A COMPARISON OF LANDSCAPE BASED METHODS

FOR CONSERVATION PLANNING

A Thesis

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ABSTRACT

A variety of well-conceived conservation planning approaches have been developed over recent years, all with the critical component of setting conservation priorities for both species protection and land acquisition and management. Ideally, the selection of the sites comprising a conservation portfolio would be based on detailed survey data, but this is not always possible due to time and monetary constraints. When it is not possible or feasible to conduct surveys of even some portion of the species in an area, basic ecological principles can be applied. The overall goal of this thesis is to examine the extent to which commonly used and easily obtained landscape metrics contribute to a conservation planning process that must be completed rapidly and with little financial resources. Specifically, I aimed to determine if landscape metrics can be used to guide the site prioritization process when the relative importance of potential conservation sites to various conservation portfolios is measured using their relative irreplaceability values. This question is examined in the context of a flexible planning process where the scope of a given project (represented by conservation portfolio size) may vary over time. I focused this

study on the conservation of riparian forest ecosystems in urbanizing Midwestern landscapes and used bird communities to indicate the ecological value of particular forest tracts.

Evaluating the usefulness of landscape metrics in the prioritization process required two steps. First, I compared two easily-applied alternate approaches to using avian species data to indicate the value of a site in a conservation planning context (as measured by relative site irreplaceability). Specifically, I compared the use of avian species richness to the use of a weighting system based on a conservation threat score developed by Partners In Flight. In addition, I compared irreplaceability scores created via two different compilation methods (1) averaged species occurrence data among multiple years versus (2) a cumulative species richness that considers all species recorded on sites over multiple years. These methods were compared using the sign test. Second, I directly examined the relative utility of landscape metrics in predicting the importance (as measured by irreplaceability) of a given site over a range of conservation portfolio sizes. Multinomial logistic regression models were created for 21 different models based on ecological principles and compared using Akaike Information Criterion.

My findings suggest that while the method of survey data compilation had little effect on irreplaceability values among sites, use of a weighting scheme that places greater emphasis on vulnerable species can significantly influence site prioritization. My results also confirm that landscape metrics are useful indicators of the value of particular sites within conservation portfolios. In particular, irreplaceability value of a site to a conservation portfolio was most consistently and simply predicted using forest coverage within a 1-km landscape surrounding potential portfolio sites. Use of an additional landscape metric describing human disturbance (e.g. number of buildings, percentage of area covered by roads, pavement, mowed surfaces, or agricultural land) improved model fit substantially (i.e. decreased the log-likelihood score) and is recommended. This combination of forest and disturbance metrics was useful across a range of portfolio sizes (protecting 6-23% of possible sites) and is therefore likely to remain useful even in the context of a planning process whose scope varies over time. These results show that simple landscape metrics can aid land managers and planners that need to make rapid decisions about prioritizing land acquisition, preservation or management activities.

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TABLE OF CONTENTS

LIST OF TABLES

Table Page

Table Page

LIST OF FIGURES

CHAPTER 1

INTRODUCTION

1.1 BACKGROUND

A variety of well-conceived conservation planning approaches have been developed over recent years (Groves 2003, Beazley et al. 2005, Borges et al. 2005, Haight et al. 2005, Shi et al. 2005, Burgess et al. 2006, Cabeza and Moilanen 2006, Freemark et al. 2006, Knight et al. 2006, Rouget et al. 2006, Turner and Wilcove 2006), all with the critical component of setting conservation priorities for both species protection and land acquisition and management. Prioritization is a necessity within every planning framework because time and financial constraints generally prevent the safe-guarding of all potentially important lands (Brooks et al. 2006, Cabeza and Moilanen 2006, Wilson et al. 2006). Planners typically work to select a conservation portfolio (Groves 2003, Burgess et al. 2006), or a group of sites that cover the full range of conservation targets, that can help guide future actions. Theoretically, the design of a conservation portfolio is much preferred over the selection of a single site as it allows the planner to protect a larger variety and number of species and habitats. However, in practice, the job of selecting a portfolio of sites can become quite complicated, especially when the number of potential sites is large. When given a collection of potential conservation areas and the task of selecting a subset of them for conservation, an intuitively attractive approach is to simply rank the sites according to their individual species diversity, species richness or other applicable criteria and then choose the top sites from the list to put into a portfolio. However, this approach overlooks the importance of three important principles for setting land priorities: complementarity, flexibility, and irreplaceability (Pressey et al. 1993, Groves 2003, Borges et al. 2005, Turner and Wilcove 2006, Williams et al. 2006).

Complementarity results from an iterative process that first examines existing conservation sites and then identifies which new conservation sites would contribute the greatest value (in terms of diversity, processes, or other conservation targets) not already represented in the existing sites (Groves 2003). Thus, failing to employ the principle of complementarity can result in the duplication of some conservation targets with the neglect of others. Portfolios designed with flexibility in mind provide multiple options for planners, allowing them to adaptively change land acquisition strategies in response to changing project scope, constraints, or opportunities (Groves 2003, Haight et al. 2005, Cabeza and Moilanen 2006, Turner and Wilcove 2006). Finally, by incorporating the principle of irreplaceability, a planner can identify in an empirical and explicit fashion the probability that any given site is contained within the portfolio of sites required to reach conservation goals. Therefore in practice, planners must consider a wide array of potential conservation portfolios and compare the overall coverage of each before making planning decisions (Groves 2003).

Ideally, the selection of the sites comprising a conservation portfolio would be based on detailed surveys consisting of presence/absence data, demography and life histories of species, interactions between species, and interactions of species with their environment (Brooks et al. 2004, Burgess et al. 2006). However, due to time and monetary constraints, decisions are usually based on more limited data. When it is not possible or feasible to conduct surveys of even some portion of the species in an area, basic ecological principles can be applied (Meffe and Carroll 1997, Wiersma et al. 2004, Hess et al. 2006). For example, measures reflecting patch size (area) and landscape composition can predict species diversity for certain taxa and for some sensitive species of conservation importance (Saveraid et al. 2001, Lombard et al. 2003, Wiersma et al. 2004, Hess et al. 2006). Research suggests that forest width, the quality of the habitat within a forested area, and the land use and cover type of areas adjacent to a forested area are the dominant factors determining the overall value of a forested area as wildlife habitat (Friesen et al. 1995, Mensing et al. 1998, Rottenborn 1999, Hennings and Edge 2003, Miller et al. 2003, Wiersma et al. 2004, Mason 2006, Rodewald and Bakermans 2006). Though these landscape-scale surrogates have been widely applied in the literature (Johnson 1995, Beazley et al. 2005, Cook 2002, Livingston et al. 2003, Kati et al. 2004), they have typically been used to identify and/or rank the importance of individual sites without considering irreplaceability.

For the comparison of the utility of landscape surrogates for site prioritization, one must have a method of determining which suite of sites represents the "truly" most important (i.e. highest priority) areas. In many situations, the development of this baseline prioritization is made possible through the use of large-scale inventory and survey efforts, such as Natural Heritage, Breeding Bird Atlas and Breeding Bird Survey databases. As these species distribution data become more readily available, they are increasingly incorporated into conservation planning efforts. Species data are useful in a planning context because they can provide important indicators of the relative condition of sites being considered for protection and, in that way, can facilitate development of conservation priorities (Noss 1993, Donovan et al. 2002, Bestelmeyer et al. 2003, Groves 2003, Bennett and Milne 2004, Brooks et al. 2004, Pressey 2004, Fleishman et al. 2006, Hess et al. 2006, Pearman et al. 2006, Tchouto et al. 2006). However, species data can be applied to the prioritization process in a variety of ways (e.g. species richness, presence of threatened and endangered species, presence of indicator species), and different approaches can yield widely different results (Dunn et al. 1999, Fleishman et al. 2006, Pearman et al. 2006). For example, use of species richness (total number of species per area) could result in overestimating the value of sites containing large amounts of edge habitat or sites containing large numbers of generalist or even exotic species.

I focused this study on mature riparian forest areas along an urban-to-rural gradient in the Midwestern state of Ohio. These areas are of interest in a planning context as over 90% of riparian habitats have been lost during the past 200 years due to disturbances caused by water management practices, agriculture, grazing, channelization, timber removal, industry, mining, urbanization and recreation (Knopf et al. 1988, Malanson 1993, Jobin et al. 2004). Governmental agencies and conservation organizations frequently invest large amounts of resources in the acquisition, retention and management of riparian forest areas due to their high ecological, social and recreational value. Riparian forests provide habitat for a variety of species, protect water quality, perform critical functions in hydrological and biogeochemical cycles, and are frequently chosen as the location for parks, bike paths and greenways. Birds were used as a focal group because their habitat requirements are often specialized, they are abundant, relatively easy to survey, toward the top of the food chain, have relatively large home ranges, and are frequently used in practice as indicators of forest habitat quality (Brooker 2002, Miller et al. 2003, Mason 2006, Rodewald and Bakermans 2006).

1.2 THESIS OBJECTIVES

In the first part of this thesis I compared two easily-applied alternate approaches to using avian species data to indicate site irreplaceability, or “the probability that a potential conservation area will be required as part of a network of conservation areas” (Groves 2003), which is a commonly used index in conservation planning efforts. Specifically, I compared the use of avian species richness to the use of a weighting system based on a conservation threat score developed by Partners In Flight (Regional Combined Score for the Breeding Season) as part of their effort to conserve landbirds in the Western Hemisphere. Their scoring system involves the assignment of scores to individual bird species based on their breeding distributions, regional and global population size, population trends and threats faced with the goal of prioritizing and focusing conservation efforts on species of highest need (Panjabi et al. 2005, Partners in Flight 2005). In addition, I compared irreplaceability scores created by averaging species occurrence data among multiple years versus a cumulative species richness that considers all species recorded on sites over multiple years.

In the second part of this thesis I examined the extent to which commonly-used and easily obtained landscape metrics (e.g. number of buildings, percent forest cover, agricultural land, mowed surfaces, paved surfaces, roadways) could contribute to a site prioritization process that considers irreplaceability and flexibility. Specifically, I aimed to (1) determine if landscape metrics can effectively guide the development of a conservation portfolio by predicting the relative importance of potential conservation sites, as measured by irreplaceability scores, and (2) evaluate flexibility by examining this question for a range of conservation portfolio sizes to determine the utility of landscape metrics in a flexible planning process where the scope of a given project may vary over time due to changes in funding and/or timelines.

1.3 LITERATURE REVIEW

Riparian areas, or “aquatic ecosystems and the portions of the adjacent terrestrial ecosystem that directly affect or are affected by the aquatic environment . . . [including] portions of hillslope that serve as streamside habitats for wildlife” (Voller 1998), are of great conservation concern in North America. Over 90% of riparian habitats have been lost during the past 200 years due to disturbances caused by water management practices, agriculture, grazing, channelization, timber removal, industry, mining, urbanization and recreation (Knopf et al. 1988, Malanson 1993, Jobin et al. 2004). Bird populations are especially dependent on these diminishing riparian areas both for breeding and migration purposes (Knopf et al. 1988, Rich 2002).

Neotropical migratory birds extensively utilize riparian areas due to the diversity of plant species and forms, presence of a wide range of foliage heights, a heterogeneous mix of open and densely vegetated areas and a relatively high frequency of nesting habitat. Habitat loss can have dramatic effects on bird populations despite their relatively high mobility due to the fact that most avian species are closely associated with particular habitats and have specific ecological needs (Farley et al. 1994). Due to declines in the population sizes of many of these migratory songbirds, Partners in Flight (an international bird conservation initiative) has sought to develop Bird Conservation Plans. One of the crucial components of these plans is the identification of high quality habitats and landscapes that promote high survival and reproduction for these species. One major research priority outlined by the group includes determining how to assess quality habitat in a cost-effective manner (Donovan et al. 2002).

Ideally, all riparian land would be protected for the continued use by wildlife. However, due to the large amount of private ownership of land and the demand for further growth, in practice it is only possible to preserve some portion of the remaining riparian areas. This leads us to the question of how best to select the land most vital to the overall survival of birds and other wildlife. Under ideal circumstances, detailed surveys incorporating knowledge of each species' demographic characteristics and interactions with other species and the environment would be used to guide conservation decisions. Unfortunately, this type of approach is not only time consuming and costly, but extremely difficult to implement in most cases (Moore et al. 2003). As management decisions often need to be made relatively quickly, less intensive shortcut approaches are more commonly used (Andelman and Fagan 2000, Hess and King 2002, Hess et al. 2006).

When it is not possible or feasible to conduct surveys of all or some portion of the species in an area it becomes necessary to apply basic ecological principles to the selection of land for conservation. Some conservation principles that are commonly used to prioritize habitat for protection include: preservation of a variety of habitat types, planning for corridors between isolated patches, the presence of waterways and associated riparian vegetation, the consideration of natural processes such as fire and flooding regimes, and proximity to currently protected areas. In general these issues can be considered in six groups: patch size, heterogeneity and dynamics, landscape context, connecting fragmented habitats, natural and modified landscape elements, and buffer zones (Meffe and Carroll 1997).

One commonly used simple approach is to preserve the largest pieces of habitat possible. This approach is based on the assumption that large habitat patches typically contain larger, more viable populations, a diversity of habitats, offer greater resources and support intact ecological processes such as disturbance regimes.

Another benefit of large patches is that they contain large amounts of interior habitat and fewer edges (when compared to a set of smaller patches covering the same area). While some species actually benefit by the existence of edge habitat, most threatened and endangered species are negatively affected due to an increase in predation and altered environmental conditions. In addition, interior areas, separated from the exterior matrix often provide critical habitat for a variety of endangered and threatened species (Dramstad 1996, Meffe and Carroll 1997, Poiani et al. 2001, Roberge and Angelstam 2004, Hess et al. 2006).

Heterogeneous areas are generally better than homogeneous ones in terms of conservation for biological diversity. This is true due to the fact that nature is dynamic and constantly changes over time through disturbances such as fires, tree-falls, disease, floods, and herbivory. Conserving a variety of different habitat types allows natural disturbances or succession to occur without destroying all suitable habitats for a given group of species (Meffe and Carroll 1997).

The type of habitat present (both within and exterior to a patch), or landscape context is another important consideration in conservation. Some species have very specific needs for food and shelter, while others are more adaptable. Some species utilize different patch types throughout a season or over different life history stages (Meffe and Carroll 1997). Habitat exterior to a patch (the matrix) is also important as it can determine the severity of edge effects, provide alternative habitats and moderate area effects. For example, some area sensitive species can inhabit smaller patches of land if that land is surrounded by agricultural land as opposed to suburban and urban development (Dramstad 1996, Hess et al. 2006, Meffe and Carroll 1997).

Connectivity and the relative proximity of patches are also important issues to consider in land preservation. While some species move between various habitat types on a daily basis, others require corridors for yearly migrations or the dispersal of juveniles into new territories. The existence of corridors and quality habitat patches clustered close to one another can increase the flow of genes between populations thereby decreasing the likelihood of extinction (Dramstad 1996, Meffe and Carroll 1997, Hess et al. 2006).

Both natural and modified landscape elements play a crucial role in the suitability of habitat. Natural elements include features such as drainage basins, ridges, slopes and canyons. Modified elements encompass features such as roads, highways, agricultural fields, industrial zones and cities. In general, increasing the diversity of natural elements and decreasing the presence of modified elements will increase the value of a given area for wildlife. In addition, the inclusion of a natural element in its entirety is much more beneficial than the inclusion of a small portion of that element (Meffe and Carroll 1997).

One last category to consider in general land preservation is the existence of buffer zones around the area to be protected. Buffer zones serve a variety of purposes including decreasing the severity of edge effects, limiting conflicts between the wildlife being protected and landowners in surrounding areas, and limiting the spread of natural disturbance events such as fire and flooding (Meffe and Carroll 1997). While many of these principles can be used in a general context, when prioritizing land for a particular group of species it is very important to specifically evaluate the needs of that species as some of the factors may be more relevant than others.

Due to the criticisms waged at each of the above approaches independently, some have suggested the possibility of designing approaches that combine several concepts together. In fact, some believe that the newest trend in conservation planning is the combination of a variety of approaches so as to meet several criteria simultaneously (Kati et al. 2004). Saveraid et al. (2001) conducted a study in Grand Teton National Park, Bridger-Teton National Forest and Yellowstone National Park to determine the utility of using many of the landscape criteria mentioned above as a predictor of bird community structure. Through comparisons of fine-scale habitat and vegetation data, coarse scale satellite imagery and bird surveys, they found several landscape and habitat variables (stem density, distance to treeline and meadow area) that were strong predictors for some of the bird species. Coarse scale satellite data were not useful on their own as predictors of avian diversity, but these data were found to be very useful for identifying potential areas that could then be examined on the ground to gather more detailed vegetation and habitat data. In general, the authors conclude that a combination of remotely sensed (coarse scale) and ground-based (fine scale) data will act as a useful indicator of species occurrence.

A wide variety of techniques, encompassing a diverse range of monetary, time and personnel resources, exist for conservation planning. As conservation decisions must often be made very quickly it is not always possible to use the most thorough method as it may be very time intense. This leads to the question of how to balance time and money constraints with the need for good science. One way to answer to this question is to compare the performance of the various methods on a landscape where the species distribution and abundances are known. While various studies have compared one of the described techniques to random land selection, only a few comparisons have been completed looking at multiple techniques at the same time (Poiani et al. 2001, Hess et al. 2006).

A large scale study in North Carolina used an inventory based plan as a standard by which to compare the success of planning using focal species and basic conservation principles (large patches, proximity to riparian areas, proximity to currently protected areas, and the diversity of forest types). The measure of effectiveness used (representation, completeness or overlap) was found to be of crucial importance in determining the success of the various methods. When representation (the proportion of species and communities represented) was used as the measure of success, the effectiveness of planning based on either focal species or basic conservation principles was comparable and relatively high. Both types of planning techniques were less effective when measured using completeness (the proportion of element occurrences captured) or by overlap (the proportion of land in the inventory-based plan included) (Hess et al. 2006).

Manne and Williams (2003) found that while choosing areas at random produced results comparable to the use of a group of indicator species when the number of areas chosen was large, the indicator species approach was much better than the random selection of areas when the number of areas chosen was small relative to the total area. Kati et al. (2004) completed an analysis of small scale reserve design in Greece and found that data intense focal species approaches performed better than simple ecological principle approaches based on general habitat and vegetation criteria. In addition, these simple approaches were found to perform better than a random selection approach.

Previous research suggests that in some situations, less time and cost intensive methods of land prioritization may be just as effective as more expensive and field intense ones. In order to determine the general applicability of these results, it will be necessary to complete more large scale studies comparing a variety of the different techniques in a variety of different ecosystems, on various spatial scales, and using diverse groups of species (Hess et al. 2006).

Birds have been used in several instances as indicator or focal species as they are abundant, charismatic, relatively easy to survey, are sensitive to environmental change, show habitat specificity, and have relatively large home ranges (Brooker 2002). In order to effectively apply the methods described above to the conservation of riparian areas for bird communities it is necessary to understand not only the specific habitat requirements of bird species, but also the threats faced by these species in the form of human development and altered ecological processes.

Brooker (2002) applied a focal species approach to the conservation and management of diminishing natural habitat in the central wheat-belt of Western

Australia. Major threats were identified (including the loss and fragmentation of habitat, the loss of critical resources, and inappropriate rates and intensities of ecosystem processes such as predation, nutrient cycling and fire) and a focal community of land birds representing these threats was chosen. Using this information, key areas of habitat were identified and an ecological neighborhood approach was used to recommend ways of building habitat patches from existing, scattered native vegetation. In the United States, urbanization has played a crucial role in the loss of riparian habitat, thereby impacting riparian bird communities. Effects on birds occur both directly and indirectly through changing ecosystem processes, disturbance regimes, habitat and food supply, and altering populations of predators, competitors and disease organisms (Marzluff et al. 1998, Marzluff 2001, Hennings and Edge 2003). These changes affect not only the population biology of specific species, but also change the structure and composition of entire bird communities (Marzluff 2001).

Some bird species are able to exploit urban environments and benefit from the less severe climate, abundant food and water, reduction in predators, and increase in nesting sites. This can lead to lengthened breeding seasons, increased survival, and increased productivity, which in turn may promote stable and dense populations. In contrast, many native species, particularly Neotropical migratory species, show declines due to the scarcity of natural habitat, increases in predators, parasites, or competitors, and a general intolerance to human activity. These two contrasting processes often lead to an uneven distribution of avian species, with communities dominated by a few, very abundant, non-native species (Marzluff et al. 1998, Marzluff 2001, Hennings and Edge 2003).

Human development results not only in habitat loss, but also decreases fragment sizes and quality, increases isolation and lowers connectivity between suitable habitats (Fernandez-Juricic 2004). Even when riparian woodlands remain intact, nearby urbanization has been found to have a strong effect on bird communities with species richness decreasing with the distance to the nearest development and width of the riparian habitat. Narrow urban forests often favor non-native plants and birds. The combination of narrow forests with high road density has been shown to favor resident and short-distance migrant species in some areas. In addition, both species richness and abundance were found to be negatively related to the proximity and abundance of bridges (Rottenborn 1999, Hennings and Edge 2003). Hennings and Edge (2003) suggest that increasing canopy cover within 450 m of important riparian habitats and decreasing street density within a 100 m radius of riparian areas might provide positive benefits to populations of Neotropical migratory birds.

