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Wildwood/Kraemer Lake County Park Observational Study: The Effects of Fragmentation on Old-Growth Forest Species

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WILDWOOD/ KRAEMER LAKE COUNTY PARK OBSERVATIONAL STUDY

THE EFFECTS OF FRAGMENTATION ON

OLD-GROWTH FOREST SPECIES

by

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WILDWOOD/ KRAEMER LAKE COUNTY PARK OBSERVATIONAL STUDY
THE EFFECTS OF FRAGMENTATION ON
OLD-GROWTH FOREST SPECIES

Rhonda H. Zimmerman

Habitat fragmentation generates forest fragments with increased ratio of "edge". This "edge effect" alters the natural community. Old growth forests support deep forest communities which rely on the closed canopy for survival, often indicators of healthy old growth communities. Putative old growth forest patches were identified in Wildwood Park and St. John's Arboretum/University. One study objective was to collect baseline data on flora and fauna in Wildwood to confirm the presence of old-growth patches. The main ecological goal of the study was to determine if bird and plant communities exhibited patterns consistent with the operating assumption of 50 acre old-growth relics surrounded by forest buffers of varying size, and if so, whether the communities in the buffers followed expectations of ecological theory. Birds and plants known to associate with old growth habitats were identified at Wildwood and then compared to similar habitats at St. John's. The research included three sites, one at Wildwood (WW) and two at St. John's (SJA & SJU), a total of 14 plots in all. Each site had a different amount of plots depending on the size of buffer zone at that site. After surveying the plots, we compared richness between the different zones (core & buffer) to determine if old growth species richness is higher in old growth core areas of the forest than the surrounding successional buffer that isolates the core from the "edge effect". Richness was inconclusive with no statistical significance seen; possibly due to the small sample size or unforeseen weather events. The strongest patterns observed (via ordination analyses) were when vegetative cluster analysis and canonical correspondence were run. Cluster analysis indicated four to five of the six old growth plots clustered in all seasons. Canonical correspondence analysis were performed, plotting CCA Axis 1 vs Axis 2; generating graphs with the points (samples) labeled three different ways (site, zone and season). In each case there appeared visually to be patterning on either Axis 1 or 2 (or both), so we ran a series of analyses of variance to determine whether these patterns were statistically significant. One-way ANOVA's indicated statistically significant differences (by the Tukey's Test) in seasons, zones and sites. The most robust patterns occurred when comparing site differences. This study found strong vegetative diversity patterns among seasons, zones and sites with what appears to be no difference between the sizes of the buffer zones but rather the land management practices.

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STATEMENT

Trust in the LORD with all your heart
and lean not on your own understanding;
in all your ways acknowledge Him
and He shall direct your paths.

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INTRODUCTION

Habitat fragmentation and loss are thought to be the main factors in species decline (Groombridge 1992; Faaborg *et al*. 1993; Bibby 1995; Ehrlich 1995; Thomas & Morris 1995; Rappole 1996). Many studies have found that loss of habitat negatively impacts a variety of species on a global scale (Groombridge 1992; Bibby 1995; Ehrlich 1995; Thomas & Morris 1995), is a well-documented recognized threat to biodiversity (Zipperer 1993; Scott & Pratini 1995; Berglund & Jonsson 2001; Ewers, Thorpe, & Didham 2011; Jamoneau *et al*. 2011) and a central issue in habitat conservation (Meffe & Carroll 1997; Trombulak 2004). However, the verbiage of "habitat loss" is specific to the type (or area) of habitat the researcher is focused on. Habitat loss of one type or site implies habitat gain for another (Rolstad 1991) thus changing the dynamics of the habitat (Schmeigelow & Monkkonen 2002). A greater decline in species can be seen when compounding the effects of habitat loss with "fragmentation". Loss of habitat and patch fragmentation each effect patch biodiversity independently (Fahrig 2003).

Habitat fragmentation is when "a large expanse of habitat is transformed into a number of smaller patches of smaller total area, isolated from each other by a matrix of habitats unlike the original" (Wilcove, McLellan, & Dobson 1986), converting "large areas of contiguous native forest to other types of vegetation and /or land use leaving remnant patches of forest that vary in size and isolation" (FD & FAO 2007) either by natural disturbance or human activity (Dale & Pearson 1997). Forest habitat fragmentation alters vegetative dynamics, impairs the functionality of the ecosystem, and diminishes the habitat's ability to support plant biodiversity (Ewers *et al*. 2011). The modern approach to habitat fragmentation (Faaborg *et al*. 1993) stems from MacArthur and Wilson's theory of island biogeography concerning habitat isolation and connectivity (MacArthur & Wilson 1963;

MacArthur & Wilson 1967), and has been studied in depth for over 40 years (Robinson 1992).

MacArthur and Wilson's original theory studied how organisms must cross the water (a dispersal

barrier) prior to inhabiting oceanic islands. The inability of a species to cross this oceanic barrier often

results in spatial isolation. The island's size and isolation are often found to be important predictors in

the likelihood of colonization. In recent years this general theory has been applied to fragmentation in

terrestrial habitats with isolation becoming distance from one patch to the next instead of the

distance from the island to the mainland (Haila 2002).

According to Zipperer, habitat "fragmentation is only one pattern of the process" known as

'deforestation'. Deforestation is "the conversion of forested areas to non-forest land use such as

arable land, urban use, logged area or wasteland" (FD & FAO 2007), and includes five specific patterns

of vegetative habitat loss: internal, indentation, cropping, fragmentation, and removal (Zipperer

1993).

- Internal habitat loss represents less internal core habitat and internal core species.

- Indentation, cutting into the habitat, creates travel corridors that may benefit plant

    species preferring edge (Zipperer 1993), increased light availability, and xeric

    microclimates (Wales 1972).

- Cropping, the removal of the extensions created above by indentation, creates smaller

    patches, adding to the matrix, and may disrupt a species ability to disperse (Zipperer

    1993).

- Removal means the habitat of original is no longer in its original condition (Zipperer

    1993) having been converted to the buffer state or some other intermediate condition.

- Fragmentation, the final deforestation pattern, is when the original patch is converted

    into smaller, less contiguous patches, thus creating more edge (Zipperer 1993).

Fragmentation results in a complex landscape interaction between multivariate elements

exclusively "reserved for changes in habitat configuration that result from the breaking apart of

habitat, independent of habitat loss" (Fahrig 2003), thus including: patch size, corridors, isolation, edge effects and ratio of size to edge (Bender, Contreras, & Fahrig 1998; Willson *et al*. 2000; Schmeigelow & Mönkkönen 2002; Fahrig 2003). The fragmented forests that remain are often smaller in size, thus excluding species that prefer interiors or those with larger territorial requirements, along with having more "edge".

Edge Effects

"Edges can be defined as the boundary between patches with differing qualities; thus, the identification of edges will depend on how researchers define patches" (Strayer, Powers, & Fagan 2003). Edge effects are some of the most thoroughly studied ecologically processes (Ries & Sisk 2004) and yet are difficult to study as there is a strong correlation between size and edge (Moen & Jonsson 2003; Ewers, Thorpe, & Didham 2007). Changes in edge amount often have ensuing effects on the plant species as different plant species respond differently to edge effects (Zipperer 1993). As fragmentation increases, so does edge amount with the total edge length of the individual patches often exceeding the length of the original patch, thus creating a need for more core habitat as the utility of the core changes (Zipperer 1993).

Creating edge provides corridors for invasive species, predators and humans, increases exposure to wind and light, as-well-as decreasing the amount of humidity to that habitat. Several studies have shown edge effects to be favorable for certain wildlife (Lay 1938; Johnston 1947). Some species responded to round fragments as they would to interior habitats, while other species react to small fragments as if to edge, concluding effects on the species to size and shape of the fragment is species specific with limitations on extrapolating from site studies to general forestry management (Moen & Jonsson 2003).

Much research has focused on edge effects in landscapes dominated by edges created between forested areas and agriculture land (Paton 1994), while less attention has been paid to edges created between mature/old-growth forests adjoining habitats dominated by younger to mid-

successional regenerating forests. In this study, edges created between old-growth forest land (core areas) abutting vegetation < 2 m high will be defined as 'hard edges' (Fenske-Crawford & Niemi 1997) and edges created between core areas abutting vegetation ≥ 2 m high will be delineated as 'soft edges'. This study hopes to show that a habitat with softer edges will provide a buffer that will actually benefit the old-growth habitat species, allowing for greater biodiversity.

Avian Habitat Loss

Habitat loss can be viewed as more than the removal of the habitat being studied; habitat is also "lost" through the breaking apart of habitat, "fragmentation" (Fahrig 2003). Fragmenting landscapes creates a larger amount of smaller, isolated patches along with changing the habitat's characteristics (van den Berg *et al*. 2001; Fahrig 2003). Studies found fragmentation and patch size to be detrimental to birds, leading to high rates of nest predation, parasitism, and decrease in breeding success (Temple & Cary 1984; Small & Hunter 1988; Askins, Lynch, & Greenberg 1990; Wilcove & Robinson 1990; Finch 1991; Robinson & Wilcove 1994), with patch sizes eventually becoming too small to support a species as a single territory or as a local population (Fahrig 2003). Other studies found the effects of patch size to be small or to have a positive effect on bird species abundance (Sallabanks, Walters, & Collazo 2000).   When considering how much habitat is required, the habitat type and the type of species must also be considered.   Before applying conservation management to an area, identifying the species in that region with the greatest vulnerability to habitat loss is paramount. Once accomplished, the amount of habitat needed for those most vulnerable species (Fahrig 2001; With & King 1999) can be estimated and policies should then be written for protecting large, unfragmented 'core' areas (Robinson *et al*. 1995).

Avian Edge Effect

Many studies have been done on nest predation and proximity to habitat edge; however, most of these studies have focused on edges that are abrupt, 'hard', (Fenske-Crawford & Nieme 1997)

such as those created by between wooded areas and agricultural lands (Paton 1994). Forest management has major influences on landscape patterns often generating mosaics of successional forest stages (Hanski, Fenske, & Niemi 1996) and creating habitat edges that are less abrupt, 'softer'(Fenske-Crawford & Nieme 1997).

Hard forest edges often result in decreased breeding success due to increased parasitism and predation (Brittingham & Temple 1983; Andre'n & Angelstam 1988; Andre'n 1992; Nour, Matthysen & Dhondt 1993; Paton 1994; Hoover, Brittingham, & Goodrich 1995). Some studies indicate decreased nest success in forests fragmented by clear cuts, in young regenerating forests (Yahner & Wright 1985; Storch 1991; Yahner 1991, Rudnicky & Hunter 1993) and within forest-dominated landscapes (Chasko & Gates 1982; Small & Hunter 1988). In contrast, other studies find no edge effect in forest-dominated habitats (Boag, Reebs, & Schroeder 1984; Yahner & Wright 1985; Yahner 1991; Rudnicky & Hunter 1993, Hanski, Fenske, & Niemi 1996) and find reproductive success near the forest edges greater than the interior (Storch 1991).

When a forested habitat is subjected to hard, abrupt edges, as in agricultural mosaics, biotic (predation, parasitism, anthropogenic disturbances) and abiotic (increased wind disturbance, decreased humidity, increased light intensity) aspects can penetrate into the core environment, thus changing the dynamics of the original habitat. Soft, non-abrupt edges, those created between old-growth and younger successional forests, buffer the effects of the biotic and abiotic factors, thus providing less risk with a matrix more similar to the core habitat.

Due to varying silviculture practices, more studies are needed for better understanding concerning reproductive success and habitat edge types.

Avian Isolation

Isolation can be detrimental to species survival (MacArthur & Wilson, 1967; Shafer, 1990) although there is less data concerning isolated fragments (Askins, Lynch, & Greenberg 1990; Opdam 1991).   Some studies found that more isolated habitats had fewer Neotropic migrant species than

less isolated woodlots of similar characteristics (Lynch & Whigham 1984), while other studies have shown extremely isolate woodlots to contain more forest migrants than was predicted (Robinson 1992).

When patch size becomes too small, inhabitants must disperse. When the distance between patches is large or inhospitable, dispersal may include extreme risk. Consistent declines in some bird populations and local extinction in others have been seen in long term bird studies correlating small fragments with isolation (Finch 1991). An on-going study at the Connecticut Arboretum (Askins & Philbrick 1987; Askins, Philbrick, & Sugeno 1987) found a decline in Neotropical migrant breeding populations when new roads, shopping centers, and highway interchanges were constructed near the arboretum, isolating the facility. Over the years, isolation of the site has been reduced through regeneration of young successional forests. With the reduction in isolation, young breeding populations of Neotropical migrants have increased in the arboretum. Access to suitable habitat during migration to replenish reserves and rest is essential, thus making corridors and stopover sites indispensable (Lindstrom 1989; Metcalfe & Furness 1984). However, with corridors comes accessibility of outside influences such as predators, unfavorable biotic factors, and disease. Debate over connectivity of isolated patches through corridors continues (Simberloff *et al.* 1992) with patch size and landscape context confounding the argument (Beier & Noss 1998).

<u>Vegetative Habitat Loss</u>

Habitat fragmentation has similar impacts on birds and plants. Fragmented habitats are at greater risk to harmful outside influences such as invasive species (Scott & Pratini 1995), elevated concentrations of shade-intolerant plant species, and more arid microclimates (Wales 1972).

