**Impacts of double-crested cormorant nesting on western Lake Erie
islands**

By

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Abstract

Double-crested cormorant populations breeding on the Great Lakes have increased over the last 25 years. On small uninhabited islands in western Lake Erie, cormorants nest in trees in mature Carolinian forest. Here I quantify forest declines spatially and temporally using high resolution aerial photos and digital image analysis in relation to nesting cormorant densities. Using stable isotopes of carbon and nitrogen in tree rings, results demonstrate that an increase in physiological stress in nesting trees coincides with an alteration of nitrogen sources consistent with deposition of seabird guano. Forest decline and tree stress are not observed in non-cormorant breeding areas. Historic seabird inputs to island ecosystems are documented, however the magnitude of impact associated with the recent cormorant population increase is a new environmental stressor in the Great Lakes ecosystem.

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Chapter 1

General Introduction

1.1 Double-crested cormorant population growth on the Great Lakes

From the beginning, both popular and scientific publications regarding cormorants have focused mainly on the practical and domestic aspects of management and pest control (see Mendall 1935). A tremendous increase in cormorant numbers over the last 25 years has occurred throughout their North American range; however, the dramatic growth in the Great Lakes population has led to new calls for management and control at both extremes of their continental range (Nettleship and Duffy 1995). From a Canadian perspective, their breeding range in the Laurentian Great Lakes coincides with Carolinian forest habitat. Almost the entire Lake Erie population breeds on islands containing some of the most undisturbed, intact examples of rare vegetation associations. This thesis will examine this interaction between nesting seabirds and Carolinian forest on islands in southern Ontario, Canada.

The double-crested cormorant (*Phalacrocorax auritus*) is a foot-propelled, piscivorous diver and is the most numerous and widely distributed of the six North American cormorants (Hatch and Weseloh 1999). They

breed early in life (beginning of 4th year), have large broods (4 eggs), are efficient predators and have flexible requirements for feeding and nesting habitats (Duffy 1995). The five main breeding populations are: 1) Alaska 2) Pacific coast from southern British Columbia to northern Mexico 3) Canadian and U.S. interior 4) Atlantic coast from Newfoundland to New York and 5) Florida and the western Caribbean; however, Hatch and Weseloh (1995) note that recent expansion of the population has blurred the boundaries between 3,4 and 5. Individuals breeding in the interior of the continent and Atlantic coast are migratory, wintering on the coast from North Carolina to the western Gulf of Mexico (Davenport 1998). The interior population which is the focus of this study comprises 60% of the North American population (Hatch 1995).

The populations of cormorants in North America have fluctuated dramatically in recent times, showing a gradual decline over two decades up until the early 1970s and then a dramatic increase starting in the early 1980s (Weseloh et al. 1995). Environmental contamination from organochlorines (such as DDE, PCBs) as well as human persecution contributed to the decline; however, the influence of contaminants was likely worst during the 1960's and 70's when cormorant populations declined dramatically and biological effects were common. The effects of contaminants on cormorants observed on Great Lakes colonies included egg-shell thinning (Anderson and Hickey 1972); elevated embryonic abnormalities and mortality (Gilbertson et al. 1991, Fox et al. 1991) and reproductive failure (Weseloh et al. 1995).

Breeding populations of cormorants that had the highest exposure to contaminants showed the most severe impacts, and the species was extirpated from Lakes Michigan and Superior (Postupalsky 1978).

The combination of efforts to reduce levels of environmental contaminants and protection of the species led to its dramatic recovery and subsequent colonization across North America, particularly in the Great Lakes (Weseloh and Ewins 1994). High productivity and survival rates contributed to this recovery, and in the Great Lakes the annual rate of population growth was about 35% or a doubling of the population every three years (Weseloh and Collier 1995). As of the most recent census in 2000, nests on the Great Lakes were as follows: Lake Ontario 24,344; Lake Erie 13,298; Lake Huron 33,914; Lake Superior 2,893 (Weseloh et al. 2002). It is probable that factors in addition to decreased concentrations of contaminants contributed to the population recovery, most likely the increased availability of prey fish. European studies on great cormorants (*Phalacrocorax carbo*) suggest that the size of a colony is directly related to the availability of food resources (van Eerden and Gregersen 1995). In the Great Lakes, forage fish abundance increased dramatically between the 1960s and 70s, following destruction of the native piscivorous species by human exploitation and sea lamprey (*Petromyzon marinus*) (Weseloh et al. 1995). Weseloh and Ewins (1994) showed that high abundance of alewife in eastern Lake Ontario led to increases in cormorant nests three years later. Glahn et al. (1999) suggested that the expansion of the channel catfish (*Ictalurus punctatus*) industry in the

Mississippi region around 1985 and increased survival of wintering cormorants has contributed to the population recovery.

The recent population growth of double-crested cormorants, most notably in the north and east parts of its North American range (Great Lakes, northeast US), has spurred renewed controversy (see www.ofah.org/Cormorant, www.peacfulparks.org). Cormorants have been persecuted for many decades and their natural history, behaviour and ecology may predispose them to conflict with human activities (Nettleship and Duffy 1995). Interspecific conflicts centre on three main areas: 1) competition with humans for recreational and commercial fish species as well as depredation at commercial aquaculture facilities; 2) impacts on other colonial nesting waterbirds and 3) an alteration of vegetation in the colonies and trees used for nesting and roosting. It is beyond the scope of this paper to address 1 or 2; readers are referred to recent studies by Belyea et al. (1999), Ross and Johnson (1999) and Trapp et al. (1999) for impacts of cormorants on fisheries and to Cuthbert et al. (2002) and Weseloh et al. (2002) for a review of impacts on other bird species.

1.2 Possible habitat consequences: Impacts on vegetation and soil

The interaction between aquatic and terrestrial ecosystems is considered an important factor in determining the structure of each ecosystem (Polis and Hurd 1996a, Polis and Hurd 1996b), so the life-history traits (e.g. multi-ecosystem use) of a colonial waterbird such as the double-

crested cormorant likely play an important role is this interaction. Energy and nutrient fluxes may be trophically linked across habitat boundaries, in that they may originate in one system and be transported to another system. This is well established in land to water transfers where nutrients or contaminants may enter water via runoff (Wetzel 1990). However, the reverse can also occur. Great Lakes cormorants feeding on aquatic organisms may transfer nutrients to their breeding colonies. In doing so, the birds act as vectors for the transfer of material from aquatic to terrestrial ecosystems.

Large populations of nesting and roosting birds may exert considerable impact on the disturbance regime and species composition of their nesting sites (Anderson and Polis 1999, Osono et al. 2002, Ellsworth and McComb 2003). Cormorants mechanically break tree branches and leaves during their breeding activities and deposit large amounts of droppings on the forest floor in the form of guano and undigestable food remnants (pellets) (Hebert et al. 2005). Some forests with great cormorant colonies in Lake Biwa, Japan, had 7-22 times the litterfall than in control forests and were estimated to drop 2.2 t/ha/nesting season of guano in high nest density areas (Hobara et al. 2001).

Since seabird diets consist primarily of fish, they excrete nutrient rich guano, high in both nitrogen (N) and phosphorus (P). For example, guano from seabird colonies in the Gulf of California averages 9% N and 11% P (Hutchinson 1950). Other studies on the impact of guano deposition show large amounts of uric acids and ureates are added to the soils (Blockstein

and Tordoff 1985). The hydrogen ion (H^+) concentration of guano varies widely (pH 5.8-8.8) depending on the species and also possibly their diet (Blakemore and Gibbs 1968). This increased deposition to the forest floor has the ability to alter soil properties such as pH and organic matter content and may also impact the biogeochemical cycling of nitrogen and phosphorus. For example, on nutrient poor desert islands in the Gulf of California, seabird colonies had 6 times higher concentrations of N and P than islands with no birds (Anderson and Polis 1999). Data from Great Lakes cormorant sites showed decreased soil pH and increased N and P collected inside colonies compared to soil collected outside the colonies (Cuthbert et al. 2002). Weseloh and Brown (1971) also suggested that seabird breeding may exert detrimental impacts on vegetation through a change in osmotic balance due to salt accumulation.

Enrichment of carbon and nutrients in soil may alter fungal communities, which in turn play a role in the formation of organic matter and nutrient release during decomposition. These fungal community alterations may also impact the symbiotic uptake of plant nutrients from the forest soils (Smith and Read 1997). Osono et al. (2002) reported changes in both the abundance and physiology of fungal communities at bird colonies, most notably a decrease of white rot fungi at colony sites, which have a major role in lignocellulose decomposition from plant tissues. Other work has shown that activity of a ligninolytic enzyme produced by white rot fungi is reduced with increased N availability (Carreiro et al. 2000). These results suggest the

decomposition of plant litter in guano-impacted soils would be influenced by the immobilization and release of nutrients and in forming soil organic matter, altering the C/N ratio.

Vegetation changes, including detrimental effects, associated with bird colonies or roosts have long been observed. Wilson (1814) reported seeing guano covered passenger pigeon roosts of thousands of hectares, where there was no living vegetation, and many fallen trees. Starlings (*Sturnus sp.*) roosting in pine plantations killed trees 2 years after the initiation of roosting activities (Stewart 1933, Young 1936). Weseloh and Brown (1971) demonstrated an inverse relationship between the amount of vegetative ground cover and nest concentration in heron rookeries. The plant communities in the colonies also had lower biodiversity (Weseloh and Brown 1971). Lemmon et al. (1994) showed that cormorants kill trees in 3 to 10 years after nesting begins, and Weseloh et al. (2002) observed that all trees and shrubby vegetation died on several islands in Lake Ontario after cormorants had colonized the area.

Empirical data from such studies have led to management actions in some areas. For example, in 2004 the Ontario Ministry of Natural Resources began culling adult cormorants at Presqu'île Provincial Park in Lake Ontario based on data showing the detrimental impact of nesting birds to vegetation on islands in the park (see www.ene.gov.on.ca/envregistry/022534ep.htm). Similar management options are currently being considered for islands in western Lake Erie (http://www.ontarioparks.com/english/planning_pdf/

east_ToR.pdf).

1.3 Island Vegetation in Western Lake Erie

The three study islands, East Sister Island, Middle Island and Middle Sister Island are situated midway between the Ontario and Ohio shorelines of Lake Erie (Fig. 1.1). The islands comprise some of the most southern parcels of land in Canada. The lake's influence produces a long growing season, comparatively mild winters and dry summers. Such favourable conditions generate a high diversity of plant and animal species, many of which are rare elsewhere in both Ontario and Canada. These islands possess a flora well represented with Carolinian species, with trees such as the common hackberry (*Celtis occidentalis*) and kentucky coffee-tree (*Gymnocladus dioica*) dominating in some areas, yet an absence of other characteristic Carolinian species, such as the tuliptree (*Liriodendron tulipifera*), sassafras (*Sassafras albidum*) and paw paw (*Asimina triloba*) which occur on the nearby Canadian mainland (Kamstra et al. 1995). The island forests have a species composition which is distinct from forests on both the Canadian and American mainland (Boerner 1984). The islands are flat and rise only a few meters above Lake Erie, and the limestone bedrock is very shallow, appearing at the surface in some places. They are affected by lake storms, fluctuating water levels and a range of human disturbance patterns (Kamstra et al. 1995).