In general, there appears to be a strong interest in how settlement patterns in urban areas affect both birds and other components of biodiversity. A variety of general ecological principles applied at the landscape level can be used by planners and managers to determine the best arrangements of settled land and conserved green space in urban areas. Key factors to consider in planning include the maintenance of native vegetation, minimization of the loss of structural diversity and an active reduction in the impacts of non-native species on bird productivity. Bird conservation is especially important as birds seem to be important signals of overall urban ecosystem health and biodiversity. While a significant amount of information pertaining to the preservation of birds in urban areas has been gathered in recent years, "the functioning of reserves and corridors needs more testing in urbanizing settings to be effectively applied. Planning based on mechanistic understanding of how bird populations respond to settlement pattern will more likely have its desired outcome" (Marzluff 2001). Conservation practices must also be dynamic, just like the ecosystems they are dealing with. Due to the fact that everything in the natural world is constantly changing, decisions concerning reserves must be regularly evaluated and modified to meet newly emerging needs and threats. It is very important to remember that "in practice, conservation decisions are informed, not dictated by science" (Kati et al. 2004).

1.4 THESIS FORMAT

Chapter 2 contains the comparison of approaches to using avian species data for the indication of the value of a site in a conservation planning context as measured by relative site irreplaceability (Objective 1). The chapter is presented in the format of a note for possible submission to Biological Conservation or Ecological Indicators. Chapter 3 examines the utility of using landscape metrics to predict site irreplaceability in a conservation planning context (Objective 2). The chapter is presented in the format of a paper for possible submission to Landscape and Urban Planning.

CHAPTER 2

APPROACHES TO USING AVIAN SURVEY DATA FOR SITE PRIORITIZATION IN A PLANNING FRAMEWORK

ABSTRACT

Prompted in part by the increasing concern for biodiversity conservation, large-scale inventory and survey efforts, such as Natural Heritage, Breeding Bird Atlas and Breeding Bird Survey databases are fast becoming commonplace. As these species distribution data become more readily available, they are increasingly incorporated into conservation planning efforts. I compared two easily-applied alternate approaches to using avian species data to indicate the value of a site in a conservation planning context as measured by relative site irreplaceability. Focusing on riparian forest conservation in Midwestern landscapes, I calculated the irreplaceability values of 35 native riparian forest stands in central Ohio using avian survey data collected in June of 2001, 2002 and 2003. Specifically, I compared irreplaceability values calculated using avian species richness to the use of a weighting system based on a conservation threat score developed by Partners In Flight, a coalition of agencies and organizations concerned with bird conservation. In addition,

I compared irreplaceability scores created by averaging species occurrence data among multiple years versus a cumulative species richness that considers all species recorded on sites over multiple years. My findings suggest that while the method of survey data compilation had little effect on irreplaceability values among sites, the use of a weighting scheme that places greater emphasis on vulnerable species significantly influenced site prioritization. Use of weighted scoring methods produced portfolios that more consistently differed from randomly-selected sites. Thus, these findings support the common perception that applying species richness data to site prioritization schemes can yield suboptimal results. Planners are encouraged to apply weighting systems that emphasize vulnerable or target species within their planning areas

2.1 INTRODUCTION

Prompted in part by the increasing concern for biodiversity conservation, large-scale inventory and survey efforts, such as Natural Heritage, Breeding Bird Atlas and Breeding Bird Survey databases are fast becoming commonplace. As these species distribution data become more readily available, they are increasingly incorporated into conservation planning efforts. Species data are useful in a planning context because they can provide important indicators of the relative condition of sites being considered for protection and, in that way, can facilitate development of conservation priorities (Noss 1993, Donovan et al. 2002, Bestelmeyer et al. 2003, Groves 2003, Bennett and Milne 2004, Brooks et al. 2004, Pressey 2004, Fleishman et al. 2006, Hess et al. 2006, Pearman et al. 2006, Tchouto et al. 2006). However, species data can be applied to the prioritization process in a variety of ways (e.g. species richness, presence of threatened and endangered species, presence of indicator species), and different approaches can yield widely different results (Dunn et al. 1999, Fleishman et al. 2006, Pearman et al. 2006). For example, use of species richness (total number of species per area) could result in overestimating the value of sites containing large amounts of edge habitat or sites containing large numbers of generalist or even exotic species.

In this paper I compared two easily-applied alternate approaches to using avian species data to indicate site irreplaceability, or "the probability that a potential conservation area will be required as part of a network of conservation areas" (Groves 2003), which is a commonly used index in conservation planning efforts. Specifically, I compared the use of avian species richness to the use of a weighting system based on a conservation threat score developed by Partners In Flight (Regional Combined Score for the Breeding Season) as part of their effort to conserve landbirds in the Western Hemisphere. Their scoring system involves the assignment of scores to individual bird species based on their breeding distributions, regional and global population size, population trends and threats faced with the goal of prioritizing and focusing conservation efforts on species of highest need (Panjabi et al. 2005, Partners in Flight 2005). In addition, I compared irreplaceability scores created by averaging species occurrence data among multiple years versus a cumulative species richness that considers all species recorded on sites over multiple years.

2.2 METHODS

2.2.1 STUDY AREA

Thirty-five mature riparian forest tracts on both public and private land within the Scioto River Watershed of central Ohio (Delaware, Franklin and Pickaway counties) were used in the study (Table 2.1, Figure 2.1). Forests ranged from approximately 50 to 300m in width and were at least 250m in length parallel to the river. Common overstory trees included American hackberry (*Celtis occidentalis*), black walnut (*Juglans nigra*), boxelder (*Acer negundo*), Eastern cottonwood (*Populus deltoides*), honey locust (*Gleditsia tricanthos*), silver maple (*Acer saccharinum*), sugar maple (*Acer saccharum*), sycamore (*Plantanus occidentalis*) and white ash (*Fraxinus americana*). Common woody understory plants included common spicebush (*Lindera benzoin*), honeysuckle (*Lonicera* spp.), multiflora rose (*Rosa multiflora*), Ohio buckeye (*Aesculus octandra*) and tall paw paw (*Asimina triloba*). Rivers were 20 to 40m in width, forests on the side of the river opposite the study sites were at least 10m wide, and study sites were separated by at least 2km. The surrounding landscape matrix ranged from primarily agricultural to very urban (Rodewald and Bakermans 2006).

2.2.2 BIRD SURVEY DATA

Bird species data came from an ongoing study conducted by a research group at The Ohio State University under the direction of Dr. Amanda Rodewald. Bird surveys were conducted three times each year in June 2001, 2002 and 2003 at each site along a 40m wide x 250m long transect located parallel and adjacent to the river's edge. Observed species include residents and short-distance migrants such as the American Robin (*Turdus migratorius*), Northern Cardinal (*Cardinalis cardinalis*), Downy Woodpecker (*Picoides pubescens*), Red-bellied Woodpecker (*Melanerpes carolinus*), Carolina Chickadee (*Poecile carolinensis*), Tufted Titmouse (*Baeolophus bicolor*) and White-breasted Nuthatch (*Sitta carolinensis*), as well as long-distance migrants such as the Acadian Flycatcher (*Empidonax virescens*), Great Crested Flycatcher (*Myiarchus crinitus*), Red-eyed Vireo (*Vireo olivaceus*), Blue-gray Gnatcatcher (*Polioptila caerulea*), and Yellow-throated Warbler (*Dendroica dominica*) (Rodewald and Bakermans 2006).

2.2.3 PARTNERS IN FLIGHT BREEDING SCORES

Partners in Flight (PIF) is a joint venture between a variety of governmental and non-governmental agencies, private industries and philanthropic foundations that is working to conserve landbirds of the Western Hemisphere. With this goal in mind PIF developed a species prioritization process that is based on the assessment of species vulnerability at continental and regional scales (Beissinger et al. 2000, Carter et al. 2000, Panjabi et al. 2005). Factors involving both global and regional population size, population trends, breeding and non-breeding distributions, and threats during the breeding and non-breeding seasons are used to assign global and regional vulnerability scores to all native North American landbirds and well-established non-native species. Each species is assigned a score between 1 (low vulnerability) and 5 (high vulnerability) for each factor and the values are summed together in various combinations to identify species of both continental and regional importance (Panjabi et al. 2005). In this study I used the regional combined score for the breeding season (hereafter referred to as PIF score) which is calculated by summing together the scores for global breeding distribution, global population size, regional population trend, breeding relative density and regional threats to breeding (Appendix C: Table C.1) (Panjabi et al. 2005, Partners in Flight 2005). This combination of factors is believed to provide a good measure of vulnerability of bird species at the regional level (Beissinger et al. 2000).

2.2.4 SCORING TREATMENTS

Four different treatments of the species count data were used to rank all possible conservation portfolios of a given size: “average”, “average weighted”, “cumulative”, and “cumulative weighted”. In both “cumulative” treatments a list of all species observed at least once during the three years of survey data collection was generated for each site. (This list will hereafter be referred to as the cumulative species list).

- In the “cumulative” treatment the score for a given group of sites, or conservation portfolio, was calculated by adding up the number of different species present in that portfolio (species richness) using the cumulative species list.
- In the “cumulative weighted” treatment the cumulative species list was again used, but the species were weighted according to their PIF score. In this case, the score for a given portfolio of sites was calculated by summing the PIF scores of the species present in that portfolio.

In both “average” treatments the survey data from each year were analyzed separately and the results were averaged together.

- In the “average” treatment the number of species present in a given portfolio of sites was calculated using each year of survey data individually. Then, the average number of species present in that portfolio (average species richness) was calculated over the three years.
- In the “average weighted” treatment the PIF breeding scores for the species present in a given portfolio in a given year were summed. The average of these three weighted sums was then calculated to come up with the overall score for a portfolio of sites under the “average weighted” treatment.

2.2.5 IRREPLACEABILITY CALCULATIONS

Conservation portfolios were created to preserve 2, 3, 4, 5, 6, 7, and 8 sites of the possible 35 representing 5.71%, 8.57%, 11.4%, 14.3%, 17.1%, 20.0% and 22.9% of the potential sites. (The maximum portfolio size of 8 sites was chosen due to computational constraints.) For each of these portfolio sizes, site irreplaceability values were calculated directly from the species counts of the survey data using computer code written in C (Metrowerks 2002). Irreplaceability was used as the measure of site importance rather than individual site species richness or diversity as it highlights the potential contribution of a site to overall conservation in an area relative to the other sites under consideration. For planning purposes, the use of an irreplaceability measure focuses efforts on the overall success of a portfolio rather than the quality of individual sites within that portfolio.

The calculation of irreplaceability involved determining the combination, or combinations, of sites producing the highest scores under each of the four scoring treatments. The overall score of a given conservation portfolio does not correspond directly to the scores of the sites making up that portfolio. It is possible that a combination of sites with fewer, yet more threatened species may have a higher overall score than a direct combination of sites with high individual scores. Prior to analysis, I examined this possibility using the cumulative weighted scoring treatment. The score of each individual site was calculated (Table 2.2) and portfolios of each size (2-8) were formed containing the top individually scoring sites. The overall scores of these portfolios were calculated and compared to the maximum possible score for the given portfolio size (Table 2.3). For all portfolio sizes, combining the top individually scoring sites produced an overall score significantly lower than the maximum possible score (11-14% less) suggesting that the direct combination of high scoring sites is a suboptimal method of portfolio formation.

For each portfolio size and scoring treatment, a ranked list was generated listing the portfolios from best to worst. Because multiple portfolios may achieve the highest score, and/or the top several portfolios may differ by only minor amounts, I did not define the top portfolio to be the optimal one. Instead, the "best set" of portfolios was defined to be the top 1%, 5% or 10% of all portfolios in the ranked list.

The irreplaceability value of each site was then defined to be the fraction of all portfolios in the “best set” that contained the given site. (In practice the percentage was not exactly equal to 1%, 5% or 10% as the number of portfolios was increased when the score of the cutoff portfolio was equivalent to that of portfolios lower in the ranked list. In this case the number of portfolios considered in the irreplaceability calculation was increased to include all portfolios with scores equal to the cutoff score.) For each of the seven conservation portfolio sizes (2-8), each of the four scoring treatments (average, average weighted, cumulative, cumulative weighted) was paired with each of the three cutoff percentages (1%, 5%, 10%) to produce 84 different measures of irreplaceability for each site.

2.2.6 STATISTICS

The sign test was used to determine if irreplaceability values calculated using the four scoring treatments were significantly different from those expected from a random ranking of portfolios. (For a random ranking the expected irreplaceability is the portfolio size divided by the number of sites, or 35 for my data.) In addition, differences between the four scoring treatments were evaluated using the sign test. Specifically I compared the average to average weighted, cumulative to cumulative weighted, average to cumulative, and average weighted to cumulative weighted.

The sign test is a nonparametric procedure that tests whether or not one half of the data from a paired data set (X_i,Y_i) is shifted in location relative to the other half. It is calculated by determining the number of observations for which $X_i>Y_i$. If the X_is are not shifted with respect to the Y_is, one expects approximately half of the pairs to satisfy $X_i > Y_i$ (Larsen and Marx 1986). A binomial probability distribution function (with n equal to the number of sites in the comparison and p equal to 0.5) is used to calculate the probability of observing the given number of pairs of irreplaceability values for which $X_i > Y_i$ where X_i and Y_i represent irreplaceability values for the same site calculated using different treatments. The sign test works under the assumption that the paired variables are never equal to one another. When the irreplaceability values of a pair are exactly equal, that comparison is eliminated from the analysis and the sample size (n) is reduced.

2.3 RESULTS

As expected, the irreplaceability value and associated rank of a site was not constant under different scoring treatments, cutoff percentages or portfolio sizes (Appendix A: Tables A.1-A.19). In addition, the use of a scoring system produced irreplaceability values significantly different from a random ranking of sites (at the 0.1 significance level) in approximately 70% of the cases examined (Table 2.4). The weighted scoring treatments were significantly different from a random ranking of sites in 74% of the comparisons, whereas the unweighted treatments were significantly different from a random ranking in only 67% of the comparisons. The cumulative weighted scoring treatment produced results that deviated the greatest amount from randomly generated portfolios for 52% of the comparisons. The average weighted treatment accounted for 38% of the most significant results followed by the average and cumulative treatments with 29% each. (Scoring treatments produced equivalent p-values for some of the comparisons accounting for the failure of these fractions to sum to one.)

The average, average weighted and cumulative weighted scoring treatments produced irreplaceability values significantly different from those produced by randomly ranking portfolios (at the 0.1 significance level) under all cutoff percentages for portfolios containing less than 7 sites (with the exception of cumulative weighted using the 10% cutoff). The cumulative scoring treatment did not perform as predictably and produced irreplaceability values significantly different from a random ranking of portfolios for 60% of the comparisons for portfolios with less than 7 sites (at the 0.1 level). For larger portfolios (containing 7 or 8 sites) and all scoring treatments, no pattern was apparent with 25% of the comparisons yielding significant p-values (at the 0.1 level).

Very few differences were apparent between the two methods of bird survey data compilation (average treatments versus cumulative treatments) as only approximately 5% of these comparisons were significant (Table 2.5). The comparison of weighted to unweighted treatments showed much more variation with approximately 26% of the comparisons having p-values less than 0.1. The weighted and unweighted scoring treatments were most significantly different for portfolios of intermediate size. The average weighted and cumulative weighted scoring treatments did not produce significantly different results for any of the portfolio sizes or cutoff percentages (at the 0.1 level). The average and average weighted treatments were significantly different for approximately 10% of the comparisons, as were the average and cumulative treatments. The cumulative and cumulative weighted scoring treatments were the most significantly different from one another producing p-values less than 0.1 for slightly more than 40% of the comparisons.

2.4 DISCUSSION

My findings suggest that while the method of survey data compilation (i.e. average versus cumulative) had little effect on irreplaceability values among sites, use of a weighting scheme that places greater emphasis on vulnerable species significantly influenced site prioritization. This result is consistent with the perception shared by many ecologists that species richness alone is not an effective indicator of the conservation value of an area (Dunn et al. 1999, Carter et al. 2000, Pressey 2004, Fleishman et al. 2006, Pearman et al. 2006).

While my study highlights the differences between weighted and unweighted scoring schemes, my findings cannot be used to widely generalize the relative worth of a given scoring scheme. Instead, the value of each scoring regime must be determined by evaluating the ecological basis for that regime. In my urbanizing Midwestern study system, urban development tends to increase species richness of resident and short distance migrants while decreasing species richness of Neotropical migratory species that are generally of more conservation concern (Blair 1999, Hennings and Edge 2003, Rodewald and Bakermans 2006). The direct use of a species richness measure in this case may select for more developed sites while neglecting those sites providing crucial habitat for threatened species, suggesting that the use of a weighted measure incorporating information on life history and threats is the preferred option (Fleishman et al. 2006).

Interestingly, despite the strong ecological support for the use of a weighted scoring scheme, differences between irreplaceability scores calculated using weighted versus unweighted schemes was less than I initially expected. The smaller-than-expected effect sizes may result from the statistical method being used. Due to the dependence of the pairs of irreplaceability values in each comparison (values are calculated using the same data set and sum to one), the data set violated the assumption of independence required by most statistical procedures. Although the sign test was useful in this context as it makes no assumption of independence between data pairs, the sign test is limited in its utility as it considers only the direction of change in the irreplaceability value from one scoring treatment to another and does not take into account the magnitude of that change. Therefore, my analysis should be viewed as conservative and likely underestimated differences between scoring schemes.

While the use of species richness information in conservation planning is of value, the use of a species richness measure on its own is not sufficient for characterizing the priority of a site for conservation purposes. On its own, species richness provides no information on the functional role of individual species in ecosystem processes and their ability to respond to stresses in the environment (Fleishman et al. 2006). This study confirms that the direct use of a species richness measure rather than a weighted one in conservation planning may produce very different results and possibly fail to identify sites that are most important to the species-level conservation targets.

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ID#	Field Site	Abbreviation	County	Waterway	Status
1	Bexley Park	bexley	Franklin	Alum Creek	Public
2	Big Walnut Park	bigwal	Franklin	Big Walnut Creek	Public
3	Camp Mary Orton	campmary	Franklin	Olentangy River	Private
4	Casto Park	casto	Franklin	Alum Creek	Public
5	Chapmans Road	chapman	Delaware	Olentangy River	Private
6	Cherrybottom Park	cherry	Franklin	Big Walnut Creek	Public
7	Three Creeks Metro Park	creeks	Franklin	Blacklick Creek	Public
8	Elk Run Park	elkrun	Franklin	Big Walnut Creek	Public
9	Galena	galena	Delaware	Big Walnut Creek	Public
10	Darby Metro Park – Gardner Road	gardner	Franklin	Big Darby River	Public
11	Darby Girl Scout Camp	girlcamp	Franklin	Big Darby River	Private
12	Heisel Park	heisel	Franklin	Big Walnut Creek	Public
13	Highbanks	highbank	Delaware	Olentangy River	Public
14	Innis Park	innis	Franklin	Alum Creek	Public
15	Kilbourn	kilbourn	Delaware	Alum Creek	Public
16	Klondike	klondike	Delaware	Scioto River	Private
17	Lockbourne	lock	Franklin	Big Walnut Creek	Public
18	Lou Berliner Park	lou	Franklin	Scioto River	Private
19	North Galena Road	ngalena	Delaware	Alum Creek	Private
20	North Olentangy Parkland	olentan	Franklin	Olentangy River	Public
21	OSU Wetlands	osuwet	Franklin	Olentangy River	Public
22	Prairie Oaks Metro Park	prairie	Franklin	Big Darby River	Public
23	Prindle Property	prindle	Delaware	Scioto River	Private
24	Darby Public Hunting Access	pubhunt	Franklin	Big Darby River	Public
25	Red Bank Road	redbanks	Delaware	Hoover Reservoir	Public
26	Rocky Creek	rocky	Franklin	Rocky Creek	Private
27	Rush Run	rushrun	Franklin	Olentangy River	Public
28	South Galena Road	sgalena	Delaware	Little Walnut Creek	Public
29	Smith Farm Metro Park	smith	Franklin	Alum Creek	Public
30	Sunbury	sunbury	Delaware	Big Walnut Creek	Private
31	Darby TNC	tnc	Franklin	Big Darby River	Public
32	Westfall	westfall	Pickaway	Scioto River	Private
33	Whetstone	whetston	Franklin	Olentangy River	Public
34	Whitehall Park	whitehal	Franklin	Big Walnut Creek	Public
35	Woodside Green Park	woodside	Franklin	Big Walnut Creek	Public

Table 2.1: Full list of study sites and their attributes.

Site	Average		Average Weighted		Cumulative		Cumulative Weighted	
	Rank	Score	Rank	Score	Rank	Score	Rank	Score
bexley	26	15.67	26	176.33	17	27	20	301
bigwal	10	18.67	17	202.67	21	26	24	288
campmary	24	16.67	24	185	13	28	14	313
casto	27	15.33	30	163.67	31	22	33	236
chapman	4	20.33	5	225.67	4	31	6	347
cherry	31	15	31	158.67	31	22	32	237
creeks	2	21.67	2	246	2	32	3	372
elkrun	21	17	22	191.33	21	26	23	293
galena	10	18.67	13	207	4	31	4	348
gardner	5	20	4	226.67	8	29	11	325
girlcamp	27	15.33	27	174	17	27	18	305
heisel	25	16	25	182	17	27	16	311
highbank	10	18.67	15	203.33	4	31	7	339
innis	32	14	32	154	30	23	30	256
kilbourn	10	18.67	10	211.33	13	28	13	314
klondike	33	13.67	33	153	27	24	27	369
lock	27	15.33	28	168.67	31	22	31	245
lou	21	17	23	188.67	27	24	27	269
ngalena	5	20	3	233	2	32	2	378
olentan	18	17.67	20	195	25	25	25	280
osuwet	3	20.67	6	225.33	8	29	12	320
prairie	18	17.67	18	201.67	21	26	20	301
prindle	8	19.33	7	220.67	8	29	9	333
pubhunt	1	24.33	1	281	1	34	1	396
redbanks	35	11.67	35	125	34	20	35	218
rocky	34	13.33	34	142.33	34	20	34	219
rushrun	10	18.67	12	210	13	28	16	311
sgalena	16	18.33	14	204.33	17	27	19	304
smith	7	19.67	8	220	4	31	4	348
sunbury	16	18.33	15	203.33	13	28	14	313
tnc	21	17	19	196.67	21	26	20	301
westfall	10	18.67	11	210.67	8	29	9	333
whetston	18	17.67	20	195	27	24	29	264
whitehal	27	15.33	29	166	25	25	26	270
woodside	8	19.33	9	218	8	29	8	335

Table 2.2: Site ranks and scores under the four different scoring treatments calculated for the individual sites based on species survey data collected in June 2001, 2002 and 2003 in central Ohio.