<u>Vegetative Edge Effect</u>

Forest fragmentation is one pattern of deforestation (Zipperer 1993) and includes the process of "degradation that reduces forest quality - the density and structure of the trees, the species

diversity and the genetic diversity" "leading to a 'temporary or permanent deterioration in the density or structure of vegetation cover or its species composition'" (FD & FAO 2007). Fragmenting a forest habitat not only reduces the amount of core habitat, it also forms 'edge' (Yahner 1988). This 'edge' is more than simply a transition between the core and the external matrix; it signifies two environmentally dissimilar habitats and their access or corridors.   These newly created spatial patterns allow for introduction of exotics, growth of xeric communities, domesticated pets and other anthropogenic disturbances (Zipperer 1993), and "spillover" colonization by vegetative species found in the buffer (Cook *et al.* 2002).

Vegetative Isolation

   Isolated patches of forest habitat are created when fragments of forest become disconnected from contiguous forests (Zipperer 1993). In the original island biogeography theory, 'islands' were not subject to colonization from the 'matrix' (Cook *et al*. 2002). When applying island biogeography isolation theory to terrestrial patches, research found fragmented patches can be influenced by colonization from the matrix (Ås 1999).   Both species abundance and composition can be affected by 'spill-over' from immigrating vegetative species found in the buffer (Cook *et al*. 2002)

   When core habitats become too small from fragmentation or edge effects, to sustain a single species or a local population, and dispersal is necessary, species that are unable to cross the "buffer" or "matrix" (non-inhabitable landscapes) between isolated habitats, risk becoming overcrowded, and eventually leading to inbreeding and/or extinction (Fahrig 2003). The ability of flora species to colonize an area is greatly dependent on distance from the source, or ancient woods (Peterken & Game 1984; Dzwonko & Loster 1992; Dzwonko 1993; Matlack 1994; Hermy & Verheyen 2007). Depending on specific seed morphology and scattering activities of birds or mammals, dispersal is frequently slow (Brunet & van Oheimb 1998). Due to the long distance seeds need to travel, those that are eaten or adhesive appear to be the best for migrating between woodlots (Dzwonko & Loster 1992; Dzwonko 1993; Matlack 1994). If seeds are not dispersed, often merely a few meters by ants,

they basically fall to the ground directly beneath the parent plant and decompose or lay dormant for years, limiting their distribution (Culver & Beattie, 1978; Bierzychudek 1982; Handel, Fisch, & Schatz 1981; Kjellsson, 1985; Hermy & Verheyen 2007). Some species are disturbance dependent and need a significant disturbance to recolonize an area. Those that lack the ability to disperse long-distances may be unable to become established between disturbance events (Dzwonko & Loster 1992; Dzwonko 1993; Matlack 1994). This decreased dispersal ability to colonize may affect the rate at which secondary succession proceeds (Dzwonko & Loster 1992) and ultimately change the dynamics and succession stages of vegetative habitats, thus finding dispersal limitations (seed dispersal morphology and ability of animal) and distance between patches (amount of isolation) an important factor in determining the structure of herb communities (Ehrle'n & Eriksson 2000).

<u>Old Growth Forest</u>

Late successional stage redwood and Douglas-fir forests in the Pacific Northwest are recognized for their species diversity, habitat structures, endemic plant and animal communities and ecological processes that are not established in younger, successional forests (Noss 1993). A substantial proportion of our forest vegetation is related to ancient forests (Hermy & Verheyen 2007).

An 1857 Ohio Senate committee decline a proposed protection bill for the Passenger Pigeon (*Ectopistes migratorius*): "The Passenger Pigeon needs no protection. Wonderfully prolific, having the vast forests of the North as its breeding grounds, traveling hundreds of miles in search of food, it is here to-day and elsewhere to-morrow, and no ordinary destruction can lessen them, or be missed from the myriads that are yearly produced." Unfortunately they were wrong. The last known living passenger pigeon died in 1914 (Jackson 1988) and the "vast forests of the North" in Ohio are also gone. Habitat loss has been significant over the last 150 years.

A substantial proportion of our forest vegetation is derived from ancient forests (Hermy & Verheyen 2007). As per Sutherland, old-growth forests are not defined by simply the age of the trees but by their function, structure and composition (2005). They contain multi-layered canopies, shade-

tolerant species, large standing snags and downed logs, and numerous stages of decomposition. Simply put, "suitability of conditions for tree growth" (Noss 1993). Over time old-growth stands increase in vascular plant diversity with peaks in old-growth forests (Halpern & Spies 1995), and with vegetative species, abundance is generally found to increase with an increase in patch size (Rosenzweig 1995).

Studies in Sweden and Finland have shown over time old-growth forest land has become greatly reduced, fragmented and deteriorated (Pulliam 1988; Berglund & Jonsson 2001). Along with the loss, fragmentation, and isolation of old-growth forests, comes the decline of species biodiversity that rely on old-growth habitats. According to a 2001 Swedish study by Berglund and Jonsson, the greatest threat to biodiversity and Threatened and Endangered (T & E) species is "the continuing fragmentation and further deterioration of the old-growth forest habitat". Due to the decrease in habitat amounts, and the impacts of global climate change, there has been increasing concern over whether or not refuge managers are protecting the biodiversity of organisms in their regions (Gaston 2008). Studies are now being done on what is termed "extinction debt", the number of species likely to go extinct in the future (Berflund & Jonsson 2005). Some organisms may be able to exist in a present day fragment while experiencing a long slow decline. Decline can often be seen in species more sensitive to habitat change, slower to reach reproductive maturity, colonial, are longer lived or require larger habitats. Current populations of these individuals are more likely to reflect habitats of the past and not of the future, thus making them prone to extinction (Sang *et al*. 2010).

<u>Old Growth Forest in Minnesota</u>

Old growth forests in the state are in peril with only "about 4% (610,800 acres) of current forest are old or old growth, compared with 51% in presettlement times" (Biodiversity 1992; MN DNR 2002). In Minnesota alone, forest acreage has fallen by 43%, old-growth forests shrank by almost 95%, with the Boundary Waters Canoe Area Wilderness containing the majority of the state's old-

growth (MN DNR, 2002). Even less habitats meet the criteria of 'old forest', that is "of natural origin with little evidence of logging" (MN DNR 2002).

Ancient forests vary considerably from region to region. An old-growth forest in Washington State looks quite different from one in northern Maine. Even within one specific area, like the state of Minnesota, several old-growth community types may be seen. "There is no generally accepted or universally applicable definition of old growth" (Hunter 1989) and that "specifying exact age ranges for late-successional and old-growth forest is impossible because of variations in climate, soil quality, disturbances, and numerous other factors" (Tuchmann *et al.* 1996). Hunter gives a general conceptual definition: "old-growth forests are relatively old and relatively undisturbed by humans".

Old-growth forests are characterized by groups of plants that interact with each other in their environment forming native plant communities (a.k.a. native habitats or natural communities). Native plant communities are those that have not been greatly altered by humans (less susceptible to the colonization of invasive species) and can interact with each other forming units that are unique yet somewhat predictable.

The Minnesota Department of Natural Resources (MNDNR) defines an "old-growth forest" in Minnesota is a stand of trees, at least 120 years old, which has developed naturally over time (2002). They generally experience few catastrophic events such as logging, with fire occurring less than every once 1,000 years, and the rotation of windthrows approximately every 380 years, with events resulting in partial tree canopy loss approximately every 40 years (MN DNR 2005).

A typical Minnesota old-growth forest contains snags (large dead standing trees), large fallen trees, and tip-up mounds or pits (piles of dirt and holes from where large trees have fallen) (MN DNR 2002). Some examples of Minnesota old-growth forest communities include: northern hardwoods, lowland hardwoods, white cedar and red/white pine (MN DNR 2002). These forests are usually dominated by sugar maples (*Acer saccharum*), white spruce (*Picea glauca*), white cedar (*Eastern Arborvitae, Northern Whitecedar)*, northern red oak (*Quercus rubra*), red pine (*Pinus resinosa)* or

white pine *(Pinus strobes)*. Maple and spruce trees reproduce under the shade of the canopy, however, oak and pine generally need a catastrophic disturbance to regenerate.

The MNDNR categorizes the forests found in central Minnesota (study area) and Northeast Iowa Morainal Section as "Central Mesic Hardwood Forest, Eastern, MHc36" (Figure 1) emerging from moraines formed by the Wisconsin glacier with soils hummocky and firm, consisting of noncalcareous glacial till (MN DNR 2005). Historically the forests have two growth phases with one transitional phase. From 0-35 years, the young forest is recovering from the disturbance and is dominated by northern red oak, quaking aspen (Populus tremuloides), big-toothed aspen (Populus grandidentata*)* and American basswood (*Tilia americana).* For the next 60 years, a transitional phase takes place. Northern red oak, quaking aspen, big-toothed aspen and paper birch decline while American basswood, American elm *(Ulmus Americana),* Hophornbeam ironwood (*Ostrya virginiana),* bur oak *(Quercus macrocarpa*) and white pine all increase with bur oak and white pines becoming established in the understory (MN DNR 2005)*.* Once the forest reaches 95+ years of age without undergoing a major disturbance, it is "mature", experiencing >75% continuous canopy coverage of basswood, American elm, northern red oak, and ironwood with few bur oak and white pines (MN DNR 2005).

The herbaceous vegetative structures have high variability with common species including early meadow-rue *(Thalictrum dioicum)*,lady fern *(Athyrium filix-femina),* large flowered bellwort *(Uvularia grandiflora),*Clayton's sweet cicely *(Osmorhiza claytonii),*Pennsylvania sedge *(Carex pensylvanica),*   large-leaved aster *(Aster macrophyllus)*, wild sarsaparilla *(Aralia nudicaulis),* zigzag goldenrod *(Solidago flexicaulis),* yellow violet *(Viola pubescens)*, chokecherry *(Prunus virginiana),*pagoda dogwood *(Cornus alternifolia),* prickly gooseberry *(Ribes cynosbati),* and beaked hazelnut *(Corylus cornuta)*. Individual forests vary with the above description a general forest composition in a MHc36 (Table 1) (MN DNR 2005).
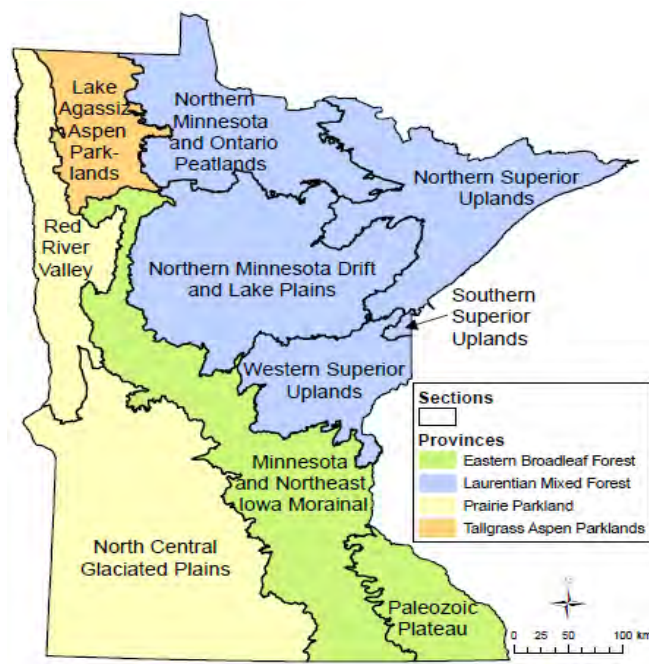
Figure 1. MNDNR classification of native plant communities of Minnesota (MN DNR, 2005).

Table 1. A list of species generally occurring in Central Mesic Hardwood Forests in Minnesota (MN DNR 2005).

Indicator Species
Blue cohosh (*Caulophyllum thalictroides)*
Maidenhair fern (*Adiantum pendatum*)
Wild geranium (*Geramium Maculatum*)
Interrupted Fern (*Osmunda claytoniana*)
Nodding fescue (*Festuca subverticilate*)
Bearded shorthusk (*Brachyelytrum erectum*)
Large-flowered trillium (*Trillium grandiflorum*)
Wild ginger (*Asarum canadense*)
Bloodroot (*Sanguinaria canadensis*)
Jack-in-the-pulpit (*Arisaema triphylium*)
Lopseed (*Phryma leptostachya*)
Common enchanter's nightshade (*Circaea lutetiana*)
Leatherwood (*Dirca palustris*)
Zigzag goldenrod (*Solidago flexicaulis*)
Long-stalked sedge (*Carex pedunculate*)
Large-leaved aster (*Aster macrophyllus*)
Pointed-leaved tick trefoil (*Desmodium glutinosum*)
Meadow-rue (*Thalictrum dioicum*)
Lady fern (*Athyrium filix-demina*)
Large-flowered bellwort (*Uvularia grandiflora*)
Clayton's sweet cicely (*Osmorhiza claytonii*)
Pennsylvania sedge (*Carex pensylvanica*)
Wild sarsaparilla (*Aralia nudicaulis*)
Yellow violet (*Viola pubescens*)
Chokecherry (*Prunus viginiana*)
Pagoda dogwood (*Cornus alternifolia*)
Prickly gooseberry (*Ribes cynosbati*)
Beaked hazelnut (*Corylus cornuta*)
Quaking aspen (*Populus tremuloides*)
Northern red oak (*Quercus rubra*)
Big-toothed aspen (*Populus grandidentata*)
Paper birch (*Betula papyrifera*)
American elm (*Ulmus Americana*)
Hoghornbeam ironwood (*Ostrya virginiana*)
Bur oak (*Quercus macrocarpa*)
Eastern white pine (*Pinus strobus*)
Sugar maple (*Acer saccharum*)
American basswood (*Tilia americana*)
Red maple (*Acer rubrum*)
American hornbeam (*Carpinus caroliniana*)
Bitternut hickory (*Carya cordiformis*)

<u>Management</u>

Increasingly studies have shown that when developing management plans, multiple aspects need to be considered: matrix characteristics (Rosenberg & Raphael 1986; Martin & LePart 1989; Saunders 1990), predator types (Hanski, Fenske, & Niemi 1996), and habitat dynamics (Rafe, Usher, & Jefferson 1985; Boecklen 1986; Hanski, Fenske, & Niemi 1996; Sallabanks, Walters, & Collazo 2000). Management plans designed for conservation of a specific species, may overlook the system on which biodiversity is maintained as a whole (Biodiversity, 1992). Landscape attributes, such as degree of isolation and food resources are extremely important when devising management strategies (Rafe, Usher, & Jefferson 1985; Boecklen 1986; Hansen & Urban 1992; Schieck *et al.* 1995). Many species are specific to one ecosystem with narrow ecological tolerances. Species in peril may have already lost genetic diversity and specific niche requirements (Biodiversity 1992).

