site characteristics included: identification of nest substrate to species (SUBSTRATE), arrangement of substrate (ARRANGEMENT, e.g. single, linear, patch [< 5 trees or  $< 3 \text{ m}^2$  of reeds], or stand [> 5 trees, or  $> 3 \text{ m}^2$  of reeds]); nest height (HEIGHT, m); nest concealment (COVER, %), distance to the nearest area of disturbance (m); type of disturbance; and distance to nearest break in foliage over 1 m at nest height (EDGE, m). We also measured water depth under nest (DEPTH, cm; not measured in 2003) and distance to shore (SHORE, m) for Redwinged Blackbirds only. Nest substrates were pooled into categories based on structure or species whenever possible (e.g. conifer, hawthorn, deciduous, vines/shrubs, or artificial for American Robin, Turdus migratorius; hawthorn, honeysuckle or buckthorn for Gray Catbird, Dumetella carolinensis; and reeds or non-reeds for Red-winged Blackbird) to ensure adequate sample size. We determined nest height and distance to edge using a measuring tape stretched along the nest-monitoring pole. Nest concealment was estimated visually laterally in all four cardinal directions, and above and below the nest, at 1 m from the nest. We took the average of the six measures to create an overall concealment score after we determined that there was varying correlation between the components. We measured the distance to nearest disturbance using a tape measure for distances smaller than 10 m (score of 3), pacing for distances between 10-20 m (score of 2), and measuring using digital aerial photographs for distances greater than 20 m (score of 1). Disturbance types were ranked based on mowing frequency and relative use by people and dogs, and assigned scores: 0 for low disturbance (e.g. seldom-used walking paths, never mowed); 1 for medium (e.g. maintained dog-walking trails, rough areas on golf course); 2 for medium-high (e.g. fairways mowed once per week, cart paths, moderate golf activity), and 3 for high (e.g. tee areas mowed every day, high golfer activity). These scores were then added to distance scores to create a disturbance index from 1-3 (DISTURBANCE; 0-1 for low, 2-3 for medium and 4-5 for high).

If nests were not found at initiation (first egg), nest age was determined using photos obtained from known-age nests or by backdating using the mean incubation period from birds in this region (Gauthier and Aubry 1996). Nesting data collected include the following: number of eggs per clutch, number of nestlings per brood, presence or absence of brood parasitism, parental activity during each nest visit, nest fate for each interval, and cause of nest failure, whenever possible.

*Nest fate.* -- Nest fate was inferred from nest contents. We assumed that nests with at least one fledgling (excluding Brown-headed Cowbird, *Molothrus ater*, fledglings) were successful (i.e. fledglings seen near the nest, adults seen carrying food or defending the area, young observed in the nest within two days of expected fledge date, or nest rim flattened and covered in feces). Failed nests were classified as depredated or 'other' (Collister and Wilson 2007, Etterson et al. 2007). Nests whose contents disappeared > 3 days before the expected fledging date or were found destroyed (e.g. eggshells, injured or dead nestlings with signs of violence) were considered depredated. All other nests were grouped together. This included nests that failed due to abandonment or poor nest construction leading to egg or nestling losses. It also included those that failed due to weather or starvation: eggs cold to the touch for two consecutive visits or > 5 days past their expected hatch date, nests found with whole eggs or nestlings strewn under or near the nest following a strong weather event, or dead nestlings found in the nest cup. This helped reduce the bias associated with the classification of causes

of nest failure, since nests that failed due to abandonment following a strong weather event may have been scavenged before the investigator visited the nest. This nest would then be classified as depredated, when in fact it failed due to weather (Etterson et al. 2007). This also eliminated the trouble of assigning one particular cause of nest failure when two causes may be closer to the truth, such as the example above. Nests observed with the eggs or young of Brown-headed Cowbird and subsequently abandoned were noted separately.

Since the identity of a predator cannot be inferred from nest remains with any certainty (Larivière 1999, Thompson III and Burhans 2003), three digital infrared motion-sensing cameras (model PM35M13, Reconyx LLP, Holmen, WI) were opportunistically deployed at 15 active nests (Mourning Dove *Zenaida macroura*, American Robin, Northern Cardinal *Cardinalis cardinalis*, and Redwinged Blackbird) between 20 April and 24 July 2006 to document predation events. The goal was to identify nest predators at a subset of nests representative of the species using these sites, rather than provide predation rates or predator abundance measures. We deployed cameras at three golf courses and one green space. However, the golf course nests were either successful or the cameras were moved by curious golfers or maintenance staff. Thus, the nests monitored on the green space provided the most information for this part of the study.

Cameras were either affixed to poles mounted next to the nest, or were tied to a nearby branch which afforded a view of the nest. The distance to the nest from the camera varied depending on the type and density of the surrounding vegetation. The camera units were relatively small (21.5 cm x 18.7 cm x 9.5 cm) and completely self-contained, reducing their possible impact on the nest environment. Camera placement usually took 5-25 min, depending on the type of installation required and the height of the nest. Monitoring began during building for six nests, during laying for two nests, during incubation for six nests, and during the nestling period for one nest. Nests were visited every 1-4 days to replace batteries and memory cards, and to monitor nest contents. These visits lasted for approximately 2-5 min depending on the accessibility of the nest. Four of the monitored nests fledged young. These cameras were in place for roughly 20

days, with one as long as 33 days. This limited the number of nests we were able to monitor with only three cameras. Nests with installed cameras were not used for nest survival analyses.

Data analysis. -- We eliminated renests (n = 5) from all analyses, despite some evidence that nest fates are independent from one another (Styrsky 2005). We used Mann-Whitney U tests to search for differences in nest density and in the number of nests per species on golf courses and on green spaces. Density was calculated as the number of nests per site divided by the amount of usable nesting habitat (defined as area covered by vegetated water for Red-winged Blackbirds, as they nested predominantly in vegetated ponds and wetlands, and as site size minus the area covered by turfgrass and rough for upland-nesting species).

We calculated the basic nest statistics for the five most common species (i.e. highest numbers of nests): American Robin (generalists), Gray Catbird and Northern Cardinal (shrub-nesters), Cedar Waxwing (*Bombycilla cedrorum*, shruband tree-nesters), and Red-winged Blackbird (primarily reed-nesters). Percent hatched and fledged were calculated by taking the ratio of the total number of eggs hatched over the number of eggs laid, and the number of birds fledged over the number of eggs hatched for each nest, regardless of the outcome. We used Kruskal-Wallis tests to determine whether these nesting statistics varied by year. We then pooled years to look for differences between types of sites (golf course vs. green space; Zar 1999).

We ran a missingness analysis on any variable missing over 10% of its values (water depth and distance to shore; Red-winged Blackbirds only) to determine whether the missingness of the variable was related to the response variable and eliminating the individuals with missing values would distort the model (Lindsey and Lindsey 2001). First, nine nests were removed due to incomplete data (missing > 40% of values). We then modelled Red-winged Blackbird nest fate as a function of all variables without any missing variables (location, substrate type, substrate arrangement, disturbance; Lindsey and Lindsey 2001). We then added a binary missingness indicator (0 or 1) for water depth and

distance to shore, respectively. In both cases the change in deviance was less than 1.0, and Chi-square tests indicated that nest fate was not dependent on the variables' missingness,  $\chi^2_1$  (n = 218) = 0.52, and 0.71, P = 0.05, respectively. To assess the effect of removing all individuals with missing values, we compared the minimal model with all individuals (n = 218) to a minimal model with the individuals missing values removed (n = 162 for water depth and n = 168 for distance to shore). Eliminating individuals that were missing values for water depth did not distort the model, as the change in deviance for each observation eliminated (1.3) did not exceed the threshold of 1.4 (the average contribution to the -log-likelihood for binary observations with a probability of success of 0.51 [111 Red-winged Blackbird nests/218]; Lindsey and Lindsey 2001). The model with deleted observations also fit better,  $\chi^2_1$  (n = 162) = 72.97, P < 0.001. Thus, individuals with missing water depth values did not influence the model and were safely eliminated (n = 56). Additionally, as missing values were a function of year (not measured in 2003) and not, for example, accessibility, we feel that eliminating these nests did not bias results. Eliminating observations with missing distance to shore values produced a distortion in the model (change in deviance 2.1 > 1.4). Since nest fate was not dependent on this value's missingness and a distortion results for the removal of observations with missing distance to shore values, these observations were retained and replaced with the average. All other missing values were replaced with either averages or mode values per site.

We tested all continuous nest-site variables for normality and applied the transformation that maximized the Shapiro-Wilks' test statistic. Though some variables remained not normally distributed after transformation, all Shapiro-Wilks' statistics were > 0.92 with the exception of water depth (W = 0.84), leading us to believe that any departure from normality was not enough to violate the requirements for parametric tests (Clark and Shutler 1999). We used t-tests to compare all nest-site characteristics between successful and depredated nests, and between nests on golf courses and green spaces for the five most common species. Differences in water depth for Red-winged Blackbirds were examined using Mann-Whitney U tests. Z-tests were used to test whether the proportion of each

categorical variable differed significantly between successful and depredated nests, and between golf courses and green spaces (Zar 1999).

We temporarily converted continuous variables into four-level categorical variables based on quartiles and tested for correlation among all variables using Spearman's rank order correlation, retaining variables that were not correlated (r < 0.70). We also performed multiple linear regressions with nest fate as the dependent variable and continuous nest-site characteristics as independent variables to test for multicollinearity, retaining all variables with tolerances > 0.7 and variance inflation factors (VIF) < 1.5 (Appendix 1). To correct for seasonal effects on vegetation, we saved residuals from linear regressions with continuous vegetation variables as dependent variables and Julian date (date of nest termination) as the explanatory variable (Clark and Shutler 1999). To remove year effects and pool data for all years, we calculated z-scores based on each variable's residuals within each year (Clark and Shutler 1999).

We used Program MARK (White and Burnham 1999) to model daily nest survival for American Robins and Red-winged Blackbirds over their respective nesting periods (30 and 27 days; derived from Gauthier and Aubry 1996). These were the only species with sufficient sample sizes for this type of analysis. We also examined the effects of temporal, landscape- and nest-site level variables on the success of all upland-nesting species pooled (average 27-day nesting period; 17 species). Pooling species allowed us to examine site-level effects that would likely affect these species at each site, and that could not be modelled with individual species due to smaller sample sizes. For all other species with n > 25nests, we calculated daily survival rates (DSR) and Mayfield nest success rates (Mayfield 1961) to compare golf courses and green spaces.

We examined possible sources of variation in nest survival through a series of hypotheses, following Dinsmore et al. (2002) and Peak et al. (2004). These were separated into four categories: temporal (YEAR, SEASON, NEST AGE), site- (LOCATION, HOUSE), patch- (SUBSTRATE, ARRANGEMENT; and their interaction for pooled species only since the two are strongly linked) and nest-site levels (NEST HEIGHT, COVER, EDGE, DISTURBANCE; and WATER and SHORE for Red-

winged Blackbirds only). We could not test both HOUSE and LOCATION simultaneously for Red-winged Blackbirds since they were absent from two sites (one golf course, one green space), rendering the codes redundant when combined. We ran two analyses, using HOUSE and LOCATION in turn as the sitelevel variable, and looked for differences between the two approaches. We did not include SEASON in the Red-winged Blackbird model due to confounding with NEST AGE, as they were synchronous nesters (64% initiated within 20 days of each other; Weatherhead and Sommerer 2001). Using  $AIC_c$  to account for small sample sizes (Burnham and Anderson 2002), we tested all possible unique 4-, 3-, and 2way combinations of the four groups, and ran each group singly. We also ran a null model (constant only) and a saturated model (all variables) to use as reference points. We tested a total of 16 models each for American Robin, Red-winged Blackbird and all upland-species pooled. By using this comparative approach, we were able to draw conclusions about the overall importance of each type of variable, as well as the effect of each covariate on open-cup nesting songbirds in suburban green spaces.

We used the logit function to confine estimates between 0 and 1, and calculated  $\Delta$ AIC and Akaike weights ( $w_i$ ) to assess model importance (Burnham and Anderson 2002). We used model averaging when examining parameter importance to account for model uncertainty when models were within  $\Delta_i < 2$  of the top model, which indicates that they have strong support (Burnham and Anderson 2002). We used the entire set of candidate models to calculate modelaveraged parameter estimates ( $\bar{\beta}$ ) and their associated unconditional standard errors, replacing any estimate and its standard error with zero when it was not present in the model (Burnham and Anderson 2004). We calculated nest success over the entire season by taking the product of all DSR estimates from the top model. We did not interpret covariates ( $\hat{\beta}$  or $\bar{\beta}$ ) whose 95% CI bounded zero, indicating little effect on the model. Only successful nests and nests that failed due to predation were analyzed.

We used Program MARK (White and Burnham 1999) to model daily nest survival and STATISTICA (version 8.0, StatSoft, Inc.) and SPSS (PASW Statistics 17, SPSS Inc.) at a significance level of  $P \le 0.05$  for all other analyses. Test results where  $0.05 \ge P \le 0.10$  were considered marginally significant. Unless otherwise indicated, all data are presented as mean  $\pm$  SE.

### RESULTS

*Nesting communities.* -- We found 865 nests of 19 species from 2003-2005 (Table 2; see Appendix 2 for complete list of observed species). Red-winged Blackbird (n = 261) and American Robin (n = 166) comprised just under half the sample (49%). The remaining half was mainly composed of the following species: Gray Catbird (n = 52), Cedar Waxwing (n = 47), Northern Cardinal (n = 46), Yellow Warbler (*Dendroica petechia*, n = 44), Common Grackle (*Quiscalus quiscula*, n = 42), Chipping Sparrow (*Spizella passerina*, n = 35), and Song Sparrow (*Melospiza melodia*, n = 29). We found less than 25 nests each for the remaining ten species.

We only found three species with biologically relevant differences (P < 0.10) in the number and density of nests between golf courses and green spaces. All Chipping Sparrow nests were on golf courses, with ensuing differences in the proportion of successful nests. The shrub-nesters (e.g. Gray Catbird, Cedar Waxwing, Yellow Warbler and Northern Cardinal) all nested at higher densities on green spaces (e.g. Gray Catbird and Yellow Warbler nests were roughly six and over 12 times denser, respectively; Table 2). Indeed, the proportion of nests placed in shrubs on green spaces (174/281; 62%) was over three times higher than the proportion on golf courses (76/421; 18%), suggesting a difference in availability. There were no large differences in any of the metrics between golf courses and green spaces when all upland-nesting species were pooled, though upland-species nest density on green spaces was more than double that of golf courses (Table 2).

Site-scale nest survival. -- The most frequent cause of nest failure was predation (n = 362; 80% of total nest failures), followed by other causes (e.g. abandonment, weather events; n = 55) and brood parasitism (n = 33; Table 2). Though nine

species were parasitized (not all leading to nest failure), only Yellow Warblers and Chipping Sparrows appeared to be negatively affected (Table 2).

Nest success rates varied widely depending on the species (Table 3), and there were variations in the difference between nest success rates on golf courses and green spaces (Fig. 2). Though differences appear large in some cases, only three species did not have (or barely had) their 95% CIs overlapping, suggesting statistically significant differences in rates (Table 3). Red-winged Blackbirds were 1.5 times more successful on golf courses, while Common Grackles were > 18 times more successful on golf courses. Cedar Waxwings, on the other hand, were almost completely unsuccessful on green spaces.

We used 476 nests (n = 249 successful, n = 227 depredated) of 17 opencup nesting species to examine the factors that influence nest survival and predation on a site scale. The most supported model indicated that nest success was influenced by temporal and patch-scale variables (Table 4), more specifically year, substrate type, substrate arrangement and their interaction (Table 5). The next most supported model differed from the top model by the inclusion of site effects, neither of which appeared to influence the model (i.e. 95% CIs crossed zero; Table 4). However, when modelled alone, nest survival was positively influenced by HOUSE ( $\hat{\beta} = 0.170, 95\%$  CI = 0.041, 0.300). Indeed, sites embedded in a suburban matrix showed lower predation rates than sites surrounded by a more natural landscape (Fig. 3). Year was the only temporal variable to affect nest survival; survival decreased from 2003-2005 (Table 5). Nest survival varied extensively with substrate type, with artificial substrates and deciduous trees providing high and low extremes, respectively (Table 6). Nest success varied according to substrate arrangement, with individual trees and shrubs providing the highest likelihood of predation (Table 6). However, substrate arrangement alone did not affect nest survival. The impact of substrate type and arrangement was modified when combined as an interaction patches of conifers provided roughly 6-7 times higher likelihoods of survival when compared to isolated deciduous trees and patches (Table 6). Overall, nest survival on golf courses, assuming constant survival, was 37% (95% CI: 29, 44%) while nest survival on green spaces was 36% (95% CI: 30, 42%). The Mayfield rate for all nests, which assumes constant survival over the entire nesting season and does not take any covariates into account, was estimated at 36% (95% CI: 32, 41%). However, the null model, which the Mayfield rate is based upon, had the least support of all models (Table 5). The top model survival estimate was 14% (95% CI: 0, 35%), and is likely more accurate than the Mayfield estimator due to better model fit.

Species-specific nesting requirements and productivity. -- The five focal species showed differences in substrate use. American Robins were generalists, nesting in 31 different species of plants. Nests (n = 166) were placed in coniferous trees (43%), deciduous trees (19%), hawthorns (16%), shrubs or vines (15%) and artificial substrates (e.g. stone pillars, barn rafters and chain-link fences; 8%). Gray Catbirds (n = 52) were much more selective shrub-nesters, using only five different plant species. The most commonly used were hawthorn (48%), honeysuckle (33%) and buckthorn (15%). Cedar Waxwings (n = 47) used 18 different species, including a variety of trees and shrubs, with hawthorns (25%) and crab-apple (11%) used most frequently. Northern Cardinals (n = 46) nested in 13 different species, including a variety of coniferous and deciduous shrubs, with hawthorns (34%) and cedar shrubs (18%) used most frequently. Red-winged Blackbirds (n = 261) nested predominantly in cattails (83%), but used 17 species in total including shrubs, reeds, and grasses.

The nest-site characteristics of the five most common species varied between golf courses and green spaces, some significantly so. American Robin nests were located higher up on golf courses than on green spaces, and were more concealed (Table 7). On golf courses, there were more placed in conifers, artificial substrates and hedges, and fewer in hawthorns, shrubs and large stands of vegetation (Table 7). Of these variables, only concealment, the use of artificial substrates and hedges (i.e. linear arrangement) differed between successful and depredated nests (Table 7). Gray Catbird nests on golf courses were placed closer to the ground, and were placed almost exclusively in honeysuckle, whereas those on green spaces were located in hawthorn (Table 8). Neither of these differences

appeared to affect success rates, as the only difference between successful and depredated nests was greater nest cover for depredated nests (Table 8). Northern Cardinals on golf courses placed their nests closer to the ground and to foliage edges, and had higher nest cover (Table 9). There were fewer Northern Cardinal nests on golf courses in hawthorns, more in conifers, and more in individual trees and shrubs. There was a higher proportion of depredated nests in conifers (Table 9). Cedar Waxwings on golf courses nested higher up and farther from foliage edges (Table 10). They also nested more in conifers and less in hawthorns on golf courses. There were more Cedar Waxwing nests depredated in conifers and fewer depredated in hawthorns (Table 10). Red-winged Blackbirds showed differences in almost every metric: nests on golf courses were placed closer to the ground and farther from an edge, and were three times closer to shore (Table 11). There was a greater proportion of nests in reeds on golf courses, and a larger proportion of nests in vegetation patches (Table 11). Few of these differences appeared related to nest success, though there was a higher proportion of successful nests in vegetation patches. For all species except Cedar Waxwing, nests on golf courses were exposed to higher levels of disturbance than those on green spaces. Only Northern Cardinals, Cedar Waxwings, and Red-winged Blackbirds were seemingly affected by disturbance. There were more depredated Northern Cardinal (Table 9) and Cedar Waxwing (Table 10) nests at low disturbance levels, more successful Cedar Waxwing and Red-winged Blackbird (Table 11) nests at intermediate disturbance levels, and more successful Cedar Waxwing nests but fewer successful Red-winged Blackbird nests at high disturbance levels. American Robins had a higher proportion of successful nests at high disturbance levels, but this difference was only marginally significant (Table 7).

Three of the five most common species showed differences in basic nest statistics between golf courses and green spaces (Table 12). American Robins and Cedar Waxwings were the only ones to show yearly variations in productivity (e.g. American Robins on Elm Ridge had smaller clutches in 2004, both Ile Perrot and Elm Ridge had a smaller number of American Robin young hatch and fledge in 2005, the average American Robin initiation date at Ile Perrot in 2003 was 20

days later than the mean initiation date in 2005, and Cedar Waxwings did not hatch any young in 2005). We examined these differences and concluded that they were due to small sample sizes and were not biologically relevant, since no overriding pattern was discernible that might influence results. However, we cannot rule out the possibility of local food supply differentially affecting species in different years. American Robins showed no differences in clutch size, mean number of young hatched and fledged, or percent hatched and fledged between golf courses and green spaces (Table 12). American Robins on golf courses, however, initiated laying an average of nine days earlier than those nesting on green spaces. Despite high overall nest success rates on both types of sites (Table 3), Gray Catbirds nesting on golf courses had consistently lower productivity values than those on green spaces, with the exception of initiation date (Table 12). Catbird clutches were 1.25 times larger on green spaces, leading to more young hatched and fledged. However, the percent of young fledged between the two site types was comparable. Cedar Waxwings experienced almost complete reproductive failure on golf courses due to predation during incubation. Neither Northern Cardinals nor Red-winged Blackbirds differed substantially in any productivity metric. Losses from partial brood reduction, either at the egg or nestling stage (failed nests and non-viable eggs left in the nest were excluded), varied between species: Red-winged Blackbirds lost 50 eggs or nestlings (19% of total eggs laid), American Robins lost 39 (24%), Northern Cardinals lost 13 (10%), Gray Catbirds lost eight (15%) and Cedar Waxwings lost four (9%).

*Nest survival.* -- Modelling results indicate that American Robin nest survival was most affected by temporal and patch-level characteristics (Table 13). We used 157 American Robin nests (85 depredated, 72 successful) for the analysis. Nest predation accounted for 91% of total nest failures. The second-most supported model differed from the top model by the inclusion of site-level variables (LOCATION and HOUSE; Table 13). Not one explanatory variable appeared to greatly affect nest success rates (i.e. all 95% CIs crossed zero) once results were model-averaged (Table 14). However, the top model ( $w_i = 0.66$ ) indicated that DSR varied with substrate type and arrangement, and 97% of model weights included temporal and patch-level effects. The likelihood of predation was nine times higher in deciduous trees than on artificial substrates, reflecting the large range between the worst and the best substrate types (Table 15). High predation rates were also apparent for nests in singletons and patches of vegetation, which showed rates 3.5 and 4 times higher than hedges (lines; Table 15). Mayfield nest success was 30% (95% CI: 23, 38%), though the null model was one of the least supported models (Table 13). Nest survival across the entire nesting period for American Robins was 22% (95% CI: 0, 46%). Apparent nest success (number of successful nests divided by total number of nests) was 40%.

Both sets of model results (LOCATION and HOUSE) indicated that Redwinged Blackbird nest survival was affected by temporal, patch- and nest-site level characteristics. We used 161 Red-winged Blackbird nests that either failed due to predation (n = 83) or succeeded (n = 78). Nest predation accounted for 76% of total nest failures. The second-most supported model in both analyses was the global model, differing from the top model by the inclusion of site-level variables (LOCATION or HOUSE; Table 13). The top model indicated that DSR varied with nest age, decreasing throughout the incubation period, hitting a low at the average hatch date, and then steadily increasing through the nestling phase (Table 14; Fig. 4). DSR also increased with nest cover (Table 14). Mayfield nest success was 35% (95% CI: 29, 42%), the null model again being the least supported model (Table 13). The Mayfield rate slightly underestimated survival, as the estimate for nest survival over the entire nesting season was 40% (95% CI: 18, 57%). Apparent nest success was 37%.