Portfolio Size	1	2	3	4	5	6	7	8
Sites	pubhunt	ngalena	creeks	creeks	creeks	chapman	chapman	chapman
		pubhunt	ngalena	galena	galena	creeks	creeks	creeks
			pubhunt	ngalena	ngalena	galena	galena	galena
				pubhunt	pubhunt	ngalena	highbank	highbank
					smith	pubhunt	ngalena	ngalena
				creeks		smith	pubhunt	pubhunt
				ngalena			smith	smith
				pubhunt				woodside
				smith				
Maximum possible score for the portfolio size	396	485	524	554	577	596	610	624
Average score for the listed portfolios	396	432	463	487.5	501	512	531	544
Average rank for the listed portfolios	1 out of 35	47 out of 595	782 out of 6,545	7,529 out of 52,360	50,473 out of 324,632	317,474 out of 1,623,160	>500,000 out of 6,724,520	>500,000 out of 23,535,820

Table 2.3: Scores for portfolios formed by taking the top scoring sites using the cumulative weighted scoring scheme (Table 2.2). Two portfolios are listed when sites shared the same score.

Portfolio Size	Expected Value	Percentage	Average	Average Weighted	Cumulative	Cumulative Weighted
2	0.05714	1%	**<0.001**	**<0.001**	**0.002**	**<0.001**
			7	7	8	6
		5%	**0.09**	**0.006**	**0.09**	**0.017**
			12	9	12	10
		10%	**0.041**	**0.041**	0.176	**0.006**
			11	11	13	9
3	0.08571	1%	**0.017**	**0.006**	**0.002**	**<0.001**
			10	9	8	7
		5%	**0.017**	**0.017**	**0.017**	**0.006**
			10	10	10	9
		10%	**0.041**	**0.017**	**0.017**	**0.006**
			11	10	10	9
4	0.1143	1%	**0.041**	**0.09**	**0.017**	**0.017**
			11	12	10	10
		5%	**0.09**	**0.09**	0.176	**0.017**
			12	12	13	10
		10%	**0.09**	**0.09**	0.176	**0.017**
			12	12	13	10
5	0.1429	1%	**0.09**	**0.09**	**0.041**	**0.09**
			12	12	11	12
		5%	**0.09**	**0.09**	0.176	**0.041**
			12	12	13	11
		10%	**0.09**	**0.09**	0.176	**0.09**
			12	12	13	12
6	0.1714	1%	**0.09**	**0.09**	**0.006**	**0.041**
			12	12	9	11
		5%	**0.09**	**0.09**	**0.09**	**0.09**
			12	12	12	12
		10%	**0.09**	**0.09**	0.311	0.311
			12	12	14	14

Continued

Table 2.4: P-values (above) and test statistics (below) for the Sign Test comparing observed irreplaceability values to the expected value for a random ranking of portfolios. Values in bold are significant at the 0.1 level (n=35 for all comparisons).

Table 2.4 continued

7	0.2000	1%	0.176	0.176	**0.09**	0.176
			13	13	12	13
		5%	0.176	0.176	**0.09**	0.176
			13	13	12	13
		10%	0.176	0.176	0.311	0.5
			13	13	14	15
8	0.2286	1%	**0.041**	0.176	**0.09**	0.176
			11	13	12	13
		5%	0.176	**0.09**	0.176	0.176
			13	12	13	13
		10%	0.176	**0.09**	0.311	0.311
			13	12	14	14

Portfolio Size	Percentage	Average & Average Weighted	Cumulative & Cumulative Weighted	Average & Cumulative	Average Weighted & Cumulative Weighted
2	1%	-----	1	0.508	1
			4 (n=8)	3 (n=9)	3 (n=7)
	5%	0.503	0.845	0.442	1
		8 (n=20)	14 (n=26)	11 (n=27)	9 (n=19)
	10%	0.585	0.281	0.487	0.377
		13 (n=30)	19 (n=31)	14 (n=33)	19 (n=32)
3	1%	0.839	0.851	0.362	0.572
		13 (n=24)	15 (n=28)	12 (n=30)	12 (n=28)
	5%	0.176	**0.002**	**0.09**	0.311
		22	27	12	14
	10%	0.311	0.851	0.736	0.311
		21	15 (n=28)	16	14
4	1%	0.311	0.736	0.311	0.392
		14	16	14	14 (n=34)
	5%	0.121	**0.002**	0.311	1
		12 (n=34)	27	14	17
	10%	0.736	**<0.001**	0.23	0.736
		16	28	13 (n=34)	16
5	1%	0.392	**0.017**	0.736	0.311
		14 (n=34)	25	16	21
	5%	0.176	**0.09**	1	1
		13	23	17	18
	10%	0.864	**0.041**	1	0.736
		18 (n=34)	24	17	19

Continued

Table 2.5: P-values (above) and test statistics (below) for the Sign Test comparing irreplaceability values calculated using the four scoring treatments. Values in bold are significant at the 0.1 level (n=35 unless otherwise indicated). (Irreplaceability values for the average and average weighted treatments of portfolio size 2 at 1% were identical.)

Table 2.5 continued

6	1%	**0.041**	1	0.736	1
		11	18	19	18
	5%	1	**0.002**	1	0.5
		17	27	17	20
	10%	0.5	**0.006**	0.736	0.5
		20	26	16	20
7	1%	**0.09**	0.176	**0.09**	0.5
		12	22	12	20
	5%	1	1	0.5	0.5
		18	18	15	20
	10%	0.176	**0.09**	0.736	0.736
		22	23	16	19
8	1%	0.736	0.311	1	1
		19	21	17	18
	5%	0.5	0.736	0.736	1
		15	19	16	18
	10%	0.5	1	0.5	0.5
		20	18	15	15

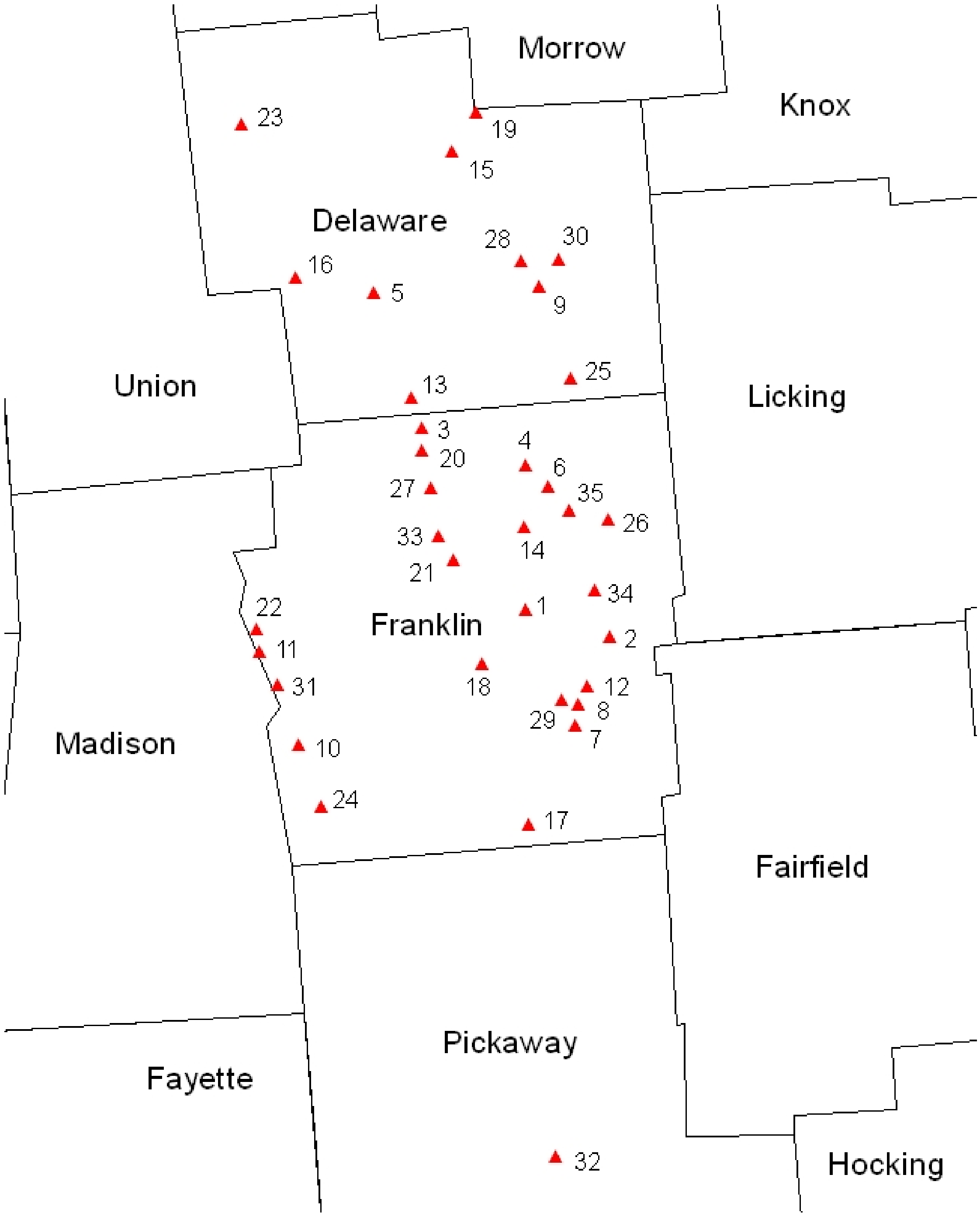

Figure 2.1: Map of the study sites in central Ohio, USA. Identification numbers refer to those listed in Table 2.1.

CHAPTER 3

TESTING THE UTILITY OF LANDSCAPE METRICS FOR CONSERVATION PLANNING

ABSTRACT

A variety of well-conceived conservation planning approaches have been developed over recent years, all with the critical component of setting conservation priorities for both species protection and land acquisition and management. Ideally, the selection of the sites comprising a conservation portfolio would be based on detailed survey data, but this is not always possible due to time and monetary constraints. When it is not possible or feasible to conduct surveys of even some portion of the species in an area, basic ecological principles can be applied. This study examined the extent to which commonly used and easily obtained landscape metrics contribute to a conservation planning process that must be completed rapidly and with little financial resources. Specifically, I aimed to determine if landscape metrics could be used to guide the site prioritization process when the relative importance of potential conservation sites to various conservation portfolios is measured using their relative irreplaceability values. This question was examined in the context of a

flexible planning process where the scope of a given project (represented by conservation portfolio size) may vary over time. I focused this study on the preservation of riparian forest land for bird species in urbanizing Midwestern landscapes. I examined 35 riparian forest sites along an urban-to-rural gradient in central Ohio and used avian survey data collected in June of 2001, 2002 and 2003 to calculate an irreplaceability score for each site. Landscape composition within a 1-km radius area centered on each site was evaluated by calculating the number of buildings and the percent forest cover, agricultural land, roadways, pavement and mowed surfaces. Multinomial logistic regression models were created for 21 different models based on ecological principles and compared using Akaike Information Criterion. Percent cover by forest most consistently appeared in the best ranked models over portfolio sizes ranging from 2 to 8 sites. Also, all of the disturbance metrics were frequently included within the top suite of models. Results suggest that when predicting the irreplaceability value of a site to a conservation portfolio, a simple yet effective method involves the use of forest coverage at the landscape level plus one metric to measure human disturbance on the landscape (number of buildings, area of roads, pavement, mowed surfaces, agricultural land). This result appears to hold across a range of portfolio sizes and is therefore likely to remain useful even in the context of a planning process whose scope varies over time.

3.1 INTRODUCTION

A variety of well-conceived conservation planning approaches have been developed over recent years (Groves 2003, Beazley et al. 2005, Borges et al. 2005, Haight et al. 2005, Shi et al. 2005, Burgess et al. 2006, Cabeza and Moilanen 2006, Freemark et al. 2006, Knight et al. 2006, Rouget et al. 2006, Turner and Wilcove 2006), all with the critical component of setting conservation priorities for both species protection and land acquisition and management. Prioritization is a necessity within every planning framework because time and financial constraints generally prevent the safe-guarding of all potentially important lands (Brooks et al. 2006, Cabeza and Moilanen 2006, Wilson et al. 2006). Planners typically work to select a conservation portfolio (Groves 2003, Burgess et al. 2006), or a group of sites that cover the full range of conservation targets, that can help guide future actions. Theoretically, the design of a conservation portfolio is much preferred over the selection of a single site as it allows the planner to protect a larger variety and number of species and habitats. However, in practice, the job of selecting a portfolio of sites can become quite complicated, especially when the number of potential sites is large. When given a collection of potential conservation areas and the task of selecting a subset of them for conservation, an intuitively attractive approach is to simply rank the sites according to their individual species diversity, species richness or other applicable criteria and then choose the top sites from the list to put into a portfolio. However, this approach overlooks the importance of three important principles for setting land priorities: complementarity, flexibility, and irreplaceability (Pressey et al. 1993, Groves 2003, Borges et al. 2005, Turner and Wilcove 2006, Williams et al. 2006). Complementarity results from an iterative process that first examines existing conservation sites and then identifies which new conservation sites would contribute the greatest value (in terms of diversity, processes, or other conservation targets) not already represented in the existing sites (Groves 2003). Thus, failing to employ the principle of complementarity can result in the duplication of some conservation targets with the neglect of others. Portfolios designed with flexibility in mind provide multiple options for planners, allowing them to adaptively change land acquisition strategies in response to changing project scope, constraints, or opportunities (Groves 2003, Haight et al. 2005, Cabeza and Moilanen 2006, Turner and Wilcove 2006). Finally, by incorporating the principle of irreplaceability, a planner can identify in an empirical and explicit fashion the probability that any given site is contained within the portfolio of sites required to reach conservation goals. Therefore in practice, planners must consider a wide array of potential conservation portfolios and compare the overall coverage of each before making planning decisions (Groves 2003).

Ideally, the selection of the sites comprising a conservation portfolio would be based on detailed surveys consisting of presence/absence data, demography and life histories of species, interactions between species, and interactions of species with their environment (Brooks et al. 2004, Burgess et al. 2006). However, due to time and monetary constraints, decisions are usually based on more limited data. When it is not possible or feasible to conduct surveys of even some portion of the species in an area, basic ecological principles can be applied (Meffe and Carroll 1997, Wiersma et al. 2004, Hess et al. 2006). For example, measures reflecting patch size (area) and landscape composition can predict species diversity for certain taxa and for some sensitive species of conservation importance (Saveraid et al. 2001, Lombard et al.

2003, Wiersma et al. 2004, Hess et al. 2006). Research suggests that forest width, the quality of the habitat within a forested area, and the land use and cover type of areas adjacent to a forested area are the dominant factors determining the overall value of a forested area as wildlife habitat (Friesen et al. 1995, Mensing et al. 1998, Rottenborn 1999, Hennings and Edge 2003, Miller et al. 2003, Wiersma et al. 2004, Mason 2006, Rodewald and Bakermans 2006). Though these landscape-scale surrogates have been widely applied in the literature (Johnson 1995, Cook 2002, Livingston et al. 2003, Kati et al. 2004, Beazley et al. 2005), they have typically been used to identify and/or rank the importance of individual sites without considering irreplaceability.

The objective of this study was to examine the extent to which commonly-used and easily obtained landscape metrics could contribute to a site prioritization process that considers irreplaceability and flexibility. Specifically, I aimed to (1) determine if landscape metrics can effectively guide the development of a conservation portfolio by predicting the relative importance of potential conservation sites, as measured by irreplaceability scores, and (2) evaluate flexibility by examining this question for a range of conservation portfolio sizes to determine the utility of landscape metrics in a flexible planning process where the scope of a given project may vary over time due to changes in funding and/or timelines.

I focused this study on mature riparian forest areas along an urban-to-rural gradient. These areas are of interest in a planning context as over 90% of riparian habitats have been lost during the past 200 years due to disturbances caused by water management practices, agriculture, grazing, channelization, timber removal, industry, mining, urbanization and recreation (Knopf et al. 1988, Malanson 1993, Jobin et al. 2004). Governmental agencies and conservation organizations frequently invest large amounts of resources in the acquisition, retention and management of riparian forest areas due to their high ecological, social and recreational value. Riparian forests provide habitat for a variety of species, protect water quality, perform critical functions in hydrological and biogeochemical cycles, and are frequently chosen as the location for parks, bike paths and greenways. Birds were used as a focal group because their habitat requirements are often specialized, they are abundant, relatively easy to survey, and sensitive to environmental change and human disturbance at local and landscape scales. Indeed others have found birds to be frequently useful and effective indicators of forest habitat quality (Brooker 2002, Mason 2006, Miller et al. 2003, Rodewald and Bakermans 2006).

3.2 METHODS

3.2.1 STUDY AREA

Thirty-five mature riparian forest tracts on both public and private land within the Scioto River Watershed of central Ohio (Delaware, Franklin and Pickaway counties) were used in the study (Table 2.1, Figure 2.1). Forests ranged from approximately 50 to 300m in width and were at least 250m in length parallel to the river. Common overstory trees included American hackberry (*Celtis occidentalis*), black walnut (*Juglans nigra*), boxelder (*Acer negundo*), Eastern cottonwood (*Populus deltoides*), honey locust (*Gleditsia tricanthos*), silver maple (*Acer saccharinum*), sugar maple (*Acer saccharum*), sycamore (*Plantanus occidentalis*) and white ash (*Fraxinus* *americana*). Common woody understory plants included common spicebush (*Lindera benzoin*), honeysuckle (*Lonicera* spp.), multiflora rose (*Rosa multiflora*), Ohio buckeye (*Aesculus octandra*) and tall paw paw (*Asimina triloba*). Rivers were 20 to 40m in width, forests on the side of the river opposite the study sites were at least 10m wide, and study sites were separated by at least 2km. The surrounding landscape ranged from agriculturally dominated to nearly urbanized matrices (Rodewald and Bakermans 2006).

3.2.2 BIRD SURVEY DATA

Bird species data came from an ongoing study conducted by a research group at The Ohio State University under the direction of Dr. Amanda Rodewald. Bird surveys were conducted three times each year in June 2001, 2002 and 2003 at each site along a 40m wide x 250m long transect located parallel and adjacent to the river's edge. Observed species included residents and short-distance migrants such as the American Robin (*Turdus migratorius*), Northern Cardinal (*Cardinalis cardinalis*), Downy Woodpecker (*Picoides pubescens*), Red-bellied Woodpecker (*Melanerpes carolinus*), Carolina Chickadee (*Poecile carolinensis*), Tufted Titmouse (*Baeolophus bicolor*) and White-breasted Nuthatch (*Sitta carolinensis*), as well as long-distance migrants such as the Acadian Flycatcher (*Empidonax virescens*), Great Crested Flycatcher (*Myiarchus crinitus*), Red-eyed Vireo (*Vireo olivaceus*), Blue-gray Gnatcatcher (*Polioptila caerulea*), and Yellow-throated Warbler (*Dendroica dominica*) (Rodewald and Bakermans 2006).

3.2.3 SITE IRREPLACEABILITY

Conservation portfolios were created to preserve 2, 3, 4, 5, 6, 7, and 8 sites of the possible 35 representing 5.71%, 8.57%, 11.4%, 14.3%, 17.1%, 20.0% and 22.9% of the potential sites. (The maximum portfolio size of 8 sites was chosen due to computational constraints.) For each of these portfolio sizes, site irreplaceability values were calculated directly from the species counts of the survey data using computer code written in C (Metrowerks 2002). I chose to use irreplaceability, which is defined as "the probability that a potential conservation area will be required as part of a network of conservation areas" (Groves 2003), as my measure of site importance. Irreplaceability was used rather than individual site species richness or diversity as it highlights the potential contribution of a site to overall conservation in an area relative to the other sites under consideration. For planning purposes, the use of an irreplaceability measure focuses efforts on the overall success of a portfolio rather than the quality of individual sites within that portfolio.

For this study, site irreplaceability was calculated using the cumulative bird survey data from the years 2001-2003 (i.e. creating species lists by pooling data among all three years) and weighting the value of each species according to its Partners in Flight Regional Combined Score for the breeding season (Partners in Flight 2005) as per the methods outlined in chapter 2 (Appendix C: Table C.1). The total score of each portfolio of sites was determined by summing the scores of the species present in that portfolio and portfolios were ranked from best to worst. The "best set" of portfolios was defined as the top 1% of all portfolios in the ranked list. (Due to the small number of portfolios of size 2, the top 10% of all portfolios was used for this portfolio size.) The irreplaceability value of each site was then defined to be the fraction of all portfolios in the "best set" that contained the given site.

