Monitoring American White Pelicans and Double-crested Cormorants in Minnesota: Assessing Status of Populations and Exploring New Survey Methodologies

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Before I recognize the people who have helped me on this challenging journey, I will tell a story about a quite fortuitous experience that, unbeknownst to me at the time, shaped the path for opportunities and relationships that would define the next several years of my life and many more into the future. During my last year as an undergraduate at the University of Minnesota, I learned about a field course in Thailand that was offered over winter break. Several friends had talked it up and were eagerly applying for a spot in the class. Traveling overseas to help with tiger and fishing cat research sounded amazing but I could not afford the class, let alone the flight overseas. After several weeks of back and forth with my overly cautious voice of reason, I decided the experience was worth the risk and enrolled. Somehow I managed to pull the funds together, and in December, I was on a plane to Thailand.

While there, my classmates and I radio-tracked fishing cats along the coast, captured amazing wildlife photos with camera traps, estimated tiger prey abundance using scat surveys, and modeled tiger home-ranges using data from GPS collars. We ate terrific food, visited beautiful temples, and had great conversations with local people including the impressive field crews at Khao Sam Roi Yot National Park and Huai Kha Khaeng Wildlife Sanctuary. The Thailand course was an incredible experience that reaffirmed my passion for the conservation field and for connecting with people. I describe this experience because if I had not taken the leap and enrolled in the course, my life would not have turned out the way it did. I would not have made friends with two women I now consider family. I would not have gotten to know my advisor and her family, and therefore this document would likely not exist. Lastly, I would not have built a friendship with the woman who is now my wife. That simple act of enrolling in a course was a decision that changed my life. I am thankful every day that I did not let fear keep me from taking a chance and I encourage people reading this to contemplate which is more terrifying: failure and potential financial loss or regretting what could have been?

With this introduction, I would like to thank my advisor Francesca Cuthbert who has been a long-time friend and colleague, and to whom I owe a great deal. She has shared her immense knowledge of birds and given me countless opportunities to explore my own abilities both in the field and while conducting research. She has often let me struggle through on my own, trusting me to accomplish what I promised, but has always been there to guide me when I got in over my head. As often as she has offered professional direction, she has also shared in friendly conversation. She has time and again gone out of her way to make sure I am financially, professionally, and personally supported leaving me free to pursue my passions. I am eternally grateful for this support.

Next, I would like to thank my committee members, Linda Wires and John Fieberg. As my supervisor during the 2010 waterbird census, Linda taught me everything she knew about censusing waterbirds in the field and using aerial photos. Her passion for these birds and the work we have done is contagious and has really endeared me to these species for which I had little experience prior to working on these projects. I am also deeply indebted to John. I first enrolled in one of John Fieberg's courses a little more than a year ago and he has helped me to demystify statistics (at least to the degree statistics can be). Despite being incredibly busy, John has always been willing to meet or send a late night email to get me unstuck and moving in the right direction.

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Deserving of special mention, I would like to express my sincerest gratitude to DNR pilots Jason Jensen and Bob Geving whose supreme flying allowed me to take great photos for our studies. I really enjoyed getting to know them during our 20+ hours of flight time and my wife really appreciates that they brought me back home safely.

Chapter 3 of this thesis would not have been possible without encouragement and guidance from my advanced remote sensing professor, Jennifer Corcoran. She helped me to take a simple idea of using satellite imagery to count pelicans and develop it into the project presented in my final chapter. I would also like to thank my peers in Jennifer's class whose prior experience and willingness to share suggestions and criticisms helped to flesh out some of the finer details of the study. I am also grateful to Jeff DiMatteo, a fellow pelican researcher based out of North Dakota State University. Jeff has studied pelicans for many years and always offered generous insight and advice related to the Marsh Lake colony and pelicans in general. Finally, I thank Gary Nohrenberg with USDA APHIS for providing information about timing, location, and scale of cormorant control efforts.

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Finally, I express gratitude to my family and friends I have neglected over the last year and a half. Thank you for your support and understanding along this journey. I look forward to catching up with all of you very soon and sharing some fancy drinks, delicious food, and hearty laughs.

DEDICATION

To my mother, Jeanna Hamilton, who taught me that doing what you love is never easy but well worth the work, and to my wife, Annie Davies Hamilton, who kept me on task while gracefully enduring my stress induced mood swings.

ABSTRACT

North American populations of American White Pelicans (*Pelecanus erythrorhynchos*) and Double-crested Cormorants (*Phalacrocorax auritus*), at one time depleted by human actions, have increased in abundance and distribution over the last 40 years. These changes resulted in a greater overlap of resource use between these bird species and humans (e.g., consumption of fish, use of islands), and managers must now balance conservation with public interests. My research informs management decisions by improving understanding of the status of these species, specifically of populations nesting in Minnesota, and by highlighting resources that can reduce cost and increase frequency of monitoring efforts. Nesting censuses conducted in Minnesota during the summer of 2015 found cormorant and pelican populations (15,421 pairs and 16,406 pairs respectively) to be comparable to previous reports. This suggests populations in the state have reached carrying capacity, at least at present, but vacation of cormorant depredation orders in 2016 may impact future populations of one or both species. In July of 2015, surveys of pelican fledglings were conducted in Minnesota as a follow up to a report released by the Minnesota Department of Natural Resources that found contaminants from the 2010 Deepwater Horizon Oil Spill in pelican eggs. We found no evidence that fledging rates observed during our survey (avg. 0.54) were below expected ranges, but cannot rule out long-term effects of contaminant exposure. Inspired by the high costs of aerial photography and manual counting employed during our studies, I also investigated alternative sources of colony imagery and computer aided analysis techniques. I found that images acquired from sources such as DigitalGlobe and Google Satellite can be used to estimate abundance of pelicans and potentially cormorants at select colony locations. I also showed that object-based image analysis can provide an estimate of pelican abundance quickly and with accuracy comparable to manual counts. Although not feasible to use at all locations, satellite imagery analysis has great potential as an inexpensive method to regularly track bird numbers at selected colony sites in Minnesota and elsewhere if images are available and meet analysis requirements.

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PREFACE

During the summer of 2015, I had the good fortune of coordinating the third statewide Minnesota colonial waterbird census which was conducted with the primary goal of monitoring Minnesota's nesting populations of Double-crested Cormorants (*Phalacrocorax auritus*, here on "cormorants" or DCCO) and American White Pelicans (*Pelecanus erythrorhynchos*, here on "pelicans" or AWPE). The 2015 survey followed a series of earlier efforts to estimate numbers of breeding cormorants and pelicans in the State. In the early 1960s, the Minnesota Department of Natural Resources (DNR) began to record data (Natural Heritage Information System) on nesting colonial waterbirds at selected individual colony sites and more recently paired with University of Minnesota researchers to formalized these records to include statewide surveys of cormorants and pelicans with the primary goal of tracking population trends (Wires et al. 2006, 2011; Hamilton and Cuthbert 2016). Despite extirpation of pelicans from Minnesota during the late 1800s, near extirpation of cormorants in the early 1900s (Roberts 1932), and sharp declines of nationwide populations of both species between the mid 1940's and the 1970's due in part to organochloride pesticide use (Anderson and Hickey 1995; Weseloh and Collier 1995; Keith 2005), nesting populations of cormorants and pelicans re-established in Minnesota and have experienced substantial growth in the last 46 years.

In Minnesota, as well as in other parts of the country, recolonization and rapid growth of cormorant and pelican populations has stirred controversy primarily due to concern that waterbodies used for sport and commercial fishing will not be able to maintain desirable levels of recreational and economic output while supporting large colonies of piscivorous birds. Conflict and controversy have been further fueled by government sanctioned population control, pressure from legislators, protests by various lobbyists, cases of illegal culling, and several federal court decisions.

Cormorants have been federally protected under the Migratory Bird Treaty Act (MBTA) since 1972, and prior to the late 1990's, individuals and agencies seeking to control cormorants, were required to apply for a permit from the U.S. Fish and Wildlife Service (USFWS). In 1998, due to growing concern for losses sustained by aquaculture operations, the Aquaculture Depredation Order (AQDO) was issued. This order allowed aquaculturalists to kill cormorants feeding at fish production areas. In 2003, the USFWS expanded the AQDO to include killing at roosting areas in proximity to catfish farms and also established the Public Resource Depredation Order (PRDO, 50 CFR 21.48) for cormorants. This order authorized "State fish and wildlife agencies, federally recognized tribes, and state directors of the Wildlife Services program of the U.S. Department of Agriculture Animal and Plant Health Inspection Service (USDA/APHIS) to prevent depredations on the public resources of fish (including hatchery stock at federal, state, and tribal facilities), wildlife, plants, and their habitats by taking without a permit double-crested cormorants found committing or about to commit, such depredations." Culling authorized by these orders and under depredation permits resulted in the take of 506,318 adult cormorants between 1999 and 2012, averaging roughly 43,000 cormorants killed per year once the depredation orders were in place (U.S. Fish and Wildlife Service 2014).

Concerns about the magnitude of culling operations and limited scientific basis for controlling cormorants were expressed by various individuals and interest groups since the inception of these programs (Wires 2014, 2015). In March 2016, a federal court case that was filed against the USFWS by Public Employees for Environmental Responsibility (PEER) found extension of the depredation orders to be in violation of the National Environmental Policy Act (NEPA) and in May 2016 ordered both orders be vacated (Public Employees for Environmental Responsibility vs. United States Fish and Wildlife Service 2016).

Compared to cormorants, pelican populations have received much less opposition from anglers and private land owners, but examples of conflict do exist. In 2011, a farmer in southern Minnesota became enraged in response to expansion of a nearby pelican colony onto the periphery of his cropland and destroyed the majority of eggs and young present in the approximately 1,458 nests along the shore (Wires 2013). Found guilty of violating the MBTA, he was fined $12,500 and sentenced to 100 hours of community service and two years of probation (Marcotty 2012). Pelicans traveling along the Pacific Flyway are managed using a combination of non-lethal and lethal measures as outlined by the Pacific Flyway Council (2012). In Idaho specifically, concern about pelicans predating trout and other game fish in rivers and hatcheries prompted Idaho Fish and

Game to begin hazing nesting pelicans in 2006. Since then, in an effort to maintain a statewide pelican population of 2,800 breeding individuals, various management methods have been adapted including culling of nesting adults and oiling of eggs (Idaho Department of Fish and Game 2016). Although pelican control efforts are currently limited in scale as compared to cormorant control, pelicans are likely to face more resistance as their populations continue to expand and grow.

As with many controversial species, management of conflicts around cormorants and pelicans is extremely complex. In addressing these issues, agencies must consider the desires of the public and the welfare of other species while also working within the confines of government procedure with limited financial and personnel resources. In anticipation of requests for cormorant control under the newly established depredation orders, the Minnesota Department of Natural Resources (MNDNR) conducted the first statewide census to obtain baseline information on the abundance and distribution of both species in 2004/05. As cormorant control got underway in the state, the census was repeated in 2010 in an effort to track distribution, abundance and trends. To build on these monitoring efforts and provide a current foundation upon which the need for and effect of management actions could be assessed, I conducted the third statewide census in Minnesota in 2015. This monitoring work also provided the basis for my thesis. By providing critical information on where these species were located, their abundance locally and statewide, and population trends through 2015, this thesis enables agencies to identify local communities where conflict may arise as well as areas that may need protection to ensure available nesting habitat for both species. A report containing the results of the 2015 census was submitted to the DNR in spring 2016 (Hamilton and Cuthbert 2016). The first chapter of this thesis describes cormorant and pelican distribution and abundance in Minnesota in 2015 and analyzes nesting population trends observed between 2004 and 2015.

The research presented in Chapter 2 was proposed following a study conducted by the MNDNR that found traces of chemicals associated with the 2010 Deepwater Horizon Oil Spill in pelican eggs collected from several Minnesota nesting colonies (Minnesota Department of Natural Resources 2014). These chemicals have been shown to be harmful to birds at multiple life stages (Leighton 1993, Paruk et al. 2016, Albers 1979, and Wooten et al. 2011) and there was concern that presence of these chemicals may lead to reduced fledging success in pelicans. We conducted a survey of select pelican colonies in Minnesota during July of 2015 to determine rates of fledging success. We then compared fledging rates observed in 2015 to rates observed in similar studies to assess whether fledging success was within expected ranges. The results of this study presented in Chapter 2 were derived from the report prepared for the Legislative-Citizen Commission for Minnesota's Resources (LCCMR) who funded the study (Hamilton and Cuthbert 2016).

Aerial surveys employed in Chapter 1 and Chapter 2 are resource intensive due to high flight costs and time consuming manual analysis of aerial photos. The study presented in Chapter 3 was conducted in an effort to identify resources that could reduce the cost of image acquisition and analysis. I first investigated alternative sources of satellite and aerial imagery that could be used to monitor populations of pelicans and potentially other colonial waterbird species. Then I explored the possibility of using computer aided analysis, specifically object-based image analysis, to expedite counting of pelicans from photos. These resources, which have primarily been used in conservation to monitor land use and develop forest inventories, have great potential to contribute to wildlife monitoring projects as well.

CHAPTER 1

DISTRIBUTION, ABUNDANCE AND POPULATION CHANGE IN THE AMERICAN WHITE PELICAN (*PELECANUS ERYTHRORYNCHOS*) AND DOUBLE-CRESTED CORMORANT (*PHALACROCORAX AURITUS*) IN MINNESOTA: 2004-2015

Double-crested Cormorants (*Phalacrocorax auritus*, “cormorants” or DCCO) and American White Pelicans (*Pelecanus erythrorhynchos*, “pelicans” or AWPE) are two of nine colonial nesting waterbirds that breed in Minnesota (Wires et al. 2011, Hamilton and Cuthbert 2016). Following overexploitation and or persecution during the 1800s, nesting populations of pelicans were considered extirpated from Minnesota and those of cormorants were greatly diminished (Roberts 1932). In the mid-twentieth century, organochloride pesticides such as DDT and Endrin, and industrial chemicals such as polychlorinated bisphenols (PCB’s) caused further declines in already stressed populations (Wires 2014). In the early 1970s, the addition cormorants and pelicans to the Migratory Bird Treaty Act protected bird list, combined with the creation of the Environmental Protection Agency and subsequent banning of DDT, PCB’s, and other toxic chemicals, as well as the development of various state and other federal protections provided conditions for cormorants and pelicans to recover on a continental scale. Recovery may also have been facilitated by other developments that occurred as these birds became protected, such as the enormous growth of catfish, baitfish, and crayfish aquaculture operations in the southern United States, the creation of thousands of new reservoirs, extensive employment of recreational fish stocking programs, and introductions of non-native fish species such as alewife (*Alosa pseudoharengus*) and round goby (*Neogobius melanostomus*). These developments created prey resources in areas where native fish were scarce, had been depleted or had never existed (Wires 2014). Today, nesting populations in Minnesota number in the thousands and range across much of the state (Wires et al. 2011, Hamilton and Cuthbert 2016).

In addition to population increases and potential or perceived negative impacts to people, several other diverse issues related to these species provide justification for tracking their population trends in MN, including:

1) Cormorants are managed at a number of sites, and monitoring the statewide population is needed to evaluate how control is affecting Minnesota’s cormorant population.
2) Pelicans are a State-listed Special Concern Species and are also classified as a species of greatest conservation need (SGCN) in Minnesota’s State Wildlife

Action Plan (Minnesota Department of Natural Resources 2006), along with the Common Tern (*Sterna hirundo*, State Threatened) and Black-crowned Night-Heron (*Nycticorax nycticorax*). Monitoring long-term changes in SGCN populations and habitats is a goal of Minnesota's State Wildlife Action Plan, and identifying and addressing the effects that emerging issues (e.g. impact of cormorant ecology and management efforts) may have on populations of SGCN species, is one of the Plan's objectives (Minnesota Department of Natural Resources 2006). Potential for disturbance to all three of these SGCN species, along with other co-nesting colonial waterbirds (e.g., herons, egrets, gulls) at cormorant colonies, is an important concern because cormorant management occurs at several diverse locations and can potentially disturb these other species. Additionally, cormorant nesting activity can change habitat for tree-nesting birds.

3) On 20 April 2010, the Deepwater Horizon oil rig explosion occurred in the Gulf of Mexico, resulting in the largest oil spill in history in U.S. waters. Sexually immature and overwintering pelicans occupy areas affected by the oil spill, leading to concern that this disaster might impact pelicans.
4) Cormorants and pelicans are sensitive to environmental contaminants (Evans and Knopf 1993, Hatch and Weseloh 1999).
5) Both cormorants and pelicans have been targets of illegal control in Minnesota and elsewhere.

The Minnesota Department of Natural Resources (DNR) has conducted surveys of cormorant and pelican colonies for nearly 50 years (Natural Heritage Information System). In 2004 and 2005 the DNR contracted researchers at the University of Minnesota--Twin Cities to conduct the first statewide census of Minnesota's nesting cormorants and pelicans (Wires et al. 2006). Providing baseline data for future monitoring efforts, this census compiled previous records into a comprehensive list of known and potential nesting locations and established census methodology based on similar waterbird census efforts (Cuthbert and Wires 2011).

In 2010, a second statewide census of both species was conducted using methods similar to the first census. This effort found distribution and abundance to be comparable to observations made during the 2004-2005 census (Wires et al. 2011). That same year, the Deepwater Horizon oil rig exploded in the Gulf of Mexico resulting in one of the largest oil spills to ever occur in US waters. Areas impacted by the oil spill and subsequent oil dispersal efforts are occupied by non-migrating sexually immature pelicans during summer months and support overwintering pelicans later in the year. In response to potential exposure to spill contaminants, the Minnesota DNR and University of Minnesota researchers conducted statewide pelican censuses in 2011 and 2012 to monitor potential impacts to pelican populations following the spill. The 2011 and 2012 censuses reported much higher nesting populations than observed in the 2004-2005 and 2010 census efforts (Wires et al. 2013). This may be due in part to the reciprocal movement between Minnesota colonies and colonies in adjacent states observed by Sovada et al. (2013).