Historical waterbird surveys have been conducted on islands in the Great lakes since the 1970s. Recent surveys have shown that the increase in populations of double-crested cormorants on Lake Erie pose some real risks to endangered plant fauna on the Canadian Erie Islands (Hebert et al. 2005, Parks Canada unpubl. data). I tested hypotheses on two scales of impact: 1) forest stand and 2) tree ecophysiology. To evaluate these hypotheses, I classified aerial photographs to assess forest declines in relation to cormorant nesting (Chapter 2). I then discuss information from tree rings to infer alteration of growth, variation in nutrient uptake and soil chemistry changes as a result of cormorant colonization (Chapter 3). The goal of this research is to assess the threat of the cormorant population explosion to one of Canada's rarest ecosystems and provide insights into the need for future management actions. As double-crested cormorant populations increase in the Great Lakes basin and continue to conflict with human recreational and commercial activities, there will be increasing pressure to manage the population.

Figure 1.1. Location of islands in western Lake Erie. Study islands are East Sister Island, Middle Sister Island and Middle Island. Pelee Island and Point Pelee National Park are labelled for reference only.

Chapter 2

Double-crested cormorant colonies are linked to Carolinian forest declines on western Lake Erie islands

2.1 Introduction

Double-crested cormorant (*Phalacrocorax auritus*) populations have increased tremendously throughout their breeding range during the last 25 years (Weseloh et al. 1995, 2002). This is exemplified in Lake Erie, part of the lower Great Lakes, where cormorant breeding populations increased 150-fold between 1979 and 2000. Approximately 80% of these birds were breeding on 2 islands (Middle and East Sister Islands) in the western basin (Hebert et al. 2005). Destruction of vegetation, in addition to impacts on fisheries and other colonial waterbirds, is cited as one of the main issues associated with the substantial increase in breeding cormorant populations (Lemmon et al. 1994, Rippey et al. 2002, Weseloh et al. 2002, Hebert et al. 2005). Carolinian vegetation is one of the rarest types in Canada, occurring only in southwestern Ontario, including the western Lake Erie archipelago. Changes in plant communities on these islands may result in a reduction in plant biodiversity, loss of wildlife habitat, and possible impact on rare species (Weseloh and Brown 1971, Kamstra et al. 1995, Hebert et al. 2005). Other research (Kamstra et al. 1995, Weseloh et al. 2002, Hebert et al. 2005) has observed negative impacts of nesting cormorants on both East Sister and

Middle Islands. Here I attempt to quantify the changes in vegetation structure (health) over time by using infrared aerial photographs.

Aerial photographs have been used for assessment of forest health since the 1950s, mostly in the context of detecting and mitigating the impact of economically important pest species (Frayer and Furnival 1999). Airborne-based remote sensing science has changed dramatically since then, with an increasing variety and number of sensors and improved digital processing techniques (Wulder and Franklin 2003). The theory behind using remote sensing for forest health assessments is that changes in foliage characteristics are potentially detectable with remote sensing instruments because the physical components of the leaf interact with the electromagnetic spectrum. For example, chlorophylls and other pigments absorb within the 0.50-0.75 μm wavelengths (visible light) and the structure of the mesophyll (inner leaf tissue) causes high reflectance of 0.75-1.35 μm wavelengths (near infrared - NIR) (Wiegand et al. 1972). Leaf moisture content results in high absorption of wavelengths between 1.35 and 2.5 μm (infrared - IR) (Franklin 2001). Knowledge of these properties and patterns allows the detection of changes in foliage characteristics using remotely sensed data (e.g. Figure 2.1). It has been shown that early signs of stress may not be visually apparent in leaves, but may be detected using radiometric sensors in that the changes are visible in the NIR portion of the spectrum as a decrease in reflectance, due to the leaf structure (mesophyll) breaking down (Wagner et al. 1989, Singhroy 1995).

In the context of forest health, damage can result from a number of different natural and anthropogenic stressors. These include extreme weather events, disease and pollution (e.g. Essery and Morse 1992, Ramsay et al. 1998, Mukai and Hasegawa 2000). Regardless of the source of stress, forest damage appears to be the result of a decrease in forest productivity and can range in scale from individual leaves to the entire forest stand (Mageau et al. 1995). Certain symptoms can become apparent almost immediately following application of stress while some are manifested only after long term stress. Leaf function breaks down and prolonged periods of stress will produce extreme leaf damage and discoloration and a tree will begin to shed leaves if foliar dysfunction is severe or prolonged (Olthof and King 2000). The approach in using remote sensing is that these changes in spectral response due to altered foliage characteristics can be related to known damage or stress conditions such as defoliation or differences in internal leaf structure (Franklin 2001).

In the Carolinian forests on the western Lake Erie islands, the apex tree species consist of Kentucky coffee tree (*Gymnocladus dioica*), common hackberry (*Celtis occidentalis*), red ash (*Fraxinus pennsylvanica*) and American elm (*Ulmus americana*) and previous work has observed that cormorants prefer to nest in these large trees (Kamstra et al. 1995). Ground surveys in 1988, showed that crown defoliation had occurred in areas where cormorants nest (Kamstra et al. 1995) and until recently (e.g. Hebert et al.

2005), these qualitative ground surveys were the only method used to monitor and assess defoliation.

Here we explore the use of colour infrared film, often called "false-color" film, which uses a combination of special film, lenses, filters and developing techniques to capture reflected near infrared energy (called Near IR) in the 0.7 to 1.3 μm range. Healthy green vegetation is a very strong reflector of infrared radiation and usually appears bright red on color infrared photographs. Colour infrared (CIR) photography is used operationally to assess vegetation health in many situations where commercial logging interests may be impacted (e.g. insect outbreaks, wind damage; Murtha 1972).

This paper examines the application of high resolution aerial photography for mapping crown defoliation on three western Lake Erie islands. I quantify the forest decline on the western Lake Erie islands where large breeding colonies of double-crested cormorants have established themselves over the past 20 years. Specifically, I describe the application of digitized colour infrared (CIR) photography and object oriented image analysis software to quantify temporal trends in defoliation and link estimates of forest cover to nesting cormorants.

From a management perspective, mapping the spatial and temporal patterns in crown defoliation serves two important aims: 1) it identifies patterns that can assess the impact of cormorant nesting on the vegetation

and 2) it establishes a current baseline of the state of vegetation health that can be used to assess efficacy of any future management practices.

2.2 Methods

2.2.1 Study Sites

The 3 study islands are located in the western basin of Lake Erie: Middle Island (18 ha) is part of Point Pelee National Park; East Sister Island (15 ha) is a provincial nature reserve; and Middle Sister Island (4 ha) is privately owned (see Figure 1.1). East Sister Island was first colonized by cormorants in 1985 and Middle Island in 1990, whereas Middle Sister Island has only recently begun to host a breeding colony (Weseloh et al. 2002). On Middle Sister Island there are areas of mature forest that are suitable for cormorant colonization; however, as of 2003 it did not have significant cormorant nests present. For detailed cormorant colonization histories of these islands, refer to Hebert et al (2005). The tree sampling was split into two groups; reference sites and colonized sites on Middle and East Sister Islands. Middle Sister Island is considered our control or reference island since as of 2003 there were negligible numbers of nests (Table 2.1).

2.2.2 Airborne Data

In June 2001 and July 2003, aerial photographs were obtained for the 3 islands in western Lake Erie (Figure 1.1). The photos were taken by the Ontario Ministry of Natural Resources (OMNR) at a scale of 1:10000 using

false-colour infrared film (Kodak Aerochrome Infrared 2443 II). Images were scanned into 24-bit red-green-blue tagged image file format (TIFF) files with a pixel size of 25cm. I then oriented the digital image to the earth (georeferenced) using known locations on the ground, or ground control points (GCPs) using PCI OrthoEngine Airphoto Module (PCI 2001). The ground locations were collected using differential GPS (Trimble 1999) with a horizontal error of less than 1m. When one image (2001) was georeferenced to have an RMS error of less than two pixels (RMS or root mean squared error indicates the discrepancy between the known ground locations and their digitized locations), the second image was georeferenced using the 2001 image, so that the images are 'co-registered', that is that they are lined up accurately.

The respective images were then classified using eCognition v3.0 (Definiens Imaging GmbH 2002). eCognition is an advanced object oriented classification software that is useful in high resolution image analysis in that it allows supervised classification using spectral information from the imagery, as well as contextual information such as texture and shape. For this analysis, ground cover was classified into 3 categories: bare ground, sparse forest and dense forest. Ground locations of the 3 classes (dense, sparse and bare) were located with GPS during field visits and these were used to train the classification algorithm.

The vegetation perimeter and wetland or meadow areas were collected using differential GPS (Trimble 1999). Since the project is focused

on forest impacts (cormorant nesting areas), masks were created to remove the wetland/meadow areas from change detection analysis and the airphotos were clipped to the vegetation boundaries on the islands (i.e. analysis does not include beach area).

2.2.3 Field data collection

In 2003, field visits were made to all three study islands to collect forest data, cormorant nest densities as well as a variety of other measures as described in Chapter 3. During these visits, specific locations (individual trees) that had a range of cormorant nest densities were recorded with GPS. The numbers of cormorant nests at these locations as well as the total number of nests in trees within a 5m radius were recorded. On East Sister Island, individual tree sampling locations were stratified so that approximately half were located in cormorant nesting areas ($n=16$) and half were reference sites ($n=19$), with little or no recent nesting activity. Again on Middle Island, we attempted to stratify the tree sampling to colony ($n=22$) and non-colony sites; however, only a limited number (5) of large trees in non-breeding areas could be located. Middle Sister Island contained only reference sites ($n=20$).

Ground-based measurements of the forest canopy were obtained using a LI-COR LAI-2000 (LI-COR 1992). Leaf Area Index (LAI) is a function of photosynthetically available radiation (PAR) under the canopy and the incoming solar radiation above the canopy (or in an open area; Franklin 2001). The instrument measures radiation passing through the canopy using

a fish-eye optical sensor with a 148° field of view. Above canopy measures were obtained on the open beach (i.e. no trees, clear view of sky) immediately prior to each below canopy reading, and leaf area index (LAI) was calculated using a model of radiative transfer in vegetative canopies (LICOR 1992). LAI is a dimensionless index and is expressed as the 1-sided foliage area per unit ground area (Franklin 2001). It defines an important structural attribute of forests due to its ability to infer energy, gas and water exchanges (Pierce and Running 1988). As such, LAI measurements are useful structural variables in defoliation and forest health assessments as they can add objectivity to normal ground-based survey methods; however predictive capabilities from LAI are limited (Baret and Guyot 1991). The relationship between under canopy PAR and incident solar radiation flattens out at a certain threshold; as more leaves are added, they are obscured by the first few layers. Most remote sensing methods, including CIR airphotos, are powerful at estimating the difference between no leaves and up to three leaves, but saturate when stand LAI exceeds 3 to 4 (Franklin 2001).