*Predator communities.* -- Infrared cameras allowed us to identify two actual predators (nest visit with consumption of eggs; Red Squirrel [*Tamia hudsonicus, n* = 3 events] and American Crow [*Corvus brachyrhynchos, n* = 1]) and eleven potential predators (nest visits only; Blue Jay [*Cyanocitta cristata*], Black-capped Chickadee [*Poecile atricapillus*], Carolina Wren [*Thryothorus ludovicianus*], Common Grackle, Red-winged Blackbird, Baltimore Oriole [*Icterus galbula*],

Brown-headed Cowbird, Human maintenance worker [Homo sapiens], Eastern Chipmunk [Tamia striatus], Red Squirrel, unidentifiable mouse [Peromyscus sp.]; Appendix 3). Predation events were captured by cameras at Stoneycroft. None occurred at nests monitored by camera on the three golf courses used for this portion of the study, though we documented several potential predators, listed above. We also observed several predatory events during field work (GC denotes golf course, GS denotes green space): two Raccoons (Procyon lotor) scaled a 10m conifer to raid an American Crow's nest (unknown whether eggs or nestlings were taken; GC); a Common Grackle depredated a Mourning Dove nest (GC), a Common Grackle attacked an adult Sora (end result was not observed; GS), and Red Foxes (Vulpes vulpes) carried off an adult Wood Duck (GC) and attacked an adult American Crow (GC). We did not document any predation events during the nestling period. Aside from the camera recordings, other potential nest and adult predators documented on both golf courses and green spaces were: Common Garter Snake (Thamnophis sirtalis), Northern Harrier (Circus cyaneus), Redshouldered Hawk (Buteo lineatus), Red-tailed Hawk (B. jamaicensis), Sharpshinned Hawk (Accipiter striatus), Cooper's Hawk (A. cooperii), Merlin (Falco columbarius), Great Horned Owl (Bubo virginianus), Gray Catbird, Gray Squirrel (Sciurus carolinensis), voles and mice (Cricetidae spp.), Raccoon, Ermine (Mustela erminea), Striped Skunk (Mephitis mephitis), House Cat (Felis catus), and White-tailed Deer (Odocoileus virginianus). Black Rat Snake (Elaphe obsoleta), Red Fox, Coyote (Canis latrans), American Kestrel (F. sparverius), Eastern Screech Owl (Megascops asio) and Long-eared Owl (Asio otus) were only observed on golf courses, while Eastern Milk Snake (Lampropeltis triangulum triangulum) and Northern Saw-whet Owl (Aegolius acadicus) were only observed on green spaces.

## DISCUSSION

Only a few species differed between golf courses and other suburban green spaces in terms of nest density, nest survival rates, productivity parameters, or all of the above. Mayfield nest success rates, though quite variable, were similar to or above rates published by Martin (1992) concerning open-cup nesting Neotropical migrants (39%). Comparing rates found here to determine how well these populations are doing relative to other suburban green spaces is difficult due to the wide variety of species and geographical locations examined (e.g. urban [11%] vs. rural [14%] American Robin and Northern Cardinal [17 and 20%, respectively] nest success in central Ohio, derived from Borgmann and Rodewald 2004, and 59% and 65% for Eastern Bluebirds [Sialia sialis] on and off Virginia golf courses, LeClerc et al. 2005). There is no simple answer regarding the impact of golf courses on breeding birds. It appears to be species-specific, with some species faring better on golf courses than on other, more natural green spaces, and others experiencing almost complete reproductive failure. Stark differences among species with relatively similar nesting habits may be due to speciesspecific preferences in nest-site placement and adult behaviour at the nest, along with the microclimate and predatory repercussions associated with each choice. Site quality likely plays a large role as well, with sparsely vegetated golf courses providing worse wildlife habitat than more natural ones. Though this study was not specifically designed to address why rates might vary between species, we can provide hypotheses based on our analyses. We believe that this question is the next logical step and should be examined if these non-traditional sites are to provide suitable nesting grounds.

*Site-scale nest survival.* -- When all species were combined, nest survival was influenced by temporal and patch-scale factors, with nest-site characteristics having little influence on survival. This lack of influence may be due to the pooling of species with different nest-site preferences, but we only included characteristics that would likely affect all upland-nesting species the same way (e.g. nest concealment), and excluded any cavity- or wetland-nesting species. In contrast with other studies that have found strong temporal effects (e.g. Grant et al. 2005, Knutson et al. 2007), we found little evidence supporting their importance. In fact, the only temporal effect that showed any support was the year effect: nest survival decreased over the three years of the study. This trend could

be due to any number of untested factors: predator attraction via investigator disturbance despite the fact that we attempted to minimize disturbance and did not approach nests when a predator was watching, changes in predator community composition, differences in food availability, changes in weather, and/or changes in landscaping and management practices. However, though nest success decreased over the years, the survival estimates' 95% CIs all overlapped, suggesting little actual difference between years.

Our results suggest that, while land use itself (golf course or green space) matters little in terms of nest survival, land use likely has a direct impact on the composition of the site at the patch-scale (substrate, arrangement and their interaction) through landscaping, which has a considerable influence on nest survival. The strength of the interaction (20  $\Delta$ AIC units, not shown) also indicates that it is important to consider not only the type of substrate available, but how that substrate relates to its environment. For example, coniferous hedges (substrate×arrangement) provided good nesting habitat, while deciduous hedges provided some of the worst. This is likely why Common Grackles fared well on golf courses; they nested exclusively in cedar or juniper hedges on golf courses, and in willow shrubs on green spaces. Though the arrangement "lines" was grouped with "patches" for analysis due to low sample sizes, univariate tests suggest the importance of this particular example, as do American Robin results where the combination did not occur: 5 of 6 nests (83%) in deciduous hedges were depredated, while only 10 of 53 nests (19%) in coniferous hedges were depredated. Deciduous hedges on golf courses require more trimming and shaping than coniferous hedges, which were often left untouched due to their slower growth (M.-A.R. Hudson, pers. obs.). This creates a sink for nesting birds, since they begin nesting only to have the top or sides of the hedges cut, revealing the nest to predators or destroying the nest directly (n = 8). These highly managed and landscaped sites also present an opportunity to examine the effect of exotic vs. native nest substrate on nest survival. Though this was not the focus of this study, we believe it to be an important question that needs to be addressed in this system, as studies have shown that exotic plants can negatively affect avian reproductive

success (Ortega et al. 2007), and that their effects can be amplified in urban landscapes (Borgmann and Rodewald 2004).

We found little evidence that the type of land use or the number of houses surrounding a site influenced nest success, but this may be due to small sample sizes or to the small difference between exurban and suburban sites in this system. On its own, the number of houses surrounding a site was positively related to nest survival. This may indicate that there is decreased predator abundance on or near these sites due to reduced habitat suitable for specialist predators (Sorace and Gustin 2009), or that the predators are present, but have access to alternate food sources (Haskell et al. 2001). There may be differences in predator communities as well, which may then affect predation rates. Few studies have compared predator communities and/or predation rates between urban, suburban and rural green spaces using consistent methods. Thus, the effect of urbanization on predators, and thus predation rates, is not consistent across studies or species, with some refuting the "safe nesting zone" hypothesis (Jokimäki et al. 2005), while others find support for it (Gering and Blair 1999, Blair 2004). Differences in study design (e.g. artificial vs. real nests, quail eggs vs. plasticine eggs, comparisons between specific land use vs. gradients) and geography may be the cause of these inconsistencies, or predator species may simply react differently to increasing levels of urbanization (Haskell et al. 2001). For example, Reidy et al. (2008) found similar predation rates between Golden-cheeked Warblers (Dendroica chrysoparia) nesting on large green spaces surrounded by high-density residential housing, and preserves with virtually no housing. However, they found that the dominant avian predators at these sites differed. This suggests that complex predator communities may render patterns in nest predation difficult to discern, as predator species respond differently to urbanization gradients (Sorace and Gustin 2009).

Our results suggest the possibility of an interaction between land use and urbanization, with golf courses in suburban areas having lower overall predation rates than suburban green spaces, but exurban golf courses having higher predation rates than exurban green spaces. Gering and Blair's (1999) results, though they focused on artificial nest predation rates over an urbanization gradient (nature preserve, recreational park, golf course, residential area, apartment complex and business district), support the first half of this interaction. They found declining predation rates with increasing urbanization, with one of the largest drops in predation rates between the recreational park and the golf course (Figure 2 in Gering and Blair 1999). Recent work also supports this hypothesis, as researchers found that Least Bell's Vireos (*Vireo bellii pusillus*) nesting within 400 m of golf course/park habitat were 80% more likely to be depredated than those nesting beyond 400 m (Kus et al. 2008). They found that urbanization interacted with the golf course/park land use so that nest survival was highest when the golf course/park was in an exurban setting. We believe that this interaction deserves further attention.

Habitat on golf courses and green spaces. -- Nest-site vegetation differed between golf courses and green spaces. Overall, on golf courses, there were fewer nests in shrubs, a higher number of nests in conifers, hedges, and artificial structures, and nests were exposed to higher levels of disturbance. There were differences in shrub and tree composition as well, with some species almost completely absent from golf courses but abundant on green spaces. For example, on golf courses, there were fewer hawthorns, likely due to the fact that their branches are covered with large spines, which may be considered a safety hazard. Gray Catbirds nested in honeysuckle on golf courses and in hawthorn on green spaces, despite the availability of both types on green spaces (M.-A.R. Hudson, pers. obs.). Though we do not know if Gray Catbirds prefer nesting in hawthorn over honeysuckle, this lack of nest substrate may account for their lower density on golf courses. In this case, the difference in species did not appear to influence nest success.

The shrub-nesters (e.g. Gray Catbird [Cimprich and Moore 1995], Cedar Waxwing [Witmer et al. 1997], Yellow Warbler [Lowther et al. 1999] and Northern Cardinal [Halkin and Linville 1999]), all nested at higher density on green spaces. Indeed, Gray Catbirds have shown linear responses to increasing

shrub cover (Cimprich and Moore 1995). The proportion of nests placed in shrubs on green spaces was over three times higher than the proportion on golf courses, suggesting there may be a real difference in availability. Continuous shrubby habitat was much more prevalent on green spaces in our study area than on golf courses (M.-A.R. Hudson, pers. obs.). However, we were not able to differentiate between deciduous tree and shrub cover when digitizing the study sites (see Hudson and Bird 2009; Chapter 2), and were thus unable to specifically quantify differences in availability or quality between site types.

Preference for a given nest substrate may have been the reason Chipping Sparrows nested exclusively on the golf courses we studied. Of the 33 nests found, 29 were located in singly growing conifers, which is typical of this species' nesting habits (Middleton 1998). We believe that the higher density of Chipping Sparrows on golf courses is due to the variable but overall superior conifer coverage on golf courses (range: 0-8% surface area) compared to green spaces (range: 0-2% surface area; see Hudson and Bird 2009; Chapter 2). Higher conifer cover on golf courses was also likely responsible for the earlier American Robin initiation dates on golf courses. Robins tend to use conifers for their first nests of the season, switching to deciduous trees for subsequent nests (Morneau et al. 1995).

Hedges and artificial structures are likely attractive to birds able to tolerate increased disturbance levels associated with high-traffic areas (e.g. walkways and buildings). They typically provide increased concealment and protection from the elements since most of the hedges in this study were of dense cedar, and most artificial substrates provided complete cover on at least one side if not more (e.g. overhangs and maintenance shed rafters). However, not all hedges are created equal; while coniferous hedges provided a lower likelihood of predation, deciduous hedges did not.

Finally, several studies have suggested that investigator disturbance can variably affect nest success, predation and abandonment rates of colonial and open-cup nesting birds (Bart 1977, Götmark 1992, Donehower and Bird 2008, but see O'Grady et al. 1996, Ortega et al. 1997, and Weidinger 2008). We were not

able to test the influence of investigator disturbance directly because any potential disturbance we exerted during nest-checks (e.g. flushing females from nests) was confounded with inadvertent disturbance by golfers and pedestrians, since we observed certain females flushing when golfers walked by. Based on field observation, we believe that our nest-checks did not unduly influence the nests' outcomes, and focus the remainder of the discussion on other causes of disturbance.

Francis et al. (2009) found that though avian assemblages were modified with high levels of noise pollution and certain species suffered reduced nest success, others actually enjoyed higher rates of nest success due to lessened predation pressure. Thus, some species, predators included, appear capable of dealing with excessive and sustained noise, while others are not. Overall, the species we studied responded differently to disturbance: American Robin and Gray Catbird proportions of successful and depredated nests did not differ substantially between disturbance levels, Cedar Waxwings and Northern Cardinals had larger proportions of depredated nests in low disturbance areas, and Red-winged Blackbirds had higher proportions of depredated nests in high disturbance areas. These differences may reflect species-specific tolerances to disturbance, may be an artefact of site type (e.g. the majority of blackbird nests on golf courses were exposed to medium and high disturbance levels), or may indicate that our method was not appropriate.

We cannot discount the possibility that the differences we found may be due to our qualitative method for measuring disturbance. Quantitative methods may allow a more in-depth analysis, allowing researchers to determine if the type of disturbance or the distance from the disturbance has more of an effect (regardless of its direction) on survival. We suggest that future studies measure disturbance levels using microphones situated near nests to measure noise levels. These data could then be paired with data loggers such as those used by Weidinger (2008) to detect whether the female has responded to an experimental disturbance by flushing from the nest. Relating the number and timing of abandoned nests to noise levels may also provide an indication of whether species are affected, provided an accurate assessment of abandonment can be obtained (see Etterson et al. 2007).

*Productivity.* -- Three of the five most common species showed no difference in overall productivity between golf courses and green spaces. They had similar clutch sizes and number of young hatched and fledged compared to previously published results for these species (Klimstra and Stieglitz 1957, Yasukawa and Searcy 1995, Halkin and Linville 1999). Gray Catbirds on golf courses had smaller average clutches and fewer young fledged per nest than reported elsewhere; those nesting on green spaces had similarly sized clutches and broods (Cimprich and Moore 1995). Cedar Waxwings had slightly above average clutch sizes, but well below average hatching and fledging rates (Witmer et al. 1997). Though we were not able to model Cedar Waxwing nest survival due to low sample size, we suspect that their habit of nesting in single, isolated trees and shrubs (30 of 44 nests [68%], Witmer et al. 1997) on both site types may have contributed to the extreme predation rates documented here.

The two studies to date that have focused on passerine reproduction on golf courses compared Eastern Bluebirds on golf courses and non-golf habitats (LeClerc et al. 2005, Stanback and Seifert 2005). Stanback and Seifert (2005) found that bluebirds nesting on golf courses initiated later, had smaller clutches, and had poorer quality nestlings than those nesting off golf courses (e.g. hayfields and pastures). They suggested that these differences were due to lower arthropod abundance on golf courses which lead to delayed and lessened breeding condition. We suspect that if food abundance were the underlying cause for smaller Gray Catbird clutches on golf courses, then the other species would have responded similarly. However, this cannot be discounted since arthropods may account for a larger proportion in the adult gray catbird diet than in these other species, or catbirds may feed on different types of arthropods that are more affected by golf course maintenance practices.

LeClerc et al. (2005), on the other hand, found that Eastern Bluebird nests on golf courses had 28% more eggs, fledged 17% more young and had more
physiologically symmetrical young (an indication of fitness) than other structurally similar habitats (campuses, farms, recreational parks). However, fitter young may not be a reliable indicator of site quality, as European Starling (*Sturnus vulgaris*) fledglings experimentally stressed as embryos developed stronger flight muscles and were thus better able to escape in flight trials than controls (Chin et al. 2008). Thus, young hatched in low-quality environments or to food-stressed mothers may actually be preconditioned to better deal with low environmental quality (Chin et al. 2008). Whether this embryonic boost in performance is happening on golf courses should be examined. We did not measure nestling condition before fledging for fear of biasing our productivity results, and thus have no information that may provide clues towards post-natal recruitment into the population (something that is sorely needed to satisfy the question of golf course productivity).

American Robin and Red-winged Blackbird nest survival. -- To our knowledge, this is the first study to examine nest survival rates of American Robins and Red-winged Blackbirds nesting on golf courses and nearby green spaces. Though both species showed higher nest success on golf courses than green spaces, overall reproduction rates were low when compared to other studies. Apparent nest survival in American Robin (40%) was either comparable or low compared to studies focusing on residential areas (21-36% from Howard 1974, 90% from Morneau et al. 1995), and similar when compared to studies focusing on more natural spaces (48% in western riparian areas, Heltzel and Earnst 2006). The nest survival estimate generated from the top model in this study (22%) was half that calculated by Knutson et al. (46%; 2007) in the Mississippi floodplain of Minnesota, Wisconsin and Iowa. Red-winged Blackbirds showed average apparent nest survival when compared to other studies (19-45% from Picman et al. 1988, average of 40% from Beletsky 1996, 30-65% derived from Weatherhead 2005). Overall though, conclusions are difficult as there is substantial variation in rates reported in the literature (fledging success varied from 40-88% in one collection of studies; Dyer et al. 1977). We recommend that future studies replicate this study to allow more meaningful comparisons. We also recommend that any study focusing on nest survival report apparent nest success, Mayfield rates and model-derived estimates of nest survival to ensure maximum comparability with other studies, regardless of the analytical method used. Though any study focusing on this question should strive to maximize sample sizes, both in terms of the number of nests found per site, as well as the number of study sites per treatment, statistical significance of differences in nest survival rates (or, more important, lack thereof), should not overwhelm the potential biological importance of a slight difference in survival for small populations (Donovan et al. 1995). Most importantly, analyses should not be restricted to simple comparisons. It is clear that understanding the factors that drive nest success on different sites should be a priority for future research..

American Robin and Red-winged Blackbird nest survival were influenced by similar factors, despite exhibiting species-specific differences in nesting behaviour. American Robin nest-site preferences are fairly broad, and they begin nesting early, leading to long nesting seasons (Sallabanks and James 1999). Redwinged Blackbirds prefer reeds for nesting and can be quite synchronous due to their polygynous mating system (Beletsky 1996). Adaptability is perhaps why we were not able to find a single variable that significantly affected American Robin nest survival. Knutson et al (2007) also failed to find variables that affected American Robin nest survival, and ascribed this to their generalist nature. Our top model suggested that substrate type and arrangement may influence nest survival, however when model uncertainty was taken into account, those effects disappeared. These results may point to a flawed model selection method, to an overly conservative model-averaging method (using all models and substituting zero when the parameter was not within a given model), or simply to a lack of effect in the chosen variables on such an adaptable species. Our Red-winged Blackbird results suggest the latter, as the same method was used to examine nest survival of both species. As Red-winged Blackbirds show a higher degree of specialization in their nest-site selection, it makes sense that our analyses were able to highlight specific variables that influenced their survival.

Nest cover influenced Red-winged Blackbird nest survival despite little difference found in univariate tests, a result similar to that found by Grandmaison and Niemi (2007). Relationships between nest concealment and predation likely exist and have been investigated extensively (Martin 1993, Filliater et al. 1994, Burhans and Thompson III 2001). Several studies have found a positive effect of nest concealment (Martin and Roper 1988, Grant et al. 2006), while others have found no effect (Seitz and Zegers 1993), or even a negative effect (Smith et al. 2007). These relationships may be difficult to discern as the underlying mechanisms influencing predation depend on habitat type, composition of the predator community and its density (Rangen et al. 1999), nest density, and behaviours such as parental nest defence and predator distraction (Flaspohler et al. 2000, Weidinger 2002). For example, if mammals are the main nest predators, concealment may have little impact on the likelihood of nest predation, as they tend to use olfaction and/or parental or nestling activity to detect nests (Rangen et al. 1999). In contrast, nest concealment may be of greater importance when highly visual avian species are the main nest predators (Sugden and Beyersbergen 1986, Clark and Nudds 1991, Colwell 1992). We suspect that blackbird nest predators in this study were most likely avian, with the exception of the Raccoon, since the two main mammalian predators in our system (Eastern Chipmunk and Red Squirrel) were never found near most blackbird habitat (wetlands and ponds).

Contrary to our predictions concerning the effects of nest age, survival rates decreased through incubation, then increased through the nestling period for Red-winged Blackbirds. This pattern is similar to that found by Grant et al. (2005), but the opposite of Caccamise's (1976) results for Red-winged Blackbirds nesting in a tidal marsh (increasing mortality throughout the nestling phase). Possible reasons for the pattern we found are reviewed in Grant et al. (2005) and include increased parental nest attendance during incubation providing cues for predators, additive exposure risk, increased activity levels near and post-hatch, increased parental nest defence as nestlings age, increased inability of predators to consume larger nestlings, and increased ability of older nestlings to fledge early if required. It is likely that our results differ from Caccamise's (1976) because of

differences in habitat (cattails vs. salt marsh grasses and shrubs) and thus predator communities. Since the avian predators common to this study are more likely to consume eggs than half-grown nestlings, it follows that predation rates should be higher during incubation. Indeed, Weatherhead and Sommerer (2001) found the opposite trend in eastern Ontario wetlands (increased predation during the nestling phase), where the majority of Red-winged Blackbird nest predation was caused by mammals.

Nesting in deeper water has been shown to increase Red-winged Blackbird nest survival (Picman et al. 1993), which was supported by the univariate analysis comparing water depth between successful and depredated nests. This suggests that water depth is probably an important component of nest-site selection in this species. However, water depth was not an important driver of nest predation. Redwinged Blackbirds in this study nested mostly in irrigation and ephemeral ponds, where the slightest rainfall or irrigation requirements on the golf course changed water levels (up to 65 cm in one case) within a few hours. This caused the drowning of 19 blackbird nests due to sudden increases in water level. It is possible that water levels varied so much over the course of the season that adding a temporal component to the analysis erased any clear effect of water depth and nest height (which is directly related to water levels) on nest predation, since measurements were taken throughout the season and were subject to this variation.

*Predator communities.* -- We documented a variety of mammalian and avian predators in our study area, though only five species were confirmed nest predators. It appeared that the Common Grackle and the Baltimore Oriole captured on film were either attempting to renest in an abandoned robin's nest, or removing nesting material from the same nest (respectively), so predation may not have been the motivation for their visits, despite the fact that both are documented nest predators (Sealy 1994). We were able to monitor a greater number of nests at Stoneycroft than on any of the golf courses, and thus captured a greater variety of predator species on film. However, nests at Stoneycroft suffered fairly high

predation rates overall, suggesting that the higher number of predators caught on film was not simply a function of the number of nests monitored. It is likely that increased habitat availability allowed for a richer predator community, since this green space is surrounded by forest and agricultural fields and may suffer from additive predation (i.e. nests are depredated by predators from both habitat types, sensu Tewksbury et al. 2006). Predator communities on more natural sites and not surrounded by urbanization may have more medium-sized mammalian predators, while sites with extensive lawn and fencing, which is typical of golf courses, may have more avian predators since mammalian predators are discouraged by lack of cover (Weatherhead and Sommerer 2001). We cannot support nor refute this hypothesis since we did not measure predator abundance. However, Sorace and Visentin (2007) compared predator abundance on three Italian golf courses and surrounding areas. They found significantly more foxes and crows on the largest of the three golf courses (all other comparisons but one were variable and nonsignificant) when compared to surrounding urban and agricultural areas (Sorace and Visentin 2007). We recommend that future studies examine predator abundance along gradients crossing golf courses and nearby green spaces embedded in both urban and exurban matrices to ascertain how predator abundance varies between land uses and landscape types. Understanding predator community dynamics at multiple scales across landscapes may be the key to managing breeding bird habitat.

*Management implications.* -- Donovan and Thompson (2001) maintain, quite logically, that as the proportion of low-quality habitat increases, nest success in high-quality habitats must increase. Their modelling results, which included a variety of demographic parameters (e.g. adult and juvenile survival, number of nesting attempts, number of young fledged per nest), indicated that with only 20% low-quality habitat, 35% nest success was required in high-quality habitats to ensure population stability. This threshold jumped to 55% nest success if low-quality habitat took up 60% of the area. Our six study sites showed variation in nest success rates ranging from 23% to 53% (median 35%; based on 27-day

nesting period). If we consider that roughly 40% of a golf course is used for play and is therefore considered low-quality habitat, nest success would have to be over 40% to maintain populations, assuming the demographic parameters were similar between this study and the values used by Donovan and Thompson (2001). Certainly, some of our study sites fall below this extrapolation, golf courses and green spaces included. However, it is important to also consider these figures in an urban context: 40% of an area used to play golf actually serves to protect 60% green space that may otherwise have been developed and lost altogether. It is critical that we understand and change how these sites are designed and managed if suburban/urban green spaces are to provide adequate breeding bird habitat.