During the summer of 2015, I organized and conducted a third statewide census of cormorants and pelicans (Hamilton and Cuthbert 2016) and identified the following objectives for this thesis chapter: First, obtain location information for all active cormorant and pelican colonies in 2015. Then, estimate the number of cormorant and pelican nests at all colonies active in 2015. Finally, use the data obtained to analyze and evaluate changes in distribution and numbers of cormorants and pelicans between the three survey periods (2004/05, 2010 and 2015 censuses). I also sought to evaluate stochastic growth models as a tool for projecting future populations of nesting waterbirds using data collected during this and previous census efforts.

METHODS

Study Area

Cormorants and pelicans prefer to nest on islands that isolate them from mainland predators and are infrequently visited by humans (Wires and Cuthbert 2010). A large number of colony sites (161) were identified during previous census efforts in Minnesota (Wires et al. 2006 and 2011); a subset of these was selected for census in 2015. This subset consisted of sites active in either 2004-05 or 2010 (or 2011/2012 in the case of pelicans) as well as sites with long-term records of active waterbird colonies as determined from a review of DNR records (Minnesota Department of Natural Resources, Natural Heritage Information System). We also surveyed a small sample of sites that had been reported previously and were determined to be inactive. Additional sites were reported by DNR staff or discovered incidentally while in the field.

Site Surveys

Protocol for census methods followed guidelines developed by Cuthbert and Wires (2011) for the Great Lakes Colonial Waterbird Survey. See this document for additional details. We surveyed the majority of sites using aerial photography to avoid disturbance to nesting birds during census efforts. I took photos from a fixed wing aircraft (either a Cessna 185 with floats or an American Champion Scout) flying at an altitude of approximately 500 feet and a flight speed of 80-115 knots (J. Jensen, personal communication, April 14, 2016). The camera used was a Nikon D200 10.20MP digital SLR camera fitted with an AF-S DX NIKKOR 18-200mm f/3.5-5.6G ED VR II lens.

When present, a dense tree canopy can obstruct views and cast shadows that make it difficult to identify nests from aerial surveys. I determined the need for ground surveys subjectively based on previous experience, expert advice, and/or by referring to satellite images available via Google Earth. Field crews used kayaks, a 17 ft. Lund boat with 40 hp motor, or an 18 ft. Zodiac with 90 hp motor to access these sites. Once at the sites, 2-4-person field teams conducted ground surveys as quickly and quietly as possible to reduce disturbance to nesting birds. DNR personnel contributing to this study also used ground counts to gather data as part of regular monitoring efforts at sites within their districts.

Field crews surveyed a small number of sites from the roadside or shoreline using 8 x 10 binoculars or spotting scopes. The majority of these were colonies suspected to be inactive and were surveyed on days when inclement weather prevented aerial or within-colony ground surveys (fair weather was required to fly; exposure to cold/rain/heat as a result of human presence can endanger chicks that cannot yet thermoregulate).

Nest Estimates

We estimated nesting population size from aerial photographs by manually counting nests with the aid of mapping software (either ArcMap 10 or QGIS v.2.12.1). After loading the digital photos into the software, we created either line-vector or polygon-vector shapefiles that were used to identify overlap between sequential photos. We assessed overlap visually by comparing sequential images side by side and placing line or polygon features to highlight areas counted in other photos. We then created point-vector shapefiles for each species and observed nesting status (Nesting, Not-Nesting, Unknown). The unknown category was created during the 2004/05 census (Wires et al. 2006) to allow for cases where the observer was unsure whether a given bird is nesting or loafing or when there is concern that the object in question may not be a bird at all (e.g., rocks, shadows). We assessed breeding status of each bird by visual inspection and placed the appropriate point feature on each bird and/or nest observed. We then determined the number of points for each point-vector shapefile by referring to the associated attribute table.

Field crews conduced all ground counts using handheld tally counters and followed protocol established by the Great Lakes Colonial Waterbird Survey (Cuthbert and Wires 2011). Surveyors avoided duplication of effort by marking trees containing nests with flagging tape (later removed) or marking ground nests with biodegradable marking spray-paint. We applied paint using paint construction wands (found at home improvement stores such as Home Depot).

Detectability of Nests

Detectability rates for counts based on ground counts have not been assessed for the statewide colonial waterbird censuses conducted in Minnesota, however field work utilizing a double-observer approach to estimate detection probability and abundance (Nichols et al. 2000) for five ground-nesting species during ground counts was conducted in the Great Lakes (Cuthbert and Wires 2007). This work demonstrated that marking nests greatly increases estimate accuracy at sites with > 25 nests by helping crews to keep track of already counted nests while ensuring that all nests have been accounted for. This study further shows that when nests were marked, detection probability was on average high (95% for single observers). Of the five species considered in the study, observers had the highest detection of ground-nesting cormorants; on average, 98% of their nests were detected in a sample by one observer. This work also compared counts between observers and concluded that trained observers were equal in their ability to detect nests; the observer sample size was small (n = 2), however. Although no information was available on detectability rates for tree nesting birds, marking trees and counting the number of nests in a tree is known to increase estimate accuracy (D.V. Weseloh, personal communication, April 15, 2016).

We are unaware of any studies that have estimated detection probabilities for nests estimated based on aerial photographs, nor any that have measured differences in observer skill in estimating birds using this method. However, measurement error for estimates of cormorants at 15 sites in the Great Lakes and MN obtained through paired ground counts and counts from aerial photographs was very low, 0.5% of total variation between colonies considered (L.R. Wires and F.J. Cuthbert, unpubl. data). Additionally, for the 15 sites considered, aerial estimates were on average within 9% of estimates based on ground counts, indicating that estimates from high quality aerial photographs are representative of numbers on the ground. Although similar comparisons were not made for pelicans, this species is large and conspicuous, and therefore aerial estimates for these birds are assumed to even more closely predict numbers on the ground. More work needs to be done to assess the accuracy of aerial census methods for tree-nesting cormorants, but preliminary data from the Great Lakes indicate that high quality photos can produce similar results as ground counts, and in some cases, may provide more accurate counts (Cuthbert and Wires 2011). For example, birds on nests are sometimes more visible when viewed from the top of the canopy than from below, and because birds typically remain on nests during aerial photography, it is often much easier to positively identify nest ownership in mixed-species colonies.

Census Timing

When scheduling survey efforts we considered several factors including sensitivity to disturbance (increased likelihood of pelican abandonment if disturbed during courting and nest initiation (Evans and Knopf 1993)), precipitation (exposure to cool rains can lead to chick mortality (Sovada et al. 2014)), air temperature (chicks cannot thermoregulate for the first 2 weeks after hatching (Evans 1984, Hatch and Weseloh 1999) and sun intensity (uncovered chicks can die in less than 11 min on warm sunny days (Hatch and Weseloh 1999)). For these reasons, suggested temperature range during surveys is 19-29ºC (Linda Wires, email communication, 12 May 2015).

We also had to take into account that peak nest numbers are not reached until a few weeks after nest initiation because cormorants and pelicans nest asynchronously (Evans and Knopf 1993, Hatch and Weseloh 1999)). To adjust for this, we considered timing of previous census efforts and contacted DNR staff to discuss colony establishment timing observed in recent years. We conducted a flight on April 29 to assess nesting status on that date. We estimated the optimal survey period to be between May 15 and June 15, although photos from the April flight were used for sites where it was believed nesting had reached peak density at the time these photos were taken.

Nest initiation begins first in southern Minnesota and progresses northward as spring moves into summer. We concentrated initial survey effort in southern Minnesota and moved northward as the survey period proceeded.

For sites surveyed by aerial photography, census timing was constrained by weather conditions and plane availability. Inclement weather results in unsafe flying conditions and can negatively affect image quality. As such, we limited aerial surveys to days with mostly clear skies and light winds.

Statistical Analysis

In the 2015 census summary report prepared for the DNR (Hamilton and Cuthbert 2016), we evaluated population trends using percent change calculations based on the following formula where x_2 is the new (2015) nest estimate and x_1 is a previous nest estimate to which the newer estimate is being compared.

$$Percent\,Change = \frac{x_2 - x_1}{x_1}$$

After submission of the DNR report, I explored stochastic growth models as a means to predict future nesting populations based on waterbird census data. I only considered data from pelican colonies that were active during all 5 census periods (2004,10,11,12,15) to maximize the number of years contributing to calculations. Using Bayesian methods outlined by Mech and Fieberg (2015), I fit two stochastic population models, a density-independent model and a Ricker (density dependent) model:

$$Density\ Independent\colon N_{t+1} = N_t \exp(a + \varepsilon_t)\,;\varepsilon_t \sim N(0, \sigma_p^2)$$

$$Ricker(Den.Dep.)\colon N_{t+1} = N_t \exp(a + bN_t + \varepsilon_t)\,;\varepsilon_t \sim N(0, \sigma_p^2)$$

For each model, I considered three options for observation error (No Observation Error, Poisson Distributed Observation error, and Log-Normal Distributed Observation Error), yielding a total of six possible models. In these models, 'a' signifies population growth for small populations, 'b' describes the strength of density dependence, and 'ε' represents process variation (i.e., variation in the true log population size), assumed to be normally distributed with mean 0 and variance σ_p^2.

I adopted code utilized by Mech and Fieberg (2015) to fit models were using statistical programs R (Version 3.3.1) and JAGS (Version 4.2.0) in concert with the R2Jags package, which allowed communication between R and Jags. I used Markov chain Monte Carlo (MCMC) simulations to generate samples from the posterior distribution of model parameters. I ran 3 independent chains for 20000 iterations following an initial burn-in of 5000 iterations. I used the Gelman-Rubin statistic for each model to check for convergence (values close to1 suggest convergence). I also visually inspected posterior distributions and trace plots to assess convergence and to evaluate whether posteriors had moved away from the priors (i.e., whether Bayesian "learning" had occurred).

I then ranked the models based on Deviance Information Criterion (DIC). Similar to Akaike's Information Criterion (AIC), a low DIC value is preferred and suggests a better fit of the data relative to other models.

For projections, I compiled nest data for Marsh Lake from the 5 statewide census efforts as well as from DiMatteo et al. (2015) resulting in 10 nest estimates between 2003 and 2015. Using the best performing model(s) from my initial calculations, I ran MCMC simulations to generate posterior distributions of model parameters for Marsh Lake. I then projected the population forward by drawing values from those posterior distributions. First, I attained 5,000 sets of parameter values ('a' and sigma for the density independent model; 'a', 'b' and sigma for the density dependent model) by randomly sampling individual iterations from the 60,000 MCMC iterations (3 chains x 20000 iterations). For each set of parameter estimates, I calculated a nesting population projection by starting from the nesting population in 2015 (10,289) and progressing forward for 10 time steps. For each time step, I sampled epsilon terms from N(0, σ_p^2). Given that 10 years of data over a 12-year period were used to estimate growth parameters, a single time step is roughly equivalent to 1 year.

RESULTS

Potential Nest Sites and Site Visits

We considered all spatially unique land masses with habitat conditions likely to attract breeding birds to be individual sites, even when islands were near one another. Since the first statewide census, 171 sites have been identified as potential breeding locations for cormorants and pelicans. 146 were identified in the 2004 census, 18 were added during the 2010 census, and 7 more were added during the 2015 census. In 2010, a few sites were determined to be outside the study area (i.e. located outside Minnesota, n=3) or to no longer exist (n=1), reducing the total potential sites to 167. During the 2010 census, 58 sites were assumed inactive based on factors that made breeding at these sites unlikely (e.g., habitat change, time since active nests were last observed, reports from professionals familiar with current conditions at these sites). In 2015, we assumed 79 sites to be inactive for similar reasons. We visited the remaining 88 sites between 29 April and 15 June 2015 to determine their status.

Active Sites, Species Composition and Nest Estimates

Of the 88 sites visited, a total of 40 had nesting cormorants (n=36), pelicans (n=15) or both (n=11). We estimated a total of 16,406 pelican nests and 15,421cormorant nests (See Table 1.1 for site list and nest totals, See Figure 1.1 for map of active sites).

Re-visits to Subset of Sites Inactive in Previous Census Efforts

In 2015, the census team revisited 31 sites that were inactive in one (n=19) or both (n=12) of the previous censuses to determine if inactive sites remained inactive. Of these 31, only 7 (23%) supported nesting colonial waterbirds in 2015 (Table 1.1). Of those 7, only 4 supported cormorants and/or pelicans (Gull Island, Goose Lake, Lake Benton, and Norway Lake).

Changes in Distribution and Abundance of the American White Pelican[1]

We found abundance of nesting pelicans in 2015 to be substantially lower than observed during the 2011 and 2012 (n=22,506 and n=22,023 respectively; Table1.1) statewide pelican censuses (Wires et al. 2013). However, the number of pelican nests estimated in this survey (16,406) was very similar to the number estimated in 2004/05 and 2010 (n=15,610 and n=15,999 respectively; Table 1.1). Despite fluctuations in total counts, breeding distribution across the state remained essentially the same, with colonies located in northern, west-central and southern portions of the state (Figure 1.2). During all survey periods, nearly the entire population (97.5-98.9%) occurred on six lakes; however, percent change at those lakes ranged between -47.6% and +114.8% from 2012 to 2015. (Table 1.1; Figure 1.3).

A marked decrease in nesting pairs occurred in the Marsh Lake Complex from 2012 to 2015 which reflected a lack of nesting on the Peninsula and Currie Island sites. Failure to nest successfully at these sites may have been due to water level fluctuations which resulted in flooding and/or increased predator access (DiMatteo et. al 2015). The

[1] Changes in pelican estimates reflect comparison between 2012 census and 2015 census unless otherwise noted.

vegetation on Big Island partially obscured our view of some birds and thus the count for this site represents a minimum. We assume 5-10% were obscured from view but were not confident enough in this assumption to adjust the counts accordingly. Overall, our results suggest that numbers at the Marsh Lake complex have declined by as much as 33.8% since 2012.

On Minnesota Lake, pelican pairs decreased overall by 47.6% since 2012, returning to an estimate similar to that reported in 2004. Nesting on the island site was limited by high water levels in 2015. Nesting on the adjacent agricultural field was initiated early in the summer (L. Gelvin-Innvaer, personal communication) but abandoned prior to the time we obtained aerial photographs.

At Lake of the Woods, numbers decreased by 15.3% (159 nests) as compared to 2012. Crowduck and Little Massacre islands decreased by 63.7% (123 nests) and 32.7% (81 nests), respectively. Nesting was not initiated on Red Lake Rock, which had been active during the three previous census efforts. Absence of pelican nesting at this site may have been due to the dense population of nesting Ring-billed Gulls (*Larus delewarensis*) occupying the island. Increases of 21.5% (20 nests) and 19.2% (85 nests) were observed at Techout and O'Dell islands (respectively) between 2012 and 2015.

We estimated a 13.7% (43 nests) increase in pelican numbers at Leech Lake from 2012 to 2015. Nesting on Little Pelican Island has decreased since 2012, but was initiated on Pelican Island and Gull Island, both of which had not been utilized by pelicans in previous census years.

Although pelican nesting on Pigeon Lake has previously occurred at two distinct sites (a site known as Bare Island to the north, and a pair of treed islands connected by dense cattails (*Typha* spp.) collectively referred to as the Vegetated Island that lie about 200 m to the south of Bare Island) pelican nesting was limited to the Vegetated Island site. Despite these changes, Pigeon Lake experienced an increase in pelican numbers of 28% (331 nests) from 2012 to 2015 resulting in the highest abundance (1,512 nests) observed at this site during the 5 pelican census periods.

Pelican numbers also increased on Swartout Lake and Lake Johanna (53.4% and 3.2%, or 202 nests and 62 nests respectively) since 2012. Nesting was also observed for the first time on a small island on Goose Lake in Pope County.

Changes in Distribution and Numbers of the Double-crested Cormorant[2]

Cormorant nest estimates in 2015 (15,421) were very similar to numbers reported in 2004 and 2010 (16,002 and 15,425 respectively; Table 1.1). As with pelicans, we found that breeding distribution of cormorants remained approximately the same, with colonies located across much of the state, except for the northwest, southwest and southeast corners. Most colonies were documented in a region running diagonally through the central portion of the state between Ottertail County in the North, and Faribault County in the South (Figure 1.4).

As with previous surveys, 10 lakes and islands comprised > 75% of the state's cormorant population; however, numbers changed appreciably at several of these locations (Table 1.1; Figure 1.5). The most striking reduction occurred at Wells Lake, where cormorant numbers declined by 76%, representing a loss of 912 nests. As was the case in 2010, nesting on Lake of the Woods declined, this time by an additional 26% (452 nests). Noteworthy declines totaling 876 nests were also recorded at Pigeon, Leech, Hawks Nest, and Minnesota lakes. In addition, no nests were recorded at 9 lakes in 2015 that supported cormorant nesting in 2010, including Long Lake and Lake Waconia.

These substantial reductions were offset, however, by increases at other sites. Nesting on Lake Johanna increased by 58% (1,059 nests). Numbers on Upper Sakatah Lake and Egret Island increased by 68% (549 nests) and 20% (412 nests) respectively. In addition to the increase in 2010, Swartout Lake experienced an additional gain of 25% (235 nests). Several smaller but notable increases also occurred on Preston Lake, Marsh Lake, Chautauqua Lake, Mille Lacs, and Gooseberry Island. In total, these five locations

[2] Changes in cormorant estimates reflect comparison between 2010 census and 2015 census unless otherwise noted

accounted for 1,372 additional nesting pairs. Four previously inactive/uncensused sites (Grotto, Goose, Norway lakes and Lake Benton) supported an additional 104 nests.

