Importance of Demographics and Vital Rates to Cavity-Nesting Waterfowl in the Lower Saint John River of New Brunswick, Canada

By

Jared Sonnleitner

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Supervisor:	Joseph Nocera, PhD, Forestry and Environmental Management
Examining Board:	Al Hanson, PhD, Environment and Climate Change Canada
	Heather Major, PhD, Biological Science

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ABSTRACT

To assist management of cavity-nesting waterfowl in the lower Saint John River, New Brunswick, Canada area, I used historical nest box data, the Motus Wildlife tracking system, and band recovery data to create a population model using a population viability analysis (PVA). I discovered that reproduction (i.e., number of eggs hatched) was positively density-dependent for common goldeneye (*Bucephala clangula*), although no density-dependent effects were detected for wood ducks (*Aix sponsa*) or hooded mergansers (*Lophodytes cucullatus*). Survival estimates revealed that juvenile female wood duck survival was lower than that of any other age or sex class and appears to have decreased since the 1960's. Finally, the PVA I created revealed similar results to a less-detailed previous model, although percent of females breeding for wood ducks appears to be more important than previously thought. Management should focus on improving hatching success and age 0-1 survival through the proper provisioning of nest boxes and habitat enhancements.

DEDICATION

This thesis is dedicated to my Mom, Lisa Sonnleitner, Dad, Norm Sonnleitner, Grandad, Richard Godber, and Nan, Susan Godber, whose support and kindness has meant more than I can ever express.

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List of Symbols, Nomenclature or Abbreviations

AHY – After hatch-year

AIC – Akaike’s Information Criterion

AR – Autoregressive

ARIMA – Autoregressive integrated moving average

BBS – Breeding bird survey

BIC – Bayesian information criterion

CWS – Canadian Wildlife Service

DD – Density dependence

DUC – Ducks Unlimited Canada

F – Site fidelity

HY – Hatch-year

LSJR – Lower Saint John River

MA – Moving average

NAWMP – North American Waterfowl Management Plan

p – Recapture probability

PVA – Population viability analysis

r – Re-capture probability

S – Survival rate

USGS – United States Geological Survey

VHF – Very high frequency

ZINB – Zero inflated negative binomial

Chapter 1. General Introduction

Due to their economic/ecological importance and broad geographical range, waterfowl in North America are managed through a system of multijurisdictional collaborations to ensure populations and their habitats are protected (Baldassare and Bolen 2006, Brasher et al. 2019). Population and habitat goals have been accomplished since 1986 through the North American Waterfowl Management Plan (NAWMP) and associated Joint Ventures (JV), which aim to conserve waterfowl populations and habitats through partnerships between government, non-government, and private institutions (U.S. Department of the Interior and Environment Canada 1986, Brasher et al. 2019).

Cavity-nesting waterfowl, including wood ducks (*Aix sponsa*), common goldeneye (*Bucephala clangula*), and hooded merganser (*Lophodytes cucullatus*) are considered by the Eastern Habitat Joint Venture (EHJV) to be priority species for conservation in eastern Canada (EHJV 2017). In New Brunswick, cavity-nesting ducks are commonly found breeding in nest boxes or natural cavities throughout the lower Saint John River (Prince 1968, Johnsen et al. 1994). Tree (natural) cavities can be formed through mechanical processes such as wind, ice storms, decay, or through active processes such as by cavity-excavating birds (e.g., pileated woodpeckers (*Dryocopus pileatus*)) (Bellrose and Holm 1994, Denton et al. 2012). Nest boxes within New Brunswick are largely placed and maintained by Ducks Unlimited Canada, although other agencies such as Delta Waterfowl, and individual landowners place and maintain boxes in certain areas.

During the early twentieth century, wood duck populations declined throughout much of North America, and were thought to be nearly extinct (Bellrose 1976), likely due to high mortality from unregulated commercial harvesting of waterfowl (Bellrose 1976) and the destruction of old growth hardwood forests and floodplains used for breeding (Prince 1968, Bellrose 1976, Johnsen et al. 1994, Denton et al. 2012). Moratoriums on hunting, specific regulations governing daily limits, and the enhancement of breeding opportunities through the provisioning of nest boxes allowed for populations to recover (Bellrose 1976). The eastern population is estimated at >1 million (U.S. Fish and Wildlife Service 2022), and wood ducks are now one of the most hunted waterfowl species within the Atlantic Flyway (Bellrose and Holm 1994, Shirkey and Gates 2020). While most conservation actions (e.g., nest boxes) target wood duck populations, other cavity-nesting waterfowl such as common goldeneye and hooded merganser have benefited from these more abundant breeding sites (Heusmann et al. 2000).