Prior to analysis, I examined the consistency of irreplaceability scores across the range of portfolio sites used in subsequent analyses (Fig. 3.1). Plots of normalized irreplaceability values showed that values were relatively consistent for most (~77%) sites across portfolio sizes ranging from 2-8 sites. However, there was a tendency for irreplaceability values to decrease as the size of the conservation portfolio increased, which may be expected given that the number of ways to combine sites into equivalent portfolios increases with increasing portfolio size. In other words, the possibility of substituting one site for another without changing the value of a given portfolio increases with increasing portfolio size causing the relative values of the sites to become more equivalent. One cautionary note resulting from this examination is that both the actual and relative irreplaceability values of the sites are not constant over changes in portfolio size. Therefore it is possible that the best landscape metric or metrics for predicting irreplaceability may change with changes in the total size of the conservation portfolio.

3.2.4 LANDSCAPE METRICS

Landscape data were either obtained by the digitization and subsequent analysis of aerial photos (purchased from county auditor's offices) using ArcGIS 9.1 (ESRI 2005), or obtained directly from pre-existing digital data provided by the county auditors' offices. Photographs were taken in 2004 for Franklin county, 2002 for Delaware county, 2004 for Madison county and 1994 for Pickaway county. The early data for Pickaway county are not believed to create a problem as the county contains only one site used in the study (westfall) and visits to the area during surveys confirmed that the landscape remained very rural.

Landscape metrics representing patch characteristics, forest availability, and cover by common land uses were quantified within a 1-km radius circle centered on the study site (Tables 3.1 and B.1). A 1-km radius scale was used as this scale has been shown to be strongly associated with bird communities in other studies (Rodewald and Bakermans 2006). Average forest width along the survey transect (as described in Rodewald and Bakermans 2006) was used as a patch characteristic metric rather than patch size because riparian forests in the study system do not occur as discrete patches and, rather, tend to be continuous along waterways simply contracting or expanding in width. Landscape-level habitat was quantified as the fraction of forest within a 1-km radius around the site center. Five disturbance metrics were used including agriculture, buildings, mowed, roads and pavement. The agriculture metric described the percentage of the 1-km radius landscape covered by agricultural land, including row crops, fallow fields, and pasture. The buildings metric was a count of the number of buildings in the 1-km radius area. The mowed metric was the fraction of the landscape covered by mowed areas including both large areas such as parks and golf courses as well as smaller residential lawns. Data for the road metric were obtained from digital data provided by the county auditor and was recorded as the fraction of the landscape covered by roadways, exclusive of driveways within apartment or business complexes, town parks and other low use roadways. The pavement metric described all paved surfaces in the given area combining the roads metric with roadways not included by the roads metric (described previously) and parking lots.

The correlation between the landscape variables was assessed using Pearson's Correlation Coefficient (Table 3.2). Despite high correlations between some of the landscape variables, all of the metrics were used in the analysis as they all are believed to be significant contributors to the overall quality of a habitat and the intent was to identify the most useful metric to explain variation in irreplaceability values among sites. Possible effects of the high correlations on my analysis and results are included in the discussion.

3.2.5 DATA ANALYSIS

Multinomial logistic regression models were constructed in R (R Development Core Team 2006) using the irreplaceability value of each site as a response variable. (Linear regression could not be used in my analysis as the normalized irreplaceability values were essentially probabilities and must all sum to one.) To avoid data dredging by testing all possible combinations of potentially important variables, 21 a priori models were constructed based on ecological principles (Table 3.3). In order to fit the multinomial distribution for each model, I assumed that the sites (numbered 1 to 35) were the potential outcomes for a given number of independent trials (approximately 10,000). In the multinomial distribution there is a probability value associated with each state (1 to 35) that describes the probability of that state occurring. These probabilities are the irreplaceability values once they have been normalized to sum to one. The landscape metrics were used as predictor variables for the 34 probabilities of the multinomial distribution (as the sum of the probabilities must equal one, one probability can be dropped from the analysis). Hence, adding one landscape predictor variable to a model adds up to 34 parameters to that model. (In the analysis with smaller portfolio sizes some sites were dropped as they had irreplaceability values of zero. In these cases the number of parameters was less than 34.)

Akaike Information Criterion (AIC_c) was used to select the best models from the 21 hypothesized ones. AIC is an information-theoretic approach that emphasizes a focus on a priori science in developing a set of hypotheses or models. Models are ranked through a consideration of the trade-offs occurring when the number of variables in a model is increased (increasing fit versus the addition of artificial complexity). The ranked models are then assigned a weighting representing the strength of evidence in favor of that model as compared to other models in the set. AIC_c is a variant on the original AIC statistic that includes a term to correct for small sample sizes (Burnham and Anderson 2002).

For each model I calculated the total number of parameters (K), the natural logarithm of the likelihood function ($\log(L(\theta))$), Akaike Information Criterion (AIC_c), distance (Δ_i) and weight (w_i). Distance is calculated as the difference between the model AIC_c and the minimum AIC_c value for the entire set of models. As Δ_i increases, the likelihood of that model being the best model in the set decreases.

Models for which Δ_i<2 are considered to have substantial support and should all be considered as potential approximating models for the data. Akaike weight, w_i indicates the likelihood that model *i* is the actual best model under the assumption that the best model is in fact in the set of models under consideration. I also calculated the sum of the Akaike weights (Σw_i) for each individual landscape metric, the average weight over all models containing the given landscape metric, and the average weight over all models with Δ_i<2 containing the given landscape metric. The sum and averages give insight into the relative importance of each landscape metric as compared to the other metrics in the set of models being compared (Burnham and Anderson 2002).

3.3 RESULTS

Across the full range of portfolio sizes, the most highly ranked models tended to include both a measure of habitat availability (i.e. forest width or forest cover within 1-km) and disturbance (e.g. cover by roads, number of buildings) (Table 3.4, Appendix B: Tables B.2a-B.8a). For portfolio sizes of 2, 4, 6 and 7 the landscape level habitat model (Forest in 1-km) was not only the best model, but was also the only model in the set with a substantial likelihood of being the true model. For portfolio sizes of 3, 5 and 8 the area plus disturbance models with one disturbance metric and the high intensity urban disturbance model (all models incorporating two landscape metrics) had substantial support (Δ_i was essentially equal to zero). Akaike weights for these models were equal suggesting that all are potential best models for the data. The patch characteristic model (Forest Width) was not given any weight as a potential best model for any of the portfolio sizes evaluated. Likewise none of the models considering only one disturbance metric were given any weight. Lastly, the null model (no landscape metrics) and all models containing more than two landscape metrics were assigned weights essentially equal to zero in all cases.

For the portfolio sizes with overwhelming support for the landscape level habitat metric (portfolio sizes 2, 4, 6 and 7) the sum of the weights was 1 for Forest in 1-km and 0 for the remaining metrics (Appendix B: Tables B.2b, B.4b, B.6b and B.7b). The average weight over all models and the best models was 0.125 and 0.2 for Forest in 1-km respectively and 0 for the remaining metrics. Looking at the portfolio sizes with multiple potential best models (portfolio sizes 3, 5 and 8), the sum of the weights was 0.5 for Forest in 1-km and Forest Width, 0.3 for Buildings, 0.2 for Roads, Mowed and Agriculture, and 0.1 for Paved (Appendix B: Tables B.3b, B.5b and B.8b). The average weight over all models was 0.0625 for the two forest metrics, 0.0375 for Buildings, 0.0667 for Roads, 0.04 for Mowed and Agriculture, and 0.02 for Paved. The average weight over the best models was 0.1 for all landscape metrics. This suggests that of the disturbance metrics considered in the analysis, Roads may be the most important followed by Agriculture and Mowed, then Buildings and then Paved.

3.4 DISCUSSION

Recent advances in landscape and spatial analyses have resulted in numerous sophisticated approaches to characterizing landscape composition and structures. However, many of these analytical techniques require software, datasets or competencies that are frequently not available to organizers engaged in conservation planning efforts. These findings suggest that relatively simple, widely available landscape metrics can be useful in planning frameworks. In particular, a simple yet effective method for predicting the irreplaceability value of a site to a conservation portfolio was the use of forest coverage at the landscape level plus one metric to measure human disturbance on the landscape. Among the disturbance metrics, the area of roadways performed best, but use of the most readily available measure of disturbance (number of buildings, area of agricultural land, mowed land or paved surfaces) should produce a reasonable result as well. The effectiveness of the forest plus disturbance metrics appears to hold across a range of portfolio sizes and is therefore likely to remain useful even in the context of a planning process whose scope varies over time.

These results are consistent with findings from several other studies that use landscape features in conservation planning. The use of satellite imagery to identify large areas of natural habitat associated with the species or group of concern was found to be a useful indicator of site quality as measured by species occurrence (Poiani et al. 2001, Saveraid et al. 2001, Lombard et al. 2003, Porej et al. 2004, Wiersma et al. 2004, Hess et al. 2006). In addition, landscape-scale habitat information was found to perform more effectively when used in conjunction with additional landscape metrics (Saveraid et al. 2001, Porej et al. 2004, Wiersma et al.

2004), especially those incorporating some measure of human influence on the surrounding landscape, such as roads, trails, railways, campgrounds, golf courses, or buildings (Porej et al. 2004, Wiersma et al. 2004).

Previous research confirms that the metrics identified in this study (landscape level forest coverage and human disturbance) are dominant factors determining the overall value of a forested area as wildlife habitat (Friesen et al. 1995, Mensing et al. 1998, Rottenborn 1999, Hennings and Edge 2003, Miller et al. 2003, Wiersma et al. 2004, Mason 2006, Rodewald and Bakermans 2006). In the same study system Rodewald and Bakermans (2006) showed that diversity and abundance of Neotropical migrants in riparian forests decreased within increasing amounts of impervious surface within landscapes. Similarly, Rottenborn (1999) found that as the distance to the nearest building and the width of the riparian habitat increased, bird species richness also increased. Mason et al. (2006) found that bird species richness and abundance in greenways decreased as the percent pavement cover increased and as the amount of mowed or maintained surfaces increased. In a study conducted on riparian areas in Colorado Miller et al. (2003) found that human settlement metrics, particularly building density in the vicinity of a site, accounted for a significant amount of the observed variation in habitat use by birds. Friesen et al. (1995) found that the abundance and number of Neotropical migratory songbirds increased with forest size, but was more influenced by the number of houses within 100m of the forest edge.

Finally, in their study of riparian forest areas near Portland, Oregon, Hennings and Edge (2003) found that variables describing forest width and road density together explained a significant amount of the observed variation in bird community structure.

Although this study emphasizes the importance of only a few simple landscape features to riparian forest systems, avian communities are certainly affected by complex interactions among numerous local and landscape-level ecological attributes. The goal of this study was not to find a comprehensive list of landscape variables to explain and predict the exact habitat needs of bird species utilizing riparian areas in and around urban areas. Instead I aimed to identify easily acquired landscape metrics that could be used in a planning framework to rapidly, inexpensively and adequately identify the most important sites for conservation from an initial list of potential sites. This was not an unreasonable goal as relatively simple rules for deciding which areas to protect have been shown to out perform both comprehensive and random conservation planning procedures in areas with high rates of land degradation, uncertainty about when and where conservation opportunities may arise, and varying budgetary constraints (Meir et al. 2004). These circumstances are exactly those often faced by conservation planners in urban settings.

Although the landscape metrics identified as important in my study may prove useful to planners, there are several important limitations of this study that should be considered as caveats. First, all sites used in this study were chosen because they were believed to potentially offer habitat suitable to forest-dependent bird species (forests greater than 50m in width and 250m in length). Therefore my conclusions are predicated on the assumption that planners are starting the planning process with a reasonable list of potential sites. Second, as mentioned in section 3.2.5, the use of multinomial logistic regression models resulted in an unusually large penalty for each additional model term. In fact, the number of parameters being fit increased by as much as 34 for each additional landscape metric included in a given model. Even so, the log likelihood of the two landscape variable models is quite small suggesting that these models are good fits to the data irregardless of the relative number of parameters. Third, another potential reason for the lack of support for models with more than two landscape variables involves the high correlation of the landscape metrics (Table 3.2). Many of the variables, especially among disturbance metrics, were highly correlated, a pattern that is not surprising given the suite of ecological and land use changes that co-occur with disturbance (e.g. greater road cover with greater numbers of buildings).

3.5 PLANNING PERSPECTIVES AND RECOMMENDATIONS

The results of my study are applicable in a variety of planning contexts. One example is the selection of land for enrollment in land retirement programs such as the Conservation Reserve Enhancement Program (CREP, USDA 2003) or conservation easement and acquisition programs administered by private land trust organizations such as the Land Trust Alliance (LTA 2006) and The Nature Conservancy (TNC 2006). These programs usually involve a commitment by a landowner to keep land out of production for a given time period with the goal of protecting environmentally sensitive land, decreasing erosion, restoring wildlife habitat and safeguarding ground and surface water. While landowners may be interested in enrollment in programs like these, they may not have the time or money to spend on the selection of the best possible parcel of land to enroll in the program. Likewise, planners working within the agencies administering these programs may not have the monetary resources or time to conduct extensive studies and determine the areas within which to focus their outreach efforts. The results presented here suggest that a rapid analysis of forest cover and one disturbance metric should provide a reasonable framework from which to base decisions.

Another useful application occurs in a city or regional planning context when monetary resources are in a state of flux over time. A city planner may be tasked with identifying potential parkland throughout a city and wishes to select areas that are not only aesthetically pleasing, but also offer the best chance of biodiversity conservation. In this instance the project may originate with the goal of choosing one or two sites and then later, as more funding becomes available, grow to encompass a much larger number of areas. Similarly a project could begin with an optimistic goal of creating a large park network and then be forced to scale back due to budgetary concerns. In either instance the use of a landscape scale forest metric in conjunction with a measure of disturbance could be successfully used despite changes in the project scope.

While these findings most clearly inform site prioritization in the context of land acquisition, they also may be useful to guide land management efforts when resources are limited. In these situations, a land manager working for a government agency already controlling large land areas can use the metrics to identify specific sites that hold the most ecological value and then focus enhancement and/or restoration activities at those locations.

As digital data describing either the area or length of roadways are often readily available from county governments, percent coverage by roads may be the most time-efficient and effective disturbance metric to use in many instances. Although I used road area in my analysis, length of roadway is expected to be highly correlated with road area and, therefore, an appropriate surrogate for most regions.

In this paper I have studied the potential utility of easily attainable landscape metrics for the prediction of site irreplaceability in a conservation planning framework. While this work suggests that the use of two metrics, one describing forest cover and the other a disturbance factor at the landscape level, should be sufficient when planning decisions must be made promptly and economically, I do not seek to minimize the value of a thoughtful and systematic prioritization and planning process for conservation. Instead, I offer a practical method for land planners making rapid decisions on limited budgets that can and should be followed by a more rigorous planning process if and when more resources eventually become available.

3.6 LITERATURE CITED

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Variable	Description
Forest Width	Average width of the forested area along the survey transect (m)
Forest in 1-km	Fraction of a 1-km radius area around the site center that contains forest cover
Agriculture	Fraction of a 1-km radius area around the site center that contains agricultural/grazing land
Buildings	Number of buildings within a 1-km radius area of the site center
Mowed	Fraction of a 1-km radius area around the site center that contains mowed surfaces (parks, golf courses, residential lawns, etc.)
Paved	Fraction of a 1-km radius area around the site center covered by pavement (includes all roadways and parking lots)
Roads	Fraction of a 1-km radius area around the site center covered by roadways (roads within apartment or business complexes, town parks etc. are not included)

Table 3.1: Landscape variables used in the analysis.

	Forest in 1-km	Agriculture	Buildings	Mowed	Paved	Roads
Forest Width	0.282	-0.124	-0.141	-0.055	-0.144	-0.1
Forest in 1-km		-0.042	-0.597	-0.424	-0.612	-0.592
Agriculture			-0.611	-0.79	-0.634	-0.643
Buildings				0.732	0.856	0.859
Mowed					0.684	0.804
Paved						0.861

Table 3.2: Correlation coefficient matrix for landscape variables across 35 riparian forest sites in central Ohio.

Hypothesis	Model	Ecological Rationale
Patch characteristics (area & edge)	Forest Width	The wider the tract of riparian forest, the greater the ecological value of the site (in terms of species diversity, ecological function etc.)
Landscape level habitat	Forest in 1-km	Forested landscapes reduce area and edge effects
Area plus habitat in the landscape	Forest in 1-km + Forest Width	Species richness and presence of sensitive species should be positively related to the amount of habitat available within the landscape
Agricultural disturbance	Agriculture	Negative impacts through contamination (pesticides/pollutants), habitat alteration, and increases in the number of generalist species (including nest predators and brown-headed cowbirds)
Roadway disturbance	Roads	Negative impacts through the bisection and fragmentation of forests, increasing sedimentation in river and riparian habitat, and increasing wildlife mortality. Positively correlated with other urban metrics.
Low-intensity urban disturbance	Mowed	Negative impacts due to pesticides, non-habitat, and the frequent visitation of residential areas by generalist predators
Human use	Buildings	Negative impacts due to increased direct human disturbance and association with residential areas frequented by generalist predators
High-intensity urban disturbance	Buildings + Paved	Reasons given above for buildings and roads

Continued

Table 3.3: Models used in the analysis and the ecological rationale behind them.

Table 3.3 continued

All disturbance	Agriculture + Buildings + Mowed + Paved	Reasons given above for all disturbance variables
Area plus disturbances	Forest Width + Agriculture	Positive impacts of patch-level characteristics are often mediated by land uses within the surrounding landscape
	Forest in 1-km + Agriculture	
	Forest Width + Road	
	Forest in 1-km + Road	
	Forest Width + Mowed	
	Forest in 1-km + Mowed	
	Forest Width + Buildings	
	Forest in 1-km + Buildings	
	Forest Width + Buildings + Paved	
	Forest in 1-km + Buildings + Paved	

Continued

Table 3.3 continued

Full model	Forest Width + Forest in 1-km + Agriculture + Mowed + Buildings + Paved	Reasons given above for area plus disturbance models
Null model	No variables	The null model both assesses the value of simple estimation of the mean, and differences between the null model and all subsequent models will identify the degree to which our understanding is improved by covariates.

Portfolio Size	Model	K	-2log(L(θ))	AIC_c	Δ_i	w_i
2	Forest in 1-km	50	15.455	115.97	0	1
3	Forest in 1-km + Buildings	78	1.23E-06	157.24	0	0.1
	Forest Width + Roads	78	4.73E-06	157.24	0	0.1
	Buildings + Paved	78	8.79E-05	157.24	0	0.1
	Forest Width + Agriculture	78	9.08E-05	157.24	0	0.1
	Forest in 1-km + Mowed	78	0.000103	157.24	0	0.1
	Forest in 1-km + Forest Width	78	0.000109	157.24	0	0.1
	Forest in 1-km + Roads	78	0.000117	157.24	0	0.1
	Forest Width + Mowed	78	0.000123	157.24	0	0.1
	Forest Width + Buildings	78	0.000165	157.24	0	0.1
	Forest in 1-km + Agriculture	78	0.000171	157.24	0	0.1
4	Forest in 1-km	68	25.813	162.76	0	1
5	Buildings + Paved	102	0	206.12	0	0.1
	Forest Width + Agriculture	102	9.95E-05	206.12	0	0.1
	Forest in 1-km + Roads	102	0.000106	206.12	0	0.1
	Forest in 1-km + Forest Width	102	0.000112	206.12	0	0.1
	Forest Width + Buildings	102	0.000132	206.12	0	0.1
	Forest in 1-km + Agriculture	102	0.000138	206.12	0	0.1
	Forest in 1-km + Mowed	102	0.00014	206.12	0	0.1
	Forest Width + Mowed	102	0.000149	206.12	0	0.1
	Forest in 1-km + Buildings	102	0.000166	206.12	0	0.1
	Forest Width + Roads	102	0.000197	206.12	0	0.1
6	Forest in 1-km	68	0.00305	136.95	0	1
7	Forest in 1-km	68	1.091	138.04	0	1
8	Forest Width + Mowed	102	1.84E-05	206.12	0	0.1
	Forest Width + Roads	102	4.29E-05	206.12	0	0.1
	Buildings + Paved	102	5.63E-05	206.12	0	0.1
	Forest in 1-km + Forest Width	102	9.23E-05	206.12	0	0.1
	Forest Width + Agriculture	102	9.68E-05	206.12	0	0.1
	Forest in 1-km + Buildings	102	0.000106	206.12	0	0.1
	Forest in 1-km + Mowed	102	0.000115	206.12	0	0.1
	Forest Width + Buildings	102	0.000115	206.12	0	0.1
	Forest in 1-km + Agriculture	102	0.000118	206.12	0	0.1
	Forest in 1-km + Roads	102	0.000199	206.12	0	0.1

Table 3.4: AIC_c analysis of multinomial logistic regression models for irreplaceability values calculated for each portfolio size. Only models with Δ_i<2 are included.

