

**Testing the existence and extent of impacts of double-crested cormorant  
(*Phalacrocorax auritus*) nesting on three islands in Lake Erie**

by

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**Author's declaration**

I hereby declare that I am the sole author of this thesis. This is a true copy of the thesis, including any required final revisions, as accepted by my examiners.

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## Abstract

Managing protected areas requires consideration of broad scale social, ecological and in some cases, political factors. Additionally, the impacts of hyperabundant species is often a concern in protected areas as increases in breeding and foraging pressures can alter habitats dramatically. The influences of hyperabundant nesting colonies of waterbird species are well documented. Many studies indicate that waterbird species can influence canopy species as especially on the shoreline of islands, where the most abundant nesting pressures occur. On Lake Erie, *Phalacrocorax auritus* populations have increased dramatically from a low of 87 nests in 1979 to 16,050 nests in 2007. However, most studies have failed to address the influences that mature tree death may have on the ground layer ecosystem, including increased litter depth, herbaceous species composition and abundance and seed bank composition and viability. Consequently this study sought to quantify the nesting colonies' influence on coarse woody litter and how nest densities and litter depth influence the herbaceous layer, the seed bank composition and viability across the extent of three Lake Erie islands.

My study was designed to quantify the nesting colonies' influence on these variables across the extent of each island, rather than just the perimeters where the majority of cormorant nests and visible damage exists. I collected my data in the summer of 2008 on East Sister Island, Middle Island and West Sister Island, using plotless-point quarter method, herbaceous layer surveys (incorporating data collected by Parks Canada and Ontario Parks in 2004 and 2007), cormorant nest counts, soil seed bank cores, and litter depth measurements. To test whether there are specific impacts of cormorant nests relegated to the island perimeters versus the rest of the island, an available subset of data on East Sister Island and Middle Island were used. These were expressed as vegetation 'damage indices' as classified by Parks Canada and Ontario Parks staff. Kruskal-Wallis tests were used to determine if the different damage indices reveal any influence of cormorant nest density in smaller localized areas as opposed to testing the entirety of the islands. Kruskal-Wallis tests indicated that, island-wide, none of the herbaceous species abundance, diversity (total, native or exotic), seed bank composition, or seed viability has been affected significantly ( $P > 0.05$ ) by either cormorant nest numbers or large-diameter litter-fall from cormorant nesting activities. However, damage indices (calculated using aerial photography) did indicate that, *P. auritus* nest density does influence litter depth, herbaceous species abundance and diversity in the island perimeter. Nest density has not significantly affected seed bank abundance or seed viability. My conclusion is that cormorants do not uniformly affect the islands as there is localized damage to the herbaceous layer, restricted primarily to the perimeters of the islands. However, there remains a risk to the interior herbaceous layer of the island if the effects of nesting pressures at the edges advance inward from this perimeter.

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## 1 Overview

My research was designed to test the total impacts of a hyperabundant<sup>1</sup> species, the double-crested cormorant (*Phalacrocorax auritus* Less. [Pelecaniformes: Phalacrocoracidae]) on forest habitat on three islands in Lake Erie: East Sister Island (under the aegis of Ontario Parks), Middle Island (managed by Parks Canada), and West Sister Island (conjointly managed by U.S. Fish and Wildlife Service and Ohio Wildlife Service). The context for this is the literature on how best to assess, measure, and act upon perceived effects of wildlife species that are recovering to the level of hyperabundance. The recovery of *P. auritus*, coincident with increased regulation of environmental impacts especially restrictions on DDT, has fuelled controversy because there are visual impacts on the perimeters of shoreline vegetation, i.e. the trees where this species prefers to nest. However, it is not clear if the damage has spread beyond a narrower perimeter across the islands, if this is likely to happen or how the nesting density increase is influencing the ground layer ecosystem. My research was the first step in a wider assessment. The longer term implications of my research will help Parks agencies determine the extent of ecological restoration needed on the islands by informing these agencies about the extent of the damage incurred to the islands from double-crested cormorant nesting on formerly unstudied components of the community. Before focusing on the specifics of my study (methods, results, discussion), I use section 2 to frame my research within the larger context of protected areas management and impacts of hyperabundant species.



Figure 1: The shoreline of East Sister Island, summer 2008 showing impacts of cormorant nesting on the perimeters of the islands © Darby McGrath.

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<sup>1</sup> The term “hyperabundance” refers to wildlife population when its size “clearly exceeds the upper range of natural variability that is characteristic of the ecosystem” (McShea et al. 1997; Parks Canada Agency, 2007). Additionally, hyperabundance is context dependent and to warrant intervention by Parks officials, the population must be: impacting the ecological integrity of the parkland; threatening a Species at Risk (SAR); influencing an ecosystem outside of historical or modeled range of variation; or the hyperabundance must be a result of an alteration to “natural population regulations” (Parks Canada Agency, 2007).



Figure 2: Picture of the shoreline of Middle Island, summer 2008; evidence of excessive guano deposition on the perimeter of the island © Darby McGrath.

## **2. Literature Review**

### **Protected Areas and Ecosystem Management**

Many issues surround protected areas and how we choose to manage them, e.g. agriculture, commercial land development, urban development, forestry, gas, oil and mining development as well as exotic species invasion, and hyperabundance of wildlife (Kiringe et al. 2007; Narayanan and Vijayan, 2008; SelKoe et al. 2008; Axini and Tofan, 2009; Warburton and Norton, 2009). The myriad issues surrounding the management of these areas often stems from anthropogenic activities (Kiringe et al. 2007). Human influences undoubtedly threaten most protected areas making management difficult. Conflicting values are entrenched in the preservation of ecosystems including when and how to manage these areas and how much money should be allocated to such endeavours (Wilshusen et al. 2002; Holl and Crone, 2004; Warburton and Norton, 2009). Additionally, these areas are important tourist destinations, especially within Canada, increasing the pressure on managers for effective conservation and restoration efforts. A core problem is that ecosystem degradation is cumulative yet also tends to occur in a sudden dramatic manner (e.g. witness the collapse of fisheries worldwide). It may take a starker series of examples of ecosystem collapses before stakeholders are willing to act

and there is difficulty in defining acceptable means for governing these areas (Wilshusen et al. 2002). The conflict over determining an action plan for management of protected areas pits people with “protectionist values” (those who want to protect nature for “nature’s sake” and scientific approaches) against those with “people-oriented” values (those groups that are focused on tourism and development) (Wilshusen et al. 2002). These types of sometimes dichotomous views can complicate, for example, the ecological restoration<sup>2</sup> of protected areas. However, restoring protected areas is often essential for the recovery or preservation those areas and species at risk therein (Plummer and Fennell, 2009; Mouillot et al. 2007; Buisson and Dutoit, 2006) – though it may be years before the relative success of protection via restoration is testable (Holl and Crone, 2004; Murphy, 2005). The following table represents a brief overview of some of the most common threats or issues associated with protected areas in Canada with the relative threatening activities and causes.

Table 1: Some of the Main Threats to Protected Areas in Canada

<b>Threat/ Issues</b>	<b>Concerns</b>	<b>Causes of Threats/ Outcomes</b>
<b>Information Gaps</b>	Information gaps regarding species numbers as well as ecosystem monitoring; information gaps between managers of federal and provincial parks; lack of comprehensive planning between managers of cross-border ecosystems (e.g. Provincial as well as U.S./ Canada ecosystems)	Differing political motivations; lacking of data sharing; financial obligations and restrictions
<b>Budget</b>	Necessary monitoring and restoration efforts often not feasible within manager's budget; requires prioritization of activities; unforeseen negative impacts	Changes in political leadership; economic downturns; increases in environmental problems

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<sup>2</sup>Ecological restoration: repairing degraded, damaged or destroyed ecosystems (SER, 2004).

<b>Visitor Impacts</b>	Visitors to parks can negatively influence ecosystems e.g. fires; dispersal of invasive species propagules; disregard for trail markers etc.; illegal visitors to restricted areas can influence Species at Risk (SAR) and nesting colonies and breeding colonies	Lack of education on impacts; disregard for regulations; lack of security and monitoring of restricted areas
<b>Agricultural Encroachment</b>	Expansion of agricultural lands into wildlife rangelands often incompatible with wildlife biodiversity requirements; pollution from runoff	Increased population and demand for resources; improvements in farming machinery makes large-scale farming feasible; increases in commercial agricultural businesses; restricts wildlife movement outside of the protected area
<b>Invasive Species</b>	Introduction of invasive species to protected areas; increases in disturbances that encourage and foster invasive species	Decrease native species abundances e.g. fragile SARs; increases in non-native species diversity
<b>Natural Resource Exploitation</b>	Overfishing, mining and logging in and around protected areas	Pressures and demands for exports; increased population and demand for resources; commercial interests in timber industry; lack of policies that focus on sustainability
<b>Blocked Migratory Corridors</b>	Human presence can divert migratory species; habitat fragmentation	Restriction of wildlife movement through rangelands and migratory pathways; permanent reduction in the size of rangelands and regional movement of species
<b>Species Density Fluctuations</b>	Alterations of habitats can widely influence species populations; persecution and conversely protection of species	Dramatic decreases in population numbers as well as dramatic increases in some cases e.g. white tailed deer and double-crested cormorants; influences food web dynamics; conflicts between human interests and wildlife

Restoration can be unpredictable in some cases; however, researchers have tried to limit the amount of uncertainty via controlled experimental designs where possible. Nonetheless, ongoing disturbances can influence restoration projects and can sometimes result in unpredictable or undesirable outcomes during regeneration (Temperton and Hobbs, 2004). Suding et al. (2004) argue that feedback loops can affect how systems reassemble after a disturbance. By researching what factors constrain community membership in a system, we can limit the uncertainty in restoration projects (Fattorini and Halle, 2004). A useful approach is to incorporate descriptions of the primary causes of damage, changes in the abiotic components, (i.e. a change in light regime, nutrients or litter depth), and biotic changes, for more targeted and successful restoration approaches. Increasing the data collection and monitoring allows for more reliable evaluation of the success of the project upon completion (Fattorini and Halle, 2004).

Holl and Crone (2004) argue that restoration success is difficult to evaluate because true experimental replicates do not exist in the natural world. Reference sites are important in ecological experimentation and restoration because similar sites (e.g. similar in age, composition, or function) can be used as suitable replicates according to Holl and Crone (2004). However, different pressures exerted on the sites chosen as replicates may be influenced by the differences of patch size and/or isolation of the ecosystem (Holl and Crone, 2004; Tanentzap et al. 2009), which are not being evaluated in this study. Studies with multiple components can increase the complexity of the study and can be difficult to test (Holl and Crone, 2004). Although management and restoration of protected areas can be complex, conservation of native biodiversity loss is an important concern for managers, often provoking restoration initiatives.

## **Wildlife Management**

Muir (1911) wrote “When One tugs at a single thing in Nature; he finds it attached to the rest of the world.” In Wildlife Management the linkages and intricacies are evident (ironically) once unplanned, unstudied reduction efforts have demonstrated wildly unpredictable impacts on the equilibrium of an ecosystem. National and provincial parks are required to protect both the ecosystems and the wildlife that dwells within those habitats. In some cases this can present a delicate balancing act which requires determining when native wildlife species have become a burden on the ecosystem while trying to conserve biodiversity. There are ethical challenges associated with reduction efforts, because decisions are made with high degrees of uncertainty and low knowledge of management outcomes (Suding et al. 2004; Warburton and Norton, 2009). Other sources of complexity stem from resource use as the relationship between population density of the critical species and the resource use is often non-linear (Warburton and Norton, 2009). Effective management requires the mitigation of habitat

pressures through reduction efforts. This requires recognition of an acceptable population level, and reduction efforts must reach that target goal (Warburton and Norton, 2009). The complexity and uncertainty in addressing the focus of wildlife population reduction efforts is not a new theme when dealing with the various cormorant populations or other nuisance species both across Canada and in fact across the globe.

Hyperabundant species can threaten biological diversity in an ecosystem (Vidal et al. 1998). Biodiversity, in this study, is defined as the number of species occurring in a community resulting from migration and species dispersal within and among ecological communities (Economo and Keitt, 2008). Managers of protected areas are often concerned that native biodiversity is threatened as a result of negative pressures on the ecosystem. Maintaining biological diversity of native species requires practical knowledge of the ecological groups as well as agreements between concerned parties (regarding elements of the economy, politics and social concerns) (Buisson and Dutoit, 2006). Pressure is often placed on decision-makers to minimize perceived impacts (Edgar et al. 2008) which may be threatening the biodiversity of protected areas. However, in an effort to conciliate these concerns, management can often overlook the loss of capacity for ecosystem function as a result of the degradation of biodiversity (Mouillot et al. 2008). Without a functioning ecosystem, the conflict between stakeholders and decision-makers over conservation will be moot. As part of a focus on functional ecosystems, conservation of biodiversity is important (Milne and Bennett, 2007; Edgar et al. 2008; Mouillot et al. 2008). Attempts to conserve biodiversity with too narrow a focus are both costly and ineffective (Edgar et al. 2008; Warburton and Norton, 2009). Biodiversity preservation that looks at larger scale community interactions to conserve important habitats, including Protected Areas, is also referred to as “the ecosystem approach” (Bruner et al. 2001; Brooks et al. 2004; Pressey, 2004; Edgar et al. 2008; Bol’shakov et al. 2009). Mouillot et al. (2008) argue that species perform unique functions in ecosystems that are essential to maintain diversity. With the loss of biodiversity, a correlative of loss of ecosystem function and general health is also likely. Protected areas provide “in-situ biodiversity conservation” in an age where global biodiversity loss is accelerating (Edgar et al. 2008; Mouillot et al. 2008; Bol’shakov et al. 2009).