Changes in Cormorant Nesting at Sites with Control Efforts

During the summer of 2015, Wildlife Services oiled 391 nests at Potato Island, Lake Vermilion, and shot 1,040 adult cormorants on Leech Lake. Control efforts in previous years were undertaken on Knife Island, Lake Superior; Little Pelican Island, Leech Lake; Potato Island, Lake Vermilion; Lake of the Woods; Wells Lake; and Lake Waconia (G. Nohrenburg, personal communication). Overall, sites with control had a 68% reduction in cormorant nests since 2010.

Pelican Population Growth Models

Initially, I fit stochastic growth models to pelican data from Lake Johanna, Lake of the Woods, Leech Lake, Marsh Lake, Minnesota Lake, Pigeon Lake, and Swartout Lake (Table 1.1). All Gelman-Rubin statistics were near 1, suggesting convergence of the Markov chains. The no observation error models consistently fit the data better (based on DIC values) than the Poisson and Log Normal observation error models. The density-independent model had a lower DIC in 5 cases, with the Ricker model giving a lower DIC in 2 cases (Table 1.2). Differences between the two models were often small, however.

Given the comparable DIC results of the two no observation error models, I used both the density independent model and the Ricker model to project the Marsh Lake population forward in time. The density-independent model predicted a steady decrease in nesting pelicans over the 10 time steps from 10,289 in 2015 to 8,618 at time step 10 (Figure 1.7). The Ricker model predicted periodic peaks and valleys with a mean population size around 15,000 (Figure 1.8).

DISCUSSION

Detecting Active Sites

Efforts to detect active sites in 2015 appear adequate to determine current distribution and abundance for both pelicans and cormorants. In Minnesota, colonies of both species are typically very conspicuous and are usually discovered shortly after initiation. Given public concern about the fish consumption behavior of both species, new colonies are frequently reported to the DNR soon after they form. Therefore, “advertising” the census effort to wildlife personnel leads to more complete information on current distribution.

Reviewing information on previous activity of individual sites helps to minimize the cost and effort of the census by eliminating visits to sites that have a high probability of inactivity. The current database now contains pertinent information to identify sites with high potential for nesting activity. Of the four cormorant/pelican sites that were inactive during previous surveys, and found to be active during this survey, one (Pelican Island) was associated with a lake (Leech Lake) that had other active colonies. Because this lake was visited to check on active sites, the previously inactive site (Pelican Island) that had recently become active had a high likelihood of being detected. The other three sites (Lake Benton, Norway Lake, and East Chain Lake) were identified as potential sites during the 2004 census, although they were found to be inactive at that time. These three sites only represented 43 cormorant nests and 48 pelican nests and do not show much potential for growth due to limited availability of nesting substrate. It is recommended that site history documented in this and the previous census be reviewed prior to subsequent surveys to help identify sites that will require visits and sites that can be assumed inactive. As shown in the 2010 census (Wires et al. 2011) the majority (92%) of sites that have been inactive in the past remain inactive during subsequent surveys.

Changes in Numbers of Pelicans

Overall it appears that pelican numbers are stable in Minnesota with few factors other than water level and predator access placing limitations on nesting. Dispersal to other active sites may make the state's population less vulnerable to site specific stochastic events such as storms, disease, and human caused disturbance.

The complex of islands in Marsh Lake constitutes the most significant area in the state for breeding pelicans, and is one of the most important sites on the continent for this species (King and Anderson 2005). However, the portion of Minnesota's pelican population that nests on Marsh Lake has steadily decreased from 2004 (84%) to 2015 (63%) as pelicans have apparently dispersed and other sites have become more established. Data from banding efforts conducted during the last decade suggest some individuals from Marsh Lake disperse to adjacent states and Canadian provinces and that Marsh Lake may even be acting as a source population for new colonization events (J. DiMatteo personal communication, May 2, 2016, unpublished data). Further efforts to color-mark or capture/recapture banded individuals from this and other sites could greatly improve our understanding of pelican movement among sites and population trends in Minnesota.

Observed fluctuations at Marsh Lake may also result from changes in water level and increased predator access as suggested by DiMatteo and colleagues (2015). Numbers may also be influenced by dispersal from other sites, especially the large colony at Chase Lake, ND. Despite lower than average numbers in 2015, the colony appears healthy (i.e. no unusual mortality; regular production of fledged juveniles) and Marsh Lake continues to support the majority of the State's population.

Several other large pelican colonies also exhibited variable numbers of nesting pairs similar to the situation observed at Marsh Lake. The population on Swartout Lake is increasing after a large decline in 2011. Nesting trends at this location appear to be driven primarily by highly variable water levels. Lake of the Woods and Minnesota Lake have returned to numbers closer to those recorded in 2004/05. At Lake of the Woods, reductions may be related to increased competition from Ring-billed Gulls for nesting habitat. Gull numbers have increased in Lake of the Woods but it is unclear if or how this species is influencing pelican numbers. The island on Minnesota Lake has become so dense with nesting cormorants and pelicans that there is little space for colony growth. As a result, nest numbers at this site appear directly related to available habitat which is closely tied to water level. Lake Johanna and Pigeon Lake colonies have been steadily increasing in size since 2004. Johanna is not as susceptible to changes in water level as many other sites because of higher elevation; it also appears to have habitat for additional growth in coming years. Pelicans were densely packed on the Vegetated Island on Pigeon Lake with nearly all suitable habitat occupied by nesting birds.

Cormorant control was undertaken at only one site used by nesting pelicans in 2015, Little Pelican Island in Leech Lake. Despite repeated cormorant control efforts at Leech Lake, pelican numbers have steadily increased since 2004. As in previous years, Little Pelican and Gull islands both supported nesting birds. In addition, nesting was initiated on the larger Pelican Island to the north. This steady increase in nests suggests that disturbance caused by Leech Lake cormorant control is having little impact on pelicans. While this observation suggests cormorant management is not impacting pelicans, it is important to note that pelicans are typically quite sensitive to human disturbance and we recommend that potential impacts of cormorant management be carefully considered prior to initiating control efforts at sites with pelicans and other nesting species.

Conditions at other colony sites in Minnesota or neighboring states may also be influencing numbers. Specifically, reciprocal movement between the colony on Chase Lake, ND, and Minnesota colonies (Sovada et al. 2013) may account for a portion of observed fluctuations.

Changes in Numbers of Cormorants

While cormorant numbers and distribution across the state remained similar overall to results reported in 2004/05 and 2010, substantial changes have occurred at individual sites. For example, declines observed at Lake of the Woods between 2004 and 2010 continued in 2015, a decline also observed in pelicans. The exact cause of these changes is uncertain. There was no obvious evidence of predator presence but there are several other factors that may have an influence on cormorants nesting on Lake of the Woods. There was an outbreak of Newcastle Disease reported on Lake of the Woods in 2008 (White et al., 2015). Concern also exists that cormorants and pelicans may have been killed illegally on this lake (F. Cuthbert, personal communication). O'Dell Island, which was abandoned in 2010 due to presence of a family of red foxes (*Vulpes vulpes*), has since been recolonized by cormorants and other nesting birds. Red Lake Rock however was not used by cormorants, but was instead occupied almost entirely by nesting Ring-billed Gulls. Cormorant numbers were greatly reduced on Techout Island but much of the habitat also supported a dense Ring-billed Gull colony. Little Massacre Island had fewer gulls in comparison to Techout but still had substantial declines in cormorant nesting.

The island on Long Lake in Kandiyohi County, which supported 1,363 cormorant nests as well as 774 pairs of other colonial waterbird species in 2004, was completely vacant during the summer of 2015. A quick survey of the island provided no clear indication of disturbance. There were a few old nests that may have been remnants of a failed attempt at nest initiation earlier in 2015 or perhaps nests still remaining from a previous year.

Changes Related to Cormorant Control

Of the 10 lakes with the largest cormorant declines in 2015, three received control in 2010. Cormorants at Leech Lake, which has been the site of multiple cormorant control efforts over the last decade, declined approximately 80% since 2004. In 2015 1,040 adults were culled from Leech Lake between April 16 and June 25 (G. Nohrenburg, personal communication, February 16, 2016).

No cormorants nested on Coney Island on Lake Waconia in 2015. Nesting was initiated by Great Blue Herons early in the season but they abandoned the site later in the season. The exact reason for the abandonment of herons in 2015 is unknown. However, the absence of cormorants, the potential for human presence on the island, and disturbance during cormorant control efforts in recent years (yearly from 2008-2011) may have discouraged nesting by all species.

Similar to Leech Lake and Lake Waconia, the number of nesting pairs also declined at Wells Lake between 2010 and 2015. It is quite clear that cormorant control efforts are reducing cormorant numbers at this site, at least temporarily. What is less clear is the impact management may have on other species. For example, in contrast to the abandonment observed on Lake Waconia, the number of nesting Great Blue Herons increased on Wells Lake between 2010 and 2015.

Forecasting Future Trends in Pelican Populations

Fitted stochastic growth models produced forecasts with realistic mean population trends. A limitation, however, was the size of the datasets used to parameterize models (5 observations for 7 sites and 10 observations for Marsh Lake). Morris et al. (1999) recommend a minimum of 10 observations when conducting population viability analysis (similar to the analyses conducted here) in order to increase precision and reduce width of confidence limits.

It important to remember that the trends shown in Figure 1.7 and 1.8 represent the mean of 5,000 individual projections which varied greatly in their predictions of future nesting populations. Several iterations resulted in complete population failure (reduction to 0) and remained at zero because the models do not account for recolonization. Alternately, some iterations predicted exponential growth resulting in unrealistically large numbers and even some instances of infinite values.

As expected, prediction intervals widened as projections progressed, illustrating the fact that uncertainty around a projected value increases as time progresses. However, intervals are much wider than desired and forecasts should be considered tentative at best.

More data, which are available for select pelican and cormorant sites, may provide more reliable projections that could potentially be used to forecast site-specific trends. Nonetheless, these models make many simplifying assumptions (e.g., no catastrophic events) and should be used along with other information to inform management decisions. In the end, I suggest that, although projections such as those employed here can provide meaningful insight into population trends in certain circumstances, less complex analysis methods, like those utilized previously in census reports, appear to be more appropriate for these monitoring efforts.

CONCLUSIONS AND RECOMMENDATIONS

American White Pelican

Results of this census indicate that the breeding population of pelicans in Minnesota may be stabilizing (at least temporarily) between 16,000 and 22,000 nesting pairs. However, several sites experienced substantial declines since the peak in 2011. Declines may be due to water level fluctuations and/or other unknown factors (e.g. predation; disease; emigration out of Minnesota; illegal control; delayed effect of the Deep Water Horizon spill; sampling error). Because the exact cause of these recent declines is unknown, we make the following recommendations:

1) Continue state-wide monitoring for pelicans every five years in order to estimate population trends and identify locations where important changes may be occurring.
2) Maintain current status as Special Concern. The limited number of breeding sites used by this species and the resulting aggregation of pairs at particular locations makes this species vulnerable to stochastic events.
3) Consider assigning conservation status for yet unprotected pelican colonies. Pelicans nest at a select few locations, therefore, protecting sites used regularly by large numbers of pelicans could greatly benefit this species. Sites to consider for protection include Minnesota and Swartout lakes, as well as several islands on Lake of the Woods, specifically Crowduck, Little Massacre, O'Dell, Red Lake Rock, and Techout islands.

Double-crested Cormorants

Results of this census indicate that the breeding population of cormorants is stable and the species continues to be abundant and widespread in Minnesota. Nevertheless, noticeable declines have occurred in specific areas, such as Lake of the Woods. Important questions regarding the effectiveness of cormorant control efforts remain unanswered and require further study. The effect of cormorant control on the productivity of non-target cormorants (and co-nesting species) is also unknown. Therefore, we make the following recommendations:

1) Continue statewide monitoring for cormorants every five years as long as cormorant management is undertaken to determine population trends, identify locations where important changes may be occurring, and to evaluate how cormorant management is affecting the state's population.
2) Initiate research regarding impact of management on cormorants to: a) obtain pre- and post-control estimates of cormorant nest numbers at all sites where cormorant control is undertaken to determine how control affects nesting numbers; b) obtain data on cormorant productivity at sites where cormorant control has and has not occurred to determine the effect of control on cormorants not taken during control efforts.
3) Consider assigning conservation status for yet unprotected cormorant priority monitoring sites (Wires 2011) because they have high colonial waterbird diversity and are typically shared with waders and pelicans. We suggest these sites are important for colonial waterbirds in general. Sites fitting this description include Chautauqua, Minnesota, Swartout, and Upper Sakatah lakes, as well as Lake of the Woods (specifically Crowduck, Little Massacre, O'Dell, Red Lake Rock, and Techout islands).

CHAPTER 2

FLEDGING SUCCESS OF AMERICAN WHITE PELICANS (*PELECANUS ERYTHRORHYNCHOS*) IN MINNESOTA IN 2015

In the 1800s and early 1900s, American White Pelicans [pelicans or AWPE] (*Pelecanus erythrorynchos*) were persecuted in many areas across the U.S. due to their perceived negative impact on fisheries. To protect their livelihoods, many anglers indiscriminately shot adult birds and destroyed nests. Individuals seeking to capitalize on the abundant eggs and plumage were also known to disturb colonies. At the same time, federal and state reclamation efforts diverted water for irrigation, power generation, and other human use causing the destruction of nesting habitat. The concurrence of these events led to a drastic decrease in the continent's pelican population (Thompson 1932). In Minnesota, a report from 1878 of 200 nests on the Mustinka River in Grant County (Roberts 1932) was the last confirmed nesting colony in the state for the next 90 years.

In 1968 a colony of 70 nesting pairs was reported on Marsh Lake in Lac Qui Parle County (Breckenridge 1968). In the years since, the colony at this site has increased to become one of the largest in the U S., averaging approximately 13,000 nesting pairs (average based on 5 census efforts from 2004 to 2015). Several smaller colonies have also been established throughout the state (Table 1.1). Today, the Minnesota population of nesting pelicans averages 18,500 pairs (average based on 5 census efforts from 2004 to 2015); an impressive recovery after extirpation from Minnesota during the early part of the 20th century.

The remarkable rebound of Minnesota's nesting pelican population is likely due largely to environmental legislation and the concerted conservation efforts by various agencies and concerned parties. In 1972, pelicans were given federal protection from harvest and trade under the Migratory Bird Species Act of 1918. They were also listed as a Species of Special Concern by the Minnesota Department of Natural Resources in 1984, a distinction that prioritizes regular monitoring of selected species. Most recently, pelicans were designated as a Species in Greatest Conservation Need (SGCN) as defined by the Minnesota State Wildlife Action Plan, which aims to prevent SGCN from becoming endangered. Legislative actions such as these, along with research, habitat preservation, and habitat restoration efforts by state and federal agencies, university researchers, and interest groups such as Audubon Society, have contributed greatly to the reestablishment of pelican nesting colonies in Minnesota.

Despite rapid recovery and population growth of this species in Minnesota, pelicans still face many challenges. While current-day illegal killing is rare, such as that reported in 2011 when a farmer destroyed a portion of a colony that he feared was encroaching on his cropland (Marcotty 2012), several other factors limiting fledging success exist. Variable water levels still pose a threat to nesting pelicans. DiMatteo et al. (2015) demonstrated that spring flooding of nests situated on low elevation islands at Marsh Lake forced adult pelicans to initiate nesting on less preferred nesting habitat that was more easily accessed by mammalian predators. West Nile virus, documented in the U.S. in 1999, has recently caused small but significant increases in chick mortality at some colony sites (Sovada et al. 2013). Additionally, Sovada et al. (2014) showed that climate change over the last four decades has resulted in earlier arrival of nesting pelicans to Chase Lake, ND. A 16-day earlier arrival time has led to earlier hatching of eggs and increased chick mortality due to increased exposure to severe spring storms.

Several potential sources of mortality that are often overlooked, perhaps due to physical distance from the majority of breeding colonies, are related to oil spills such as the 2010 Deepwater Horizon Oil Spill. This spill resulted in the release of 205.8 million gallons of oil (United States 2011) and 1.84 million gallons of chemical dispersant (United States 2010) into the Gulf of Mexico.

Prior to the Deepwater Horizon spill, Leighton (1993) reviewed several studies ranging several bird species and reported that oil coming into contact with incubating eggs, even at low levels, can interrupt embryo development and result in mortality or malformation of chicks. Paruk et al. (2016) recently suggested that polycyclic aromatic hydrocarbons in the oil released during the Deepwater Horizon Spill, when absorbed into the blood of prey (i.e. marine invertebrates and fish) exposed to oil, are linked to reduced body mass in Common Loons (*Gavia immer*). These authors further posit that this reduction in body mass may contribute to reduced reproductive success.

The chemical dispersants used following the Deepwater Horizon Spill, Corexit 9500 and 9527, were applied to break up oil slicks and reduce beached oil by converting hydrophobic bulk oil into small hydrophilic droplets that can disperse through the water column rather than concentrating at the water's surface. (United States 2011). A study by

Albers (1979) examined the effect of Corexit 9527 and oil on Mallard *(Anas platyrhynchos)* hatching success. He reported that Corexit 9527 combined with oil in certain concentrations is more toxic to mallard eggs as compared to oil alone. His study also found that mallard eggs exposed to undiluted Corexit, an unlikely event in a real world oil spill scenario, suffer swift mortality. Wooten et al. (2011) observed similar results when conducting an almost identical experiment on Mallard eggs using Corexit 9500.