Suburban sites in this study provided safer nesting grounds than those situated in an exurban matrix (small-scale agriculture, large-lot residences, or deciduous forest). Clearly this cannot be managed at a site-level, so we suggest that this be investigated further, especially with regards to possible interactions between land use and the surrounding matrix. It is possible that highly managed golf courses have more isolated trees and grass cover, as design usually dictates, and thus offer an increased risk of predation when embedded in a natural environment capable of supporting large numbers of predators. Indeed, artificial nest predation rates were higher in managed parks than unmanaged parks in Finland (Jokimäki and Huhta 2000), likely due to simplified vegetation structure and lack of shrub and ground vegetation layers. If this relationship holds in this system, it should be taken into account when planning a golf course: golf courses should be sited in suburban or urban areas, so that they may take advantage of the "safe nesting zone" which appears to be provided by dense residential housing. Not only would this reclamation improve the aesthetics of the area and thus the quality of life for those living nearby, but it might provide an oasis in urban areas for breeding, wintering and migratory wildlife.

At the site scale, several simple steps can be taken to improve the quality of nesting habitat provided by golf courses. Our results indicated that nests in isolated shrubs or trees are at substantially higher risk of predation, regardless of plant species. Thus, the propensity for planting isolated or small patches of any substrate type, both on golf courses and recreational parks, should be strictly avoided. Whether nest survival increases with the addition of shrub cover (i.e. increased complexity), or whether the addition of other trees (i.e. increased patch size) is required should be investigated. Landscaping and vegetation trimming, while sometimes a necessity for safety reasons, should be kept to a minimum. Landscaping directly caused the failure of at least two nests through accidental but complete removal of the entire nest, and indirectly caused the failure of at least six others through reduced concealment and disturbance. Granted these numbers are not very high, but these represent an absolute minimum; it is likely that more were affected and were never found. Nests in deciduous hedges were particularly prone to predation, thus we suggest that deciduous hedges either be trimmed after the breeding season, or be replaced with native coniferous hedges. These would provide safer nesting sites than deciduous hedges, would require minimal maintenance, and would provide roosting cover in winter. Increasing native shrub cover and understory through reduced mowing and maintenance of rough areas is highly recommended as well.

Our results also indicate that water bodies must be well-vegetated, not only for increased nest survival, but also for increased species richness (White and Main 2005, Hudson and Bird 2009; Chapter 2). Red-winged Blackbirds were completely absent from the two study sites without vegetated water bodies; the importance of emergent vegetation is clear. For both golf courses and green spaces with water bodies, large cattail beds with intact old growth and interspersed water are critical for Red-winged Blackbirds, both in terms of presence (Hudson and Bird 2009; Chapter 2) and success (Short 1985). We witnessed the effects of increased vegetation firsthand, as a golf course manager decided not to cut and remove the summer's growth of cattails in preparation for winter as they had in previous years. The rate of blackbird occupation and success the following year rose with increasing cattail cover (e.g. increase from seven nests which all failed, to 25 nests, seven of which fledged young). Thus, encouraging cattail growth on golf course ponds to covert small reed patches to larger stands interspersed with open water would likely benefit Red-winged Blackbirds through increased protection from predators afforded by colonial nesting (Caccamise 1976, Olendorf et al. 2004). Managers should also strive to stabilize water levels as much as possible to reduce the number of drowned nests.

The simple and fairly straightforward results and recommendations presented here indicate the necessity for additional research and active management. It is critical that we change how these sites are designed and managed if they are to provide adequate breeding bird habitat. Managers should keep both the general recommendations and the species' specific requirements presented here in mind when modifying or creating wildlife friendly habitat, so that recreation and species conservation can coexist.

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## LITERATURE CITED

- Balogh, J. C., V. A. Gibeault, W. J. Walker, M. P. Kenna, and J. T. Snow. 1992.
  Background and overview of environmental issues. Pages 1-37 *in* Golf
  Course Management & Construction: Environmental Issues. (J. C. Balogh and W. J. Walker, Eds.). CRC Press, Lewis Publishers, Boca Raton, FL.
- Bart, J. 1977. Impact of human visitation on avian nesting success. Living Bird 16: 187-192.
- Beletsky, L. 1996. The Red-winged Blackbird: The Biology of a Strongly Polygynous Songbird. Academic Press, San Diego, CA.

- Best, L. B. and D. F. Stauffer. 1980. Factors affecting nesting success in riparian bird communities. Condor 82: 149-158.
- Blair, R. 2004. The effects of urban sprawl on birds at multiple levels of biological organization. Ecology and Society 9, article 2. [Online.] Available at www.ecologyandsociety.org/vol9/iss5/art2/
- Blair, R. B. 1996. Land use and avian species diversity along an urban gradient. Ecological Applications 6: 506-519.
- Borgmann, K. L. and A. D. Rodewald. 2004. Nest predation in an urbanizing landscape: the role of exotic shrubs. Ecological Applications 14: 1757-1765.
- Burhans, D. E. and F. R. Thompson III. 2001. Relationship of songbird nest concealment to nest fate and flushing behaviour of adults. Auk 118: 237-242.
- Burnham, K. P. and D. R. Anderson. 2002. Model Selection and Multi-Model Inference: A Practical Information-Theoretic Approach. Springer-Verlag, New York, NY.
- Burnham, K. P. and D. R. Anderson. 2004. Multimodel inference: understanding AIC and BIC in model selection. Sociological Methods & Research 33: 261-304.
- Caccamise, D. F. 1976. Nesting mortality in the Red-winged Blackbird. Auk 93: 517-534.
- Chin, E. H., O. P. Love, J. J. Verspoor, T. D. Williams, K. Rowley, and G. Burness. 2008. Juveniles exposed to embryonic corticosterone have enhanced flight performance. Proceedings of the Royal Society B: Biological Sciences 276: 499-505.
- Cimprich, D. A. and F. R. Moore. 1995. Gray Catbird (*Dumetella carolinensis*) in The Birds of North America Online. (A. Poole, Ed.). Cornell Lab of Ornithology, Ithaca, NY. [Online.] Available at bna.birds.cornell.edu/bna/species/167

- Clark, R. G. and T. D. Nudds. 1991. Habitat patch size and duck nesting success: the crucial experiments have not been performed. Wildlife Society Bulletin 19: 534-543.
- Clark, R. G. and D. Shutler. 1999. Avian habitat selection: Pattern from process in nest-site use by ducks? Ecology 80: 272-287.
- Collister, D. M. and S. Wilson. 2007. Contributions of weather and predation to reduced breeding success in a threatened northern Loggerhead Shrike population. Avian Conservation and Ecology 2: 11.
- Colwell, M. A. 1992. Wilson's Phalarope nest success is not influenced by vegetation concealment. Condor 94: 767-772.
- Dinsmore, S. J., G. C. White, and F. L. Knopf. 2002. Advanced techniques for modeling avian nest survival. Ecology 83: 3476-3488.
- Donehower, C. E. and D. M. Bird. 2008. Gull predation and breeding success of Common Eiders on Stratton Island, Maine. Waterbirds 31: 454-462.
- Donovan, T. M. and F. R. Thompson III. 2001. Modeling the ecological trap hypothesis: a habitat and demographic analysis for migrant songbirds. Ecological Applications 11: 871-882.
- Donovan, T. M., F. R. Thompson III, J. Faaborg, and J. R. Probst. 1995. Reproductive success of migratory birds in habitat sources and sinks. Conservation Biology 9: 1380-1395.
- Dyer, M. I., J. Pinowski, and B. Pinowska. 1977. Population dynamics. Pages 53-105 *in* Granivorous Birds in Ecosystems: Their Evolution, Populations, Energetics, Adaptations, Impact and Control. (J. Pinowski and S. C. Kendeigh, Eds.). Cambridge University Press, Cambridge, UK.
- Etterson, M. A., L. R. Nagy, and T. R. Robinson. 2007. Partitioning risk among different causes of nest failure. Auk 124: 432-443.
- Filliater, T. S., R. Breitwisch, and P. M. Nealen. 1994. Predation on Northern Cardinals nests: does choice of nest site matter? Condor 96: 761-768.
- Flaspohler, D. J., S. A. Temple, and R. N. Rosenfield. 2000. Relationship between nest success and concealment in two ground-nesting passerines. Journal of Field Ornithology 71: 736-747.

- Francis, C. D., C. P. Ortega, and A. Cruz. 2009. Noise pollution changes avian communities and species interactions. Current Biology 19: 1-5.
- Gauthier, J. and Y. Aubry. 1996. The Breeding Birds of Quebec: Atlas of the Breeding Birds of Southern Quebec. Association québécoise des groupes d'ornithologues, Province of Quebec Society for the Protection of Birds, Canadian Wildlife Service, Environment Canada, Quebec Region, Montreal, QC.
- Gering, J. C. and R. B. Blair. 1999. Predation on artificial bird nests along an urban gradient: predatory risk or relaxation in urban environments? Ecography 22: 532-541.
- Götmark, F. 1992. The effects of investigator disturbance on nesting birds. Current Ornithology 9: 63-104.
- Grandmaison, D. D. and G. J. Niemi. 2007. Local and landscape influence on Red-winged Blackbird (*Agelaius phoeniceus*) nest success in Great Lakes coastal wetlands. Journal of Great Lakes Research 33: 292-304.
- Grant, T. A., E. M. Madden, T. L. Shaffer, P. J. Pietz, G. B. Berkey, and N. J. Kadrmas. 2006. Nest survival of Clay-Colored and Vesper Sparrows in relation to woodland edge in mixed-grass prairies. Journal of Wildlife Management 70: 691-701.
- Grant, T. A., T. L. Shaffer, E. M. Madden, and P. J. Pietz. 2005. Time-specific variation in passerine nest survival: new insights into old questions. Auk 122: 661-672.
- Green, B. H. and I. C. Marshall. 1987. An assessment of the role of golf courses in Kent, England, in protecting wildlife and landscapes. Landscape and Urban Planning 14: 143-154.
- Halkin, S. L. and S. U. Linville. 1999. Northern Cardinal (*Cardinalis cardinalis*) *in* The Birds of North America Online. (A. Poole, Ed.). Cornell Lab of Ornithology, Ithaca, NY. [Online.] Available at bna.birds.cornell.edu/bna/species/440
- Haskell, D. G., A. M. Knupp, and M. C. Schneider. 2001. Nest predator abundance and urbanization. Pages 241-258 in Avian Ecology and

Conservation in an Urbanizing World. (J. M. Marzluff, R. Bowman, and R. Donnelly, Eds.). Kluwer Academic Publishers, Norwell, MA.

- Heltzel, J. and S. L. Earnst. 2006. Factors influencing nest success of songbirds in aspen and willow riparian areas in the Great Basin. Condor 108: 842-855.
- Heske, E. J., S. K. Robinson, and J. D. Brawn. 2001. Nest predation and Neotropical migrant songbirds: piecing together the fragments. Wildlife Society Bulletin 29: 52-61.
- Howard, D. V. 1974. Urban robins: a population study. Pages 67-75 in A Symposium on Wildlife in an Urbanizing Environment. (J. H. Noyes and D. R. Progulske, Eds.). Cooperative Extension Service, University of Massachusetts, Springfield, MA.
- Hudson, M.-A. R. and D. M. Bird. 2006. An affordable computerized camera technique for monitoring bird nests. Wildlife Society Bulletin 34: 1455-1457.
- Hudson, M.-A. R. and D. M. Bird. 2009. Recommendations for design and management of golf courses and green spaces based on surveys of breeding bird communities in Montreal. Landscape and Urban Planning 92: 335-346.
- Jokimäki, J. and E. Huhta. 2000. Artificial nest predation and abundance of birds along an urban gradient. Condor 102: 838-847.
- Jokimäki, J., M.-L. Kaisanlahti-Jokimäki, A. Sorace, E. Fernández-Juricic, I. Rodriguez-Prieto, and M. D. Jimenez. 2005. Evaluation of the "safe nesting zone" hypothesis across an urban gradient: a multi-scale study. Ecography 28: 59-70.
- Klimstra, W. D. and W. O. Stieglitz. 1957. Notes on reproductive activities of robins in Iowa and Illinois. Wilson Bulletin 69: 333-337.
- Knutson, M. G., B. R. Gray, and M. S. Meier. 2007. Comparing the effects of local, landscape, and temporal factors on forest bird nest survival using logistic-exposure models. Pages 105-116 *in* Studies in Avian Biology no. 34: Beyond Mayfield, Measurements of nest-survival data. (C. D. Marti, S.

L. Jones, and G. R. Geupel, Eds.). Cooper Ornithological Society, Ephrata, PA.

- Kus, B. E., B. L. Peterson, and D. H. Deutschman. 2008. A multiscale analysis of nest predation on Least Bell's Vireos (*Vireo bellii pusillus*). Auk 125: 277-284.
- Larivière, S. 1999. Reasons why predators cannot be inferred from nest remains. Condor 101: 718-721.
- LeClerc, J. E., J. P. K. Che, J. P. Swaddle, and D. A. Cristol. 2005. Reproductive success and developmental stability of eastern bluebirds on golf courses: evidence that golf courses can be productive. Wildlife Society Bulletin 33: 483-493.
- LeClerc, J. E. and D. A. Cristol. 2005. Are golf courses providing habitat for birds of conservation concern in Virginia? Wildlife Society Bulletin 33: 463-470.
- Lindsey, J. K. and P. J. Lindsey. 2001. Detecting covariates with non-random missing values in a survey of primary education in Madagascar. Journal of the Royal Statistical Society. Series A (Statistics in Society) 164: 327-338.
- Lowther, P. E., C. Celada, N. K. Klein, C. C. Rimmer, and D. A. Spector. 1999. Yellow Warbler (*Dendroica petechia*) in The Birds of North America Online. (A. Poole, Ed.). Cornell Lab of Ornithology, Ithaca, NY. [Online.] Available at bna.birds.cornell.edu/bna/species/454
- Martin, T. E. 1992. Breeding productivity considerations: what are the appropriate habitat features for management? Pages 455-473 *in* Ecology and Conservation of Neotropical Migrant Landbirds.( J. M. Hagan III and D. W. Johnston, Eds.). Smithsonian Institution Press, Washington, DC.
- Martin, T. E. 1993. Nest predation and nest sites. BioScience 43: 523-532.
- Martin, T. E. 1995. Avian life history evolution in relation to nest sites, nest predation, and food. Ecological Monographs 65: 101-127.
- Martin, T. E. and G. R. Geupel. 1993. Nest-monitoring plots: methods for locating nests and monitoring success. Journal of Field Ornithology 64: 507-519.

- Martin, T. E. and J. J. Roper. 1988. Nest predation and nest-site selection of a western population of the Hermit Thrush. Condor 90: 51-57.
- Mayer-Gross, H., H. Q. P. Crick, and J. J. D. Greenwood. 1997. The effect of observers visiting the nests of passerines: An experimental study. Bird Study 44: 53-65.
- Mayfield, H. 1961. Nesting success calculated from exposure. Wilson Bulletin 73: 255-261.
- Melampy, M. N., E. L. Kershner, and M. A. Jones. 1999. Nest Predation in Suburban and Rural Woodlots of Northern Ohio. American Midland Naturalist 141: 284-292.
- Merola-Zwartjes, M. and J. P. DeLong. 2005. Avian species assemblages in New Mexico golf courses: surrogate riparian habitat for birds? Wildlife Society Bulletin 33: 435-447.
- Middleton, A. L. 1998. Chipping Sparrow (*Spizella passerina*) in The Birds of North America Online. (A. Poole, Ed.). Cornell Lab of Ornithology, Ithaca, NY. [Online.] Available at bna.birds.cornell.edu/bna/species/334
- Miller, J. R. and N. T. Hobbs. 2000. Recreational trails, human activity, and nest predation in lowland riparian areas. Landscape and Urban Planning 50: 227-236.
- Morneau, F., C. Lépine, R. Décarie, M.-A. Villard, and J.-L. DesGranges. 1995. Reproduction of American Robin (*Turdus migratorius*) in a suburban environment. Landscape and Urban Planning 32: 55-62.
- Morrison, S. A. and D. T. Bolger. 2002. Lack of an urban edge effect on reproduction in a fragmentation-sensitive sparrow. Ecological Applications 12: 398-411.
- Moul, I. E. and J. E. Elliott 1992. A Survey of Pesticide Use and Bird Activity on Selected Golf Courses in British Columbia. Technical Report Series No. 163. Canadian Wildlife Service, Pacific and Yukon Region, BC.
- O'Grady, D. R., D. P. Hill, and R. M. R. Barclay. 1996. Nest visitation by humans does not increase predation on Chestnut-collared Longspur eggs and young. Journal of Field Ornithology 67: 275-280.

- Olendorf, R., T. Getty, and K. Scribner. 2004. Cooperative nest defence in Redwinged Blackbirds: reciprocal altruism, kinship or by-product mutualism? Proceedings of the Royal Society B: Biological Sciences 271: 177-182.
- Ortega, C. P., J. C. Ortega, C. A. Rapp, S. Vorisek, S. A. Backensto, and D. W. Palmer. 1997. Effect of research activity on the success of American Robin nests. Journal of Wildlife Management 61: 948-952.
- Ortega, Y. K., K. S. McKelvey, and D. L. Six. 2007. Invasion of an exotic forb impacts reproductive success and site fidelity of a migratory songbird. Oecologia 2006: 340-351.
- Patten, M. A. and D. T. Bolger. 2003. Variation in top-down control of avian reproductive success across a fragmentation gradient. Oikos 101: 479-488.
- Peak, R. G., F. R. Thompson III, and T. L. Shaffer. 2004. Factors affecting songbird nest survival in riparian forests in a midwestern agricultural landscape. Auk 121: 726-737.
- Picman, J., M. Leonard, and A. Horn. 1988. Antipredation role of clumped nesting by marsh-nesting Red-winged Blackbirds. Behavioral Ecology and Sociobiology 22: 9-15.
- Picman, J., M. L. Milks, and M. Leptich. 1993. Patterns of predation on passerine nests in marshes: effects of water depth and distance from edge. Auk 110: 89-94.
- Picozzi, N. 1975. Crow predation on marked nests. Journal of Wildlife Management 39: 151-155.
- Pribil, S. 1998. Reproductive success is a misleading indicator of nest-site preferences in the Red-winged Blackbird. Canadian Journal of Zoology 76: 2227-2234.
- Rangen, S. A., R. G. Clark, and K. A. Hobson. 1999. Influence of nest-site vegetation and predator community on the success of artificial songbird nests. Canadian Journal of Zoology 77: 1676-1681.
- Reidy, J. L., M. M. Stake, and F. R. Thompson III. 2008. Golden-cheeked Warbler nest mortality and predators in urban and rural landscapes. Condor 110: 458-466.

- Ricklefs, R. E. 1969. An analysis of nesting mortality in birds. Smithsonian Contributions to Zoology 9: 1-48.
- Rodewald, A. D. 2004. Nest-searching cues and studies of nest-site selection and nesting success. Journal of Field Ornithology 75: 31-39.
- Sallabanks, R. and F. C. James. 1999. American Robin (*Turdus migratorius*) in The Birds of North America Online. (A. Poole, Ed.). Cornell Lab of Ornithology, Ithaca, NY. [Online.] Available at bna.birds.cornell.edu/bna/species/462
- Schmidt, K. A. 1999. Foraging theory as a conceptual framework for studying nest predation. Oikos 85: 151-160.
- Sealy, S. G. 1994. Observed acts of egg destruction, egg removal, and predation on nests of passerine birds at Delta Marsh, Manitoba. Canadian Field-Naturalist 108: 41-51.
- Seitz, L. C. and D. A. Zegers. 1993. An experimental study of nest predation in adjacent deciduous, coniferous and successional habitats. Condor 95: 297-304.
- Shafer, C. L. 1995. Values and shortcomings of small reserves. BioScience 45: 80-88.
- Short, H. L. 1985. Habitat Suitability Index Models: Red-winged Blackbird. Biological Report no. 82. U.S. Fish and Wildlife Service, Fort Collins, CO.
- Smith, L. A., E. Nol, D. M. Burke, and K. E. Elliott. 2007. Nest-site selection of Rose-breasted Grosbeaks in southern Ontario. Wilson Journal of Ornithology 119: 151-161.
- Smith, M. D., C. J. Conway, and L. A. Ellis. 2005. Burrowing owl nesting productivity: a comparison between artificial and natural burrows on and off golf courses. Wildlife Society Bulletin 33: 454-462.
- Sorace, A. and M. Gustin. 2009. Distribution of generalist and specialist predators along urban gradients. Landscape and Urban Planning 90: 111-118.
- Sorace, A. and M. Visentin. 2007. Avian diversity on golf courses and surrounding landscapes in Italy. Landscape and Urban Planning 81: 81-90.

- Soulé, M. E., D. T. Bolger, A. C. Alberts, J. Wright, M. Sorice, and S. Hill. 1988. Reconstructed dynamics of rapid extinctions of chaparral-requiring birds in urban habitat islands. Conservation Biology 2: 75-92.
- Stanback, M. T. and M. L. Seifert. 2005. A comparison of eastern bluebird reproductive parameters in golf and rural habitats. Wildlife Society Bulletin 33: 471-482.
- Styrsky, J. N. 2005. Influence of predation on nest-site reuse by an open-cup nesting Neotropical passerine. Condor 107: 133-137.
- Sugden, L. G. and G. W. Beyersbergen. 1986. Effect of density and concealment on American Crow predation of simulated duck nests. Journal of Wildlife Management 50: 9-14.
- Tanner, R. A. and A. C. Gange. 2005. Effects of golf courses on local biodiversity. Landscape and Urban Planning 71: 137-146.
- Terman, M. R. 1997. Natural links: naturalistic golf courses as wildlife habitat. Landscape and Urban Planning 38: 183-197.
- Tewksbury, J. J., L. Garner, S. Garner, J. D. Lloyd, V. Saab, and T. E. Martin. 2006. Tests of landscape influence: Nest predation and brood parasitism in fragmented ecosystems. Ecology 87: 759-768.
- Tewksbury, J. J., S. J. Hejl, and T. E. Martin. 1998. Breeding productivity does not decline with increasing fragmentation in a western landscape. Ecology 79: 2890-2903.
- Thompson III, F. R. and D. E. Burhans. 2003. Predation of songbird nests differs by predator and between field and forest habitats. Journal of Wildlife Management 67: 408-416.
- Vierling, K. T. 2000. Source and sink habitats or Red-winged Blackbirds in a rural/suburban landscape. Ecological Applications 10: 1211-1218.
- Weatherhead, P. J. 2005. Effects of climate variation on timing of nesting, reproductive success, and offspring sex ratios of Red-winged Blackbirds. Oecologia 144: 168-175.
- Weatherhead, P. J. and S. J. Sommerer. 2001. Breeding synchrony and nest predation in Red-winged Blackbirds. Ecology 82: 1632-1641.

- Weidinger, K. 2002. Interactive effects of concealment, parental behaviour and predators on the survival of open passerine nests. Journal of Animal Ecology 71: 424-437.
- Weidinger, K. 2008. Nest monitoring does not increase nest predation in opennesting songbirds: inference from continuous nest-survival data. Auk 125: 859-868.
- White, C. L. and M. B. Main. 2005. Waterbird use of created wetlands in golfcourse landscapes. Wildlife Society Bulletin 33: 411-421.
- White, G. C. and K. P. Burnham. 1999. Program MARK: survival estimation from populations of marked animals. Bird Study 46 (suppl.): S120-S139.
- Wilcove, D. S. 1985. Nest predation in forest tracts and the decline of migratory songbirds. Ecology 66: 1211-1214.
- Witmer, M. C., D. J. Mountjoy and L. Elliot. 1997. Cedar Waxwing (*Bombycilla cedrorum*) in The Birds of North America Online. (A. Poole, Ed.). Cornell Lab of Ornithology, Ithaca, NY. [Online.] Available at bna.birds.cornell.edu/bna/species/309
- Yasukawa, K. and W. A. Searcy. 1995. Red-winged Blackbird (Agelaius phoeniceus in The Birds of North America Online. (A. Poole, Ed.). Cornell Lab of Ornithology, Ithaca, NY. [Online.] Available at bna.birds.cornell.edu/bna/species/184
- Zar, J. H. 1999. Biostatistical Analysis. Upper Saddle River, New Jersey, Prentice Hall.



Fig. 1. Map of the six sites studied in 2003-2005 in the greater Montreal area, Quebec. Sites shaded in black represent non-certified golf courses, those shaded with the black-and-white pattern are Audubon-certified golf courses, and those highlighted in white are green spaces. See Table 1 for site codes and descriptions.

Table 1. Landscape and site-use description of six Montreal study sites, listed from smallest to largest. Site size, usable area and number of houses within a 200-m buffer is presented for all sites; number of holes on each golf course and average number of rounds played per year (1 round = 1 person playing 18 holes) is presented for golf courses.