The complexity of managing “hyperabundant” species may be related to the conflict of individuals attempting to legitimize their own view when faced with the lethal reduction of a species. Values surrounding wildlife populations are diverse and often pose the problem of delicately balancing perceptions of unnecessary killings in conjunction of protection of biological diversity (Warburton and Norton, 2009). An inherent complexity is entrenched in managing wildlife populations because of the human tendency towards individualization of wildlife. These individual based theories, according to Warburton and Norton (2009), do not allow for consideration for control when dealing with hyperabundant species nor the

distinction between indigenous and exotic species in an ecosystem. The expansion of suburban development, for instance, has increased the conflicts between home owners and large herbivores in North America. People feel conflicted over this issue because white tailed deer (*Odocoileus virginianus* L.) and elk (*Cervus elaphus* L.) maintain a connection between homeowners and the natural landscape (Chase et al. 2002). However, other people are concerned about property damage inflicted by the wandering ungulates but options for action can be limited because seasonal hunting or prescriptive culling can be anything from illegal to not feasible (for safety reasons) to not socially acceptable in some cases (Chase et al. 2002). Alternatives such as fertility control, have been researched, but are often costly and inefficient (Merrill et al. 2006). In many of these cases, differing social values makes cooperation in solving the problem a lengthy challenge at best (Warburton and Norton, 2009). Complexity of wildlife management issues are entrenched with both the competition for legitimacy of values and the uncertainty in population management objectives.

Reaching a target population for a hyperabundant species via management is difficult because of the complexity of the effects that prescriptive reductions may have on the ecosystem. Accounting for uncertainty in reduction efforts is essential in order to minimize the collateral impacts on the ecosystem, including other wildlife inhabitants. Control efforts with little consideration of these impacts, or with too narrow a focus, can cause unpredictable perverse outcomes (Warburton and Norton, 2009). A “tragedy of the commons” can result in cases where game shooting is unrestricted unless there is a permit based or at least well organized (Jesper and Bregnballe, 2007).

A specific example of an unforeseen outcome occurred after the legal unrestricted shooting of the grey seal (*Halichoerus grypus* Fabricus) in Scotland that nearly caused extirpation in the early 1990s. Seals can be shot by any person with an authorized firearm license and suitable equipment under the United Kingdom’s Conservation of Seals Act (1970) (Butler et al. 2008). Since the seals feed on the Atlantic salmon (*Salmo salar* L.) they are shot to protect the fisheries. An intense increase in the seals in the late 20<sup>th</sup> century caused an increase in management of the population. The management plan required no target number for reduction, no record-keeping of the numbers taken and no overall monitoring plan for the harvest; this reduction effort provided no reliable way of assessing the human control impacts (Butler et al. 2008). Consequently, the tourism based on marine mammals suffered during the near collapse of the seal populations and forced managers to deal with the uncertainty embedded in the current conservation plan. The newly designed adaptive management plan addresses conservation for the salmon and the seals, tourism, stakeholders as well as the complexity of the issue by commencing adequate monitoring of harvest and conservation efforts (Butler et al. 2008). When managing in cases of great uncertainty, it is important to



have a target threshold or reduction goal based on scientific research and is essential to monitor the biodiversity response to such efforts (Warburton and Norton, 2009).

Other situations have shown more success. In New Zealand and in the United States, bovine Tuberculosis (bTB) is prevalent in both wildlife and livestock. Transmission of the disease occurs between both domestic animals and wildlife, threatening the economic stability of production for farmers and ranchers. Managing for a targeted reduction has proved successful in New Zealand because elimination of disease has provided a clear threshold to attain in control efforts (Warburton and Norton, 2009). The brush tail possum (*Trichosurus vulpecula*) has been targeted for reduction because the species transmits bTB to domestic animals. According to Warburton and Norton (2009) these reduction efforts have been successful because a reduction threshold level was targeted at which prevents persistence of the disease in the ecosystem. Horan et al. (2008), however, argue that it is not enough to target the vectors for the disease to eliminate the spread. Wildlife populations, although considered to be the reservoir for the disease, cannot merely be managed below the exogenous host density threshold according to Horan et al. (2008). Too much uncertainty exists in a model that merely attempts to eliminate the disease because the ecological and economic systems are jointly determined (Horan et al. 2008). In Michigan, white tailed deer (*Odocoileus virginianus* L.) transmit the disease to livestock. Horan et al. (2008) argue that modeling for the reduction of bTB must assume responsibility of the endogenous nature of ecological relations and the associated management tradeoffs; such as the ecosystems services and damages provided by the host and pathogens. Target thresholds for population reductions are only one component of management efforts, because social, economic and ecological complexities are embedded in wildlife management.

### **Impacts of *Phalacrocorax* as a Management Issue in Protected Areas**

Management of *Phalacrocorax* is a good example of the uncertainty and complexity that surrounds population hyperabundance. Variable management responses in differing jurisdictions have been used to mitigate the social, economic and ecological conflicts associated with increases in densities of nesting cormorants. Manitoba, for example, culled *Phalacrocorax auritus* as early as 1945 on Lake Winnipegosis because of complaints from commercial fishers; here numbers of nesting adults was reduced from 39,000 (1945) to 19,000 (1951) (Keith, 1995). In contrast, during this same period, *Phalacrocorax auritus* was not a candidate for culling as part of management in the Great Lakes provinces and states – though culls had occurred before 1940, even as early as 1634 because of perceived impacts on smaller fish populations (Wires et al. 2001). Toxic chemicals (mainly DDT) in the Great Lakes likely was the cause for severe decreases in the number of breeding pairs - from ca 900 (1950s) to 125 (1973) (Environment Canada, 2005). In the 1960s and 1970s this changed. *P. auritus* was added to the National

Audubon Society Blue List and Migratory Bird Act in the USA and environmental laws reduced uses of DDT (1972 ban for agricultural uses) and similar pollutants (Wires et al. 2001, Cuthbert et al. 2002). In addition, increased aquaculture and fish stocking in the rivers and Great Lakes probably meant increased food sources for *P. auritus*, especially since 90 percent of the channel catfish production (*Ictalurus punctatus* L.) in the United States occurs in Mississippi, Arkansas, Alabama and Louisiana, all located directly in the interior migratory path (which comprises 61 percent of the total North American breeding population) that the species follows<sup>3</sup> (Environment Canada 2005). Global climate change may exacerbate this, e.g. via lower winter mortality on their migratory path (Frederiksen et al. 2001).

Management varies across the continent. In New Brunswick, an open season was issued in the early 1990s for double-crested cormorants from the beginning of October to the end of February with no bag limit (Keith, 1995). Similarly, P.E.I. declared an open season (in 1992) for two weeks at the beginning of October with a bag limit of six cormorants per hunter (Keith, 1995). In Nova Scotia and Newfoundland, fishermen can apply for permits to cull nuisance birds although no open season is in place (Keith, 1995). Cormorants are protected and managed under provincial law, not federal, in Canada (Keith, 1995). This leads to variability in the pursuit of management because in all provinces (not including the territories) cormorants are listed as protected under provincial wildlife acts (Keith, 1995). Management of cormorant populations thus ultimately falls in the jurisdiction of each province. In Quebec reduction techniques have been used on the islands in the St. Lawrence estuary. The methods include (but are not limited to) oiling of eggs and a lethal cull (Keith, 1995); non lethal tactics have also been used across Canada, such as scarecrows and bird-bangers (the effectiveness of these tools are currently being researched). Similar techniques have been used in Ontario in Presqu'île Provincial Park as included in a five year management plan of the park (T. Dobbie, personal communication, March 4, 2009).

In the United States the conflict over management of cormorant populations is mainly due to the protected status in Migratory Bird and Game Mammal Treaty with Mexico (1972) (Trapp et al. 1995). The legislation complicates management efforts by U.S. Fish and Wildlife Service forcing the agency and other individuals to pursue reduction efforts through other avenues e.g. protection of individual property. U.S. Fish and Wildlife Service possess conflicting responsibilities regarding cormorants and aquaculture. In response to this conflict, the U.S. Fish and Wildlife Service can issue depredation permits to aquaculturists in accordance with federal regulation (Trapp et al. 1995). This allows the aquaculturist to shoot the number of cormorants specified on the permit to counteract the detrimental economic impacts that the cormorants may be causing. The government also mandates other depredation orders which

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<sup>3</sup> The other flyway for *P. auritus* is off the Atlantic coast; 12 346 pairs at 12 colonies use this route (Wires et al. 2001).

are not directly associated with aquaculture predation. It is under the Public Resource Depredation Order (PRDO) that the U.S. Fish and Wildlife Service have carried out the cull on West Sister Island (D. Sherman, personal communication. July 25, 2008). Variable numbers of cormorants are taken each year differing on a state by state basis. As a result, the United States lacks a comprehensive approach to cormorant management whereby the numbers of taken birds can be used for modeling to monitor the way the cull influences cormorant populations and the linkages between the wintering and nesting ecosystems.

The increase in *Phalacrocorax carbo* L. in the United Kingdom has led to extended research on density dependence modeling to account for the uncertainty in reduction efforts. Despite this accounting, Green (2008) still found that density dependence modeling of *P. carbo* does not always give reliable estimates of the test's strength which can cause an underestimate of the potential impact that culling can have on the colony in question. Smith et al. (2008), however, also focused on modeling approaches for the *P. carbo* to account for the uncertainty prevalent in wildlife management. Smith et al. (2008) argue that regardless of the uncertainty in wildlife management, decisions still need to be made based on the available evidence; which is not ideal especially in highly controversial situations. Collecting useful and accurate data prior to, during, and post management is essential to monitor the impacts of the reduction for social, ecological and economic responsibility. Scientific research helps to address the political uncertainties entrenched in both the reaction to hyperabundant species and the reaction to management efforts, especially lethal operatives.

### ***Phalacrocorax auritus* and the State of the Environment of Lake Erie Islands**

There has been average annual population increase in *P. auritus* of 33 per cent on the Great Lakes demonstrating the rapid and dramatic rise in population i.e. from 39 breeding pairs in 1970 to 38,115 in 1991 (Frederiksen et al. 2001, Wires et al 2001). Due to nest site fidelity, the birds return to the same islands, possibly even to the same nest or tree each year after birth (Wires et al. 2001; Environment Canada, 2005; T. Dobbie, personal communication, June 19, 2008). This means impacts, like guano, will continue to build season and after season (Verkoeyen et al. 2009 unpublished data). Additionally, the intense increase in recent years has perpetuated the illegal use of lethal means to reduce cormorant numbers mainly by fishermen in Canada (Keith, 1995). Nesting colonies of Lake Erie islands has been monitored by Canadian Wildlife Service (CWS) since 1979 using annual nest counts. The peak year for cormorant nest numbers in total seems to be 2004, with each island included in the census expressing the highest numbers of all the years. The total nests counted in the western basin for 2004 was 17,170 nests. In 2007, the total nest numbers for the western basin was 13, 948. The drop in nest numbers on West Sister Island is a result of continued annual culling efforts by U.S. Fish and Wildlife Service and Ohio Fish and Wildlife Service beginning in 2006 (T. Dobbie, personal

communication, October 23, 2008). Managers of the island plan to use the cull to manage the numbers on the island each season as deemed necessary (D. Sherman, personal communication, July 24, 2008). The following table is a brief adaption of nesting information from the annual Canadian Wildlife Service census on nesting cormorants. Included is some indicative data on increases in nesting numbers on the western basin of Lake Erie and the islands included in the study.

Table 2: Nest numbers of *P. auritus* on three Lake Erie Islands (adapted from Canadian Wildlife Service census)

Year	East Sister Island	Middle Island	West Sister Island	Total Nests in the Western basin
1993	2770	(N/A)	307	3,077
1994	2998	1011	580	4,932
2002	4824	6635	2787	14,666
2004	6028	6611	3780	17, 170
2007	4197	4688	1967	13,948

The increase in the breeding population has marked a change in consciousness of the people in the Great Lakes areas, as cormorants are seen dominating shorelines and isolated islands (Hebert et al. 2005; Duffe, 2006). These concerns have increased the research on cormorants on the Great Lakes; including how the increased nesting populations affect other colonial nesting birds and how these larger numbers may be influencing food web dynamics in the lakes. Cuthbert et al. (2002) found that *P. auritus* did not affect populations of other colonial waders (great blue heron or black crowned night heron) but parts of their studies show that the forest structure is being altered through forest cover loss and soil chemistry alterations. Marine focused research conducted by Stapanian et al. (2002) indicates that the cormorant foraging impacts are localized based on the depth of foraging as well as distance from shore, and therefore, was not then outcompeting walleye for similar forage class.

However, the main question that faces the agencies that manage these islands are not related to fish or other aquatic issues. In particular, Parks Canada and Ontario Parks are mandated to achieve ecological integrity of ecosystems and on the islands the dominant ecosystem is late successional 'Carolinian' forests (and the historical ecosystem will be the seres within that Carolinian succession). While there is some concern over species that are at their

northern range limit or otherwise unusual - the common tree species consist of Kentucky coffee tree (*Gymnocladus dioica* L. K.Koch), common hackberry (*Celtis occidentalis* L.), red ash (*Fraxinus pennsylvanica* Marsh.) and American elm (*Ulmus americana* L.) – readers should remain focused on the question of impacts on the whole ecosystem. It is too easy to get distracted by the idea that the islands are of concern simply because they are Carolinian in structure and function; the real issue is that, as with any natural feature, the Parks agencies are legally required not to allow ecological integrity to decline – and it appears cormorants may do just that.

The initial phase of putative impacts from cormorants is related to their preference to nest in the larger individuals of the three most common species listed above (Kamstra et al. 1995), so habitats with a mature late successional forest may experience any impacts. The visuals of mounds of guano and dead trees at the perimeter of the islands evoke a visceral reaction that leads to a demand for action. The trees are directly impacted by cormorant nesting activities during nest construction and also through the direct contact with highly acidic guano (Hebert et al. 2005; Koh and Hudson, 2006). Studies focusing on the impacts of double-crested cormorants on the canopy layer of island ecosystems on Lake Erie have demonstrated that they decrease canopy cover through nesting activities (Hebert et al. 2005; Duffe, 2006; Koh and Hudson, 2006). One of the components of mature tree mortality that has not been studied in these ecosystems is the effect the increase in coarse woody litter depth may have on germination of herbaceous species as well as the influence it may have on seed bank composition and viability.

The main issue, often typical of these situations, is that empirical quantitative evidence of impacts beyond the canopy layer on the island perimeter is limited (Chapdelaine and Bedard, 1995; Hebert et al. 2005; Duffe, 2006). This does not mean there are no impacts; it means that the visible impacts (the perimeter canopy) has been studied more intensively and the subtle ones (in the interior understory) have been neglected. The lag between perception of a problem and evidence for decision making is rather a common one. However, in absence of conclusive evidence, it is possible that even if the cormorant populations are within historical variations in numbers, the rapid recovery itself may be enough to cause detrimental long-term impacts on flora, especially on isolated islands in Lake Erie. More specifically, one possible ecological issue is that large numbers of nesting *P. auritus* may alter the successional trajectory of the islands (Hebert et al. 2005; Environment Canada, 2005; Frederiksen et al, 2001). Studies on other islands affected by other piscivorous colonial waterbirds during nesting and roosting, highlight some of these influences; guano coverage and nutrient input, plant clipping to construct nests, fallout from plant clipping activities, as well as increased canopy openness from nest construction, and trampling or “bird induced erosion” (McColl and Burger, 1976; Hogg and

Morton, 1983; Anderson and Polis, 1999; Sanchez-Pinero and Polis, 2000; Vidal et al. 2000; Wait et al. 2004). These problems are apparent on the three Lake Erie islands being studied.