In a multivariate world, understanding interactive regional factors is important to understanding broad-spectrum habitat dynamics; however isolating site specific local factors are paramount to appropriate management decisions. Management strategies need to include general applications of research data along with how to specifically apply and augment the data as it pertains to the: region, habitat, local migratory patterns, individual populations, and species dynamics. Developing relationships between theory, experiments, and field work is vital (Margules & Austin 1991). A combination of modeling, field work, and experimentation should all be considered in analysis whenever possible (Usher 1987; Murdoch & Walde 1989) and general studies should be used to provide information on species vulnerability and potential conservation concerns (Haila & Hanski 1984; Haila, Hanski, & Raivio 1989). Combining all of this data with habitat type (old-growth, successional, prairie, grasslands, deserts, etc.) and regional characteristics can create a daunting task for perspective managers.

Objective

The principal objective of this observational study (Wolff 2000) is to collect baseline at

Wildwood/Kraemer Lake Stearns County Park, St. Joseph, Minnesota, USA, at St. John's Arboretum,

Stearns County, Collegeville, Minnesota, USA, and at St. John's University, Stearns County, Collegeville,

Minnesota, USA to compare and determine to what extent avian and vegetative deep forest specialists

(specifically T & E and Minnesota' State List of "species of special concern") are utilizing old growth

core sections of these sites. The main focus of this study is to assist in understanding the degree of

importance inhabitants place on the buffer surrounding the core areas.   The three sites differ in the

amount of 'soft edges' created from adjoining landscapes.

Other objectives include:

- Collecting baseline data

- Establishing permanent coordinates in sequestered landscapes for possible future
  studies

- Presence/Absence of species listed on national T & E list or Minnesota's list of "Species of
  Greatest Concern"

- Documenting the species diversity of the core site

- Understanding the species habitat utilization of the old-growth core and surrounding
  buffer

- The collection of data to assist Stearns County Parks in drafting future management
  plans.

Prediction/Hypothesis

The first 50 acre core of old-growth forest at Wildwood/Kraemer Lake Park (WW) is

surrounded by various types and amounts of buffers. To the north there is several acres of soft

successional buffers, grassland and a county highway, to the west, agricultural land and a small stand

of planted pines, to the south, forest regeneration from selectively harvested timber and a small lake, and the to the east, an area of silviculture for the production of maple sugar. In total the buffer area described above totals 265 acres. The second 50 acre core of old-growth forest at St. John's Arboretum (SJA) is surrounded by a relatively consistent 2,500 acre buffer of selectively harvested timber for use on the St. John's University campus. This timber is of varying ages and varieties. Because of the larger contiguous, analogous buffer surrounding the core section in these plots, higher species diversity may be seen. The third 50 acre site, St. John's University (SJU), is located between, and in close proximity to, the other sites and differs only in the amount of buffer surrounding the core sections. This site has minimal to no soft buffer, bordered by Sagatagan Lake, two county roads, and tillable monoculture agricultural land. However, since the three plots are within one mile, avian diversity should be comparable.

Prediction

 Scenario One: Neither Wildwood/Kreamer Lake Park, St. John's Arboretum, nor St. John's University core sections will have mature forest species (bird or vegetative).

 Scenario Two: Wildwood/Kraemer Lake Park, St. John's Arboretum, and St. John's University core sections will all have mature forest species.

 Scenario Three: St. John's Arboretum will have mature forest species in the core area, but Wildwood/Kraemer Lake Park and St. John's University will not.

 Scenario Four: St. John's Arboretum and Wildwood/Kraemer Lake Park will have mature forest species in the core area, but St. John's University will not.

 Null hypothesis is that species richness in mature forest habitat shows no relationship to the amount of young buffer forest surrounding it.

 Alternative hypothesis is that species richness in mature forest habitat does show a relationship to the amount of young buffer forest surrounding it.

I expect to find:

1. There will be a difference in species richness and ground cover between the central older core areas and the outer younger successional buffer areas.

2. The larger the buffer area surrounding the central old growth core area, the greater the species richness in that core area.

3. That species richness will increase from the old growth core areas outward through the successional younger forest buffer zones.

4. The forest fragments of Wildwood/Kreamer Lake Park and St. John's Arboretum will be enough territory to function as usable habitat for species specific to old-growth forests (scenario four).

METHODS

History

This study was initiated in 2012 in response to questions Stearns County Parks Department had concerning future development of Wildwood/Kraemer Lake County Park (WW). Parks manager, Chuck Wolken, asked to have the park evaluated concerning utilization by vegetative and avian species.

Wildwood/Kraemer Lake County Park site history was obtained verbally from the previous owners. Wally and Dorothy Honer purchased WW in the 1950s, raising a family there until Wally's death in 1999. Occasionally cattle and horses grazed, standing timber was harvested and dead trees were removed for the sawmill operation. The county purchased the land, 265 acres, from the children in 2007; with the only management practice having been implemented since 1985 is the removal of downed trees.

Pioneering monks purchased St. John's Arboretum (SJA) (now renamed St. John's Outdoor University) and St. John's University (SJU) in the 1850s. "From the beginning, the faculty—mostly monks—engaged in manifold projects of research and restoration" (Kulas 2007), leaving the land as they found it, engaging in conservative forestry management practices to this day. Forestry manager, Tom Kroll, oversees removal of some timber to make furniture, fuel their maple sugar operation and fire their pottery kiln.

Location

The research area two miles west of St. Joseph, Minnesota, USA (Figures 2 & 3) included sites at: Wildwood/Kraemer Lake County Park (45°33'13.03"N, 94°22'30.66"W), St. John's University

Arboretum (45°33'56.02"N, 94°22'52.80"W), St. John's University (45°34' 50" N, 94° 23' 33" W)

(Figure 4).

Figure 2. Location of study site in USA, Minnesota (google.com).

Figure 3. Stearns County in Central Minnesota (google.com).

Figure. 4. Site locations, central Stearns County, 2 miles west of St. Joseph, MN: Wildwood/Kraemer Lake County Park (WW), St. John's University (SJU) and St. John's Arboretum (SJA). The yellow arcs represent targets, beginning with the putative old growth (red-zone 1) in center transitioning outward to the continually presumed younger forest (blue-zone 2, yellow-zone 3, and pink-zone 4).   These arcs infer edges between the buffer zones.

Study Site

Each site (forest fragment) varied in size: SJA=2,500 acres, WW=265 acres, SJU=50 acres. Roughly 50 acres of old growth core with varying amounts of successional buffer were identified per site. One pair of old growth (zone one) target plots were established per site, with the amount of successional buffer plots per site dependent on overall total area of that site: SJA had two deep forest and six buffer plots, WW had two deep forest and two buffer plots, and SJU had two deep forest and no buffer plots. The buffer zones consisted of younger hardwood (red/white oaks) and softwood (basswood, poplar, box elder) tree species as compared to mostly older hardwood in the mature core forest zones. As successional zones proceeded out from the core zones, the area became more populated with younger trees and shrub species. A total of fourteen 20 X 20 meter plots were sampled with each site and zone containing two replicate plots.

To census plant and bird communities, plots were established in accordance with standard sampling procedures from *A Handbook for Collecting Vegetative Plot Data in Minnesota: The Releve' Method* and *A Habitat-Based Point-Count Protocol for Terrestrial Birds* (MN DNR, 2007,USDA & FS, 2000). Plot locations were permanently recorded via global positioning system (GPS) (Table A-Appendix). Releve' and point count survey stations were selected according to protocols with the first site randomly located (Figure 5). Old growth plots were selected first, with subsequent successional plots located off these. All plots were 125m to the stand edge, 150m between plots, 50m from roads or water and 75m from sharp breaks in vegetation. Releve' and point count surveys were in the same plots.
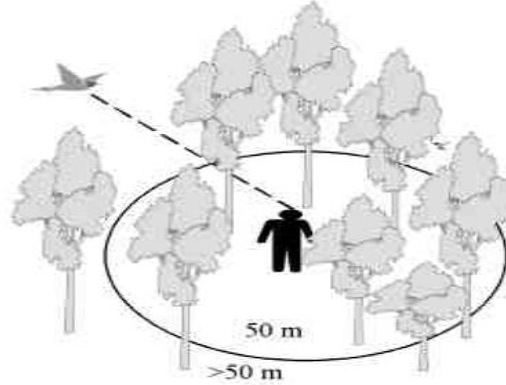
Figure 5. Point-count survey station divided into bands of 50m and >50m (USDA & FS 2000).

Vegetative surveys were conducted three times yearly: May, July, and September. Each survey recorded presence/absence species data in 2012 and 2013, with vegetative ground cover estimation data collected in 2013. In 2013, ground cover was estimated for each species in each plot. Avian surveys were conducted three times per season May/June (spring) and September/October (fall), in predawn hours prior to 10:00 a.m. (Ralph *et al*. 1993; USDA & FS 2000; MN DNR 2007; Stucker 2012). All surveys were conducted using safety precautions and in seasonable weather, devoid of significant wind or precipitation.

Species Categorizations

Species were categorized prior to sampling according to expected habitat affiliations. Expected plant species were categorized as with deep forest or ecotone species. For birds, habitat affiliations consisted of: old growth, generalist, successional and incidental. The species were placed into these categories due to their preference for a specific type of forest habitat. Old growth species preferring mature woods for reproductive fitness, generalist were found throughout the forest with no partiality noted, successional birds selecting brushy cover and younger forest trees with the incidentals being those that were seen or heard flying over. Species that did not belong to the habitats (incidentals) were recorded but omitted from analysis (Table F).

Species categorizations were made via consulting various sources and local authorities (Hanowshi 1995; Bedford Audubon Society 2013; Bell State University 2013; Cornell Ornithology Lab. 2013; MN DNR Data Plant Base 2013, Dr. Jorge Arriagada, Dr. Stephen Saupe; National Parks Service 2013;). Inclusive species lists for birds, plants and trees were compiled for posterity (see Appendix).

Data Analysis

Due to small sample size, non-parametric Kruskal-Wallis Tests were run in Microsoft Excel to compare species richness between sites and between zones. Diversity was investigated via cluster analysis, Shannon Wiener Diversity Index (H') , and canonical correspondence analysis using Minitab and PAST version 2.17c.

Prior to sampling, predictions were generated for each species regarding occurrence patterns in the putative core and buffer zones. Species were categorized into 'Fit' and 'Not Fit' using the data with respect to the spatial predictions. For each species the percentage of observed occurrences in each zone was calculated and then used to evaluate fit. To check correctness of vegetative categorizations, species that 'Fit' into deep forest patterns were found in mature woods; species that did 'Not Fit' were found only in younger successional buffer zones. Plants that 'Fit' into ecotone patterns were recorded in successional zones; species categorized 'Not Fit', were recorded only in deep forest areas.

To determine accuracy of avian classification, the same habitat affiliation was used as it pertains to times a bird was recorded as an: old growth, generalist, successional, or incidental siting. Species were placed into 'Fit' or 'Don't Fit' classes as to the percentage of time they were recorded in a particular zone. Species that 'Fit' into old growth patterns were found in mature woods; species that did 'Not Fit' were found only in younger successional zones. Birds that 'Fit' into generalist patterns were recorded randomly in all zones. Species identified as 'Fit' in successional zones were those recorded outside old growth cores areas; species categorized 'Not Fit', were recorded in deep forest areas only with incidental siting data not used in this analysis.

<u>Plot Measurements</u>

Tree measures were recorded to establish relative stand age and maturity. Diameter at breast height (DBH) of the largest individual of each species was recorded for each plot using a tape measure. These were then averaged for each zone (Table B-Appendix). A total of 58 trees were measured. Plot canopy cover was recorded using a forest densiometer. Readings were taken at each 90° angle and plot center, the five readings were then averaged, multiplied by 1.04 and subtracted from 100 to give percent canopy cover (Table C).