Study Site	Size (ha)	Upland nesting habitat (ha) <sup>a</sup>	Wetland nesting habitat (ha) <sup>b</sup>	Houses /area	# holes	Rounds played per year	Year created	Landscape-level site description
Stoneycroft Wildlife Area (SW)	22	22	2.1	0.5	-	-	1971	Rural/Exurban: private wildlife research area bordered by mixed forest, agriculture
Terra Cotta Park (TC)	34	31	0.0	10.6	-	-	1971	Suburban/Urban: recreational nature park surrounded by residential development
Beaconsfield Golf Club (BG)	65	28	0.9	11.7	18	6,468	1904	Suburban/Urban: Audubon-certified, surrounded by residential development
Ile Perrot Golf and Country Club (IP)	69	33	1.3	1.3	18	8,289	1974	Rural/Exurban: non-certified, surrounded by large-lot residential development, deciduous forest, agriculture
Elm Ridge Country Club (EG)	163	85	0.0	0.3	36	13,444	1959	Rural/Exurban: non-certified, surrounded by agriculture, deciduous forest, large-lot residential development
Royal Montreal Golf Club (RM)	215	107	5.0	0.3	45	50,000	1959	Rural/Exurban: Audubon-certified, surrounded by agriculture, deciduous forest, large-lot residential development

<sup>a</sup> Defined as site size minus area covered by grass; represents minimum nesting area available to upland-nesting birds.

<sup>b</sup> Defined as minimum area covered by vegetated water bodies; represents minimum nesting area available to Red-winged Blackbird.

Table 2. Mean nest density, proportion of successful nests, number of nests parasitized by the Brown-headed Cowbird (BHCO), and causes of nest failure (number failed due to predation, brood parasitism, and other causes) for the most common open-cup nesters found on two Montreal green spaces and four golf courses studied from 2003-2005. Shaded lines represent golf courses, clear lines represent green spaces. Bolded *P*-values represent marginal significance at  $\alpha = 0.10$ .

	Mean nest density	Mean nest density Proportion		D :/: 1	Causes of nest failure			
Species <sup>a</sup>	(nests/ha) ± SE	successful nests $\pm$ SE	fate <sup>b</sup>	nests <sup>c</sup>	Predation	Brood parasitism <sup>d</sup>	Other <sup>e</sup>	
AMRO	$0.96\pm0.14$	$0.35\pm0.10$	4	1	26	1	1	
	$0.81\pm0.36$	$0.44\pm0.04$	15	0	64 <sup>f</sup>	0	7	
M-W U test	U = 2.0 P = 0.355	U = 3.0 P = 0.643						
GRCA	$0.71\pm0.11$	$0.78\pm0.06$	2	1	6	0	1	
	$0.12\pm0.07$	$0.58\pm0.21$	1	1	4	0	0	
M-W U test	U = 0.0 <b>P</b> = <b>0.064</b>	U = 3.0 P = 0.643						
CEDW	$0.49\pm0.33$	$0.54\pm0.26$	1	0	12	0	1	
	$0.15\pm0.08$	$0.13\pm0.13$	3	0	24 <sup>f</sup>	0	2	
M-W U test	U = 1.0 P = 0.165	U = 1.0 P = 0.165						
YWAR	$0.84\pm0.48$	$0.26\pm0.02$	2	16	17	9	2	
	$0.07\pm0.04$	$0.19\pm0.11$	2	5	$3^{\rm f}$	5	0	
M-W U test	U = 0.0 <b>P</b> = <b>0.064</b>	U = 4.0 P = 1.000						
NOCA	$0.41\pm0.09$	$0.37\pm0.23$	0	1	11	0	1	
	$0.19\pm0.10$	$0.53\pm0.17$	1	7	15 <sup>f</sup>	2	2	

M-W U test	U = 2.0 P = 0.355	U = 3.0 P = 0.643					
CHSP	$0.00 \pm 0.00$	$0.0 \pm 0.0$	0	0	0	0	0
	$0.14\pm0.03$	$0.75\pm0.42$	2	16	12	11	2
M-W U test	U = 0.0 <b>P = 0.064</b>	U = 0.0 <b>P</b> = 0.064					
SOSP	$0.26\pm0.20$	$0.60\pm0.10$	1	2	3	0	0
	$0.10\pm0.04$	$0.43\pm0.14$	1	8	6	1	0
M-W U test	U = 2.0 P = 0.355	U = 2.5 P = 0.487					
RWBL	$27.1 \pm 27.1^{ m g}$	$0.41\pm0.09$	11	3	59	2	6
	$37.3\pm22.6^{\rm g}$	$0.27\pm0.11$	23	2	57 <sup>f</sup>	0	29
M-W U test	U = 3.5 P = 0.817	U = 2.0 P = 0.355					
COGR	$0.18\pm0.18$	$0.06\pm0.06$	0	0	7	0	0
	$0.26\pm0.23$	$0.45\pm0.26$	0	0	6	0	1
M-W U test	U = 4.0 P = 1.000	U = 3.0 P = 0.643					
All species	$4.7\pm1.5^{\rm h}$	$0.44\pm0.10$	24	27	161	13	12
All species	$2.0\pm0.8^{\rm h}$	$0.41\pm0.05$	52	40	201	20	43
M-W U test	U = 1.0 P = 0.165	U = 2.0 P = 0.355					
Total	2.9 <sup>h</sup>	0.41	76	67	362	33	55

<sup>a</sup> AMRO = American Robin, GRCA = Gray Catbird, CEDW = Cedar Waxwing, YWAR = Yellow Warbler, NOCA = Northern

Cardinal, CHSP = Chipping Sparrow, SOSP = Song Sparrow, RWBL = Red-winged Blackbird, COGR = Common Grackle.

<sup>b</sup> Nest found empty; fate unknown.

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<sup>c</sup> Parasitized nests did not necessarily fail.

<sup>d</sup> Brood parasitism led to nest failure when: nest abandoned after BHCO egg laid, only BHCO nestling fledged, host eggs destroyed following BHCO egg ejection by host species.

<sup>e</sup>Other causes of failure include abandonment, poor nest construction or weather.

<sup>f</sup>Landscaping likely caused predation of two nests or less.

<sup>g</sup> Red-winged Blackbird absent from one golf course and one green space. Mean golf course density without those sites =  $49.7 \pm 26.7$ and for green spaces =  $54.3 \pm 0.0$ .

<sup>h</sup> Excludes Red-winged Blackbirds.

Table 3. Nest success rates on golf courses and green spaces for the most common species nesting in Montreal from 2003-2005. Nest success rates were calculated by raising the DSR from the model constraining survival to site type (golf course vs. green space) to the power of the length of the nesting period. Mayfield nest success rates based on the null model using Program MARK are also provided for comparison.

Species	Number	Nesting	Golf course nest success		Green space nest success		Mayfield	
Species	of nests	(days)	Estimate (%)	95% CI	Estimate (%)	95% CI	Estimate (%)	95% CI
American Robin	156	30	32.7	23.7, 42.0	22.9	11.2, 37.4	30.1	22.6, 37.9
Cedar Waxwing	44	31	0.0	0.0, 2.0	22.3	13.6, 52.6	4.3	3.6, 18.4
Gray Catbird	51	27	73.9	39.3, 90.7	74.5	52.0, 87.6	74.3	56.5, 85.7
Yellow Warbler	31	24	49.0	11.3, 79.6	29.6	12.4, 49.5	34.1	17.4, 51.9
Northern Cardinal	44	24	27.5	11.8, 46.1	39.2	14.2, 64.1	31.6	17.4, 46.9
Chipping Sparrow	25	24	24.0	8.2, 44.6	-	-	24.0	8.2, 44.6
Song Sparrow	27	26	31.7	10.2, 56.5	60.4	21.2, 85.0	42.6	21.6, 62.5
Red-winged Blackbird	226	27	42.1	32.6, 51.3	26.9	18.3, 36.1	35.1	28.5, 41.8
Common Grackle	40	28	75.1	50.4, 88.8	4.1	0.1, 22.3	54.4	34.3, 70.8



Fig. 2. Percent differences in nest success rates between birds nesting on Montreal golf courses and green spaces in 2003-2005. Light gray bars represent golf course rates > green space rates; dark gray bars represent golf course rates < green space rates. Asterisks represent species with < 4% overlap of 95% CI, suggesting statistically significant differences in rates.

Table 4. Model support for hypotheses that temporal, site- and nest-scale variables influence the daily survival rates of all uplandnesting species found on Montreal golf courses and other suburban green spaces in 2003-2005. Akaike's Information Criterion corrected for small sample size (AIC<sub>c</sub>),  $\Delta i$  (AIC<sub>c</sub> model *i* - AIC<sub>c</sub> minimum), Akaike weights (*w<sub>i</sub>*), the number of model parameters (K), and model deviance are shown for all models.

Model	AICc	$\Delta i$	Wi	K	Deviance
{temporal <sup>a</sup> + patch}	1261.20	0.00	0.74	18	1225.08
$\{\text{temporal} + \text{site}^{\text{b}} + \text{patch}^{\text{c}}\}$	1263.53	2.34	0.23	20	1223.39
$\{\text{temporal} + \text{patch} + \text{nest-site}^d\}$	1268.00	6.81	0.02	23	1221.81
{global}	1270.42	9.22	0.01	25	1220.19
{site + patch}	1299.33	38.14	0.00	14	1271.26
{patch}	1299.73	38.53	0.00	12	1275.68
{temporal + site + nest-site}	1300.56	39.37	0.00	14	1272.49
$\{site + patch + nest-site\}$	1302.12	40.92	0.00	12	1278.06
{temporal + site}	1304.44	43.24	0.00	19	1266.31
{patch + nest-site}	1304.50	43.31	0.00	17	1270.40
{temporal + nest-site}	1306.66	45.46	0.00	9	1288.63
{temporal}	1307.72	46.52	0.00	7	1293.70
{site + nest-site}	1345.97	84.78	0.00	8	1329.95
{nest-site}	1349.66	88.46	0.00	6	1337.64
{site}	1357.28	96.09	0.00	3	1351.28
<i>{</i> . <i>}</i>	1360.34	99.14	0.00	1	1358.34

<sup>a</sup> temporal = age + age<sup>2</sup> + season + season<sup>2</sup> + year

<sup>b</sup> site = location + house

<sup>c</sup> patch = substrate + arrangement + substrate\*arrangement

<sup>d</sup> nest-site = edge + cover + disturbance + nest height

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Table 5. Beta estimates ( $\beta$ ) and 95% CIs are presented for all parameters occurring in the top model influencing nest survival of all upland-nesting species on Montreal golf courses and green spaces in 2003-2005. Influential variables (i.e. 95% CIs does not cross zero) are bolded.  $\dagger$  denotes reference category for categorical variables.

Variable	β	95% CI
Age	0.002	-0.070, 0.074
Age <sup>2</sup>	0.002	-0.070, 0.074
Season	0.005	-0.005, 0.015
Season <sup>2</sup>	0.000	0.000, 0.000
Year		
2003	0.441	0.037, 0.845
2004	0.375	0.056, 0.694
2005	†	*
Substrate		
Conifer	-3.400	-5.607, -1.193
Deciduous	-2.852	-4.917, -0.786
Shrub	-1.489	-3.494, 0.516
Artificial	†	Ť
Arrangement		
Single	-1.175	-3.456, 1.106
Patch	-1.330	-3.613, 0.954
Stand	†	Ť
Substrate*arrangement		
Single conifer	2.039	-0.458, 4.537
Patch of conifers	3.292	0.768, 5.816
Single deciduous	1.129	-1.277, 3.535
Patch of deciduous	1.610	-0.868, 4.088
Single shrub	0.413	-1.919, 2.745
Patch of shrubs	0.735	-1.605, 3.075
All combinations of artificia	l and stand	Ť



Fig. 3. Estimated predation rates for a 27-day nesting period derived from daily survival rates plotted against the log-transformed number of houses within a 200m buffer around each site divided by site area. RM: Royal Montreal Golf Club; EG: Elm Ridge Country Club; SW: Stoneycroft Wildlife Area; IP: Ile Perrot Golf and Country Club; TC: Terra Cotta Park; BG: Beaconsfield Golf Club. Dotted lines indicate the limits of the 95% CI regression bands.

Table 6. Level-specific nest success rates with associated 95% CIs for influential categorical variables for all upland-nesting species on Montreal golf courses and green spaces in 2003-2005. Nest success was calculated by raising DSR estimates generated from the null model to the power of the average nesting period (27 days).

Variable	Nest success (%)	95% CI (%)
Year		
2003	46.3	34.1, 57.7
2004	41.5	33.1, 49.7
2005	29.0	22.8, 35.6
Substrate		
Conifer	33.8	25.9, 41.9
Deciduous	8.7	3.6, 16.7
Shrub	41.0	33.8, 48.0
Artificial	72.7	51.3, 85.9
Arrangement		
Single	23.8	17.7, 30.5
Patch	42.6	33.8, 51.5
Stand	48.1	38.2, 57.3
Substrate*arrangement		
Single conifer	18.8	11.2, 28.0
Patch of conifers	58.4	44.2, 70.2
Single deciduous	8.0	2.0, 19.8
Patch of deciduous	9.5	1.0, 31.3
Single shrub	30.0	18.4, 42.5
Patch of shrubs	30.8	19.3, 43.2

Table 7. Univariate comparisons (mean  $\pm$  SE) and proportions of habitat variables at successful and depredated nests of American Robin, and at nests on golf courses and green spaces in Montreal from 2003-2005. Bolded *P*-values indicate t- or z-test comparisons that were significant at  $\alpha = 0.05$ .

Habitat variable	Golf course nests (n = 115)	Green space nests (n = 42)	Test statistic	Successful nests (n = 72)	Depredated nests (n = 85)	Test statistic
Nest height (m)	$2.8\pm0.1$	$2.1 \pm 0.1$	t <sub>155</sub> = 2.52, <b><i>P</i> = 0.013</b>	$2.5\pm0.1$	$2.7\pm0.2$	$t_{155} = -0.8, P = 0.417$
Concealment (%)	$63.2\pm1.7$	$56.8\pm2.5$	t <sub>155</sub> = 1.99, <b><i>P</i> = 0.048</b>	$65.5\pm2.1$	$58.2\pm1.9$	$t_{155} = 2.59, P = 0.010$
Edge (m)	$1.5 \pm 0.2$	$1.5 \pm 0.1$	$t_{155} = -0.85, P = 0.399$	$1.3\pm0.2$	$1.7\pm0.1$	t <sub>155</sub> = -2.63, <b><i>P</i></b> = <b>0.009</b>
Proportion of nests	by substrate cla	SS				
Conifer	0.53	0.10	<i>P</i> < 0.001	0.44	0.40	P = 0.614
Deciduous	0.21	0.12	P = 0.205	0.10	0.26	P = 0.011
Artificial	0.11	0.00	<i>P</i> < 0.001	0.14	0.04	P = 0.027
Shrub or vine	0.09	0.29	P = 0.002	0.15	0.14	P = 0.859
Hawthorn	0.05	0.49	<i>P</i> < 0.001	0.17	0.16	P = 0.867
Proportion of nests	by substrate arr	angement class				
Single	0.43	0.37	P = 0.504	0.32	0.49	P = 0.033
Line	0.22	0.00	P = 0.001	0.28	0.07	<i>P</i> < 0.001
Patch	0.14	0.17	P = 0.642	0.10	0.19	P = 0.116
Stand	0.21	0.46	P = 0.002	0.31	0.25	P = 0.404
Proportion of nests	by disturbance	class				
Low	0.00	0.46	<i>P</i> < 0.001	0.08	0.15	P = 0.177
Medium	0.53	0.54	P = 0.912	0.49	0.56	P = 0.383
High	0.47	0.00	<i>P</i> < 0.001	0.43	0.28	P = 0.051

Table 8. Univariate comparisons (mean  $\pm$  SE) and proportions of habitat variables at successful and depredated nests of Gray Catbird, and at nests on golf courses and green spaces in Montreal from 2003-2005. Bolded *P*-values indicate t- or z-test comparisons that were significant at  $\alpha = 0.05$ .

Habitat variable	Golf course nests (n = 16)	Green space nests (n = 35)	Test statistic	Successful nests (n = 42)	Depredated nests (n = 9)	Test statistic	
Nest height (m)	$1.5 \pm 0.1$	$2.3 \pm 0.1$	t <sub>49</sub> = -3.63, <i>P</i> < 0.001	$2.0 \pm 0.1$	$1.9 \pm 0.3$	$t_{49} = 0.49, P = 0.626$	
Concealment (%)	$63.4\pm5.0$	$54.5\pm2.7$	$t_{49} = 1.72, P = 0.092$	$54.3\pm2.6$	$71.5\pm4.6$	$t_{49} = -2.88, P = 0.006$	
Edge (m)	$1.6 \pm 0.3$	$1.9 \pm 0.3$	$t_{49} = -0.67, P = 0.503$	$1.8\pm0.2$	$2.1 \pm 0.3$	$t_{49} = -1.11, P = 0.271$	
Proportion of nests	by substrate cl	ass					
Buckthorn	0.13	0.18	P = 0.657	0.15	0.25	P = 0.491	
Honeysuckle	0.87	0.12	<i>P</i> < 0.001	0.34	0.38	P = 0.829	
Hawthorn	0.00	0.71	<i>P</i> < 0.001	0.51	0.38	P = 0.504	
Proportion of nests	by substrate ar	rangement class	5				
Single or line	0.13	0.03	P = 0.174	0.05	0.11	P = 0.497	
Patch	0.25	0.29	P = 0.769	0.26	0.33	P = 0.671	
Stand	0.63	0.69	P = 0.674	0.69	0.56	P = 0.456	
Proportion of nests by disturbance class							
Low	0.19	0.57	P = 0.015	0.50	0.22	P = 0.132	
Medium	0.50	0.43	P = 0.643	0.43	0.56	P = 0.481	
High	0.31	0.00	<i>P</i> = 0.001	0.07	0.22	P = 0.173	

Table 9. Univariate comparisons (mean  $\pm$  SE) and proportions of habitat variables at successful and depredated nests of Northern Cardinal, and at nests on golf courses and green spaces in Montreal from 2003-2005. Bolded *P*-values indicate t- or z-test comparisons that were significant at  $\alpha = 0.05$ .

	Golf course	Green space		Successful	Depredated	
Habitat variable	nests	nests	Test statistic	nests	nests	Test statistic
	( <i>n</i> = 28)	( <i>n</i> = 16)		( <i>n</i> = 22)	( <i>n</i> = 22)	
Nest height (m)	$2.0\pm0.1$	$2.5\pm0.2$	$t_{42} = -2.21, P = 0.033$	$2.3\pm0.2$	$2.1 \pm 0.2$	$t_{42} = 0.57, P = 0.570$
Concealment (%)	$66.2\pm2.9$	$48.9\pm5.0$	$t_{42} = 3.24, P = 0.002$	$58.1\pm4.1$	$61.7\pm4.0$	$t_{42} = -0.64, P = 0.526$
Edge (m)	$1.2\pm0.2$	$1.6 \pm 0.2$	$t_{42} = -2.18, P = 0.035$	$1.4 \pm 0.2$	$1.4\pm0.3$	$t_{42} = -0.05, P = 0.962$
Proportion of nests	s by substrate o	class				
Hawthorn	0.11	0.75	P = 0.001	0.45	0.23	P = 0.131
Conifer	0.57	0.06	P = 0.018	0.23	0.55	P = 0.035
Shrub or vine	0.32	0.19	P = 0.346	0.32	0.23	P = 0.508
Proportion of nests	s by substrate a	arrangement cla	ass			
Single	0.43	0.07	P = 0.017	0.23	0.41	P = 0.208
Patch or line	0.39	0.14	P = 0.090	0.36	0.32	P = 0.781
Stand	0.18	0.36	P = 0.193	0.41	0.27	P = 0.333
Proportion of nests	s by disturbanc	ce class				
Low	0.04	0.11	P = 0.352	0.00	0.18	P = 0.043
Medium	0.57	0.46	P = 0.498	0.77	0.55	P = 0.131
High	0.39	0.00	<i>P</i> = 0.006	0.23	0.27	P = 0.761

Table 10. Univariate comparisons (mean  $\pm$  SE) and proportions of habitat variables at all successful and depredated nests of Cedar Waxwing, and at nests on golf courses and green spaces in Montreal from 2003-2005. Bolded *P*-values indicate t- or z-test comparisons that were significant at  $\alpha = 0.05$ .

	Golf course	Green space		Successful	Depredated	
Habitat variable	nests	nests	Test statistic	nests	nests	Test statistic
	( <i>n</i> = 23)	( <i>n</i> = 21)		( <i>n</i> = 10)	( <i>n</i> = 34)	
Nest height (m)	$4.5 \pm 0.4$	$2.3 \pm 0.2$	$t_{42} = 6.01, P < 0.001$	$2.8\pm0.6$	$3.7\pm0.3$	$t_{42} = -1.55, P = 0.129$
Concealment (%)	$62.1\pm4.6$	$57.7\pm4.2$	$t_{42} = 0.70, P = 0.489$	$60.2\pm7.9$	$59.9\pm3.4$	$t_{42} = 0.046, P = 0.964$
Edge (m)	$1.4 \pm 0.2$	$0.9 \pm 0.1$	t <sub>42</sub> = 3.14, <b><i>P</i> = 0.003</b>	$0.9\pm0.2$	$1.2 \pm 0.1$	$t_{42} = -1.29, P = 0.206$
Proportion of nests	by substrate cl	ass				
Conifer	0.48	0.05	P = 0.003	0.00	0.35	P = 0.033
Deciduous	0.35	0.19	P = 0.246	0.20	0.29	P = 0.561
Hawthorn	0.04	0.52	<i>P</i> < 0.001	0.70	0.15	P = 0.001
Shrubs or vines	0.13	0.24	P = 0.359	0.10	0.21	P = 0.449
Proportion of nests	by substrate an	rangement clas	s			
Single	0.65	0.71	P = 0.530	0.60	0.71	P = 0.662
Line	0.09	0.00	P = 0.436	0.00	0.05	P = 0.174
Patch	0.13	0.10	P = 0.876	0.10	0.12	P = 0.716
Stand	0.13	0.19	P = 0.174	0.30	0.12	P = 0.589
Proportion of nests	by disturbance	e class				
Low	0.50	0.29	P = 0.234	0.00	0.71	<i>P</i> < 0.001
Medium	0.50	0.53	P = 0.873	0.74	0.29	P = 0.005
High	0.00	0.18	P = 0.161	0.26	0.00	<i>P</i> = 0.016

Table 11. Univariate comparisons (mean  $\pm$  SE) and proportions of habitat variables at all successful and depredated nests of Redwinged Blackbird, and at nests on golf courses and green spaces in Montreal from 2003-2005. Bolded *P*-values indicate t-, z-test or Mann-Whitney U comparisons that were significant at  $\alpha = 0.05$ .

	Golf course	Green space		Successful	Depredated	
Habitat variable	nests	nests	Test statistic	nests	nests	Test statistic
	( <i>n</i> = 96)	( <i>n</i> = 66)		( <i>n</i> = 78)	( <i>n</i> = 84)	
Nest height (m)	$0.7 \pm 0.1$	$0.8\pm0.1$	$t_{160} = -2.35, P = 0.020$	$0.7\pm0.1$	$0.8\pm0.1$	$t_{160} = -0.07, P = 0.944$
Concealment (%)	$57.0\pm2.2$	$51.4\pm2.3$	$t_{160} = 1.76, P = 0.080$	$55.1 \pm 2.4$	$54.4\pm2.2$	$t_{160} = 0.197, P = 0.843$
Edge (m)	$1.6 \pm 0.6$	$0.9 \pm 0.2$	$t_{160} = 2.66, P = 0.009$	$1.9\pm0.8$	$0.8 \pm 0.1$	$t_{160} = 1.92, P = 0.056$
Water depth (cm)	$14.3\pm2.0$	$17.7\pm2.8$	U = 3079.5, <i>P</i> = 0.763	$19.2\pm2.8$	$12.4\pm1.9$	U = 2677.0, <b>P</b> = <b>0.045</b>
Distance to	$25 \pm 02$	$68 \pm 09$	t. co1 07 P < 0 001	$4.0 \pm 0.5$	$4.4 \pm 0.7$	$t_{\rm rec} = 1.66 P = 0.907$
shore (m)	$2.3 \pm 0.2$	$0.0 \pm 0.7$	$t_{160} = -4.07, 1 < 0.001$	$+.0 \pm 0.3$	4.4 ± 0.7	$t_{160} = 1.00, T = 0.007$
Proportion of nests	by substrate cla	ass				
Reeds	0.84	0.71	P = 0.049	0.85	0.74	P = 0.086
Non-reeds	0.16	0.29	P = 0.049	0.15	0.26	P = 0.086
Proportion of nests	by substrate ar	rangement class	6			
Single or line	0.22	0.35	P = 0.070	0.10	0.27	P = 0.006
Patch	0.64	0.15	<i>P</i> < 0.001	0.60	0.44	P = 0.043
Stand	0.15	0.50	<i>P</i> < 0.001	0.29	0.29	P = 1.000
Proportion of nests	by disturbance	class				
Low	0.01	0.92	<i>P</i> < 0.001	0.31	0.45	P = 0.069
Medium	0.76	0.08	<i>P</i> < 0.001	0.63	0.35	<i>P</i> < 0.001
High	0.23	0.00	<i>P</i> < 0.001	0.06	0.20	P = 0.010

Table 12. Summary nesting statistics  $\pm$  SE for American Robin (AMRO), Gray Catbird (GRCA), Cedar Waxwing (CEDW), Northern Cardinal (NOCA) and Red-winged Blackbird (RWBL) nests on two green spaces (GS) and four golf courses (GC) in Montreal from 2003-2005. Averages are presented with associated Mann-Whitney U statistics. Significant *P*-values are bolded. Unless otherwise indicated in parentheses, sample sizes are presented in the # nests column.