While these threats may seem far removed, investigators funded by the Legislative Citizens Commission for Minnesota Resources (LCCMR) detected polycyclic aromatic hydrocarbons ("PAH's", a family of organic pollutants released from fossil fuels such as oil) and dioctyl sodium sulfosuccinate ("DOSS", a major component of the chemical dispersants used to clean up the 2010 Deepwater Horizon Oil Spill) in pelican eggs collected from Marsh Lake, Minnesota, in 2012 (Minnesota Department of Natural Resources 2014). Whether these traces are linked to the Gulf oil spill is unknown. Equally unknown is whether the eggs were exposed through contact with soiled feathers or by deposition of the chemicals into the egg shell during egg formation via absorption by females after they fed on contaminated prey (Minnesota Department of Natural Resources 2014). Regardless of the method of exposure, presence of these chemicals in pelican eggs suggests a possible direct link between the Deepwater Horizon Spill and pelican reproduction in Minnesota.

In an effort to assess the health of Minnesota's breeding pelicans, a statewide census effort was undertaken during the summer of 2015. The objective of this census was to determine abundance and distribution of the American White Pelican in the state, which was accomplished by conducting aerial and ground surveys during the laying and incubation stage of breeding season (Chapter 1 of this thesis). The results were then compared to results from previous census efforts to assess changes, if any, in the breeding population. In conjunction with this census effort, a study was undertaken to develop an index of fledging success that could serve as another indicator of the health of the Minnesota breeding population. The index was developed by revisiting a subsample of sites during the fledging stage of the breeding season, estimating the number of juveniles present and then relating this estimate to the number of nests observed earlier in the breeding season. This measure of fledging success was then compared to earlier rates estimated at Minnesota and other colonies outside the state to determine if pelicans breeding in Minnesota demonstrate evidence of reduced productivity, possibly related to the 2010 Deepwater Horizon Oil Spill. For the purpose of this thesis, this chapter (Chapter 2) will focus primarily on the fledging success study with occasional references to the nesting census highlighted in Chapter 1.

METHODS

Timing of Colony Visits

Pelicans typically begin nesting in Minnesota in April and incubate for approximately 30 days and fledglings begin leaving their natal colony about two months after hatching (Knopf and Evans 2004). With the aim of observing young near fledging, I determined timing for flights based on review of annual cycles (Evans and Knopf 1993), prior experience of Francesca Cuthbert and Linda Wires, and email correspondence with Jeff DiMatteo (personal communication, June 10, 2016). I conducted two flights (one in early July and one in late July) to improve chances of proper timing (see Chapter 1 for more factors effecting timing of colony visits).

Fledgling Counts

Of the 11 lakes with active pelican nests, I revisited only four (Lake Johanna, and Marsh, Minnesota, and Swartout Lakes) at a later date after the initial pelican census (Chapter 1) to estimate fledging success. Multiple visits to all sites was cost prohibitive due to the wide geographic range of sites. I chose this subset because these lakes are four of the largest colonies in Minnesota, have limited vegetation that obscures pelicans, and are in relative proximity to one another allowing for all four to be visited during a single flight.

I counted fledglings using similar methodology to that used to count nests. I identified adult pelicans that were present based on size (noticeably larger than nearby fledglings), location (often "standing guard" around groups or crèches of fledglings), and coloration (adults have vibrant white plumage with vivid yellow-orange bills and feet as compared to the dull white plumage and dusky grey feet and bills of fledglings). Once adults were marked on images, counts of fledglings proceeded quickly.

Determining Fledging Success

I defined fledging success as the number of young per nest surviving to fledging (in this case, young present at the time of the late summer flights were considered to have survived to fledging). I estimated their age to range from 40 to 60 days post hatching. Because fledglings had departed their nests and aggregated into crèches, an estimate of fledging success could not be obtained for individual nests. Instead, I calculated fledging success by dividing the total number of fledglings present at a site by the total nests counted at that site earlier in the season. Due to the close proximity of islands on Marsh Lake, I summed nest and fledgling counts of the individual islands to account for possible movement among islands by swimming juveniles.

Detectability

Although detectability of fledglings was not measured during this study, earlier studies relating to the census methods used and observations made during previous census efforts suggest that the methods utilized result in high detection rates (see Chapter 1 for more information). Furthermore, I assumed that the high quality of the photos limited errors of differentiation between adults and fledglings and thus made no attempt to correct for this error.

RESULTS

The four lakes surveyed (Lake Johanna, Marsh Lake, Minnesota Lake, and Swartout Lake) contained a substantial portion (83%) of the total nests observed during the early portion of the 2015 breeding season. I observed much greater numbers of fledgling pelicans at all sites during the July 9 flight, likely due to dispersal from the breeding site in preparation for the fall migration prior to the July 31 flight. As such, I used counts from July 9 in fledging success calculations. I estimated site-specific fledging success rates of 1.06 (Lake Johanna), 0.45 (Marsh Lake), 0.53 (Minnesota Lake) and 0.19 (Swartout Lake) chicks fledged per nest. For all sites surveyed, I calculated a combined average of 0.54 fledglings present per nest observed (Table 2.1).

DISCUSSION

The statewide average of 0.54 fledglings per nest estimated at the four Minnesota study sites is comparable to rates observed in previous studies. In Minnesota, estimates of pre-oil spill fledging success are limited to a study by Jeff DiMatteo (2015) for Marsh Lake in Lac Qui Parle county. DiMatteo observed an average fledging success rate of 0.55 fledglings per nest during 4 years (2006, 2007, 2009, and 2010) prior to the 2010 Deepwater Horizon Oil Spill. I included data for 2010 in this calculation as birds nesting in Minnesota would not have come into contact with oil spill contaminants until returning to the Gulf in the fall of 2010. Johnson and Sloan (1978) conducted a review of nine studies that reported young produced per nest ranging from 0.21 to 1.23 (mean=0.71, median=0.45). Sidle et al. (1984) reported annual fledging success rates (therein referred to as productivity) ranging from 0.38 to 0.64 fledglings per nest during 3 years at Chase Lake, North Dakota (~300km NW of the Marsh Lake colony). In the Chase Lake study, colony-wide fledging success rates were averages of rates observed at a subset of nests monitored throughout the breeding season to determine nest specific fledging success.

Intraspecific differences in fledging success at pelican colonies have been attributed largely to environmental factors, specifically variability in water level. DiMatteo et al. (2015) found that 84% of variability in colony fledging success at Marsh Lake sites was related to water levels in April (i.e. when nest initiation begins).

The island on Lake Johanna, the site with the highest fledging success, is elevated about 6m above the water on the north side and slopes gently until it meets the water on the south side. This condition, paired with limited water level fluctuation on Lake Johanna provides the pelicans at this site with reliable access to relatively safe nesting habitat. In contrast, the island on Swartout Lake is extremely flat and rises only moderately (<1m) above the lake’s surface. The colony occupying this island has the lowest fledging success and is subject to frequent flooding. In fact, the area occupied by the colony was partially under water when nest photos were taken. The island on Minnesota Lake has a long trailing spit that is susceptible to flooding, but the majority of the island is typically well out of the water. Due to high water levels later in the year, the lower areas were flooded and some nesting habitat was inundated.

DiMatteo et al. (2015) show that variations in topography of the six islands on Marsh Lake offer pelicans some flexibility to nest on less preferred habitat (closer to or connected to the mainland). They point out, however, that this flexibility comes at risk of increased exposure to terrestrial mammalian predators. Compared to the range of fledgling success DiMatteo (2015) observed over six years (0.50-0.61 fledglings per nest), our observed fledging success rate of 0.45 fledglings per nest seems quite low.

While the 2015 statewide average of 0.54 fledglings per nest falls well within the ranges observed by DiMatteo (2015), Johnson and Sloan (1978), and Sidle et al. (1984), several considerations should be taken into account when interpreting the results. First, high sensitivity to disturbance makes it difficult to census pelicans without having at least a small effect on colony fledging success. Although I believe that my impact was limited, it is possible that my presence influenced the outcome of this study.

Second, DiMatteo's study assumed that all non-nesting pelicans disperse and therefore all observed pelicans were tending nests. DiMatteo's study also assumed that adult pelicans disperse toward the end of the season and that pelicans present in later months are by default fledglings. In contrast, my study assumed mixed age demographics during the entire season and thus effort was made to differentiate between nesting, fledgling, and loafing pelicans. This difference in methodology suggests that the results from the two studies are not directly comparable. I observed lower numbers of loafing/non-nesting adults early in the nesting season as compared to later in the season, which would skew the ratio of fledglings to nests in favor of more fledglings had I operated under the assumptions of DiMatteo's 2015 study. Given this observation, I predict higher estimates of fledging success for DiMatteo's study as compared to mine, which may account for some of the difference between observed rates.

Finally, because pelicans nest asynchronously, determining fledging success can be problematic. Schreiber (1979), in his study of brown pelicans, which are similarly asynchronous, suggests “a reasonably accurate measure of reproductive performance can be obtained by making weekly surveys of the colony and checking contents of individually marked nests.” Unfortunately, this level of census effort would greatly increase project cost, labor, and potential for disturbance. Knowing I would likely only be able to obtain a single measure of nests and fledglings, I attempted to census during periods of peak numbers of both nests and fledglings. Based on the high density of nesting adults/fledglings observed during my flights, I feel confident in my timing but recognize that some early and late nests/fledglings were likely not taken into account.

CONCLUSIONS AND RECOMMENDATIONS

Based on comparison to similar studies, I found no evidence that pelicans nesting in Minnesota in 2015 experienced fledging rates outside of expected ranges. However, the possibility of carry-over effects or impacts to individual pelicans as a result of the 2010 Deepwater Horizon Oil Spill cannot be ruled out (Bagby et al. 2016). If further research on the long term impact of this disaster on nesting pelicans is to be pursued, I suggest more in-depth analysis of the effects of spill contaminants on individual bird survival and reproductive success.

I would also suggest that managers utilize lessons learned following the 2010 spill to develop a response plan that could be enacted following a future spill event or similar disaster. This plan should include increased pelican population monitoring for a given number of years following the event, tracking nest productivity, and/or testing of feathers, eggs, water, and/or soil in nesting colonies. Such a plan should be implemented as close to the event date as possible.

State and local population trend information is important for evaluating the overall status of the species, especially before and after environmental or natural disasters. I encourage managers to continue regular monitoring of distribution and abundance of pelican populations. In addition to disaster response, population data will be invaluable when making informed management decisions in the face of problems such as human conflict and global climate change.

Finally, to facilitate management of pelicans in Minnesota and elsewhere, I encourage agency staff, nearby landowners, and casual observers to report changes in the status of pelican colonies. Because frequent population censuses are cost prohibitive, anecdotal information about changes in water level, bird abundance, nesting behavior, and other variables can help managers better understand trends observed during censuses.

CHAPTER 3

COMPLEMENTARY MONITORING TECHNIQUES FOR CENSUSING AMERICAN WHITE PELICANS (*PELECANUS ERYTHRORHYNCHOS*) IN MINNESOTA: ALTERNATIVE SOURCES OF COLONY IMAGERY AND OBJECT-BASED IMAGE ANALYSIS

For more than a decade, the Minnesota Department of Natural Resources (DNR) has funded an ongoing census of colonial waterbirds that focused, in part, on the American White Pelican (*Pelecanus erythrorhynchos*, here on 'pelican'). Census data, obtained approximately every 5 years since 2004, are used to monitor distribution and abundance of target species and to inform decisions related to habitat management and mitigation of human/wildlife conflict. Currently the statewide census of nesting pairs is conducted using a combination of ground counts and counts obtained from low-altitude aerial photography. The aerial photos are acquired using a hand-held digital camera with a zoom lens. Photos are then analyzed manually using GIS software (a point feature is placed on each bird). During this process, individuals are identified as nesting (physically sitting on a nest) or loafing (non-nesting/standing away from nests). This level of differentiation can be negatively impacted by individual subjectivity, poor photo quality and density of vegetation. As such, the project requires high resolution imagery or ground data that are expensive to collect and time consuming to process. These challenges limit the frequency of the census efforts. However, alternative sources of remotely sensed imagery and computer aided image analysis could serve as tools to provide cost effective population estimates during non-census years and/or speed up photo analysis.

To date, wildlife related applications of imagery gathered by large-scale remote sensing operations (i.e. imagery collected using satellites or aircraft with specialized imaging devices covering vast swaths of land) have mostly focused on habitat mapping and have not been widely applied to estimate populations. Of the limited applications that have been reported, the use of commercially available satellite imagery to detect and count penguins in Antarctica has demonstrated the most promising results. Fretwell et al. (2012) used DigitalGlobe Inc. ('DigitalGlobe') imagery to locate potential Emperor Penguin (*Aptenodytes forsteri*) colonies. A commercial satellite was then tasked (i.e. 'hired' to acquire imagers for a specific purpose) to acquire high resolution imagery of the colonies. Imagery was then processed to create 61 cm resolution 4-band imagery (typically consists of red, green, blue, and near infrared wavelengths). Supervised classification (form of pixel-based image classification that requires the user to 'train' the applied software to recognize certain pixels within the image as belonging to a user defined category or class) was then employed to differentiate penguins from surrounding substrates. Following classification, penguin population estimates were calculated from the area of the ‘penguin’ class using an index of penguins to area that had been derived using information from previous studies. This process yielded an estimate of the entire penguin population with a confidence interval of ± 13%.

Lynch et al. (2014) applied decision rule and transition matrix algorithms to automate detection of Adélie Penguin (*Pygoscelis adeliae*), colonies using Landsat-7 imagery. This method was successful in identifying colonies that made up roughly 97% of the population that had been censused during a previous study using high resolution imagery. Additionally, the study demonstrated that estimates of penguin abundance could be determined given previously derived measures of bird density within colonies. Clearly, remotely sensed imagery and related analysis applications like those used by Fretwell et al. (2012) and Lynch et al. (2014) are powerful tools that have the potential to contribute to a wide array of wildlife monitoring efforts.

For the purpose of my research involving pelican monitoring, Heather Lynch, associate professor in Ecology and Evolution at Stoney Brook University, and senior author of one of the penguin studies cited above, suggested that a combination of supervised classification and object-based image analysis could potentially be used to classify images (personal communication, February 22, 2016). Supervised classification works fairly well with low resolution images but can perform poorly when using high resolution images due to the complexity of color patterns when fine details are visible. Object-based image analysis (OBIA), while similar, first segments the image into ‘objects’ using algorithms to recognize colors, textures, shapes, and other patterns within the image. These objects are then classified based on a ruleset, which is a sequential set of ‘rules’ that dictate the criteria an object has to meet (e.g. brightness, color, size, linearity, etc) to be included in a given class. OBIA works well with high resolution images because of its ability to recognize patterns prior to classification.

I developed the study outlined in this chapter based on my interest in whether alternative image sources and computer aided analysis, like that suggested by Lynch, could aid in pelican monitoring efforts. First, I explored possible alternative sources of colony imagery, specifically various depositories of aerial and satellite imagery such as DigitalGlobe. I then assessed the utility of computer aided analysis to assist in counting pelicans. For this objective, I conducted analysis using both an image taken during the most recent pelican monitoring effort and an image from one of the alternative image sources. Finally, I compared costs associated with current survey methods versus costs estimated for proposed alternative methods.

METHODS

Exploration of Alternative Imagery

For budgetary reasons, I limited my search for alternative imagery to sources that are free to the public or for which the University of Minnesota has a license to access. Free sources that were considered are USGS Earth Explorer, USGS Global Visualization Viewer (GloVis), and Google Earth. As a University of Minnesota student I was also granted basic access to DigitalGlobe, a commercial imagery provider.

While exploring these resources, I considered several criteria that images would need to meet to be suitable for monitoring purposes. I first searched for sources that offered imagery for the desired colony locations. As of summer 2015, 15 islands (sites) on 8 lakes supported pelican colonies in Minnesota. During our statewide census efforts, sites with heavy vegetation were visited by ground crews because dense vegetation obscured nesting birds and limited image interpretation. For this reason, sites known to have dense vegetation were excluded from the search. I then narrowed my search based on timing of image acquisition. Our census efforts generally occur between late April and late June when pelican nests reach peak concentrations (Evans and Knopf 1993). This relatively narrow date range creates a challenge when searching for non-tasked imagery (i.e. imagery collected for no specific purpose) because a good portion of available imagery is obtained during early spring or late fall when vegetation cover does not obstruct the underlying physical features. In considering timing, I also looked for imagery that was not obstructed by cloud cover or impacted by other complications (e.g. sun glare, fog, smoke from forest fires). Finally, I considered resolution of images (i.e. the ground area represented by a single image pixel). A nesting pelican measures a little over a meter from tail to beak and a little more than a half meter wide (with wings tucked away). As such, I focused my search on imagery with <1m resolution imagery so pelicans would register as more than a single pixel.

After a thorough investigation, a single high-altitude aerial image was selected from DigitalGlobe for use during the assessment of computer aided analysis. See Results below for an explanation of why this source was selected.

Acquisition and Selection of Imagery from the 2015 Pelican Monitoring Efforts

I acquired all traditional aerial photo imagery for the 2015 monitoring efforts using a Nikon D200 10.20MP digital SLR camera fitted with an AF-S DX NIKKOR 18-200mm f/3.5-5.6G ED VR II lens. The plane, an American Champion Scout, was flown at an altitude of about 500 feet (I will refer to this imagery as low-altitude aerial imagery to avoid confusion with the high-altitude aerial imagery acquired from DigitalGlobe) to acquire images that would allow for species differentiation as well as activity interpretation (nesting vs. loafing). During aerial surveys, I made several passes and took photos at various angles and levels of zoom, often taking as many as 100 photos per location.