2.3.4 Statistical Analysis

Analysis of Variance (ANOVA) or Students t-tests were used to test for differences between site – impact groups. Regression analysis was performed to test for relationships between LAI, % dense canopy and cormorant nests, fitting a 2nd order polynomial model. All analyses satisfied the assumptions (normality, homoscedasticity) of standard parametric

statistics (Zar 1984). All statistics were performed in Statistica v7.1 (Statsoft 2005).

2.3 Results

2.3.1 East Sister Island

The number of nests on East Sister Island between 2001 and 2003 decreased from 5300 to around 4200 (Table 2.1). The airphoto analysis focused on the forested areas of the island, which are the preferred nesting habitat for the birds and it showed that there was a slight decrease in dense forest canopy, from 76% to 71% (Table 2.2). This analysis also showed a significant increase in the area within the forested section that was bare ground (Table 2.2), from 2% in 2001 to 15% in 2003.

The average nest density within 5m of the study tree was significantly greater ($t= 8.18$ $p=0.007$) in the colony areas: 6.1 nests versus 3.2 at the reference sites. Leaf Area Index (LAI) was significantly lower ($t=40.8$, $p<0.001$) within the colonized areas than in the non-colonized areas (Table 2.3) and LAI measurements on East Sister Island were negatively correlated to the number of cormorant nests within a 5m radius of the sampled tree ($r^2=0.40$, $p<0.001$, $n=35$). LAI measurements were positively correlated to the percent of dense forest canopy within 5m as derived from the airphoto analysis ($r^2=0.28$, $p<0.01$, $n=35$; Figure 2.2).

A closer look at the colonized sites on East Sister Island seems to reveal a geographical difference in the vegetation patterns. For instance, the

group of sites ($n=9$) at the eastern part of the island seemed to have more canopy classified as unhealthy in 2001 to healthy in 2003, whereas the colony sites in the mid-section of the island had a large portion of unhealthy vegetation in 2001 and much of this vegetation appeared as bare ground in 2003 (Figure 2.3).

2.3.2 Middle Island

The breeding population on Middle Island was around 1100 pairs in 1995, increased to approximately 5200 pairs in 2001 and increased by 27% from 2001 to 2003. (Weseloh unpubl data; Table 2.1). Airphoto analysis (Figure 2.4) demonstrated a corresponding decline in dense canopy from 93% in 1995 to 84% in 2001 and 61% in 2003 and an increase in sparse canopy from 6% in 1995 to 32% in 2003 (Table 2.2).

Average number of nests in breeding areas was 4.3 ($n=22$) and 2.0 in non-colony areas ($n=5$). LAI measurements in colony areas were significantly lower (mean =1.9) than in the reference areas (mean=3.0; $t=10.3$, $p<0.01$). There was a significant negative correlation between LAI and the number of cormorant nests within a 5m radius of the sampled tree ($r^2=0.341$, $p<0.01$, $n=27$; Figure 2.5a). There was a positive relationship between LAI and the percentage of dense canopy within 5m of the sampled tree ($r^2=0.23$, $p=0.043$, $n=27$; Figure 2.5b).

2.3.3 Middle Sister Island

Middle Sister Island was the smallest of the study islands and is considered my reference site as it only contained approximately 20 cormorant nests in 2001 and 120 nests in 2003 (Weseloh unpubl. data). Airphoto analysis showed a decrease in dense canopy from 96% to 91% during this time, mostly in the southern section of the island (Figure 2.6). The average number of cormorant nests within 5m of our study sites was 0.35 nests and the LAI averaged 2.8 (Table 2.3). The majority ($n=17$) of the trees had 0 nests so correlation analysis with vegetation parameters was not completed due to the limited range of data. All of our sampling points ($n=20$) on Middle Sister had dense canopy in 2001 and all but two had negligible ($< 2\%$) amounts of sparse canopy in 2003 as inferred from aerial photos.

2.4 Discussion

Previous work on the western Lake Erie islands has demonstrated a spatial correlation between areas with nesting cormorants and crown defoliation (Hebert et al. 2005), and other studies have noted qualitative declines in vegetation in cormorant colonies (Kamstra et al. 1995, Rippey et al. 2002). Results presented here corroborate these findings; I showed that declines in forest cover were related to numbers of breeding cormorants on both East Sister and Middle Islands, whereas in the absence of large colonies, forest cover remained stable on Middle Sister Island. In addition, in this study, the temporal airphoto analysis quantitatively demonstrated the negative temporal impacts of cormorant nesting on forest cover.

Declines in dense canopy on Middle Island over the study period were substantial. A 34.4% decrease occurred between 1995 and 2003 when cormorant nest numbers on the island were expanding exponentially. A majority of the decline occurred between 2001 and 2003, (23%) when the nesting population peaked at around 6600 nests. This decrease translates to a 6 hectare loss of a total area dense canopy of 16.8 hectares over 8 years. This change becomes even more striking when we consider the 'preferred' nesting habitat available on the island. Until recently, Middle Island was owned privately and in the 1930s there was a house and an airstrip on the island. On the airphotos (Figure 2.4) there is clearly an 'X' shaped area in the centre of the island, which is the remnant of the runway and taxi area which is now classified as a field and scrub vegetation community, with dogwood (*Cornus spp.*) and sumac (*Rhus spp.*) dominating, as well as younger, smaller specimens of apex species, mainly common hackberry (*Celtis occidentalis*) (Kamstra et al. 1995, Hebert et al. 2005) Cormorants are thought to prefer larger trees for nesting and for that reason few nests are found in this disturbed 4.5 ha area. The classifications from both 2001 and 2003 (Fig. 2.4) show this area to be intact dense canopy with few exceptions. Considering this, the percentage of island habitat that is mature, closed canopy Carolinian forest is 6.5 ha or 36% of the original forested area, much less than the 61% dense canopy forest calculated in Table 2.2.

On East Sister Island, perhaps even more striking than the decline in dense canopy (76% to 71% between 2001 and 2003), is the increase in bare

ground in historically forested areas. This increased from 2 to 15% or 1.5 ha of the island in 2003. This is likely due to the fact that cormorant nesting began in 1981, so there has been a cumulative impact from nesting over 20 years and the destruction of vegetation is at a more advanced stage. A potential weakness in the choice of CIR film as an imagery source is highlighted by the long-term changes. Hebert et al. (2005) hypothesized that the colonization history of cormorant nesting on East Sister Island proceeded from east to west; the eastern most part of the island was colonized first and the western areas are now beginning to be colonized. This was corroborated in our data, where some of the 'non-colony' sites at the western end of the island have new cormorant nests. This may be occurring because as trees die, they become unstable and birds abandon them in favour of healthier more stable nesting platforms (Rippey et al. 2002). Close examination of the classified aerial photos showed that some areas at the eastern end of the island had reverted to dense vegetation in 2003 from sparse in 2001.

The opening of the canopy in these areas as trees died allowed light penetration to the forest floor and the lack of guano deposition following nest abandonment may have acted in concert to allow recolonization and growth of understory species. On East Sister Island these species are most often large dense areas of stinging nettles (*Urtica dioica*) and American pokeweed (*Phytolacca americana*) (pers. ob.). Wilson and Shure (1993) found that *P. americana* was among the most abundant pioneer species in Appalachian forest ecosystems following disturbance where high nutrient levels were

present. Stinging nettle is well known as a nitrophilous (nitrogen loving) species (Hogg et al. 1995). Nutrient enrichment is expected to decrease species diversity and increase vegetation biomass (Odum 1969). Weseloh and Brown (1974) demonstrated this in blue heron rookeries, noting lower plant diversity in areas where colonial waterbirds nested, and qualitative observations from our study sites corroborated this.

This progression from dense forest canopy to tall understory species biomass is a potential source of error and perhaps accounts for some of the variability in the plots relating ground based measures of canopy cover (LAI) to the airphoto data. The lack of spectral resolution of the CIR airphotos, in that it only collects data in 3 bands, may potentially limit its use when combined with digital analysis to reliably discriminate between dense tree cover and dense understory.

The slight vegetation declines on Middle Sister Island also demonstrate where more research is required. These declines were likely due to cormorant impacts; however, these effects were probably associated with loafing and roosting activities rather than nesting. Impacts of juveniles and non-breeding adults during the nesting season as well as adults and juveniles during the time from fledging to migration were not considered in our study design. In 2003, the number of nests in the western basin was 11,112 (22,224 breeding adults). Literature suggests that roughly 15% of the population are non-breeders ($22,224 \times 15\% = 3334$ non-breeders) and if we use the average fledging success of 2 chicks/nest ($2 \times 11,112 = 22,224$

juveniles) from the literature (coefficients from Weseloh et al 1995, Hatch and Weseloh 1999), the estimated cormorant population on western Lake Erie post-fledging would be almost 50,000 birds (47,782).

Other works have suggested that possible impacts associated with roosting may be more intense than nesting, due to the higher densities that are possible when roosting (Ellsworth and McComb 2003), so the impacts of this number of cormorants is likely significant. During the early fall, large congregations of roosting cormorants have been observed at various islands in Lake Erie, including in trees along the south shore of Middle Sister island where most of the reductions in canopy closure were seen (Laird Shutt CWS, pers. comm.). Fall roosting surveys are required to fully quantify the impact of cormorants on island vegetation.

Another possible cause of vegetation change detected from airphoto analysis is climate variability or water stress. Total precipitation data (April to July) from Kingsville (Ontario mainland) show that area received double the amount of rain in 2003 compared to 2001, suggesting that water stress is not responsible for the declines observed on the islands.

The development of a three-class forest canopy classification system for the Carolinian forest on these islands is an initial step toward a long-term forest monitoring program based on remote sensing data. The key property of such a remote sensing monitoring program is that it provides a consistent means to make reliable spatial and temporal comparisons. This study demonstrates the usefulness and limitations of CIR airphotos as well as the

challenges of using infrared photography for island ecosystems. As mentioned above, near-infrared wavelengths (as in CIR airphotos) are responsive to water content, so it is likely that the calibration of an imaging system trying to detect subtle changes in leaf moisture content in forests that are surrounded by water is challenging. This is partially demonstrated by the large variability in the range of colours in the imagery between years (e.g. Figs 2.3, 2.4, 2.6) and also contributes to the problems in discerning between dense forest canopy and dense understory. Imaging systems with more spectral resolution such as multi-spectral or even hyper-spectral sensors would likely remedy this problem and refine the classification; however, this would be accomplished at a higher cost. Continued collaboration with the OMNR and their current CIR system is a cost effective option; however, more attention to camera calibration is required to gain annual consistency of the imagery.

A challenge for park and protected areas managers is to operate with increasingly limited resources while establishing and implementing scientifically sound management plans. Assessment of crown defoliation has traditionally relied on ground surveys and manual interpretation of aerial surveys. Despite their subjectiveness and labour intensiveness they are still commonly used due to a lack of more efficient and accurate technologies. The possibility of using high resolution remote sensing data in combination with automated object extraction and classification for the assessment of vegetation health may allow monitoring programs to be more quantitative and

cost effective. The capability of such a system to provide information on canopy condition through an assessment of physiological functioning (i.e. stress) and to provide a spatial context which can be merged with other spatial information (e.g. location or potential location of listed species) represents a powerful tool for managers and decision makers.