In summary, the context here is the management concerns over the severity and extent of the impact of cormorants on island flora in the recent past on the entirety of the islands and the composition and diversity of the herbaceous layer and likely impacts in the future. Again, many of species on the islands are Carolinian, typical of those found further south in the mid-Atlantic states of the United States. Being the most southerly part of Canada, the Lake Erie islands harbour some species that are found nowhere else in the country. An example of this is the red mulberry (*Morus rubra* L.), a rare tree that is found almost exclusively on Middle Island, and may be impacted by cormorant nesting. Certain exotic invasive species threaten the biodiversity of the Carolinian life zone through habitat fragmentation and propagule pressures (Meloche and Murphy, 2006). In both federal and provincial parks in Canada, managers are legally mandated to protect the ecological integrity of park lands<sup>4</sup>. Ultimately, the question is whether the impact of double-crested cormorants will disrupt the island ecosystems to an extent that they require a large ecological restoration effort or will the impacts be restricted mainly to the perimeter where smaller scale management, including ecological restoration, is all that may be needed. The first step to answering these questions requires tests to determine if there are detectable impacts already in areas away from the island perimeter.

### **Description of 3 Main Lake Erie islands**

The Lake Erie Archipelago located between the Canadian and American shores in the western basin of Lake Erie is a composite of shoals and reefs and 22 islands (on Devonian Dundee limestone bedrock) (Kamstra et al. 1995; North-South Environmental Inc., 2004). According to most accounts, the climate on the islands is much warmer than both the surrounding mainland and other locations of the same latitude due to the climatic effect of Lake Erie (see discussion in North-South Environmental Inc., 2004). The climate is classified as temperate, humid-continental (Cooper and Herdendorf 1977, in Boerner 1984). These islands, as a result, support species of flora and fauna characteristic of West Virginia, resulting in a plant community that is found nowhere else in Canada or in any of the adjacent U.S. states (Bird Studies Canada 2000; North-South Environmental Inc., 2004). The shorelines of the islands have characteristic exposed rock ledges with limestone shelves and cobble beaches (Kamastra et al. 1995; North-South Environmental Inc., 2004).

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<sup>4</sup> Parks Canada (2009) defines Ecological Integrity as "...a condition that is determined to be characteristic of its natural region and likely to persist, including abiotic components and the composition and abundance of native species and biological communities, rates of change and supporting processes."

### **East Sister Island**

East Sister Island is the smallest (15 ha) of the islands in the study and had been subjected to less anthropogenic impacts than Middle Island and West Sister Island; East Sister Island was often cited as the most characteristic and undisturbed island in the archipelago (Hebert et al. 2005). But this may not be the case with the return of large populations of *P. auritus*. Between 1981 and 2000, double-crested cormorant nests on the island increased from 6 to 5485 on the island (Weseloh et al. 2002). Post 2000, data collection began again in 2005 by Ontario Parks.

### **Middle Island**

Middle Island (18.5 ha) was disturbed by the construction of a lighthouse to warn of dangerous shoals surrounding the island. It was also a haven for smugglers during prohibition. In 1935 a north-south strip was cleared in the center of the island to allow planes to taxi on Middle Island and a clubhouse/casino/hotel was erected. Flora and fauna not characteristic of the island were introduced, such as pigs, pheasants, squirrels as well as livestock. Further disturbances occurred such as an east-west runway along with a lagoon to harbour boats (all information was courtesy of North-South Environmental Inc., 2004).

### **West Sister Island**

A lighthouse was constructed on West Sister Island in 1821 and a lighthouse keeper and family lived on the island until 1937; in 1945 the island was used by the War Department as a military test site (U.S. Fish and Wildlife Service, 2008). Military use ended in 1951 and in 1975 the island was declared a 'wildlife refuge', where the use of any motors are prohibited (D. Sherman, personal communication). The 36 hectare island is located just north of the Ottawa Wildlife Refuge, in the Western basin of Lake Erie and is one of the largest nesting colonies for waterbirds in the U.S. portion of the Great Lakes. According to Sherman (2007) West Sister Island "hosts one of Ohio's two primary breeding colonies of black-crowned night-herons along with three state-listed birds and a bird of special concern".

## **Vegetation on the Main Lake Erie Islands**

### *Invasive Species*

Excrement levels on islands with *P. auritus* nesting colonies express a decrease in native plant diversity and an increase in exotic invasive species (Hebert et al. 2005, Rippey et al. 2002, Weseloh et al. 2002). Luh and Pimm (1993) suggest that ecological communities that are damaged are more susceptible to changes in their composition through invasions or extinctions, ultimately changing the pathway for assembly of that community. The extant data that the managers have collected is very important in this study. Managers of Middle Island

claim that they have noticed changes in the assemblage of the herbaceous layer each year that the cormorants occupy the islands for nesting (T. Dobbie, personal communication June 17, 2008.) Although the above ground data does not provide a clear picture of what is happening in the seed bank, it is important because the composition of the vegetation seems to be shifting to species that capitalize on disturbances, such as pokeweed (*Phytolacca americana* L.), lamb's quarters (*Chenopodium album* L.), garlic mustard (*Alliaria petiolata* (Bieb) Cav. & Gran.) and common motherwort (*Leonurus cardiaca* L.). For the purposes of this study, an invasive species or what I am calling an exotic species refers to a non-native species that has become established in a new location (D'Antonio and Meyerson, 2002), spread with the potential to alter the long term successional trajectory of the site (Chornesky and Randall, 2003). It is important to understand that we cannot be sure what the "original" community was comprised of, and ecosystems should be viewed as systems subject to continual change (Larson, 2005). The Carolinian forest is not a static assemblage of species, but an ever changing aggregation of species that varies over time and from place to place.

Invasive species can often outcompete native species, through the dispersal of a large number of seeds and their ability to withstand nutrient rich soil (Cole et al. 2007). Many biologists have argued that invasive species are a serious threat to native biodiversity (Trottier 1986; Douglas et al. 1990; Wilson and Pärtel 2003; Renne et al. 2006; Clements and Catling, 2007). Fragmentation facilitates invasion of weedy species, according to Murphy (2005). Murphy also argues that non-native species often colonize new areas causing a decrease in native herbaceous densities (not to be confused with species richness) and an increase in densities of these invading species (Murphy, 2005; Sax and Gaines, 2008). In many cases, species invasions have led to native species extinctions, particularly on islands (Sax and Gaines, 2008). Sax and Gaines (2008), argue, however, that the naturalization of exotic plant species on islands is an ongoing process, and the presence of new species on islands ultimately increases the species richness of the islands. In most cases, the native flora does not go extinct, although native abundances may be inhibited. However, the added dynamic functioning within these systems is the increased cormorant nesting pressures. The concern is that increased *P. auritus* nesting pressures may inhibit native species regeneration of the herbaceous layer, which is not unrealistic, given that the native canopy species are declining (Hebert et al. 2005; Duffe 2006; Koh and Hudson, 2006) and at the same time, foster non native invasive species. Additionally, Murphy (2005) argues, that garlic mustard, a common woodland invader in Southern Ontario, has the ability to compromise what he calls the vernal dam. He describes the vernal dam as the spring ephemerals that necessarily fix nutrients in an ecosystem post-winter during decomposition. These more adaptable invasive species, in conjunction with cormorant nesting pressures, may impede native herbaceous species regeneration on these islands. Murphy (2005) also suggests that in order to actively restore a site colonized by such invaders, it is important to simultaneously manage these species while pursuing restoration.



### *Soil Seed Bank*

The soil seed bank composition on these islands has never been studied. Understanding the dynamics of what causes the composition and density of a seed bank will help to uncover the most effective way to reassemble an ecosystem post-perturbation (Baskin and Baskin, 1998). This knowledge about the soil seed bank is important in an ecosystem for maintaining genetic diversity (Wang, 1997). Seed banks are constructed via a delicate relationship between the “inputs and outputs of seed to the soil” which “determines numbers, longevity, and the characteristics of the seeds that are present” (Harper, 1977; Carver, 1983; Louda, 1989). In this very important horizontal community, new species colonization may be a consequence of a disturbance in the system, drastically altering the seed bank composition (Templeton and Levin, 1979; Herbert, 1989).

The seed bank composition and density may not be a reflection of what is happening in the above ground community (Nakagoshi, 1984; Baker, 1989). According to Baskin and Baskin (1998) seed banks often persist at a site without seed immigration because disturbances or other environmental factors may reduce seed set. Hence, the extant vegetation data is an important component of this study as data has been collected for three years on Middle Island and East Sister Island and may indicate important changes or trends plus presages plant communities’ response to future disturbances or environmental conditions (Pickett and McDonnell, 1989; Fisher et al. 2009).

In these systems, I predict impacts by *P. auritus* nesting activities, has altered the dynamics of the above ground community (*viz* Fenner and Thompson, 2005; Koh and Hudson, 2006 – they also are the sources for the remaining concepts in this paragraph). In most forests the seed banks that predominate in the landscape are transient in nature because the disturbances are too infrequent to select for persistent seeds; seed bank density in woodlands is often lower than in arable fields. This may have to do with the types of herbaceous species found in arable fields compared to terrestrial uplands, because small seeds are produced in higher numbers than large seeds, and in general are more persistent in the seed bank (. The chronic disturbance regime on the islands may alter the seed bank composition. For instance, Middle Island and East Sister Island both support large quantities of common pokeweed – where pokeweed accumulates into a persistent seed bank, which is contrary to hypotheses of transient seed bank composition in woodland ecosystems.

The soil seed bank has been called the “ecological memory” of an ecosystem (Templeton and Levin, 1979; Herbert, 1989; Fisher et al. 2009). Soil seed banks do not represent an immediate rendering of what is currently occurring in a given ecosystem (Templeton and Levin, 1979; Herbert, 1989) however, sampling the soil seed bank provides a more complete picture of the evolution of a given system.

Seed bank germination is dependent on a variety of conditions since different species have different physiological dormancies (Baskin and Baskin, 1989 – with this reference as the basis for the ideas that follow). Since different species have different abiotic requirements to break dormancy, it is important to understand how different seeds in different states of dormancy respond to dormancy breaking mechanisms (i.e. light or darkness, temperature or moisture). Certain abiotic factors may be altered in the environment ultimately retarding the seeds ability to break dormancy for successful germination. Although seeds are capable of germinating over a wide array of lightness and darkness, “with physical dormancy, germination can be prevented with a lack of imbibition of water”. This suggests that a variety of consequences exist for the germination potential of the native seed bank if the litter depth is too thick, preventing necessary dormancy breaking episodes.

### *Litter*

Studies on nesting colonies of cormorants refer to the influence that nesting activities may have on the input of woody debris (hereafter described as the main component of the litter layer) into the communities housing the colonies (Meier 1981; Hobara et al. 2001; Stapanian et al. 2002; Hebert et al. 2005; Koh and Hudson, 2006)<sup>5</sup>. This increase may influence the seed bank composition and decrease herbaceous species diversity and abundance (Bedard 1995; Chapdelaine and Bedard, 1995; Rippey et al. 2002; Hebert et al. 2005; Koh and Hudson, 2006). In this study, coarse woody litter is the twigs broken off during nest construction, as well as the bark and general pieces of decaying trees that create a mulch-like consistency around the base of dead or dying nesting trees. During nest construction and due to influences of guano on nesting trees (Sanchez-Pinero and Polis, 1999) the death of mature trees is accelerated altering the composition of ground layer material and creating gaps in the canopy. Tanentzap et al (2009) found that disturbances that increase even minor canopy gaps can increase non-native seed pressures and alter the native seed bank composition of an ecosystem a common phenomenon in forest ecosystems.

Restricted seed set and germination is common especially in conifer plantations, where the combination of the acidity of the needles and the increase in the duff layer prevents regeneration of the herbaceous layer (Kauffman, 2004). Leaf litter, and litter in general, when increased can affect soil temperature, reduce light (Watt, 1970; Molofsky and Augspurger, 1992; Rinkes and McCarthy, 2007), as well as interfere with the regeneration process (Sydes and Grime, 1981; Rinkes and McCarthy, 2007). Messasoud and Houle (2006) and Mountford et al. (2006) discuss the importance of measuring litter depth as a variable affecting seedling recruitment and regeneration. The increased amount of tree litter may decrease regeneration

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<sup>5</sup> Litter can include much smaller sizes of debris but in the context of *P. auritus* nesting on these islands, the focus tends to be on the larger [coarser woody debris can inhibit seed germination according to Ellsworth et al. 2004] woody debris from the mature trees where nests are built.

of native saplings and discourage ground layer vegetation from germinating (Parent et al. 2006). Coarser litter may diminish the capability of seeds to germinate where the coarseness of the debris obstructs the emergence of the radicle and cotyledon, causing the seed to expend an excessive amount of energy on penetration of the litter layer by the hypocotyl (Sydes and Grime, 1981; Facelli and Pickett, 1991; Molofsky and Auspurger, 1992; Ellsworth et al. 2004). In order to test how the amount of litter on the floor contributes, hinders or affects regeneration and seed bank composition, the litter depth was measured. Germination is the most vulnerable stage in seedling recruitment in terms of impacts from litter (Xiong and Nilsson, 1999; Bartuszevige, 2007). The litter layer, therefore, may become an essential factor in a restoration plan for these areas. For example, the deeper litter under the areas most colonized and damaged by cormorants may actually have the most complete pre-disturbance seed bank and greater litter depth may prevent invasions by exotic species (Bray and Graham, 1964; Brothers and Spingarn, 1992; Bartuszevige, 2007).