I selected census imagery based on what was available through the alternative image sources. This involved an iterative process of comparing dates and locations to identify images that exhibited comparable pelican density. I also selected survey images containing an entire island (rather than zoomed-in partial-coverage photos) because the vast majority of alternatively sourced imagery capture areas much larger than a single site and therefore offer a single snapshot of an entire colony.

Computer Aided Analysis using Object-Based Image Classification

I decided to employ object-based image analysis for two reasons. First, as stated above, object-based image analysis is particularly useful for classifying high resolution images. Second, I had observed demonstrations of eCognition software (an OBIA application) and was intrigued by the user friendly and intuitive nature of the software.

As a result of the software's approachability, I was able to conduct relatively complex image analysis despite limited experience with remote sensing concepts and software.

Using eCognition software (ver. 9.1.1), I developed a ruleset (Figure 3.2) for the DigitalGlobe image primarily through trial and error. I started by classifying objects that were the most dissimilar to pelicans, in this case water and vegetation. First I segmented the image into large homogenous objects and classified water based on low values of 'intensity' (water absorbs the majority of light, so reflected light is represented by low intensity value from 0 to 255) and 'max difference' (i.e. measure of variability between pixels in an object). I then merged the remaining unclassified objects and resegmented the unclassified area into smaller segments. With these smaller segments, I first classified vegetation based on low brightness level, which is a relative measure based on how bright an object appears. In this case, I specified a sum of the green and blue values < 228 (as with intensity, color values range from 0-255 and have no unit associated) to capture the dark green vegetation. I then created a pelican class based on high brightness levels. I excluded some of the larger bright areas by limiting pixels per object allowed in this class. I then placed the remaining unclassified objects into a class labeled bare earth (Figure 3.3).

For the low-resolution imagery, I used the ruleset developed for the DigitalGlobe image rather than create a new ruleset. I then adjusted the ruleset to account for differences in resolution, light intensity, and color richness but features used for classification remained largely the same (i.e. dark items classified as water; green as vegetation) (See Figure 3.4 for ruleset; see Figure 3.5 for classified image).

Population Estimation and Accuracy Assessment

I calculated total pixels per class using information extracted from eCognition. I estimated the average number of pixels per object in the pelican class (used as a proxy for the number pixels per pelican) by averaging the size of the middle 95% of objects (very large and very small objects were excluded from this calculation) in that class. I then calculated a raw population estimate (no adjustment for errors of omission and commission) by dividing the pelican class by the average pixels per object in that class.

After the raw population was derived, I conducted an accuracy assessment to obtain measures of commission and omission error that could then be used to adjust the population estimate to account for errors in classification. I was unable to locate clear methods for assessing accuracy that were applicable for my classification and so I developed a set of methods based on methods utilized for pixel-based classifications. First, I segmented each class into individual pixels. I then output the segments as two vector shapefiles; one was created from the pelican class and another by combining the other classes into what was deemed the non-pelican class. I discovered that including water in later calculations led to extreme overestimation of populations. This is likely because water occupied a disproportionately large portion of the image and contained almost no pelicans. To remedy this bias, sampling was limited to a region of interest that included the island but excluded water. Next, I imported the vector shapefiles to ArcMap 10 along with the original imagery. I manually lined up the images, which were not georeferenced, with the shapefiles using the shift and resize tools. To randomly select individual pixels for accuracy assessment, I first derived a count of the pixels in each class from the attribute table for each shapefile. Then, I utilized the statistical software R (version 3.2.3) to generate random numbers between zero and the total pixels in each class. I stratified my sampling by generated more random numbers for the non-pelican class than for the pelican class (120 for non-pelican, 60 for pelican). I used a stratified sample because the non-pelican class was much larger (>10 times more pixels) than the pelican class. Once numbers were generated, I identified the vector feature with the corresponding number by searching the attribute table and locating that feature on the image. I recorded class membership (1 for pelican, 0 for non-pelican) as well as the actual value observed (1 for pelican, 0 for non-pelican). A third column was generated with values indicating whether the pixel was properly classified (1 for correctly classified, 0 for incorrectly classified).

I only completed classification accuracy assessment for the low-altitude aerial image due to problems registering the DigitalGlobe image and the eCognition shapefile output. For an unknown reason, I did not experience this issue when importing files for the low-altitude aerial imagery. I used the accuracy assessment results to calculate commission and omission error which were then used to recalculate the population estimates using the equation below (Table 3).

$$Adjusted\ Population\ Estimate = \frac{Px\ in\ Pelican\ Class\ +\ ((Total\ Px - Water) * Omission) - (Px\ in\ Pelican\ class * Commission)}{Average\ Px\ Per\ Pelican\ Object}$$

Estimation of Costs

I estimated the cost of utilizing remotely sensed imagery based on time required for imagery acquisition and analysis. I assumed that access to imagery and software required for analysis would be provided by a university or agency, and therefore I did not include the rather substantial costs of DigitalGlobe subscription and software licensing.

I estimated costs associated with current monitoring methods based on historical operational costs for cormorant and pelican censuses in Minnesota. I conducted a sample manual count of a handful of previously uncounted photos was to estimate the time required for manual analysis. This sample included not only counting of several photos, but also time spent determining overlap and creating of shapefiles.

RESULTS

Alternative Sources of Colony Imagery

The USGS websites provide a vast database of both aerial photographs and satellite images. However, I found that the imagery available was of a much lower resolution than required for monitoring pelicans. Additionally, the poorly designed user interface made searching through available imagery cumbersome and time consuming.

I found images available through Google Earth to be of a much higher resolution, although at the time this study was originally developed, I was unable to locate images acquired during the breeding season. I discovered shortly after completion of this study in May of 2016 that Google had updated their imagery and the new images showed quite clearly not only pelicans but also cormorants sitting on nests at several colony locations in Minnesota.

The last source I explored was DigitalGlobe, a commercial imagery provider. I found DigitalGlobe's database to be very extensive, offering many years of imagery from a variety of sources; it was also very "user friendly". The user interface for selecting and downloading data is highly intuitive which allows for quick location of relevant imagery. DigitalGlobe also offers fairly high resolution images even with the basic subscription. The biggest issue I encountered while locating usable imagery was cloud cover and seasonal timing, which are problematic factors regardless of resolution. Nonetheless, I eventually located a single usable image that was then used to assess the utility of OBIA.

Image Acquisition for Object-Based Image Analysis Assessment

I selected an image from DigitalGlobe that captured a snapshot of a pelican colony on Lake Johanna in Pope County, Minnesota (see Figure 3.1 for a site map). The image identified was a natural color (RGB) image with 30 cm resolution. Date of acquisition was 6 July 2011. Low-altitude aerial photos of Lake Johanna were acquired twice during the summer of 2015 (April 29 and July 9). The July 9th images were selected to best match the nesting stage at the time the DigitalGlobe imagery was collected. Of the 92 photos taken at Lake Johanna on that date, a photo (#15) taken during one of our approaches was selected because it captured the entire extent of the island and therefore provided a good comparison to the DigitalGlobe image.

Preprocessing and Image Classification

Prior to classification of the DigitalGlobe image, I cropped the image to the island margins to reduce the total pixels being analyzed. Due to high light intensity caused by reflectance in rocky areas of the island I also used image adjustment tools available in Erdas IMAGINE (ver. 13.00.00) to increase the contrast of the DigitalGlobe image prior to importing the image to eCognition. I did not make these adjustments to the low-altitude aerial imagery because the photo was already cropped fairly tight to the island margins and the angle of the photo minimized reflectance issues.

I realized early in the classification that I needed to address the presence of other white bird species that could be misclassified as pelicans because of their similar coloration, specifically Great Egrets (*Ardea alba*) and Ring-billed Gulls (*Larus delawarensis*). Fretwell et al. (2012) suggested creating areas of interest to omit undesired objects/species. Fortunately, due to small size (gulls are smaller than the pixels at 30cm resolution) and/or nesting location (egrets nest within trees and are often partially or entirely obstructed), many of these individuals were automatically classified as other cover types. See Figure 3.2 for a screen capture of the DigitalGlobe image classification ruleset; see Figure 3.3 for pre- and post- classification images.

Given the high resolution of the low-altitude image, gulls and egrets were quite conspicuous. I excluded these species from the pelican class by adding a step to remove objects below a certain pixel count (both gulls and egrets are much smaller than pelicans) and placing them into their own class. See Figure 3.4 for a screen capture of the low-altitude image classification ruleset; see Figure 3.5 pre- and post-classification images.

Population Estimates and Classification Accuracy

Based on the classification of the DigitalGlobe image, I estimated a colony population of 3,343 individuals, which is nearly three times that observed following a census of this site conducted in 2011 (1,203). As mentioned previously, I did not conduct accuracy assessment for this location and so an estimate adjusted for classification error was not calculated. See Table 3.1 for classification summary and populations estimates.

For the low-altitude aerial image, I calculated a population of 2,351, which only differed from 2015 survey observations (2,504) by 153 individuals. Following accuracy assessment, I calculated my overall accuracy for the classification of the low-altitude aerial image to be 88% (Table 3.2). Using derived values of commission and omission error, I recalculated the pelican population adjusting for error resulting in an estimate of 4,268 individuals. (Table 3.1).

Cost Comparison

Table 3.3 summarizes the primary expenditures for both census methods. For the OBIA approach, location of a single usable image took upwards of half an hour due to difficulty locating imagery that was acquired while pelicans were present and visible.

Once located, the imagery had to be compiled and downloaded from DigitalGlobe, which took approximately 6 hours. Development of a working ruleset required an additional 8 hours (mostly due to the learning curve of orienting myself within eCognition). It should be mentioned that once the ruleset was developed, it was much easier to adapt it to new sites/photos. Running a complete ruleset ranged from 30 seconds to 2.5 minutes, with variation in time depending largely on the number of pixels in an image. Exporting classifications as shapefiles and randomly generating numbers required less than 5 minutes, while visually determining whether pixels had been classified correctly required nearly 3 hours of effort.

Time to count low-altitude photos ranged from 0.5-2 hours per photo depending on density of birds, number of species, and the quality of the photo. Given an average of 5 photos per site, the total time spent is estimated to be around 6.25 hours. Image acquisition involved visiting a total of 65 colonies over the course of roughly 39 hours of flight time, resulting in an average of 36 minutes of flight time per site. At $320 per hour, this translates to a flight cost of approximately $192 per colony. Factoring in travel to the airport at approximately $0.55 per mile and an hourly wage of $15.8/hr adds $18.57 per colony (($0.55/mile* 260 miles*4 flight days+15.8*39)/65 colonies) to acquisition costs (total = $210.57 per colony). A conservative approach that spreads costs for visiting empty sites among the sites with birds (39) would result in a per active colony acquisition price of approximately $424.33.

DISCUSSION

Object-based classification of a high resolution image acquired during the most recent statewide pelican survey (2015) provided a very accurate estimate of pelicans present, albeit with reduced precision as compared to manual counts. Although the image acquired from DigitalGlobe performed poorly during OBIA classification, the image did illustrate that alternative image sources could be used to locate images that can be used to estimate pelican abundance. A cost comparison of methods suggests that object-based image analysis and alternative imagery sources are cost effective resources that could be employed to provide supplemental data during periods between statewide surveys.

Sources of Remotely Sensed Imagery

My search for alternative colony imagery was not as productive as I had hoped, but this may not have been the case had I searched today or a month from now. These databases are constantly updated with new imagery and new platforms are being launched with new capabilities and higher resolutions. As I alluded to previously, Google Earth updated its images shortly after the initial completion this study providing high resolution images of several colony locations that would be ideal for monitoring overall pelican colony abundance.

The biggest issue I foresee with using the image resources I explored is the inability to locate what you need when you need it. Achieving proper timing for monitoring pelicans in Minnesota is especially problematic because their short breeding season coincides with frequent and unpredictable cloud cover. Even on a day with intermittent clouds, an entire colony can easily be obscured by a small cloud formation. Ancel et al. (2014) demonstrated that reliance on satellite imagery can lead to omission of features due to suboptimal viewing angle, inclement weather, and poor timing. In their study, several penguin colonies were missed during their search due to landscape obstructions, movement of colonies, and masking of spectral signatures by snow cover. At pelican colonies, physical features as well as presence of several other nesting species also have the potential to complicate analysis. With these limitations in mind, low-altitude aerial surveys allow more flexibility than most alternative imagery sources because flights take place below most cloud formations images can be captured from multiple angles.

Viability of Estimating Populations with Object-Based Image Analysis

Overall the classifications performed well but there is room for improvement of rulesets. For example, the extreme over-estimation of the population in the DigitalGlobe image is mostly due to a large portion of bare ground being incorrectly placed into the pelican class (Figure 3.4). With more time, the ruleset could be further refined and some of this error could likely be resolved.

An enormous benefit of eCognition and similar software is that a well-developed ruleset can be applied to similar imagery (i.e. comparable light intensity, resolution, level of zoom) and retain much of the accuracy with little to no adjustment. This concept is illustrated in Figure 3.6, which displays a low-altitude aerial image taken at another pelican site (Big Island, Marsh Lake, Lac Qui Parle County, MN) before and after classification using the same ruleset as was used for the low-altitude aerial imagery of Lake Johanna. Re-use of previously developed rulesets would likely be more reliable in satellite and high-altitude imagery due to consistencies in distance and resolution. For low-altitude aerial imagery, distance and angle change considerably while photographing sites. As such, the ruleset will need to be adjusted manually from one photo to the next unless efforts to standardize flight altitude and image angle are adopted.

Given imagery with sufficient resolution, object-based image analysis can be used to derive abundance estimates comparable to those of manual counts with regards to accuracy, at least for pelicans. However, limitations of alternative imagery and object-based analysis must be considered when planning future monitoring efforts. As expected, fine analysis such as activity interpretation (nesting v. non-nesting) is not possible using OBIA methods or when using low resolution images. As is, use of these resources may be limited in scope to ascertaining presence/absence or determination of relative abundance. However, this limitation may be offset by the potential to obtain counts on an annual rather than 5-year basis; a capability that may be beneficial in response to certain natural or anthropogenic events.

Reliability of Accuracy Assessment

Adjusting for classification errors after accuracy assessment resulted in a gross overestimate of the pelican population present in the low-altitude image of Lake Johanna. I suspect that this overestimation is a result of deficient accuracy assessment methods rather than a reflection of errors in the classification, based on the accuracy apparent upon visual inspection of the classification and the proximity of the unadjusted estimate to the manual count. A truly stratified random sample with many more control points may produce better results, but assessing a large number of random points would be very time consuming and would defeat the purpose of using OBIA to automate the counting process. Changing the accuracy assessment methodology from pixel-based sampling to object-based sampling is an option that may greatly reduce the number of samples needed but I am unaware of a process for implementing this method.

Cost Considerations and Method Alternatives

If free[3] access to eCognition, or similar software, is available and imagery can be acquired for free[3] through alternative sources, estimation of pelican populations could potentially be far less expensive than project-specific acquisition and manual analysis of low-altitude aerial imagery. The admittedly conservative costs for these alternative methods could be further reduced if efficiencies in download time (DigitalGlobe files could be compiled overnight and downloaded in the morning) and the ability to re-use rulesets are considered. However, due to the limited time window of nesting, use of third party image acquisition for a statewide census will likely require tasking of satellites, commercial aerial operations, or an Unmanned Aerial System (UAS). Depending on the platform used and the resolution requested, imagery could get very expensive, especially if repeated surveying is required due to weather or other interference. Some of the cost of tasked imagery collection could potentially be offset by coordinating with other parties who could benefit from the imagery.

Technologies for Future Consideration

Unmanned Aerial Systems, perhaps the most practical alternative given the safety benefits, mobility, and flexibility a UAS could provide, are ideal for monitoring species with small ranges or for projects concerned about a single population/location (Hodgson et. al. 2016). Unfortunately, utilizing a UAS specifically for a statewide census would likely be cost prohibitive even if we already owned a unit due to several factors. Federal Aviation Administration (FAA) regulations require that a continuous line of site to the craft be maintained by one or more project personnel (Federal Aviation Administration

[3] Access to resources provided free of charge to student/employee by institution/employer.

2016) which would require two or more personnel be present at the majority of site surveys. In addition to survey time, this would require wages be paid to all personnel for travel time and expenses. Because of the huge area covered by the statewide census, lodging would be required when visiting some of the more distant sites or areas where there are more sites than can be surveyed in a single day. Temporary lodging may also be required for personnel if inclement weather or mechanical issues cut operations short. A UAS may still be a viable option if a subset of the most populous sites is selected for UAS survey rather than conducting visits to all sites.

Incorporating other technology such as band specific sensors, LIDAR, radar, or thermal imaging could offer additional options for classification of various features. For instance, Cynthia Berger (2012) has shown that many birds have plumage that looks quite different when viewed in ultraviolet (UV) wavelengths. In the case of cormorants, which have black iridescent plumage, imagery with UV bands may provide a reliable means of differentiating them from shadow and other dark objects. Lidar data would add significant power to object-based analysis as it would allow much more accurate removal of objects based on height (e.g. differentiating ground-nesting pelicans from tree-nesting egrets).

CONCLUSIONS

Multiple sources of high quality imagery that could be used to monitor wildlife populations exist and many offer images for little to no investment. Unfortunately, using these sources limits control over when, where, and how the images are collected. However, as providers of remotely sensed images increase the resolution of imagery, as well coverage and capture frequency, the ability to utilize imagery for individual and population level studies (in addition to habitat-focused wildlife research) will also increase.