This study demonstrates that digital imagery coupled with object oriented image analysis can clearly map and quantify canopy cover. Here, it was used to effectively evaluate the impact of nesting cormorants on their breeding islands. While there are limitations associated with this approach, the current method provides independent and objective estimates of forest loss and subsequent habitat change associated with large breeding colonies of cormorants. This type of spatial data can be updated relatively cheaply and in a semi-automated fashion and will be invaluable in monitoring future change in these island ecosystems. In addition, such an approach could be applied more generally to address planning activities for other protected areas, e.g. providing baseline information for further research (i.e. climate change) and placement of long term monitoring plots. Clearly, the potential for the application of remotely-sensed data to address conservation issues will increase in the future.

Figure 2.1. Example of radiomagnetic spectrum of healthy and stressed foliage. Adapted from Franklin (2001).

Figure 2.2. Leaf Area Index (LAI) vs. percent dense forest canopy within 5m of sampled tree for East Sister Island, 2003.

Figure 2.3. Colour infrared airphotos and classified images from East Sister Island for 2001 and 2003. In the classified images, green is dense canopy, yellow is sparse canopy, and red is bare ground.

Figure 2.4. Colour infrared airphotos and classified images from Middle Island for 1995, 2001 and 2003. In the classified images, green is dense canopy, yellow is sparse canopy, and red is bare ground.

Figure 2.5. a) Leaf Area Index (LAI) vs number of cormorant nests within 5m of sampled tree for Middle Island, 2003. b) LAI vs percent of dense canopy within 5m of sampled tree.

Figure 2.6. Colour infrared airphotos and classified images from Middle Sister Islands for 2001 and 2003. In the classified images, green is dense canopy, yellow is sparse canopy, and red is bare ground.

Table 2.1. Cormorant populations and density in 2001 and 2003 on the study islands.

Island	Size (ha) (forested areas)	Year	Total Nests	Nest Density (nest/ha)
East Sister Island	10	2001	5300	530
		2003	4200	420
Middle Island	18	2001	5200	289
		2003	6600	367
Middle Sister Island	4	2001	20	5
		2003	120	30

Table 2.2. Vegetation health classes (%) of the study islands in 2001 and 2003 as interpreted from classified airphotos.

Island	Year	%Dense	%Sparse	%Bare
East Sister Island	2001	76	22	2
	2003	71	13	15
Middle Island	1995	93	6	0.6
	2001	84	13	3
	2003	61	32	7
Middle Sister Island	2001	96	3	1
	2003	91	8	1

Table 2.3. Averages and standard error (in parentheses) from sample trees in each of the island/treatment combinations. Superscripts denote statistically significant difference from *t*-test.

Island	Treatment	N	Cormorant nests/sample tree	Cormorant nests/ 5m radius	LAI
East Sister Island	Colonized	16	4.2 ^a (0.70)	6.1 ^a (0.73)	1.3 ^a (0.2)
	Reference	19	2.6 ^b (0.64)	3.2 ^b (0.67)	2.9 ^b (0.15)
Middle Island	Colonized	22	3.3 ^a (0.60)	4.3 ^a (0.62)	1.9 ^a (0.15)
	Reference	5	1.4 ^a (1.23)	2.0 ^a (1.31)	3.0 ^b (0.30)
Middle Sister Island	Reference	20	0.2 (0.62)	0.3 (0.65)	2.8 (0.15)

Chapter 3.

Ecophysiological and biogeochemical impacts of nesting double-crested cormorants on rare island forests in western Lake Erie

3.1 Introduction

The double-crested cormorant (*Phalacrocorax auritus*) population breeding on the Great Lakes has increased tremendously over the last 30 years (Weseloh et al. 2002). On Lakes Ontario and Erie, impact on nesting island vegetation has become apparent (Hebert et al. 2005, Ontario Ministry of Natural Resources 2004). Seabird nesting activities have long been recognized to modify vegetation at nesting and roosting sites (Hutchinson 1950, Ishizuka 1966, Weseloh and Brown 1971). Cormorants specifically have been implicated in forest declines in Japan and Australia (Hobara et al. 2001, Rippey et al. 2002). Other work in North America has demonstrated a decline in forest health at a stand level (Ch. 2, Hebert et al 2005); however, the physiological impacts at a tree level and specific mechanisms of seabird impact have not been explored.

Here we examine physiological variables (tree growth rates), ecological biomarkers (stable isotopes) as well as abiotic parameters (soil chemistry) as indicators of disturbance caused by double-crested cormorant

nesting activities on western Lake Erie islands. The vegetation communities on these islands are rare in Canada. With cormorant management actions underway in Canada and the U.S. (Glahn et al. 2000, Ontario Ministry of Natural Resources 2004), such work is timely to evaluate recovery potential and to monitor the rehabilitation of these ecosystems.

Annual variations in tree-ring width have been used to analyze variations in climate, to date historical events and recently as tools to detect environmental stress and forest decline. Variations in radial growth rates are induced by variations in environmental signals interacting with plant processes (Fritts 1976). The study of tree ring chronologies has greatly increased understanding of disturbance processes, stand dynamics and forest history (Fritts and Swetnam 1989). Dendroecological analysis has been used to examine natural disturbance patterns such as insect outbreaks and fire (e.g. Swetnam and Betancourt 1998, Ryerson et al. 2003) as well as anthropogenic impacts such as land-use change, air pollution and increased nitrogen deposition (Cook 1988, Ruffner and Abrams 1998, Juknys et al. 2002). Of particular relevance to this study is some recent work using tree-ring analysis to detect growth changes induced by increased nitrogen inputs to forest systems. For example, Juknys et al. (2002) studied forests surrounding a fertilizer plant in Lithuania which began operation in 1965 and dramatically lowered emissions in the 1990s with the fall of the Soviet Union. They were able to single out three distinct periods of anthropogenic transformations in tree growth using tree rings: fertilization period when

emitted compounds had a positive impact on tree growth; a depression period when increased emissions suppressed growth; and a recovery period when a reduction in N emissions led to normal growth rates (Juknys et al. 2002). Other work on the Pacific coast of Canada and the U.S. has documented the importance of inputs of marine-derived nutrients to riparian communities in salmon streams (Hilderbrand et al. 1999, Reimchen 2000). Growth rates of dominant riparian trees have been shown to be 1.5 to 3 times higher on rivers where salmon spawn (Helfield and Naiman 2001) and tree chronologies have been used to reconstruct historical salmon runs in the Pacific Northwest (Drake et al. 2002).

Recently, some advanced chemical tracer techniques have been used in conjunction with tree ring research to infer environmental conditions through the period in which the tree has grown (Nabais et al. 1999, McCarroll and Loader 2004). Stable isotopes of carbon, hydrogen, oxygen and nitrogen are fixed in annual growth rings. Their isotope values reflect air, soil and water conditions. It is assumed that isotope values in annual growth rings provide a record of the chemical composition of the sap during the period of annual growth (Momoshima and Bodiatti 1990). The physiological controls that influence isotope ratios in tree rings are well understood and relatively simple (see McCarroll and Loader 2004), especially in comparison to the multiple factors controlling annual growth increments (e.g. O'Leary 1981, Evans 2001).

Leaf carbon isotope discrimination indicates the ratio of leaf internal to external CO₂ concentration and integrates the effects of factors affecting inward CO₂ diffusion (leaf conductance via stomata) and CO₂ consumption (photosynthetic rate via RUBISCO) (Ehleringer et al. 1993). It is known that $\delta^{13}\text{C}$ of C-3 plants changes with various environmental stressors and is dependent on the type and degree of stress. Water stress, sulphur dioxide and ozone pollution have been shown to cause higher $\delta^{13}\text{C}$ values (Farquhar et al. 1989, Elsik et al. 1993, Saurer et al. 1995) whereas stress from low light levels and acid misting cause a decrease in $\delta^{13}\text{C}$ values (Ehleringer et al. 1987, Heaton and Crossley 1995). Carbon isotope values have been correlated with a variety of factors regulating gas exchange including leaf conductance, hydraulic conductivity, water-use efficiency and photosynthetic capacity (Farquhar et al. 1989, Ehleringer et al. 1993). Abiotic factors such as soil moisture availability and humidity have also been shown to impact $\delta^{13}\text{C}$ values in the plant (Ehleringer et al. 1993).

Recent work has found that the frequently limiting element nitrogen (N) is found in tree rings and, although some inter-annual translocation between rings may occur, stable isotope analysis of tree rings may allow for the assessment of long term changes in soil N availability at a particular site (Hart and Classen 2003, Elhani et al. 2003). Saurer et al. (2004) measured $\delta^{15}\text{N}$ values in tree rings and observed alterations (increase in $\delta^{15}\text{N}$) in nitrogen dynamics in forests as a result of car exhaust. Bukata and Kyser (2005)

demonstrated that biogeochemical changes in nitrogen cycling induced by land-use alteration could be detected using nitrogen isotopes in tree rings. Nitrogen isotopes in vegetation have also been used to demonstrate the importance of marine-derived nutrients for many ecosystems, including salmon streams on the Pacific coast of North America (Mathewson et al. 2003), coral cay vegetation in Australia (Schmidt et al. 2004) and around Lake Biwa in Japan (Mizutani and Wada 1988). Nitrogen inputs in the form of seabird guano, with uric acid as the main excretory product, have the potential to impact biogeochemical cycling of nutrients and the biotic functioning of the soil, which in turn may be detected in vegetation (Lindeboom 1984). Ornithogenic nitrogen additions at seabird colonies are estimated to be higher than inputs in many modern agricultural systems; up to $1000 \text{ kg N ha}^{-1} \text{ y}^{-1}$ (Hobara et al. 2001, Schmidt et al. 2004).

Given the possibility of high nitrogenous inputs at these sites, the soil may experience nitrogen saturation, defined as the situation where N inputs exceed biotic and abiotic fixation (Aber et al. 1989). Increasing amounts of N in ecosystems may lead to a release of N from the soil-N pool, as a result of changes in the mineralization and nitrification rate (McNulty et al. 1990). Excess N input results in acidification and/or eutrophication of the soil which may lead to nutrient imbalances (Roelofs et al. 1985). Other research has shown that initially, increased N inputs to N-limited ecosystems will lead to a stimulation of tree growth (Tamm 1991, Juknys et al. 2002). However, excess N inputs can lead to conditions that lower growth rates of trees,

nutritional imbalances, deterioration of the root systems, disappearance of mycorrhizal fungi and replacement of a nutrient-poor adapted ground vegetation community by a nitrophilous one (Boxman et al. 1998). As a result, trees may become increasingly susceptible to secondary stress factors such as drought, frost, fungi and parasites (Long and Davis 1999, Juknys et al. 2002). Magill and Aber (1998) also demonstrated that soil fauna involved in decomposition are negatively affected by high N concentrations in soil. Using N isotopes, Schmidt et al. (2004) found that altered nutrient cycling in seabird colonies suggested nitrogen saturation of island soils. Studies on Laurentian Great Lakes cormorant colonies found a significant increase in the amount of soil nitrogen, phosphorus, and decreased soil pH within the colony boundaries, however vegetation impacts were not discussed (Cuthbert et al. 2002).