RESULTS

Plant Community Composition

A total of 82 understory herb and shrubs (Table D-Appendix) and 15 tree species (Table E-Appendix) were recorded over the two year period May to October, 2012-2013. Of the 18 species categorized as deep forest plants, only four were never recorded in Zone 1 and of the 64 ecotone buffer plants, only four were recorded in Zone 1 only.

In both years, the number of species that fit expected patterns was greater than the number that did not fit (Figure 6). In 2012, 40% more 'Fit' in the deep forest than did 'Not Fit' and 80% more in the ecotone categorization. In 2013, 60% more in the deep forest and 95% in the ecotone 'Fit' than did 'Not Fit'.

Plant Species Richness

It was predicted that the greatest old growth species richness would occur in Zone 1 plots in the sites with the greatest amount of buffer (i.e., SJA and WW). Zone 1 plot species richness did not follow this pattern in 2012 ($H = 2.00$, $DF = 2$, $P = 0.368$) or in 2013 ($H = 2.00$, $DF = 2$, $P = 0.368$) (Figure 7). This prediction was thus not supported. The lowest ecotone species richness was expected to be found in the Zone 1 plots of the site with the greatest buffer, increasing from the largest buffer (SJA) to the smallest (SJU). This pattern was not observed in 2012 ($H = 2.00$, $DF = 2$, $P = 0.368$) or 2013 ($H = 2.00$, $DF = 2$, $P = 0.368$).

When comparing zones, old growth species richness was expected to be highest in Zone 1 plots and decrease moving outward through Zones 2-4 (Figure 8). No significant consistent pattern was detected in 2012 ($H = 3.00$, $DF = 3$, $P = 0.392$) or 2013 ($H = 3.00$, $DF = 3$, $P = 0.392$).

When comparing successional species richness among zones, the expected pattern was for richness to be lowest in Zone 1 plots and to increase when moving farther away. This pattern did not follow expectations (in 2012, H = 3.00, DF = 3, P = 0.392 or 2013, H = 3.00, DF = 3, P = 0.392).

Kruskal-Wallis Tests were run to compare total plant species richness among zones, however there were no consistent patterns and differences were not statistically significant in 2012 (H=6.31, DF=3, p=0.097) or in 2013 (H=3.34, DF=3, p=0.339) (Figure 9). Plant species diversity and ordinations.

Shannon-Weiner Diversity (including all plant species) increased slightly from Zone 1 plots through Zones 2-4 (Figure 10), but this trend was not statistically significant (H = 41.00, DF = 41, P = 0.471). Cluster analyses were run in PAST to determine affiliation between samples collected in the various zones, sites, and seasons. In the first cluster analysis four of the six Zone 1 plots clustered together in the spring, four of the six in the summer and five of the six in the fall (Figure 11, a., b., and c.). One Zone 1 plot (WWA1) consistently did not cluster with the others. After a canonical correspondence analysis was performed, we plotted CCA Axis 1 vs Axis 2, and generated graphs with the points (samples) labeled three different ways (by site, zone and season) (Figures 12, 13, and 14). In each case there appeared visually to be patterning on either Axis 1 or 2 (or both), so we ran a series of analyses of variance to determine whether these patterns were statistically significant. When the CCA Axis 2 scores were labelled by season, a one-way ANOVA indicated a statistically significant difference (DF=2, F=12.03, P<0.001, differences between spring and both summer and fall by Tukey's multiple comparisons of means). When the points were labelled by zone, one-way ANOVA indicated a statistically significant difference on Axis 1 (DF=3, F=3.22, P=0.033, differences between Zone 1 and Zones 2-4 using the Tukey's Test). When labelled by site, one-way ANOVAs showed significant differences on both axes of the CCA. For CCA Axis 1, ANOVA indicated statistically significant differences (DF=2, F=18.32, P<0.001) and Tukey's procedure indicated significant differences between WW and both SJA and SJU. For CCA Axis 2, ANOVA indicated statistically significant differences (DF=2, F=3.49, P=0.040) and the Tukey's procedure indicated significant differences between WW and SJA.

[We also performed ANOVAs by season and zone on scores on the other axes, but no other patterns were statistically significant.] This suggests that proper interpretation of patterning on the CCA by site requires simultaneous consideration of both CCA axes, while Axis 2 better represents seasonal patterning and Axis 1 better represents differences among zones.
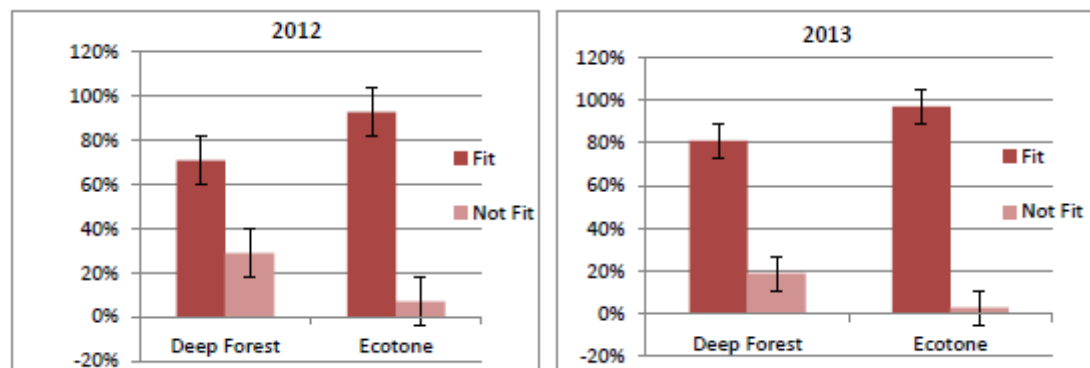
Figure 6. Deep forest and ecotone affiliated plant species that either "fit" or did "not fit" the predicted pattern in 2012 and 2013.
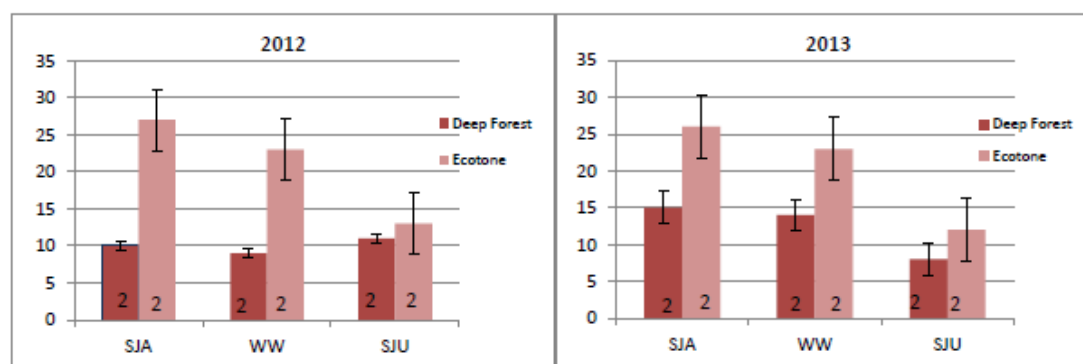
Figure 7. Old growth and successional habitat affiliated plant species richness per site in 2012 and 2013.
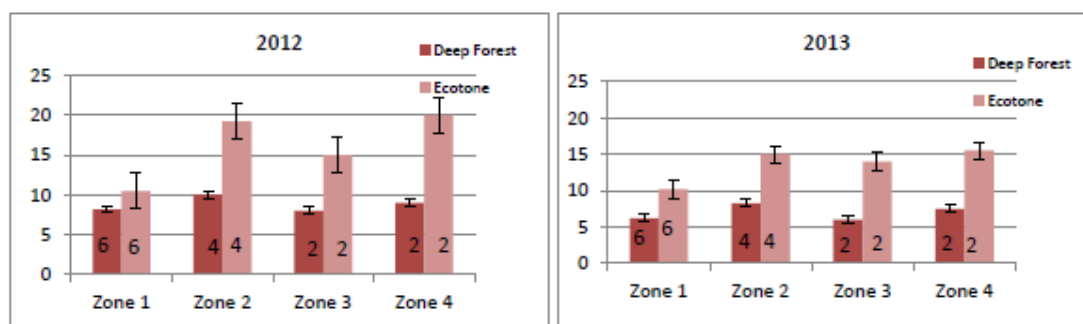
Figure 8. Plot richness in zones 1-4 in 2012 and 2013.

Kruskal-Wallis Richness Test 2012				Kruskal-Wallis Richness Test 2013			
Zone	N Median	AveRank	Z	Zone	N Median	AveRank	Z
1	6 23.00	4.7	-2.19	1	6 17.00	5.5	-1.55
2	4 33.00	10.8	1.84	2	4 23.00	10.3	1.56
3	2 26.50	6.5	-0.37	3	2 16.00	6.8	-0.27
4	2 32.00	10.5	1.10	4	2 19.50	8.8	0.46
Overall	14	7.5		Overall	14	7.5	
H = 6.31 DF = 3 P = 0.097				H = 3.34 DF = 3 P = 0.342			