Object-based image analysis software such as eCognition can offer a great deal of aid in classifying images with the intention of quantifying wildlife. The methods used in this exploratory analysis are relatively simple yet provided a large amount of information given the effort and expense.

A final, and key difference between nest counts from low-altitude photos and the OBIA is the inability to differentiate incubating birds from those not on a nest when using OBIA. The resolution is simply not good enough from most satellite data sources. However, it could be argued that, if the primary goal of a census is overall abundance, then the accuracy and precision of current methods is unnecessary and the less discriminating method may suffice.

Finally, wildlife scientists have only begun to explore how remotely sensed imagery and related analysis techniques can contribute to studies of free-ranging animals, and the prospect of where these technologies can take us, is truly exciting.

TABLES

Table 1.1: Nest estimates of American White Pelicans (censused in 2004, 2010, 2011, 2012, and 2015) and Double-crested Cormorants (censused 2004, 2010, and 2015). Only sites with nesting pelicans or cormorants in 1 or more census periods are shown. * indicates individual count, not nest count.

	AWPE					DCCO		
Site Name	**2004**	**2010**	**2011**	**2012**	**2015**	**2004**	**2010**	**2015**
Barry Lk WPA	0	0	NC	NC	0	79	42	131
Big Twin Lake	16	0	0	NC	0	0	0	0
BLM 80-Knife Is	0	0	NC	NC	0	26	103	116
Bolland Slough	0	P	NC	NC	0	50	19	0
Chautauqua Lk	0	0	NC	NC	0	401	414	693
Clifford Lk (Swim Lk)	0	0	NC	NC	0	48	30	0
Coney Is, Waconia Lk	0	0	NC	NC	0	250[1]	425	0
Dark River Tailings Pond	0	NA	NC	NC	0	70	NA	0
Egret Is	0	0	NC	NC	0	1385	1653	2065
Elysian Lk	0	NA	NC	NC	0	205	NA	0
Goose Lk	0	NC	NC	NC	48	0	NC	37
Gooseberry Is	NA	0	NC	NC	0	NA	50	415
Grotto Lk	NC	NC	NC	NC	0	NC	NC	46
Haldorsen Lk	0	NA	NC	NC	0	20	NA	0
Hanska Lk	NA	3	P	P	P	NA	38	P
Hawks Nest Lk	NA	0	NC	NC	0	NA	408	238
Lindquist WMA	0	0	NC	NC	P	32	3	0
Little Elk WMA	0	0	NC	NC	P	49	0	0
Little Pine Is, Voyageurs N.P.	0	0	NC	NC	0	173	123	P
Lk Alice	0	0	NC	NC	0	49	88	42
Lk Benton	NC	0	NC	NC	P	NC	0	6
Lk Hassel	19	0	0	NC	0	54	264	307
Lk Johanna	97	735	1203	1904	1966	580	782	1841
Long Lk	0	0	NC	NC	0	1363	747	0
Mink Lk	NA	0	NC	0	0	NA	43	37
MNDNR 36 Guano Rock	0	0	NC	NC	0	24	14	0
Norway Lk	NC	NC	NC	NC	0	NC	NC	15
O'Brian Lk	NA	0	NC	NC	P	NA	43	25
Pigs Eye Lk	0	0	NC	NC	0	150[2]	169	118
Preston Lk	0	P	NC	NC	0	186	253	450
Red Lake	340	0	P	P	0	0	0	0
Swartout Lk	49	913	11	176	378	86	703	937

Table 1.1 (cont.): Nest estimates of American White Pelican (censused in 2004, 2010, 2011, 2012, and 2015) and Double-crested Cormorant (censused 2004, 2010, and 2015). Only sites with nesting pelicans or cormorants in 1 or more census periods are shown. * indicates individual count, not nest count.

	AWPE					DCCO		
Site Name	**2004**	**2010**	**2011**	**2012**	**2015**	**2004**	**2010**	**2015**
Swenson Lk	0	P	P	NC	P	271	230	142
Upper Sakatah Lk	NA	P	P	P	P	NA	259	808
Vinge Lk	NA	0	NC	NC	0	NA	12	0
Wells Lk	0	0	0	P	0	472	1197	285
West Two Rivers Reservoir	0	0	NC	NC	P	47	102	51
Lake Vermilion								
Potato Is	NA	P	NC	NC	0	NA	307	339
Vermillion Rocks #1	0	NA	NC	NC	0	32	NA	0
Lake Vermilion Total	**0**	**0**	**0**	**0**	**0**	**32**	**307**	**339**
Lake of the Woods								
Crowduck Is	242	408	160	193	70	447	73	P
Gull Rock	0	0	0	0	0	66	65	64
Little Massacre Is	277	185	533	248	167	1363	918	323
O'Dell Is.	25	0	450	442	527	1889	0	823
Red Lk Rock	NA	292	43	60	0	NA	159	0
Techout Is	25	143	126	93	113	605	477	30
Lake of the Woods Total	**569**	**1028**	**1312**	**1036**	**877**	**4370**	**1692**	**1240**
Leech Lake								
Gull Is	0	NC	NC	NC	23	0	NC	105
Little Pelican Is	11	174	239	314	108	2524[3]	688[3]	391
Pelican Is	NC	NC	NC	NC	226	NC	NC	0
Leech Lake Total	**11**	**174**	**239**	**314**	**357**	**2524**	**688**	**496**
Marsh Lake								
Banding Is	4160[4]	684	1074	3579	4159	NA	0	0
Big Is	5292[4]	1082	279	6465	5376	264	303	914
Peninsula	2706[4]	4650[5]	8983	0	0	NA	103	0
Rock Is	0	0	0	0	P	414	504	225
Currie Island	0	4813[5]	6245	5163	0	0	0	0
Small Is	1020[4]	4	0	337	754	NA	0	0
Marsh Lake Total	**13178**	**11233**	**16581**	**15544**	**10289**	**678**	**910**	**1139**
Mille Lacs								
Hennepin Is	0	0	NC	NC	0	5	16	5
Spirit Is	0	0	NC	NC	0	95	201	514
Mille Lacs Total	**0**	**0**	**0**	**0**	**0**	**100**	**217**	**519**
Minnesota Lake								
Minnesota Lk	974	622	429	1868	979	725	1252	1097
Ag Field	0	748	1458	0	0	0	0	0
Minnesota Lake Total	**974**	**1370**	**1887**	**1868**	**979**	**725**	**1252**	**1097**

Table 1.1 (cont.): Nest estimates of American White Pelican (censused in 2004, 2010, 2011, 2012, and 2015) and Double-crested Cormorant (censused 2004, 2010, and 2015). Only sites with nesting pelicans or cormorants in 1 or more census periods shown. * indicates individual count, not nest count.

	AWPE					DCCO		
Site Name	**2004**	**2010**	**2011**	**2012**	**2015**	**2004**	**2010**	**2015**
Pigeon Lake								
Bare Is	357	24	6	115	0	1450	1846	826
Vegetated Is	0	519	1267	1066	1512	53	299	960
Pigeon Lake Total	**357**	**543**	**1273**	**1181**	**1512**	**1503**	**2145**	**1786**
Total Nests	**15610**	**15999**	**22506**	**22023**	**16406**	**16002**	**15425**	**15421**
Total Sites	**16**	**17**	**16**	**15**	**15**	**38**	**42**	**36**

NA=Not applicable, no cormorants present at site. NC=Species was not censused, P = Present Not-Nesting

[1]Estimate was obtained based on weekly counts; number of nests estimated ranged between 250-400. Used minimum number in this report.

[2]Estimate was obtained based on weekly counts; number of nests estimated ranged between 150-200. Used minimum number in this report.

[3]Estimates for DCCO combine numbers on Little Pelican and Gull islands.

[4]Pelican counts for these sites were conducted in 2005.

[5]Based on total count and extrapolation.

Table 1.2: Deviance information criterion (DIC) associated with stochastic growth models. Models were fit to five years of pelican data at 7 Minnesota lakes.

	Model	Density Independent			Ricker		
	Obs. Error	None	Poisson	Log Normal	None	Poisson	Log Normal
Lakes	Lake Johanna	13.9	44.2	22.9	14.5	44.2	44.2
	Swartout Lake	23.6	35.8	29.2	15.6	35.5	35.8
	Lake of the Woods	9.1	43.1	18.1	12.9	43.1	43.1
	Leech Lake	16.3	37.7	27.5	15.4	37.7	37.7
	Marsh Lake	8.4	6431.4	15.4	12.9	6431.4	6431.4
	Minnesota Lake	10	44.5	17.9	13.6	44.5	44.5
	Pigeon Lake	9.1	43.2	15.6	13.6	43.3	43.2

Table 2.1: American White Pelican productivity rates for 4 lakes surveyed in Minnesota in 2015. Nests counted between late April and Mid June. Fledglings counted on July 9th.

Site	Nest Count	Fledgling Count	Fledging Success (Fledglings/Nest)
Lake Johanna	1,966	2,089	1.06
Marsh Lake	10,289	4,617	0.45
Minnesota Lake	979	523	0.53
Swartout Lake	378	71	0.19
Total	**13,612**	**7,300**	**0.54**

Table 3.1: Summary of object-based classification results and derived population estimates.

Image Source	**DigitalGlobe Image**	**Low-Altitude Aerial Photo**
Date Collected	7/6/2011	7/9/2015
Avg. Pixels/ Pelican Object	4.69	94.97
Pixels in Pelican Class	15,673	223,274
Total Pixels (Minus Water)	223,975	3,736,230
Manual Count	1,203	2,504
Raw Estimate	3,343	2,351
Population Estimate	NC[1]	4,268

[1] NC= Not calculated due to technical problems while conducting accuracy assessment. See page 42 for details.

Table 3.2: Accuracy assessment confusion matrix for object-based classification of low-altitude aerial imagery.

Reference Map		No. Pixels Classified as:			
	# of Pixels	Pelican	Not Pelican	Total	Omission Error
Pelican	60	42	3	45	0.0667
Not-Pelican	120	18	117	135	0.1333
Total	180	60	120		
Commission Error		0.3	0.025		
Overall Accuracy			**0.883**		

Table 3.3: Cost analysis of alternately sourced imagery with object-based image analysis versus current survey methods using low-altitude aerial imagery. Assumes hourly wage of $15.8/hr.

Alternative Methods	Time (Hrs)	Total Cost
DigitalGlobe Image Acquisition	6.5	$102.70
OBIA Ruleset development	8	$126.40
Accuracy Assessment	3	$47.40
	Per Colony Cost	**$276.50**

Current Census Methods	Distance	Rate	Time (Hrs)	Total Cost
Flight time (Total)		$320/hr	39	$12480.00
Travel to Airport (Total)	260 mi	$0.55/mile	4.73	$217.69
Manual Counting (Active Sites)			243.75	$3,851.25
			Total Cost	$16,548.94
			Per Active Colony Cost	**$424.33**

FIGURES

Figure 1.1: Minnesota sites with breeding Double-crested Cormorants or both Double-crested Cormorants and American White Pelicans in 2015. Sites censused Late April-Mid June using ground counts and aerial photography.

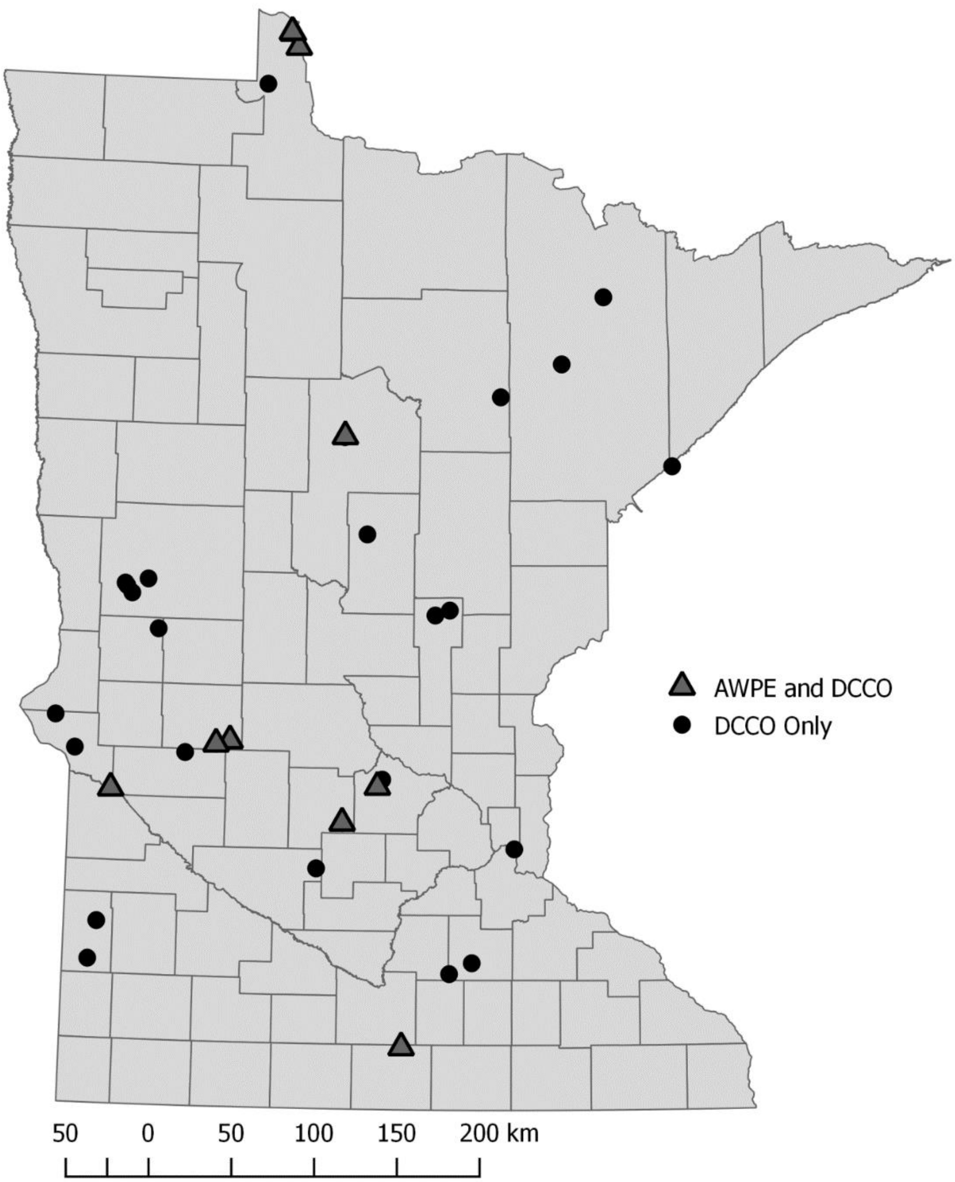
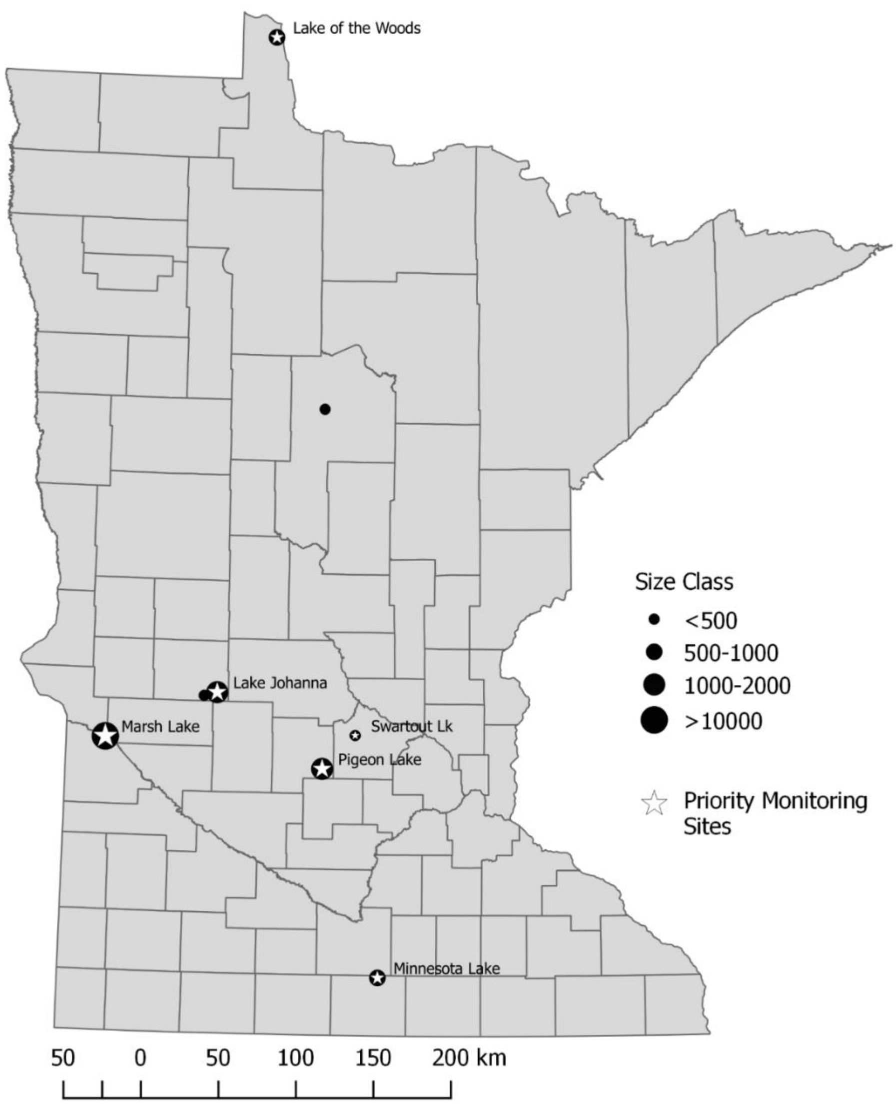
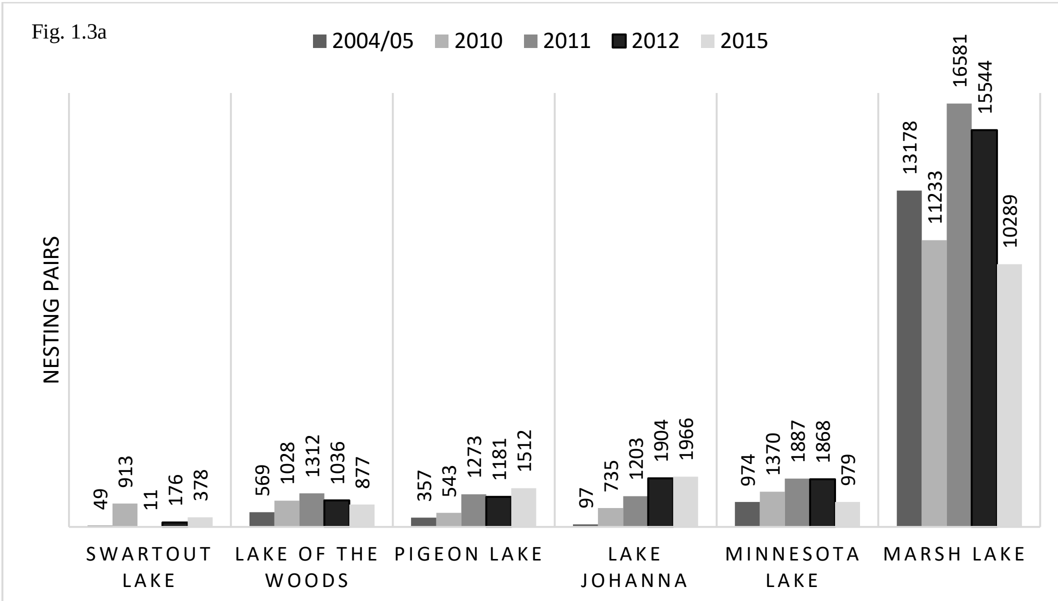