### **3 Study Objectives**

My research was focused on assessing large scale impacts of *P. auritus* on understorey vegetation across three federally/provincially/state protected islands in Lake Erie. In the United States, most concerns are focused on how the increased population of *P. auritus* influences food web dynamics, especially desirable fish species, in the Great Lakes and other inland lakes. In Canada, managers are concerned with fish populations but are also concerned with the influences that nesting double-crested cormorants may have on unique island ecosystems on the Great Lakes. Recent studies have focused on the way that cormorant nesting can influence mature nesting trees and forest succession (Hebert et al. 2005; Duffe, 2006; Koh and Hudson, 2006) but little research has studied the way that nesting colonies may influence the interactions between the increased coarse woody debris and the regeneration of native ground layer ecosystems on these islands. I used three Lake Erie islands (Middle Island, East Sister Island and West Sister Island) to test if double-crested cormorant nesting density increases the depth of large diameter coarse woody litter. In turn, I tested if this resulted in effects on (a) above-ground relative species abundance and species diversity and (b) relative species abundance and viability of the seed bank. To provide some comparisons, I tested relative 'damage' to understorey species and seed banks between the interior and perimeter samples from my study.

### **Hypotheses**

Many interconnections exist between the nesting cormorants and their habitat. Below I highlight the ecological connections between the cormorants and their habitat on the islands; including the variables I have studied. These linkages form the basis of my hypotheses,

including the increasing litter depth in waterbird nesting colonies (Meier 1981; Hobara et al. 2001); and how herbaceous species diversity and abundance (Bedard 1995; Chapdelaine and Bedard, 1995; Rippey et al. 2002; Hebert et al. 2005; Koh and Hudson, 2006) and seed bank abundance and viability (Kauffman, 2004; Parent et al. 2006; Rinkes and McCarthy, 2007) are influencing by coarse woody litter in hyperabundant nesting colonies.

### Ecological Linkages between Variables used in the Study

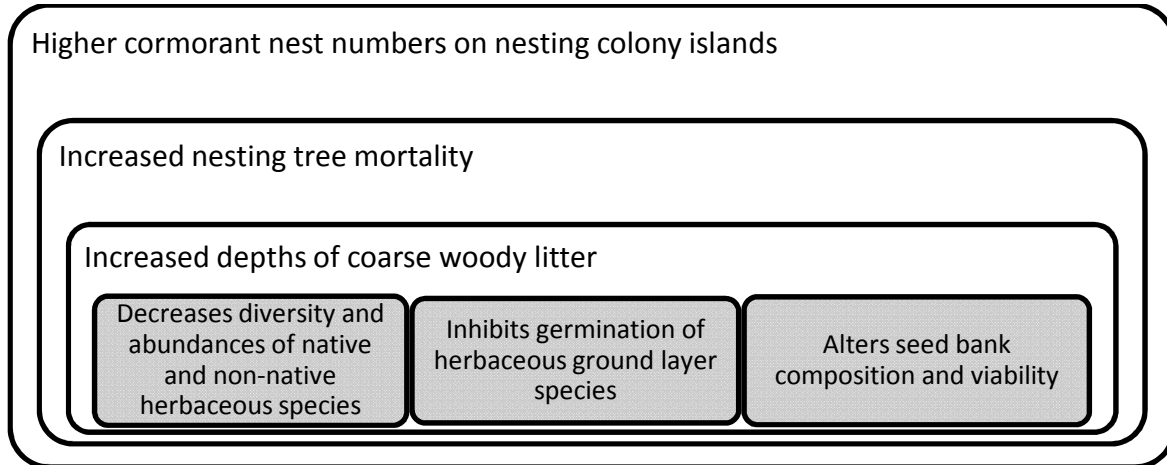


Figure 3: Descriptions of the ecological linkages between the cormorants and the ecosystem including the variables used in the study.

The hypotheses were<sup>6</sup>:

- The responses tested below will be most apparent where prolonged management has not yet occurred (East Sister and Middle Island) and where plots are under the canopy where *P. auritus* nest, i.e. where nesting densities are highest and litter likely deepest. These findings will be compared to the responses on West Sister Island (where there has been management and population reduction of *P. auritus*). As demonstrated in the Figure 3, the increase in *P. auritus* nesting densities on the Lake Erie islands may increase large pieces of coarse woody litter that then can reduce germination and seed bank recruitment on the forest floor (Hobara et al. 2001; Ellsworth et al. 2004).

<sup>6</sup> These are the main alternate hypotheses and stated in this format for a clearer narrative; readers may be assured that in statistical tests, I actually used the null version for each ('no significant differences') as the default and thus did a proper test of the null and alternate hypotheses.

- Higher densities of *P. auritus* nests significantly:
  - decreases abundance of native and exotic herbaceous species
  - decreases diversity of native and exotic herbaceous species
  - decreases seed viability of native and exotic species
  - increases litter depth
- Increased litter depth caused by higher densities of *P. auritus* nests significantly:
  - decreases abundance of native and exotic species
  - decreases diversity of native and exotic herbaceous species
  - decreases the abundance of both native and exotic seeds in the seed bank
  - prolongs viability of native species in the seed bank
- Because of lag times in impacts by cormorants, species abundances of both native and exotics in the seed bank is significantly different than the herbaceous layer

## 4 Study Design

### Site Selection

The islands in the study were chosen based on current ecological similarity as Middle Island and East Sister Island are both severely impacted by cormorant nesting colonies. West Sister Island was chosen as the control because it had similar forest composition historically to both East Sister Island and Middle Island. The islands are all located in the western basin of Lake Erie, and provide suitable replicates. I have made the assumption that West Sister Island, with population control methods being employed on the island, makes a suitable control site for the study because in the western basin of Lake Erie, it is not only similar to East Sister Island and Middle Island, it is also significantly less damaged than those islands. This assumption was made based on personal accounts from Parks Canada, U.S. Fish and Wildlife Service, and the cormorant nest count annual census numbers collected and provided by Canadian Wildlife Service.

All three islands support diverse and unique communities; on Middle Island and East Sister Island, Kamastra et al. (1995) found 73 and 190 native species respectively. According to Hebert et al. (2005) a considerable number of these species are of national and provincial significance. For example, on East Sister Island, in 1995, 22 nationally rare plant species were

identified, while on Middle Island, 26 rare plant species had been identified (Kamstra et al. 1995). Some of the species of concern on Middle Island and East Sister Island are red mulberry (*Morus rubra* L.), blue ash (*Fraxinus quadrangulata* Michx.), as well as herbaceous species like wild hyacinth (*Camassia scilloides* Raf.) Cory and Miami mist (*Phacelia purshii* Buckley).

### **My study design needed to be consistent with earlier studies**

Since some of the data collection procedures originated prior to my study, I chose to match those earlier studies for long term comparison (this issue is a common constraint, best illustrated by Price and Weltzin 2003). The same is true for the time of year of the original sampling; I elected to sample at similar times for consistency and comparability of data collected previously by professional staff ecologists at the Parks. The goal was to ensure sampling efforts were equal for each year of data collection because differential efforts can bias the herbaceous species detected and bias the amounts of nests counted or the types of nests identified. The drawback was that there are perhaps sampling designs better suited to my specific study but this would mean I had one and one half seasons' worth of data and would sacrifice the usability of the earlier data by future researchers. Readers should also be aware that access to these islands is both legally restricted and the islands are difficult and expensive to reach – this may not be obvious because of the word “lake” but Lake Erie is called a “Great Lake” for its size and resultant rough weather and waters. This limited the number of times I could sample.

**Field Surveys: Overview.** On June 17 and 19, 2008 nest count data and the ground layer surveys were completed on Middle Island. On June 18, 2008 nest count data and the ground layer surveys were completed on East Sister Island. On July 21, 22 and 24, 2008 the protocols for nest counts and ground layer vegetation surveys (along with the supplementary data collection methods Parks Canada and Ontario Parks use, not included in my study) were set up and data was collected for the initial season on West Sister Island with the aid of U.S. Fish and Wildlife Service and Ohio Wildlife Service.

### **Nest Counts**

Nest count data was collected using a plotless point centered quarter method (PCQ) as per Mueller-Dombois and Ellenberg (1974). For Middle Island, Parks Canada had established twelve North- South transects for a total of 52 sampling sites along each transect, with each site located at a distance of 50m apart along the transects. On East Sister Island, in 2004, a total of 11 North-South transects for a total of 29 sampling sites were established, with each site located at a distance of 50m apart along the transects (Koh and Hudson, 2006). On West Sister Island, the plot locations were already established along transects and a sub-sample of 60 plots was chosen on a representative basis.

At each of the sites, PCQ was used to obtain nest counts along with tree health data used by the parks. According to the data collection records by the park, (as dictated by PCQ methods in Cottam and Curtis, 1956), the area surrounding each central point was divided into four quadrants; north-west, north-east, south-east and south-west. A minimum distance of 10m was established for proximity to the forest edge. Two pieces of rope ten metres in length were attached to the centre of the plot where a piece of rebar resides to mark the permanent plot. Two researchers pulled each piece of rope out until they were fully extended. The researchers used the lengths of rope to indicate a circle with a ten metre radius, and a circumference of 62.83m, wherein the amount of nests were counted. This allows the researchers navigate the edge of the circular plot while counting the number of nests that fall inside the circle, and ovetop of the herbaceous plot. Each nest was then identified as either a double-crested cormorant nest, a great egret (*Ardrea alba* L.) nest, a black crowned night heron nest or a great blue heron nest. The totals were calculated and recorded for the given plot.

### **Ground Layer Vegetation**

At the centre point of each point quarter plot a 1m by 1m quadrat was placed in the northeast quadrant of the PCQ. In the 1 by 1m plot, each plant was identified and counted; regenerating woody species were counted but were not used in the analysis. The focus of the herbaceous data collection was wildflower species, and therefore grasses and sedges were excluded from the analysis (practical constraints forced me to limit my data collection to wildflower species). This data collection provides information on abundance of native species and non native species as well as diversity data. To see the herbaceous plots for Middle Island please see Annex I, Figure 7; to see the herbaceous plots for East Sister Island please Annex I, Figure 8. To see the herbaceous plots for West Sister Island, please see Annex I, Figure 9.

### **Seed Bank Samples**

If seeds accumulate in the soil and remain viable, the assembly of the seeds in the soil can create a bank which can be activated during a soil disturbance or other restorative techniques (Ellsworth et al. 2004). At each site, seed bank samples were taken based on a representative plot basis. This sampling structure was used in order to account for the patchy distribution and spatial variation of seed dispersal in order to avoid overestimating or underestimating seed bank density and composition (Wiles et al. 1992; Ambrosio et al. 2004). In order to combat these problems Ambrosio et al. (2004) recommend using a systematic sampling design, which matches the sampling design already used on Middle Island and East Sister Island. Bigwood and Inouye (1988) recommend taking many small samples as opposed to few large ones in order to increase the precision of the seed number estimations. Seed bank samples were taken on August 28, 2008 and September 2 on Middle Island, September 3, 2008 on East Sister Island and September 10, 2008 on West Sister Island. The seed bank was sampled at this time because it was after the majority of the species in the community had

dispersed their seeds and I wanted to sample both the transient and persistent seed banks (Baskin and Baskin, 1998). On Middle Island of the 52 vegetation plots 42 plots were chosen as a subsample for seed bank analysis, the remaining vegetation plots were not sampled due to limited soil depth; 38 samples were processed in the lab for the study (the samples were shared with Canadian Wildlife Service and were processed by a Parks Canada archaeologist which reduced my useable sample size to 38 samples). All 27 of the vegetation plots on East Sister Island were sampled for seed bank analysis. On West Sister Island, of the 60 plots chosen for vegetation sampling, 42 plots were chosen as a sub sample due to soil depth limitations and logistical restraints regarding sampling time. Of the 42 samples taken, 38 were processed for seed bank viability and composition; as four samples were contaminated in the lab and were removed from the analysis (the 38 useable samples from both Middle Island and West Sister Island was coincidental).

At each site, the 1 by 1m plot in the North east corner of the plot used in the point-centre quarter method (where the vegetation data was collected) was avoided during seed bank coring. A core sampler was used with a 4.5cm diameter and 13 cm depth.



Figure 4: Collection of soil seed bank samples around the plot centre © Darby McGrath.

Around the center samples were collected to be used for seed bank sampling, in an attempt to account for the patchy distribution of the buried seeds (Schenkeveld and Verkaar, 1984; Thompson, 1986) five samples were taken at each plot (Baskin and Baskin 1998). As the patchiness increases, the density decreases and the number of samples required to accurately sample the seed bank increases (Fenner and Thomspson, 2005). The cores were placed in one large Ziploc, thoroughly mixed and then transferred into a paper bag in the lab to avoid fungal growth on the samples; 50mg of soil was used for the actual analysis.





Figure 5: Example of the soil sample pattern used to collect the seed bank samples © Darby McGrath.

The seed bank samples were then cleaned using a 1mm by 1mm screen to remove the soil in order to reduce the volume of the soil in order to sieve out the seeds - physical extraction of seeds from the soil in general reveals more species than germination (Fenner and Thompson, 2005). The seeds were removed from the remaining debris and placed in envelopes. Once the seeds had been cleaned the samples were identified and counted using a microscope.

Each seed was then crushed using forceps (*viz* Borza et al. 2007) and were deemed viable or non viable. Seed viability is the “capacity of the seed to germinate and produce a normal seedling under favourable conditions” (Borza et al. 2007). The crush method has been found to be a useful and less laborious way to test seed viability (Buhler et al. 2001; Forcella et al. 1996; Nurse and DiTomasso 2005; Rothrock et al. 1993; Sawma and Mohler 2002; Borza et al. 2007). Since the study attempts to survey the seed bank for viability the unimbibed crush test is acceptable (Sawma and Mohler 2002; Borza et al. 2007). To perform the crush test, the researcher applies gentle but steady pressure to the seed using forceps. The seeds that collapse under this pressure are categorized as nonviable, and those that remain firm and intact are considered viable (Borza et al. 2007). This method is preferred to germination, flotation and tetrazolium tests because the crush does not require as much training for the analyst, it is less labour intensive than other tests, and may result in fewer errors (Borza et al. 2007). Germination, aside from being labour intensive, time intensive as well as requiring more resources than the crush test, may still not provide accurate results of viability due to specific dormancy breaking requirements for different species (Borza et al. 2007; Baker 1989). The International Seed Testing Association (1985) finds fault with the tetrazolium tests because performing the test requires substantial skill at dissecting the seed coat longitudinally to reveal the embryo with a high probability of ruining the seed. Sawma and Mohler (2002) also report that the tetrazolium tests can have potential sources of error including false positives due to

microbial precipitates and false negatives due to embryo separation from the rest of the seed in the species with smaller seed sizes.