An understanding of key vital rates (birth and death) and their influence on population growth is important to ensure the sustainability of populations (Mills and Lindberg 2002, Mills 2013, Hepp et al. 2020). For example, due to the abundance of grain crops grown during the latter half of the twentieth century, many regions have experienced increased snow geese (*Anser caerulescens*) populations. These agricultural landscapes improved conditions for increased survival and fitness at non-breeding areas, and in turn northern breeding grounds have experienced increased environmental degradation over time due to burgeoning snow geese populations (Robertson and Slack 1995, Abraham et al. 2005). Identifying specific life stages or attributes most likely to cause population reductions has since become a focus of waterfowl research (Rockwell et al. 1997, Aubrey et al. 2010). Conversely, for mallards (*Anas platyrhynchos*) where population increases in certain areas are desired, life stages and vital rates likely to increase population growth are of importance; female survival during the breeding season, nest success, and duckling survival were all found to be critical rates for population growth (Hoakman et al. 2002).

Female wood duck survival, nest success, and recruitment of females are all highly important for population growth (Hepp et al. 2020), while for common goldeneye, the local density of nest boxes is a large contributor to recruitment (Lawson et al. 2017). Harding (2021) evaluated the influence of artificial and natural cavities on growth rates of cavity-nesting ducks in New Brunswick using a population viability analysis (PVA) and found that age 0 (hereafter referred to as "hatch year") survival was more important than nest site abundance for all three species. While results from Harding (2021) provided an understanding of the interactions among vital rates, due to limited data, some estimates were imprecise. For instance, while hatch year survival was determined to be important, estimates have not been established for any of the three cavity-nesting waterfowl species in the Maritimes. Estimates for New Brunswick in particular are important, as these species are listed as priorities under the EHJV implementation plan (EHJV 2017). Additionally, there remains mixed support for density dependence in the nesting of waterfowl (Gunnarson et al. 2012), which may have substantial impacts on cavity-nesting waterfowl whose reproduction is largely managed through the provisioning of nest boxes. While Harding's (2021) PVA suggested nest site abundance was of lesser importance than hatch-year survival, density dependent effects were not included within this model, which could affect predicted outcomes (Brook et al. 1997, Beissinger and Westphal 1998).

Effective conservation of cavity-nesting waterfowl in New Brunswick requires: 1) data deficiencies in the most recent PVA be reduced; 2) identification of the most limiting life stages for survival of cavity nesting waterfowl, and; 3) improved understanding of regional nesting ecology. This thesis pursues these three objectives and will result in more precise population models, suggestions for future conservation and management, and improved understanding of these species' natural history.

Target Species

Wood ducks are dabbling ducks found in diverse habitats, ranging from swamps and marshes to creeks and river systems (Bellrose and Holm 1994). There appears to be no preference for certain habitats by different age classes or sexes, although slight seasonal differences exist with open water habitat being avoided later in the year (Thompson and Baldassare 1988). Most wood ducks form pair bonds from fall to winter or early spring, usually once they have gained their nuptial plumage (Armbruster 1982, Heitmeyer and Frederickson 1990). Egg laying begins as early as mid-April and may be initiated later than mid-May (Haramis and Thompson 1985). Once a female has finished laying eggs, she begins incubation which lasts an average of 30 days (Haramis 1990).

Common goldeneye are diving ducks distributed from Canada and the United States to Russia and parts of China, often inhabiting large lakes or wetlands (Eadie et al. 2020). Habitat selection at various stages is tied to specific food requirements, for instance, many females will move broods from nesting to brood-rearing lakes shortly after hatching (Wayland and McNicol 1994). Maturation is often delayed and breeding is not attempted until their second year (Eadie and Gauthier 1985), where they will form a pair bond in the late winter or early spring (Snow and Perrins 1998). Females begin laying eggs between mid April – mid May, producing clutches of 9.8 eggs on average (SénéChal et al. 2008). Incubation lengths vary, although most have been found to last around 30 days (Mallory and Weatherhead 1993, Eadie et al. 2020).