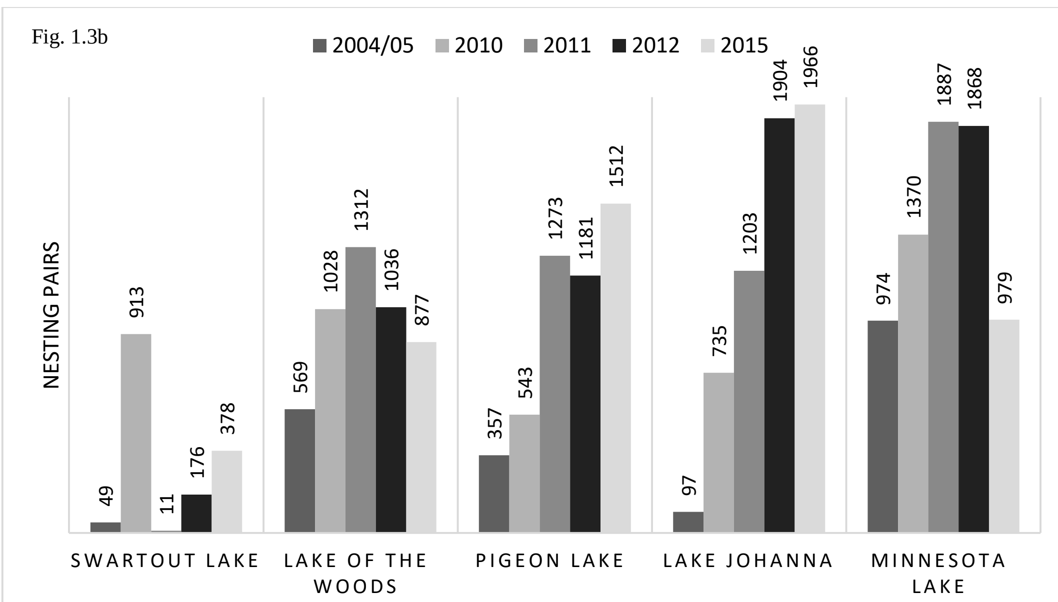
Figure 1.2: Minnesota American White Pelican colony distribution in 2015 by number of nesting pairs. Sites starred and labeled identify location of priority monitoring sites[1] with nesting AWPE. Sites censused Late April-Mid June using ground counts and aerial photography.



[1] "Priority Monitoring Sites" are sites with the greatest number of cormorant nests in 2015. These sites tend to exhibit high waterbird diversity and are often shared by wading birds and pelicans, which suggests that these are important sites for colonial waterbirds in general. The six priority sites with nesting pelicans also happen to support the greatest numbers of nesting pelicans.

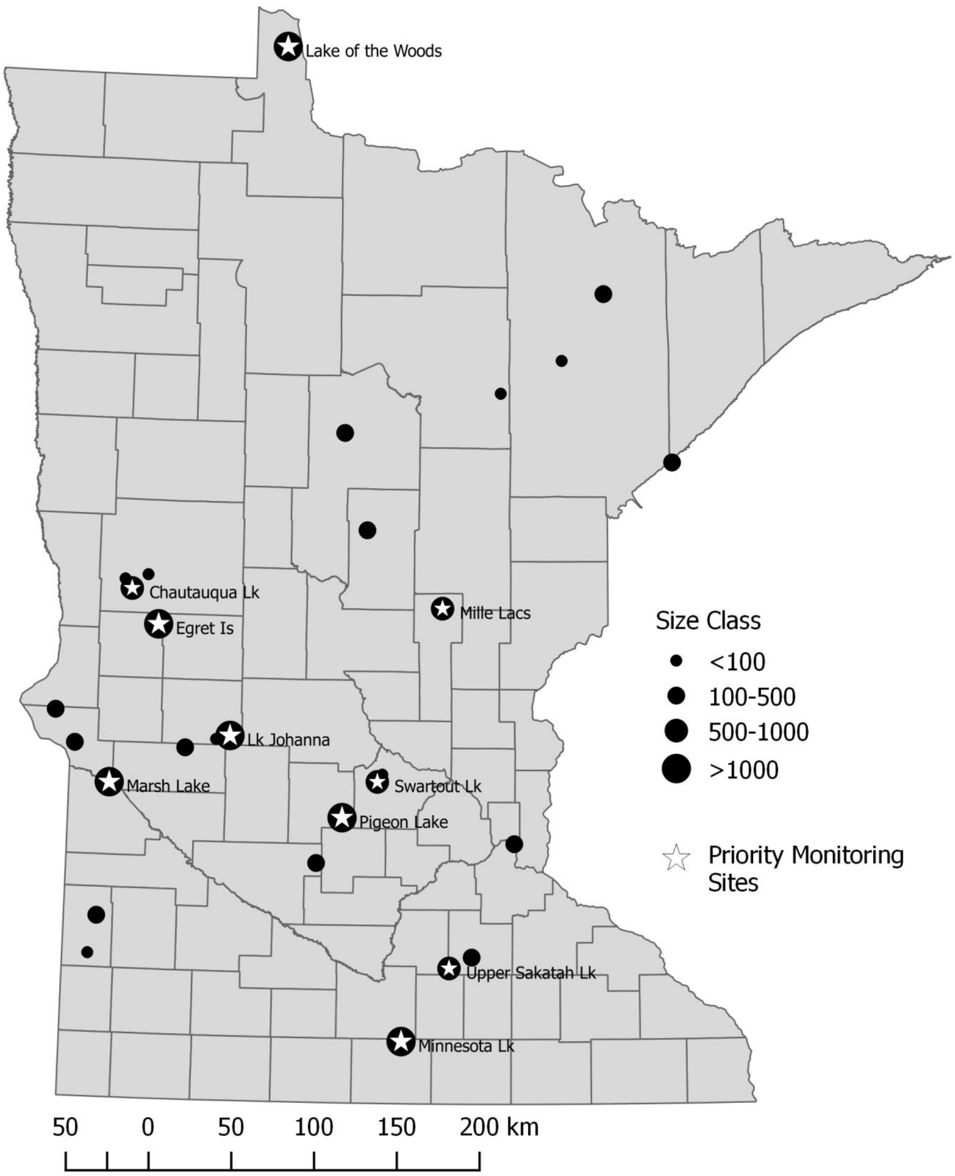
Figures 1.3a and 1.3b*: Number of American White Pelican nests at priority monitoring sites[1] in Minnesota, 2004-2015. *Figure values identical. Marsh Lake has been excluded from 1.3b for better visualization of yearly variation at smaller colonies.





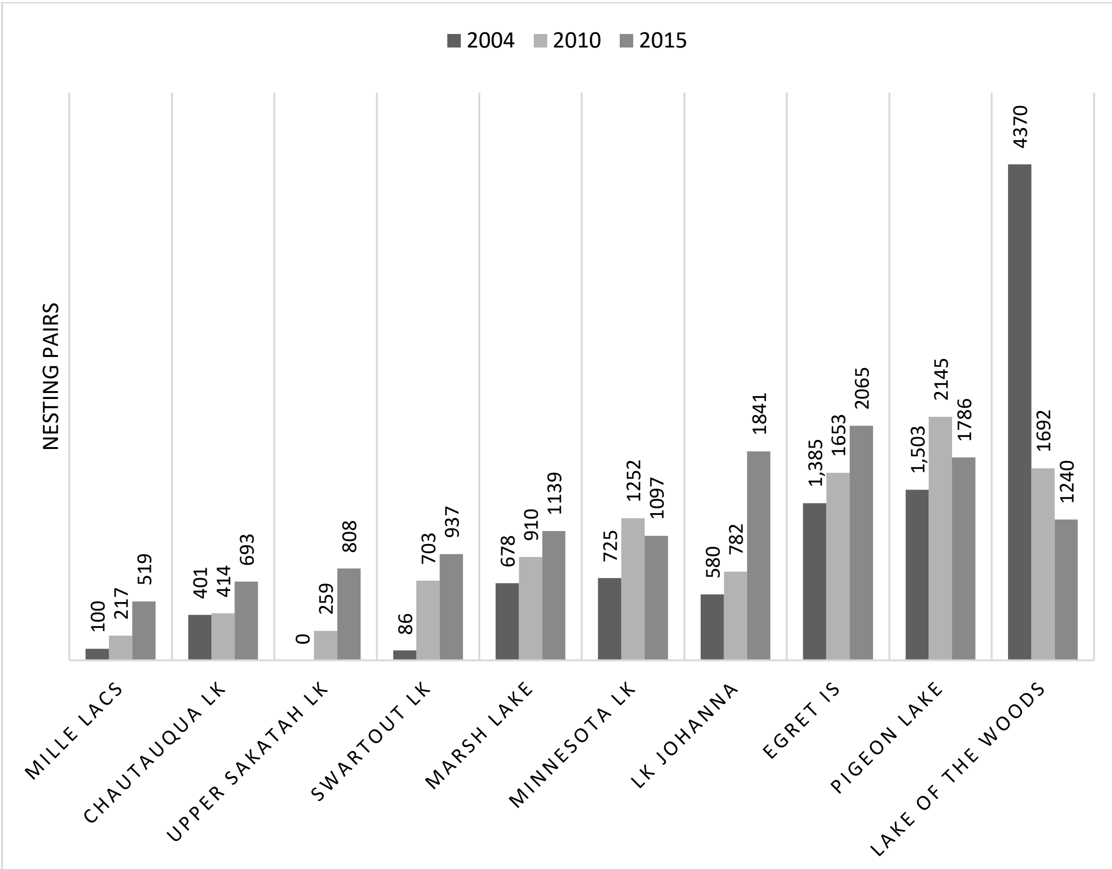
[1] "Priority Monitoring Sites" are sites with the greatest number of cormorant nests in 2015. These sites tend to exhibit high waterbird diversity and are often shared by wading birds and pelicans, which suggests that these are important sites for colonial waterbirds in general. The six priority sites with nesting pelicans also happen to support the greatest numbers of nesting pelicans.

Figure 1.4: Minnesota Double-crested Cormorant colony distribution in 2015 by number of nesting pairs. Lakes and islands starred and labeled are priority monitoring sites[1]. Sites censused Late April-Mid June using ground counts and aerial photography.



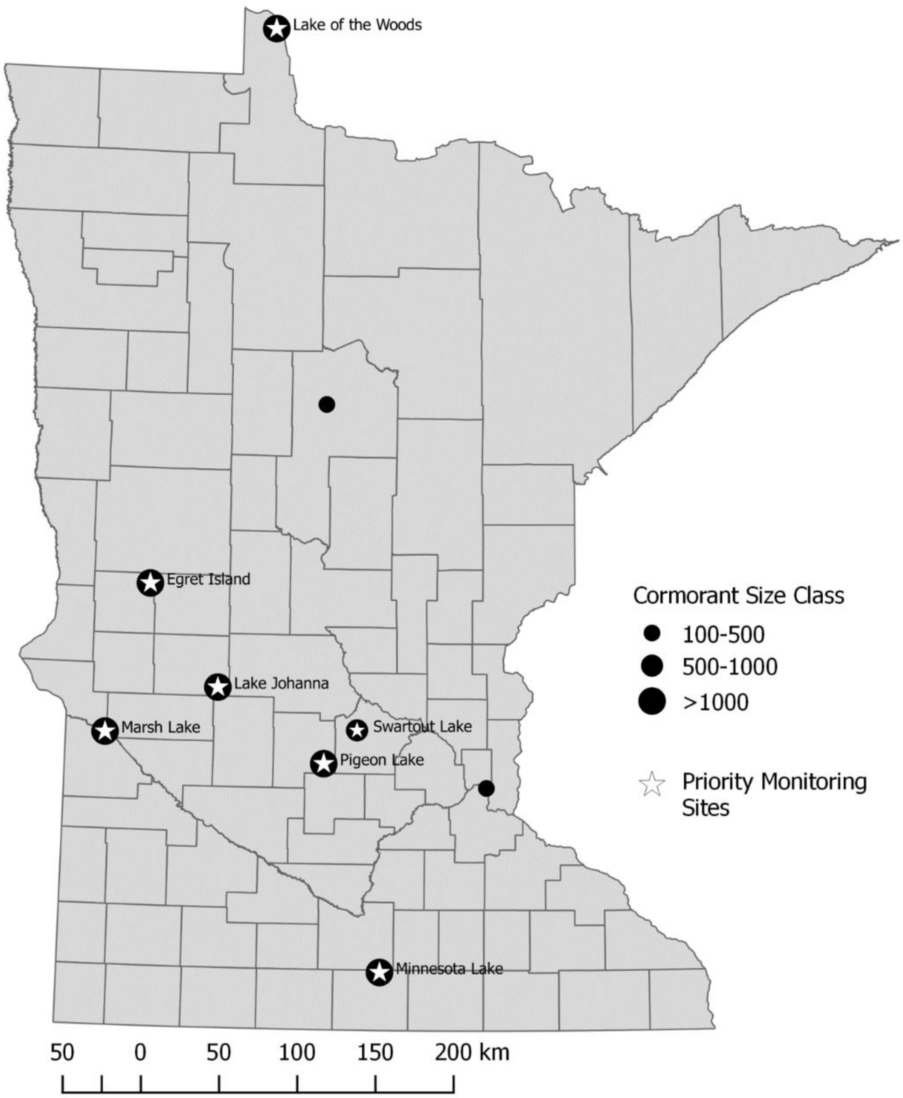
[1] "Priority Monitoring Sites" are sites with the greatest number of cormorant nests in 2015. These sites tend to exhibit high waterbird diversity and are often shared by wading birds and pelicans, which suggests that these are important sites for colonial waterbirds in general. The six priority sites with nesting pelicans also happen to support the greatest numbers of nesting pelicans.

Figure 1.5: Number of Double-Crested Cormorant nests at priority monitoring sites[1] in Minnesota, 2004-2015.



[1] "Priority Monitoring Sites" are sites with the greatest number of cormorant nests in 2015. These sites tend to exhibit high waterbird diversity and are often shared by wading birds and pelicans, which suggests that these are important sites for colonial waterbirds in general. The six priority sites with nesting pelicans also happen to support the greatest numbers of nesting pelicans.

Figure 1.6: Double-crested Cormorant sites with four or more colonial waterbird species in 2015 by number of nesting pairs. Lakes and islands starred and labeled are priority monitoring sites. Sites censused Late April-Mid June using ground counts and aerial photography.



[1] "Priority Monitoring Sites" are sites with the greatest number of cormorant nests in 2015. These sites tend to exhibit high waterbird diversity and are often shared by wading birds and pelicans, which suggests that these are important sites for colonial waterbirds in general. The six priority sites with nesting pelicans also happen to support the greatest numbers of nesting pelicans.

Figure 1.7: Projection of Marsh Lake nesting population based on density independent stochastic growth model.