Previous work has demonstrated the detrimental effect of nesting double-crested cormorants on forest canopy in western Lake Erie (Chapter 2, Hebert et al. 2005). However, little is known about the forest ecology on these islands and the impact of factors such as fluctuating water levels, lake storms and historic seabird communities that would not affect mainland forest stands. The relative impacts of parameters that have been shown to limit tree growth, such as precipitation and nutrient cycling, are not known for the Erie islands. This chapter examines the impact of nesting double-crested cormorants on tree eco-physiology and nutrient dynamics on these unique island ecosystems.

First I use dendrochronology to test for changes in growth rates of trees, before and after cormorants began nesting on the islands. I then examine stable carbon and nitrogen isotopes in tree rings and foliage for potential signs of stress (carbon) and altered nutrient uptake (nitrogen). The heavier ^{13}C isotope is discriminated against during photosynthesis, mediated by the diffusion of CO_2 from the atmosphere and the rate of carboxylation during photosynthesis. The mechanism by which cormorant nesting may induce a stress response is not known, so the direction of response is unclear. Cormorant guano is enriched in ^{15}N , so the presence of nesting cormorants would be expected to increase $\delta^{15}\text{N}$ values in tree rings. Finally, I describe the differences in various soil chemistry parameters that occur in cormorant colonized areas and potential outcomes for vegetation.

3.2 Methods

3.2.1 Sample Sites and Field Methods

Common hackberry (*Celtis occidentalis*) (referred to hereafter as 'hackberry') is a deciduous tree in the elm family (Ulmaceae) and was chosen as my study species due to its presence in many Carolinian forest associations and ubiquity on all 3 study islands (Kamstra et al 1995). A literature review and query of the ITRDB (International Tree Ring Database)

found no previous dendrochronological studies performed using *C. occidentalis*. For a complete description of the islands, see Chapters 1 & 2.

My study examined three islands, two of which had areas with large cormorant colonies. A total of 5 sites were established: a) Middle Sister Island – non-colonized; b) East Sister Island – non-colonized; c) East Sister Island – cormorant colony; d) Middle Island – non-colonized and e) Middle Island – cormorant colony. I attempted to core 20 trees in each group, and two cores were taken from each tree at breast height using a 5mm increment borer. Trees were not chosen as a true random sample, but rather the principle criterion was ‘old-growth’ as judged from circumference. Soil cores were taken with a 2.5cm diameter step-in soil corer. Forest floor litter was removed by hand before taking the core, and cores were taken to a depth of 15cm when possible. GPS locations were collected using a Trimble GeoExplorer 3 (Trimble 2001). Diameter at breast height (1.3m) was measured using a standard forestry DBH tape. The Canadian Wildlife Service attempted to visit and census all of the known cormorant colonies on the Great Lakes and all apparently occupied nests were counted individually (Weseloh et al. 2002).

3.2.2 Climate and Lake Level data

Regional records of total monthly precipitation and temperature were obtained from the Pelee Island recording station for 1889 to 1987

(http://www.climate.weatheroffice.ec.gc.ca/climateData/canada_e.html).

Monthly Lake Erie water levels were obtained from the NOAA website (Great Lakes Environmental Laboratory)

((<http://www.glerl.noaa.gov/data/now/wlevels/levels.html>) from 1918-2003.

Summer (June, July, August) Palmer Drought Severity Index (PDSI) from instrumental records (1900-current) from northern Ohio were obtained from the National Climate Data Centre (Boulder, CO)

(<http://www.ncdc.noaa.gov/paleo/pdsidata.html>).

3.2.3 Tree Ring Analysis

In order to assess changes in growth increments that may be caused by cormorant nesting activities, I evaluated normal growth in the absence of cormorants and looked for deviations from normal growth in cormorant impacted sites. Specifically, I established a climate–tree ring relationship for the period preceding cormorant colonization (i.e. pre-1990) and then compared this relationship with that from trees in colony sites. The assumption here, as noted by Cook (1987), is that if there is a breakdown in the relationship between tree growth and climate, an external stressor is involved. The key is to verify the accuracy and stability of the climate-tree ring model prior to predicting tree growth from climate during the suspected period of abnormal decline.

All cores were mounted, dried, sanded and dated according to standard dendrochronological techniques (see Fritts 1976). Cores were mounted using masking tape rather than glue to ensure the validity of stable isotope measurements. Ring width increments were measured to the nearest 0.001 mm with a Velmex measuring system and Measure J2X™. The accuracy of the series (individual core) measurements were verified using COFECHA, a computer based quality control cross dating program (Holmes 1986). The program creates a master chronology of mean ring widths from a series of input trees and flags potentially problematic (i.e. incorrectly dated) series based on correlation with the master chronology.

Detrending, a statistical operation to remove temporal trend from tree-ring series, is often applied in dendrochronology to remove features thought to distort or obscure the relationships of interest (Fritts 1976). For this study, a two step detrending was used. The first detrending used a negative exponential or regression line to fit the data (Fritts 1976) to remove the influences of age-related growth. This standard procedure is based on knowledge of the physical system: if the same amount of biomass is produced on an ever increasing circumference, the ring width would be expected to decline (Fritts 1976). The second detrending involved fitting cubic splines with 50% frequency responses at wavelengths of 50 years to remove high frequency variability associated with stand dynamics (Cook and Holmes 1986). This procedure recognizes that within-stand processes over time may produce growth anomalies. For example, during natural forest

succession, local effects due to tree removal and recruitment alter variables such as competition for light and nutrients as well as micro-climate.

Recognizing that tree growth in a given year may be dependent on physiological processes from the previous year(s) (i.e. stored photosynthate); statistical removal of autocorrelation in tree ring indices is routine (Fritts 1976). To accomplish this, the individual detrended series were averaged into a 'standard' final chronology. The program ARSTAN then applies autoregressive-moving average models to the tree-ring indices to produce a "residual" chronology without significant autocorrelation (Cook 1987). These residual chronologies are better reflections of the original environmental inputs that affect growth independent of internal biological processes. Removal of significant autocorrelation also simplifies significance tests of the relationship between tree-rings and climate (Cook and Cole 1991). Thus, to examine the relationship of tree rings to climate variables, the residual ring width index after detrending and standardization was used as the dependent variable in a regression model.

In order to maximize the number of series and the time period covered for the climate predictions, individual series (trees) from Middle Island and Middle Sister Island were pooled to generate a pre-cormorant chronology. To remove any possible growth effects caused by cormorants, all series used in the chronology were truncated at the earliest date of colonization (1989; Weseloh et al. 2002). Climate records from Pelee Island were available from 1889 to 1987 (<http://www.climate.weatheroffice.ec.gc.ca/climateData/>

canada_e.html). Climate factors used as predictor variables in multiple regression analysis included mean monthly temperatures and monthly precipitation from the year of growth and the year previous; monthly Lake Erie water levels; and the summer Palmer-Drought Severity Index (PDSI). Other work in the Great Lakes basin has shown tree ring relationships with similar variables (Buckley et al. 2004).

A modelled chronology up to 2002 was generated using coefficients from the above regression model. To test the stability of the tree ring – climate relationship in the absence of cormorants, a precolonization verification period of 10 years (1978-1988) was selected. Modeled pre-cormorant tree ring indices were generated to 2002 and compared to chronologies from Middle Island from the post colonization period (1990-2002). Product-moment correlations were used to compare the correspondence between annual index values of the actual and predicted chronologies in the pre- and post cormorant periods.

On East Sister Island, chronologies were not long enough to use the climate modeling approach. However, adequate samples were obtained for both reference and cormorant colonized sites to allow development of standard chronologies using a similar process as described above. Due to the short time series on East Sister, I used a single negative exponential or linear detrending method and autoregressive modeling to remove autocorrelation in the chronologies. I used product-moment correlations

before and after cormorant colonization (1985) to look for correspondence in the two chronologies.

At the colony sites, I compared the variance of the tree ring index for the period before colonization (pre-1989 for Middle Island and pre-1986 for East Sister Island) to the variance post colonization; statistical significance was assessed using an F-test.

3.2.4. Stable Isotope Analysis

East Sister Island and Middle Sister Island were the focus of the stable isotope study in tree rings; East Sister provided reference and colonized sites and Middle Sister Island provided a long-term reference site. On East Sister Island, 5 cores from each site were selected for isotope analysis. Due to low N content in wood a large amount of sample is required (30mg) to obtain a reliable estimate of $\delta^{15}\text{N}$; so the entire core was pooled into 5-year groups for analysis. $\delta^{13}\text{C}$ in tree rings from 1986 to 2002 were analyzed individually, and years previous to that were pooled into 5 year bins. For Middle Sister Island, 1 core was analyzed in 10 year increments for both C and N isotopes.

Foliage data is presented for Middle Sister Island (reference) and Middle Island (cormorant colonized). Foliage samples from East Sister Island were not analyzed; all collected foliage was heavily soiled with guano in what were considered to be reference sites as well as sites colonized by

cormorants. Guano samples from the leaves were submitted for $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ analysis.

Cores selected for isotope analysis were cut into individual years using scissors. Individual rings were ground using a new commercially available coffee grinder. Foliage samples for isotope analysis were dried at room temperature for 1 week and pulverized using a Wig-L-BugTM ball mill (Dentsply Ltd, Elgin, IL, USA). Guano samples were retrieved from leaves collected from cormorant colonized areas and did not require homogenization after drying at room temperature. Samples were weighed on a microbalance (Mettler Toledo MX5) into tin capsules: 30mg for wood; 3mg for foliage; and 1mg for guano.

The following isotope procedures were completed by the G.G. Hatch Stable Isotope Laboratory at the University of Ottawa. Tin capsules were flash combusted at 1800°C in a Vario EL III (Elementar, Germany) elemental analyzer (E.A.) and resulting gases were carried via helium through the E.A. to purify and separate into N_2 and CO_2 . Gases were carried from the E.A. into a DeltaPlus (ThermoFinnigan, Germany) isotope ratio mass spectrometer (IRMS). $\delta^{13}\text{C}$ are expressed relative to VPDB (Vienna Peedee Belemnite). The $\delta^{15}\text{N}$ values were background corrected and reported relative to air. Data were normalized using internal standards and analytical precision (2 sigma) was $\pm 0.2\text{‰}$.

One-way ANOVAs were used to look at differences in $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ values in foliage between sites. To look at trends in stable isotopes in tree rings on East Sister Island we used regressions in individual trees, examining $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ vs. year in non-colonized and colonized sites

3.2.5 Soil Nutrient Analysis

Soil nutrients were analyzed from samples taken at all 3 study islands. One gram of soil was placed in a flask with 250ml of Milli-Q water (distilled) and allowed to sit for 24 hours. A portion of the solution was filtered through a GF/B filter (for nitrate (NO_3) analysis) or a GF/C filter for total phosphorus analysis. Nitrates (NO_3) and total phosphorus were analyzed on a Bran+Luebbe AutoAnalyzer 3. Ammonium ion (NH_4^+) was analyzed using a Turner Designs handheld fluorometer as outlined by Holmes et al (1999). pH was measured using a Corning pH meter 220. Percent (%) organic content was determined using weights before and after heating in a muffle furnace according to the following equation: % organic content = $(B-C)/B \times 100$, where B is the weight of dry soil and C is the weight of soil after muffle furnace incineration.