### **Litter Depth**

Mountford et al. (2006) found that the depth of litter is a variable affecting sapling regeneration and that seedling density was higher in quadrats where litter depth was lower. Facelli and Carson (1991) argue that litter depth can affect the amount of moisture and temperature as well as the “light profile close to the soil” which can ultimately restrict the ability of seeds to germinate in areas of high litter depth.

At each site where a seed bank core was collected, immediately after the soil core was removed, a litter depth reading was taken using a ruler. The litter depth was taken at the northern edge of each soil core sample. The five readings were then averaged and the standard deviation was calculated to account for any samples collected that might vary dramatically from the mean due to the variable distribution of litter depths around the sampling points.



Figure 6: Measurement of the litter depth after soil core removal © Darby McGrath.

### **Damage Indices**

Parks Canada and Ontario Parks established a set of damage indices on East Sister Island and Middle Island in 2005. The areas were classified as zero damage, low damage, medium damage or high damage. The damage classifications based on a ranking system were established for a separate study by a student at the University of Windsor focused on soil chemistry analysis including East Sister Island and Middle Island. The classifications are related to specific soil plot locations used for soil chemistry tests and not the herbaceous vegetation survey plots. Therefore, I chose only the herbaceous plots which are strictly associated with the

damage rankings based on the GIS maps of the islands. Cormorant nest densities were ranked based on aerial photographs as well as expert opinion during the assessment (A. Fisk., personal communication April 3, 2009). The plots designated highly damaged on East Sister Island are 64, 66, 68, 72; Medium damage: 79, 80, 52, 82; Low: 52; zero: 53, 81, and 84. The plot designations are as follows for Middle Island: High damage: 9, 28, 40, 12; Medium damage: 17, 37, 24, 46; Low damage: 29, 30, 35, 5, 22; and zero damage: 15, 31, and 32. Although this method can only provide limited results because it uses a limited data set, it may help to justify studying more extensively the areas considered to be more highly damaged through cormorant nesting activities. The data collected for the herbaceous plots and the seed bank samples was run separately based on the damage indices to determine if there is in fact a significant difference in damage in localized areas across these two islands. West Sister Island was not used for this data analysis because damage indices had not been designated to the plots prior to the start of my study.

### **Data Analysis**

In most ecological studies, it is unrealistic to assume that the data sets will conform to a parametric test because of spatial autocorrelation or skewness (Anderson 2001). That was true in my study as data were non parametric (failed the Kolmogorov-Smirnov test for normality), hence Kruskal-Wallis tests were used for testing equality of population medians among groups (Vargha and Delaney, 1998). This test does not focus on measures of central tendencies like the parametric ANOVA (Ruxton and Beauchamp, 2008). The Kruskal-Wallis test is referred to as the ranked equivalent of the ANOVA (Analysis of Variance). Regardless of the shape of the distributions among groups, “if the sample observations across all groups are ranked, and the variances of these ranks are similar for all the groups”, then the Kruskal-Wallis test can be used (Ruxton and Beauchamp 2008).

For one test where I compared litter depth and herbaceous species diversity, the Pearson Product Moment test was used (Rodgers and Nicewander, 1988). It measures bivariate association or linear dependence between two variables e.g. litter depth against native species diversity. It is important to note that the test is limited in application because it does not determine a cause and effect relationship, but determines correlative relationships. The coefficients range from -1 to 1 and a reading of 1 demonstrates that the linear equation describes the given relationship exactly (positively). Conversely, -1 demonstrates that one variable (Y) increases as the other decreases (X) (Rodgers and Nicewander, 1988). A reading of 0 indicates that no relationship exists between the two variables (Rodgers and Nicewander, 1988).

## 4 Results

**Overview.** Generally, cormorant nest density did not contribute to increased litter depth on the islands. Neither cormorant nest density nor litter depth caused any significant responses in species using above-ground or seed bank measures (abundance, composition, diversity, seed viability). Cormorant nest numbers decreased the abundance of *Alliaria petiolata* on all three islands. Tables 3 through 12 contain the results from the data analysis from the entire island data sets as explained in the following section in depth. The selected plot analysis revealed that cormorant nest density and litter depth does influence the herbaceous layer abundance and diversity but does not influence seed bank viability or abundance. Table 13 is a summarization of the data from the selected damage indices plots.

### 1. *P. auritus* Nest Density and Herbaceous Species Abundance (Table 3 pg. 30)

Herbaceous data and associated cormorant nesting numbers for the years 2004, 2007, and 2008 demonstrated that *P. auritus* nesting numbers do not affect herbaceous species abundance on East Sister Island. The Kruskal Wallis test for all species on was  $P=0.541$ ; year and nest number are not significant in any case so all the data was combined. This demonstrates that higher density cormorant nest numbers does not impact abundance of herbaceous species. The P value indicates that there is not a statistically significant relationship between cormorant nest density and herbaceous abundance and diversity.

For Middle Island, cormorant nesting numbers did not affect herbaceous species abundance. (Kruskal-Wallis  $P=0.426$ ). Year and nest numbers are not significant in any of the data and therefore all the data were combined for the test.

On West Sister Island, *P. auritus* nesting numbers were found to negatively correlate with herbaceous species abundance. The Kruskal Wallis test found  $P < 0.05$  for all species combined for one year of data, 2008. This finding may be subject to the fact that only one year of data was collected and run in the Kruskal Wallis test and will be analyzed further in the discussion.

Additionally, when the data for the herbaceous species was run separately against the cormorant nest numbers, I found that cormorant nesting numbers do not affect any one species on the islands except for garlic mustard. In the case of garlic mustard on all of the islands, *P. auritus* was found to be significantly influential in decreasing numbers of this herbaceous species; on East Sister Island  $P < 0.01$ ; on Middle Island  $P < 0.01$  and on West Sister Island  $P < 0.05$ .

## **2. *P. auritus* Nest Density and Litter Depth (Table 4 pg. 30)**

Only one year of data was used, 2008, for litter depth for all three islands. The analysis demonstrated that litter depth is not significantly influenced by *P. auritus* nest numbers anywhere. The findings were as follows: East Sister Island  $P=0.334$ , Middle Island  $P = 0.298$  and West Sister Island  $P=0.671$ .

## **3. Native Seed Bank Abundance and Exotic Seed Bank Abundance (Table 5 pg. 30)**

This hypothesis compared native seed bank abundance against exotic seed bank species. Middle Island has the highest exotic seed bank,  $P < 0.001$ ; West Sister Island has the lowest exotic seed bank for all three islands,  $P < 0.01$ . The exotic seed bank is significantly higher than the native seed bank on all islands; on East Sister Island  $P < 0.01$ ; on West Sister Island  $P < 0.05$ ; on Middle Island  $P < 0.001$ . The native seed bank is the same for all three islands,  $P = 0.317$ . This suggests that where double-crested cormorants have been culled, there is not a significant difference in the species abundance of native species in the seed bank, namely on the control island, West Sister Island. There is however, a significant difference between West Sister Island and Middle Island and exotic species abundance.

## **4. Litter Depth and Seed Bank Abundance (Table 6 pg. 31)**

Litter depth does not affect native or exotic species abundance on any of the islands in any statistically significant ways. There was no significant difference between any of the islands according to the analysis, so the data from the islands were combined and the  $P$  value for all the islands is 0.338.

## **5. Seed Bank Abundance and Herbaceous Vegetation Abundance (Table 7 pg. 31)**

Using the Kruskal-Wallis test, I tested native seed bank versus native herbaceous vegetation; native seed bank versus non native herbaceous vegetation; non native seed bank versus native herbaceous vegetation and non native seed bank versus non native herbaceous vegetation. For East Sister Island there are significantly more native species in the seed bank than exotic species. The Kruskal-Wallis test for East Sister Island found  $P < 0.05$ . On East Sister Island there is significantly more native species in the herbaceous species count than exotics,  $P < 0.05$ . On Middle Island  $P < 0.05$ , demonstrating that there are significantly more exotics in the seed bank than native species. Additionally on Middle Island, the exotic species found in the herbaceous count is significantly higher than the native species found in the count;  $P < 0.01$ . On West Sister Island, there is no statistically significant difference between the seed bank and the herbaceous counts; Kruskal Wallis test found that seed bank  $P = 0.188$  and herbaceous count  $P = 0.247$ .

## **6. Litter Depth and Herbaceous Species Abundance (Table 8 pg. 31)**

The Kruskal Wallis tested native compared to non native herbaceous species abundance; native species abundance alone and non native species abundance to litter depth. In no case were there significant differences in mean litter depth and counts for exotics ( $P = 0.302$ ) or natives ( $P = 0.259$ ).

## **7. *P. auritus* Nest Density and Seed Bank Viability (Table 9 pg. 31)**

The islands were run separately to determine if there are any statistically significant effects on any one of the islands. However, there were no difference and pooling the islands ultimately provided a larger data set and therefore a more significant test. Cormorant nest density compared to the viable native seed bank found  $P = 0.348$ . Cormorant nest density compared to the viable exotic seed bank found  $P = 0.192$ .

## **8. Litter Depth and Seed Bank Viability (Table 10 pg. 32)**

The Kruskal-Wallis test pooled the islands and still found that the litter depth does not impact the viable native or exotic seed bank. Litter depth versus the native viable seed bank found the  $P = 0.261$ . Litter depth versus the exotic viable seed bank found the  $P = 0.204$ .

## **9. *P. auritus* Nest Density and Herbaceous Species Diversity (Table 11 pg. 32)**

The P values on all the islands and for all the tests, was not statistically significant. On Middle Island, cormorant nest density compared to total diversity of herbaceous species found  $P = 0.167$ ; cormorant nest density compared to native herbaceous species diversity  $P = 0.108$ ; and cormorant nest density compared exotic herbaceous species diversity  $P = 0.184$ .

On East Sister Island, cormorant nest density versus total diversity of herbaceous species found  $P = 0.191$ ; cormorant nest density versus native herbaceous species diversity  $P = 0.135$ ; and cormorant nest density versus exotic herbaceous species diversity  $P = 0.243$ .

On West Sister Island, cormorant nest density versus total diversity of herbaceous species found  $P = 0.166$ ; cormorant nest density versus native herbaceous species diversity  $P = 0.142$ ; and cormorant nest density versus exotic herbaceous species diversity  $P = 0.194$ .

## **10. Litter Depth and Herbaceous Diversity (Table 12 pg. 32)**

I used the Pearson Product Moment for the analysis which combined of the islands, because there was no statistically significant finding on the islands individually. The findings were as

follows: litter depth versus total herbaceous diversity P=0.415; litter depth versus native species diversity P=0.307; litter depth versus non native species diversity P=0.338.

Table 3: Results from Herbaceous Species Abundance and Nest Density Analysis

<b>1. ESI, MI, WSI<sup>7</sup></b>	<b>Years Run</b>	<b>P Value</b>	<b>Significance</b>
Higher densities of <i>P .auritus</i> nests significantly decrease the abundance of herbaceous plant species.	For ESI & MI 2004, 2007, 2008 & 2008 for WSI	P=0.541	ESI: No MI: No WSI: No

Table 4: Results from Nest Density and Litter Depth Analysis

<b>2. ESI, MI, WSI</b>	<b>Years Run</b>	<b>P Value</b>	<b>Significance</b>
Higher densities of <i>P .auritus</i> nests will significantly increase litter depth because of mature tree death and subsequent downed woody debris resultant from cormorant nesting activities.	2008	ESI: P=0.334, MI:P=0.298 WSI:P=0.671	No

Table 5: Significant Results from Nest Density and Seed Bank Abundance Analysis

<b>3. ESI, MI, WSI</b>	<b>Years Run</b>	<b>P Value</b>	<b>Significance</b>
Where <i>P .auritus</i> have been culled, the native seed bank abundance is higher than those islands where the birds have not been culled, and the exotic seed bank abundance is lower than the islands where no cull has taken place.	2008	Exotic species: ESI P<0.01, MI P<0.001, WSI P<0.05 and Native species P=0.317	Yes

<sup>7</sup> ESI stands for East Sister Island; MI stands for Middle Island; WSI stands for West Sister Island

Table 6: Results from Litter Depth and Seed Bank Abundance Analysis

<b>4. ESI, MI, WSI</b>	<b>Years Run</b>	<b>P Value</b>	<b>Significance</b>
Increased litter depth decreases the amount of seeds found in the seed bank (Native seed bank vs. Non native seed bank).	2008	P=0.338	No

Table 7: Significant Results from Seed bank and Herbaceous Layer Abundance Analysis

<b>5. ESI, MI, WSI</b>	<b>Years Run</b>	<b>P Value</b>	<b>Significance</b>
The species abundances found in the seed bank is significantly different than that of the herbaceous layer (Native seed bank vs. exotic seed bank compared to Native herbaceous vs. Non native herbaceous)	2008	ESI seed bank: native versus non natives P<0.05; ESI herbaceous: native versus non natives P<0.05 MI seed bank: non natives vs. natives P<0.05; MI herbaceous: non natives vs. natives P<0.01; WSI there is no difference in the seed banks (P=0.188) or the counts (P=0.247)	Yes ( for ESI and MI)

Table 8: Results from Litter Depth and Herbaceous Layer Abundance Analysis

<b>6. ESI, MI, WSI</b>	<b>Years Run</b>	<b>P Value</b>	<b>Significance</b>
At what litter depth is herbaceous layer abundance highest? (total abundance, native species abundance, non native species abundance).	2008	In no case were there significant differences in mean litter depth and counts for exotics (P = 0.302) or natives (P = 0.259)	No

Table 9: Results from Nest Density and Seed Bank Abundance Analysis

<b>7. ESI, MI, WSI</b>	<b>Years Run</b>	<b>P Value</b>	<b>Significance</b>
Cormorant nesting density affects the amount of seeds that are viable in the seed banks on the islands (native seed bank viability; non native seed bank viability).	2008	Native viable P=0.348, Non native viable P=0.192	No



Table 10: Results from Litter depth and Seed Bank Viability Analysis

8. ESI, MI, WSI	Years Run	P Value	Significance
Increased litter depth prolongs viability of native species in the seed bank.	2008	Litter depth vs. Native viable: (P = 0.261); Litter depth vs. Non native viable: (P = 0.204)	No