Hooded merganser, another diving duck species, is often found nesting in similar cavities and nest boxes to wood ducks and common goldeneyes, although in lower densities (Prince 1968, SénéChal et al. 2008). As with common goldeneye, maturation is often delayed and breeding is not attempted until their second year (Morse et al. 1969). Females begin laying eggs between mid April - mid May, although they produce slightly fewer eggs (9.2) on average than common goldeneye (SénéChal et al. 2008). In a different nest box study where female experience was known, Morse et al. (1969) determined that first time breeders had slightly smaller clutch sizes (9.4 eggs) when compared to experienced breeders (10.8 eggs).

Nest Box Management

Nest boxes for waterfowl are built from a variety of materials such as wood, plastic, and metal (Bellrose and Holm 1994). Nests box usage rates vary due to regional differences in resources, predators, and nest densities (Jones and Leopold 1967, Haramis and Thompson 1985, Soulliere 1986, Pöysä and Pöysä 2002), although maintenance and placement of boxes can also be important factors (Lacki et al. 1987, Utsey and Hepp 1997). For example, boxes placed higher and closer to water often have improved hatching success (Bellrose et al. 1964, Gilmer et al. 1978). Habitat type can be another determinant of usage rates by waterfowl. For instance, within South Carolina, boxes placed in open swamps had the highest usage by wood ducks, followed by marshes, impoundments, and riverine systems (Bellrose and Holm 1994). Densities of boxes in an area can influence reproductive success by increasing rates of nest parasitism and depredation (Haramis and Thompson 1985, Bellrose and Holm 1994).

Hatching success and clutch sizes are often higher among individuals nesting earlier in the season when compared to those nesting later (Dow and Fredga 1984, Hepp and Kennamer 1993). Experience is often a strong determinant of nest success and box usage. For instance, inexperienced breeding buffleheads (*Bucephala albeola*) have lower hatching success and clutch sizes than experienced breeders (Gauthier 1989). Further, inexperienced breeding buffleheads show higher use of nest boxes, where they experience lower reproductive rates compared to natural cavities (Gauthier 1989). Predation is one of the main reasons for nest failure in artificial boxes as shown in central Illinois, where between 1938-1940 and 1958-1961 predation accounted for 65% and 87% of wood duck nest failures, respectively (Bellrose et al. 1964). The use of metal poles and predator guards are two common methods to reduce predation by raccoons (*Procyon lotor*; Bellrose and Holm 1994, Bailey and Bonter 2017).

Survival

Hunting is one of the highest mortality factors for many waterfowl species in North America (Baldassare and Bolen 2006). However, rates of non-hunting mortality and their sources are far more difficult to study and quantify. In Missouri, of 675 waterfowl carcasses, 18% had no definitive causes of death, and predation was often difficult or impossible to identify (Humburg et al. 1983, Baldassare and Bolen 2006).

Hen and duckling survival have been studied extensively; often hen survival has been found to be high (0.90-0.92) during the brood rearing period (Davis et al. 2001, Dyson et al. 2018), while duckling survival to the flight stage is often low (Heusmann 1972, Bellrose and Holm 1994, Davis et al. 2001, Dyson et al. 2018). One life history period that has rarely been examined is that of juvenile birds (fully flighted, but <1 year old). Birds of this stage may experience high hunting-related mortality (Reed and Boyd 1974, Longcore et al. 1991, Parker 1991), and will have increased risk during their first migratory flight due to inexperience. For instance, in whooping cranes (*Grus americana*), 1 year old birds migrating with older individuals deviated less from the most optimal migratory route when compared to those migrating without older more experienced birds (Mueller et al. 2013). In northern gannets (*Morus bassanus*), one third of juveniles undergoing their first migratory flights were found to have died shortly after leaving the colony, likely due to inexperience with threats encountered during migration (Lane et al. 2021).

Population Viability Analysis

Population viability analysis (PVA) provides the opportunity to examine the relationship(s) among various life history and ecological parameters that allow for long-term population projections. PVA's use simulation-based models to predict thresholds for extinction risk. One common critique regarding the use of PVAs is that datasets used for model construction are often incomplete and contain large uncertainties in estimating demographic parameters (Hamilton and Moller 1995, Beissinger and Westphal 1998, Beissinger and McCullough 2002, Ellner et al. 2002, Ellner and Fieberg 2003). Due to these concerns, a second, and arguably more robust use for these models has been the comparison of management actions for target species (Beissinger and Westphal 1998, Ellner et al. 2002, Lacy 2018). For instance, models for the critically endangered northern spotted owl (*Strix occidentalis caurina*) predicted how protecting critical habitat would influence their long-term survival and revealed that >21% of habitat needs to be old growth forest for a population to persist (Lande 1988). For cavity-nesting ducks (which are relatively abundant and harvested recreationally), additional data are available through Breeding Bird Surveys (BBS), band recoveries, government organizations (e.g., Environment and Climate Change Canada, U.S Fish and Wildlife Service), and the literature to meet most needs of this type of model. Therefore, by using a PVA to compare management actions rather than extinction risk, many critiques of classic PVAs can be avoided.