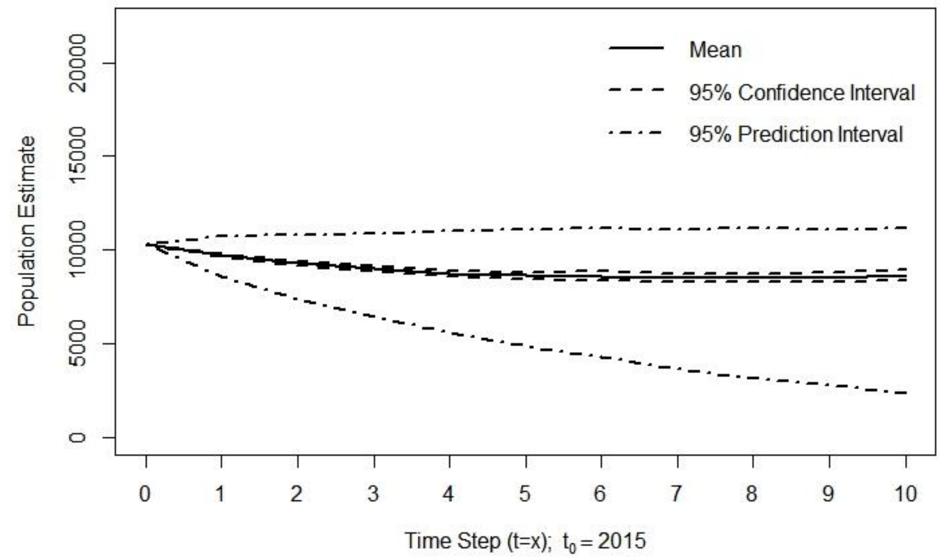


Figure 1.8: Projection of Marsh Lake nesting population based on Ricker density dependent stochastic growth model.

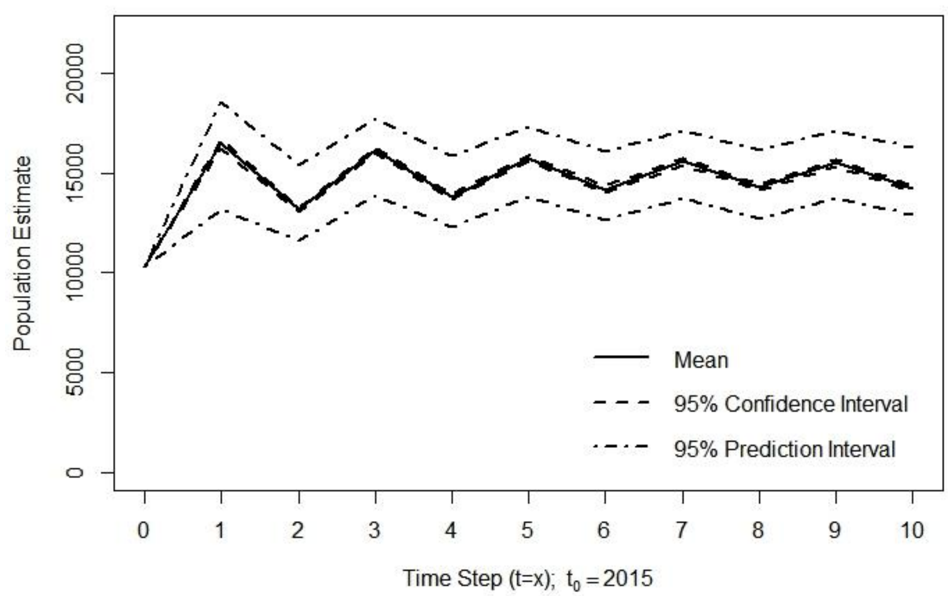


Figure 2.1: Minnesota American White Pelican active colony distribution in 2015 by number of nesting pairs.

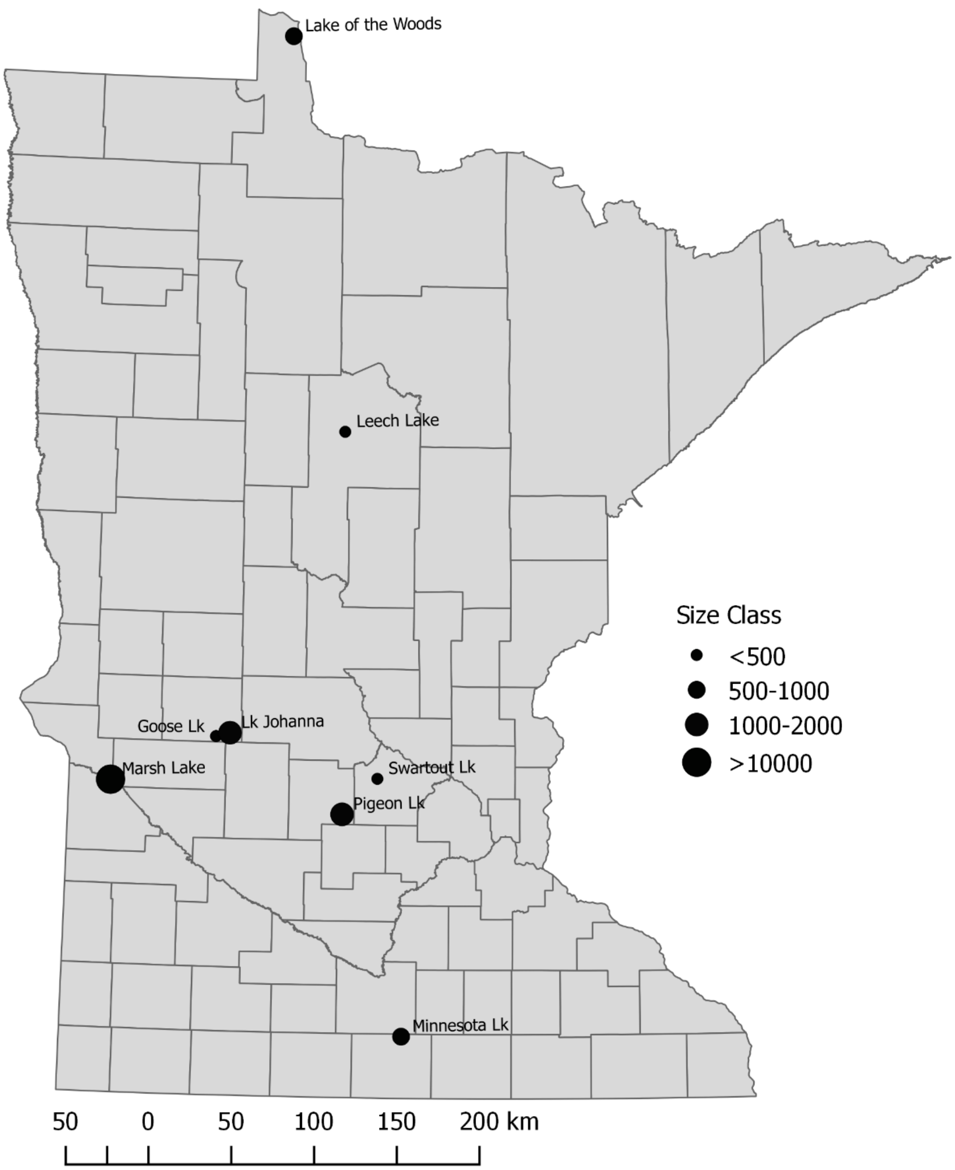


Figure 3.1: Lake Johanna, located in the southeast corner of Pope County, is one of several active pelican sites in Minnesota. The pelican colony resides on a large island on the south side of the lake.

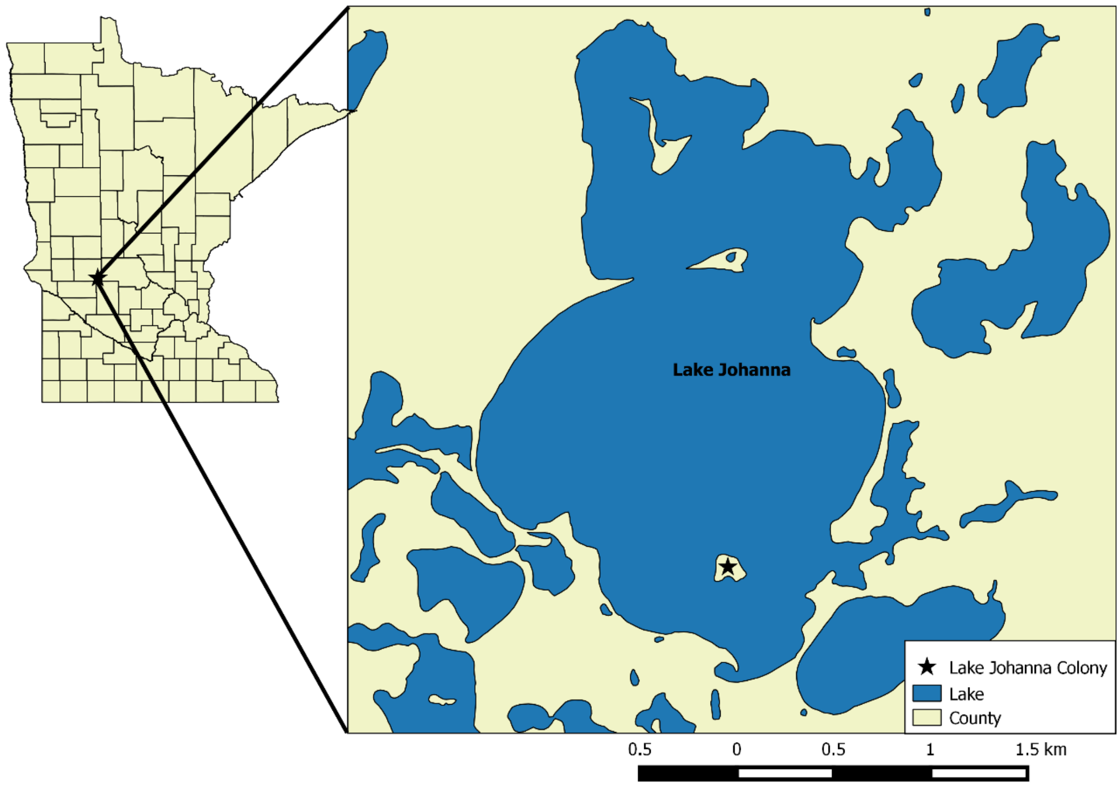


Figure 3.2: eCognition ruleset used for object-based classification DigitalGlobe imagery of Lake Johanna.

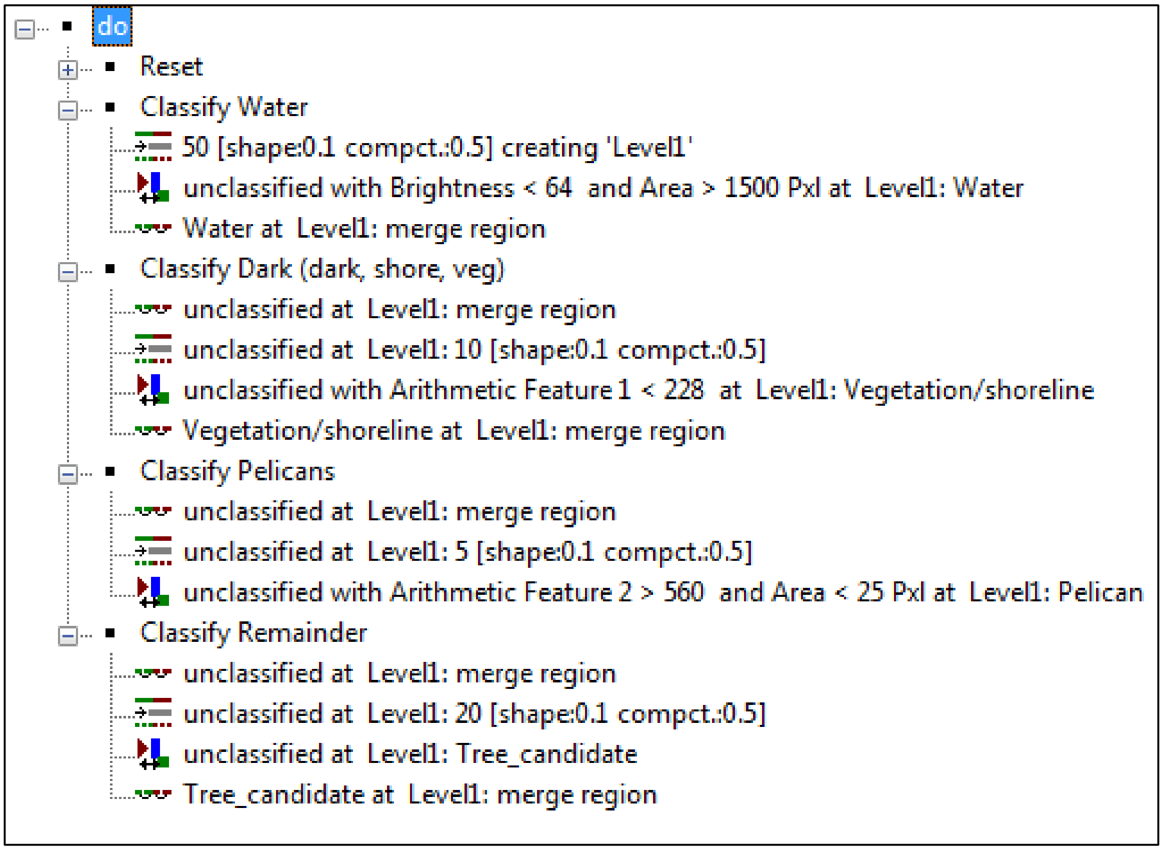


Figure 3.3: DigitalGlobe imagery of island on Lake Johanna pre- and post- object-based classification using eCognition software.

Figure 3.4: eCognition ruleset used for object-based classification of low-altitude aerial image of Lake Johanna.

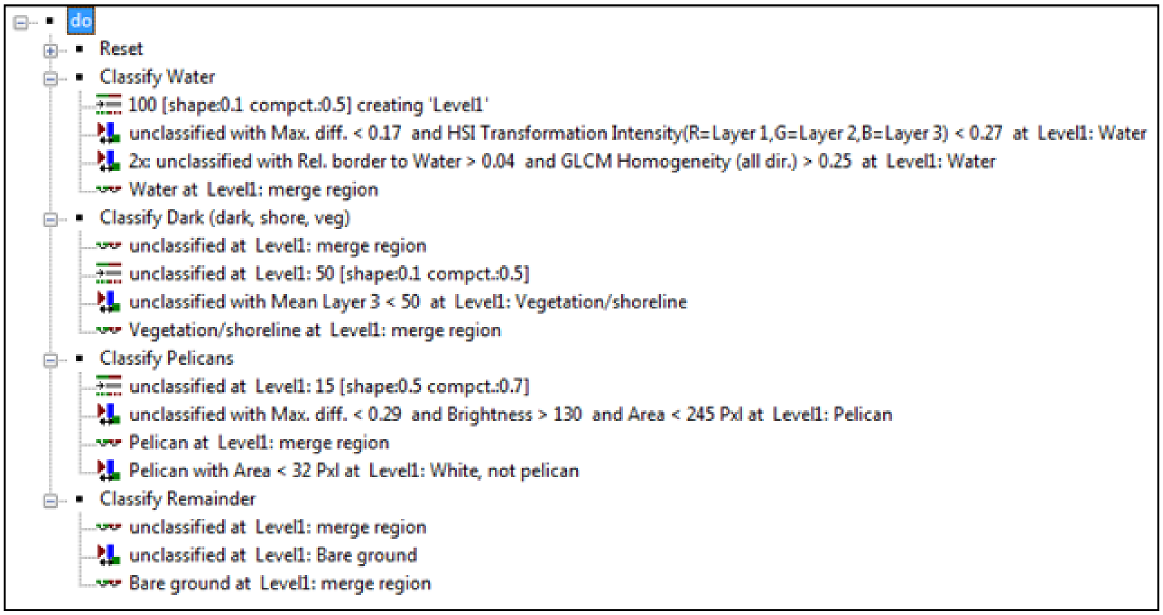
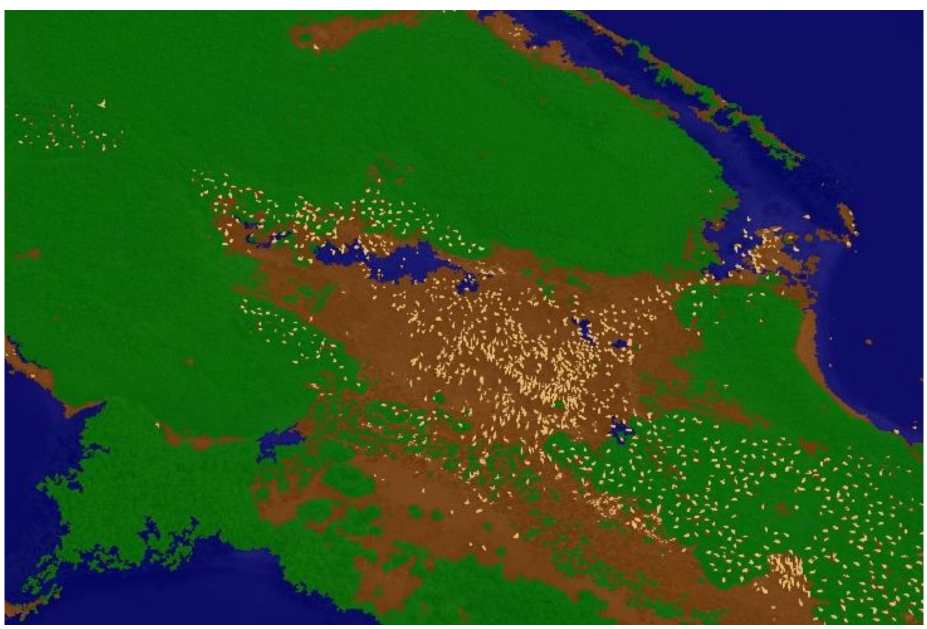


Figure 3.5: Low-altitude aerial image of island on Lake Johanna pre- and post- object-based classification using eCognition software.

Figure 3.6: Object-based classification of a portion of Big Island on Marsh Lake, Lac Qui Parle County, MN using eCognition software. *Classification uses same ruleset as that used to classify the Lake Johanna Photo without making site specific adjustments.



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