Table 11: Results from Nest Density and Herbaceous Species Diversity Analysis

9. ESI, MI, WSI	Years Run	P Value	Significance
Cormorant nesting densities decrease diversity on the islands (total species diversity; native species diversity; non native species diversity).	For ESI & MI 2004, 2007, 2008 & 2008 for WSI	ESI: Total diversity P=0.191; Native species P=0.135; Non native species P=0.243 MI: Total diversity P=0.167; Native species P=0.108; Non native species P=0.184 WSI: Total diversity P=0.166; Native species P=0.142; Non native species P=0.194	No

Table 12: Results from Litter Depth and Herbaceous Species Diversity Analysis

10. ESI, MI, WSI	Years Run	P Value	Significance
Increased litter depth decreases herbaceous species diversity (total species diversity, native species diversity, non native species diversity).	2008	Litter depth vs. Total diversity P=0.415; Litter depth vs. Native species diversity P=0.307; Litter depth vs. Non native species diversity P=0.338	No

Table 13: Significant Results from Selected Damage Indices Plots Analysis

Possible Effects	ESI		MI	
	P value	Progression of Significance	P Value	Progression of Significance
<b>a. Cormorant nest density vs. Litter depth</b>	P<0.001	High>Medium>Low=Zero	P<0.001	High>Medium>Low=Zero
<b>b. Cormorant nest density vs. Native herbaceous species abundance</b>	P<0.001	High>Medium=Low=Zero	P<0.001	High>Medium=Low=Zero

<b>c. Cormorant nest density vs. Exotic herbaceous species abundance</b>	P<0.001	High>Medium=Low=Zero	P<0.001	High>Medium=Low=Zero
<b>d. Cormorant nest density vs. Native seed bank viability.</b>	P=0.197	No significance.	P=0.248	No Significance.
<b>e. Cormorant nest density vs. Exotic seed bank viability.</b>	P=0.265	No significance.	P=0.314	No Significance.
<b>f. Cormorant nest density vs. Total diversity of herbaceous species.</b>	P<0.05	High>Medium=Low=Zero	P<0.05	High>Medium=Low=Zero
<b>g. Cormorant nest density vs. Diversity of native herbaceous species.</b>	P<0.05	High>Medium=Low=Zero	P<0.05	High>Medium=Low=Zero
<b>h. Cormorant nest density vs. Diversity of exotic herbaceous species.</b>	P<0.05	High>Medium=Low=Zero	P<0.05	High>Medium=Low=Zero
<b>i. Litter depth vs. Native herbaceous species abundance.</b>	P=0.228	No significance.	P=0.291	No Significance.
<b>j. Litter depth vs. Exotic herbaceous species abundance.</b>	P=0.157	No significance.	P=0.212	No Significance.
<b>k. Litter depth vs. Native seed bank viability.</b>	P=0.310	No significance.	P=0.238	No Significance.
<b>l. Litter depth vs. Exotic seed bank viability.</b>	P=0.113	No significance.	P=0.149	No Significance.
<b>m. Litter depth vs. Total diversity of herbaceous species.</b>	P=0.339	No significance.	P=0.250	No Significance.
<b>n. Litter depth vs. Diversity of native herbaceous species.</b>	P=0.374	No significance.	P=0.238	No Significance.
<b>o. Litter depth vs. Diversity of exotic herbaceous species.</b>	P=0.301	No significance.	P=0.294	No Significance.

## 5 Discussion

***P. auritus* nesting has not yet affected the herbaceous vegetation across the extent of the islands studied.** Herbaceous species abundance and diversity (total, native or exotic), seed bank composition, and seed viability were not significantly affected by either cormorant nest numbers or large-diameter litter-fall from cormorant nesting activities. After analyzing the damage indices data on the specific plots on Middle Island and East Sister Island only, I determined that higher nest densities decreases herbaceous abundance (native and exotic); decreases herbaceous species diversity (total, native and exotic) and increases litter depth. I found that higher nest densities do not necessarily affect the soil seed bank. I hypothesize that not enough time has lapsed on either Middle Island or East Sister Island since the increase in nesting numbers, to significantly affect the soil seed bank to any measurable degree.

I did not find that higher densities of nesting cormorants significantly affected abundance or diversity of above or below ground flora - seemingly contrary to much of the literature (Bedard et al. 1995; Chapdelaine and Bedard, 1995; Rippey et al. 2002; Hebert et al. 2005; Koh and Hudson, 2006). However, this is not surprising because unlike these studies, I tested the extent of effect of cormorants on herbaceous vegetation on an entire island. The other studies have focused on cormorant nesting impacts on tree species and were focused on canopy damage across the islands. I was asking a different scale of question.

Nonetheless, there is some consensus between my study and others where my (limited) tests of selected plots with associated damage indices indicate that cormorant nest densities significantly affect the herbaceous layer within high density nesting zones. Further, the highly damaged plots correspond with the increasing “bare ground” zones and high nesting areas demonstrated in the aerial photographs and infrared photographs used by Hebert et al. (2005) and Duffe (2006). In both of those studies, damage to the canopy from cormorant nesting activities occurs primarily at the perimeter of the islands but the sparseness of the canopy is straying inwards from the edge. As nesting trees die they become unstable nesting platforms for the birds and they are abandoned for another more stable and healthier nesting tree in proximity to their foraging location (Rippey et al. 2002; Hebert et al. 2005; Duffe, 2006).

The immediate impacts on understory herbaceous species are not yet measurable or even visible; this is probably due to the immediate impacts on the canopy layer. Hebert et al. (2005) found that cormorant nest density was negatively correlated with tree cover on both East Sister Island and Middle Island. Koh and Hudson (2006) also found that trees in which cormorants are nesting become increasingly damaged until they are killed, and when they are

too decayed and little nesting material is left for construction, they abandon the tree. The direct impacts, therefore, are on the canopy and supercanopy (Koh and Hudson, 2006) and the impacts on the herbaceous layer are more indirect and perhaps take longer to become apparent.

### **Litter Depth is Increased in High Density Nesting Areas**

Hobara et al. (2002) found that cormorant nesting activities may lead to increased litter fall as a result of nest construction debris. I did not find that litter depth had a consistent or even significant impact on herbaceous species diversity, abundance, seed bank composition or viability. There was no significant correlation between *P. auritus* nest density and increased litter depth in the herbaceous species plot. Based on the findings from the damage indices section, although limited in scope, I predict that more localized testing would provide a better indication of influences from cormorant nesting on increased coarse woody litter. Observable increases in litter depth around mature tree deaths corroborate the damage indices findings (D. McGrath pers. ob.), suggesting that cormorants may increase the litter depth on the islands, but not in a significantly damaging way for the herbaceous species across the entire island ecosystem.

There seems to be a consensus among those who have long studied these islands that a focus on the perimeter nesting areas has yielded sufficient data for determining impacts on the canopy layer (Hebert et al. 2005; Duffe 2006). I cannot disagree but note that this is not the case for understorey herbaceous vegetation. This information gap exists because there has been more attention given to impacts on trees where nests exist and on the guano that builds near the shoreline where there was inherently little understorey growth (because of exposure to winds and waves on porous beach soil). I tested as much of this as possible in the perimeter areas, with the constraint that I also had to study the entire islands. This meant that while I could draw some limited conclusions, more extensive comparisons of the perimeter and interior areas will be needed. Using the areas with associated damage rankings, (please refer to page 25 for ranking methods) I found that litter depth is increased in the areas of high nesting density, and also that herbaceous diversity (total, native and exotic) and abundance (total, native and exotic) is decreased in these areas (please refer to Table 13 for specific results). To address this issue of litter depth and dead snags, I recommend sampling in concentric circles moving outward from a dead snag to allow for more accurate information on the amount of litter depth variation there is in areas of high nesting usage on the islands. Consequently, in these same sample areas it would be informative to also take soil seed bank samples for analysis to determine what is happening in the seed bank, explicitly in these highly damaged areas.

### **Invasive Species Seed Bank was not Affected by Nesting Densities**

There were no significant differences between the native seed bank and the non native seed bank between West Sister Island and either East Sister Island or Middle Island (which had the most exotic species in the seed bank). West Sister Island had the lowest exotic herbaceous species abundance count; Middle Island had the highest. The reason may be the relative locations of the islands to the mainland. Invasion of exotic species may be predicted by the connectivity or relative isolation to continental areas. Although it seems counterintuitive studies show, that isolated islands tend to be more invisable than islands nearer to a source of colonists (Lonsdale, 1999; Gillespie et al. 2008). Furthermore, on these isolated islands the indigenous biota may prove more susceptible to invasive species that are newly introduced to the ecosystem (Lonsdale, 1999; Sax, 2001; Denslow, 2003; Gillespie et al. 2008). West Sister Island was found to have the lowest abundances of exotic species in the herbaceous layer and is also the closest island to the continent; it is approximately 14km from the United States. Middle Island is approximately 17km to the United States and 30km to Canada. East Sister Island is approximately 21km to Canada and approximately 25km to the United States. The distances to the closest continental areas may help to explain the invasion potential of the islands to some degree.

On Middle Island the exotic seed bank is increasing (2004-2008) as a result of the presence of exotic herbaceous species on the island; however, this may not be unique to the archipelagos. For example, Sax and Gaines (2008) argue that naturalization of exotic species on islands is not a new phenomenon, as it has persisted for over two centuries. They also found that the number of naturalized plant species has increased linearly over time. On many of the islands included in their study, they found that “the mean ratio of naturalized to native plant species across the islands has changed steadily” for nearly two hundred years and furthermore they predict that more species will continue to be naturalized on the islands in the future (Sax and Gaines, 2008). On islands invaded by exotic species, the exotics tend to be widely distributed across the island (Brown, 1984; Gaston, 1990, 1994; Lawton, 1993; Blackburn et al. 1997; Koze et al. 2003; Gillespie et al. 2008). Changing amounts of plant richness, specifically, exotic species, on these islands is not unique to Lake Erie, nor does it seem to be uniquely characteristic of the cormorant nesting colonies. Increasing amounts of exotic species becoming naturalized on islands all over the world, both continental and oceanic, are predicted with varying magnitudes of native plant extinction in future years (Sax and Gaines, 2008).

Although East Sister Island does not have as many exotic species and has the most natives in both the seed bank and the herbaceous count; in general East Sister Island has less overall vegetation cover than Middle Island. Duffe (2006) found that East Sister Island expressed increasing percentages of bare soil from 2001 to 2003. In 2001, only 2% of the ground was considered bare, and by 2003, 15% of the ground was considered bare, or 1.5ha in

total (Duffe, 2006). East Sister Island has less overall herbaceous cover than Middle Island, which may suggest that all herbaceous species, including native species, have low survivorship.

Viable native seeds do remain on all of the islands in the soil seed bank. Restoration potential for these systems however, may be limited by the density of these species and also by the requirements needed to break dormancy. Additionally, local scale dispersal of native species is an important limiting factor in community invasibility (Tanentzap et al. 2009). These “fine scale” variations in the native species seed pressures are significant for limiting or restricting the invasion of restoration areas by non native species. Although I did not find significant species turnover from native to non native species in either the seed bank or the herbaceous layer, non native species are present on the islands and need consideration for control during restoration initiatives (Murphy, 2005). Fisher et al (2009) consider a soil seed bank to be “healthy” when all the species of the given ecosystem are adequately represented. Arguably, representation of native species of the community is important for conservation of genetic material as well as survival of those individual species and the plant community as a whole (Fisher et al. 2009). However, limitations of native seed species can also constrain restoration efforts, and ultimately may require employing a combination of restoration techniques to foster the native seed bank.

#### **Garlic mustard has Decreased on all Islands**

The main significant result was that higher densities of cormorant nests decrease the abundance of garlic mustard. While it may seem counterintuitive that disturbance impedes garlic mustard, this is consistent with the literature. Garlic mustard colonizes unoccupied or sparsely occupied habitats – though it also will invade any disturbed habitat with higher initial densities of invasives - (Anderson et al. 1996). However, garlic mustard is also sensitive to litterfall (more often in the form of mulched leaves but nest litterfall may suffice) and it is possible that garlic mustard is more sensitive to even small litterfall changes, and changes in the pH which may not be affecting the other herbaceous species. Bartuszevige (2007) found that garlic mustard survival and seedling establishment was significantly diminished by increasing litter amounts. In this study where different litter treatments were used to test garlic mustard establishment and survival, the findings demonstrated that garlic mustard responds well to increased light and nutrient influxes, but adding litter to the plots decreased establishment and survival.

Changes in the soil chemistry may also be a factor in the relationship of garlic mustard and cormorant nest densities. Ornithogenic soils are found to be extremely high in phosphate, nitrate and ammonium as well as exhibiting generally low pH values over long periods of time (Wait and Aubrey, 2005). While this evidence may seem contradictory is important to keep in mind that studies on seabird effects on islands in arid climates in salt water ecosystems may

yield different results from the islands on Lake Erie; additionally compressing the time scale from such a study would also yield contradictory results in soil chemistry composition. The findings from Verkoeyen et al. (2009) will provide more appropriate comparisons for the islands in this study. Duffe (2006) found a significant negative relationship between cormorant nests in 2003 and pH; conversely he also found a significant positive relationship between cormorant nests and soil ammonia and nests and nitrate concentrations. Wait and Aubrey (2005) also found that soils inundated by guano deposition can inhibit some herbaceous species while facilitating the growth of other species. Long term guano deposition is associated with a decrease in soil pH (Wait and Aubrey, 2005) which ultimately could inhibit the survivorship of garlic mustard. Vidal et al. (2000) found that species turnover is positively correlated to gull nesting density and also that seabird activities tend to select and facilitate some adapted plant species at the expense of others, including native indigenous flora as well as other exotic flora species. Hobara et al. (2001) found that excreted Nitrogen (N) was quickly mineralized and taken up and cycled throughout the ecosystem. Hobara et al. (2001) also found that plant carbon: nitrogen ratios and leaf litter carbon: nitrogen ratios decrease with an advanced stage of bird colonization. What my findings indicate, however, is that there is not any major change in the nutrient cycling because the plant compositions remain relatively the same island-wide. Garlic mustard may be more sensitive to these changes than other herbaceous species. Although no chemical testing of the soil was taken into account for this study. On East Sister Island and Middle Island, soil samples are taken frequently for chemical testing, and could be examined in relation to garlic mustard abundance counts.