	Site	# nests	Mean initiation date	Mean clutch size <sup>a</sup>	Mean young hatched <sup>b</sup>	Hatched (%) <sup>c</sup>	Mean young fledged <sup>d</sup>	Fledged (%) <sup>e</sup>
AMRO	GC	123	May 22 ± 1.9	$3.5(114) \pm 0.1$	$2.9(72) \pm 0.1$	$48.7\pm4.0$	$3.0(56) \pm 0.1$	$37.3\pm3.9$
	GS	43	May $31 \pm 2.9$	$3.4(35) \pm 0.1$	$2.9(29) \pm 0.2$	$57.2\pm6.9$	$2.9(17) \pm 0.2$	$33.7\pm6.7$
	M-W U		U = 1927.0	U = 1839.0	U = 1018.0	U = 2340.0	U = 452.0	U = 2523.0
			P = 0.008	P = 0.432	P = 0.845	P = 0.262	P = 0.754	P = 0.666
GRCA	GC	16	June 11 ± 4.7	$2.8(16) \pm 0.2$	$2.4(12) \pm 0.2$	$68.2\pm9.4$	$2.2(13) \pm 0.2$	$75.0\pm9.9$
	GS	36	June $12 \pm 3.2$	$3.5(35) \pm 0.1$	$3.2(29) \pm 0.2$	$83.3\pm5.4$	$3.1(29) \pm 0.2$	$77.8\pm6.8$
	M-W U		U = 54.0	U = 16.0	U = 85.5	U = 206.5	U = 81.0	U = 261.0
			P = 0.893	P = 0.018	P = 0.011	P = 0.106	P = 0.003	P = 0.592
CEDW	GC	23	July $1 \pm 1.7$	$4.2(12) \pm 0.2$	4.0 (1)	$4.3\pm4.3$	4.0 (1)	$4.3\pm4.3$
	GS	21	June $25 \pm 2.3$	$4.2(18) \pm 0.2$	$4.2(12) \pm 0.2$	$55.0\pm10.7$	$3.7(9) \pm 0.4$	$38.1 \pm 10.4$
	M-W U		U = 214.0	U = 101.0		U = 1150.0		U = 149.0
			P = 0.807	P = 0.767	_	P = 0.003	-	P = 0.031
NOCA	GC	28	May $29 \pm 4.8$	$3.0(26) \pm 0.2$	$2.4(18) \pm 0.3$	$53.1\pm8.3$	$2.5(12) \pm 0.3$	$40.4\pm9.3$
	GS	16	May $31 \pm 6.5$	$3.0(15) \pm 0.3$	$2.9(10) \pm 0.3$	$57.6 \pm 11.8$	$2.4(9) \pm 0.3$	$48.3 \pm 11.7$
	M-W U		U = 214.0	U = 188.5	U = 66.5	U = 201.5	U = 52.5	U = 198.5
			P = 0.807	P = 0.860	P = 0.259	P = 0.716	P = 0.915	P = 0.660

RWBL	GC	157	May 31 ± 1.0	$3.6(137) \pm 0.1$	$3.3(72) \pm 0.1$	$54.9\pm3.7$	$3.1(72) \pm 0.1$	$38.6\pm3.5$
	GS	102	May $28 \pm 1.2$	$3.6(98) \pm 0.1$	$3.2(37) \pm 0.2$	$51.5\pm4.6$	$3.0(37) \pm 0.2$	$29.5\pm4.1$
	M-W U		U = 7153.5 P = 0.116	U = 6472.5 P = 0.639	U = 1272.0 P = 0.701	U = 7764.0 P = 0.588	U = 1272.5 P = 0.703	U = 7125.0 P = 0.134

<sup>a</sup> Average clutch size calculated using nests with complete clutches (i.e. initiated incubation).

<sup>b</sup> Average young hatched calculated using nests that survived the incubation period (i.e. were not depredated).

<sup>c</sup> Percent hatched calculated by dividing number of nestlings with number of eggs laid for each nest.

<sup>d</sup> Average young fledged calculated using nests that survived the nestling period (i.e. were not depredated).

<sup>e</sup> Percent fledged calculated by dividing number of fledglings with number of eggs for each nest.
Table 13. Model support for hypotheses that temporal and nest-scale variables influence the daily survival rates of American Robin and Red-winged Blackbird nests on Montreal golf courses and other suburban green spaces in 2003-2005. Akaike's Information Criterion corrected for small sample size (AIC<sub>c</sub>),  $\Delta i$  (AIC<sub>c</sub> model *i* - AIC<sub>c</sub> minimum), Akaike weights (*w<sub>i</sub>*), the number of model parameters (K), and model deviance are shown for all models. Results using the LOCATION variable for site are shown for Red-winged Blackbird, as the two analyses produced almost identical results.

Model	AICc	$\Delta i$	Wi	K	Deviance
American Robin					
$\{\text{temporal}^{a} + \text{patch}\}$	485.23	0.00	0.66	14	457.03
$\{\text{temporal} + \text{site}^{b} + \text{patch}^{c}\}$	486.95	1.72	0.28	15	456.73
$\{\text{temporal} + \text{patch} + \text{nest-site}^{d}\}$	491.74	6.52	0.03	19	453.39
{global}	493.19	7.97	0.01	20	452.80
{patch + nest-site}	493.66	8.43	0.01	13	467.49
{patch}	493.94	8.71	0.01	8	477.87
{site + patch + nest-site}	495.11	9.88	0.00	14	466.91
{site + patch}	495.54	10.31	0.00	9	477.46
{temporal + nest-site}	499.78	14.55	0.00	12	475.63
{temporal + site + nest-site}	501.33	16.10	0.00	13	475.16
{temporal + site}	501.93	16.71	0.00	8	485.87
{temporal}	502.39	17.16	0.00	7	488.34
{nest-site}	512.21	26.98	0.00	6	500.17
{site + nest-site}	514.11	28.88	0.00	7	500.06
{.}	516.00	30.77	0.00	1	514.00
{site}	516.39	31.16	0.00	2	512.38
Red-winged Blackbird					
{temporal <sup>e</sup> + patch + nest-site}	409.38	0.00	0.50	15	379.16
{global}	411.41	2.03	0.18	16	379.16

{temporal + nest-site}	411.56	2.18	0.17	12	387.42
$\{\text{temporal} + \text{site}^{f} + \text{nest-site}\}$	413.49	4.11	0.06	13	387.32
$\{patch^g + nest-site^h\}$	414.04	4.67	0.05	11	391.92
{nest-site}	415.96	6.59	0.02	8	399.90
{site + patch + nest-site}	415.99	6.61	0.02	12	391.85
{site + nest-site}	417.74	8.36	0.01	9	399.65
{temporal + patch}	469.15	59.78	0.00	8	453.09
$\{\text{temporal} + \text{site} + \text{patch}\}$	471.12	61.74	0.00	9	453.04
{patch}	479.33	69.95	0.00	4	471.31
{site + patch}	481.05	71.67	0.00	5	471.03
{temporal + site}	482.42	73.04	0.00	6	470.38
{temporal}	482.83	73.45	0.00	5	472.80
{site}	494.42	85.04	0.00	2	490.41
{.}	496.12	86.74	0.00	1	494.12

<sup>a</sup> temporal = age + age<sup>2</sup> + season + season<sup>2</sup> + year

<sup>b</sup> site = location + house

<sup>c</sup> patch = substrate + arrangement

<sup>d</sup> nest-site = edge + cover + disturbance + nest height

e temporal = age + age<sup>2</sup> + year

<sup>f</sup> site = location or house

<sup>g</sup> patch = substrate + arrangement

<sup>h</sup> nest-site = edge + cover + disturbance + nest height + distance to shore + water depth

Table 14. Weighted beta estimates ( $\overline{\beta}$ ) and unconditional 95% CIs for all modelaveraged parameters within the top model influencing nest survival of American Robin nesting on Montreal golf courses and green spaces in 2003-2005. Beta estimates ( $\beta$ ) and 95% CIs are presented for Red-winged Blackbird since modelaveraging was not required. Influential variables (i.e. 95% CI of  $\overline{\beta}$  or  $\beta$  does not cross zero) are bolded. † denotes reference category for categorical variables.

Variable	$\overline{\beta}$ or $\hat{\beta}$	95% CI
American Robin	•	
Age	-0.047	-0.123, 0.029
$Age^2$	0.003	-0.001, 0.008
Season	0.019	-0.006, 0.044
Season <sup>2</sup>	0.000	0.000, 0.000
Year		
2003	0.244	-0.222, 0.711
2004	0.288	-0.129, 0.705
2005	Ť	Ť
Substrate		
Shrub or vine	0.169	-0.279, 0.618
Conifer	0.132	-0.295, 0.560
Deciduous	-0.641	-1.459, 0.177
Artificial	1.359	-0.215, 2.934
Hawthorn	Ť	Ť
Arrangement		
Single	-0.297	-0.754, 0.161
Line	1.118	-0.184, 2.420
Patch	-0.175	-0.587, 0.237
Stand	Ť	Ť
Red-winged Blackbird		
Age	-0.236	-0.391, -0.082
$Age^{2}$	0.011	0.004, 0.018
Year		
2003	-1.077	-2.339, 0.184
2004	-0.203	-0.709, 0.303
2005	Ť	Ť
Substrate		
Non-reeds	-0.422	-1.146, 0.302
Reeds	Ť	Ť
Arrangement		
Single or line	-0.756	-1.513, 0.001
Patch	-0.218	-0.902, 0.466
Stand	Ť	Ť
Disturbance		

Low	0.201	-0.619, 1.021
Medium	-0.295	-0.220, 1.242
High	Ť	÷
Nest height	-0.295	-3.343, 2.754
Cover	1.457	0.141, 2.774
Edge	-1.150	-2.896, 0.595
Distance to shore	2.637	-0.425, 5.698
Water depth	-1.355	-3.530, 0.820

Table 15. Level-specific nest success rates with associated 95% CIs for all potentially influential categorical variables for American Robin nesting on Montreal golf courses and green spaces in 2003-2005. Nest success was calculated by raising DSR estimates to the power of the American Robin nesting period (30 days).

Variable	Nest success (%)	95% CI (%)
Substrate		
Shrub or vine	36.6	17.2, 56.6
Conifer	35.4	23.4, 47.7
Deciduous	8.4	2.3, 19.7
Artificial	73.8	39.2, 90.7
Hawthorn	28.2	11.9, 47.4
Arrangement		
Single	20.2	11.5, 30.7
Line	71.3	47.2, 85.9
Patch	17.0	5.6, 33.9
Stand	37.7	22.4, 53.0



Fig. 4. Estimated daily survival estimates from the most supported model for Red-winged Blackbird nest success over the 87-day nesting period, with average initiation, hatch and fledge dates.

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		Collinearity statistic		
	Variable	Tolerance	VIF	
n	edge	0.93	1.08	
ric <i>ɛ</i> bin	nest height	0.56	1.80	
Rol	disturbance	0.89	1.12	
A	cover	0.89	1.13	
_	nest height	0.79	1.26	
ged rd	water	0.81	1.23	
/ing kbi	edge	0.85	1.18	
d-w lacl	distance to shore	0.75	1.33	
B	disturbance	0.72	1.38	
	cover	0.91	1.10	
S	nest height	0.96	1.04	
scie ed	edge	0.98	1.02	
spe	cover	0.95	1.05	
JII ⊅q	house	0.99	1.01	
1	tree	0.94	1.07	

Appendix 1. Multicollinearity statistics for nest- and site-scale habitat variables for American Robin, Red-winged Blackbird and all species pooled. All variables had tolerance scores > 0.7 and variance inflation factors (VIF) < 1.5.

Appendix 2. Bird species observed on Audubon-certified golf courses (BG and RM; dark gray), non-certified golf courses (IP and EG; dark gray) and reference sites (SW and TC; light gray) during all field seasons (2003-2005). All scientific names based on the A.O.U. Check-list of North American Birds, 7<sup>th</sup> Edition (2004). All species have been assigned a breeding code adapted from Gauthier and Aubry (1996). This system infers a hierarchy based on breeding-activity evidence and is listed in order (when observed over multiple years, the species was assigned the highest ranking breeding code).

Non-breeding species:	N – excavation of hole by woodpecker
X - observed in breeding season without evidence of	Confirmed breeding:
breeding	CN - nest-building
Fly - species seen flying over the site	NU - used nest or egg shell found
Possible breeding:	AT - adult carrying food for young
H - observed in suitable nesting habitat	PH - physiological evidence of breeding (e.g. brood
Probable breeding:	patch)
P - pair observed in suitable nesting habitat	DD - distraction display
T - territoriality (seen/heard at least 2 days, a week or	NO - adults around nest indicating that it is occupied
more apart)	FE - adults carrying faecal sac
C - courtship display	JE - recently fledged young
V - visiting probable nest site	NJ - nest containing eggs or young
A - agitated adult behaviour	

Common names	Scientific names			Si	tes		
		BG	RM	IP	EG	SW	TC
Common Loon	Gavia immer	Fly	Fly			Fly	Fly
Pied-billed Grebe	Podilymbus podiceps		Т			NJ	
Double-crested Cormorant	Phalacrocorax auritus	Fly	Fly	Fly	Fly		Fly
American Bittern	Botaurus lentiginosus		Х	Х		Х	
Great Blue Heron	Ardea herodias	Fly	Х	Х	Х	Х	Fly
Green Heron	Butorides virescens	-	Т	Х	Х	Т	X
Black-crowned Night Heron	Nycticorax nycticorax		Fly				
Turkey Vulture	Cathartes aura		X	Fly	Х	Fly	
Snow Goose	Chen caerulescens			Fly			
Canada Goose	Branta canadensis	Х	Х	X	Х	NJ	
Wood Duck	Aix sponsa		JE	JE	Р	JE	
Gadwall	Anas strepera		Х	Р			
American Black Duck	Anas rubripes		JE				
Mallard	Anas platyrhynchos	NJ	NJ	NJ	Р	NJ	Fly
Northern Pintail	Anas acuta					Х	
Blue-winged Teal	Anas discors		Р	Р			
Ring-necked Duck	Aythya collaris		Х				
Hooded Merganser	Lophodytes cucullatus		JE				
Red-breasted Merganser	Mergus serrator			Fly			
Osprey	Pandion haliaetus				Fly		Fly
Bald Eagle	Haliaeetus leucocephalus	Fly					-
Northern Harrier	Circus cyaneus	-			Х	Х	
Sharp-shinned Hawk	Accipiter striatus	X	Η	Х	JE	Η	NJ
Cooper's Hawk	Accipiter cooperii		NO	Η	Η	Т	Η
Red-shouldered Hawk	Buteo lineatus		Р		Т	Р	

	Broad-winged Hawk	Buteo platypterus	Х	Х		Х		Х
	Rough-legged Hawk	Buteo lagopus			Х			
	Red-tailed Hawk	Buteo jamaicensis		Х	Х	Fly	Fly	
	Golden Eagle	Aquila chrysaetos		Fly		•	•	
	American Kestrel	Falco sparverius	Х	X				
	Merlin	Falco columbarius	Х				Х	
	Virginia Rail	Rallus limicola					Х	
	Common Moorhen	Gallinula chloropus		Х				
	Sandhill Crane	Grus canadensis		Fly				
	Ruffed Grouse	Bonasa umbellus		Ċ				
	Sora	Porzana carolina					Х	
	Killdeer	Charadrius vociferous	Х	JE	NJ	Х	Х	Х
	Solitary Sandpiper	Tringa solitaria	Х	Х	Х	Х	Р	
	Spotted Sandpiper	Actitis macularia	Х	JE		Х	Х	
	Least Sandpiper	Calidris minutilla			Х			
	American Woodcock	Scolopax minor					Х	
	Bonaparte's Gull	Larus philadelphia		Х				
	Ring-billed Gull	Larus delawarensis	Х	Х	Х	Х	Fly	Х
	Common Tern	Sterna hirundo		Х				
	Black Tern	Chlidonias niger		Х				
	Rock Pigeon	Columbia livia	NJ	Х	Х		Fly	Fly
	Mourning Dove	Zenaida macroura	Р	NJ	NJ	NJ	NJ	NJ
	Black-billed Cuckoo	Coccyzus erythropthalmus		Х	Х		Х	Х
	Yellow-billed Cuckoo	Coccyzus americanus					Х	
	Eastern Screech-Owl	Megascops asio	JE					
	Great Horned Owl	Bubo virginianus				NJ	Х	
	Long-eared Owl	Asio otus		Х				
10	Northern Saw-whet Owl	Aegolius acadicus						Х
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Chimney Swift	Chaetura pelagica	Fly	Fly			Fly	Fly
Ruby-throated Hummingbird	Archilochus colubris		Ĥ	Н	Н	P	2
Belted Kingfisher	Ceryle alcyon	Х	Η	Т	Н	Н	
Yellow-bellied Sapsucker	Sphyrapicus varius	AT	Н	Т	JE	Т	Х
Downy Woodpecker	Picoides pubescens	Т	JE	JE	Р	Н	Η
Hairy Woodpecker	Picoides villosus	Η	Η	Н	Η	Н	Η
Northern Flicker	Colaptes auratus	Η	NO	С	Η	Н	С
Pileated Woodpecker	Contopus virens	Т	Т	Т	Т	Т	Т
Eastern Wood-Pewee	Dryocopus pileatus	Х	Т	Т	Т	Н	Η
Yellow-bellied Flycatcher	Empidonax flaviventris				Х	Х	
Alder Flycatcher	Empidonax alnorum	Х	Х		Х	Х	Х
Willow Flycatcher	Empidonax traillii						Х
Least Flycatcher	Empidonax minimus	Х	Х	Х	Х	Х	Х
Eastern Phoebe	Sayornis phoebe	Х	Η	Х	NJ	NJ	Х
Great Crested Flycatcher	Myiarchus crinitus	Т	Т	Т	Т	С	Η
Eastern Kingbird	Tyrannus tyrannus	Η	NJ	NJ	NJ	JE	Х
Blue-headed Vireo	Vireo solitaries		Х	Х	Х	Х	
Warbling Vireo	Vireo gilvus	Т	Т	А	Т	Н	Н
Philadelphia Vireo	Vireo philadelphicus	Х					Х
Red-eyed Vireo	Vireo olivaceus	А	NU	NU	Т	NU	NJ
Blue Jay	Cyanocitta cristata	Η	Η	NJ	Η	NJ	NJ
American Crow	Corvus brachyrhynchos	JE	JE	NO	JE	Η	Η
Common Raven	Corvus corax				Fly		
Purple Martin	Progne subis		NO	Х	Х		
Tree Swallow	Tachycineta bicolor	NJ	NJ	NO	NO	NJ	Х
Northern Rough-winged Swallow	Stelgidopteryx serripennis	Х	Х			Х	
Bank Swallow	Riparia riparia		Х				
Cliff Swallow	Petrochelidon pyrrhonota		Х				

	Barn Swallow	Hirundo rustica	NJ	NJ	NO	NJ	Х	Х
	Black-capped Chickadee	Poecile atricapilla	NO	NJ	NO	NO	Н	JE
	Tufted Titmouse	Baeolophus bicolor	X					
	Red-breasted Nuthatch	Sitta canadensis		Т	Н	Т	Н	
	White-breasted Nuthatch	Sitta carolinensis	Т	Н	JE	CN	Н	Η
	Brown Creeper	Certhia americana	Х		Х		Н	
	House Wren	Troglodytes aedon	Н	NJ	NO	CN	NU	Η
	Golden-crowned Kinglet	Regulus satrapa	X			Х	Х	
	Ruby-crowned Kinglet	Regulus calendula	Х	Х	Х	Х	Х	Х
	Eastern Bluebird	Sialia sialis		NJ		Х		
	Veery	Catharus fuscesce		Х	Т	Т	Т	Т
	Swainson's Thrush	Catharus ustulatus						Х
	Hermit Thrush	Catharus guttatus	X	Х	Х			Х
	Wood Thrush	Hylocichla mustelina		Х	Т	Х	Х	Х
	American Robin	Turdus migratorius	NJ	NJ	NJ	NJ	NJ	NJ
	Gray Catbird	Dumetella carolinensis	NJ	NJ	NJ	А	NJ	NJ
	Brown Thrasher	Toxostoma rufum		Х	NJ	Х	NJ	Т
	European Starling	Sturnus vulgaris	NJ	NO	JE	NO	Н	JE
	Bohemian Waxwing	Bombycilla garrulus			Х			
	Cedar Waxwing	Bombycilla cedrorum	NJ	NJ	NJ	NJ	NJ	NJ
	Tennessee Warbler	Vermivora peregrina		Х		Х	Х	
	Nashville Warbler	Vermivora ruficapilla	Χ	Х		Х	Х	Х
	Northern Parula	Parula americana	Х	Х	Х	Х		Х
	Yellow Warbler	Dendroica petechia	Х	NJ	NJ	NJ	NJ	NJ
	Chestnut-sided Warbler	Dendroica pensylvanica		Х	Х	Х	Х	Х
	Magnolia Warbler	Dendroica magnolia	Χ	Х	Х	Х	Х	Х
	Cape May Warbler	Dendroica tigrina			Х			
10	Black-throated Blue Warbler	Dendroica caerulescens	X	Х	Х	Х	Х	Х
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	Yellow-rumped Warbler	Dendroica coronata	X	X	X	X	X	X
	Black-throated Green Warbler	Dendroica virens	X	X	X	X	X	X
	Blackburnian Warbler	Dendroica fusca				X		
	Pine Warbler	Dendroica pinus				X		
	Palm Warbler	Dendroica palmaru		Х		Х	Х	Х
	Bay-breasted Warbler	Dendroica castanea	Х			Х		Х
	Blackpoll Warbler	Dendroica striata	Х	Х	Х	Х	Х	Х
	Black-and-white Warbler	Mniotilta varia	Х	Х	Х	Х	Х	Х
	American Redstart	Setophaga ruticilla	Х	Х		Х	Т	Х
	Ovenbird	Seiurus aurocapilla	Х	Т	Т	Т	Т	Т
	Northern Waterthrush	Seiurus noveboracensis			Х	Х	Х	Х
	Common Yellowthroat	Geothlypis trichas	Х	AT	Η	Т	А	Т
	Wilson's Warbler	Wilsonia pusilla					Х	Х
	Canada Warbler	Wilsonia canadensis	Х					Х
	Scarlet Tanager	Piranga olivacea		Х	Х		Х	
	Eastern Towhee	Pipilo erythrophthalmus		Х				
	American Tree Sparrow	Spizella arborea	Х	Х	Х	Х	Х	
	Chipping Sparrow	Spizella passerina	NJ	NJ	NJ	NJ	Х	
	Field Sparrow	Spizella pusilla		Х				
	Savannah Sparrow	Passerculus sandwichensis					Х	
	Song Sparrow	Melospiza melodia	NJ	NJ	NJ	NJ	NJ	NJ
	Swamp Sparrow	Melospiza georgiana					NU	
	White-throated Sparrow	Zonotrichia albicollis	Х	Т	Т	Т	Х	Т
	White-crowned Sparrow	Zonotrichia leucophrys	Х	Х	Х	Х	Х	Х
	Dark-eyed Junco	Junco hyemalis	Х	Х	Х	Х	Х	
	Northern Cardinal	Cardinalis cardinalis	NJ	NJ	NJ	NJ	NJ	NJ
	Rose-breasted Grosbeak	Pheucticus ludovicianus	Х	Η	JE	Т	NJ	Η
10	Indigo Bunting	Passerina cyanea		Х		Т	Х	JE
57								

Bobolink	Dolichonyx oryzivorus	Х	Х	Х		Η	Х
Red-winged Blackbird	Agelaius phoeniceus	NJ	NJ	NJ	Х	NJ	NJ
Rusty Blackbird	Euphagus carolinus		Х	Х		Х	
Common Grackle	Quiscalus quiscula	NJ	JE	NJ	Η	NJ	JE
Brown-headed Cowbird <sup>a</sup>	Molothrus ater	NJ	NJ	NJ	NJ	NJ	NJ
Baltimore Oriole	Icterus galbula	Χ	NO	NO	JE	NJ	JE
Purple Finch	Carpodacus purpureus	Х	Х		Х		
House Finch	Carpodacus mexicanus	Х				Х	Х
American Goldfinch	Pinus tristis	Η	Η	NJ	CN	NJ	Η
House Sparrow	Passer domesticus	NJ	NO	NO	NO	NJ	Η
Total number of confirmed breeding species		19	33	30	23	25	17
Total number of species		76	106	84	88	96	73

<sup>a</sup> Brown-headed Cowbird is a brood parasite. Both eggs and young were found in nests of other species on all sites.