Research Objectives

I seek to address two main issues outlined from previous work on cavity-nesting ducks in New Brunswick which are: 1) nest boxe availability does not appear to be limiting to population growth, yet little is known about the dynamic processes (e.g., density dependence) regulating this system, and; 2) survival of juveniles appears to be important for population growth of wood ducks, although no local survival estimates currently exist. I will therefore re-construct previously created population viability models (Harding 2021) and conduct a sensitivity test with these updated model parameters. The results of this work will improve our understanding of cavity-nesting duck ecology and population dynamics within the LSJR. This information will be useful in meeting the population goals outlined in the 2012 NAWMP and better inform Ducks Unlimited Canada, the EHJV, and other relevant partners on how to best manage these species.

Overview and Design of Thesis

1) I hypothesized that nest box availability is limiting to cavity-nesting duck populations. If that is true, density dependence should be observable in nest use and hatching success out of these boxes. These patterns are likely to be more pronounced within diving ducks as has been shown by previous studies.
2) I hypothesized that inexperience with hunters and migration induces increased costs and mortality for juvenile wood ducks during migration (the hunting season).
3) I hypothesized that my PVA would produce similar results to Harding's (2021) PVA (which used only literature-derived values), although with increased precision because I use locally-measured values.

My hypotheses yield the following testable predictions:

1) If nesting in boxes is regulated by density dependent processes for common goldeneyes and hooded mergansers, then I expect usage rates and hatching success would be autocorrelated over time and have opposing trends.
2) If juvenile wood ducks have higher risks when compared to adults as the result of hunting or migration, then their survival rates will be lower over this period.

3) If the results from the PVA are the same but with more precise parameter estimates, then we should be able to make more confident management recommendations based on the results.

Thesis Structure

In chapter 2, I use 36 years of nesting data on wood ducks, common goldeneyes, and hooded mergansers from nest boxes to test for density dependent regulation. Using results from an ARIMA model, I discuss the implications of my findings on the future of nest box management within New Brunswick, Canada.

In chapter 3, I use the MOTUS automated telemetry system and 63-years of band-recovery data to estimate survival of wood ducks in the Atlantic flyway. I discuss how different cohorts may differ in their ability to migrate and respond to mortality factors such as hunting.

In chapter 4, I re-constructed a previous population viability analysis using updated information from chapters 2 and 3. I discuss that improved parameter estimates increased model precision, which facilitates some of my regional management recommendations.

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Chapter 2. Exploring density dependence in cavity-nesting ducks: Insights from 36 years of nesting data

Abstract

Nest boxes for cavity-nesting waterfowl have proven highly effective in supplementing habitat with low natural cavity availability. Nest box management can be costly due to labour and materials to maintain boxes; understanding nest box efficacy provides managers with information for planning future expenditures. As an exemplar, I examined the efficacy of a nest box program in the lower Saint John River (LSJR; New Brunswick, Canada). I used 36 years of hatching and box usage data to better understand direct and delayed density dependence in cavity-nesting waterfowl. Wood duck (*Aix sponsa*) numbers in New Brunswick have been increasing since the 1970's, while common goldeneye (*Bucephala clangula*) and hooded merganser (*Lophodytes cucullatus*) populations have seen little or no growth, indicating a population nearing carrying capacity for which the addition of nest boxes may provide few conservation benefits if nest sites are not a limiting factor. To examine density dependent patterns in these three species, I used an autoregressive integrated moving average model which allows for the detection of autoregressive events occurring at different lags. I found that common goldeneyes, but not wood ducks or hooded mergansers, demonstrated density dependence in their nesting success over the last 36 years. Increasing the number of nest boxes in the LSJR would have positive effects on local populations of common goldeneye, but less so for wood ducks or hooded mergansers.

Introduction

Density dependence (DD), as first described by Nicholson (1933), is a process in which a population increases in density, triggering per capita growth rates to decrease, and thus setting limitations on population size. These limitations experienced by wild populations are important for understanding trends in life history traits which influence conservation and management decisions (Eikeset et al. 2016). Knowing whether a species' survival, dispersal, and/or reproduction is density dependent can be important information in wildlife management (Fuller et al. 2007; Lawson et al. 2016), as DD can lead to population regulation or destabilization.