Differences in concentrations between islands and colony/non-colony status were analyzed using one way ANOVAs. I then looked at relationships between the number of cormorants nesting within 5m of the tree in 2003 and soil chemistry parameters using regression analysis.

3.3 Results

3.3.1 Dendrochronology

A total of 81 growth series from individual trees (41 reference and 40 cormorant colony) were used to develop chronologies. ANOVA was used to test for differences among means and Tukey's HSD post hoc comparisons were used (Statistica v 7.0). Descriptive statistics of raw ring width series for each island site are listed in Table 3.1. Diameters at breast heights were not significantly different between any of the 4 sites ($p=0.21$, $F=1.53$, $n=76$). The mean ring widths for the individual series were significantly lower on Middle Sister and Middle Islands than on either site on East Sister Island ($p<0.0001$, $F=36.9$, $n=81$). Mean sensitivities were significantly different among islands ($p<0.0001$, $F=34.8$, $n=81$). Given that sensitivity is a measure of variance within the raw tree ring widths of a given tree, sensitivity is likely related to mean ring width. Trees on East Sister Island were significantly younger on average than either Middle or Middle Sister Islands ($p=0.002$, $F=5.64$, $n=81$) (see Table 3.1 for details and post-hoc comparisons).

The master residual chronology (Middle and Middle Sister pre 1989) was developed using double detrending and autoregressive modeling and is shown in Figure 3.1. The trend shows lower than normal growth during the

following periods: 1880-1900, 1918 to 1940 and from the late 1950's to around 1970.

A stepwise regression model showed that 25% of the variance in the master chronology could be explained by PDSI, with no additional variance in tree ring width explained by instrumental temperature or precipitation averages from Pelee Island ($r^2=0.25$, $p<0.0001$, $n=90$; TR index = $0.076 \times \text{PDSI} + 0.981$).

Predicted index to 2002 and actual index for Middle Island chronologies are shown in (Figure 3.2). To verify the predictions pre colonization, a 10 year period pre-colonization was selected, during this period product-moment correlations were significant between the modeled index and the actual chronology ($p<0.05$, $r=0.68$, $n=10$). A similar analysis for the 12 years post colonization also showed a significant relationship between the modeled index and the actual chronology ($p<0.05$, $r=0.56$, $n=13$). There was no significant difference detected in the growth ring variance pre and post colonization ($F_{(0.05(2), 24, 18)} = 1.81$ $p > 0.05$).

On East Sister Island, pre cormorant correspondence between the modeled chronology and the chronology derived from colonized trees were significantly correlated ($p<0.05$, $r=0.53$, $n=24$). Post cormorant periods of the chronologies were also significantly correlated ($p<0.05$, $r=0.56$, $n=18$). The variance in tree ring chronologies (age detrended) were significantly lower pre-1986, suggesting an increase in variability with nesting cormorants ($F_{(0.05(2), 24, 18)} = 2.58$, $p<0.05$).

3.3.2. Stable Isotopes

On East Sister Island, the $\delta^{13}\text{C}$ values of tree rings for the pre-1986 period ranged from -28‰ to -23.5‰ at the non-colonized sites and from -28‰ to -25‰ at the colonized sites (Figure 3.3a, b). $\delta^{13}\text{C}$ values for all 5 individual trees from colony sites were positively related to year, whereas all 5 non-colony trees showed no trend through time (Table 3.2; Figure 3.3). Similar trends were observed in tree $\delta^{15}\text{N}$ values (Figure 3.4 a, b); results from regression analysis performed on individual trees were similar; 75% of colony trees showed an increase in $\delta^{15}\text{N}$, whereas only 20% of non-colony trees increased over the study period (Table 3.2). However, a homogeneity of slopes test (ANCOVA model including a dummy variable for tree) revealed no significant difference in slopes between trees from reference or colonized sites for either $\delta^{13}\text{C}$ or $\delta^{15}\text{N}$ ($\delta^{13}\text{C}$; $F=0.21$, $p=0.66$, $\delta^{15}\text{N}$; $F=0.46$, $p=0.46$). As mentioned above, foliage samples collected from both reference and cormorant trees were heavily coated in guano (approx. > 50% by area) and were not analyzed for isotopic signatures. A lab error resulted in the loss of one non-colonized $\delta^{15}\text{N}$ sample.

Middle Sister Island had a range of approx. 2.5‰ in $\delta^{13}\text{C}$ over 110 years of growth rings, from -26.5‰ to -24.0‰ (Figure 3.5a). There was a positive relationship between $\delta^{13}\text{C}$ and PDSI averaged by decade ($p<0.05$,

$r^2=0.42$, $n=10$). $\delta^{15}\text{N}$ increased from around 8‰ in the late 1800s-early 1900s to around 12‰ around the 1920s. Values since then have been stable between 12-13‰ (Figure 3.5b).

$\delta^{13}\text{C}$ values in foliage averaged -27.4‰ on Middle Sister Island and were significantly lower than those on cormorant impacted Middle Island (mean = -26.05‰; $F(1,11)=10.981$, $p<0.05$, $n=13$). There was no significant difference in $\delta^{15}\text{N}$ in foliage between Middle and Middle Sister Island ($p=0.57$, $n=13$).

$\delta^{15}\text{N}$ values in guano averaged 24.4 $\pm$ 1.0‰ ($n=3$).

3.3.3. Soil chemistry

Soil nutrients were analyzed from non-colonized and colonized areas on East Sister Island, colonized areas on Middle Island and non-colonized areas on Middle Sister Island. Mean concentration and other information on soil parameters analyzed are summarized in Table 3.3. All parameters demonstrated statistically significant differences between areas, but patterns between islands and colonization status were not consistent. On East Sister Island, colonized areas had higher NO_3^- and NH_4^+ compared to non-colonized areas but no difference was detected for phosphorus (Tukey's HSD post hoc; see Table 3.3 for details). Middle Island soils in colonized areas had higher NO_3^- and phosphorus, but no difference was detected in NH_4^+ concentrations compared to reference areas (Tukey's HSD post hoc; see Table 3.3 for

details). On East Sister Island, pH was significantly lower at colonized areas than at non-colonized areas ($p=0.012$, $F=4.99$, $n=20$). % Organic matter was significantly higher on Middle Island in colonized areas than in non-colonized areas on East Sister island or Middle Sister Island ($p<0.001$, $F=22.74$, $n=20$; Table 3.3). There were no differences detected between the reference sites (East and Middle Sister Islands) in any parameters measured (Table 3.3).

When including data from all islands, significant positive relationships were observed between the number of cormorant nests and both soil ammonia and nitrate concentrations (Figure 3.6 a,b). There was a significant negative relationship between cormorant nests in 2003 and pH ($r^2= 0.32$, $p<0.01$, $n = 20$; Figure 3.6c).

3.4 Discussion

Annual growth rings in trees are a function of several interrelated biological, physical, stand and climate factors. The isotopic composition of carbon dioxide and water used by a tree for growth are modified by the tree in response to environmental conditions. These modifications are recorded in the isotopic composition of wood which provides an archive for reconstructing past environments. Many activities associated with colonial waterbird nesting have the potential to alter environmental conditions including: nutrient/carbon additions to soil; defoliation from physical breaking of limbs and leaves or acidic guano deposition on foliage. Studies examining the impact of

cormorants on breeding habitats found that they altered soil parameters and negatively affected forest health (Cuthbert et al. 2002, Hebert et al. 2005). In eastern Lake Ontario, cormorant management is underway as a result of concern regarding their impact on rare trees in a protected area (OMNR 2004), yet few data exist to link the presence of cormorants to physiological endpoints in vegetation. Results presented here demonstrate, that in areas where cormorants nest, nitrogen dynamics were altered and there was an increase in stress in hackberry trees used for nesting.

The weather pattern associated with south-western Ontario island locations is characterized by relatively mild winters (average minimum temperature -7.1°C), warm summers (average maximum temperature 29.9°C) and ample precipitation (mean annual precipitation around 818.5mm). This is expressed mathematically in the low mean sensitivity of the series (Table 3.1). Mean sensitivity is considered an indicator of the responsiveness of a tree to prevailing environmental conditions (Fritts 1976). Relationships between tree rings and climatic factors are usually weak even in the best 'dendrochronological' situation; the coefficient of correlation usually does not exceed 0.3-0.4, making the interpretation of other environmental stressors using tree rings a challenge. The conditions present on the Lake Erie islands are considered sub optimal for dendrochronological research: the temperate deciduous forest is the climax vegetation with the tempering influence of a large body of water and possibly a high water table. Many tree ring studies often use conifers as the ring boundaries are easier to

discriminate; often in extreme environments at the limits of their ranges (e.g. altitude, latitude; Fritts 1976).

In our study, the summer Palmer Drought Severity Index (PDSI) explained 25% of the variation in ring width over the period 1900 to 1989. The PDSI measures accumulated effects of monthly rainfall deficit or surplus relative to the monthly 'climatologically appropriate' rainfall. This is defined as the rainfall needed to maintain adequate soil water content for normal growth of plants in a region and is calculated from the soil water balance from such processes as evaporation, transpiration and runoff (Palmer 1965). The PDSI is an index that measures the departure of the moisture supply and varies roughly between -6.0 and +6.0, positive values denoting 'wetter' periods and negative meaning 'drier' periods. None of the local weather data variables (Pelee Island) explained any of the remaining variability in tree growth. Other work in the Great Lakes basin has also shown weak climate relationships. Buckley et al. (2004) inferred summer precipitation for southern Ontario using eastern white cedar (*Thuja occidentalis*) where they were only able to explain 21% of the variability in this climate sensitive species in an ideal dendrochronological location (i.e. rock outcrops on Niagara Escarpment). They also noticed an unexplained weakening of the climate – tree growth relationship in recent decades (post 1960).

Significant correlations using both the climate forecast chronology and measured chronology for both pre and post cormorant colonization periods suggested that impacts from nesting cormorants were not detectable using

tree ring analyses; no alteration in the climate-tree ring relationship was observed for either colonized area (Middle or East Sister Island). Other work has shown that radial growth measurements taken at the base of the tree (i.e. breast height) may underestimate growth reduction in cases where stress to tree growth is from crown defoliation (e.g. insects; Swetnam et al. 1985). Additionally, growth reduction may lag 1-3 years behind the onset of defoliation in the case of spruce budworm (*Choristoneura sp*). It is important to note that in other studies successfully using tree rings to illustrate impacts of an environmental stressor other than climate (i.e. insects, air pollution), the stressor usually impacts the entire stand or region in a synchronous fashion (Ryerson et al. 2003). Even though the islands studied here are small (< 20ha), the impact of cormorant nesting on tree health is localized and has changed over time. Although we have total nest estimates on these islands through time, we do not have detailed data regarding the spatial dynamics of the colonies. As populations increased over the last 20 years, in some areas the colonized area has shifted as trees died and became unstable nesting platforms (Hebert et al. 2005). Although our analysis was divided into pre and post colonization periods, it is difficult to know the chronology of nesting in our study trees. It is unlikely that all the trees in our sample experienced synchronous stress from nesting, further diluting any change in growth signal inferred from tree rings. A goal of the nitrogen isotope analysis was to examine if we could estimate the date of colonization of an individual tree by the onset of change in $\delta^{15}\text{N}$ in the growth ring. The large amount of variability

through time in both the colonized and non-colonized trees (Figure 3.4) and the documented translocation of N across annual growth rings (Hart and Classen 2003) make this an unlikely possibility without further refinement of the method.