Figure 9. Kruskal-Wallis Richness Test on richness in zones 1-4 in 2012 and 2013.
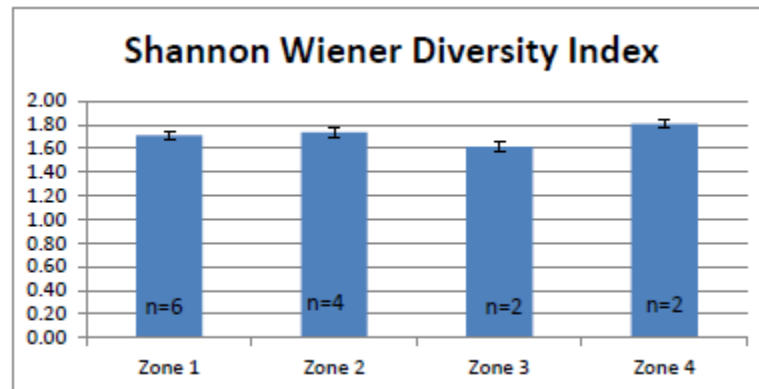
Figure 10. Shannon Wiener Diversity Index comparing total biomass in zones 1-4 in 2013.
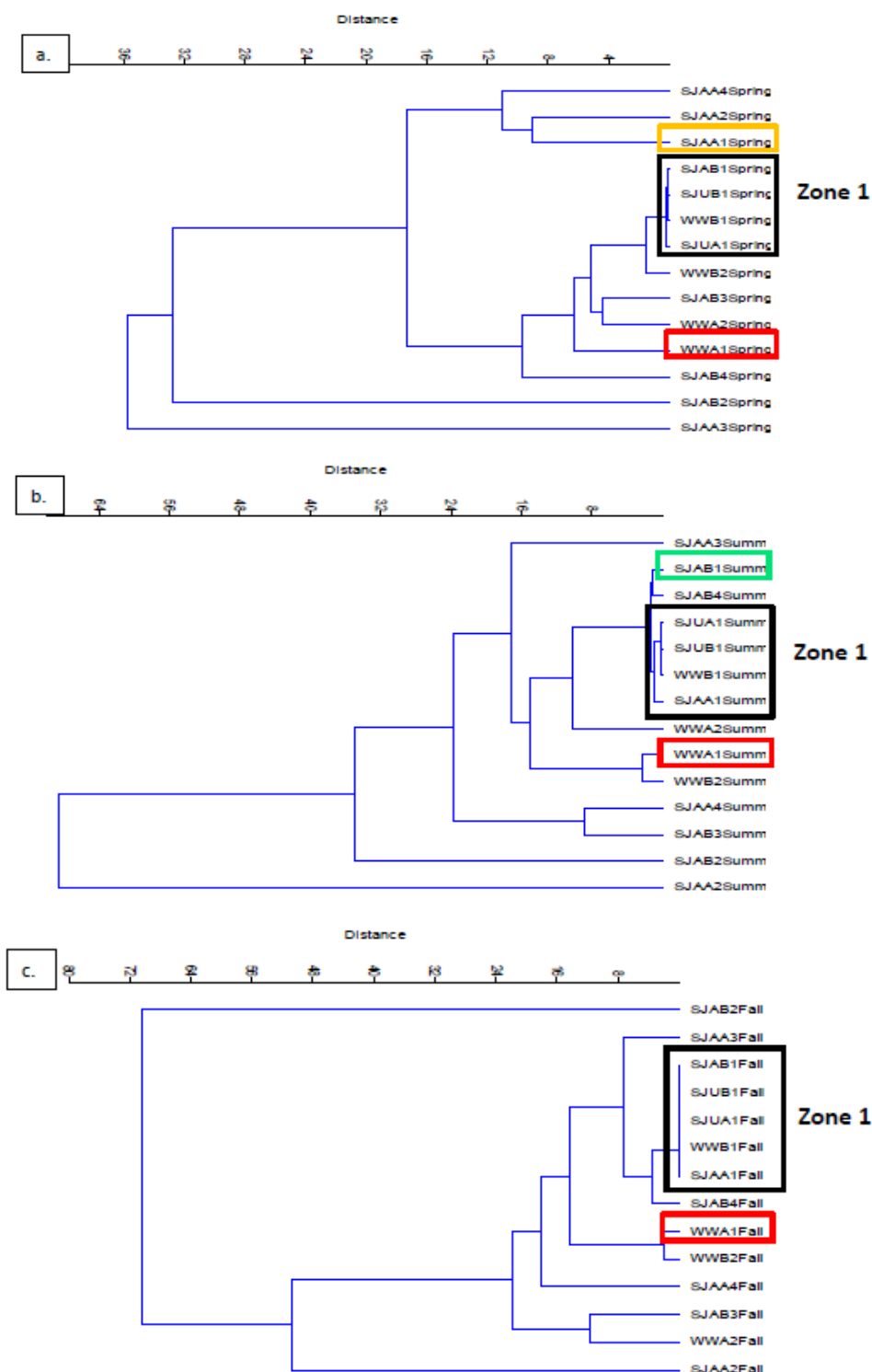
Figure 11. Cluster analysis graphs showing zone 1 relationships among seasons (a., b. and c.). The first 2-3 letters of the code represent the plots (SJA, SJU, or WW). Following the plot letters, the A or B represents the plot within each site and zone. The number codes for the zone followed by the season.
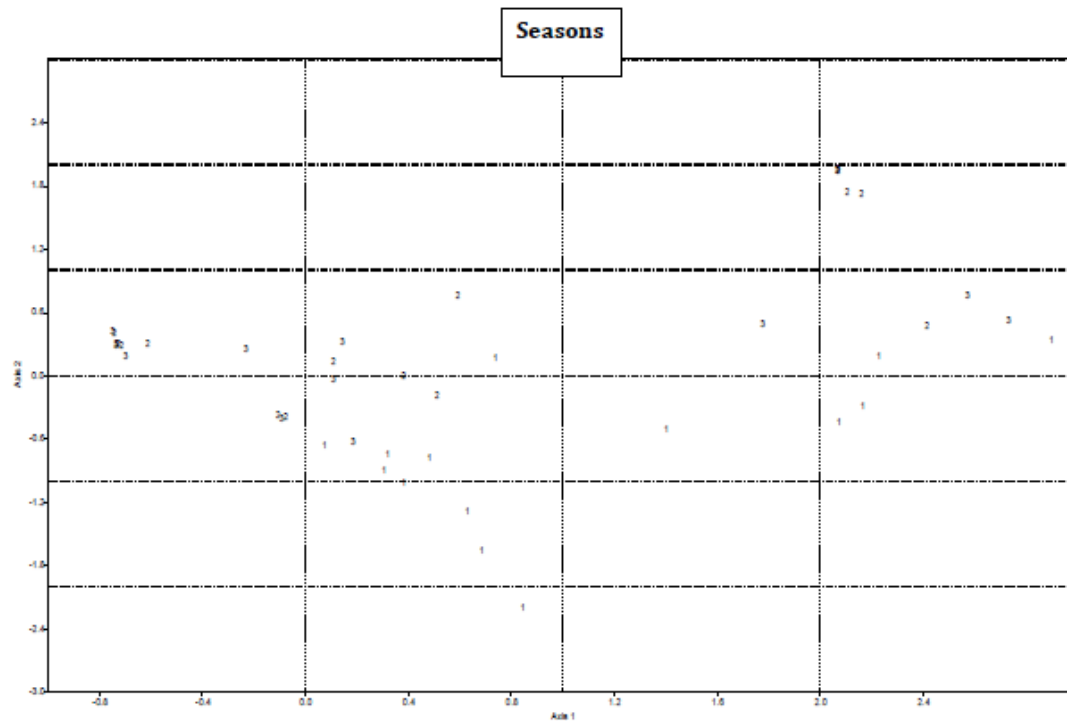
Figure 12. CCA Axis 1 and 2 with samples labeled by season (spring=1, summer=2, and fall=3).
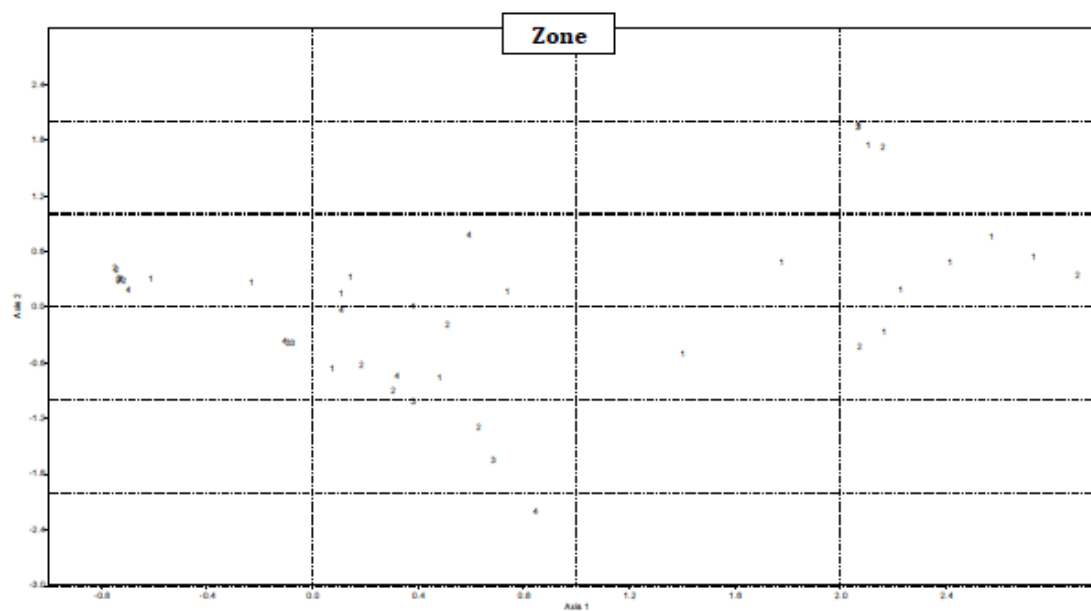
Figure 13. CCA Axis 1 and 2 with samples labeled by zone (1, 2, 3, 4).
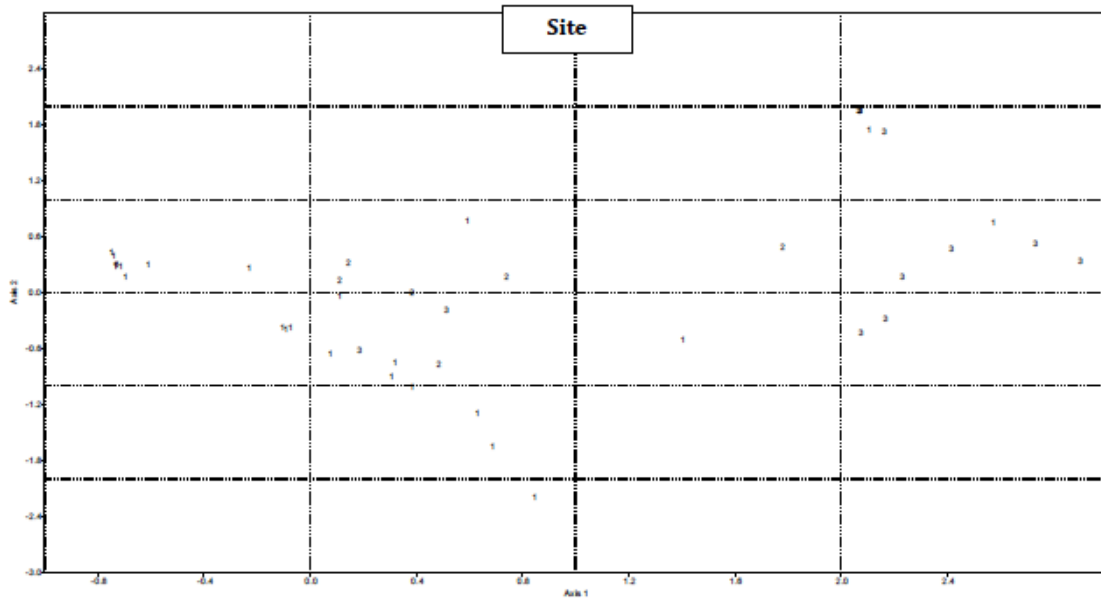
Figure 14. CCA Axis 1 and 2 with samples labelled by site (SJA=1, SJU=2, and WW=3).

Bird Community Composition

A total of 52 avian species were recorded (Table F-Appendix) over the two years, May to October 2012 and 2013. Of the 16 species classified prior to sampling as old growth affiliated, five were never seen in Zone 1 plots and of the eleven categorized as successional habitat species, only one was never seen outside the Zone 1 plots.

In 2012, old growth-affiliated bird species 'Fit' into the expected patterns (occurring more in Zones 1 & 2 than in 3 & 4) as often as they did 'Not Fit' and in 2013 slightly more species did 'Not Fit' (Figure 15). The generalist community had marginally more species that 'Fit' than did 'Not Fit' in 2012, with a stronger pattern in 2013. Successional species followed expectations most closely; at least 80% of species fit the expected pattern in both years.

Bird Species Richness

Species originally categorized as old growth-affiliated were expected to occur more in Zone 1 plots with larger buffers (i.e., SJA and WW) than in those with smaller buffers (SJU). Successional

species richness was predicted to increase from Zone 1 of the site with the largest buffer area (SJA) to Zone 1 of the site with the smallest buffer area (SJU) (Figure 16). Both old growth and successional bird richness in the Zone 1 plots varied little (and not significantly) among the three sites in 2012 (old growth H = 2.00, DF = 2, P = 0.368 and successional H = 2.00, DF = 2, P = 0.368)) and in 2013 (H = 2.00, DF = 2, P = 0.368 and successional H = 2.00, DF = 2, P = 0.368).

When comparing richness among zones, old growth habitat affiliated bird richness was predicted to decrease from Zone 1 outward through Zones 2 through 4 (assumed to be the successional buffer areas) and successional habitat affiliated bird richness was expected to increase when moving from Zone 1 to Zone 4 (Figure 17). This pattern was not seen in 2012. In 2013 both the expected old growth affiliated and successional habitat affiliated bird patterns appeared to occur in Zones 1-3 but did not extend to Zone 4. Overall, this pattern was not statistically significant in 2012 in old growth (H = 3.00, DF = 3, P = 0.392) or successional (H = 3.00, DF = 3, P = 0.392) nor in 2013 old growth (H = 3.00, DF = 3, P = 0.392) or successional (H = 3.00, DF = 3, P = 0.392).

Total bird species richness did not vary significantly by Kruskal-Wallis test in 2012 (H = 1.89, DF = 3,P = 0.596) or 2013 (H = 1.16, DF = 3, P = 0.762) (Figure 18).
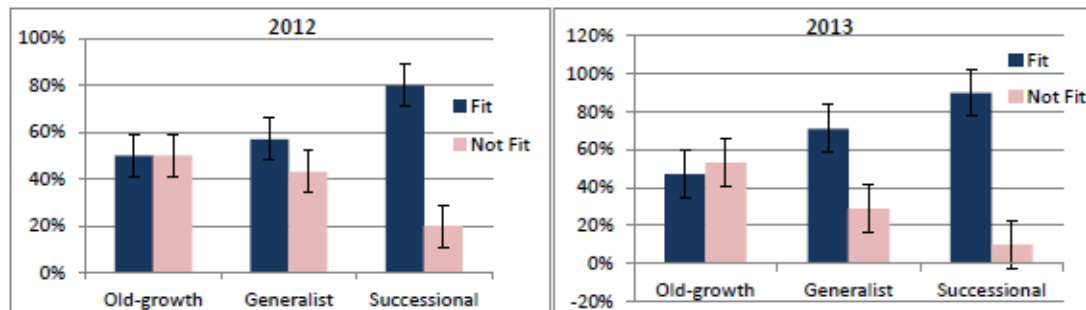
Figure 15. Old growth, generalist and successional habitat affiliated bird species that either "fit" or did "not fit" the predicted pattern in 2012 and 2013.
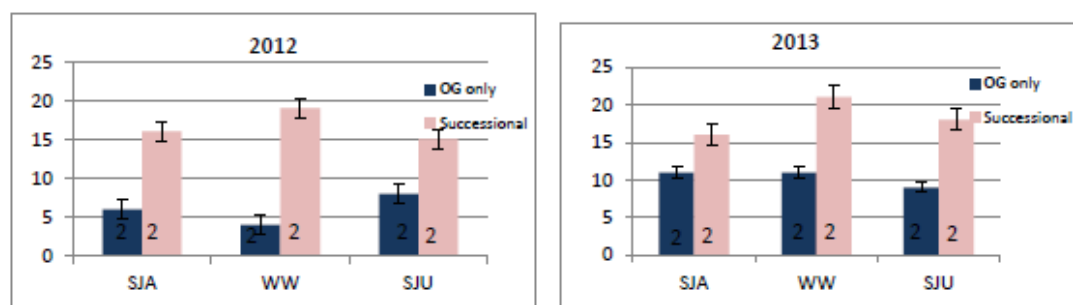
Figure 16. Comparing site richness between old growth and successional habitat affiliated bird species in 2012 and 2013.
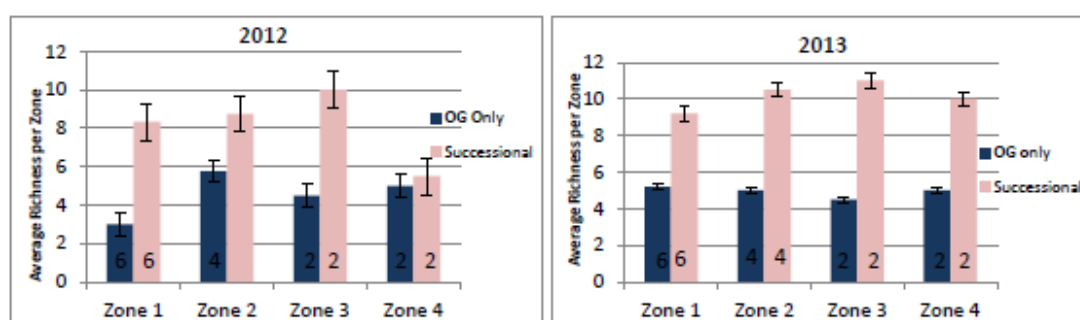
Figure 17. Comparing zone richness between old growth and successional habitat affiliated species in 2012 and 2013.