Many wildlife and plant species tend to experience decreased reproductive success at high population densities due to increased competition or lower resource quality (Kunin 1997; Norris et al. 2004). One taxonomic group that seems to show inconsistent patterns of DD is waterfowl; studies examining the influence of breeding densities on reproductive success of waterfowl have found little consensus (Kaminski and Gluessing 1987; Viljugrein et al. 2005; Gunnarsson et al. 2012; Nummi et al. 2015). In a review of DD literature for waterfowl, Gunnarsson et al. (2012) found that of 30 studies, 15 identified DD processes occurring while another 15 did not. These differences could be the result of studies examining DD at different spatial scales, or a lack of methodological consistency among studies (Gunnarsson et al. 2012). More studies found support for DD while examining populations at a landscape scale rather than smaller spatial scales (e.g., wetland). Gunnarsson et al. (2012) also noted some differences when comparing different groups of ducks; diving ducks had higher rates of

DD compared to dabbling ducks, likely due to higher stability in their breeding habitats (Vickery and Nudds 1984; Johnson and Grier 1988).

Rates of DD influencing the reproductive success of cavity-nesting ducks may be higher than for other groups of waterfowl, possibly due to limitations in nest site availability resulting from historic habitat destruction and the loss of many natural cavities (Prince 1968, Johnsen et al. 1994, Denton et al. 2012). Some of this habitat loss can be offset by providing artificial nest boxes (Bellrose 1976). Support for DD during reproduction in waterfowl using nest boxes has been found at the wetland scale in Sweden (Pöysä and Pöysä 2002), where lakes were monitored for a four-year period before nest box installation and five-years after to examine changes in the number of common goldeneye (*Bucephala clangula*) breeding pairs and ducklings. The number of breeding pairs increased after nest box installation, but duckling abundance did not (Pöysä and Pöysä 2002). In wood ducks (*Aix sponsa*), increases in the number of nest boxes on a wetland resulted in lower nesting success over time (Haramis and Thompson 1985). In both cases, evidence supports DD acting on reproductive success, although how these processes operate at the landscape level has yet to be identified. This may be an important process to examine further as the causes of nest failure seem to differ at different spatial scales. For instance, the effects of nest parasitism are often apparent at the wetland level (Haramis and Thompson 1985; Eadie et al. 1988; Semel et al. 1988), while effects from predation tend to be most easily observed at the population and landscape levels (Schmidt and Whelan 1999; Gunnarsson et al. 2012).

Identification of DD could be difficult if time lags are ignored (i.e., second or third order density dependence) (Hassell et al. 1989; Turchin 1990). Among diving ducks, immature birds often delay reproduction until their second or third spring, as compared to dabbling ducks that often reproduce in their first spring (Zicus and Hennes 1989; Baldassare and Bolen 2006). For diving ducks, the effects of high densities on reproduction may be delayed (i.e., lagged) which could explain some variation in detection rates among studies. Examining population data, Turchin (1990) detected delayed DD in seven of nine species of invertebrates previously thought to be density-independent, demonstrating that when attempting to identify DD, models could benefit from including lagged effects. Delayed density dependence can occur when negative reproductive consequences resulting from increased densities are not detectable during the following breeding season or reproductive attempt of that species. It may however be detectable during subsequent breeding seasons or reproductive attempts. There has yet to be support for delayed DD among waterfowl (Viljugrein et al. 2005), although no studies have examined this among cavity-nesting ducks (which is especially surprising considering that breeding density can be easily determined from provisioned nest boxes).

Within the lower Saint John River floodplain of New Brunswick, Canada, cavity nesting ducks have been managed through nest box programs due to historically declining natural cavity abundance (Prince 1968; Johnsen et al. 1994). Despite monitoring programs for over 30 years, few efforts have been made to understand the dynamic processes underlying this nest box program and any benefit to local populations. Here, examined 36 years (1987-2022) of nest box data to outline possible trends in both the usage and reproductive success of wood ducks, common goldeneyes, and hooded mergansers (*Lophodytes cucullatus*). The use of long-term data such as those used here, is often required for statistical models to accurately detect DD processes acting on a population (Hassell et al. 1989; Woiwod and Hanski 1992; Hanski et al.