### ***P. auritus* and Invasive Species Dispersal**

MacArthur and Wilson (1967) introduced the concept of Island Biogeography Theory which attempted to explain the species richness of actual islands; the theory now extends beyond islands to fragmented ecosystems in landscape ecology. Post MacArthur and Wilson (1967), other factors influence species richness. One of the major factors in Island Biogeography Theory is related to dispersal mechanisms, and especially vectors for dispersal. Humans, wildlife, and wind are well studied dispersal vectors (refer to Gillespie et al. 2008). On Lake Erie, the data collected on the islands indicates that dispersal of invasive species has allowed these species to colonize the islands. I have found no studies that indicate whether cormorants are vectors for invasive species to the islands. Nesting materials used by cormorants in nest construction can be collected from “diverse” locations, it is reported that they may collect material from locations several kilometres away (Wires et al. 2001). We do not know, however, whether the cormorants are bringing in invasive species from the mainland. It can be hypothesized that cormorants bring in nesting material, including some invasive species (during a visit in April 2008 I observed that garlic mustard was a commonly used in cormorant nests on both Middle Island and East Sister Island). They may also perhaps, unwittingly transport the seeds in their plumage from their migratory route.

Seed abundances of native species may be reduced by declines or absence of plants in proximity to the islands (few species can send seeds via the water or via longer-distance dispersal on wind or animals). Still, both native and invasive species could potentially be transported to the islands via the cormorants, and they may be an important vector for seed source input within the island archipelagos (see also Laurence, 2008). However, the likely vectors for seed transport to the islands are humans (see Gillespie et al. 2008). Many seeds can “hitch-hike” in on unwitting visitors to the islands as this is a common mechanism for dispersal for many species (Bartuszevige, 2007). In general, islands, as isolated habitats, will tend to be colonized by exotic invasive species due to their larger dispersal ranges and higher propagule pressures than the indigenous species of the area (Gillespie et al. 2008). Although visitation is restricted to the islands during nesting and breeding seasons, trespassing still occurs on the islands. Additionally, the data collection teams that visit the island may also bring in seeds on their clothing and most likely, on their footwear. The cormorant nest census that takes place brings out dozens of individuals to sweep the islands in order to limit the time spent on the island; however to my knowledge, there is no protocol in place to eradicate foreign material from entering the island on the research teams.

### **Landscape Ecology and the Islands**

Landscape Ecology looks at ecological processes and functions through spatial variation. Islands, historically, have been the subject of studies in this field because of their unique locations and restricted species interaction. Island Biogeography Theory (IBT) (MacArthur and Wilson, 1967) originally discussed factors influencing species richness, colonization and extinction on islands but the theory has more recently been applied to habitat fragments, which are discussed as “figurative” islands in a landscape. Current research on islands uses concepts of IBT to study factors facilitating or restricting dispersal of invasive species (Matlack, 1993; Collinge, 1996; Bartuszevige, 2007, Gillespie et al. 2008). Islands are subject to different species pressures and propagule pressures than continental habitats (Gillespie et al. 2008) which can often make these areas more susceptible to invasion of exotic species.

My data suggest that the canopy openings may originally encourage exotic species (Peters et al. 2006) but as nesting pressures continue to stress a localized area there will be a decrease in herbaceous species abundance and diversity. The continued decreases in canopy cover on Middle Island and East Sister Island (Hebert et al. 2005; Duffe, 2006) however, may facilitate invasions because non native species can place pressure on the ecosystem through seed dispersal during intense forest fragmentation (Tanentzap et al. 2009). Fragmentation is also commonly associated with changing patch sizes and shapes of habitats. A decrease in species richness is also commonly associated with the decrease in patch size due to disturbances creating new edges and openings in ecosystems (Lovejoy et al. 1986; Bierregard et al. 1992; Fagan et al. 1999).



Invasion and dispersal of species can be influenced by the patch size or shape of an ecosystem (Forman, 1995). One predicted outcome of cormorant nesting on the islands is a modification of patch size and shape of the habitable parts of the islands. Modification of the shape of the patches can influence the colonization of exotic species. Certain shapes, such as rounded patches, can be more effective for conserving internal resources in an ecosystem through the minimization of the exposed perimeters to outside effects (Forman, 1995). Patch creation or modification is linked to disturbance agents (Peters et al. 2006) such as the cormorants. Cormorant nesting behaviour indicates that nest creation and maintenance will be close to the shoreline and foraging locations (Quinn et al. 1995; Wires et al. 2001; Hebert et al. 2005; Duffe, 2006; Koh and Hudson, 2006). Cormorants stress the edge of the island ecosystems with nesting activities and may be modifying the perimeter of the island ecosystems through these pressures. Landscape dynamics are governed by the types of transitions that are contained within the landscape (Peters et al. 2006). This indicates that the hypothesized cormorant nesting impacts may drive the landscape changes in the systems.

### **Nesting may Increase Edge Effects on the Islands**

The research on cormorant impacts on trees qualifies the hypothesis that nesting activities cause a decrease in forest ecosystem cover with a decrease in the “health” of individual trees (Hebert et al. 2005; Koh and Hudson, 2006; Duffe, 2006). Duffe (2006) found that Middle Island lost 16.8 ha of dense canopy in 8 years, with an increase of sparse canopy from 6% in 1995 to 32% in 2003. Research completed on cormorant nesting site location and behaviour suggests that cormorants nest in specific locations on uninhabited islands such as the islands in this study. The nest site preference indicates fidelity to nesting areas and proximity to foraging locations due to their aversion to human presence.

Additionally, it is hypothesized that cormorants nest in the higher canopy for proximity to escape routes if they feel threatened, which is apparent as cormorants were flushed out of their nests during data collection on all of the islands during my research. Quinn et al. (1995) studied and characterized cormorant nesting behaviour in order to facilitate nesting on a constructed island in Hamilton harbour. They found that cormorants build their nests in open and highly visible trees most commonly those trees found along the shoreline. Additionally they also nest along the shore close to their foraging areas (Wires et al. 2001). Additionally, cormorants are also believed to have intense nest site fidelity to colony sites (Wires et al. 2001). Young cormorants are often found to gravitate to their natal colonies, and initiate their first breeding season at that site (Wires et al. 2001). Hebert et al. (2005) hypothesized that colonization history of East Sister Island proceeded from east to west, which was corroborated by Duffe (2006), where some of the formerly non-colonized areas of the western end, now host cormorant nests.

Duffe's (2006) infrared photos support my conclusion that the herbaceous layer and seed bank is not yet affected by cormorant nesting densities across the entirety of the islands. The photos indicate that the canopy remains fairly densely intact on the interior of the islands, but that the edge is collapsing inward. On both East Sister and Middle Island, the red area in the infrared aerial photos, classified as bare ground is less prolific in 2001 and is spread evenly around the perimeter. By 2003, however, the yellow area (sparse canopy) is increasing substantially along the perimeter and further into the interior. On Middle Island, the western edge demonstrates more sparse canopy and bare soil with a solid perimeter of bare soil around the entire island in 2003 as compared to the photo from 1995. Although Duffe (2006) demonstrated that the canopy is becoming sparser on the islands, further research on these island ecosystems should look to focus on the highly damaged primary nesting areas of the cormorants and determine the rate of collapse of the edges. To see the infrared photos, please refer to Duffe (2006) or Hebert et al. (2005). As of 2008, cormorant nesting has not spread dramatically into the interior this may be the reason that my study did not demonstrate nesting impacts across the islands extensively. This may also explain why my study did not find a dramatic species turnover or an overall species abundance decline on the islands in their entirety as a result of cormorant nest numbers or increased litter depth.

Although the selected plot analysis provides only a limited data set (for the damage indices), it did provide a comparison between areas considered to be impacted either with low, medium or high nesting activities. This does, however, indicate that a larger comparison between areas with zero or low damage (e.g. the interior) compared to the highly degraded areas where the cormorants primarily nest would help to determine the rate of collapse of the perimeter of the islands. There is not (in the years of data collection 2004-2008) a massive wave of invasion of exotic species as predicted, and some exotic species (garlic mustard) are even declining in abundance across all islands. It is important to note, that I am not explicitly testing the specific damage areas, I am looking at the overall-island herbaceous and seed bank ecosystem, which I have found to remain relatively unchanged in the years of data collection. Furthermore, it is important to discuss the impacts that edge effects can have on an ecosystem, and how this, potentially, could become an issue on these islands. According to Peters et al. (2006) the type of transition that a boundary is influenced by can help to predict the environmental drivers in the system. A boundary that is experiencing, what the authors call a "shifting transition zone" will respond to certain environmental conditions (Peters et al. 2006). If the vegetated boundary of the island is shifting due to cormorant nesting pressures, it can be predicted, that species interactions within the habitat will be influenced (Peters et al. 2006). If pressure is placed on the boundary of the system it can influence the shape of the patch which can change the permeability of the boundary to propagule pressures (Fagan et al. 1999). Disturbances associated with edge creation can also influence ecological mechanisms, functions and processes at a variety of scales (Fagan et al. 1999). Modification of edge areas have been

found to influence biogeochemical nutrient transport (Kitchell et al. 1979) and can also influence the outcomes of species interactions (Kareiva 1987; Roland 1993).

Island size and shape may determine the amount of damage cormorants may exude on a system. Cormorants, in island ecosystems, are primarily edge nesters (Quinn et al. 1995; Wires et al. 2001). Literature on forest fragmentation talks about fragmentation shape and the edge to interior ratio as having an impact on disturbance predictability and invasibility. The impacts of disturbances along the edge of an ecosystem can often lead to “modifications” in the interior (Malcolm, 1994). Edge effects can very easily impact species by increasing the abundance of some species while decreasing the abundance of other species (Malcolm, 1994). As in the island ecosystems, the edge can be pushed back further into the interior, which causes new areas to become exposed to the adjacent area that is disturbed and cleared with the edge impacts that may permeate the habitat (Collinge 1996; Chen et al. 1992; Lovejoy et al. 1986; Harris 1984; Ranney et al. 1981). This can lead to a whole host of changes to the interior area such as changes in temperature, light, moisture and wind (Collinge 1996). Matlack (1993) found edge effects altered humidity and litter moisture up to 50 metres into the interior of the forest. These types of changes can affect the herbaceous communities in proximity to the disturbed areas (Collinge 1996); suggesting that further research on the islands should be focused on these areas.

In forests, edge effects are a major concern because dramatic structural changes can be wrought by altering the vertical constitution of the ecosystem (Murcia 1995). Additionally, smaller fragments, or in this case, islands, have relatively more perimeter and therefore may receive relatively more edge effects compared to larger islands (Malcolm 1994). Smaller islands with a smaller interior and therefore a larger edge to interior ratio, may suffer more impacts to the interior due to sheer nesting density on the edges. My findings, therefore, should be restricted to islands of similar size (e.g. 15 ha to 36 ha in size, in respect to the size of the islands included in my study.) Additionally, island shape, that is, islands with large edge to interior ratios may also be impacted differently than the islands in this study. In a longer term study, it will be important to look at the damaged areas, those areas currently experiencing high nesting densities and those areas abandoned due to mortality of mature trees chosen for nesting. These areas should be compared to the overall health of the herbaceous ecosystems of the island. Using the findings from this study in comparison to a study focused on the areas frequented or chronically used for nesting provide a more complete picture on the impacts of cormorant nesting activities on this island.

### **Alternative Stable States and the Islands**

Suding et al. (2004) offer a possible future scenario that can be tested as a result of my data collection, and current findings on the islands, as I now have a baseline data set for the

entirety of the islands. Suding et al. (2004) discuss the unpredictability of ecosystem responses to restoration efforts where the system has reached an alternative stable state in their paper focused on herbaceous ecosystems. They suggest that failed restoration efforts are often a result from the narrow focus we place on abiotic factors while ignoring or misinterpreting the biotic factors that may have changed or the feed backs between the biotic and abiotic factors that may have developed in the perturbed state (Suding et al. 2004).

Degraded systems can be forced into what Suding et al (2004) call *persistent alternative stable states*. Systems can have different phases after a disturbance degrades an ecosystem. As a result the dynamics of the degraded state are very different from what they call the “pristine” or the “target state”. Systems can shift between two or more stable states. System state variables, according to the authors, are biotic factors like abundance, composition and diversity and are influenced by environmental conditions such as fire or nutrient loading which can be shifted to cause a change in the ecosystem. In the case of the islands, my study clearly indicates that there is no statistically significant change in the herbaceous data between 2004-2008 (the unmeasured years, 2006-2007, might have shown change but one would expect this to still be detectable in 2008). What this may represent is an interim period of an alternative stable state. No change is evident within these years, perhaps, because prior to the data collection commencing in 2004 a new stable equilibrium may have been reached. It is possible that I was measuring the post-shift stable state in 2008. More likely, however, if the cormorant effects create collapsing edges into the interior, then, and only then, will there be island-wide impacts unless the nutrient cycling becomes altered in the meantime or if a newly introduced pest propagates faster than the edge collapse. There most likely have already been local state shifts along the perimeter, but not the whole island.

The overall effect on the islands suggests a variation of impacts on the abundance herbaceous layer and subsequent seed bank composition. The ecological integrity of the herbaceous layer within the years of data collection indicates that the ecological integrity of the island is not that different with the double-crested cormorant nesting colony or without. In the data set of the four year window, there is not a large shift from native to invasive species and therefore we are not really seeing a shift in herbaceous species composition. This indicates that on cormorant nesting islands we should expect more damage along the shoreline areas, and more direct damage on the trees (Hebert et al. 2005; Koh and Hudson, 2006; Duffe, 2006).

The next step is to ask what is happening in the areas were cormorant nesting activities is in fact impacting the herbaceous layer and canopy. Koh and Hudson (2006) found that cormorants decrease crown density and crown ratio as well as foliage transparency and found from 2004 to 2005 there was an overall increase in damage to trees and poles. They also report, however, consistent with my findings, that damage is not uniform across the islands.

Ultimately, cormorants are damaging the canopy trees in which they nest, but the fallout from nesting activities along the shore is not impacting the herbaceous layer or the seed bank island-wide. Most importantly, for future studies on these islands, we should try to determine if this area is expanding and at what rate. It is possible that the interior is acting as an independent entity and also that the edge of the islands where the cormorants primarily nest, may be declining at a non linear rate. As a result of my study, we now have a synthesis of the data collected on each island (by myself and the respective agencies) and some baseline findings about the herbaceous layer and the seed bank.