Kruskal-Wallis Richness Test 2012				Kruskal-Wallis Richness Test 2013			
Zone	N Median	Rank	Z	Zone	N Median	Rank	Z
1	6 12.50	6.8	-0.58	1	6 14.50	6.2	-1.03
2	4 16.50	9.5	1.13	2	4 15.50	8.9	0.78
3	2 14.50	8.3	0.27	3	2 15.50	8.5	0.37
4	2 11.50	5.0	-0.91	4	2 15.00	7.8	0.09
Overall 14		7.5		Overall 14		7.5	
H = 1.89 DF = 3 P = 0.596				H = 1.16 DF = 3 P = 0.762			

Figure 18. Kruskal-Wallis Richness Test comparing zones 1 through 4 in 2012 and 2013.

DISCUSSION

After monitoring avian and vegetative communities for two seasons at Wildwood County Park, we came to the conclusion that the park is a forest fragment with some evidence of relictual old growth communities as defined by the Minnesota Department of Natural Resources (MN DNR, 2005) as well as additional communities of species not associated with old growth habitats (Zone 1) moving outward into the buffer areas (Zones 2-4).

Ecological succession is the process by which species of communities change over time by varying degrees often brought on by disturbances. When wind, floods, fire, insect outbreaks or humans cause structural change in the forest, this causes other changes involving light, wind, soil nutrients, erosion, canopy cover and moisture availability. When nutrients become limiting or plentiful, plant communities are altered, leading to habitat changes for animal species. When animal biodiversity is altered, this has an impact on the forest, leading continued successional changes.

Fragmentation results in complex landscape interactions among a variety of characteristics and processes including patch size, edge effects, habitat loss and gain, matrix interactions, corridors, isolation and ratio of size to edge (Bender *et al*. 1998; Willson *et al*. 2000; Schmeigelow & Mönkkönen 2002; Fahrig 2003). Due to the complexity and the interaction of these elements, teasing apart each component as it relates to the others is both problematic and necessary. Studies have shown variously that area, isolation, matrix characteristics, intensity of former agricultural use, stand size and structure all have strong influences on species richness and diversity (Berglund & Jonsson, 2001; Dupouey *et al*. 2002; Ockinger *et al*. 2012).

A main ecological goal of the study was to determine if the bird and plant communities at the several sites exhibited patterns consistent with the operating assumption of 50 acre old-growth relicts

surrounded by forest buffers of varying size, and if so, whether the communities in the buffers followed expectations of ecological theory. In protected forest fragments, understanding processes in buffers is imperative to the understanding of health of the fragments (Spies & Franklin 1996). In theory a larger area of buffer habitat should offer more protection for species specific to old growth forest (Chapman & Reich unpublished data; Groombridge 1992; Faaborg *et al*. 1993; Ruggiero, Hayward, & Squires 1994; Bibby, 1995; Ehrlich 1995; Thomas & Morris 1995; Rappole 1996; Fahrig 2001). If a buffer exists, there should be species in the buffer that prefer the surrounding habitats and the distance through the buffer should decrease the number of these species found in central old growth areas where they do not usually occur (Bowman 2011). If the assumed buffer areas around the core function according to theory, buffer areas and old growth areas should contain different communities of species with some overlap (NRCS, 2007). In this study, predictions concerning species richness patterns were not consistently supported by either bird or plant communities. However, ordination analyses detected some significant patterns among vegetative communities which better support some of the general predictions. Ordinations also detected community differences among seasons and study sites which may be meaningful for evaluation of management practices.

Sampling at Wildwood and St. John's found bird species associated with old growth habitats, but with unpredictable occurrence patterns similar to a study in West Virginia where some species specific to old growth exhibited predicted patterns while others did not (Finch 1991). In Minnesota there were equal numbers of old growth bird species that did 'Not Fit' the pattern as 'Did Fit.' In analysis of the generalist and successional forest species, most birds 'Fit' predictions (Figure 15). When comparing bird and plant species patterns we found that the plant species 'Fit' the predicted classifications better and had a stronger association with Zone 1 plots, than the bird species did (Figure 6). The particular guild of deep forest understory plant species is ecologically better equipped to exhibit predicted patterns in forest fragments: they are long-lived, can reproduce colonially and thus persist if pollinators are missing, are tolerant of isolation, and they do not disperse well (Dzwonko

& Loster 1992; Ehrlen 1999; Rees *et al.* 2001; Hermy & Verheyen 2007). Thus, these plants often persist in forest fragments long after the original fragmentation event.

When old growth forest fragments start to degrade, the relatively mobile birds can simply fly away (Haila *et al*. 1993). As these fragments are not particularly large, it seems unsurprising that the old growth bird community has degraded noticeably, and that the successional bird community better fits expectations at the Wildwood and St. John's sites. The plants do not have the luxury of easy dispersal, and must remain where they are planted or rely on animals, wind or rain events for dispersal and the appropriate weather conditions for germination (Ehrle'n & Eriksson 2000; Dzwonko & Loster 1992).

<u>Vegetative Species Richness and Diversity Patterns</u>

Jamoneau *et al*. (2011) indicates that preserving large old forest fragments should better conserve species specific to old growth habitats because the size of habitat patches is known to affect resource availability, species richness and diversity (Estades & Temple 1999; Bowman 1980; Berglund & Jonsson 2001, D'Amato, Orwig, & Foster 2009). In this study, plant species richness patterns in Zone 1 areas (assumed to be old growth) did not follow the prediction that old growth species richness should be greater when surrounded by a greater buffer area (Figure 7). In 2012 the sites with the largest buffer zones (WW and SJA) seemed to fit the pattern but it did not extend to SJU. In 2013 the general pattern seemed to hold as expected but it was not statistically significant. Overall, the size of the buffer did not have consistent or strong effects on plant richness in the "old growth" plots. It is possible that this is because this study included three relatively small forest fragments. During habitat fragmentation, if fragments are reduced in size sufficiently, all fragments can become small enough that they experience the same processes of resource allocation, relocation and predation regardless of relative size (Chapman & Reich unpublished, Kouki *et al.* 2001).

When comparing zones, vegetative old growth species richness was predicted to decrease in the plots as they progress outward from Zone 1 plots (the assumed old growth areas) through Zones 2 through 4 (assumed to be younger successional forest) (Figure 8). In both years, the predicted pattern was not seen as Zone 1 had lower old growth species richness than Zones 2-4, with Zone 2 consistently exhibiting the greatest richness (this may have been a result of heavy equipment driving through our plots for tree removal from severe wind and storm damage). When our study compared ecotone species richness among zones, the predicted pattern of species richness increasing when moving from Zone 1 outward through Zones 2-4 was not observed (Figure 8) and in a similar study, Mabry (2009) reports that even though he found "differences" in the two forest fragments, he found no difference between richness and diversity between an old growth forest and a younger planted forest.   Mabry's study also found that variations in plant richness between areas, can be attributed to the method by which the data is collected, line intersect or meter squared plot (Mabry 2009). Unlike our study, White and Lloyd found that when managing forests for protection of old growth areas, that surrounding old growth core areas with managed buffer stands, protects old growth specific species in those core areas (1994), resulting in greater old growth species richness in the core than in the surrounding buffer. In a 2001 study, Battles *et al.* found that managed sites have higher understory plant richness than 80 year old non-managed sites and Spies and Franklin (1996) found that many different old growth forest types and stages of development can be occurring in the same area and at the same time. Other studies find that vascular plant diversity increases over time, peaking in old growth (Halpern & Spies 1995).

When comparing total overall species richness among the variously sized sites, this study found marginally statistically significant differences among zone richness in 2012 by Kruskal-Wallis test, but differences were not significant in 2013 (Figure 9). In our study, overall plant richness increased with increasing fragment size (although not significantly) similar to Berglund *et al.*, who found area species richness usually increases when overall territory increases (2001). When

comparing total plant richness among the zones, Zone 2 had the greatest species richness in both years, possibly due to ground disturbance and tree removal after several large storms which specifically impacted Zone 2 plots (vehicles were driven through the plots to facilitate tree removal) (Figure 8). Not only did Zone 2 have the greatest richness, deep forest and ecotone plants alike were both found in Zone 2 more often than the other zones, possibly suggesting that Zone 2 is transitioning between from an ecotone to an old growth stage or some other more complex pattern.

Shannon diversity (H') of plants increased from Zone 1 outward through Zones 2-4. The plant cover data available in 2013 showed that H' was greatest in Zone 4 (Figure 10). The outer zones (Zone 2-4), furthest from the core old growth areas (Zone 1), are subject to greater diversity possibly due to edge effects, increased sunlight and decreased old growth tree competition for moisture and nutrients (Spies & Franklin 1996; Kouki *et al*. 2001; Fiona *et al*. 2002; Kennedy *et al*. 2010).   A cluster analysis showed affiliation among the plant communities from the Zone 1 plots, with four to five of the six zone 1 plots appearing together in all analyses (Figure 11). This suggests the Zone 1 plots (presumed old growth) are different from the Zone 2-4 (presumed buffer) areas when species abundances are taken into account, and that the Zone 1 plots had been categorized correctly as relative old growth habitat (Figure 11). [One plot [WWA1] consistently did not cluster with the others, possibly due to higher elevation as it is the only putative old growth plot on a hill with conceivably decreased moisture in the soil.] Canonical correspondence analysis of plant community differences among zones, seasons and sites, showed several clear patterns among samples. The CCA with points labeled by zone (Figure 13) supports the result from cluster analysis, suggesting that communities in Zone 1 plots were in fact more similar to each other than to the communities on other plots in Zones 2-4. Given the lack of resolution among samples from Zones 2-4, we conclude that samples reflect two predominant habitats, relative old growth (Zone 1) and buffer or successional habitat (Zones 2-4). The CCA where samples were coded by season showed similarities among the samples recorded in the spring and affiliations among those from the summer and fall communities together (Figure 12). This

appears logical as spring ephemerals usually emerge first under the deep forest canopy and are gone before the summer and fall plants appear, leading to the differentiation of spring from summer and fall samples. Meanwhile, most plant species active in summer continue to be present in the fall, so samples from these seasons appear together. The sample sites (WW, SJA, and SCU) also showed some degree of differentiation (Figure 14). Samples from each site clustered together somewhat, but samples from the SJA and SJU sites were more similar to each other than they both were to samples from the WW site. One-way ANOVAs of ordination scores on Axes 1 and 2 showed statistically significant differences among seasons, zones and sites with the strongest pattern among the sites on both CCA Axes 1 and 2. On both axes WW plots differentiated strongly from plot from the two St. John's sites (SJA and SJU). The pattern suggests management may play a role in biodiversity patterns, similar to Dupouey *et al*.'s study (2002), which found land management to have irreversible impacts on species richness and plant communities.

Avian Species Richness Patterns

This study's original prediction of greater species richness occurring in Zone 1 plots (putative old growth areas) surrounded by the larger buffer zones was not well supported by the avian data (Figure 16). When comparing old growth species richness patterns in Zone 1 sites to each other and successional species richness patterns in Zone 1 sites to each other, no patterns were found in 2012 and only weak patterns were present in 2013. Successional species richness in Zone 1 at sites with the largest buffers (e.g., SJA) was expected to be smaller in Zone 1 plots than in sites with smaller buffers (e.g., SJU). This pattern was not seen in either year (Figure 16). Some studies indicate landscape structure, size, composition, and management all influence habitat selection by birds (Jones 2001; Kennedy *et al.* 2010). Other studies find that avian species in smaller fragments will relocate yearly irrespective of population trends as small forest predators have been shown to target the smaller fragments (Santos & Telleria 1992; Haila *et al.* 1992).

When comparing zones, old avian growth species bird richness did not follow the predicted pattern of Zone 1 having greater old growth richness than the other three zones (Figure 17). Successional bird species richness was predicted to increase as the plots moved away from Zone 1 into the surrounding buffer. This pattern was observed to some degree in both 2012 and 2013, however neither was statistically significant. Our study is similar findings by Finch (1991) who did not find increased species richness, but actually found that long term studies of avian populations in small isolated forest fragments showed consistent declines or extinctions and this may be explained partially by the association between bird community structure and forest fragment size. These smaller isolated fragments may not be compatible with positive reproductive growth of those species due to: increased predation, decreased food and nesting resources, increased anthropogenic disturbances and increased weather influences. While our study did not show a decline, it did not show an increase, this possibly due to the short term of the study, or because all forest fragments were sufficiently small that they have suffered more or less equivalent degradation.