Appendix 3. Photographs taken with infrared nest-cameras in 2006. Figs. 1 and 2 are of actual predators (removed eggs from nests); Figs. 3-8 show potential predators (no eggs available for removal.



Fig. 1. Red Squirrel eating Northern Cardinal egg.



Fig. 3. Eastern Chipmunk.



Fig. 5. Baltimore Oriole.



Fig. 7. Peromyscus sp.



Fig. 2. American Crow swallowing American Robin egg.



Fig. 4. Brown-headed Cowbird.



Fig. 6. Blue Jay.



Fig. 8. Human maintenance worker.

## CONNECTING STATEMENT 4

Many factors can influence whether a nesting attempt will be wholly successful, somewhat successful (e.g. suffer from partial egg or brood losses) or a complete failure. In the previous chapter, I compared nest success rates of birds breeding on golf courses and on green spaces, and identified potential factors that may influence nest survival. In this chapter, I focus on the effects that historical pesticide use may have on current avian nest success rates on these sites, in an attempt to rule out this factor as a potential cause of avian reproductive failure.

# CHAPTER 5

ORGANOCHLORINE AND POLYCHLORINATED BIPHENYL CONTAMINATION OF PASSERINE EGGS COLLECTED ON SUBURBAN GOLF COURSES AND GREEN SPACES IN MONTREAL, QUEBEC

Hudson, M.-A. R., K. J. Fernie, D. Burniston, C. Marvin and D. M. Bird. Organochlorine and polychlorinated biphenyl contamination of passerine eggs collected on suburban golf courses and green spaces in Montreal, Quebec. Will be submitted to Environmental Pollution.

#### Abstract

Birds nesting in suburban/urban areas are affected by a myriad of factors that may influence their ability to successfully breed and raise young. Historically, organochlorine (OC) pesticides were used extensively to control insect pests on golf courses and polychlorinated biphenyl (PCB) compounds were used in a wide spectrum of commercial, industrial and household applications. Despite this former ubiquity and their persistent nature, very few studies have quantified the level of contamination of organochlorines and other persistent compounds in birds using golf courses and other suburban green spaces to breed. We opportunistically collected non-viable and addled eggs as part of a larger study examining the factors affecting birds breeding on golf courses and on other suburban green spaces, and measured the levels of OCs and PCBs in a subset of these eggs from five species (Barn Swallow Hirundo rustica, American Robin Turdus migratorius, Gray Catbird Dumetella carolinensis, Northern Cardinal Cardinalis cardinalis and Red-winged Blackbird Agelaius phoeniceus Thirty-two of the 191 compounds measured were detected in all egg samples; most compounds were found at trace levels or were below detection limits. The concentrations and relative proportions of each pesticide varied between and within species, but overall concentrations peaked with several PCB congeners and 1,1-dichloro-2,2bis(p-chlorophenyl)ethylene (p,p'-DDE) in all species. Though  $\Sigma$  PCBs were higher in Red-winged Blackbird eggs collected on golf courses, reproductive parameters did not differ between golf courses and green spaces, suggesting that chemical contamination is not the main driver affecting nest success on these sites. This study is the first to quantify OCs and PCBs in bird eggs found on golf courses; additional in-depth studies are required to determine whether our results are typical of other areas.

The negative effects of organochlorine and PCB contamination on avian reproduction are well established, especially for raptorial and fish-eating species (e.g. Ratcliffe 1967, Hickey and Anderson 1968, Heath et al. 1972, Keith and Gruchy 1972). Though environmental concentrations have been declining in North America since their ban in the early 1970s (Sun et al. 2006), organochlorines are still found in the atmosphere (Aulagnier and Poissant 2005), in sediments (Mast et al. 2007) and in wildlife (Bishop et al.1995, Harper et al. 1996, Bishop et al. 1999, Harris et al. 2000, Gill et al. 2003), for several possible reasons proposed by Elliott et al. (1994). First, organochlorines are still being used in several parts of the world against, for example, malaria-carrying mosquitoes (Shaw and Chadwick 1998). Thus, birds may be directly exposed to these chemicals on their wintering grounds (but see Mora 1997, Capparella et al. 2003), or compounds may be taken into the atmosphere, circulated around the globe, and deposited on the breeding grounds where soils are contaminated through precipitation (Aulagnier and Poissant 2005). Second, since many organochlorines and their metabolites (e.g. 1,1-dichloro-2,2-bis(pchlorophenyl)ethylene, or p, p'-DDE) are long-lived, lipid-soluble compounds (Blus 2003), they may be picked up by foraging birds on their breeding grounds after previously contaminated soil is exposed through landscaping and construction, and/or through atmospheric deposition. Third, the legal use of dicofol and other similar compounds, which contain DDT (1,1,1-trichloro-2,2bis(*p*-chlorophenyl)ethane) isomers, may be contributing to current organochlorine contamination (Qiu et al. 2005).

These hypotheses, and the possible effects of these contaminants on wildlife, have been examined over the past two decades. Many studies examining the concentrations and effects of organochlorines on birds have used cavity-nesting insectivores due to the ease of using nest-boxes to attract breeders and because of their dietary and migratory habits which apparently increase their susceptibility to these compounds (e.g. Tree Swallows *Tachycineta bicolor*, and Great Tit *Parus major*; DeWeese et al. 1985, Bishop et al. 1995, Bishop et al. 2000, Bouvier et al. 2005). Open-cup nesting birds in orchards and agricultural

regions have also been extensively investigated due to the pre-ban high use of organochlorine pesticides, and because organophosphates are still being used today (Elliott et al. 1994, Fluetsch and Sparling 1994, Gill et al. 2003, Bouvier et al. 2005, Mineau and Whiteside 2006). Resident birds in Latin America and the southern United States have been analyzed to address the possibility that continued use of organochlorine compounds in Latin America might be the source of current loads (Mora 1997). This seems unlikely since studies indicate that historical contamination of the breeding grounds is the source of current contaminant exposure (Bishop et al. 1995, Harper et al. 1996, Bishop et al. 1999, Harris et al. 2000, Bartuszevige et al. 2002). Overall, there is a paucity of data concerning organochlorine pesticide loads in suburban/urban breeding birds (but see Okoniewski and Novesky 1993), where large amounts of construction and landscaping may make previously used and buried compounds once again available to wildlife.

Suburban areas are often home to golf courses, where pesticide use was, until recently with by-law changes, substantially higher than in agricultural areas (Knopper and Lean 2004). For example, pesticide application rates, including herbicides, insecticides and fungicides, on a coastal golf course in Maryland in the early 1990s were over 20 times higher than on a nearby agricultural field planted with corn or soy (golf course = 135.3 kg/ha/year vs. cropland = 6.5 kg/ha/year, derived from Schueler 1994). Indeed, there have been several reports of dead or dying birds found on golf courses after application of organophosphates (Zinkl et al. 1978, Stone and Koch 1982, Kendall et al. 1992). Aside from these reports and a handful of studies on the contaminant loads of watersheds (Metcalfe et al. 2008), groundwater (Wong et al. 1998, Cohen et al. 1999), Meadow Voles (Microtus pennsylvanicus, Knopper and Mineau 2004) and insects (Okoniewski and Novesky 1993) on or near golf courses, and measures of avian response to organophosphate spraying on golf courses (e.g. Brewer et al. 1988, Frank et al. 1991, Kendall et al. 1993, Rainwater et al. 1995), very little work has been published on the impact of golf course chemicals on wildlife and the environment. Results concerning the possibility of avian exposure to organophosphates are

mixed, but overall it appears that there is a risk to birds using golf courses (Rainwater et al. 1995). However, evidence of acute effects is restricted to waterfowl (Kendall et al. 1992, Kendall et al. 1993).

To our knowledge, there is no information published regarding organochlorine and PCB exposure of birds using golf courses. It is now well established that birds, like many other organisms, use golf courses for breeding and feeding (Moul and Elliott 1992, Terman 1997, Cristol and Rodewald 2005, LeClerc et al. 2005, LeClerc and Cristol 2005, Porter et al. 2005, Stanback and Seifert 2005; Chapters 2 and 4). Through landscaping and soil-turnover, historically used persistent organochlorine and PCB compounds may be resurfacing and contaminating the organisms inhabiting these sites. One of the major conclusions from a collection of research studies focusing on golf courses recently published in the Wildlife Society Bulletin was that the determination of pesticide contamination is essential in order to achieve a complete understanding of the factors affecting the breeding success of birds on golf courses (Cristol and Rodewald 2005, LeClerc et al. 2005, LeClerc and Cristol 2005, Porter et al. 2005, Rodewald et al. 2005, Stanback and Seifert 2005).

The collection and use of eggs is a minimally invasive and wellestablished method to detect contamination in birds since compounds like DDT are extremely lipophilic and are passed from female to egg (Bishop et al. 1999, Harris et al. 2000). Studies have also shown the link between concentrations in eggs and the environment (Bishop et al. 1995). Thus, the goal of this study was to quantify OC and PCB contamination of birds nesting on golf courses by screening their eggs, and comparing these values to those found in eggs collected from other types of suburban green spaces, whenever possible. We acknowledge that organophosphates, fungicides, herbicides, fertilizers and brominated flameretardants are currently used to some degree on some golf courses and may pose a chemical threat to bird life, but non-viable or addled eggs are not always reliable sources for these compounds. Also, as this study was part of a larger study examining passerine nest success on golf courses, testing for these chemicals, which often requires taking blood from nestlings or collecting fresh eggs, was not feasible, as it could have potentially biased the productivity metrics under study. However, as the number of new chemicals found to affect wildlife increase (e.g. Fernie et al. 2009, Sughrue et al. 2008), it is imperative that we quantify the contaminant loads in wildlife with a high probability of exposure (i.e. those inhabiting urbanized, agricultural and industrial areas).

#### MATERIALS AND METHODS

Data collection. -- This study was carried out on six sites in the Montreal area, Quebec, Canada from 2004-2005. (Beaconsfield Golf Club, Ile Perrot Golf and Country Club, Elm Ridge Country Club and Royal Montreal Golf Club) and two green spaces (Stoneycroft Wildlife Area and Terra Cotta Park) were intensively searched for nests from early April to late August each year. Nest searching and monitoring was conducted by four observers between 0500 and 1600 EST. Nests were located by systematically searching the area and cueing in on adult behaviour such as food or nesting-material transportation or vocalization (Martin and Geupel 1993). We monitored nests every 2-5 days until they either fledged at least one young or failed. Nest monitoring was done with a small colour camera attached to an extendible pole (Hudson and Bird 2006; Chapter 3). Focal species included Barn Swallow (Hirundo rustica), American Robin (Turdus migratorius), Gray Catbird (Dumetella carolinensis), Northern Cardinal (Cardinalis cardinalis) and Red-winged Blackbird (Agelaius phoeniceus). Unfertilized or unhatched eggs were opportunistically collected following fledging or failure of a nest, and were placed in clean bottles and frozen at -20 °C. All eggs were collected under both federal and provincial permits.

To prepare the contents for analysis, eggs were thawed and cut or cracked around the equator. The contents were placed in chemically cleaned jars for chemical analysis, and the shells were air-dried for two days. Shell thickness was measured at three points around the equator with a set of callipers (Mitotoyo, Japan; smallest unit of measurement: 0.0254 mm). These measures were then averaged to determine mean eggshell thickness.

Chemical analysis. -- Egg contents were prepared and homogenized by laboratories at the National Water Research Institute (Burlington, Ontario), and the chemical analysis was conducted by Axys Analytical Services Ltd. (Victoria, British Columbia). All samples were prepared by being weighed, filtered and liquid-liquid extracted. The samples were then spiked with isotopically labelled surrogate standards and then fractionated into two parts using a Florisil column. One fraction of each sample was analyzed for low and medium polarity organochlorine (OC) pesticides and polychlorinated biphenyls (PCBs) using highresolution gas chromatography with detection by low-resolution mass spectrometry (HRGC/LRMS). These include: 160 PCB congeners (CB-1, 2, 3, 4/10, 6, 7/9, 8/5, 11, 12/13, 14, 15, 16/32, 17, 18, 19, 22, 24/27, 25, 26, 28, 29, 30, 31, 33/20/21, 34/23, 35, 36, 37, 38, 39, 40, 41/71/64/68, 42/59, 44, 45, 46, 47/48/75, 49/43, 50, 51, 52/73, 53, 54, 55, 56/60, 57, 58, 63, 65/62, 66/80, 67, 69, 70/76, 72, 74/61, 77, 78, 79, 81, 82, 83/108, 84, 85/120, 87/115/116, 88/121, 90/101/89, 91, 92, 94, 95/93, 96, 97/86, 98/102, 99, 100, 103, 104, 105/127, 107/109, 110, 111/117, 112, 113, 114, 118/106, 119, 122, 123, 124, 125, 126, 128, 129, 130, 131/142, 132/168, 133, 134/143, 136, 137, 138/163/164, 140, 141, 144/135, 145, 146, 147, 148, 149/139, 150, 151, 152, 153, 154, 155, 156, 157, 158/160, 159, 161, 162, 165, 166, 167, 169, 170/190, 171, 172/192, 173, 174/181, 175, 176, 177, 178, 179, 180, 183, 184, 185, 186, 187/182, 188, 189, 191, 193, 194, 195, 196/203, 197, 198, 199, 200, 201, 202, 204, 205, 206, 207, 208, 209); 1,3-DiCB, 1,4-DiCB, 1,2-DiCB, 1,3,5-TricCB, 1,2,4-TriCB, 1,2,3-TriCB, 1,2,35/1,2,4,5-TetCB, 1,2,3,4-TetCB; hexachlorobutadiene;  $\beta$ -,  $\alpha$ -,  $\gamma$ - and  $\delta$ -HCH; heptachlor; aldrin; trans-, cis- and oxychlordane; octachlorostyrene; trans- and cis-nonachlor; DDT, DDE and DDD isomers; and mirex. The HRGC/LRMS was performed on a gas chromatograph equipped with a quadruple mass spectrometer and a J&W 60-m DB-5 chromatography column (0.25 mm ID, 0.10 µm film thickness). The second fraction of each sample was analyzed for high polarity OCs, specifically  $\delta$ -HCH; heptachlor epoxide;  $\alpha$ - and  $\beta$ -endosulphan; dieldrin; endrin; endrin aldehyde; endosulphan sulphate; endrin ketone and methoxychlor. This analysis involved using gas chromatography with electron capture detection

(GC/ECD). The GC/ECD was performed on a gas chromatograph, a <sup>63</sup>Nickel electron capture detector and an integrator. We again used a J&W 60-m DB-5 capillary column (0.25 mm ID, 0.10 µm film thickness), confirming with a simultaneous analysis on a J&W 30-m DB-17MS capillary column (0.25 mm ID, 0.25 µm film thickness) when required. Analytical run time and ramping of the temperatures throughout the analysis were conducted according to standardized methodologies of Axys Analytical Laboratories (T. Rawsthorne, pers. comm., Axys Analytical Ltd.). Detection limits ranged from 0.055 - 30.476 ng/g wet weight (an exceptionally high value for the p,p'-DDT concentration of one sample); most values fell between 0.1 and 0.5 ng/g wet weight. When we detected peaks of certain compounds which failed to meet quantification criteria, we estimated the maximum possible concentration for these compounds and reported these values as detection limits. When summing chemical classes or providing averages, we used half the value of each detection limit to avoid showing false non-detects or zeros. All chemical concentrations are expressed in ng/g wet weight.

Data analysis. -- We used Mann-Whitney U tests to look for differences between eggshell thicknesses on golf courses and green spaces. After verifying each species individually and log-transforming DDT concentrations to improve normality, we pooled all eggshell thickness measures, which were normally distributed, and regressed them against  $\Sigma$  DDT to look for evidence of eggshell thinning. Hatching success was calculated by dividing the number of young hatched by the number of eggs laid. Fledging success was calculated using two methods: first, by dividing the number of young fledged per nest by the number of young hatched (% fledged) and second, by dividing the number of young fledged per nest by the number of eggs laid (% total fledged). Differences in Red-winged Blackbird clutch size and the mean number of young hatched and fledged between golf courses and one of the green spaces were examined for statistically significant differences using Mann-Whitney U tests; Red-winged Blackbirds were only found in sufficient number on one of the green spaces examined. We used zscores to compare hatching and fledging success rates. Apparent nest success rates (successful nests/all nests) were calculated for both Red-winged Blackbird and American Robin. We used Mann-Whitney U tests to compare concentrations of the most commonly detected chemical compounds in Red-winged Blackbird eggs collected on golf courses and on the green space. Finally, we used Spearman Rank Order correlation to measure the relationship of these chemicals to Red-winged Blackbird nest success statistics, and to each other. We did not apply Bonferroni corrections to the correlation matrix, opting instead to carefully inspect the number and degree of significant *P*-values (Moran 2003). We used STATISTICA (StatsSoft, Inc., Tulsa, OK) for all analyses, with significance levels set at P < 0.05. Means are presented  $\pm$  standard error (SE) unless otherwise specified.

#### RESULTS

We found 126 non-viable eggs left in the nests of 14 open-cup nesting passerine species, which represented 4.9% of all eggs laid during the course of this study (n = 2598). Despite a much larger proportion of non-viable eggs on golf courses, the number of non-viable eggs observed per site type was similar (Table 1). This is due to the larger number of target species' nests found on golf courses, and not necessarily a reflection of site quality. A total of 57 eggs was collected from 11 of these species, though only five species' eggs were analyzed (n = 41): Barn Swallow (n = 4), American Robin (n = 12), Gray Catbird (n = 5), Northern Cardinal (n = 5) and Red-winged Blackbird, n = 15). As eggs from the same clutch were homogenized and considered as one sample, this reduced the samples to n = 1, n = 8, n = 4, n = 2 and n = 10, respectively, for a total of n = 25. Observed embryological development included non-fertile (yolk only), fertilized but undeveloped (yolk with dark spot), and moderately developed (embryo) (Caldwell and Snart 1974). The number of eggs collected per site was: Beaconsfield Golf Club, n = 3; Royal Montreal Golf Club, n = 2; Ile Perrot Golf and Country Club, n = 2; Elmridge Golf Club, n = 9; Stoneycroft Wildlife Area, n= 7; and Terra Cotta Park, n = 2.

*Eggshell thickness.* -- American Robin eggshell thicknesses averaged 0.118 mm  $\pm$  0.003. Red-winged Blackbird eggshell thicknesses averaged 0.102 mm  $\pm$  0.003, and did not differ between eggs collected on golf courses (n = 5, mean thickness = 0.107 mm  $\pm$  0.003) and the green space (n = 8, mean thickness = 0.098 mm  $\pm$  0.004; Mann Whitney U = 10.5, P = 0.164). The mean eggshell thickness measures of the other species were: Barn Swallow, 0.080 mm  $\pm$  0.007; Gray Catbird, 0.102 mm  $\pm$  0.002; and Northern Cardinal, 0.116 mm  $\pm$  0.005. The log of the sum of all DDT congeners was not related to eggshell thicknesses for any of the species individually (P value range = 0.073-0.835) or when pooled (Fig. 1).

Contaminant levels. -- Of the 191 chemical compounds assayed, 71 (10 OCs and 61 PCBs) were below detection limits in all samples, while 32 (three OCs:  $p_{,p}$ ) -DDE, *t*-nonachlor, endrin ketone; and 29 PCBs) were detected in all samples. Though concentrations and relative proportions of each pesticide varied substantially among and within species, p,p'-DDE and  $\Sigma$  PCBs were found at higher concentrations across all species (Tables 2 and 3). P,p'-DDE dominated the relative chemical burdens in all species but the Gray Catbird (Table 3). However, the mean concentration of p, p'-DDE for catbirds was one of the highest measured, second only to Northern Cardinal (Fig. 2). The relatively low p,p'-DDE burden for catbirds is due to high concentrations of oxychlordane and t-nonachlor in one of the catbird samples (4870.0 and 5760.0 ng/g wet weight, respectively), which then inflated the overall mean (Fig. 2). The sum of all PCB congeners represented the second largest contribution to overall burdens (Tables 2 and 3). PCB congeners 153 and 138/13/164 and 180 were found more consistently at higher levels than the other congeners, contributing between 5.8 - 23.7% of the PCB total (Fig. 3). The American Robin was the only species to show peaks for Mirex (Fig. 2). This peak, much like those for oxychlordane and *t*-nonachlor, is due to two contaminated samples, which inflated the average concentration for the species. Two robins had Mirex concentrations 2-3 orders of magnitude higher than the other samples (1476.6 and 825.0 ng/g wet weight vs. 1.0 - 70.3 ng/g wet weight).

We compared the chemical concentrations of the most commonly found compounds in individual Red-winged Blackbird eggs found on golf courses and on one of the green spaces (Fig 4). Individual variation was extensive in some cases, causing large standard errors (e.g. p,p'-DDE concentration ranged from 13.8 - 1015.5 ng/g wet weight for green space samples). Despite this intra-site variability, differences were found between sites; the  $\Sigma$  PCB concentration in eggs from golf courses was double that from eggs collected off green spaces (U = 1.0, P = 0.019), with ensuing differences in the major components of  $\Sigma$  PCB (Fig. 4).

The ratio of DDE to DDT concentrations can indicate how recently an organism was exposed to DDT. DDE:DDT varied between individuals and species: American Robin, range = 3.3 - 63.3, mean = 21.7; Red-winged Blackbird, range = 4.4 - 1188.6, mean = 274.1; Gray Catbird, range = 20.1 - 33.0, mean = 26.0; Northern Cardinal, range = 2.5 - 375.7, mean = 189.1; and Barn Swallow, DDE:DDT = 13.8. As lower ratios indicate more recent exposure and higher ratios point to a more historical exposure, the Red-winged Blackbirds studied here were likely exposed the earliest, followed by Northern Cardinal, Gray Catbird, American Robin and Barn Swallow. The average DDE:DDT for Red-winged Blackbirds eggs from golf courses was 351.02 (range = 51.3 - 1188.6), and was 222.88 (range = 27.7 - 620.8) for green spaces.

*Reproductive success.* -- We calculated various reproductive parameters for the most commonly collected species, American Robin and Red-winged Blackbird. American Robin clutch size, based on the nests containing the collected eggs, was  $3.8 \pm 0.2$ . The number of young hatched and fledged were  $1.1 \pm 0.4$  and  $0.9 \pm 0.4$ , respectively, leading to a hatch rate of  $38\% \pm 12\%$  and a total fledge rate of  $27\% \pm 11\%$  (80%  $\pm 20\%$  from successful nests).

We tested for differences in reproductive parameters between golf courses and the green space with the Red-winged Blackbird. There were no statistically significant differences between any of the measures from nests on golf courses and nests on the green space, though the percent fledged on golf courses was more than double that on the green space (Table 4). Apparent nest success was 75% on golf courses and 50% on the green space, which was not statistically different (P = 0.440). Average clutch size based on all nests was  $3.5 \pm 0.2$ . The number of young hatched and fledged were identical ( $1.3 \pm 0.4$ ), leading to a hatch rate of  $43\% \pm 13\%$  and a total fledge rate of  $44\% \pm 15\%$  ( $100\% \pm 0\%$  from successful nests). Red-winged Blackbird apparent nest success was 60%.