1993). I therefore examine changes in the usage and hatching success in nest boxes spanning a relatively large spatial and temporal scale using an autoregressive integrated moving average model (ARIMA) which has proven useful in detecting temporal patterns in population regulation (Brigatti et al. 2016). The use of time series analysis has been used for the detection in DD in waterfowl species (Vickery and Nudds 1984; Viljugrein et al. 2005), and the use of an ARIMA model with autoregressive component allows for the detection of time lags such as second or third-order DD.

Since 1961, eastern wood duck populations have been increasing and in 2020 were 110% higher than the long-term average (Canadian Wildlife Service (CWS) 2022) indicating a population not severely limited in their reproduction. Despite conditions at the landscape level likely differing from those at the flyway level (the spatial scale at which wood duck population size is estimated by federal managers), I predict that due to the increasing population size of wood ducks, that their reproduction has not been DD in the last 36 years. However, I predict that delayed DD may be detected in both common goldeneyes and hooded mergansers over the last 36 years, due to higher stability in their breeding habitats and delayed maturity.

Methods

Study Site

The lower Saint John River (LSJR) floodplain is a complex of wetlands including the Grand Lake meadows and Portobello Creek (Rideout et al. 2021). Historically these areas were dominated by silver maple (*Acer saccharinum*) and American elm (*Ulmus americana*), which could tolerate the mesic conditions and provide a substantial number of cavities for waterfowl and other species (Prince 1968, Johnsen et al. 1994). However, since the 1970's many areas within New Brunswick have lost American elm or have remnant populations that are of considerably smaller age classes. Additionally, silver maple stands have decreased within the province during this period (Johnsen et al. 1994, Loo et al. 2007). Water levels at these sites increase through flooding in the spring, summer, and fall as the result of freshets. This creates habitat for a variety of cavity nesting waterfowl such as wood ducks, common goldeneye, and hooded mergansers which are often characterized by their use of floodplain forest habitat (Prince 1968).

Nest box checks

I inspected nest boxes from 15 May-21 June in 2021 (n = 157) and 17 April-7 July in 2022 (n = 73). I assessed occupancy by determining if a female was present, eggs were present, or down was discovered inside the nest box. By examining egg size and color, I determined the species nesting in each box. I identified interspecific egg dumping when eggs from different species were found in the same box. Intraspecific egg dumping was only occasionally identifiable by super-numerary clutches. I checked nests once during the egg-laying stage to determine occupancy and which species was present in each box. I then checked eggs approximately once every week to ensure the nest had not been abandoned or depredated, until hatching. I identified successfully hatched clutches based on the presence of egg caps and eggs liners; I also used the presence of egg liners to discern between a clutch that had successfully hatched versus one that had been depredated.

Nest Box Data Management

Nesting data from 1987-2022 were gathered from three sources: a volunteer dataset spanning 1987-2018, from Harding (2021; *unpublished dataset*) spanning 2019-2020, and data I collected from 2021-2022. I calculated proportional nest box use by each species by taking the number of boxes occupied by either a common goldeneye, wood duck, or hooded merganser and dividing that by the total number of occupied boxes each year. I calculated hatching success for each species by dividing the number of successful clutches by the total number of occupied boxes of each species. The fates of a few boxes were unknown and therefore I did not include them when calculating hatching success, leading to some discrepancies between sample sizes when calculating nest box use versus hatching success. For both nest box use and hatching success, I excluded boxes containing intraspecific dump nests from both calculations as intraspecific parasitism is often difficult to identify.

Testing for correlation

To examine the relationship between hatching success and nest box usage from the 1987-2022 datasets, I ran a simple linear model using the statistical software R (v 4.2.1) to determine if there was an underlying relationship. I then tested the residuals for both the hatching success and usage models for normality. If the residuals were not normally distributed, I followed up by using generalized linear modeling. For hooded mergansers, residuals were not normally distributed, and both usage and hatching success datasets were zero-inflated. I therefore used a zero-inflated negative binomial (ZINB) regression using R package pscl (v 1.5.5, Zeileis et al. 2008) to account for inflated counts of zeroes. For a ZINB, this typically means using both a count component and a Bernoulli distribution (point mass at zero) (Zeileis et al. 2008). If there appeared to be no relationship between hatching success and the usage of boxes (i.e., they are independent of each other), then those species were candidates for an ARIMA model to assess trends over time for each of the variables separately.