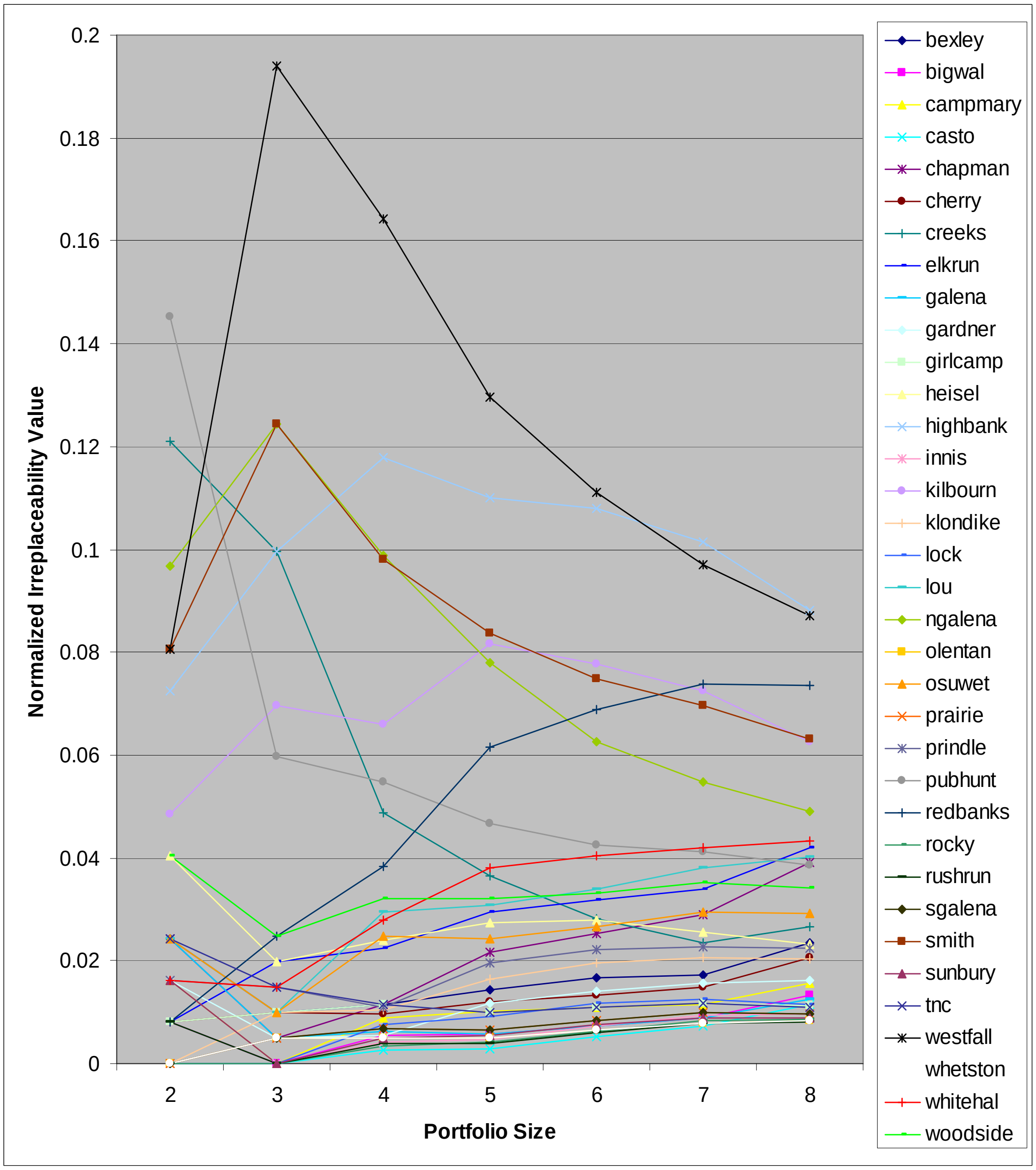

Figure 3.1: Irreplaceability values plotted as a function of portfolio size (normalized so that the sum of all irreplaceability values within a portfolio size is equal to 1).

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APPENDIX A

IRREPLACEABILITY VALUES AND RANKINGS FOR ALL SCORING TREATMENTS

Site	Irreplaceability value for the given portfolio size						
	2	3	4	5	6	7	8
bexley	0	0	0.0195	0.0338	0.0525	0.073	0.1004
bigwal	0	0.0312	0.023	0.0506	0.0791	0.1091	0.144
campmary	0	0	0.023	0.0466	0.0659	0.087	0.1133
casto	0	0	0	0.0031	0.0086	0.0194	0.0398
chapman	0.1667	0.2396	0.2938	0.3028	0.3314	0.3512	0.3643
cherry	0	0	0.0071	0.0341	0.0719	0.1243	0.1845
creeks	0	0.1354	0.1646	0.1812	0.1885	0.2042	0.2247
elkrun	0	0.0833	0.1292	0.1982	0.2389	0.2767	0.3046
galena	0	0	0.0283	0.0585	0.0794	0.0993	0.1246
gardner	0	0.0312	0.069	0.0923	0.1242	0.1577	0.1986
girlcamp	0	0.0208	0.0354	0.0488	0.0563	0.0689	0.0938
heisel	0	0.0521	0.1062	0.1698	0.2291	0.2748	0.3134
highbank	0.1667	0.1771	0.2708	0.3627	0.4662	0.5272	0.5487
innis	0	0	0.0018	0.0105	0.0222	0.0406	0.0689
kilbourn	0	0.1354	0.2142	0.2942	0.3643	0.4292	0.4739
klondike	0	0	0.0088	0.0383	0.0735	0.1162	0.1637
lock	0	0	0.0035	0.0128	0.0233	0.038	0.0636
lou	0	0.0521	0.0743	0.1207	0.1565	0.1896	0.2195
ngalena	0.1667	0.4167	0.6673	0.8069	0.8653	0.8886	0.8848
olentan	0	0.0208	0.0195	0.0227	0.0314	0.0444	0.0696
osuwet	0.3333	0.2292	0.2248	0.2119	0.2174	0.2338	0.2493
prairie	0	0	0.0248	0.0463	0.0671	0.0915	0.1167
prindle	0	0.0938	0.1522	0.1874	0.2178	0.2366	0.2602
pubhunt	0.8333	0.5312	0.3699	0.269	0.2299	0.2125	0.2143
redbanks	0	0	0.0283	0.0832	0.1616	0.2552	0.3371
rocky	0	0	0.0035	0.0216	0.0399	0.0633	0.0939
rushrun	0	0.0833	0.0761	0.0878	0.1132	0.132	0.1536
sgalena	0	0.0208	0.0531	0.0753	0.1004	0.1229	0.1457
smith	0.1667	0.2188	0.2673	0.3252	0.3432	0.3606	0.3751
sunbury	0	0.0208	0.0248	0.0542	0.0743	0.102	0.1328
tnc	0	0.0312	0.0442	0.069	0.0706	0.0835	0.1076
westfall	0.1667	0.3542	0.4354	0.4825	0.5553	0.6085	0.6208
whetston	0	0.0104	0.0195	0.0346	0.0515	0.0723	0.1039
whitehal	0	0	0.0159	0.0415	0.0776	0.13	0.1884
woodside	0	0.0104	0.1009	0.1218	0.152	0.176	0.2018

Table A.1: Irreplaceability values based on the top 1% of portfolios using the average scores with no weighting.

Site	Irreplaceability value for the given portfolio size						
	2	3	4	5	6	7	8
bexley	0	0	0.0217	0.0394	0.0637	0.0851	0.1071
bigwal	0	0.0143	0.0307	0.0468	0.0793	0.1093	0.1378
campmary	0	0	0.0307	0.0587	0.0793	0.0982	0.1177
casto	0	0	0.0036	0.0049	0.0122	0.0245	0.0386
chapman	0.1667	0.2	0.2315	0.2511	0.2718	0.2997	0.3376
cherry	0	0	0.0108	0.0379	0.0746	0.1229	0.1807
creeks	0	0.1286	0.1609	0.1835	0.2011	0.2145	0.2286
elkrun	0	0.0714	0.1302	0.1982	0.2362	0.2778	0.3227
galena	0	0	0.0253	0.0529	0.078	0.0983	0.1178
gardner	0	0.0286	0.0741	0.0899	0.11	0.1343	0.1587
girlcamp	0	0.0143	0.0362	0.0517	0.0589	0.0702	0.0812
heisel	0	0.0714	0.1248	0.1856	0.2468	0.2944	0.3365
highbank	0.1667	0.1143	0.2007	0.2618	0.333	0.4021	0.4612
innis	0	0	0.0072	0.0174	0.0318	0.0532	0.0753
kilbourn	0	0.0857	0.1754	0.2453	0.3089	0.3704	0.4275
klondike	0	0	0.0108	0.0428	0.0764	0.1157	0.1592
lock	0	0	0.0018	0.0153	0.0251	0.0391	0.0548
lou	0	0.0571	0.0669	0.1196	0.1573	0.1929	0.2302
ngalena	0.1667	0.4857	0.7523	0.8813	0.9292	0.9527	0.965
olentan	0	0.0143	0.0271	0.026	0.0376	0.0501	0.0651
osuwet	0.3333	0.1857	0.1899	0.1847	0.1945	0.2175	0.244
prairie	0	0.0143	0.0289	0.0612	0.0856	0.109	0.13
prindle	0	0.1286	0.1754	0.222	0.2486	0.2635	0.2738
pubhunt	0.8333	0.5429	0.3635	0.2612	0.2383	0.2244	0.2163
redbanks	0	0	0.0181	0.0734	0.1505	0.2369	0.3261
rocky	0	0	0.0072	0.0229	0.0437	0.0644	0.0869
rushrun	0	0.0571	0.0687	0.0771	0.1013	0.12	0.1398
sgalena	0	0.0286	0.0542	0.0801	0.106	0.1279	0.1499
smith	0.1667	0.2143	0.2984	0.3434	0.3654	0.3908	0.4099
sunbury	0	0	0.0199	0.0456	0.0698	0.0935	0.1176
tnc	0	0.0429	0.0633	0.0832	0.0874	0.099	0.1101
westfall	0.1667	0.4714	0.481	0.5434	0.6287	0.6953	0.7435
whetston	0	0.0143	0.0199	0.0336	0.0531	0.0727	0.0952
whitehal	0	0	0.0127	0.0346	0.0623	0.0994	0.1434
woodside	0	0.0143	0.0759	0.1235	0.1536	0.18	0.2103

Table A.2: Irreplaceability values based on the top 1% of portfolios using the average scores with weighting.

Site	Irreplaceability value for the given portfolio size						
	2	3	4	5	6	7	8
bexley	0	0.0316	0.0466	0.0729	0.0881	0.1592	0.1584
bigwal	0	0	0.0137	0.0388	0.0413	0.1031	0.1063
campmary	0	0	0.0288	0.065	0.0635	0.1301	0.1241
casto	0	0	0.0055	0.0236	0.0338	0.0838	0.0929
chapman	0	0.0211	0.0658	0.126	0.1602	0.2641	0.2658
cherry	0	0.0211	0.0342	0.0652	0.0664	0.1609	0.1578
creeks	0.3636	0.2842	0.2219	0.2135	0.1655	0.2407	0.2002
elkrun	0	0.0632	0.1164	0.1637	0.22	0.2992	0.2951
galena	0	0.0211	0.0219	0.0427	0.0434	0.1031	0.1058
gardner	0	0.0105	0.0384	0.0913	0.0928	0.1745	0.1795
girlcamp	0	0.0316	0.0479	0.0642	0.0737	0.0929	0.121
heisel	0	0.0632	0.1055	0.1365	0.1629	0.1726	0.2077
highbank	0.3636	0.4526	0.6534	0.6559	0.8266	0.702	0.7584
innis	0	0	0.0151	0.0384	0.0481	0.0795	0.1156
kilbourn	0.0909	0.2632	0.3753	0.4075	0.5222	0.4822	0.5361
klondike	0	0.0211	0.0534	0.086	0.1095	0.137	0.1758
lock	0	0.0105	0.0342	0.0612	0.0776	0.0922	0.1264
lou	0	0.0316	0.0795	0.1203	0.1588	0.1967	0.2591
ngalena	0.1818	0.2737	0.2562	0.2591	0.279	0.2772	0.3096
olentan	0	0.0105	0.0151	0.0333	0.0391	0.0635	0.096
osuwet	0.0909	0.0421	0.074	0.1108	0.1408	0.172	0.2207
prairie	0	0.0211	0.026	0.0465	0.0562	0.0768	0.1087
prindle	0	0.0211	0.0507	0.0984	0.1199	0.1493	0.1884
pubhunt	0.2727	0.1684	0.1795	0.1929	0.2236	0.2235	0.2592
redbanks	0	0.0737	0.1699	0.28	0.411	0.4374	0.5265
rocky	0	0	0.011	0.0315	0.0423	0.0646	0.0985
rushrun	0	0	0.011	0.0303	0.0379	0.0613	0.0943
sgalena	0	0.0211	0.026	0.0469	0.0562	0.0768	0.1088
smith	0.2727	0.3474	0.3767	0.3743	0.4091	0.4067	0.4547
sunbury	0	0.0105	0.0274	0.0466	0.0506	0.0739	0.1049
tnc	0	0.0316	0.0507	0.0643	0.0737	0.0912	0.121
westfall	0.3636	0.4526	0.4685	0.4841	0.5534	0.524	0.5543
whetston	0	0.0105	0.0151	0.0331	0.0391	0.0632	0.0959
whitehal	0	0.1158	0.174	0.2475	0.3445	0.3574	0.4207
woodside	0	0.0737	0.111	0.1476	0.1689	0.207	0.2519

Table A.3: Irreplaceability values based on the top 1% of portfolios using the cumulative scores with no weighting.

Site	Irreplaceability value for the given portfolio size						
	2	3	4	5	6	7	8
bexley	0	0.0299	0.0455	0.0712	0.0994	0.1197	0.1882
bigwal	0	0	0.0219	0.0292	0.0438	0.0621	0.1054
campmary	0	0	0.0354	0.0507	0.065	0.0807	0.1242
casto	0	0	0.0101	0.0149	0.032	0.0517	0.0925
chapman	0	0.0149	0.0455	0.1082	0.1518	0.2029	0.3132
cherry	0	0.0299	0.0387	0.0596	0.0802	0.1037	0.164
creeks	0.1667	0.2985	0.1953	0.1827	0.1685	0.1641	0.2136
elkrun	0	0.0597	0.0892	0.1475	0.1902	0.2375	0.335
galena	0	0.0149	0.0253	0.0283	0.044	0.0623	0.1002
gardner	0	0.0149	0.0219	0.0593	0.085	0.1098	0.1297
girlcamp	0	0.0299	0.0455	0.0495	0.0663	0.0823	0.089
heisel	0	0.0597	0.096	0.1374	0.1679	0.1794	0.1856
highbank	0.1667	0.2985	0.4714	0.5508	0.6482	0.7097	0.7042
innis	0	0	0.0168	0.0238	0.044	0.0652	0.0773
kilbourn	0	0.209	0.2643	0.4077	0.4657	0.5068	0.4997
klondike	0	0.0299	0.0421	0.0826	0.1178	0.1444	0.1623
lock	0	0	0.0303	0.045	0.0705	0.0876	0.0921
lou	0	0.0299	0.1178	0.1541	0.2038	0.2674	0.3208
ngalena	0.3333	0.3731	0.3956	0.3899	0.3759	0.3831	0.3915
olentan	0	0.0149	0.0202	0.0244	0.0389	0.0554	0.0661
osuwet	0	0.0299	0.0993	0.1207	0.159	0.2058	0.2346
prairie	0	0.0149	0.0269	0.0328	0.0504	0.0686	0.077
prindle	0	0.0448	0.0438	0.0972	0.1323	0.1591	0.1785
pubhunt	0.5	0.1791	0.2189	0.2334	0.2555	0.288	0.3085
redbanks	0	0.0746	0.1532	0.3076	0.413	0.517	0.5874
rocky	0	0	0.0135	0.0206	0.0383	0.058	0.0682
rushrun	0	0	0.0152	0.0194	0.0358	0.0539	0.065
sgalena	0	0.0149	0.0269	0.0328	0.0503	0.0686	0.0769
smith	0.3333	0.3731	0.3923	0.4182	0.4485	0.4875	0.5043
sunbury	0	0	0.0202	0.0271	0.0446	0.062	0.0717
tnc	0	0.0448	0.0455	0.0495	0.0663	0.0823	0.0885
westfall	0.5	0.5821	0.6566	0.6477	0.6667	0.6786	0.6972
whetston	0	0.0149	0.0202	0.0244	0.0389	0.0554	0.0661
whitehal	0	0.0448	0.1111	0.1908	0.2428	0.2932	0.3472
woodside	0	0.0746	0.1279	0.161	0.1986	0.2462	0.2741

Table A.4: Irreplaceability values based on the top 1% of portfolios using the cumulative scores with weighting.

Site	Irreplaceability value for the given portfolio size						
	2	3	4	5	6	7	8
bexley	0	0.0185	0.0363	0.0572	0.0758	0.099	0.1203
bigwal	0.0294	0.0439	0.0579	0.09	0.1158	0.1451	0.1674
campmary	0	0.0254	0.0443	0.0701	0.0937	0.1169	0.1349
casto	0	0.0092	0.0068	0.0142	0.0253	0.0436	0.0603
chapman	0.1471	0.2009	0.2403	0.2681	0.2889	0.3085	0.3345
cherry	0	0.0115	0.0265	0.0545	0.0934	0.1405	0.1893
creeks	0.2059	0.1848	0.2067	0.216	0.229	0.2417	0.2506
elkrun	0.0294	0.0716	0.1238	0.163	0.2075	0.2424	0.2778
galena	0	0.0254	0.0503	0.0754	0.1004	0.1244	0.1441
gardner	0	0.0393	0.0689	0.1107	0.1469	0.1816	0.2131
girlcamp	0	0.0346	0.0636	0.0842	0.1032	0.1231	0.134
heisel	0.0294	0.0785	0.1257	0.1788	0.2251	0.2667	0.305
highbank	0.0294	0.1293	0.2123	0.2874	0.3494	0.3973	0.4561
innis	0	0.0069	0.0132	0.0254	0.043	0.0674	0.091
kilbourn	0.1176	0.1594	0.2305	0.2853	0.3415	0.3879	0.4377
klondike	0	0.0185	0.0326	0.0613	0.095	0.1335	0.1722
lock	0	0.0162	0.0242	0.0365	0.053	0.0752	0.0926
lou	0.0294	0.0485	0.0818	0.1155	0.1489	0.1823	0.215
ngalena	0.1471	0.3025	0.4788	0.5947	0.6806	0.7226	0.7828
olentan	0.0294	0.0531	0.0454	0.0523	0.0634	0.0812	0.0961
osuwet	0.1471	0.1732	0.184	0.1892	0.201	0.2161	0.2382
prairie	0	0.0185	0.0348	0.0559	0.0815	0.1058	0.1279
prindle	0.0588	0.0924	0.1351	0.1712	0.2072	0.2353	0.2616
pubhunt	0.5	0.4342	0.3535	0.2907	0.2571	0.2464	0.2404
redbanks	0	0.0139	0.0443	0.0979	0.1615	0.2311	0.3056
rocky	0	0.0069	0.0159	0.0375	0.0617	0.0905	0.1141
rushrun	0.0588	0.0716	0.0916	0.1094	0.1255	0.1456	0.1635
sgalena	0	0.0485	0.0613	0.0852	0.1064	0.1296	0.1511
smith	0.1471	0.2032	0.2562	0.2913	0.3228	0.3446	0.368
sunbury	0.0588	0.0531	0.0708	0.0979	0.118	0.1424	0.1593
tnc	0	0.0346	0.0598	0.0846	0.1054	0.1258	0.1398
westfall	0.1176	0.2286	0.3418	0.3872	0.4294	0.4681	0.5265
whetston	0.0294	0.0346	0.0454	0.0602	0.0819	0.1062	0.1271
whitehal	0	0.0254	0.0307	0.063	0.1008	0.1464	0.1938
woodside	0.0882	0.0831	0.1048	0.1382	0.1601	0.1851	0.2086

Table A.5: Irreplaceability values based on the top 5% of portfolios using the average scores with no weighting.

Site	Irreplaceability value for the given portfolio size						
	2	3	4	5	6	7	8
bexley	0	0.0213	0.0405	0.0618	0.0835	0.1063	0.1307
bigwal	0.0333	0.0304	0.056	0.0856	0.1123	0.1378	0.164
campmary	0	0.0304	0.0514	0.0762	0.1012	0.1234	0.1441
casto	0	0.0091	0.0091	0.0192	0.0325	0.0494	0.0696
chapman	0.1	0.1793	0.1959	0.2252	0.2484	0.2733	0.2977
cherry	0	0.0091	0.0269	0.055	0.093	0.1382	0.1863
creeks	0.2333	0.1793	0.2057	0.2184	0.2284	0.2395	0.2523
elkrun	0.0333	0.076	0.1271	0.1652	0.2052	0.2397	0.2708
galena	0	0.0213	0.0465	0.0709	0.0961	0.1205	0.143
gardner	0.0333	0.0486	0.0749	0.1098	0.1424	0.1713	0.1978
girlcamp	0	0.0395	0.0666	0.0868	0.1087	0.1243	0.139
heisel	0.0333	0.0851	0.1418	0.1851	0.2347	0.2793	0.3167
highbank	0.0333	0.1125	0.1698	0.2245	0.2743	0.3222	0.3663
innis	0	0.0061	0.017	0.0345	0.0538	0.0777	0.1054
kilbourn	0.1	0.1307	0.2023	0.2534	0.3051	0.3503	0.394
klondike	0	0.0182	0.0367	0.0629	0.0958	0.1322	0.1705
lock	0	0.0213	0.0272	0.0422	0.0595	0.0798	0.1
lou	0.0333	0.0365	0.0772	0.1107	0.1451	0.1815	0.2164
ngalena	0.2	0.3708	0.5374	0.6619	0.7516	0.8124	0.859
olentan	0.0333	0.0426	0.048	0.0554	0.0677	0.0835	0.102
osuwet	0.1	0.1672	0.1611	0.1707	0.186	0.2048	0.2282
prairie	0	0.0182	0.045	0.0665	0.0921	0.1171	0.142
prindle	0.0333	0.0942	0.1547	0.1929	0.228	0.2587	0.2842
pubhunt	0.6	0.4802	0.3585	0.2927	0.2631	0.2501	0.2473
redbanks	0	0.0122	0.045	0.0927	0.1512	0.216	0.2846
rocky	0	0.003	0.0174	0.04	0.0652	0.0914	0.117
rushrun	0.0667	0.0638	0.0825	0.1008	0.1161	0.1352	0.1553
sgalena	0	0.0486	0.062	0.0859	0.1102	0.1321	0.1555
smith	0.1333	0.2006	0.2583	0.3034	0.3338	0.3603	0.3848
sunbury	0.0333	0.0334	0.0628	0.0888	0.1097	0.1317	0.1516
tnc	0	0.0365	0.0643	0.0968	0.1168	0.1351	0.1512
westfall	0.1	0.2675	0.3623	0.4184	0.4654	0.5176	0.5755
whetston	0.0333	0.0334	0.0454	0.061	0.0814	0.1028	0.1263
whitehal	0	0.0091	0.0257	0.0536	0.0847	0.1211	0.1608
woodside	0.0333	0.0638	0.0968	0.1309	0.1572	0.1835	0.2101

Table A.6: Irreplaceability values based on the top 5% of portfolios using the average scores with weighting.