When comparing overall species richness among zones, Kruskal-Wallis tests were non-significant and failed to detect the predicted trends of increased richness in the zones further away from the core (Figure 18). This suggests minimal ecological difference among the three successional zones with respect to bird habitat. This study found that occurrence patterns of old growth species generally known to prefer closed canopies were more difficult to predict than those of generalist and successional species (Figure 15). Mobile organisms, however, such as birds, tend to be unreliable indicators of richness as their habitat affiliation may be affected by overall migratory range as they are subjected to different physical requirements (i.e., nesting, feeding, and predator protection). Fiona *et al*. (2002) found that seasonal diversity fluctuation is seen and that regional trends are not always consistent in forest remnants. This may account for some differences seen between Wildwood and St. John's even though the sites are in close proximity. Data in this study was collected uniformly, with all sites surveyed the same time of year and for the same amount of time. This inconsistent pattern of

fluctuations and trends in the bird data was consistent with avian data collected during this research as there were no significant statistical patterns found among sites, zones, or years as summarized in Figure 18.

Summary

The hypothesis that species richness is greatest in larger forest fragments was not supported by either avian or vegetative species richness measures. Some minimal plant richness differences were detected, but no overall richness patterns in bird or plant data showed significance, meaning site size did not appear to make a difference.   CA and CCA ordinations revealed similarities in community composition and composite measures of diversity between zones, confirming the occurrences of two somewhat dissimilar habitats, Zone 1 (old growth) and Zones 2-4 (successional).

The strongest pattern observed (via ordination analyses) was that of differences between Wildwood plant community and the St. John's plant communities (both SJA and SJU), indicating that historical land ownership and management practices seem to be associated with greater plant community variation than season or habitat type. Even though SJA may be a larger expanse of intact forest, taking the standing timber may cause successional regeneration to occur faster than would normally occur if the woods are left to age naturally and it may cause areas of the forest to be difficult to place into one particular category. This causes plant and bird communities to shift, attracting different species than would occur in a natural setting. Managing the woods for faster regeneration may detour species that require old growth. This may have been the reason that although this forest covers approximately 3,000 acres, few old growth plant and bird species were identified. It may also be why birds did not "Fit" into old growth categories as had been predicted, with successional birds being found throughout the forest. WW may be a smaller tract of forested land (265 acres), but by not harvesting the mature trees, it has old growth species biodiversity similar to the larger tract of forest (SJA). SJU is by far the smallest and the least managed area in the study. This may be why it

had greater old growth species richness in 2012 than either SJA or WW and had more richness per acre in 2013 than either SJA or WW. Even though SJA is 30 X larger and WW is 11 X than SJU, greater biodiversity was found in the smaller old growth forest tract because the old growth trees had been left alone and the forest in this area has been allowed to age naturally. Management in this study appeared to make a significant difference in plants and bird species communities. Each of the forest fragments in this study has undergone some amount of 'management' over the last 50 years (and beyond). At the present time, WW has less active management than SJA, while SJU is managed as a 'quiet zone'.

WW currently allows some windfalls, standing dead and snags to be harvested. Previous owners have operated a maple sugar operation and have been allowed by the county to continue to 'manage' the land even after it was sold to the county. This management involves cutting standing dead and fallen timber to fuel their maple sugar operation, and to clear the cross country ski and hiking trails often driving off road vehicles to assist with tree removal. Many of these management practices undoubtedly impact animal and plant species via changes in vegetation structure and soil disturbance, as well as via visual, auditory and pollution disturbances during and after harvest (White & Lloyd 1994; Dupouey *et al.* 2002). Cutting the standing dead and fallen timber increases light, moisture and wind effects along with decreasing species and nutrient resource availability (Spies & Franklin 1996; Kouki *et al*. 2001; Fiona *et al.* 2002; Kennedy *et al*. 2010). Driving vehicles increases noise and CO_2 emissions, disrupts plant growth, along with adding overall anthropogenic disturbances (Canterbury *et al*. 2000; Francis, Ortega, & Cruz 2009).

SJA, at this time, as per forest manager Tom Kroll, cuts standing live timber but not snags or standing dead timber. Management practices allow heavy equipment to harvest trees to make furniture, fire a large kiln and cook maple syrup. This also includes many miles of hiking trails to clear timber from and collect sap for producing maple sugar (tapping approximately 800 trees yearly). St. John's management allows a limited archery deer hunting season and various educational

opportunities for students both on and off campus. The edges created by the removal of standing live timber impacts soil nutrients, erosion, light and weather affects decreases habitat resources for resident plant and animal communities, while facilitating forest inhabiting predators (Santos & Telleria 1992; Haila *et al.* 1992; Hanski, Fenske, & Niemi 1996). The operation of heavy equipment compacts soils disturbs animal and bird communities by emitting noise, CO_2 and visual pollution, chainsaws are loud and emit gas and oil smells, along with multiple anthropogenic disturbances from hunting, hiking, and allowing outdoor classes on the grounds for educational opportunities (Fahrig 2001; Slabbekoorn & Peet 2003; Slabbekoorn & Ripmeester 2007).

SJU has no management practices at this time (or for approximately the last 50 years) and is termed 'the quiet zone' by forest manager Kroll. This area is away from the central campus and experiences little anthropogenic disturbances with no hiking or cross country trails available. The data for this site showed high species biodiversity for both plants and birds even though this site has no buffer zones as seen in SJA and WW. The lack of an active management (removal of trees, etc.), and the reduction in anthropogenic disturbances, appear to improve habitat quality and 'attractability' for old growth specific species.

Some of the varying results in plant and bird data over just two seasons can probably be attributed to management practices in clearing debris from the storms that affected our survey plots. There was an increase in storms and windfalls and a decrease in rain events in 2013, with subsequent increased management at SJA. Due to tree removal, machinery was driven through survey plots, disrupting plant life and plot markers; changing plant communities, richness, and biodiversity (plots in Zone 2 had increased bird edge species richness conceivably due to increased sunlight and community disturbances). Many vegetation studies research the effects of how prescribed burns and short rotation timber harvests have replaced wildfires, insect outbreaks and windstorms, low-intensity under burns and human fire suppression have changed forests, how forest structure, productivity and understory composition varies among moist availability and different environments with little

research specifically assessing the effects of management on vegetative diversity with most research focusing on needs of fauna, not flora (Halpern & Spies 1995).

This study's original prediction that old growth species would be found only in WW and SJA but not in SJU was not supported. Old growth specific species were found at WW, SJA and SJU. For these specific sites, each specific habitat type (old growth or successional) is generally similar in structure. This study found strong vegetative diversity patterns among seasons, zones and sites with what appears to be no difference between the sizes of the outer buffer zones. The strongest pattern occurred when comparing the sites differences possibly due to management. Both SJA and SJU are management by Tom Kroll (but with different directives), while WW is managed by the county parks commission.

The initial expectation of our study was that buffer size would explain variation in species composition, but ultimately site location and assumedly, land management influenced the results more than anticipated. A comprehensive scientific understanding of natural forest succession is imperative if managers are to design successful management plans regardless of fragment size (Franklin *et al*. 2002). Although statistical significance was rare in our study, these forest fragments are significant to the species that live there.

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APPENDIX

Table A. Global Positioning System (GPS) coordinates for the research plots.

GPS Coordinates		
PlotCode	N	W
SJA A 1	45.57866	-94.37758
SJA A 2	45.57592	-94.38094
SJA A 3	45.57364	-94.38244
SJA A 4	45.57162	-94.38055
SJA B 1	45.57939	-94.37638
SJA B 2	45.58025	-94.37241
SJA B 3	45.58159	-94.36874
SJA B 4	45.58128	-94.36563
WW A 1	45.55618	-94.38226
WW A 2	45.55352	-94.38223
WW B 1	45.55576	-94.37690
WW B 2	45.55325	-94.37720
SJU A 1	45.56766	-94.39815
SJU B 1	45.56702	-94.39722

Table B. List of tree species with average diameter at breast height (DBH) in cm per zone noted in table. Tape measure used to exact DBH.

Average DBH (cm) per Zone	Zone 1	Zone 2	Zone 3	Zone 4
American basswood (Tilia americana)	35	36	0	29
Black ash (Fraxinus nigra)	33	0	0	0
Black cherry (Prunus serotina)	0	8	0	0
Green ash (Fraxinus pennsylvanica)	0	0	0	45
Hophornbeam ironwood (Ostrya virginiana)	8	8	8	0
Northern red oak (Quercus rubra)	71	64	59	51
Red maple (Acer rubrum)	0	15	38	0
Silver maple (Acer saccharinum)	36	0	0	0
Sugar maple (Acer saccharum)	36	31	30	15
White oak (Quercus alba)	44	48	71	0
Yellow birch (Betula alleghaniensis)	0	38	0	0

Table C. Comprehensive listing of averaged percent canopy cover of plot obtained using a forest densitometer with readings taken between midmorning and early afternoon on a sunny day in early September, 2013.

Site-Target-Zone	% Canopy Cover
SJA A 1	99.74
SJA B 1	100
WW A 1	98.7
WW B 1	98.18
SJU A 1	99.74
SJU B 1	100
SJA A 2	92.46
SJA B 2	98.7
WW A 2	97.66
WW B 2	96.1
SJA A 3	95.84
SJA B 3	98.7
SJA A 4	98.18
SJA B 4	98.96

Table D. A comprehensive list of the 82 species documented and the percentage of time each was detected in Zones 1-4 in 2012 and 2013.

Vegetative Species Classification	2012				2013			
(Deep Forest-Zone 1)	Zone1	Zone 2	Zone 3	Zone 4	Zone 1	Zone 2	Zone 3	Zone 4
Canada Mayflower *Maianthemum canadense*	0.00	0.50	0.00	0.50	0.00	0.25	0.00	0.50
Clayton's sweet cicely *Osmorhiza claytonii*	0.50	1.00	1.00	1.00	0.50	0.75	1.00	1.00
Common blue violet *Viola sororia*	0.33	0.50	0.00	0.50	0.33	0.75	0.00	0.50
Early meadow-rue *Thalictrum dioicum*	0.50	0.75	0.00	0.00	0.17	0.50	0.00	0.00
Eastern leatherwood *Dirca palustris*	0.67	0.25	0.00	0.00	0.50	0.25	0.00	0.00
Eastern prickly gooseberry *Ribes cynosbati*	1.00	1.00	1.00	1.00	0.83	1.00	1.00	1.00
Fern - Common Lady *Atherium filix-femina*	1.00	1.00	1.00	0.50	0.33	0.75	0.50	0.00
Fern - Spinulose wood *Dryopteris carthusiana*	0.00	0.25	0.50	0.50	0.00	0.25	0.00	0.00
Fragrant bedstraw *Galium triflorum*	0.50	0.75	1.00	1.00	0.33	0.25	0.50	1.00
Indianpipe *Monotropia uniflora*	0.00	0.25	0.50	0.00	0.00	0.00	0.00	0.00
Jack-in-the-pulpit *Arisaema triphyllum*	1.00	0.75	1.00	0.50	1.00	0.75	1.00	0.50
Large-leaved aster *Aster macrophyllus*	0.50	0.50	0.50	1.00	0.17	0.75	0.50	0.50
Largeflowered bellwort *Uvularia grandiflora*	0.00	0.00	0.00	0.00	0.17	0.00	0.00	0.00
Nodding trillium *Trillium cernuum*	0.17	0.25	0.00	0.00	0.67	0.25	0.00	0.50
Rough bedstraw *Galium asprellum*	0.83	0.75	0.50	1.00	0.50	0.75	0.50	1.00
Showy orchis *Galearis spectabilis*	0.00	0.00	0.00	0.50	0.00	0.00	0.00	0.00

Yellow violet *Viola pubescens*	1.00	1.00	1.00	1.00	0.67	0.50	1.00	1.00
Zigzag goldenrod *Solidago flexicaulis*	0.17	0.50	0.00	0.00	0.00	0.75	0.00	0.00
(Ecotone-Zones 2-4)								
American ginseng *Panax quinquefolius*	0.00	0.25	0.00	0.00	0.00	0.50	0.00	0.00
American hog peanut *Amphicarpa bracteata*	0.17	0.75	0.50	1.00	0.00	0.75	0.50	0.50
Baneberry - red *Actaea rubra*	0.00	0.25	0.00	0.00	0.00	0.25	0.00	0.00
Baneberry – white *Acetaea pachypoda*	0.00	0.25	0.00	0.50	0.00	0.00	0.00	0.00
Beggarslice *Hackelia virginiana*	0.17	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Black snake root *Sanicula canadensis*	0.33	0.50	0.50	0.50	0.33	0.50	0.50	0.00
Bloodroot *Sanguinaria canadensis*	0.17	0.75	0.50	1.00	0.33	0.50	0.50	0.50
Blue cohosh *Caulophyllum thalictroides*	0.17	0.25	0.00	0.00	0.17	0.25	0.00	0.00
Broadleaf wild leek *Allium ampeloprasum*	0.00	0.00	0.00	0.00	0.17	0.25	0.50	0.50
Buckthorn *Rhamnus*	0.17	0.00	0.50	0.00	0.17	0.50	0.50	0.50
Bull thistle *Cirsium vulgare*	0.00	0.25	0.00	1.00	0.00	0.25	0.00	0.00
Burdock *Asteraceae arctium*	0.00	0.25	0.00	0.00	0.00	0.00	0.00	0.00
Burr cucumber *Sicyas angulatus*	0.00	0.00	0.00	0.50	0.00	0.00	0.00	0.50
Canada goldenrod *Solidago canadensis*	0.00	0.25	0.00	0.50	0.00	0.00	0.00	0.00
Canada thistle *Cirsium arvense*	0.00	0.25	0.00	0.00	0.00	0.00	0.00	0.00
Canadian honewort *Cryptotaenia canadensis*	0.17	0.00	0.50	1.00	0.00	0.00	0.00	0.00
Canadian wildginger *Asarum canadense*	0.00	0.25	0.00	0.00	0.00	0.00	0.00	0.00
Carrion flower *Smilax herbacea*	0.17	0.50	0.50	0.50	0.50	0.50	0.50	0.00