#### *Assembly Theory and the Future of the Islands*

Assembly theory can be thought of as a sub-framework of restoration ecology. In response to former ecological practices, focused on climax communities and linear successional pathways, assembly theory attempts to integrate the repertoire of possibilities in an ecosystem through function, structure and processes. It can be understood as a reconciling of a variety of concepts into a new explanatory vision, which focuses on how communities are put together and how they function (Temperton and Hobbs, 2004). The theory also includes the interaction of the “environment with the organisms of a community, and the interactions among organisms,” and how these interactions restrict community structure and development” (Temperton and Hobbs, 2004). This theory essentially studies how communities assemble and reassemble during or after the conclusion of a disturbance. Reassembly is important in the reintroduction of native species into damaged or recovering ecosystems and is therefore very pertinent to the preservation of protected areas. Reassembly can happen either naturally, through natural dispersal mechanisms characteristic to ecosystems (i.e. wind dispersal, water dispersal or wildlife dispersal) or can be manipulated through efforts of reintroduction of native biota. This study evaluated the assembly of herbaceous species during the ongoing cormorant nesting disturbances and sought to determine the community membership of the ecosystem.

Re-assembly of ecosystems is an ongoing dynamic process. Given that a disturbance is influencing the trajectory of these ecosystems (even if it is restricted to the perimeter of the islands) re-assembly may be influenced by the nesting activities. Re-assembly can actively be pursued on the island confluent with wildlife management strategies, which may prove to be the most effective management strategy. Murphy (2005) found when studying the selection pressures from changes in the edge environment that planting intermediate densities of *Sanguinaria Canadensis* L. can manage invasions of *A. petiolata*. *S. Canadensis* in certain densities were successful in impeding and sometimes halting the dominance of garlic mustard and providing an opening for restoration efforts. This study indicates that re-assembly of native herbaceous communities often needs to be pursued in concordance with supplementary restoration efforts; including management of invasive species.

Since assembly theory attempts to understand the dynamics of a system, and the rules which enable certain assemblages, the theory is contrasted against former equilibrium paradigms; conceptualized organization and stability in ecosystems (Hobbs and Norton, 2004). Hobbs and Norton (2004) call this idea, a *nonequilibrium paradigm*, suggesting that systems are frequently stressed to varying degrees by disturbances. The magnitude, frequency, quality etc., of the disturbance will force a system to respond by maintaining a characteristic makeup or by shifting to a new basic identity with new processes, structure and function. The basic identity across the extent of the islands has not been altered according to this study. Shifts in certain regions, primarily shifts in the microenvironment through increasing the relative edge to interior ratio of the forest (Murphy, 2005), may be occurring due to nesting pressures (as indicated by the limited data set run against the damage indices). This theory recognizes that the natural world is an uncertain place in which disturbances are constantly causing alterations in the composition of assemblages and in the spatial patterns of the environment (Hobbs and Norton 2004).

The concept of a threshold in this sense is the biotic and abiotic factors (referred to as filters) that have been unduly altered by disturbance, causing a shift in the system, or an alternative state. It can be argued that thresholds should be thought of as permeable barriers between alternative states but thresholds can also be understood as a set of dependent variables that have been reassembled to alter community composition and change the basic identity, as suggested by Fattorini and Halle (2004). If a threshold is reached by a disturbance, then the filters that were formerly affecting species composition may also be altered, subsequently altering the community makeup. In the case of Lake Erie archipelagos and the double-crested cormorants, it should be these filters that are targeted for restoration to interrupt the feedback loop the cormorants may have created. This is the dynamic view of restoration ecology Suding et al. (2004) discuss; understanding and incorporating the dynamics of each system in order to limit the uncertainty during human intervention will allow for restoration ecologists to aid ecosystems in regaining a natural resilience cycle.

## **6 Management Options**

Protected areas require varying degrees of ecosystem management. As the human population grows, according to Sarr and Puettman (2008), it will be imperative to “conserve, restore, create, and sustainably manage ecosystems”. The environment found on the islands may require management techniques including wildlife management and also restoration of the highly damaged areas. Restoration can be pursued actively where the system is actually manipulated in order to repair or recover the system. Sarr and Puettman (2008) describe active restoration forestry as “re-creation of a set of specific ecological conditions that were believed to be present in the past or that exist in remaining reference areas”(Pilarski, 1994; Keddy and

Drummond, 1996). The term “management” has become very unpopular due to the perceptions of anthropocentric projections onto ecosystems.

In many cases managing ecosystems is necessary based on our previous mismanagement of the same ecosystems. Keddy and Drummond (1996) argue that restoration of parks is important in order to “represent remnants of the original forest and to provide baselines for scientific research”. Our mismanagement of forest ecosystems can have perilous consequences. An example of this is our restriction of forest fires in many western forests. The overall restriction of forest fires has resulted in a change in the natural disturbance regime, which has altered the structures and functions of many forests where fire is an inherent and essential part of the ecosystem (Kauffman 2004). Kauffman argues that management must be understood as much more than mere mechanical means. In respect to forest fires, thinning is no substitute for the chemical, biological and physical impacts a fire can have on the ecosystem. Restoration efforts on the island may require a manipulation of the litter layer including removal of both coarse litter and even soil removal in areas of high guano deposition. It may also require translocation of native species and focusing on increasing canopy cover in the areas where nesting trees have been abandoned. To determine the best possible restoration and management pathway, further research needs to be pursued towards determining restoration and management goals on the islands. This study has proven that, within this limited data set that the soil seed bank has not yet been affected by cormorant nesting densities, and therefore, restoration pursuing the revival of the soil seed bank may prove fruitful.

**Is four years of data enough?** One question that arises is whether the data window (2004-2008) has been long enough to detect any changes in the ground layer community. It is important to take into consideration the time frame (Magurran, 2008) as well as the spatial features of a study. Magurran (2008) argues that the time interval over which data is collected can influence the shape of the species abundance distribution. The apparent lack of change in the herbaceous community from 2004-2008 may be artefact of having only four years of data for a shift that may span several decades

### **Recommendations for Further Research**

Following from the issue of how many years of data are needed, it would be beneficial to create a follow up project where the baseline data from the entire island is compared to the significantly damaged areas. Setting up transects to sample the perimeter overtime will indicate whether the edge is in fact collapsing in on the interior on the island. Using concepts of landscape ecology would help to determine the spatial variation of damage resulting from cormorant pressures over time. It would be interesting to determine the rate of collapse of the perimeters on the islands. This could assist in further planning for restoration of the islands.

Specific to the current monitoring protocols used on the island, continuation of monitoring may provide useful data on trajectory of the islands. It remains to be seen if sampling the localized damaged areas would provide more useful data on the spatial variation of the degradation, but I hypothesize that this may be the case.

In regard to cormorant nesting densities, although the nesting densities are not immediately impacting the herbaceous ecosystems across the extent of the island, it is apparent that they are still influencing the highly used nesting zones. Edge to interior ratio theory indicates that the modification of edges can threaten the health of the interior of habitats (Malcolm, 1994). Additionally, although I used a limited data set, I did find that increased cormorant nesting densities does influence the herbaceous ecosystem in these localized high density nesting areas. From my research it is apparent that it is important to further research the influence that cormorant nest densities can have on patch dynamics on these islands. Management activities may help to minimize the potential impacts of cormorant nest densities to the interior of the island.

Studies focusing on restoration initiatives may prove beneficial to the islands. If management of the cormorant population is pursued, restoration to the areas with significant damage may prove worthwhile on the islands. Since the islands are not uniformly damaged, if management of the population can halt the degradation of the nesting influences, restoration should be pursued in the areas that have been abandoned by cormorants because these areas are unlikely to be impacted further by nesting activities. On all the islands, the herbaceous layer is congruous with the soil seed bank. I hypothesize that this is an indicator that there has not been a dramatic species turnover in the herbaceous layer because the seed bank functions as an “ecological memory” (Fisher et al. 2009) of the island, and these layers are not drastically different. Also, the soil seed bank does contain viable native species on all the islands. Fisher et al (2009) found that alterations in the density and composition of the native seed bank can alter the growth and regeneration of these native species, ultimately altering ecosystem function. Native seed bank limitations, therefore, will constrain local native species recruitment. Additionally, non native species are present in the seed banks of all the islands in the study. It is unlikely that the ecosystems can be restored solely from the native seed banks due to the persistence of the introduced soil seed bank (Fisher et al. 2009). However native soil seed banks are essential in the control of introduced seed sources while fostering the growth of the currently restricted native seeds abound in the bank (Tanentzap et al. 2009). Manipulation of the native soil seed bank is especially important in developing conservation strategies for these protected areas (Tanentzap et al. 2009). This indicates that given the proper research of techniques and efforts towards re-assembly, there is potential for restoration that exposes the soil seed bank. Researching litter removal and the introduction of a native seed source may provide interesting results on the islands.



### **Scientific Recommendations**

- Comparative study to measure the edge effects on the island systems over time (i.e. determine the influence of the high nesting perimeter areas on the islands and rates of degradation)
- Use different experimental treatments on the areas of high nesting density (i.e. litter removal and possible chemical treatments; see Verkoeyen et al. 2009 and other studies on guano in waterbird colonies)
- Continued monitoring of the herbaceous layer including sedges, grasses and a focus on species with variable reproduction means (i.e. a focus on species reproducing clonally)
- Creation of a study that uses treatments between areas of high use for nesting, moderate use, zero use or those areas that are abandoned
- Creation of a study that looks at islands of different sizes and shapes to determine nesting habits and configurations as well as the variable rates of decline of these islands with these different parameters

### **Policy Recommendations**

- Initiate work on how increased development along the Lake Erie shoreline has influenced cormorant nesting locations and possible restrictions on development
- Begin to focus on this issue with a broader ecosystem approach that requires data sharing and mandates that look beyond federal, provincial, state and international borders (i.e. a management plan for instance, similar to the Interagency Bison Management Plan in Yellowstone National Park, that includes managers within the interior range breeding zone including breeding and overwintering areas)
- Most importantly, communicate with stakeholders about what the issue is really about in terms of federal, state and provincial agencies. It is not about commercial fishing, rare species, magnificent trees, or the cormorants themselves though there will be stakeholders interested in focusing on these and that is their right. What needs to be emphasized is that islands, in particular, are vulnerable to hyperabundant species that have impacts that may start along shorelines but then may propagate inward. For agencies that must ensure ecosystem integrity, the fate of one species is not as relevant. This does not mean that management should proceed with cupidity – and there is no evidence of this despite occasional salvos in the media. It does mean that it is a

“Hobson’s choice” – do something or do nothing<sup>8</sup>. Doing nothing in similar situations has not worked. Doing something, however unpopular with some groups, is the option chosen. It will indeed affect a formerly-declining species (the cormorant). But species are not the proper scale for management of protected areas. And longer-term, it may be that human habitation shifts away from shorelines and perhaps some of the other predators or competitors of the cormorants will either increase in response or be deliberately restored. That would in fact be the preferred option – but it will take decades at least and this is decades too long to risk in terms of the islands’ ecological integrity. It comes down to the age-old problem of there not being sufficient time to allow this rather anthropocentric experiment to run with complete controls. As with most ecosystem scale conservation, the risk of doing nothing is too great.

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<sup>8</sup> This is not to be confused, as often happens in vernacular use, with what is properly called a ‘Morton’s Fork’ where all outcomes are not desirable or a false dilemma – there are, in the short term, only two choices: intervene or not.

## Annexed Section: Maps and Images of the Islands

### Maps and Images of the Herbaceous Monitoring Plots



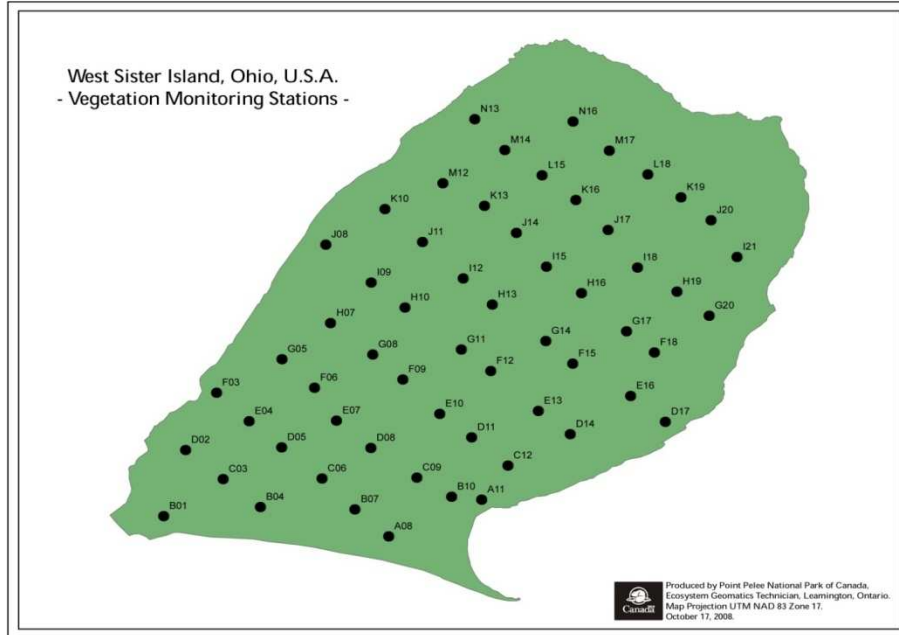
The map and image are both property of Parks Canada, 2006.

Figure 7: GIS Map of Middle Island displaying the numbered herbaceous vegetation plots.



The image is the property of the Ministry of Natural Resources, Ontario Parks, 2006.

Figure 8: Aerial photo of East Sister Island (with GIS overlay) displaying the numbered herbaceous vegetation plots.



The image is the property of Parks Canada, 2008.

Figure 9: GIS Map of West Sister Island displaying the numbered herbaceous vegetation plots.

## Maps and Images of Damage Indices/ Soil Plots



The map and image are both property of Parks Canada, 2006.

Figure 10: GIS Map of Middle Island displaying both unnumbered vegetation plots and the soil damage indices locations.



Image is the property of Ministry of Natural Resources, Ontario Parks, 2006.

Figure 11: Aerial Image of the soil locations on East Sister Island used for classification of the damage indices (please refer to Figure 10, the Middle Island map for corroborating legend for damage indices.)

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