To determine if OCs or PCBs contamination of Red-winged Blackbird eggs influenced reproductive parameters, we used Spearman Rank Order Correlation to test the most commonly found compounds against site type (golf course vs. green space), clutch size, the number of young hatched and fledged, and the percent hatched and fledged. As many of the top compounds were PCBs, we first determined that they were all significantly correlated to  $\Sigma$  PCB, and then removed them from the analysis. Of the top five compounds,  $\Sigma$  PCBs was positively correlated to golf courses (P = 0.008), and to the number and percent of Red-winged Blackbird young hatched and fledged (P = 0.014-0.016; Table 5). Oxychlordane and *c*-nonachlor were also correlated with  $\Sigma$  PBCs (P = 0.029 and 0.033, respectively; Table 5).

#### DISCUSSION

*Eggshell thickness.* -- Bunck et al. (1985) found that eggshell thickness was not related to embryological development and could therefore act as a method to assess the effects of organochlorine contamination. This was later confirmed by Bennett (1995), though neither study examined passerines. Since DDT is known to affect raptor eggshell thickness, contributing to eggshell breakage, this measure is commonly available for raptorial species. However, few researchers have published information on passerine eggshell thickness. Of the species studied here, we were only able to find published values for Red-winged Blackbird and Gray Catbird. Spaw and Rohwer (1987) provided mean values based on six and five samples, respectively: 0.094 mm  $\pm$  0.002 for Red-winged Blackbird and 0.100 mm  $\pm$  0.006 for Gray Catbird. Our values are slightly thicker than those reported. It is likely that, due to their trophic position in the terrestrial food web, most passerines will not experience the level of eggshell thinning documented in

raptors or aquatic birds (Blus et al. 2003). Indeed, we found no relationship between eggshell thickness and  $\Sigma$  DDT.

Contaminant levels. -- We found several species-specific differences in chemical concentrations. Differences in contaminant loads may be due to differences in the birds' ability to metabolize compounds (Blus et al. 2003), or differences in exposure depending on diet, home range size and content (Klemens et al. 2000). Feeding habits are often identified as a major factor affecting a passerine's exposure to pesticides, with insectivores and omnivores usually accumulating more toxins than granivores (Klemens et al. 2000, Bartuszevige et al. 2002). Indeed, American Robin and Gray Catbird (both consume invertebrates and fruit) eggs had roughly twice and six times the total contamination than Red-winged Blackbirds (granivores outside the breeding season, insectivores during the breeding season; Yasukawa and Searcy 1995). Based on this prediction, we would also expect to see higher concentrations for Barn Swallow (aerial insectivores) eggs and lower values for Northern Cardinal (consume primarily seeds and fruits, with a smaller proportion of insects; Halkin and Linville 1999) eggs, but they actually had slightly lower (0.7 times) and higher (1.7 times) concentrations than the Red-winged Blackbird, respectively. This discrepancy, as well as individual variation, may be due to dietary shifts before and within the breeding season, population-specific differences in the proportion of insects vs. berries and seeds (and thus differences in lipid-rich prey items), variation in prey selection and local prey availability (Morrissey et al., *in press*), or small sample sizes.

Overall, the levels of organochlorine and PCB contamination found in our study were variable but substantially lower than those found in by Bishop et al. (1995, 2000), Elliott et al. (1994), Harris et al. (2000) and Gill et al. (2003), who focused their analyses on passerine eggs collected from fruit orchards and/or sites around the Great Lakes. Other studies using bird carcasses instead of eggs have generally found lower contaminant concentrations (e.g. Klemens et al. 2000, Bartuszevige et al. 2002), reflecting the chemical off-loading into eggs by laying birds. In general, contaminant loads seem low in Quebec compared to other

northeastern states and provinces, as shown in Great Blue Herons (*Ardea herodias*; Laporte 1982). This is potentially a reflection of the less intense land use (urban, agriculture) and industrial base of the province. However, given that many studies quantifying OC and PCB contamination were done in the mid-1990s to early 2000s, it may simply be that more time has elapsed, allowing for additional degradation and dispersal into the environment, reducing overall contaminant loads.

DDE to DDT ratios found in this study are higher than those found in American Robins nesting in British Columbia fruit orchards (Gill et al. 2003), indicating less recent exposure. After 15-20 years, DDE:DDT should be higher than 20:1 (Elliott et al. 1994). Since the ban on DDT was enacted in the early 1970s, this ratio should now be even higher. However, Barn Swallow showed a much lower ratio, and American Robin and Gray Catbird showed ratios close to 20, suggesting that some of these birds are still being exposed. We have no evidence to support the hypothesis that birds were subject to a more recent exposure to DDT on golf courses, since the ratios for golf course-nesting Redwinged Blackbirds were overall higher than those nesting on the green space. It appears that birds nesting in these suburban green spaces are all prone to being contaminated by historically used pesticides, regardless of whether they are nesting on golf courses or other less-managed green spaces.

*Reproductive success.* -- Though based on small samples, our results suggest that the chemical burdens found here do not significantly affect nest success endpoints. Others have found similar results in Red-winged Blackbirds and Tree Swallows (*Tachycineta bicolor*) in the St. Lawrence River and Great Lakes (Bishop et al. 1995, Bishop et al. 1999) and in American Robins nesting in Canadian orchards (Harris et al. 2000, Gill et al. 2003), despite finding significantly higher contaminant concentrations. Rather, it is likely that the high predation pressure found in this system (see Chapter 4) affects birds nesting on golf courses and on green spaces much more than historically used pesticide contamination.

However, we did find that  $\Sigma$  PCBs correlated positively with golf courses and with the number and percent of young Red-winged Blackbirds hatched and fledged. We believe the latter to be a spurious correlation, since we found, as part of a larger study, that Red-winged Blackbirds had significantly higher nest success rates on golf courses than on the green space we studied (see Chapter 4). Thus, it appears that Red-winged Blackbird eggs on golf courses simply carried higher PCB loads than those on green spaces; it is unlikely that PCB contamination was responsible for higher nest success given their known effects on avian reproduction (Rice et al. 2003). The fact that oxychlordane and cnonachlor concentrations were also higher on golf courses and significantly correlated with  $\Sigma$  PCBs suggests similar chemical origins or metabolic properties (Laporte 1982). Though the contamination rates found here did not negatively affect reproduction, is it important to consider why these chemicals were found at higher concentrations in blackbird eggs from golf courses than from the green space. If these chemicals were picked up on-site, then three different golf courses are implicated. Though traditionally not regarded as pesticides due to their numerous industrial applications, PCBs were formerly used as pesticide extenders (Nisbet and Sarofim 1972), possibly explaining their higher concentrations on golf courses. Extremely mobile in water and very persistent (Rice et al. 2003), these PCBs may have then been continuously re-circulated through the intensive irrigation typical of a golf course, and picked up by insectivorous Red-winged Blackbirds nesting and feeding in or near golf course ponds. We have no way of knowing definitively whether the PCBs found here are due to historical use, due to their overall ubiquity in the environment, or due to contamination at another site shared by these particular birds prior to laying. However, Morrissey et al. (in *press*), found strong correlations between the blood and egg contents of American (Cinclus mexicanu) and European (C. cinclus) Dippers. This suggests that passerines might be faithful indicators of on-site contamination, as chemicals are likely picked up right before and during egg laying.

### CONCLUSIONS

Our results, the first of their kind to the best of our knowledge, suggest that golf courses are of comparable "quality" to other suburban green spaces in terms of the role that historical contamination might play in passerine reproductive success. However, we stress that further testing, using the systematic collection of a large number of eggs or blood samples, is required. We were typically able to analyse one egg per nest, though some exceptions did occur. Reynolds et al. (2004) found large variations in DDE concentrations within European Starling (Sturnus vulgaris) and Prothonotary Warbler (Protonotaria *citrea*) clutches in Colorado and Alabama, respectively. Since passerines typically lay one egg every 24 hrs, the lipid content, and thus the contaminant concentration in each egg, is dependent on the female's diet prior to laying (Reynolds et al. 2004). This means that changes made in the female's foraging location may be reflected in each egg, producing different chemical burdens within each egg, and leading to possibly different conclusions depending on which egg is analyzed. Certain avian species feed extensively on golf course fairways and in the rough (I. Julian, unpublished data, Rainwater et al. 1995). To determine whether feeding locations are placing birds at risk to historical pesticide contamination, we suggest that individual feeding preferences are examined and related to possible differences in egg contaminant loads (e.g. Morrissey et al. 2004). We also suggest that future studies analyse sediment samples taken from the soil surrounding golf course irrigation ponds, and water samples from both golf courses and other suburban green spaces to directly quantify OC and PCB concentrations for a definitive answer. Contaminant levels in various arthropod species should be also assessed at these types of sites. Finally, nestlings hatched on golf courses should be monitored post-fledging to ascertain whether *in ovo* exposure to OCs and PCBs affects their fitness and ability to survive long-term. Clearly, there is still much to learn about the role pesticides play in these environments. We hope that this work will stimulate others to conduct more in-depth investigations of the levels of historically and currently used chemicals in wildlife inhabiting urban green spaces in general (e.g. Décarie et al. 1993), and golf courses in particular.

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#### LITERATURE CITED

- Aulagnier, F. and L. Poissant. 2005. Some pesticides occurrence in air and precipitation in Québec, Canada. Environmental Science and Technology 39: 2960-2967.
- Bartuszevige, A. M., A. P. Capparella, R. G. Harper, J. A. Frick, B. Criley, K. Doty, and E. Erhart. 2002. Organochlorine pesticide contamination in grassland-nesting passerines that breed in North America. Environmental Pollution 117: 225-232.
- Bennett, R.S. 1995. Relative sensitivity of several measures of eggshell quality to the stage of embryonic development. Bulletin of Environmental Contamination and Toxicology 54: 428-431.
- Bishop, C. A., B. Collins, P. Mineau, N. M. Burgess, W. F. Read, and C. Risley. 2000. Reproduction of cavity-nesting birds in pesticide-sprayed apple orchards in southern Ontario, Canada, 1988-1994. Environmental Toxicology and Chemistry 19: 588-599.
- Bishop, C. A., M. D. Koster, A. A. Chek, D. J. T. Hussell, and K. Jock. 1995.
  Chlorinated hydrocarbons and mercury sediments, Red-winged Blackbirds (*Agelaius phoeniceus*) and Tree Swallows (*Tachycineta bicolor*) from wetlands in the Great Lakes-St. Lawrence River basin. Environmental Toxicology and Chemistry 14: 491-501.
- Bishop, C. A., N. A. Mahony, S. Trudeau, and K. E. Petit. 1999. Reproductive success and biochemical effects in Tree Swallows (*Tachycineta bicolor*)

exposed to chlorinated hydrocarbon contaminants in wetlands of the Great Lakes and St. Lawrence River Basin, USA and Canada. Environmental Toxicology and Chemistry 18: 263-271.

- Blus, L. J. 2003. Organochlorine Pesticides. Pages 313-340 in Handbook of Ecotoxicology. (D. J. Hoffman, B. A. Rattner, G. A. Burton Jr., and J. Cairns Jr., Eds.). Lewis Publishers, CRC Press LLC, Boca Raton, FL.
- Bouvier, J.-C., J.-F. Toubon, T. Boivin, and B. Sauphanor. 2005. Effects of apple orchard management strategies on the Great Tit (*Parus major*) in southeastern France. Environmental Toxicology and Chemistry 24: 2846-2852.
- Brewer, L. W., C. J. Driver, R. J. Kendall, T. E. Lacher Jr., and J. C. Galindo. 1988. Avian response to a turf application of Triumph® 4E. Environmental Toxicology and Chemistry 7: 391-401.
- Bunck, C. M., J. W. Spann, O. H. Pattee, and W. J. Fleming. 1985. Changes in eggshell thickness during incubation: implications for evaluating the impact of organochlorine contaminants on productivity. Bulletin of Environmental Contamination and Toxicology 35: 173-182.
- Caldwell, P. J. and A. E. Snart. 1974. A photographic index for aging Mallard embryos. Journal of Wildlife Management 38: 298-301.
- Capparella, A. P., J. A. Klemens, R. G. Harper, and J. A. Frick. 2003. Lack of widespread organochlorine pesticide contamination in South American resident passerines. Bulletin of Environmental Contamination and Toxicology 70: 769-774.
- Cohen, S., A. Svrjcek, T. Durborow, and N. LaJan Barnes. 1999. Water quality impacts by golf courses. Journal of Environmental Quality 28: 798-809.
- Cristol, D. A. and A. D. Rodewald. 2005. Introduction: Can golf courses play a role in bird conservation? Wildlife Society Bulletin 33: 407-410.
- Décarie, R., J. L. DesGranges, C. Lépine, and F. Morneau. 1993. Impact of insecticides on the American Robin (*Turdus migratorius*) in a suburban environment. Environmental Pollution 80: 231-238.

- DeWeese, L. R., R. R. Cohen, and C. J. Stafford. 1985. Organochlorine residues and eggshell measurements for Tree Swallows *Tachycineta bicolor* in Colorado. Bulletin of Environmental Contamination and Toxicology 35: 767-775.
- Elliott, J. E., P. A. Martin, T. W. Arnold, and P. H. Sinclair. 1994. Organochlorines and reproductive success of birds in orchard and nonorchard areas of Central British Columbia, Canada, 1990-91. Archives of Environmental Contamination and Toxicology 26: 435-443.
- Fernie K.J., J. L. Shutt, R. J. Letcher, J. I. Ritchie, K. Sullivan, and D. M. Bird. 2009. Environmentally relevant concentrations of DE-71 and HBCD alter eggshell thickness and reproductive success of American kestrels. Environmental Science and Technology 43: 2124-2130.
- Fluetsch, K. M. and D. W. Sparling. 1994. Avian nesting success and diversity in conventionally and organically managed apple orchards. Environmental Toxicology and Chemistry 13: 1651-1659.
- Frank, R., P. Mineau, H. E. Braun, I. K. Barker, S. W. Kennedy, and S. Trudeau.1991. Deaths of Canada geese following spraying of turf with diazinon.Bulletin of Environmental Contamination and Toxicology 46: 852-858.
- Gill, H., L. K. Wilson, K. M. Cheng, and J. E. Elliot. 2003. An assessment of DDT and other chlorinated compounds and the reproductive success of American Robins (*Turdus migratorius*) breeding in fruit orchards. Ecotoxicology 12: 113-123.
- Halkin, S. L. and S. U. Linville. 1999. Northern Cardinal (*Cardinalis cardinalis*) *in* The Birds of North America Online. (A. Poole, Ed.). Cornell Lab of
  Ornithology, Ithaca, NY. [Online.] Available at
  bna.birds.cornell.edu/bna/species/440.
- Harper, R. G., J. A. Frick, A. P. Capparella, B. Borup, M. Nowak, D. Biesinger, and C. F. Thompson. 1996. Organochlorine pesticide contamination in Neotropical migrant passerines. Archives of Environmental Contamination and Toxicology 31: 386-390.

- Harris, M. L., L. K. Wilson, J. E. Elliot, C. A. Bishop, A. D. Tomlin, and K. V. Henning. 2000. Transfer of DDT and metabolites from fruit orchard soils to American Robins (*Turdus migratorius*) twenty years after agricultural use of DDT in Canada. Archives of Environmental Contamination and Toxicology 39: 205-220.
- Heath, R. G., J. W. Spann, J. F. Kreitzer, and C. Vance. 1972. Effects of polychlorinated biphenyls on birds. Proceedings of the XVth International Ornithological Congress, The Hague, The Netherlands. E. J. Brill, Leiden, Netherlands.
- Hickey, J. J. and D. W. Anderson. 1968. Chlorinated hydrocarbons and eggshell changes in raptorial and fish-eating birds. Science 162: 271-273.
- Hudson, M.-A.R. and D. M. Bird. 2006. An affordable computerized camera technique for monitoring bird nests. Wildlife Society Bulletin 34: 1455-1457.
- Keith, J. A. and I. M. Gruchy. 1972. Residue levels of chemical pollutants in North American birdlife. Proceedings of the XVth International Ornithological Congress, The Hague, The Netherlands. E. J. Brill, Leiden, Netherlands.
- Kendall, R. J., L. W. Brewer, and R. R. Hitchcock. 1993. Response of Canada Geese to a turf application of diazinon AG500. Journal of Wildlife Diseases 29: 458-464.
- Kendall, R. J., L. W. Brewer, R. R. Hitchcock, and J. R. Mayer. 1992. American widgeon mortality associated with turf application of diazinon AG500. Journal of Wildlife Diseases 28: 263-267.
- Klemens, J. A., R. G. Harper, J. A. Frick, A. P. Capparella, H. B. Richardson, and M. J. Coffey. 2000. Patterns of organochlorine pesticide contamination in Neotropical migrant passerines in relation to diet and winter habitat. Chemosphere 41: 1107-1113.
- Knopper, L. D. and D. R. S. Lean. 2004. Carcinogenic and genotoxic potential of turf pesticides commonly used on golf courses. Journal of Toxicology and Environmental Health, Part B 7: 267-279.
- Knopper, L. D. and P. Mineau. 2004. Organismal effects of pesticide exposure on Meadow Voles (*Microtus pennsylvanicus*) living in golf course ecosystems: developmental instability, clinical hematology, body condition, and blood parasitology. Environmental Toxicology and Chemistry 23: 1512–1519.
- Laporte, P. 1982. Organochlorine residues and eggshell measurements of Great Blue Heron eggs from Quebec. Colonial Waterbirds 5: 95-103.
- LeClerc, J. E., J. P. K. Che, J. P. Swaddle, and D. A. Cristol. 2005. Reproductive success and developmental stability of eastern bluebirds on golf courses: evidence that golf courses can be productive. Wildlife Society Bulletin 33: 483-493.
- LeClerc, J. E. and D. A. Cristol. 2005. Are golf courses providing habitat for birds of conservation concern in Virginia? Wildlife Society Bulletin 33: 463-470.
- Martin, T.E. and G. R. Geupel. 1993. Nest-monitoring plots: methods for locating nests and monitoring success. Journal of Field Ornithology 64: 507-519.
- Mast, M. A., W. T. Foreman, and S. V. Skaates. 2007. Current-use pesticides and organochlorine compounds in precipitation and lake sediment from two high-elevation National Parks in the western United States. Archives of Environmental Contamination and Toxicology 52: 294-305.
- Metcalfe, T. L., P. J. Dillon, and C. D. Metcalfe. 2008. Detecting the transport of toxic pesticides from golf courses into watersheds in the Precambrian Shield region of Ontario, Canada. Environmental Toxicology and Chemistry 27: 811–818.
- Mineau, P. and M. Whiteside. 2006. Lethal risk to birds from insecticide use in the United States -- A spatial and temporal analysis. Environmental Toxicology and Chemistry 25: 1214–1222.
- Mora, M. 1997. Transboundary pollution: persistent organochlorine pesticides in migrant birds of the southwestern United States and Mexico. Environmental Toxicology and Chemistry 16: 3-11.

- Moran, M.D., 2003. Arguments for rejecting the sequential Bonferroni in ecological studies. Oikos 100, 403-405.
- Morrissey, C. A., L. I. Bendell-Young, and J. E. Elliott. 2004. Linking contaminant profiles to the diet and breeding location of American dippers using stable isotopes. Journal of Applied Ecology 41: 502-512.
- Morrissey, C.A., J. E. Elliott, and S. J. Ormerod. 2009. Diet shifts during egg laying: Implications for measuring contaminants in bird eggs. Environmental Pollution In press: doi:10.1016/j.envpol.2009.08.040.
- Moul, I. E. and J. E. Elliott. 1992. A survey of pesticide use and bird activity on selected golf courses in British Columbia. Technical Report Series No. 163. Canadian Wildlife Service, Pacific and Yukon Region, BC.
- Nisbet, I.C.T, and A. F Sarofim. 1972. Rates and routes of transport of PCBs in the Environment. Environmental Science Perspectives 1: 21-38.
- Okoniewski, J. C. and E. Novesky. 1993. Bird poisonings with cyclodienes in suburbia: links to historic use on turf. Journal of Wildlife Management 57: 630-639.
- Porter, E. E., J. Bulluck, and R. B. Blair. 2005. Multiple spatial-scale assessment of the conservation value of golf courses for breeding birds in southwestern Ohio. Wildlife Society Bulletin 33: 494-506.
- Qiu, X., T. Zhu, B. Yao, J. Hu, and S. Hu. 2005. Contribution of dicofol to the current DDT pollution in China. Environmental Science and Technology 39: 4385–4390.
- Rainwater, T. R., V. A. Leopold, M. J. Hooper, and R. J. Kendall. 1995. Avian exposure to organophosphorus and carbamate pesticides on a coastal South Carolina golf course. Environmental Toxicology and Chemistry 14: 2155-2161.
- Ratcliffe, D. A. 1967. Decrease in eggshell weight in certain birds of prey. Nature 215: 208-210.
- Reynolds, K. D., S. L. Skipper, G. P. Cobbs, and S. T. McMurry. 2004. Relationship between DDE concentrations and laying sequence in eggs of

two passerine species. Archives of Environmental Contamination and Toxicology 47: 396-401.

- Rice, C.P., P. W. O'Keefe, and T. J. Kubiak. 2003. Sources, pathways, and effects of PCBs, dioxins, and dibenzofurans. Pages 501-575 *in* Handbook of Ecotoxicology. (D. J. Hoffman, B. A. Rattner, G. A. Burton Jr., and J. Cairns Jr., Eds.). Lewis Publishers, CRC Press LLC, Boca Raton, FL.
- Rodewald, P. G., M. J. Santiago, and A. D. Rodewald. 2005. Habitat use of breeding red-headed woodpeckers on golf courses in Ohio. Wildlife Society Bulletin 33: 448-453.
- Schueler, T. R. 1994. Minimizing the impact of golf courses on streams. Watershed Protection Techniques 1: 73-75.
- Shaw, I. C. and J. Chadwick. 1998. Principles of Environmental Toxicology. Taylor & Francis Ltd., London, UK.
- Spaw, C. D. and S. Rohwer. 1987. A comparative study of eggshell thickness in cowbirds and other passerines. Condor 89: 307-318.
- Stanback, M. T. and M. L. Seifert. 2005. A comparison of eastern bluebird reproductive parameters in golf and rural habitats. Wildlife Society Bulletin 33: 471-482.
- Stone, W. B. and H. Koch. 1982. American Brant killed on golf courses by diazinon. New York Fish and Game Journal 29: 95-96.
- Sughrue, K. M., M. C. Brittingham, and J. B. French Jr. 2008. Endocrine effects of the herbicide linuron on the American Goldfinch (*Carduelis tristis*). Auk 125: 411-419.
- Sun, P., S. Backus, P. Blanchard, and R. A. Hites. 2006. Temporal and spatial trends of organochlorine pesticides in Great Lakes precipitation. Environmental Science and Technology 40: 2135-2141.
- Terman, M. R. 1997. Natural links: naturalistic golf courses as wildlife habitat. Landscape and Urban Planning 38: 183-197.
- Wong, J. W. C., C. W. Y. Chan, and K. C. Cheung. 1998. Nitrogen and phosphorus leaching from fertilizer applied on golf course: lysimeter study. Water, Air and Soil Pollution 107: 335-345.

- Yasukawa, K. and W. A. Searcy. 1995. Red-winged Blackbird (Agelaius phoeniceus in The Birds of North America Online. (A. Poole, Ed.). Cornell Lab of Ornithology, Ithaca, NY. [Online.] Available at bna.birds.cornell.edu/bna/species/184
- Zinkl, J. G., J. Rathert, and R. R. Hudson. 1978. Diazinon poisoning in wild Canada Geese. Journal of Wildlife Management 42: 406-408.

Table 1. Number and proportion of non-viable eggs left in nests for the three most common passerine species and all species pooled on Montreal golf courses and green spaces from 2004-2005. Since four golf courses and two green spaces were studied, the number of eggs divided by the number of sites for each type is also presented.