Site	Irreplaceability value for the given portfolio size						
	2	3	4	5	6	7	8
bexley	0.0196	0.0442	0.0682	0.0857	0.1161	0.1693	0.1751
bigwal	0.0196	0.0244	0.046	0.0597	0.0859	0.1269	0.1313
campmary	0.0392	0.0579	0.0837	0.0922	0.1142	0.1533	0.1488
casto	0	0.0091	0.0197	0.0339	0.0587	0.0978	0.11
chapman	0.0784	0.0899	0.1175	0.1443	0.1856	0.2606	0.2649
cherry	0	0.0335	0.0703	0.0957	0.1344	0.194	0.1939
creeks	0.2549	0.2774	0.2547	0.2326	0.2277	0.2619	0.229
elkrun	0.0392	0.0838	0.1217	0.1596	0.1989	0.2752	0.2783
galena	0.0392	0.0518	0.0577	0.0636	0.0869	0.126	0.1301
gardner	0.0392	0.0579	0.0931	0.1156	0.1549	0.1994	0.2106
girlcamp	0.0196	0.0473	0.0623	0.0749	0.0955	0.108	0.1358
heisel	0.0588	0.093	0.1232	0.143	0.1684	0.1827	0.2156
highbank	0.2745	0.2957	0.3653	0.5033	0.5361	0.5661	0.633
innis	0	0.0168	0.0374	0.0541	0.0841	0.1015	0.1375
kilbourn	0.0588	0.1799	0.2639	0.3558	0.3872	0.4156	0.4645
klondike	0	0.0351	0.0657	0.0929	0.126	0.1479	0.1848
lock	0	0.0274	0.0498	0.07	0.0953	0.1061	0.1395
lou	0.0392	0.0488	0.091	0.1272	0.1683	0.1993	0.2536
ngalena	0.1569	0.1966	0.2187	0.2496	0.2623	0.2687	0.2997
olentan	0	0.0168	0.0339	0.0458	0.0684	0.0816	0.1147
osuwet	0.0588	0.0793	0.0954	0.1201	0.1538	0.1781	0.2212
prairie	0	0.0229	0.0393	0.055	0.0791	0.0923	0.1246
prindle	0.0196	0.0518	0.0793	0.1046	0.1393	0.1607	0.1963
pubhunt	0.2157	0.1814	0.1773	0.1839	0.2028	0.2171	0.2502
redbanks	0.0196	0.0854	0.1709	0.2573	0.3292	0.3872	0.4635
rocky	0	0.0152	0.0295	0.0426	0.067	0.081	0.1152
rushrun	0.0196	0.0244	0.0309	0.0415	0.0644	0.0785	0.1121
sgalena	0.0196	0.029	0.0399	0.0553	0.0797	0.0928	0.1249
smith	0.1961	0.3049	0.3246	0.3596	0.3748	0.3882	0.4303
sunbury	0.0196	0.0335	0.0487	0.0595	0.0822	0.0932	0.125
tnc	0.0392	0.0442	0.0602	0.0749	0.0954	0.1056	0.1357
westfall	0.1176	0.2378	0.3455	0.422	0.4473	0.4727	0.5081
whetston	0	0.0168	0.0324	0.0454	0.0681	0.0812	0.1146
whitehal	0.0588	0.0854	0.1539	0.2192	0.2707	0.3164	0.3734
woodside	0.0784	0.1006	0.1282	0.1597	0.1915	0.213	0.2539

Table A.7: Irreplaceability values based on the top 5% of portfolios using the cumulative scores with no weighting.

Site	Irreplaceability value for the given portfolio size						
	2	3	4	5	6	7	8
bexley	0.0333	0.0419	0.0664	0.0816	0.1098	0.1435	0.2054
bigwal	0	0.021	0.0424	0.0618	0.0788	0.099	0.1425
campmary	0	0.0479	0.0714	0.0918	0.1056	0.1206	0.1626
casto	0	0.009	0.0174	0.0347	0.0518	0.0753	0.1186
chapman	0.0667	0.0539	0.0885	0.1178	0.1655	0.2044	0.2903
cherry	0	0.0269	0.0547	0.0922	0.1255	0.1548	0.2128
creeks	0.3	0.2575	0.2371	0.2221	0.2137	0.2117	0.2519
elkrun	0.0333	0.0689	0.1004	0.1417	0.1844	0.2157	0.3021
galena	0.0333	0.0359	0.0468	0.0625	0.0774	0.097	0.1352
gardner	0	0.0269	0.0598	0.0966	0.1282	0.1558	0.1775
girlcamp	0.0333	0.0509	0.0602	0.076	0.0885	0.1063	0.1198
heisel	0.0333	0.0898	0.1265	0.1421	0.1666	0.1958	0.2106
highbank	0.1667	0.2485	0.3129	0.3952	0.4625	0.5307	0.5599
innis	0	0.021	0.033	0.0524	0.0726	0.0979	0.1161
kilbourn	0.1	0.1647	0.2487	0.3219	0.3829	0.4144	0.4213
klondike	0	0.0269	0.058	0.0842	0.1193	0.1514	0.1741
lock	0	0.0269	0.0457	0.0689	0.0869	0.1098	0.1212
lou	0.0333	0.0599	0.1008	0.1493	0.1942	0.2473	0.2865
ngalena	0.2	0.2216	0.2821	0.3113	0.3312	0.3518	0.3498
olentan	0	0.015	0.0312	0.0455	0.0619	0.0818	0.0966
osuwet	0.1	0.0629	0.0914	0.1288	0.1602	0.1985	0.2257
prairie	0	0.021	0.0373	0.0554	0.0716	0.0926	0.1074
prindle	0	0.0449	0.0761	0.0999	0.1364	0.1645	0.1877
pubhunt	0.3	0.2425	0.2277	0.2203	0.2478	0.2723	0.2893
redbanks	0.0333	0.0629	0.1512	0.2417	0.3354	0.4118	0.4631
rocky	0	0.009	0.0221	0.0418	0.0585	0.0812	0.0972
rushrun	0.0333	0.015	0.0247	0.0398	0.0563	0.0778	0.0943
sgalena	0	0.018	0.0348	0.054	0.0711	0.0923	0.1072
smith	0.2333	0.3473	0.3825	0.3963	0.4312	0.4509	0.4748
sunbury	0	0.0329	0.0381	0.0521	0.0681	0.0883	0.1034
tnc	0.0333	0.0539	0.062	0.0762	0.0886	0.1063	0.1188
westfall	0.1333	0.4102	0.5069	0.5571	0.5839	0.6121	0.6248
whetston	0	0.015	0.0308	0.0453	0.0619	0.0818	0.0966
whitehal	0	0.0629	0.0939	0.1695	0.216	0.2638	0.2913
woodside	0.1	0.0868	0.1363	0.1721	0.2057	0.2409	0.2638

Table A.8: Irreplaceability values based on the top 5% of portfolios using the cumulative scores with weighting.

Site	Irreplaceability value for the given portfolio size						
	2	3	4	5	6	7	8
bexley	0.0145	0.0264	0.0425	0.0659	0.0898	0.1154	0.1383
bigwal	0.029	0.0554	0.0763	0.1065	0.1344	0.1621	0.1847
campmary	0.029	0.039	0.0646	0.0881	0.1106	0.1355	0.1548
casto	0.0145	0.0101	0.0135	0.0248	0.0407	0.0631	0.0834
chapman	0.1449	0.1864	0.2272	0.2486	0.2688	0.2872	0.3115
cherry	0	0.0151	0.0328	0.0663	0.105	0.1489	0.1939
creeks	0.1304	0.199	0.2085	0.2268	0.2403	0.2542	0.2664
elkrun	0.029	0.0756	0.125	0.1623	0.1953	0.2293	0.2644
galena	0.029	0.0403	0.0632	0.0898	0.1127	0.1374	0.1587
gardner	0.029	0.0542	0.0771	0.1147	0.1513	0.1871	0.22
girlcamp	0.0145	0.0504	0.0729	0.1024	0.1274	0.1526	0.1699
heisel	0.0145	0.073	0.1333	0.1809	0.2221	0.2591	0.2955
highbank	0.0435	0.1322	0.2003	0.2556	0.3075	0.3472	0.3945
innis	0	0.0126	0.0201	0.0382	0.0593	0.0868	0.1129
kilbourn	0.1014	0.165	0.2165	0.2712	0.3194	0.3605	0.4054
klondike	0	0.0214	0.0427	0.0738	0.1077	0.1444	0.1802
lock	0.0145	0.0189	0.0326	0.0516	0.0736	0.1006	0.1221
lou	0.0145	0.0542	0.0859	0.1194	0.1505	0.1818	0.2131
ngalena	0.1739	0.262	0.3798	0.4644	0.5413	0.587	0.6544
olentan	0.0435	0.0542	0.0633	0.0683	0.0825	0.1024	0.1197
osuwet	0.1304	0.1511	0.1658	0.177	0.1915	0.2098	0.2318
prairie	0.0145	0.0365	0.0472	0.07	0.0918	0.1177	0.1401
prindle	0.0725	0.1008	0.1378	0.1711	0.1992	0.2274	0.2561
pubhunt	0.3913	0.3363	0.3295	0.2876	0.2682	0.262	0.2613
redbanks	0	0.0214	0.0547	0.1043	0.1625	0.2211	0.2827
rocky	0	0.0139	0.0253	0.0486	0.0756	0.1058	0.1318
rushrun	0.0435	0.0856	0.1051	0.1183	0.1365	0.1558	0.1741
sgalena	0.029	0.0491	0.0725	0.0932	0.1148	0.1385	0.1599
smith	0.1159	0.1902	0.2391	0.2759	0.3056	0.3294	0.3552
sunbury	0.0725	0.0756	0.0906	0.1195	0.1432	0.1687	0.1873
tnc	0.0435	0.0504	0.0708	0.0964	0.1211	0.146	0.165
westfall	0.1159	0.1927	0.2792	0.3304	0.3744	0.4059	0.4519
whetston	0.0145	0.0378	0.0503	0.0743	0.0966	0.1234	0.1459
whitehal	0.0145	0.0214	0.0416	0.0745	0.1126	0.1559	0.1992
woodside	0.0725	0.0919	0.1126	0.1392	0.1659	0.19	0.2136

Table A.9: Irreplaceability values based on the top 10% of portfolios using the average scores with no weighting.

Site	Irreplaceability value for the given portfolio size						
	2	3	4	5	6	7	8
bexley	0.0167	0.0272	0.046	0.0698	0.0933	0.1171	0.1422
bigwal	0.0167	0.0393	0.0704	0.0973	0.1256	0.1514	0.176
campmary	0.0167	0.0378	0.0674	0.09	0.1133	0.1354	0.1573
casto	0.0167	0.0091	0.017	0.0278	0.0445	0.0646	0.087
chapman	0.1167	0.1543	0.1888	0.2139	0.2365	0.2611	0.2864
cherry	0	0.0136	0.0331	0.0651	0.1023	0.1445	0.1901
creeks	0.1667	0.1906	0.2083	0.2296	0.2416	0.2516	0.2634
elkrun	0.05	0.0817	0.1274	0.1611	0.195	0.2302	0.2621
galena	0	0.0287	0.0559	0.0825	0.1053	0.129	0.1534
gardner	0.0333	0.053	0.0799	0.1134	0.1463	0.1771	0.2073
girlcamp	0.0167	0.0484	0.0776	0.1052	0.129	0.1501	0.1697
heisel	0.0167	0.0787	0.1441	0.1922	0.2318	0.2712	0.3079
highbank	0.0333	0.1059	0.1583	0.2102	0.2561	0.2992	0.3385
innis	0	0.0166	0.0265	0.0435	0.0656	0.0914	0.1201
kilbourn	0.1	0.1483	0.2014	0.2509	0.2968	0.3392	0.379
klondike	0	0.0197	0.0454	0.0744	0.1057	0.1402	0.1767
lock	0.0167	0.0197	0.0381	0.0559	0.0788	0.1011	0.1247
lou	0.0167	0.0469	0.0784	0.1155	0.1471	0.1793	0.2136
ngalena	0.2167	0.3359	0.4351	0.5397	0.628	0.7012	0.7551
olentan	0.05	0.0514	0.0629	0.0678	0.0819	0.0991	0.1191
osuwet	0.1167	0.1392	0.1471	0.1611	0.1793	0.2007	0.2241
prairie	0.0167	0.0363	0.0528	0.0745	0.0986	0.123	0.147
prindle	0.0667	0.1165	0.1585	0.189	0.2184	0.247	0.275
pubhunt	0.4333	0.3797	0.3425	0.2953	0.2758	0.2663	0.2656
redbanks	0	0.0227	0.0526	0.0982	0.1529	0.2102	0.2693
rocky	0	0.0151	0.0265	0.0486	0.0753	0.1027	0.1301
rushrun	0.05	0.0832	0.0913	0.1083	0.1263	0.1444	0.1647
sgalena	0.0167	0.0424	0.0706	0.0927	0.1146	0.1373	0.1603
smith	0.1	0.1952	0.241	0.2813	0.3166	0.3465	0.3721
sunbury	0.0333	0.059	0.0825	0.1078	0.1313	0.1521	0.1743
tnc	0.0667	0.059	0.0788	0.1018	0.1271	0.1489	0.1692
westfall	0.1	0.2057	0.3016	0.3644	0.4103	0.4532	0.4966
whetston	0.0167	0.0348	0.05	0.0721	0.093	0.1163	0.1403
whitehal	0.0167	0.0242	0.0392	0.0636	0.0943	0.1303	0.1694
woodside	0.0667	0.0802	0.1032	0.1354	0.1617	0.1871	0.2124

Table A.10: Irreplaceability values based on the top 10% of portfolios using the average scores with weighting.

Site	Irreplaceability value for the given portfolio size						
	2	3	4	5	6	7	8
bexley	0.0505	0.0442	0.0787	0.099	0.1295	0.1799	0.1885
bigwal	0.0101	0.0244	0.0596	0.0783	0.1068	0.149	0.1549
campmary	0.0303	0.0579	0.1014	0.1132	0.1345	0.1725	0.1706
casto	0	0.0091	0.0285	0.0451	0.0731	0.114	0.1281
chapman	0.0808	0.0899	0.1319	0.1586	0.1946	0.2581	0.2645
cherry	0.0101	0.0335	0.0828	0.1157	0.1573	0.2155	0.2201
creeks	0.1818	0.2774	0.2437	0.2424	0.2381	0.2726	0.2485
elkrun	0.0606	0.0838	0.1221	0.1588	0.194	0.2593	0.2673
galena	0.0707	0.0518	0.076	0.0842	0.1083	0.1481	0.1534
gardner	0.0404	0.0579	0.111	0.1373	0.1723	0.2149	0.2312
girlcamp	0.0202	0.0473	0.075	0.0852	0.1082	0.1241	0.1509
heisel	0.0808	0.093	0.1277	0.1505	0.1759	0.1921	0.2215
highbank	0.1818	0.2957	0.2935	0.3825	0.4167	0.4482	0.5124
innis	0	0.0168	0.0525	0.0714	0.1027	0.1228	0.1582
kilbourn	0.1111	0.1799	0.2184	0.2998	0.3334	0.361	0.4065
klondike	0.0101	0.0351	0.0728	0.0991	0.1331	0.1579	0.1944
lock	0	0.0274	0.0623	0.0795	0.1075	0.1225	0.1544
lou	0.0303	0.0488	0.0971	0.13	0.17	0.1994	0.2487
ngalena	0.1414	0.1966	0.1916	0.2283	0.2449	0.2546	0.288
olentan	0.0101	0.0168	0.044	0.0591	0.0844	0.1005	0.1338
osuwet	0.0606	0.0793	0.1066	0.1285	0.1601	0.1833	0.2234
prairie	0.0202	0.0229	0.0502	0.0664	0.0926	0.1087	0.1409
prindle	0.0202	0.0518	0.0909	0.1149	0.1463	0.1687	0.2046
pubhunt	0.2222	0.1814	0.1755	0.1839	0.1989	0.2135	0.2448
redbanks	0.0101	0.0854	0.1576	0.2375	0.2952	0.3429	0.4089
rocky	0	0.0152	0.0388	0.0548	0.0818	0.099	0.133
rushrun	0.0202	0.0244	0.044	0.0547	0.0798	0.0969	0.1306
sgalena	0.0303	0.029	0.054	0.0675	0.0933	0.1093	0.1415
smith	0.1717	0.3049	0.2841	0.3284	0.3474	0.3632	0.4025
sunbury	0.0404	0.0335	0.066	0.075	0.1003	0.1143	0.1454
tnc	0.0404	0.0442	0.0736	0.0847	0.108	0.1213	0.1508
westfall	0.1212	0.2378	0.2792	0.3674	0.393	0.4187	0.4579
whetston	0.0101	0.0168	0.0417	0.058	0.084	0.1	0.1337
whitehal	0.0404	0.0854	0.1376	0.2003	0.2426	0.2807	0.3335
woodside	0.0707	0.1006	0.1296	0.1599	0.1916	0.2124	0.2527

Table A.11: Irreplaceability values based on the top 10% of portfolios using the cumulative scores with no weighting.

Site	Irreplaceability value for the given portfolio size						
	2	3	4	5	6	7	8
bexley	0.0484	0.0457	0.0659	0.0962	0.1257	0.1544	0.2091
bigwal	0.0161	0.0244	0.0507	0.0748	0.0949	0.1163	0.1623
campmary	0.0323	0.0564	0.0838	0.1079	0.1228	0.1385	0.181
casto	0	0.0107	0.0253	0.0405	0.0636	0.0878	0.1332
chapman	0.0484	0.0808	0.1014	0.1386	0.172	0.2137	0.2825
cherry	0.0161	0.0335	0.069	0.1105	0.1407	0.1723	0.2341
creeks	0.2419	0.2744	0.2378	0.2316	0.2221	0.2229	0.2662
elkrun	0.0161	0.0732	0.1076	0.1535	0.183	0.2207	0.2846
galena	0.0484	0.0534	0.0598	0.0765	0.0936	0.1142	0.1537
gardner	0.0323	0.0518	0.0834	0.113	0.1374	0.1661	0.193
girlcamp	0.0161	0.0549	0.0716	0.0864	0.1011	0.1194	0.1349
heisel	0.0806	0.1037	0.1322	0.1582	0.1834	0.2038	0.2157
highbank	0.1452	0.2317	0.278	0.3182	0.3906	0.4429	0.4647
innis	0	0.0183	0.0419	0.0656	0.0914	0.1163	0.1354
kilbourn	0.0968	0.1723	0.2372	0.2928	0.3284	0.3674	0.3952
klondike	0	0.0351	0.0608	0.0974	0.1286	0.1615	0.182
lock	0	0.0259	0.0573	0.0768	0.0998	0.1203	0.1366
lou	0.0484	0.0595	0.0993	0.1419	0.1933	0.2391	0.2725
ngalena	0.1935	0.2073	0.2408	0.2673	0.3017	0.325	0.3319
olentan	0	0.0137	0.0343	0.053	0.0731	0.0952	0.113
osuwet	0.0484	0.0793	0.093	0.1288	0.1643	0.2016	0.2255
prairie	0	0.0335	0.0467	0.0632	0.0846	0.1057	0.1222
prindle	0.0323	0.0518	0.0791	0.1168	0.1441	0.1766	0.1939
pubhunt	0.2903	0.2043	0.2301	0.2126	0.2416	0.2628	0.2799
redbanks	0.0161	0.0793	0.1541	0.2347	0.3028	0.3724	0.4206
rocky	0	0.0152	0.0322	0.0479	0.0709	0.094	0.1128
rushrun	0.0161	0.0198	0.032	0.0472	0.0679	0.0908	0.1102
sgalena	0	0.029	0.0427	0.0619	0.0841	0.1056	0.1222
smith	0.1613	0.2957	0.3448	0.3895	0.4053	0.4326	0.4508
sunbury	0.0323	0.032	0.0514	0.0644	0.0831	0.1045	0.1207
tnc	0.0484	0.061	0.0716	0.0867	0.1013	0.1195	0.1337
westfall	0.1613	0.2942	0.4138	0.4803	0.5372	0.565	0.5757
whetston	0	0.0137	0.0333	0.0524	0.073	0.0951	0.113
whitehal	0.0323	0.0595	0.1109	0.1472	0.1874	0.238	0.2778
woodside	0.0806	0.1052	0.1261	0.1659	0.2054	0.2379	0.2592

Table A.12: Irreplaceability values based on the top 10% of portfolios using the cumulative scores with weighting.