Stable carbon isotopes in tree rings may provide a tool to uncouple climatic from local effects on tree growth. Methods using $\delta^{13}\text{C}$ records in annual growth rings have been used to assess various environmental stresses in mature forest trees (e.g. Sakata and Suzuki 1998, McLaughlin and Downing 1995). Plant $\delta^{13}\text{C}$ varies due to ecophysiological responses to environmental conditions that influence rates of carbon fixation and stomatal conductance (McCarroll and Loader 2004). Our results suggest that trees in cormorant colonized areas are experiencing increased levels of stress manifested as a shift in $\delta^{13}\text{C}$ detected in both foliage and tree ring samples. Foliage $\delta^{13}\text{C}$ were significantly greater from trees in colonized areas on Middle Island which suggests less isotopic discrimination was occurring than in leaves collected from a non-colonized area on Middle Sister Island. This result must be qualified, however, because other variables such as micro-climate and soil characteristics (Saurer et al 1995, 1997) are known to affect $\delta^{13}\text{C}$ values and these are not accounted for here.

Stable carbon isotopes in tree rings on East Sister provided more conclusive evidence of the impact of cormorants on nest trees. In colonized trees there was an indication of increased stress over time. Baseline $\delta^{13}\text{C}$

(pre-cormorant) exhibit a large range of values, perhaps indicative of some of the micro-site variability that may regulate carbon isotope fractionation (soil properties, competition, light). For example, Kamstra et al. (1995) identified two mesic forest associations and two wet forest associations on East Sister Island, both of which contain common hackberry, but trees in these areas may have significantly different water use efficiencies and thus different $\delta^{13}\text{C}$ signatures. Due to the nature of these island forests, there may also be high variability in $\delta^{13}\text{C}$ values depending on the location of the tree. For example, exposed trees closer to the shoreline would likely experience lower humidity, higher evapotranspiration and possibly higher irradiance than protected trees on the interior of the island, all of which have been shown to impact $\delta^{13}\text{C}$ in trees (Leavitt and Long 1991, Saurer and Siegenthaler 1989). The data presented here does not allow for robust testing of these alternate hypotheses for the variability in baseline C isotope values. Despite inter-tree differences in baseline $\delta^{13}\text{C}$ values, all 5 nesting trees exhibited an increase in $\delta^{13}\text{C}$ over time, whereas the same trend for the reference trees was not observed (Fig 3.3 a,b).

The heavier ^{13}C isotope is discriminated against during photosynthesis and in C3 plants, this is mediated by the diffusion of CO_2 from the atmosphere and the rate of carboxylation during photosynthesis. Several mechanisms of stress induced by cormorant nesting are possible. As mentioned previously, all surfaces were frequently covered in guano at

colony sites, including foliage of trees and understory vegetation. This poses a potential physical barrier to the diffusion of CO₂ through the stomata, leading to increased $\delta^{13}\text{C}$ values through decreased C isotope fractionation.

The $\delta^{15}\text{N}$ value of nitrogen that is bioavailable to a tree can be affected by either a) the introduction of a different nitrogen source with a different isotopic signature or b) altering the processes that make nitrogen available for uptake (Hogberg 1997). $\delta^{15}\text{N}$ values in tree rings have primarily been used to demonstrate a), mostly in the context of anthropogenic N emissions (e.g. Saurer et al 2004). In bird colonies, many studies have shown that foliar $\delta^{15}\text{N}$ is enriched, approaching the signal evident in organic inputs (Mizutani and Wada 1988, Schmidt et al 2004). Inorganic nitrogen inputs through precipitation and N₂ fixation seem relatively unimportant in penguin rookeries (Lindboom 1984), and Mizutani and Wada (1988) noted that nitrification and denitrification were minor in the nitrogen flow in gull colonies. Instead, microbial degradation of organic nitrogen of avian origin and volatilization of resulting ammonia were quantitatively the most important nitrogen processes in seabird rookeries (Mizutani and Wada 1988). Our results corroborate this; significantly higher $\delta^{15}\text{N}$ values were found in tree rings from islands with nesting cormorants. On such nesting islands, $\delta^{15}\text{N}$ values in wood were similar to those measured in cormorant guano. High $\delta^{15}\text{N}$ values in guano were likely the result of ammonia volatilization which discriminates against

the heavier isotope, similar to data from soils in Antarctic seabird rookeries (Wada and Shibata 1981).

On East Sister Island, absolute differences in wood $\delta^{15}\text{N}$ values between colonized and non-colonized areas were less clear. Positive relationships between time and $\delta^{15}\text{N}$ values in trees from colonized areas were more evident (75% of samples) than in trees from non-colonized areas (20%).

A potential problem with our experimental design was the use of nest density as an index of seabird nutrient input. There are a number of other activities associated with nesting, such as foraging trips, roosting, and loafing, that may lead to input of guano to other areas of the islands, areas that we may have considered 'reference' sites. Historically there have been other colonial waterbirds that nest in trees on all of these islands, in fact the largest Great Egret colony on the lower lakes is on East Sister Island (Weseloh et al 1988). Our study cannot account for these possible additions in the past since these birds also deposit ^{15}N enriched guano to the forest floor.

There are other processes that may affect the distribution of N on the Erie islands. East Sister Island is only slightly elevated above the lake making it prone to water inundation from lake storms, combined with the thin layer of soil over limestone makes the leaching of nitrogen compounds beyond the extent of the colony possible. Leaching of N compounds, specifically nitrates in soils, is well documented (Schmidt et al 2004). In

addition, volatilized ammonia may provide nitrogen for the plant community outside the colony boundaries; heavy rains and westerly winds in rookeries of Macaroni penguins on Marion Island produced an ammonia shadow around the rookeries, where rich vegetation and peat layers were present (Lindboom 1984). Mizutani and Wada (1988) reported high standard deviations of $\delta^{15}\text{N}$ in herbaceous plants, soils and guano, suggesting that the processes that enrich soils in $\delta^{15}\text{N}$ were locally heterogeneous.

Dense seabird colonies are often responsible for strong enrichment of soil nutrients and organic matter through guano and organic refuse deposition (Wainright et al. 1998). In the case of cormorant colonies on the Lake Erie islands, our results suggest that cormorants have transferred nitrogen and phosphorous to the soil. Soil concentrations of nitrogen were 4-6 times higher at colony sites and phosphorous was up to 2 times higher (Table 3.3). In addition, organic matter was greatly enriched at nesting sites. Our results indicated that the deposition of uric acid in guano led to an increase in soil acidity. This observation is somewhat surprising given the well buffered soil conditions likely arising from the shallow limestone bedrock.

Schmidt et al. (2004) concluded that NH_4^+ was the principal source of N for plants growing at seabird rookeries. Plants vary greatly in their ability to absorb and use ammonium and nitrate as their sources of nitrogen (Haynes and Goh 1978). Experimental observations indicate that some acid-loving species are adapted to, or at least tolerate a nitrogen supply predominantly

consisting of ammonium (Gigon and Rorison 1972). Other work has shown that increases in ammonium concentrations are deleterious to both root and above ground growth (Haynes and Goh 1978). Maynard and Barker (1969) reported that roots of ammonium grown plants were poorly developed, but addition of CaCO_3 was effective in preventing the adverse effects. Specifics of nitrogen nutrition or pH tolerance of *C. occidentalis* or other 'Carolinian' flora was not located. However as noted in Chapter 2, extremely dense growth of American pokeweed (*Phytolacca americana*) and stinging nettles (*Urtica dioica*), both well adapted to high nitrogen and disturbance regimes, are common occurrences in areas of cormorant colonies (*pers. obs.*). Soil enrichment is likely to have favoured the establishment and expansion of ruderal plant species (i.e. plants adapted to high disturbance and low stress levels; Grime 1977), perhaps to the detriment of indigenous taxa. Recent vegetation surveys on Middle Island suggest that many rare herbaceous species known to have existed previously (Kamstra et al. 1995), could not be located in areas of cormorant colonization (Parks Canada, unpubl. data).

Results from studies on Mediterranean islands off the south coast of France where large colonies of yellow-legged gulls (*Larus cachinnans*) have become established over the last 30 years indicate large floristic changes have coincided with seabird colonization (Vidal et al 1998a). The impact on flora and vegetation around nesting sites was severe and possibly irreversible at the largest colonies. Small islands were particularly affected, some

containing more than 50% newly established plant species, while many indigenous species underwent severe decline (Vidal et al 1998b).

Disturbance caused by nesting seabirds can result in major alterations in island vegetation communities. Results from this study suggest that cormorants breeding on the Lake Erie islands are no exception. Our main objective was to evaluate the impact of nesting cormorants on tree physiology and soil chemistry in Carolinian forest on western Lake Erie islands. Results demonstrated that stable isotopes of carbon and nitrogen were useful for studying alterations in nitrogen cycling and detecting physiological stress in individual trees. By using dated tree cores I was able to put recent changes in nutrient dynamics and tree health into a historical perspective. Over time, trees in cormorant colonized areas experienced increased stress and an enrichment in ^{15}N consistent with N inputs from seabird guano. Whereas non-colony trees in relatively close proximity on a relatively small island showed no trend in a physiological stress response. Many traditional tree-ring studies are focused on species and locations at the limits of their ecological range (e.g. latitude, altitude), often examining impacts that have direct and sometimes catastrophic effects (e.g. drought, fire, insects) on tree growth. The value in analyzing stable isotopes in tree rings are: a) the ability to detect subtle environmental changes that may cause 'sub-lethal' stress and that would be difficult to discern using tree ring data alone; and b) the possibility of using species and environments that may be considered sub-optimal for traditional dendrochronology.

Colonial seabirds can exert tremendous pressure on island ecosystems and both direct and indirect impacts on vegetation communities have been documented. My results demonstrate that soil conditions in areas of the forest with cormorant colonies were different from non-colony sites. Colony sites could be considered highly disturbed areas, with increases in soil nitrogen, phosphorus and organic matter. On shallow limestone-derived soils (i.e. highly buffered), decreases in soil pH was indicative of the levels of uric acid that were deposited by cormorants and their magnitude of impact. These results in conjunction with field observations documenting the invasion of nitrophilous vegetation species in these cormorant colonies were consistent with other findings that nesting seabirds affect many ecosystem processes that regulate the composition of vegetation communities. Although the chosen study species (*C. occidentalis*) is considered rare in Canada, it does not appear listed Federally or Provincially. However, a number of species associated with the climax forest ecosystem studied here are protected. The results of our study quantitatively documenting alterations in nutrient dynamics and ecological stress warrant further intensive study on the resilience of native 'desirable' plant species and the recovery potential of these unique island habitats.