ARIMA model

An ARIMA model is a mixture of three terms: an auto-regressive (AR), differencing, and moving average (MA), often denoted in the same order as p, d, q. Unlike the original ARMA model developed by Box and Jenkins (1970), an ARIMA model includes the differencing (integration) term, to meet the assumption of stationarity when data may be non-stationary. For instance, with nest box data, trends in usage and hatching success over time are likely, therefore violating assumptions of stationarity. In this case, I found that a single integration term was sufficient to meet the assumptions of stationarity within the data. The AR process can be used to describe the effects of previous years' breeding on that of the current year and will allow for the detection of lagged effects within the data (Ives et al. 2010). The detection of lagged effects is important for common goldeneye and hooded merganser as they do not begin breeding until their second or third year (Morse et al. 1969, Eadie and Gauthier 1985). I used MA terms to assess the level of 'noise' present in the data and how that may influence the detection of DD. A random process (i.e., contains no AR/MA components) would take the form (0,0,0) or (0,1,0) (Hamblin et al. 2010), indicating a lack of temporal structure to nesting and therefore no DD. Models containing MA, but no AR terms would take the form (0,0,1) or (0,1,1) and further examinations of coefficients would need to be calculated to assess if past variation is influencing the detection of DD by the AR term. For this analysis, I used the R package forecast (v 8.20, Hyndman and Khandakar 2008) to run the ARIMA models. I ran models with p, q components ranging from 1-3 (total of 16 models) and used Bayesian information criterion (BIC) to select the best fitting model. BIC was used due to its generalizability and low number of parameters used within this model.

Results

In analyzing the 36 years of data (n = 3068 nest boxes), I found that common goldeneyes were the most common species nesting in boxes (they were found using 45% of boxes in the study area, on average) and had a mean hatching success rate of 63%. Wood ducks were the second most common species found to be nesting in boxes, using on average 8% of boxes and had a mean hatching success rate of 85%. Hooded mergansers were found the least often to be nesting in these boxes, occupying just 4% of boxes on average with a mean hatching rate of 76% (Table 2.1).

For each species, the relationship between hatching success and rate of box usage was modeled to determine the strength of the relationship. For common goldeneyes there was a statistically significant relationship between hatching success and usage of nest boxes ($r^2 = 0.09$, $p = 0.04$), although the low r^2 value indicates little predictive power. Conversely, hatching success and usage of nest boxesby wood ducks ($r^2 = 0.02$, $p = 0.2$) and dump nests ($r^2 = -0.01$, $p = 0.42$) were not related. Residuals were tested for normality and were found to be normally distributed for common goldeneyes ($p = 0.8$), wood ducks ($p = 0.07$), and dump nests ($p = 0.15$). For hooded mergansers, residuals were not normally distributed ($p = 0.002$) and due to the zero-inflated nature of their hatching success and one-inflated nature of their usage, I used a ZINB model to analyze the relationship. Use of nest boxes had no influence on hatching success (z value = -1.312, $p = 0.19$) when examining the count model and the zero inflated model indicated that zero hatching success did not influence the usage of nest boxes (z value = 0.403, $p = 0.69$).

Due to the lack of relationship between hatching success and the use of nest boxes for wood ducks, hooded mergansers and a weak relationship for common goldeneye, I used an ARIMA model to evaluate the trends in each of these components from 1987-2022. For common goldeneye use of nest boxes, the best fitting model was (1,1,0), which indicates a single AR term with a coefficient of AR1 $\theta = -0.38$ from 1987-2022 (Figure 2.1A.). For hatching success of common goldeneyes over the same period the best fitting model was (1,1,0) with a single AR term with a coefficient of AR1 $\theta = -0.58$ from 1987-2022 (Figure 2.1B.). For wood ducks, the best fitting model for both use and hatching success was (0,1,1) indicating no AR terms, but a single MA coefficient of MA1 $\theta = -0.99$ for usage from 1987-2022 (Figure 2.2A.), and MA1 $\theta = -0.96$ for hatching success from 1987-2022 (Figure 2.2B.). Hooded mergansers use and hatching success was similar to that of wood ducks with the best fitting model being (0,1,1) indicating again no AR but a single MA term with a coefficient of MA1 $\theta = -0.93$ for usage from 1987-2022 (Figure 2.3A.) and MA1 $\theta =-0.85$ for hatching success from 1987-2022 (Figure 2.3B.). The best fitting models for common goldeneye dump nest usage was (0,1,1), the singe MA term had a coefficient of MA1 $\theta =-0.83$. For hatching success of common goldeneye dump nests the best fitting model was a (2,1,0) model although the (0,1,1) model was within 2 delta BIC values and was therefore chosen due to the lower number of parameters (n=2 vs. n=3). The single MA term of the (0,1,1) model had a coefficient of MA1 $\theta = -0.54$.