Site	Average		Average Weighted		Cumulative		Cumulative Weighted	
	Rank	Value	Rank	Value	Rank	Value	Rank	Value
bexley	22	0.0145	18	0.0167	14	0.0505	10	0.0484
bigwal	16	0.029	18	0.0167	26	0.0101	21	0.0161
campmary	16	0.029	18	0.0167	19	0.0303	16	0.0323
casto	22	0.0145	18	0.0167	32	0	27	0
chapman	3	0.1449	4	0.1167	8	0.0808	10	0.0484
cherry	31	0	30	0	26	0.0101	21	0.0161
creeks	4	0.1304	3	0.1667	2	0.1818	2	0.2419
elkrun	16	0.029	12	0.05	12	0.0606	21	0.0161
galena	16	0.029	30	0	10	0.0707	10	0.0484
gardner	16	0.029	15	0.0333	15	0.0404	16	0.0323
girlcamp	22	0.0145	18	0.0167	22	0.0202	21	0.0161
heisel	22	0.0145	18	0.0167	8	0.0808	8	0.0806
highbank	12	0.0435	15	0.0333	2	0.1818	6	0.1452
innis	31	0	30	0	32	0	27	0
kilbourn	8	0.1014	6	0.1	7	0.1111	7	0.0968
klondike	31	0	30	0	26	0.0101	27	0
lock	22	0.0145	18	0.0167	32	0	27	0
lou	22	0.0145	18	0.0167	19	0.0303	10	0.0484
ngalena	2	0.1739	2	0.2167	5	0.1414	3	0.1935
olentan	12	0.0435	12	0.05	26	0.0101	27	0
osuwet	4	0.1304	4	0.1167	12	0.0606	10	0.0484
prairie	22	0.0145	18	0.0167	22	0.0202	27	0
prindle	9	0.0725	9	0.0667	22	0.0202	16	0.0323
pubhunt	1	0.3913	1	0.4333	1	0.2222	1	0.2903
redbanks	31	0	30	0	26	0.0101	21	0.0161
rocky	31	0	30	0	32	0	27	0
rushrun	12	0.0435	12	0.05	22	0.0202	21	0.0161
sgalena	16	0.029	18	0.0167	19	0.0303	27	0
smith	6	0.1159	6	0.1	4	0.1717	4	0.1613
sunbury	9	0.0725	15	0.0333	15	0.0404	16	0.0323
tnc	12	0.0435	9	0.0667	15	0.0404	10	0.0484
westfall	6	0.1159	6	0.1	6	0.1212	4	0.1613
whetston	22	0.0145	18	0.0167	26	0.0101	27	0
whitehal	22	0.0145	18	0.0167	15	0.0404	16	0.0323
woodside	9	0.0725	9	0.0667	10	0.0707	8	0.0806

Table A.13: Site ranks and irreplaceability values under the four different scoring treatments calculated using the top 10% of all portfolios of size 2.

Site	Average		Average Weighted		Cumulative		Cumulative Weighted	
	Rank	Value	Rank	Value	Rank	Value	Rank	Value
bexley	24	0	24	0	14	0.0316	15	0.0299
bigwal	15	0.0312	18	0.0143	30	0	28	0
campmary	24	0	24	0	30	0	28	0
casto	24	0	24	0	30	0	28	0
chapman	4	0.2396	5	0.2	18	0.0211	21	0.0149
cherry	24	0	24	0	18	0.0211	15	0.0299
creeks	8	0.1354	7	0.1286	4	0.2842	4	0.2985
elkrun	11	0.0833	11	0.0714	11	0.0632	10	0.0597
galena	24	0	24	0	18	0.0211	21	0.0149
gardner	15	0.0312	16	0.0286	25	0.0105	21	0.0149
girlcamp	18	0.0208	18	0.0143	14	0.0316	15	0.0299
heisel	13	0.0521	11	0.0714	11	0.0632	10	0.0597
highbank	7	0.1771	9	0.1143	1	0.4526	4	0.2985
innis	24	0	24	0	30	0	28	0
kilbourn	8	0.1354	10	0.0857	6	0.2632	6	0.209
klondike	24	0	24	0	18	0.0211	15	0.0299
lock	24	0	24	0	25	0.0105	28	0
lou	13	0.0521	13	0.0571	14	0.0316	15	0.0299
ngalena	2	0.4167	2	0.4857	5	0.2737	2	0.3731
olentan	18	0.0208	18	0.0143	25	0.0105	21	0.0149
osuwet	5	0.2292	6	0.1857	13	0.0421	15	0.0299
prairie	24	0	18	0.0143	18	0.0211	21	0.0149
prindle	10	0.0938	7	0.1286	18	0.0211	12	0.0448
pubhunt	1	0.5312	1	0.5429	7	0.1684	7	0.1791
redbanks	24	0	24	0	9	0.0737	8	0.0746
rocky	24	0	24	0	30	0	28	0
rushrun	11	0.0833	13	0.0571	30	0	28	0
sgalena	18	0.0208	16	0.0286	18	0.0211	21	0.0149
smith	6	0.2188	4	0.2143	3	0.3474	2	0.3731
sunbury	18	0.0208	24	0	25	0.0105	28	0
tnc	15	0.0312	15	0.0429	14	0.0316	12	0.0448
westfall	3	0.3542	3	0.4714	1	0.4526	1	0.5821
whetston	22	0.0104	18	0.0143	25	0.0105	21	0.0149
whitehal	24	0	24	0	8	0.1158	12	0.0448
woodside	22	0.0104	18	0.0143	9	0.0737	8	0.0746

Table A.14: Site ranks and irreplaceability values under the four different scoring treatments calculated using the top 1% of all portfolios of size 3.

Site	Average		Average Weighted		Cumulative		Cumulative Weighted	
	Rank	Value	Rank	Value	Rank	Value	Rank	Value
bexley	26	0.0195	25	0.0217	20	0.0466	15	0.0455
bigwal	24	0.023	20	0.0307	32	0.0137	27	0.0219
campmary	24	0.023	20	0.0307	24	0.0288	22	0.0354
casto	35	0	34	0.0036	35	0.0055	35	0.0101
chapman	4	0.2938	5	0.2315	15	0.0658	15	0.0455
cherry	31	0.0071	30	0.0108	22	0.0342	21	0.0387
creeks	9	0.1646	10	0.1609	6	0.2219	7	0.1953
elkrun	11	0.1292	11	0.1302	10	0.1164	14	0.0892
galena	20	0.0283	24	0.0253	28	0.0219	26	0.0253
gardner	16	0.069	14	0.0741	21	0.0384	27	0.0219
girlcamp	19	0.0354	19	0.0362	19	0.0479	15	0.0455
heisel	12	0.1062	12	0.1248	12	0.1055	13	0.096
highbank	5	0.2708	6	0.2007	1	0.6534	2	0.4714
innis	34	0.0018	32	0.0072	29	0.0151	32	0.0168
kilbourn	8	0.2142	8	0.1754	4	0.3753	5	0.2643
klondike	30	0.0088	30	0.0108	16	0.0534	20	0.0421
lock	32	0.0035	35	0.0018	22	0.0342	23	0.0303
lou	15	0.0743	16	0.0669	13	0.0795	10	0.1178
ngalena	1	0.6673	1	0.7523	5	0.2562	3	0.3956
olentan	26	0.0195	23	0.0271	29	0.0151	29	0.0202
osuwet	7	0.2248	7	0.1899	14	0.074	12	0.0993
prairie	22	0.0248	22	0.0289	26	0.026	24	0.0269
prindle	10	0.1522	8	0.1754	17	0.0507	19	0.0438
pubhunt	3	0.3699	3	0.3635	7	0.1795	6	0.2189
redbanks	20	0.0283	28	0.0181	9	0.1699	8	0.1532
rocky	32	0.0035	32	0.0072	33	0.011	34	0.0135
rushrun	14	0.0761	15	0.0687	33	0.011	33	0.0152
sgalena	17	0.0531	18	0.0542	26	0.026	24	0.0269
smith	6	0.2673	4	0.2984	3	0.3767	4	0.3923
sunbury	22	0.0248	26	0.0199	25	0.0274	29	0.0202
tnc	18	0.0442	17	0.0633	17	0.0507	15	0.0455
westfall	2	0.4354	2	0.481	2	0.4685	1	0.6566
whetston	26	0.0195	26	0.0199	29	0.0151	29	0.0202
whitehal	29	0.0159	29	0.0127	8	0.174	11	0.1111
woodside	13	0.1009	13	0.0759	11	0.111	9	0.1279

Table A.15: Site ranks and irreplaceability values under the four different scoring treatments calculated using the top 1% of all portfolios of size 4.

Site	Average		Average Weighted		Cumulative		Cumulative Weighted	
	Rank	Value	Rank	Value	Rank	Value	Rank	Value
bexley	30	0.0338	27	0.0394	19	0.0729	18	0.0712
bigwal	22	0.0506	24	0.0468	29	0.0388	27	0.0292
campmary	24	0.0466	21	0.0587	21	0.065	21	0.0507
casto	35	0.0031	35	0.0049	35	0.0236	35	0.0149
chapman	5	0.3028	6	0.2511	13	0.126	15	0.1082
cherry	29	0.0341	28	0.0379	20	0.0652	19	0.0596
creeks	11	0.1812	12	0.1835	8	0.2135	9	0.1827
elkrun	9	0.1982	9	0.1982	10	0.1637	12	0.1475
galena	20	0.0585	22	0.0529	28	0.0427	28	0.0283
gardner	15	0.0923	15	0.0899	17	0.0913	20	0.0593
girlcamp	23	0.0488	23	0.0517	23	0.0642	22	0.0495
heisel	12	0.1698	10	0.1856	12	0.1365	13	0.1374
highbank	3	0.3627	4	0.2618	1	0.6559	2	0.5508
innis	34	0.0105	33	0.0174	30	0.0384	32	0.0238
kilbourn	6	0.2942	7	0.2453	3	0.4075	4	0.4077
klondike	27	0.0383	26	0.0428	18	0.086	17	0.0826
lock	33	0.0128	34	0.0153	24	0.0612	24	0.045
lou	14	0.1207	14	0.1196	14	0.1203	11	0.1541
ngalena	1	0.8069	1	0.8813	6	0.2591	5	0.3899
olentan	31	0.0227	31	0.026	31	0.0333	30	0.0244
osuwet	8	0.2119	11	0.1847	15	0.1108	14	0.1207
prairie	25	0.0463	20	0.0612	27	0.0465	25	0.0328
prindle	10	0.1874	8	0.222	16	0.0984	16	0.0972
pubhunt	7	0.269	5	0.2612	9	0.1929	7	0.2334
redbanks	17	0.0832	19	0.0734	5	0.28	6	0.3076
rocky	32	0.0216	32	0.0229	33	0.0315	33	0.0206
rushrun	16	0.0878	18	0.0771	34	0.0303	34	0.0194
sgalena	18	0.0753	17	0.0801	25	0.0469	25	0.0328
smith	4	0.3252	3	0.3434	4	0.3743	3	0.4182
sunbury	21	0.0542	25	0.0456	26	0.0466	29	0.0271
tnc	19	0.069	16	0.0832	22	0.0643	22	0.0495
westfall	2	0.4825	2	0.5434	2	0.4841	1	0.6477
whetston	28	0.0346	30	0.0336	32	0.0331	30	0.0244
whitehal	26	0.0415	29	0.0346	7	0.2475	8	0.1908
woodside	13	0.1218	13	0.1235	11	0.1476	10	0.161

Table A.16: Site ranks and irreplaceability values under the four different scoring treatments calculated using the top 1% of all portfolios of size 5.

Site	Average		Average Weighted		Cumulative		Cumulative Weighted	
	Rank	Value	Rank	Value	Rank	Value	Rank	Value
bexley	29	0.0525	27	0.0637	19	0.0881	18	0.0994
bigwal	20	0.0791	21	0.0793	31	0.0413	30	0.0438
campmary	27	0.0659	21	0.0793	24	0.0635	24	0.065
casto	35	0.0086	35	0.0122	35	0.0338	35	0.032
chapman	6	0.3314	6	0.2718	13	0.1602	15	0.1518
cherry	24	0.0719	25	0.0746	23	0.0664	20	0.0802
creeks	12	0.1885	11	0.2011	11	0.1655	12	0.1685
elkrun	7	0.2389	10	0.2362	9	0.22	11	0.1902
galena	19	0.0794	23	0.078	29	0.0434	28	0.044
gardner	16	0.1242	16	0.11	18	0.0928	19	0.085
girlcamp	28	0.0563	29	0.0589	21	0.0737	22	0.0663
heisel	9	0.2291	8	0.2468	12	0.1629	13	0.1679
highbank	3	0.4662	4	0.333	1	0.8266	2	0.6482
innis	34	0.0222	33	0.0318	28	0.0481	28	0.044
kilbourn	4	0.3643	5	0.3089	3	0.5222	3	0.4657
klondike	23	0.0735	24	0.0764	17	0.1095	17	0.1178
lock	33	0.0233	34	0.0251	20	0.0776	21	0.0705
lou	14	0.1565	13	0.1573	14	0.1588	9	0.2038
ngalena	1	0.8653	1	0.9292	7	0.279	6	0.3759
olentan	32	0.0314	32	0.0376	32	0.0391	31	0.0389
osuwet	11	0.2174	12	0.1945	15	0.1408	14	0.159
prairie	26	0.0671	20	0.0856	25	0.0562	25	0.0504
prindle	10	0.2178	7	0.2486	16	0.1199	16	0.1323
pubhunt	8	0.2299	9	0.2383	8	0.2236	7	0.2555
redbanks	13	0.1616	15	0.1505	4	0.411	5	0.413
rocky	31	0.0399	31	0.0437	30	0.0423	33	0.0383
rushrun	17	0.1132	18	0.1013	34	0.0379	34	0.0358
sgalena	18	0.1004	17	0.106	25	0.0562	26	0.0503
smith	5	0.3432	3	0.3654	5	0.4091	4	0.4485
sunbury	22	0.0743	26	0.0698	27	0.0506	27	0.0446
tnc	25	0.0706	19	0.0874	21	0.0737	22	0.0663
westfall	2	0.5553	2	0.6287	2	0.5534	1	0.6667
whetston	30	0.0515	30	0.0531	32	0.0391	31	0.0389
whitehal	21	0.0776	28	0.0623	6	0.3445	8	0.2428
woodside	15	0.152	14	0.1536	10	0.1689	10	0.1986

Table A.17: Site ranks and irreplaceability values under the four different scoring treatments calculated using the top 1% of all portfolios of size 6.

Site	Average		Average Weighted		Cumulative		Cumulative Weighted	
	Rank	Value	Rank	Value	Rank	Value	Rank	Value
bexley	28	0.073	28	0.0851	18	0.1592	18	0.1197
bigwal	22	0.1091	21	0.1093	22	0.1031	29	0.0621
campmary	26	0.087	26	0.0982	21	0.1301	24	0.0807
casto	35	0.0194	35	0.0245	27	0.0838	35	0.0517
chapman	6	0.3512	6	0.2997	9	0.2641	13	0.2029
cherry	19	0.1243	18	0.1229	17	0.1609	20	0.1037
creeks	13	0.2042	13	0.2145	10	0.2407	15	0.1641
elkrun	7	0.2767	8	0.2778	7	0.2992	11	0.2375
galena	24	0.0993	25	0.0983	22	0.1031	28	0.0623
gardner	16	0.1577	16	0.1343	14	0.1745	19	0.1098
girlcamp	30	0.0689	30	0.0702	24	0.0929	22	0.0823
heisel	8	0.2748	7	0.2944	15	0.1726	14	0.1794
highbank	3	0.5272	3	0.4021	1	0.702	1	0.7097
innis	33	0.0406	32	0.0532	28	0.0795	27	0.0652
kilbourn	4	0.4292	5	0.3704	3	0.4822	4	0.5068
klondike	21	0.1162	20	0.1157	20	0.137	17	0.1444
lock	34	0.038	34	0.0391	25	0.0922	21	0.0876
lou	14	0.1896	14	0.1929	13	0.1967	9	0.2674
ngalena	1	0.8886	1	0.9527	8	0.2772	6	0.3831
olentan	32	0.0444	33	0.0501	33	0.0635	32	0.0554
osuwet	11	0.2338	12	0.2175	16	0.172	12	0.2058
prairie	25	0.0915	22	0.109	29	0.0768	25	0.0686
prindle	10	0.2366	9	0.2635	19	0.1493	16	0.1591
pubhunt	12	0.2125	11	0.2244	11	0.2235	8	0.288
redbanks	9	0.2552	10	0.2369	4	0.4374	3	0.517
rocky	31	0.0633	31	0.0644	32	0.0646	31	0.058
rushrun	17	0.132	19	0.12	35	0.0613	34	0.0539
sgalena	20	0.1229	17	0.1279	29	0.0768	25	0.0686
smith	5	0.3606	4	0.3908	5	0.4067	5	0.4875
sunbury	23	0.102	27	0.0935	31	0.0739	30	0.062
tnc	27	0.0835	24	0.099	26	0.0912	22	0.0823
westfall	2	0.6085	2	0.6953	2	0.524	2	0.6786
whetston	29	0.0723	29	0.0727	34	0.0632	32	0.0554
whitehal	18	0.13	23	0.0994	6	0.3574	7	0.2932
woodside	15	0.176	15	0.18	12	0.207	10	0.2462

Table A.18: Site ranks and irreplaceability values under the four different scoring treatments calculated using the top 1% of all portfolios of size 7.

Site	Average		Average Weighted		Cumulative		Cumulative Weighted	
	Rank	Value	Rank	Value	Rank	Value	Rank	Value
bexley	29	0.1004	28	0.1071	19	0.1584	15	0.1882
bigwal	22	0.144	22	0.1378	28	0.1063	22	0.1054
campmary	26	0.1133	25	0.1177	22	0.1241	21	0.1242
casto	35	0.0398	35	0.0386	35	0.0929	24	0.0925
chapman	6	0.3643	6	0.3376	9	0.2658	10	0.3132
cherry	18	0.1845	16	0.1807	20	0.1578	18	0.164
creeks	12	0.2247	13	0.2286	15	0.2002	14	0.2136
elkrun	9	0.3046	9	0.3227	8	0.2951	8	0.335
galena	24	0.1246	24	0.1178	29	0.1058	23	0.1002
gardner	16	0.1986	18	0.1587	17	0.1795	20	0.1297
girlcamp	31	0.0938	31	0.0812	23	0.121	26	0.089
heisel	8	0.3134	7	0.3365	14	0.2077	16	0.1856
highbank	3	0.5487	3	0.4612	1	0.7584	1	0.7042
innis	33	0.0689	32	0.0753	25	0.1156	28	0.0773
kilbourn	4	0.4739	4	0.4275	3	0.5361	5	0.4997
klondike	19	0.1637	17	0.1592	18	0.1758	19	0.1623
lock	34	0.0636	34	0.0548	21	0.1264	25	0.0921
lou	13	0.2195	12	0.2302	11	0.2591	9	0.3208
ngalena	1	0.8848	1	0.965	7	0.3096	6	0.3915
olentan	32	0.0696	33	0.0651	32	0.096	33	0.0661
osuwet	11	0.2493	11	0.244	13	0.2207	13	0.2346
prairie	25	0.1167	23	0.13	27	0.1087	29	0.077
prindle	10	0.2602	10	0.2738	16	0.1884	17	0.1785
pubhunt	14	0.2143	14	0.2163	10	0.2592	11	0.3085
redbanks	7	0.3371	8	0.3261	4	0.5265	3	0.5874
rocky	30	0.0939	30	0.0869	31	0.0985	32	0.0682
rushrun	20	0.1536	21	0.1398	34	0.0943	35	0.065
sgalena	21	0.1457	19	0.1499	26	0.1088	30	0.0769
smith	5	0.3751	5	0.4099	5	0.4547	4	0.5043
sunbury	23	0.1328	26	0.1176	30	0.1049	31	0.0717
tnc	27	0.1076	27	0.1101	23	0.121	27	0.0885
westfall	2	0.6208	2	0.7435	2	0.5543	2	0.6972
whetston	28	0.1039	29	0.0952	33	0.0959	33	0.0661
whitehal	17	0.1884	20	0.1434	6	0.4207	7	0.3472
woodside	15	0.2018	15	0.2103	12	0.2519	12	0.2741

Table A.19: Site ranks and irreplaceability values under the four different scoring treatments calculated using the top 1% of all portfolios of size 8.