Figure 3.1. Master ring width index and the summer Palmer Drought Severity Index. The Master ring width index was generated using tree cores from Middle Island and Middle Sister Island up to 1989 (pre-cormorant).

Figure 3.2. Modelled chronology and the Middle Island chronology. The dotted line represents the modelled tree ring chronology based on regression model with climate factors (PDSI). The solid line is the detrended, standardized chronology from cores on Middle Island ($n=23$). Arrow shows time of earliest breeding by cormorants on the island (1990).

Figure 3.3. $\delta^{13}\text{C}$ values from trees on East Sister Island through time. a) non-colonized trees b) colonized trees. Each symbol and line represents a single tree core. Data from 1985 to 2002 are displayed as 3 year averages.

Figure 3.4. $\delta^{15}\text{N}$ ^{13}C values from trees on East Sister Island through time. a) non-colonized trees b) colonized trees.

Figure 3.5. Stable isotopes in tree rings from Middle Sister Island. a) $\delta^{13}\text{C}$ b) $\delta^{15}\text{N}$. Samples were pooled in 10 year groups for analysis.

Figure 3.6. Simple linear regressions of soil chemistry parameters and the numbers of cormorant nests within a 5m radius of the study tree. These graphs include data from all 3 study islands. A) NH₄⁺ b) NO₃⁻ c) pH.

Table 3.1. Raw tree ring width statistics for the study islands and colony status. Superscripts denote statistically similar groups from Tukey's HSD *post hoc* comparisons.

Site	Series (N)	Mean DBH (cm)	Mean ring-width	Mean Sensitivity	Max. Year Range	Mean Series Length (range)
Middle Sister Island (reference)	22	17.7 ^a	1.66 ^a	0.32 ^a	1881-2002	95.4 ^a
East Sister (reference)	19	15.9 ^a	4.11 ^c	0.37 ^b	1960-2002	31.8 ^c
Middle Island (colony)	23	14.8 ^a	2.32 ^b	0.31 ^a	1875-2002	56.7 ^b
East Sister (colony)	17	16.6 ^a	4.13 ^c	0.40 ^b	1939-2002	37.0 ^c

Table 3.2. Results from simple regressions between $\delta^{13}\text{C}$ or $\delta^{15}\text{N}$ and time. Only r^2 from significant regressions are given.

Site	Tree	$\delta^{13}\text{C}$			$\delta^{15}\text{N}$		
		r^2	p	n	r^2	p	n
Non-colonized	4		0.33	20	0.61	0.04	7
	5		0.26	21		0.10	6
	6		0.19	20		0.09	7
	7		0.19	21		0.14	7
	16		0.59	21		0.35	7
Colonized	1	0.48	0.001	25	0.82	<0.001	12
	6	0.33	0.006	21	0.91	<0.001	8
	10	0.29	0.014	20		0.21	7
	11	0.34	0.006	20	0.55	0.045	7
	16	0.60	<0.001	20		na	na

Table 3.3. Soil chemistry concentrations. Statistically significant differences are denoted by superscripts (Tukey's HSD).

	Island	%OM	pH	NO ₃ uMol/g	NH ₄ ⁺ uMol/g	P uMol/g	nests (5m)
Reference	East Sister	0.02 ^a	6.42 ^b	0.70 ^a	0.62 ^a	4.0 ^a	0.33
	Middle	0.09 ^a	6.31 ^a	1.66 ^a	1.03 ^a	4.45 ^a	0.0
	Sister						
Cormorant	East Sister	0.37 ^a	5.85 ^a	3.48 ^b	6.22 ^b	3.76 ^a	6.7
	Middle	1.09 ^b	6.45 ^b	4.21 ^b	1.70 ^a	7.98 ^b	2.6
	Island						

Chapter 4

General Discussion and Conclusions

Islands often support large seabird colonies because they provide birds with the tranquility necessary to accomplish their nesting cycle (Jouventin and Mougin 1981). However, nesting birds affect their breeding habitat through physical disturbance (collection of nest materials; Sobey and Kenworthy 1979) and chemical disturbance (guano enrichment; Weseloh and Brown 1971). Many studies have demonstrated direct and indirect impacts of nesting seabirds on herbaceous vegetation on marine islands off Australia (Rippey et al. 2002), New Zealand (Markwell and Daugherty 2003), France (Vidal et al. 1998a), in the Gulf of California (Anderson and Polis 1999) as well as in freshwater systems including inland lakes (Weseloh and Brown 1971) and the Laurentian Great Lakes (Hogg and Morton 1983). A few studies have demonstrated qualitative impacts of nesting birds on woody species, however most of these are limited to shrub species (Lemmon et al. 1994). In western Lake Erie, cormorants nest on islands in large deciduous trees that represent climax species in some of the rarest vegetation associations in Canada.

My objective was to document the impacts that may be associated with double-crested cormorants nesting on these western Lake Erie islands. In Chapter 2, I used digitized airphotos and image analysis processing to quantitatively demonstrate that cormorant nesting colonies have led to substantial forest declines on these islands. In Chapter 3, I demonstrated the effect of nesting on individual trees using stable isotopes of carbon and nitrogen in tree growth rings. Results suggested an increase in physiological stress ($\delta^{13}\text{C}$) in common hackberry trees (*C. occidentalis*) situated in cormorant nesting areas.

Nutrient inputs (nitrogen, phosphorus and micronutrients) and their cycling are important processes regulating productivity in ecosystems. Nitrogen is frequently a limiting nutrient in terrestrial northern ecosystems (Chapin et al 1986). Principal sources of nitrogen are from atmospheric deposition and N-fixing plants (Van Cleve and Alexander 1981). However, in some terrestrial systems, most notably island and riparian ecosystems, organic nutrients derived from aquatic sources may constitute important inputs to the terrestrial domain. Transport can occur via physical (wind, water) or biotic factors.

The biogenic movement of carbon and nutrients by mobile consumers is an important ecosystem-level process producing substantial and rapid redistribution across habitat boundaries (Kitchell et al 1979). Among the best characterized process is the bear-salmon interaction in coastal regions of western North America (see Reimchen 2000). Not only does this represent a

major source of protein for bears (Hildebrand et al 1996), but this annual pulse of nutrients can represent a major source of nitrogen for riparian vegetation (Ben-David et al 1998, Hilderbrand et al 1999).

Seabirds and pinnepeds feeding on fish also transport large quantities of nutrients and organic matter, including guano, undigested food (pellets), eggs, feathers and bodies of dead individuals, to terrestrial ecosystems. In some cases, entire food webs on many islands and coastal areas depend on allochthonous inputs from birds (e.g. Hutchinson 1950, Polis and Hurd 1996a,b).

In most environments the natural abundance of ^{15}N is not a suitable tracer to detect plant N sources (Robinson 2001). However, systems with high N inputs with a distinct $\delta^{15}\text{N}$ value are well suited to ^{15}N tracer studies. High values of $\delta^{15}\text{N}$ are the result of enrichment along the food chain due to isotope fractionation (Kelly 2000). The position of seabirds at the top of food webs makes $\delta^{15}\text{N}$ an ideal tracer in colony soils and vegetation where the dominant N input may be ornithogenic.

Annual growth rings combined with reasonable longevity make trees possible long-term sentinels of nitrogen cycle disturbance in forested ecosystems. Improved techniques to examine stable isotopes in tree rings, particularly of carbon and nitrogen, provide a method to examine stressors on ecosystems. For example, stable N isotopes can provide insights into subtle changes in biogeochemical cycling that may not have been evident using

traditional growth ring analysis. The avenue could be particularly useful in areas where suboptimal conditions for dendrochronological analysis may exist.

As mentioned in Chapter 1, there are three main aspects associated with the increase in cormorant populations that conflict with human values: a) impacts on fisheries b) impacts on other vertebrate species and c) vegetation destruction. Data reviewing a) and b) are cited in Chapter 1 and are beyond the scope of this paper. However, relatively few studies exist on the vegetation effects of cormorant nesting, and most of them consist of qualitative field observations. The results reported here are the first to quantify the forest declines and to demonstrate the physiological stress in trees related to cormorant nesting. Two of the three islands studied here are protected areas. Middle Island is managed by Parks Canada (Point Pelee National Park) and East Sister Island is an Ontario Ministry of Natural Resources Provincial Nature Reserve (Wheatley office). Scientifically sound park management plans, including cormorant management, require objective studies with reproducible data for creditability.

There is a popular outcry for the management of over-abundant cormorants in eastern North America, with commercial and sport fishing industry groups (<<http://www.ofah.org/cormorants>>) considering them a pest. In the case of vegetation impacts, those vested with management authority have been lacking precise studies to corroborate or refute claims. In fact, some groups (Peaceful Parks Coalition: <http://www.peacefulparks.org/ppc/>

vegetativeimpacts_corm.htm) suggest that premature killing of trees by guano is a naturally occurring process, likening it to swamp succession caused by beavers and wildfire for rejuvenating forests. Dendrochronological techniques similar to those used here have been extremely successful in documenting the important role of a natural disturbance like fire in forest ecology (see Swetnam and Betancourt 1998). My data from Middle Sister Island suggest that cormorants have not been a part of western Lake Erie island ecosystems for the last 120 years. Impacts such as those currently observed on other western basin islands which I have linked to cormorant nest presence are not observed in the tree ring chemical record back to the late 1800s. Death by guano as an integral stage in natural succession is thus untenable.

So the question remains - why do cormorants in particular seem so destructive? Islands in the Great Lakes have hosted other large seabird colonies (gulls, terns, herons and egrets) that have co-existed over the longer term with native vegetation communities. My data from Middle Sister Island indicating a $\delta^{15}\text{N}$ value of 12-13‰ suggests that allochthonous nutrient inputs are important, possibly from seabird colonies. One hypothesis is that the guano from cormorants is different, perhaps more acidic or more nutrient rich. Unfortunately, there is no literature to test this. Another hypothesis stems from the high nest densities observed on cormorant colonies. On Scotch Bonnet Island in eastern Lake Ontario a large herring gull (*Larus argentus*) colony has existed for a long period in conjunction with healthy herbaceous

vegetation. However, recently cormorants displaced part of the gull colony and within a few years, the area was devoid of vegetation (G. Fox pers. comm.).

I suggest the vegetation destruction may be due to another aspect of cormorant life-history, namely that they may deposit the majority of their guano on the nesting colony, whereas other bird species do not. Hutchinson (1950) outlined particular requirements, both biological and geophysical, that must be satisfied for colonial seabirds to produce guano deposits or guano 'islands'. He noted that not all seabirds produce significant deposits of guano; that the largest deposits were produced by a relatively few species in the Pelicaniformes, namely certain pelicans, boobies and cormorants (Hutchinson 1950). These birds appear especially prone to deposit the majority of their droppings on the breeding colony. Evidence suggests that this is not accidental, but rather a feature of the behaviour of the bird, and that the guano material is used during the breeding season for nest material. It is possible that a tendency for the development of such behaviour underlies the fact that Pelicaniformes rather than gulls and terns are the predominant guano birds (Hutchinson 1950). This may explain why large colonies of other birds such as those found on western Lake Erie islands (terns, gulls, herons and egrets) may co-exist with island vegetation, whereas cormorants do not.

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