Species	# eggs on golf courses	# eggs on green spaces	# eggs /	# eggs /	
L	(proportion of total)	(proportion of total)	golf course	green space	
American Robin	23 (0.77)	7 (0.23)	5.8	3.5	
Gray Catbird	6 (0.60)	4 (0.40)	1.5	2.0	
Red-winged Blackbird	20 (0.71)	8 (0.29)	5.0	4.0	
All species $(n = 14)$	83 (0.66)	38 (0.30)	20.8	19.0	

Table 2. Mean and range (ng/g wet weight) concentrations of organochlorine (OC) pesticides and polychlorinated biphenyls (PCBs) in American Robin and Red-winged Blackbird eggs collected on four golf courses and two green spaces in Montreal, Quebec in 2004-2005. Percent of total contaminant burden contributed by the mean of each compound is also shown (%); only compounds exceeding 1.0% of total burden are included.

Compounds <sup>a</sup>	A	merican Roł	pin (n = 8)		Red-w	vinged Bla	ckbird ( $n = 1$	10)
Compounds	[Mean]	[Min] <sup>b</sup>	[Max]	%	[Mean]	[Min] <sup>b</sup>	[Max]	%
PentaCB	29.49	< 0.31	67.25	2.4	25.04	< 0.27	187.04	4.0
HexaCB	7.44	0.60	18.99	0.6	8.42	0.39	45.8	1.4
Oxychlordane	60.15	7.90	139.81	4.8	23.61	<1.37	186.11	4.0
<i>t</i> -Nonachlor	87.44	5.73	261.17	7.0	27.77	0.66	226.85	4.5
c-Nonachlor	19.37	2.16	49.61	1.6	7.52	< 0.12	59.91	1.2
Mirex	307.83	1.03	1476.64	24.8	0.84	0.28	1.72	0.1
<i>p,p'</i> -DDE	350.70	158.25	642.99	28.2	256.08	13.80	1015.22	42.5
p,p'-DDD	8.65	1.45	42.34	0.7	2.74	0.13	<24.67	0.4
<i>p,p'</i> -DDT	32.08	5.29	77.50	2.6	3.03	< 0.23	30.48	0.5
$\Sigma$ DDT	392.04	166.42	720.71	31.5	271.26	14.60	1027.44	43.9
Heptachlor epoxide	11.50	1.02	43.30	0.9	1.84	< 0.27	11.11	0.3
Dieldrin	42.96	0.73	225.24	3.5	1.66	< 0.23	6.75	0.3
Endrin ketone	5.06	4.09	6.75	0.4	6.27	3.47	13.59	1.0
PCB 89/90/101	6.74	3.52	17.57	0.5	9.98	1.85	21.44	1.6
PCB 99	6.46	2.88	18.25	0.5	8.52	1.07	18.56	1.4

PCB 106/118	4.28	1.47	12.33	0.3	14.81	2.06	30.19	2.4
PCB 138/163/164	36.98	12.66	89.42	3.0	23.58	3.66	48.65	3.9
PCB 139/149	4.45	2.57	9.90	0.4	4.33	0.94	10.29	0.7
PCB 146	7.49	2.54	15.73	0.6	4.32	0.62	9.23	0.7
PCB 153	40.64	11.47	99.04	3.3	26.06	4.58	48.56	4.2
PCB 170/190	9.07	2.76	23.08	0.7	6.68	1.10	14.30	1.1
PCB 180	19.85	4.86	70.00	1.6	14.48	2.68	29.00	2.4
PCB 182/187	31.69	11.53	64.13	2.5	9.45	1.47	18.75	1.5
$\Sigma PCBs^{c}$	266.12	114.83	522.25	21.4	235.73	82.46	436.81	38.1
Total	1243.54			90.9	618.52			80.2

<sup>a</sup> Endrin aldehyde, aldrin, hexachlorobutadiene,  $\alpha$ -,  $\beta$ - and  $\gamma$ -HCH, octachlorostyrene,  $\alpha$ -endosulphan, *o*,*p*'-DDE, *o*,*p*'-DDD and several PCB congeners (1, 2, 3, 4/10, 5/8, 6, 11, 12/13, 14, 15, 16/32, 17, 18, 19, 23/24, 24/27, 25, 26, 29, 30, 35, 38, 39, 40, 45, 46, 50, 51, 53, 54, 55, 57, 58, 67, 69, 72, 78, 79, 81, 83/108, 88/121, 96, 104, 112, 113, 122, 125, 129, 131/142, 134/142, 145, 152, 155, 161, 169, 173, 184, 186, 188, 200, 204) were below detection limits for all samples.

<sup>b</sup> Represents detection limit.

<sup>c</sup> Sum of all PCB congeners (for congener # see methods).

Table 3. Mean and range (ng/g wet weight) concentrations of organochlorine (OC) pesticides and polychlorinated biphenyls (PCBs) in Gray Catbird, Northern Cardinal and Barn Swallow eggs collected on four golf courses and two green spaces in Montreal, Quebec in 2004-2005. Percent of total contaminant burden contributed by the mean of each compound is also shown (%); only compounds exceeding 1.0% of total burden are included.

Compounds <sup>a</sup>	G	ray Catbi	rd(n = 4)		Northern Cardinal $(n = 2)$				Barn Swallow $(n = 1)$		
compounds	[Mean]	[Min]	[Max]	%	[Mean]	[Min] <sup>b</sup>	[Max]	%	[]	%	
PentaCB	7.55	0.59	22.88	0.2	11.44	10.84	12.04	1.0	33.28	6.8	
HexaCB	3.11	1.86	4.74	0.1	4.50	3.49	5.52	0.4	5.93	1.2	
Oxychlordane	1228.40	4.38	4870.00	30.0	45.27	11.94	78.59	4.2	18.02	3.7	
<i>t</i> -Nonachlor	1455.15	7.49	5760.00	35.5	14.26	8.98	19.53	1.3	15.95	3.2	
c-Nonachlor	91.71	0.82	359.00	2.2	2.13	1.22	3.03	0.2	9.14	1.9	
Mirex	7.84	0.90	23.18	0.2	2.73	< 0.25	5.34	0.3	6.85	1.4	
<i>p,p'</i> -DDE	684.29	30.00	2440.00	16.7	737.62	125.24	1350.00	67.6	177.59	36.1	
<i>p,p'</i> -DDD	19.59	0.19	66.70	0.5	18.49	6.66	30.31	1.7	1.42	0.3	
<i>p,p'</i> -DDT	26.04	1.49	93.60	0.6	26.80	3.59	50.00	2.5	12.85	2.6	
Σ DDT	736.68	31.89	2626.38	18.0	783.93	182.11	1385.76	71.9	192.18	39.0	
Heptachlor epoxide	47.69	0.99	183.00	1.2	4.84	2.28	7.39	0.4	4.44	0.9	
Dieldrin	4.35	0.73	8.13	0.1	10.14	5.46	14.89	0.9	7.78	1.6	
Endrin ketone	6.13	3.29	9.83	0.1	7.30	5.34	9.25	0.7	4.28	0.9	
PCB 89/90/101	5.47	1.96	12.1	0.1	5.47	3.22	7.719	0.5	6.44	1.3	
PCB 99	29.43	1.23	111.00	0.7	5.91	1.01	10.81	0.5	3.85	0.8	

PCB 106/118	62.34	2.26	235.00	1.5	12.77	2.11	23.44	1.2	10.60	2.2
PCB 138/163/164	53.76	2.62	198.00	1.3	24.03	10.87	37.19	2.2	20.43	4.2
PCB 139/149	2.13	1.08	3.43	0.1	5.77	4.80	6.74	0.5	5.42	1.1
PCB 146	10.35	0.56	37.70	0.3	3.76	1.42	6.11	0.3	5.41	1.1
PCB 153	118.81	3.91	449.00	2.9	27.10	13.11	41.09	2.5	29.05	5.9
PCB 170/190	15.24	0.89	55.50	0.4	6.80	3.26	10.34	0.6	7.43	1.5
PCB 180	29.13	2.16	103.00	0.7	14.91	7.00	22.81	1.4	15.95	3.2
PCB 182/187	11.01	1.381	34.70	0.3	9.08	5.18	12.97	0.8	11.55	2.3
$\Sigma PCBs^{c}$	500.81	48.47	1718.20	12.2	198.64	92.04	305.24	18.2	191.59	38.9
Total	4103.08			95.7	1090.59			91.8	492.295	84.0

<sup>a</sup> Endrin aldehyde, aldrin, hexachlorobutadiene, α-, β- and γ-HCH, octachlorostyrene, α-endosulphan, *o*,*p*'-DDE, *o*,*p*'-DDD and several PCB congeners (1, 2, 3, 4/10, 5/8, 6, 11, 12/13, 14, 15, 16/32, 17, 18, 19, 23/24, 24/27, 25, 26, 29, 30, 35, 38, 39, 40, 45, 46, 50, 51, 53, 54, 55, 57, 58, 67, 69, 72, 78, 79, 81, 83/108, 88/121, 96, 104, 112, 113, 122, 125, 129, 131/142, 134/142, 145, 152, 155, 161, 169, 173, 184, 186, 188, 200, 204) were below detection limits for all samples.

<sup>b</sup> Represents detection limit.

<sup>c</sup> Sum of all PCB congeners (for congener # see methods).



Fig. 1. Scatterplot illustrating the non-significant relationship between log  $\Sigma$  DDT and mean eggshell thickness values for five species of passerine breeding on golf courses and green spaces in Montreal in 2004-2005. The gray bands illustrate 95% confidence limits.



Fig. 2. Concentrations (ng/g wet weight)  $\pm$  SE of the top five OC and PCB compounds measured in the eggs of five passerine species nesting on golf courses and green spaces in Montreal.



Fig. 3. Relative proportion of PCBs (%) for all PCB congeners contributing >5% to the overall PCB average.



Fig. 4. Concentrations in ng/g wet weight ( $\pm$  SE) of the 15 most commonly found chemical compounds detected in Red-winged Blackbird eggs collected on three golf courses (n = 4) and one green space (n = 6) in the Montreal area (2004-2005). Asterisks represent significant Mann-Whitney U values at P < 0.05 indicating concentration differences between eggs from golf courses and from the green space.

Table 4. Nesting statistics for Red-winged Blackbirds nesting on three golf courses and one green space in the Montreal area from 2004-2005 ( $\pm$  SE). Mann-Whitney U tests were used to compare clutch sizes and mean number of young hatched and fledged; z-scores were used to compare % hatching and fledging success.

	Golf courses	Green space	D voluo
	( <i>n</i> = 4)	( <i>n</i> = 6)	r-value
Clutch size	$3.3 \pm 0.5$	$3.7\pm0.2$	0.522
Mean number young hatched	$1.8\pm0.6$	$1.0\pm0.5$	0.393
Mean number young fledged	$1.8\pm0.6$	$1.0\pm0.5$	0.393
Hatching success (%)	$58.3\pm21.0$	$33.3 \pm 17.2$	0.457
Fledging success from hatch (%)	$100.0\pm0.0$	$100.0\pm0.0$	1.000
Fledging success from egg (%)	$66.7\pm31.2$	$29.2 \pm 14.2$	0.270

Table 5. Spearman correlations (R) between Red-winged Blackbird nest site (golf courses vs. green space), reproductive parameters and the five most commonly found compounds in 10 egg samples collected from Montreal sites in 2004-2005. Significant values at P < 0.05 are bolded.

	Site	Clutch	Hatched	Fledged	%	%	<i>p</i> , <i>p</i> ′-	Σ	t-	Oxy-	С-
	Site Cluten	Clutch	Hateneu	Tieugeu	hatched	fledged <sup>a</sup>	DDE	PCBs <sup>b</sup>	Nonachlor	chlordane	Nonachlor
Site	1.000										
Clutch	-0.244	1.000									
Hatched	0.298	-0.542	1.000								
Fledged	0.298	-0.542	1.000	1.000							
% hatched	0.298	-0.542	1.000	1.000	1.000						
% fledged <sup>a</sup>	0.294	-0.646	0.987	0.987	0.987	1.000					
p,p'-DDE	-0.071	0.166	0.102	0.102	0.102	0.119	1.000				
$\Sigma PCBs^b$	0.782	-0.472	0.744	0.744	0.744	0.734	0.139	1.000			
<i>t</i> -Nonachlor	0.569	0.243	0.254	0.254	0.254	0.194	0.406	0.527	1.000		
Oxychlordane	0.569	-0.069	0.464	0.464	0.464	0.414	0.358	0.673	0.867	1.000	
c-Nonachlor	0.640	0.055	0.369	0.369	0.369	0.339	0.539	0.685	0.952	0.879	1.000

<sup>a</sup> Percent fledged from eggs.

<sup>b</sup> PCBs are represented by ΣPCBs, which was correlated to all other PCBs in the 15 most commonly found compounds.

### CONNECTING STATEMENT 5

This thesis contains a series of manuscripts addressing the general question: are golf courses able to support breeding bird populations as well as other urban green spaces? These manuscripts focus on breeding bird communities present on both golf courses and on green spaces and how they relate to one another, a novel method for monitoring birds' nests, nest success rates and the factors that influence these rates, and finally, the current contaminant loads in birds breeding on sites with historical pesticide use. In this chapter, I summarize my major findings, and provide recommendations for future studies based on my research.

# CHAPTER 6

# SUMMARY, CONCLUSIONS AND FINAL RECOMMENDATIONS FOR MANAGEMENT AND FUTURE STUDIES

#### SUMMARY

From 2003-2006, I compared avian communities and reproduction on Montreal-area golf courses and green spaces to determine whether golf courses could support breeding bird communities as effectively as other types of suburban green space. First, I compared breeding bird communities on six golf courses and six green spaces. The mean number of avian species on each site type was similar, but golf courses supported fewer total species. However, one exceptional green space contributed nine species to the cumulative total; without that site the green space total dropped to within two species of the golf course total. Avian communities differed between golf courses and green spaces, and were most correlated with the size of the site, housing density surrounding the site, and the extent of coniferous tree, grass, and vegetated water cover.

Second, I described a novel, portable nest monitoring method that allowed live video and images to be archived with minimal disturbance to nesting birds and third, I used this method to monitor almost 900 nests on four golf courses and two green spaces. Nesting density and Mayfield survival rates for the most common open-cup nesting species using these sites varied extensively, but there was no difference between nesting density and Mayfield survival rates between golf courses and green spaces when all upland-nesting species were pooled together. Gray Catbird and Cedar Waxwing were less productive on golf courses than green spaces, while American Robin, Red-winged Blackbird and Northern Cardinal were equally productive on both site types. I identified the temporal, landscape and micro-habitat characteristics influencing nest survival rates of the two most common species, American Robin and Red-winged Blackbird, as well as all upland-nesting species pooled together. Modelling of all species indicated that nest survival was influenced by temporal and patch-level factors such as year, type of nesting substrate and arrangement, and the interaction between these two variables. While American Robin nest success was not influenced by any of the variables we modelled, Red-winged Blackbird nest success was related to nest age and the degree of concealment around the nest. Suburban green spaces, including golf courses, can play a role in maintaining healthy bird populations by following carefully designed management plans, at both planning and landscaping scales.

However, the risk of exposure to historically used contaminants must be kept in mind, especially in these urban/suburban areas. I quantified the amounts of organochlorine and polychlorinated biphenyl compounds in addled or infertile eggs of five passerine species opportunistically collected in 2004-2005. Of the 191 different chemicals assayed, 32 were detected in all samples. Though concentrations and relative proportions of each pesticide varied substantially between and within species, most chemicals were found at trace levels or below detection levels. Reproductive parameters for Red-winged Blackbird such as clutch size and the number of young hatched and fledged did not differ between golf courses and green spaces, nor did they appear affected by pesticide contamination despite higher PCB levels in eggs collected from nests on golf courses.

## CONCLUSIONS

Management recommendations for the creation and maintenance of wildlife-friendly habitat on golf courses have existed for years, but have largely been based on inferred (e.g. Tietge 1992, Smart et al. 1993, Dodson 2000, Gillihan 2000) and hypothetical data (e.g. Mankin 2000). Many of these recommendations have never been directly tested on these sites to see whether they affect avian presence and breeding. Understanding the influence of site-, patch- and nest-site level characteristics on nest success, and how these are modified in an urban environment and through different land uses, may provide both land managers and conservationists with the tools required to make more informed decisions. Despite the large amount of inter-site variability in this system, two main conclusions are apparent:

1) Avian communities differ between Montreal-area golf courses and green spaces, but show a fair degree of overlap; these communities are driven by specific land-cover characteristics. This suggests that these communities may be enriched and/or modified by managing golf courses to more closely resemble the high-quality green spaces highlighted in this study. These sites differed from golf courses by having highly vegetated water bodies, less highly managed turfgrass and unusable surface cover (e.g. buildings, paved roads), and more deciduous tree cover. However, they were similar in that they were heterogeneous; containing a wide variety of vegetation types. If, as a first step, vegetation volume (not including highly managed turfgrass) on golf courses can simply be increased, habitat quality will increase as a function of vegetation complexity. More specifically, our results suggest that, to increase avian species diversity, on-site management techniques should include: i) increasing the amount of continuous dominant vegetation by planting patches of coniferous trees, iii) reducing the amount of highly managed turfgrass; and iv) increasing the area covered by vegetated water and encouraging the growth of emergent, aquatic and terrestrial vegetation within and surrounding all water bodies.

To increase nest success rates over all sites, species-specific nesting and feeding requirements must be understood if management is to be beneficial for target species. Overall, we only found three species with large differences in nest success rates between golf courses and green spaces. Rather, patch-level vegetation characteristics, regardless of site type, appear to influence most species' nest survival. For example, conifer patches and hedges provided high nest success rates when compared with other nest substrates, whereas isolated deciduous trees provided very low success rates. Conifer cover was also associated with higher species richness, so increasing the number of hedges and conifer patches may benefit both richness and nest success. Avoiding the structural simplicity of singly planted deciduous trees by either increasing the amount of understory vegetation or by planting additional trees to create stands will not only help naturalize the site, but will likely also positively influence nest success rates by providing more protected nest sites. Overall, managers should strive for more natural golf courses and green spaces that appear minimally managed to improve the quality of habitat provided by these sites. Fortunately, historical use of pesticides does not appear to impact avian reproduction at these sites, so managers can focus their efforts on habitat management.

2) Species richness, though influenced by on-site vegetation and landscape, is ultimately capped by increasing urbanization. Avian nest success, on the other hand, appears to increase with higher levels of urbanization, at least within a suburban landscape. There appears to be a trade-off between increasing species richness and increasing the likelihood of nest survival at the sites we studied. Thus, management goals should be outlined early: do managers want to promote avian diversity or nest success? This may depend on location within a landscape (i.e. whether the site is well positioned to act as a migratory stopover site) as well as the type of vegetation present on-site. However, it makes more sense to promote higher nest success in fewer species than to manage a site in such a way as to incite failure across a greater number of species. Future studies should examine this apparent balancing act between species diversity and nest success in a suburban landscape, and whether this result is truly applicable to other landscapes and cities.

To complicate matters, there might be an interaction between land-use type (e.g. golf course or green space) and level of urbanization. It is possible that golf courses embedded in a natural setting may be exposing birds to a higher likelihood of predation than green spaces due to the predominance of singly planted trees, which provided the highest likelihood of predation for nesting birds. However, golf courses embedded in a suburban landscape may provide safer breeding grounds than more natural green spaces surrounded by urbanization, perhaps due to reduced predator habitat availability both on and off the site. This interaction deserves further study, as the results may change the way suburban sites are cited, planned and managed, with different management techniques designed for urban-embedded sites vs. non-urban embedded sites. *Recommendations for future study.* -- This collection of studies was a step in identifying the factors that affect avian reproduction on suburban golf courses and green spaces. Despite learning a great deal about how these factors may impact nesting birds, these studies highlight how much there is left to investigate. I have recommended many topics for future study throughout this dissertation, and highlight additional ones here.

Our results indicate that some avian species appear to breed quite successfully on golf courses, while others do very poorly. We were not able to completely disentangle the variety of factors that may have affected these birds' survival. Thus, I recommend that future studies continue and expand the work presented in these pages: how do suburban/urban golf courses and green spaces support common, open-cup nesting birds, and how these sites can be improved for maximum benefit to humans and wildlife?

It is very difficult to understand how a microcosm functions without understanding the ecosystem to which it belongs. Thus, wildlife groups both above and below birds in the food web should be examined. For example, where do golf courses lie along urban gradients in terms of predator composition and abundance? Does the simplified vegetation structure on golf courses increase the likelihood of predation in a natural landscape while having little effect when embedded in an urbanized area?

Virtually nothing is known of the insect life on golf courses. The few studies that have taken preliminary steps towards describing insect life on a golf course have simply looked at differences in gross abundance between golf course and non-golf habitats (Stanback and Seifert 2005). However, New (2005) suggested that older golf courses in Victoria, Australia, may provide important habitat for grassland insects. Through their establishment, golf courses have preserved large tracts of native grassland, one of the most endangered habitat types in Australia. It is important to understand, not only from a biodiversity point of view, but also from an avian physiology and food web point of view, how well insects can survive on sites where insect pests are controlled for the sake of recreation. Higher food availability generally leads to earlier and better breeding condition, as well as increased reproductive success (Martin 1987). Insects are also an essential part of a nestling's diet. If there is less food available on golf courses, then birds will have to forage farther from the nest, which will likely reduce both adult and nestling survival. Future studies should measure how the various management techniques suggested here affect insect richness and abundance. In addition to reduced food availability, the risk of contamination by various pest control agents cannot be ignored, either through ingestion or dermal absorption, as organophosphates and other chemical compounds are still being used on golf courses and recreational parks worldwide. I recommend studies examine whether organophosphates and other sources of contamination (e.g. fungicides and herbicides) are found in breeding and migrating birds using golf courses and other suburban/urban green spaces.

Use of a golf course by breeding birds accounts for roughly one third of the year (May through August in eastern North America), when most defend a territory to the exclusion of others. One could argue that it is more important to create and maintain golf courses and other urban green spaces to encourage usage by migrants as stop-over sites, and during the winter as little disturbed (in temperate areas) roosts and feeding areas. Thus, a single site could play host to thousands of birds and not restrict itself to a few hundred during the breeding season. Fortunately, most recommendations made to improve breeding habitat will also improve habitat for the needs of migrants and winter residents.

A final footnote. -- Despite not directly testing the various way that management techniques could be implemented, I found that the most effective techniques appear to be the ones proposed and explained to planners and managers in person, with results improving with follow-up. Though arguably not recommended during the course of a study to preserve standardization, encouraging management changes on-site appeared to be the most effective way to modify management techniques. We found that managers were much more likely to modify their techniques if someone took the time to explain the costs and benefits of each action, as opposed to relying on published work. The importance of correct

management techniques on these sites is paramount, and can certainly impact nesting birds. For example, the number of nests lost due to landscaping can be reduced or even eliminated by substituting native plants that do not require as much maintenance (e.g. conifers) or by restricting landscaping activities to outside the breeding season. We found that showing workers how to check for birds' nests (both in shrubs and in cattails) was a simple and effective way to prevent accidental nest losses, and workers seemed interested and protective of the birds found nesting on courses. The modification to cattail management at one of the sites also proved effective and resulted in almost immediate changes on-site, with the number of Red-winged Blackbird nests increasing with the amount of vegetation left un-cut from the previous year. The management techniques proposed here also likely have the advantage of producing almost immediate results, as birds are highly mobile and are quick to exploit new habitats and food sources.

#### LITERATURE CITED

- Dodson, R. G. 2000. Managing Wildlife Habitat on Golf Courses. Ann Arbor Press, Chelsea, MI.
- Gillihan, S. W. 2000. Bird Conservation on Golf Courses: A Design and Management Manual. Ann Arbor Press, Chelsea, MI.
- Mankin, K. R. 2000. An integrated approach for modelling and managing golf course water quality and ecosystem diversity. Ecological Modelling 133: 259-267.
- Martin, T. E. 1987. Food as a limit on breeding birds: a life-history perspective. Annual Review of Ecology and Systematics 18: 453-487.
- New, T. R. 2005. Recreation and reserves: values of golf courses for insect conservation. Journal of Insect Conservation 9: 1-2.
- Smart, M., J. D. Spencer, R. N. Calvo and C. H. Peacock. 1993. Working with nature for better golf developments. Urban Land 52: 17-22.

- Stanback, M. T. and M. L. Seifert. 2005. A comparison of eastern bluebird reproductive parameters in golf and rural habitats. Wildlife Society Bulletin 33: 471-482.
- Tietge, R. M. 1992. Wildlife and golf courses. Pages 441-478 in Golf Course Management & Construction: Environmental Issues. (J. C. Balogh and W. J. Walker, Eds.). Lewis Publishers, Chelsea, MI.