Common chickweed *Stellaria media*	0.00	0.25	0.00	0.50	0.00	0.00	0.00	0.00
Common dandelion *Taraxacum officinale*	0.33	1.00	1.00	1.00	0.50	1.00	1.00	1.00
Common enchanter's nightshade *Circaea lutetiana*	0.33	0.50	1.00	0.50	0.17	0.25	0.50	1.00
Common yarrow *Achillia millefolium*	0.00	0.25	0.00	0.00	0.00	0.00	0.00	0.00
Corn gromwell *Buglossoides arvensis*	0.17	0.50	1.00	0.50	0.00	0.00	0.00	0.00
Dewey's Sedge *Carex deweyana var.* *deweyana*	0.67	0.50	1.00	1.00	0.67	0.50	1.00	1.00
False solomon seal *Smilacina racemosa*	0.33	0.25	0.50	0.50	0.67	0.75	1.00	1.00
Fern - Interrupted *Osmunda claytoniana*	0.17	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Fern – Ostrich *Matteuccia struthiopteris*	0.00	0.25	0.00	0.50	0.00	0.25	0.00	0.00
Fern - Rattlesnake *Botrychium virginianum*	0.50	0.75	0.00	0.00	0.17	0.00	0.00	0.50
Green headed cone flower *Rudbeckia laciniata*	0.00	0.25	0.00	0.00	0.00	0.00	0.00	0.00
Hairgrass *Aira L.*	0.17	0.50	1.00	1.00	0.17	0.50	1.00	1.00
Heartleaved aster *Aster cordifolius*	0.00	0.25	0.00	0.00	0.00	0.00	0.00	0.00
Helianthus *Helianthus*	0.17	0.50	1.00	0.50	0.00	0.25	0.00	0.00
Hepatica *Hepatica acutiloba*	0.83	0.25	0.00	0.00	0.67	0.25	0.00	0.50
Honeysuckle *Lonicera*	0.00	0.25	0.00	0.00	0.00	0.00	0.00	0.00
Hooked crowfoot *Ranunculus recurvatus*	0.00	0.00	0.50	0.00	0.00	0.00	0.00	0.00

Jewelweed *Impatiens pallida*	0.33	0.75	1.00	0.50	0.17	0.50	1.00	0.50
Kidney-leaved buttercup *Ranunculus abortivus*	0.00	0.00	0.50	0.00	0.00	0.00	0.00	0.00
Little-leaf buttercup *Ranunculus abortivus*	0.00	0.00	0.00	0.00	0.17	0.00	0.00	0.00
Longstalk sedge *Carex pedunculate*	0.17	0.00	0.00	0.50	0.00	0.00	0.00	0.00
Lopseed *Phryma leptostachya*	0.17	0.00	0.00	0.50	0.00	0.00	0.00	0.00
Mead's sedge *Carex meadii*	0.33	0.50	1.00	0.50	0.33	0.50	1.00	0.50
Narrowleaf whitetop aster *Sericocarpus linifolius*	0.17	0.00	0.00	0.50	0.00	0.00	0.00	0.00
Nodding onion *Allium cernuum*	0.17	0.25	0.00	0.00	0.00	0.00	0.00	0.00
Perennial sow thistle *Sonchus arvensis*	0.00	0.50	0.00	0.50	0.00	0.00	0.00	0.00
Perfoliate bellwort *Uvularia perfoliata*	0.83	0.75	1.00	0.50	0.67	0.75	0.50	0.50
Prickly ash *Zanthoxylum americanum*	0.00	0.75	0.00	0.00	0.00	0.25	0.00	0.50
Red columbine *Aquilegia canadensis*	0.00	0.50	0.00	0.00	0.00	0.25	0.00	0.00
Riverbank grape *Vitis riparia*	0.00	0.25	0.00	0.00	0.00	0.00	0.00	0.00
Sessile leaf bellwort *Uvularia sessilifolia*	0.00	0.00	0.50	0.00	0.00	0.00	0.00	0.00
Smooth solomon's seal *Polygonatum biflorum*	1.00	1.00	1.00	1.00	1.00	0.75	1.00	1.00
Spotted coralroot *Corallorhiza maculata*	0.17	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Starflower *Trientalis borealis*	0.33	0.25	0.00	1.00	0.00	0.00	0.00	0.00
Stinging nettle *Urtica dioica*	0.83	1.00	1.00	1.00	0.00	0.00	0.00	0.00

Tall meadow rue *Thalictrum polygamum*	0.17	0.25	0.00	0.00	0.00	0.50	0.00	0.00
White arrowleaf aster *Symphyotrichum urophyllum*	0.00	0.00	0.00	0.00	0.17	0.25	0.00	0.00
White snakeroot *Ageratina altissima*	0.00	0.25	0.00	1.00	0.00	0.00	0.00	0.00
Wild cucumber *Ageratina altissima*	0.00	0.00	0.50	0.50	0.00	0.00	0.00	0.00
Wild geranium *Geramium Maculatum*	0.50	0.50	0.50	0.50	0.67	0.50	0.50	0.00
Wild sarsaparilla *Aralia nudicaulis*	0.00	0.25	0.00	0.00	0.00	0.00	0.00	0.00
Wild strawberry *Fragaria virginiana*	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.50
Wood anemone *Anemone quinquefolia*	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
Wood nettle *Laportea canadensis*	0.17	0.25	0.00	0.00	0.67	0.50	0.50	1.00
Woodbine *Parthenocissus vitacea*	0.00	0.75	0.00	0.00	0.00	0.25	0.00	0.50
Woodland muhly grass *Muhlenbergia sylvatica*	0.67	0.50	1.00	1.00	0.67	0.50	1.00	1.00

Table E. Tree species identified at all three sites.

Tree Species
American basswood *(Tilia americana)*
Black ash *(Fraxinus nigra)*
Black cherry *(Prunus serotina)*
Green ash *(Fraxinus pennsylvanica)*
Hophornbeam ironwood (*Ostrya virginiana*)
Northern red oak *(Quercus rubra)*
Red maple *(Acer rubrum)*
Silver maple *(Acer saccharinum)*
Sugar maple *(Acer saccharum)*
White oak *(Quercus alba)*
White pine *(Pinus strobus)*
Yellow birch *(Betula alleghaniensis)*

Table F. An inclusive listing of 52 bird species identified and the percentage of time each was detected in Zones 1-4 in 2012 and 2013.

Avian Species Classification	2012				2013			
(Old-Growth- Zone 1)	Zone 1	Zone 2	Zone 3	Zone 4	Zone 1	Zone 2	Zone 3	Zone 4
Barred Owl *Strix varia*	0.17	0.25	0.0	0.0	0.17	0.25	0.0	0.0
Black-and-white Warbler *Mniotilta varia*	0.0	0.25	0.0	0.0	0.0	0.25	0.0	0.50
Blue-headed Vireo *Vireo solitarius*	0.0	0.0	0.0	0.0	0.17	0.0	0.0	0.0
Cerulean Warbler *Setophaga cerulea*	0.0	0.0	0.50	0.0	0.0	0.0	0.0	0.0
Great Crested Flycatcher *Myiarchus crinitus*	0.67	0.75	0.50	0.50	0.33	0.50	0.0	0.0
Hairy Woodpecker *Picoides villosus*	0.50	0.50	0.0	0.50	0.33	0.0	0.0	0.0
Nashville Warbler *Oreothlypis ruficapilla*	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.50
Ovenbird *Seiurus aurocapilla*	1.00	1.00	.50	1.00	1.00	1.00	1.00	1.00
Pileated Woodpecker *Dryocopus pileatus*	0.50	0.50	0.50	0.50	0.83	0.0	0.0	0.0
Red-eyed Vireo *Vireo olivaceus*	1.00	1.00	1.00	1.00	1.00	0.75	1.00	1.00
Red-shouldered Hawk *Buteo lineatus*	0..50	0.0	0.50	0.0	0.33	0.50	0.0	0.0
Scarlet Tanager *Piranga olivacea*	0.0	0.25	0.50	0.50	0.67	0.50	0.50	0.50
Wood Duck *Aix sponsa*	0.0	0.0	0.0	0.0	0.33	0.25	0.0	0.0
Wood Thrush *Hylocichla mustelina*	0.0	1.00	0.50	1.00	0.0	0.50	1.00	0.50
Yellow-rumped Warbler *Setophaga coronata*	0.0	0.0	0.0	0.0	0.0	0.50	0.50	0.0

Yellow-throated Vireo *Vireo flavifrons*	0.50	0.50	0.0	0.0	0.0	0.0	0.50	1.00
(Generalist-Zone 2)								
American Crow *Corvus brachyrhynchos*	1.00	0.50	1.00	0.0	0.83	1.00	0.50	0.50
American Goldfinch *Carduelis tristis*	0.50	0.50	0.0	0.50	0.50	0.75	1.00	1.00
American Robin *Turdus migratorius*	0.0	0.25	0.50	0%	0%	0%	0%	0.50
Baltimore Oriole *Icterus galbula*	0.0	0.25	0.0	0.0	0.0	0.25	0.50	0.50
Black-capped Chickadee *Poecile atricapilla*	0.33	0.75	1.00	1.00	0.33	1.00	0.50	0.50
Bluejay *Cyanocitta cristata*	1.00	0.75	1.00	1.00	1.00	1.00	0.50	1.00
Chipping Sparrow *Spizella passerina*	0.17	0.25	0.0	0.0	0.17	0.0	0.0	0.0
Downy Woodpecker *Picoides pubescens*	0.33	0.0	0.0	0.0	0.67	0.50	0.0	0.50
Northern Flicker *Colaptes auratus*	0.0	0.25	0.0	0.0	0.0	0.25	0.0	0.0
Red-bellied Woodpecker *Melanerpes carolinus*	0.83	0.50	1.00	0.0	0.50	0.75	1.00	0.50
Red-winged Blackbird *Agelaius phoeniceus*	0.17	0.25	0.50	0.0	0.17	0.50	1.00	0.0
breasted Grosbeak *Pheucticus ludovicianus*	0.0	0.0	0.50	0.0	0.0	0.0	0.50	0.50
Turkey Vulture *Cathartes aura*	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.50
White-breasted Nuthatch *Sitta carolinensis*	1.00	1.00	1.00	0.50	1.00	1.00	1.00	0.0

White-throated Sparrow *Zonotrichia albicollis*	0.0	0.25	0.0	0.0	0.0	0.0	0.0	0.0
(Successional-Zone 3)							
American Redstart *Setophaga ruticilla*	0.17	0.50	0.50	0.50	0.50	0.25	0.0	0.50
Chestnut-sided Warbler *Setophaga pensylvanica*	0.0	0.0	0.0	0.0	0.0	0.25	0.0	0.0
Common Yellowthroat *Geothlypis trichas*	0.50	0.75	0.50	0.50	0.17	0.25	0.50	0.50
Eastern Wood Pewee *Contopus virens*	1.00	1.00	0.50	0.50	1.00	0.50	1.00	0.50
Golden-winged Warbler *Vermivora chrysoptera*	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.50
Least Flycatcher *Empidonax minimus*	0.0	0.0	0.0	0.0	0.0	0.0	0.50	0.0
Tennessee Warbler *Oreothlypis peregrina*	0.0	0.0	0.0	0.0	0.33	0.0	0.0	0.50
Veery *Catharus fuscescens*	0.0	0.0	0.0	0.50	0.0	0.0	0.0	0.0
Yellow Warbler *Setophaga petechia*	0.0	0.0	0.0	0.0	0.17	0.25	0.0	1.00
Yellow-bellied Sapsucker *Sphyrapicus varius*	0.0	0.0	0.0	0.0	0.17	0.0	0.0	0.0
Yellow-billed Cuckoo *Coccyzus americanus*	0.0	0.0	0.0	0.50	0.0	0.25	0.50	0.0
(Incidental-Zone 4)							
Blackpoll Warbler *Setophaga striata*	0.17	0.0	0.0	0.0	0.0	0.25	0.0	0.0
Canada Goose *Branta canadensis*	0.67	0.75	0.50	0.0	1.00	0.50	1.00	0.50

Common Loon *Gavia immer*	0.0	0.25	0.0	0.0	0.33	0.50	0.0	0.0
Killdeer *Charadrius vociferus*	0.0	0.0	0.0	0.0	0.17	0.0	0.0	0.0
Mallard *Anas platyrhynchos*	0.0	0.0	0.50	0.0	0.0	0.0	0.50	0.0
Red-headed Woodpecker *Melanerpes erythrocephalus*	0.17	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Ring-necked Pheasant *Phasianus colchicus*	0.0	0.25	0.0	0.0	0.0	0.0	0.0	0.0
Sand hill Crane *Grus canadensis*	0.17	0.0	0.50	0.0	0.17	0.0	0.0	0.0
Spotted Sandpiper *Actitis macularia*	0.17	0.0	0.0	0.50	0.0	0.25	0.0	0.0
Swamp Sparrow *Melospiza georgiana*	0.0	0.25	0.50	0.50	0.0	0.25	0.50	0.0