APPENDIX B

RAW LANDSCAPE DATA AND FULL STATISTICAL RESULTS OF MODEL FITS

Site	Forest Width (m)	Forest in 1-km	Agriculture	Number of Buildings	Mowed	Paved	Road
bexley	133	0.143	0	1692	0.505	0.141	0.082
bigwal	115	0.165	0	2233	0.449	0.159	0.078
campmary	565	0.463	0	681	0.342	0.074	0.045
casto	202	0.186	0	1776	0.422	0.200	0.078
chapman	87	0.537	0.243	92	0.163	0.015	0.015
cherry	165	0.222	0.020	997	0.364	0.155	0.075
creeks	133	0.533	0.095	92	0.104	0.041	0.021
elkrun	167	0.223	0.308	812	0.273	0.061	0.053
galena	277	0.352	0.155	262	0.223	0.040	0.024
gardner	125	0.552	0.195	248	0.130	0.022	0.010
girlcamp	200	0.502	0.232	377	0.148	0.022	0.012
heisel	144	0.345	0.059	603	0.267	0.174	0.055
highbank	235	0.483	0.059	166	0.284	0.043	0.025
innis	69	0.324	0.022	959	0.455	0.062	0.034
kilbourn	106	0.462	0.299	115	0.163	0.020	0.020
klondike	88	0.246	0.535	107	0.122	0.025	0.021
lock	256	0.236	0.391	333	0.118	0.018	0.014
lou	156	0.113	0	2272	0.277	0.234	0.079
ngalena	135	0.543	0.363	21	0.046	0.011	0.010
olentan	102	0.198	0	1373	0.504	0.146	0.121
osuwet	87	0.110	0	2886	0.347	0.286	0.087
prairie	148	0.285	0.469	58	0.124	0.026	0.022
prindle	158	0.114	0.805	29	0.027	0.014	0.014
pubhunt	194	0.499	0.322	210	0.080	0.008	0.008
redbanks	279	0.528	0.153	140	0.196	0.020	0.020
rocky	150	0.558	0.168	266	0.224	0.016	0.013
rushrun	150	0.316	0	1611	0.409	0.090	0.060
sgalena	163	0.434	0.143	69	0.297	0.017	0.012
smith	144	0.104	0.347	729	0.328	0.087	0.025
sunbury	129	0.290	0.299	500	0.257	0.050	0.028
tnc	292	0.377	0.414	340	0.107	0.028	0.027
westfall	56	0.142	0.772	11	0.027	0.008	0.008
whetston	154	0.196	0	2017	0.456	0.162	0.083
whitehal	106	0.177	0	545	0.405	0.203	0.052
woodside	104	0.250	0.113	1227	0.398	0.067	0.045

Table B.1: Landscape variables recorded as the fraction of the total area within a 1-km radius that is of the given type (with the exception of Forest Width and Buildings which are measured as length and number within the 1-km radius).

Model	K	-2log(L(θ))	AIC_c	Δ_i	w_i
Forest in 1-km	50	15.455	115.97	0	1
Forest in 1-km + Buildings	75	5.05E-05	151.15	35.18	0
Forest in 1-km + Forest Width	75	6.04E-05	151.15	35.18	0
Forest Width + Agriculture	75	7.76E-05	151.15	35.18	0
Forest Width + Buildings	75	0.000108	151.15	35.18	0
Forest in 1-km + Roads	75	0.000114	151.15	35.18	0
Forest Width + Roads	75	0.000115	151.15	35.18	0
Buildings + Paved	75	0.000115	151.15	35.18	0
Forest in 1-km + Mowed	75	0.000124	151.15	35.18	0
Forest Width + Mowed	75	0.000131	151.15	35.18	0
Forest in 1-km + Agriculture	75	0.000152	151.15	35.18	0
Forest in 1-km + Buildings + Paved	100	5.77E-05	202.04	86.07	0
Forest Width + Buildings + Paved	100	9.96E-05	202.04	86.07	0
Agriculture + Buildings + Mowed + Paved	125	0.000104	253.19	137.22	0
Forest Width + Forest in 1-km + Agriculture + Buildings + Mowed + Paved	175	7.78E-05	356.27	240.30	0
Mowed	50	961.056	1061.57	945.60	0
Buildings	50	1310.065	1410.58	1294.61	0
Forest Width	50	4294.43	4394.94	4278.98	0
Agriculture	50	4511.909	4612.42	4496.45	0
Roads	50	8540.749	8641.26	8525.29	0
Null model	25	57408.28	57458.41	57342.44	0

a.

Variable	Σw_i	Average over all models	Average over models with Δ_i <2
Forest in 1-km	1	0.125	0.2
Forest Width	0	0	0
Buildings	0	0	0
Roads	0	0	0
Agriculture	0	0	0
Mowed	0	0	0
Paved	0	0	0

b.

Table B.2: a. AIC_c analysis of multinomial logistic regression models for irreplaceability values calculated for portfolios of size 2 (n=10,001).
b. Sum of the individual variable weights (Σw_i) from the AIC_c analysis in Table B.2a. Average over all models is the average weight of all models containing the given variable. Average over models with Δ_i <2 is the average weight of all models in the set of potential best models containing the given variable.

Model	K	-2log(L(θ))	AIC_c	Δ_i	w_i
Forest in 1-km + Buildings	78	1.23E-06	157.24	0	0.1
Forest Width + Roads	78	4.73E-06	157.24	0	0.1
Buildings + Paved	78	8.79E-05	157.24	0	0.1
Forest Width + Agriculture	78	9.08E-05	157.24	0	0.1
Forest in 1-km + Mowed	78	0.000103	157.24	0	0.1
Forest in 1-km + Forest Width	78	0.000109	157.24	0	0.1
Forest in 1-km + Roads	78	0.000117	157.24	0	0.1
Forest Width + Mowed	78	0.000123	157.24	0	0.1
Forest Width + Buildings	78	0.000165	157.24	0	0.1
Forest in 1-km + Agriculture	78	0.000171	157.24	0	0.1
Forest in 1-km + Buildings + Paved	104	2.65E-09	210.21	52.96	0
Forest Width + Buildings + Paved	104	1.96E-06	210.21	52.96	0
Agriculture + Buildings + Mowed + Paved	130	0.000103	263.45	106.21	0
Forest Width + Forest in 1-km + Agriculture + Buildings + Mowed + Paved	182	0.00014	370.78	213.54	0
Buildings	52	406.975	511.53	354.29	0
Forest in 1-km	52	860.217	964.77	807.53	0
Mowed	52	1598.312	1702.87	1545.62	0
Agriculture	52	1948.76	2053.31	1896.07	0
Forest Width	52	2805.699	2910.25	2753.01	0
Roads	52	8569.17	8673.72	8516.48	0
Null model	26	52798.1	52850.24	52693	0

a.

Variable	Σw_i	Average over all models	Average over models with Δ_i <2
Forest Width	0.5	0.0625	0.1
Forest in 1-km	0.5	0.0625	0.1
Buildings	0.3	0.0375	0.1
Roads	0.2	0.0667	0.1
Mowed	0.2	0.04	0.1
Agriculture	0.2	0.04	0.1
Paved	0.1	0.02	0.1

b.

Table B.3: a. AIC_c analysis of multinomial logistic regression models for irreplaceability values calculated for portfolios of size 3 (n=10,005).
b. Sum of the individual variable weights (Σw_i) from the AIC_c analysis in Table B.3a. Average over all models is the average weight of all models containing the given variable. Average over models with Δ_i <2 is the average weight of all models in the set of potential best models containing the given variable.

Model	K	-2log(L(θ))	AIC_c	Δ_i	w_i
Forest in 1-km	68	25.813	162.76	0	1
Buildings + Paved	102	6.23E-07	206.12	43.36	0
Forest in 1-km + Forest Width	102	1.92E-05	206.12	43.36	0
Forest Width + Agriculture	102	4.44E-05	206.12	43.36	0
Forest Width + Mowed	102	4.85E-05	206.12	43.36	0
Forest Width + Roads	102	7.24E-05	206.12	43.36	0
Forest in 1-km + Roads	102	0.000131	206.12	43.36	0
Forest in 1-km + Buildings	102	0.000145	206.12	43.36	0
Forest in 1-km + Agriculture	102	0.000146	206.12	43.36	0
Forest in 1-km + Mowed	102	0.000159	206.12	43.36	0
Forest Width + Buildings	102	0.000159	206.12	43.36	0
Forest Width + Buildings + Paved	136	2.48E-06	275.78	113.02	0
Forest in 1-km + Buildings + Paved	136	0.000119	275.78	113.02	0
Agriculture + Buildings + Mowed + Paved	170	8.58E-05	345.91	183.16	0
Forest Width + Forest in 1-km + Agriculture + Buildings + Mowed + Paved	238	1.31E-05	487.65	324.89	0
Buildings	68	668.679	805.62	642.87	0
Mowed	68	1662.058	1799.00	1636.25	0
Forest Width	68	3499.723	3636.67	3473.91	0
Agriculture	68	5109.35	5246.3	5083.54	0
Roads	68	9433.371	9570.32	9407.56	0
Null model	34	58722.38	58790.62	58627.86	0

a.

Variable	Σw_i	Average over all models	Average over models with Δ_i <2
Forest in 1-km	1	0.125	0.2
Forest Width	0	0	0
Buildings	0	0	0
Agriculture	0	0	0
Roads	0	0	0
Mowed	0	0	0
Paved	0	0	0

b.

Table B.4: a. AIC_c analysis of multinomial logistic regression models for irreplaceability values calculated for portfolios of size 4 (n=10,002).
b. Sum of the individual variable weights (Σw_i) from the AIC_c analysis in Table B.4a. Average over all models is the average weight of all models containing the given variable. Average over models with Δ_i <2 is the average weight of all models in the set of potential best models containing the given variable.

Model	K	-2log(L(θ))	AIC_c	Δ_i	w_i
Buildings + Paved	102	0	206.12	0	0.1
Forest Width + Agriculture	102	9.95E-05	206.12	0	0.1
Forest in 1-km + Roads	102	0.000106	206.12	0	0.1
Forest in 1-km + Forest Width	102	0.000112	206.12	0	0.1
Forest Width + Buildings	102	0.000132	206.12	0	0.1
Forest in 1-km + Agriculture	102	0.000138	206.12	0	0.1
Forest in 1-km + Mowed	102	0.00014	206.12	0	0.1
Forest Width + Mowed	102	0.000149	206.12	0	0.1
Forest in 1-km + Buildings	102	0.000166	206.12	0	0.1
Forest Width + Roads	102	0.000197	206.12	0	0.1
Forest Width + Buildings + Paved	136	9.87E-05	275.78	69.66	0
Forest in 1-km + Buildings + Paved	136	0.000117	275.78	69.66	0
Forest in 1-km	68	184.437	321.38	115.26	0
Agriculture + Buildings + Mowed + Paved	170	3.45E-06	345.92	139.79	0
Forest Width + Forest in 1-km + Agriculture + Buildings + Mowed + Paved	238	1.82E-06	487.66	281.53	0
Buildings	68	807.886	944.83	738.71	0
Mowed	68	1968.436	2105.38	1899.26	0
Forest Width	68	4086.943	4223.89	4017.77	0
Agriculture	68	5488.29	5625.24	5419.11	0
Roads	68	11348.52	11485.47	11279.34	0
Null model	34	61103.99	61172.23	60966.11	0

a.

Variable	Σw_i	Average over all models	Average over models with Δ_i <2
Forest in 1-km	0.5	0.0625	0.1
Forest Width	0.5	0.0625	0.1
Buildings	0.3	0.0375	0.1
Agriculture	0.2	0.04	0.1
Mowed	0.2	0.04	0.1
Roads	0.2	0.0667	0.1
Paved	0.1	0.02	0.1

b.

Table B.5: a. AIC_c analysis of multinomial logistic regression models for irreplaceability values calculated for portfolios of size 5 (n=9,999).
b. Sum of the individual variable weights (Σw_i) from the AIC_c analysis in Table B.5a. Average over all models is the average weight of all models containing the given variable. Average over models with Δ_i <2 is the average weight of all models in the set of potential best models containing the given variable.

Model	K	-2log(L(θ))	AIC_c	Δ_i	w_i
Forest in 1-km	68	0.00305	136.95	0	1
Forest Width + Mowed	102	1.23E-09	206.12	69.17	0
Forest Width + Roads	102	8.69E-09	206.12	69.17	0
Buildings + Paved	102	1.44E-06	206.12	69.17	0
Forest Width + Agriculture	102	8.15E-05	206.12	69.17	0
Forest Width + Buildings	102	0.000118	206.12	69.17	0
Forest in 1-km + Buildings	102	0.000126	206.12	69.17	0
Forest in 1-km + Agriculture	102	0.000154	206.12	69.17	0
Forest in 1-km + Forest Width	102	0.00017	206.12	69.17	0
Forest in 1-km + Roads	102	0.00018	206.12	69.17	0
Forest in 1-km + Mowed	102	0.000193	206.12	69.17	0
Forest Width + Buildings + Paved	136	0	275.78	138.83	0
Forest in 1-km + Buildings + Paved	136	2.32E-05	275.78	138.83	0
Agriculture + Buildings + Mowed + Paved	170	0.000197	345.92	208.97	0
Forest Width + Forest in 1-km + Agriculture + Buildings + Mowed + Paved	238	2.28E-07	487.656	350.71	0
Buildings	68	811.439	948.38	811.44	0
Mowed	68	2269.501	2406.45	2269.5	0
Forest Width	68	4202.022	4338.97	4202.02	0
Agriculture	68	7671.305	7808.25	7671.3	0
Roads	68	10958.99	11095.93	10958.99	0
Null model	34	63082.84	63151.08	63014.13	0

a.

Variable	Σw_i	Average over all models	Average over models with Δ_i <2
Forest in 1-km	1	0.125	0.2
Forest Width	0	0	0
Buildings	0	0	0
Roads	0	0	0
Mowed	0	0	0
Agriculture	0	0	0
Paved	0	0	0

b.

Table B.6: a. AIC_c analysis of multinomial logistic regression models for irreplaceability values calculated for portfolios of size 6 (n=10,000).
b. Sum of the individual variable weights (Σw_i) from the AIC_c analysis in Table B.6a. Average over all models is the average weight of all models containing the given variable. Average over models with Δ_i <2 is the average weight of all models in the set of potential best models containing the given variable.

Model	K	-2log(L(θ))	AIC_c	Δ_i	w_i
Forest in 1-km	68	1.091	138.04	0	1
Forest Width + Mowed	102	1.69E-08	206.12	68.08	0
Buildings + Paved	102	4.82E-06	206.12	68.08	0
Forest in 1-km + Forest Width	102	7.7E-05	206.12	68.08	0
Forest Width + Roads	102	8.44E-05	206.12	68.08	0
Forest Width + Buildings	102	0.000115	206.12	68.08	0
Forest in 1-km + Agriculture	102	0.000116	206.12	68.08	0
Forest in 1-km + Buildings	102	0.000132	206.12	68.08	0
Forest Width + Agriculture	102	0.000139	206.12	68.08	0
Forest in 1-km + Roads	102	0.000171	206.12	68.08	0
Forest in 1-km + Mowed	102	0.000184	206.12	68.08	0
Forest Width + Buildings + Paved	136	6.53E-05	275.78	137.74	0
Forest in 1-km + Buildings + Paved	136	0.000104	275.78	137.74	0
Agriculture + Buildings + Mowed + Paved	170	0.000111	345.92	207.88	0
Forest Width + Forest in 1-km + Agriculture + Buildings + Mowed + Paved	238	5.24E-14	487.66	349.62	0
Buildings	68	721.264	858.21	720.17	0
Mowed	68	2401.86	2538.81	2400.77	0
Forest Width	68	4193.364	4330.31	4192.27	0
Agriculture	68	7444.029	7580.97	7442.94	0
Roads	68	13274.78	13411.73	13273.69	0
Null model	34	64376.42	64444.66	64306.62	0

a.

Variable	Σw_i	Average over all models	Average over models with Δ_i <2
Forest in 1-km	1	0.125	0.2
Forest Width	0	0	0
Buildings	0	0	0
Mowed	0	0	0
Agriculture	0	0	0
Roads	0	0	0
Paved	0	0	0

b.

Table B.7: a. AIC_c analysis of multinomial logistic regression models for irreplaceability values calculated for portfolios of size 7 (n=9,999).
b. Sum of the individual variable weights (Σw_i) from the AIC_c analysis in Table B.7a. Average over all models is the average weight of all models containing the given variable. Average over models with Δ_i <2 is the average weight of all models in the set of potential best models containing the given variable.

Model	K	-2log(L(θ))	AIC_c	Δ_i	w_i
Forest Width + Mowed	102	1.84E-05	206.12	0	0.1
Forest Width + Roads	102	4.29E-05	206.12	0	0.1
Buildings + Paved	102	5.63E-05	206.12	0	0.1
Forest in 1-km + Forest Width	102	9.23E-05	206.12	0	0.1
Forest Width + Agriculture	102	9.68E-05	206.12	0	0.1
Forest in 1-km + Buildings	102	0.000106	206.12	0	0.1
Forest in 1-km + Mowed	102	0.000115	206.12	0	0.1
Forest Width + Buildings	102	0.000115	206.12	0	0.1
Forest in 1-km + Agriculture	102	0.000118	206.12	0	0.1
Forest in 1-km + Roads	102	0.000199	206.12	0	0.1
Forest Width + Buildings + Paved	136	7.81E-08	275.78	69.65	0
Forest in 1-km + Buildings + Paved	136	0.000141	275.78	69.65	0
Agriculture + Buildings + Mowed + Paved	170	0.000123	345.91	139.79	0
Forest in 1-km	68	243.7861	380.73	174.61	0
Forest Width + Forest in 1-km + Agriculture + Buildings + Mowed + Paved	238	5.2E-06	487.65	281.53	0
Buildings	68	890.0025	1026.95	820.82	0
Mowed	68	2347.429	2484.37	2278.25	0
Forest Width	68	4300.007	4436.95	4230.83	0
Agriculture	68	8527.533	8664.48	8458.36	0
Roads	68	9463.899	9600.84	9394.72	0
Null model	34	65618.2	65686.44	65480.32	0

a.

Variable	Σw_i	Average over all models	Average over models with Δ_i <2
Forest Width	0.5	0.0625	0.1
Forest in 1-km	0.5	0.0625	0.1
Buildings	0.3	0.0375	0.1
Mowed	0.2	0.04	0.1
Agriculture	0.3	0.04	0.1
Roads	0.2	0.0667	0.1
Paved	0.1	0.02	0.1

b.

Table B.8: a. AIC_c analysis of multinomial logistic regression models for irreplaceability values calculated for portfolios of size 8 (n=10,001).
b. Sum of the individual variable weights (Σw_i) from the AIC_c analysis in Table B.8a. Average over all models is the average weight of all models containing the given variable. Average over models with Δ_i <2 is the average weight of all models in the set of potential best models containing the given variable.

APPENDIX C

PARTNERS IN FLIGHT SPECIES SCORES

Species Common Name	PS-g	BD-g	TB-r	PT-r	RD-b	RCS-b
Turkey Vulture	3	1	1	1	2	8
Cooper's Hawk	3	1	2	1	4	11
Red-tailed Hawk	3	1	1	1	4	10
Mourning Dove	1	1	1	2	5	10
Yellow-billed Cuckoo	2	1	4	4	4	15
Barred Owl	3	1	4	2	3	13
Chimney Swift	2	1	3	4	5	15
Ruby-throated Hummingbird	2	1	3	2	3	11
Belted Kingfisher	3	1	3	3	4	14
Red-bellied Woodpecker	2	2	2	1	3	10
Downy Woodpecker	2	1	2	2	4	11
Hairy Woodpecker	2	1	2	3	3	11
Northern Flicker	2	1	4	5	4	16
Pileated Woodpecker	3	1	4	1	2	11
Eastern Wood-Pewee	2	1	4	2	4	13
Acadian Flycatcher	3	2	4	5	2	16
Eastern Phoebe	2	1	3	1	4	11
Great Crested Flycatcher	2	1	4	4	3	14
Eastern Kingbird	2	1	3	4	5	15
White-eyed Vireo	2	2	3	3	1	11
Yellow-throated Vireo	3	2	3	1	3	12
Warbling Vireo	2	1	3	1	3	10
Red-eyed Vireo	1	2	3	2	2	10
Blue Jay	2	1	1	4	4	12
American Crow	2	1	1	2	4	10
Northern Rough-winged Swallow	2	1	2	1	5	11
Carolina Chickadee	2	3	2	2	2	11
Tufted Titmouse	2	2	2	2	4	12
White-breasted Nuthatch	2	1	2	1	3	9
Carolina Wren	2	2	2	1	2	9
House Wren	2	1	1	1	5	10
Blue-gray Gnatcatcher	1	1	4	3	2	11
Eastern Bluebird	2	1	3	1	4	11
Veery	2	2	2	3	1	10
Wood Thrush	2	2	5	3	2	14
American Robin	1	1	1	1	5	9
Gray Catbird	2	1	3	3	3	12
Brown Thrasher	2	1	4	4	5	16

Continued

Table C.1: Partners in Flight species assessment scores. The regional combined score for the breeding season (RCS-b) is calculated as the sum of scores for the global population size (PS-g), global breeding distribution (BD-g), regional threats to breeding (TB-r), regional population trend (PT-r) and breeding relative density (RD-b).

Table C.1 continued

European Starling	1	1	1	2	5	10
Cedar Waxwing	2	1	2	1	3	9
Northern Parula	2	2	3	3	2	12
Yellow Warbler	2	1	3	1	2	9
Yellow-throated Warbler	3	3	3	3	2	14
Prairie Warbler	3	3	4	3	1	14
Prothonotary Warbler	3	3	4	3	2	15
Ovenbird	2	2	4	3	1	12
Louisiana Waterthrush	4	2	4	3	2	15
Common Yellowthroat	2	1	2	4	4	13
Summer Tanager	3	2	3	1	2	11
Scarlet Tanager	3	2	3	3	2	13
Eastern Towhee	2	2	3	3	2	12
Chipping Sparrow	1	1	1	1	4	8
Field Sparrow	2	2	4	5	4	17
Song Sparrow	1	1	2	2	4	10
Northern Cardinal	1	1	1	2	4	9
Rose-breasted Grosbeak	3	2	3	1	4	13
Indigo Bunting	2	1	2	2	4	11
Red-winged Blackbird	1	1	2	4	5	13
Common Grackle	1	1	1	3	5	11
Brown-headed Cowbird	1	1	1	2	5	10
Baltimore Oriole	2	2	3	2	5	14
House Finch	2	1	1	1	2	7
American Goldfinch	2	1	2	4	4	13
House Sparrow	1	1	1	5	5	13