Discussion

I found support for the reproduction of common goldeneyes, but not wood ducks or hooded mergansers, being DD during 1987-2022. This supports my prediction that the reproduction of common goldeneyes, but not wood ducks would be DD. It should be noted that unlike the more common negative feedback density dependent system, I found that both reproductive success and usage rates appear to be declining over the 36-year period, and is therefore positive feedback DD. My analyses did not provide any support of the prediction that reproduction of hooded mergansers would be negatively DD. Nesting appearing to have DD patterns for common goldeneyes is similar to what was found by Nummi et al. (2015) who examined DD between common goldeneye and another dabbling duck species, the Eurasian teal (*Anas crecca*). Brood numbers were compared to breeding pair density and food availability at a 39 km^2 study area in the boreal region of Sweden, where common goldeneyes have higher predictability of lake use than Eurasian teal and displayed more pronounced patterns of DD. Although in contrast to what was found here, this study in the boreal region found that as usage increased, reproductive success decreased. Similar to Eurasian teal, wood ducks are a ‘fast’ species characterized by chaotic population trends and therefore provides support for differences in the strength of DD between diving and dabbling ducks (Nummi et al. 2015).

Lack of DD in hooded mergansers may be due to their relatively low usage of nest boxes in eastern North America (Savard and Robert 2007). Mean usage of nest boxes in my study area by hooded mergansers was only 4% and therefore breeding populations were likely too widely spatially distributed to be under DD in nest boxes. However, it should be noted that models for both hooded merganser and wood ducks had MA terms with high coefficients, therefore high levels of variation or noise may be partially hiding patterns of usage and reproductive success associated with DD (Figures 2.2, 2.3).

I did not detect support for the prediction of delayed DD in reproduction of common goldeneyes and hooded mergansers. Both these species delay their maturation until their second or third breeding season (Morse et al. 1969, Eadie and Gauthier 1985), which may result in delayed DD (Brigatti et al. 2016). I therefore predicted that population densities would also exhibit a delayed pattern, although the best fitting ARIMA model for common goldeneye contained a single p component, indicating previous years' breeding was most autocorrelated with the current year's breeding success. Evidence for delayed DD has been shown in small mammals (Agrell et al. 1995), and insects (Turchin 1990), but based on previous work (Viljugrein 2005, Gunnarsson et al. 2012) including my results here, there appears to be little support for lags in DD processes acting on cavity-nesting waterfowl.

Nesting of common goldeneyes appears to be positively DD and reproductive performance appears to increase with higher nest densities. Of the nest boxes examined since 1987, average usage was 45%; other studies have reported higher usage rates of 85-90% (Semel et al. 1988, Malanchuck and Straub 2020). Therefore, while these areas can differ in the degree of nest box usage for various reasons, this suggests there are enough boxes within the LSJR for increased densities of common goldeneyes should they not be limited by other means.

The declining usage and hatching success of common goldeneyes within nest boxes could be due to higher rates of predation or intraspecific parasitism in nest boxes than in natural cavities, or the result of overall decreases in habitat quality which was not measured in this study. Barrow's goldeneyes (*B. islandica*) have been found to have higher reproductive success, lower nest predation, and reduced parasitism rates in natural cavities when compared to nest boxes (Evans et al. 2002). Natural cavities for waterfowl often result from abandoned pileated woodpecker (*Dryocopus pileatus*) cavities, or lost tree limbs (Prince 1968, Bellrose and Holm 1994), which cause them to be more randomly distributed throughout the forest and hidden in the canopy compared to artificial nest boxes (Evans et al. 2002).

Many of the boxes I examined were placed at high densities along habitat edges (e.g., shorelines, agricultural fields), and were generally placed < 3m in height. Raccoons (*Procyon lotor*), a common nest predator of waterfowl, tend to inhabit and feed along these edges (Bider et al. 1968, Dijak and Thompson 2000) causing a potential ecological trap because of edge effects when densities are high (Gates and Gysel 1978, Pasitschniak-Arts and Messier 1995). Conversely, a study on common goldeneye in Finland did not find increased predation along forest edges, although predator communities differed, and interior nests were boxes, not natural cavities (Pöysä et al. 1999). Due to natal philopatry and site fidelity among common goldeneye (Savard and Eadie 1989, Pöysä et al. 1997, Ludwichowski et al. 2002), declining reproductive output may cause lower usage rates of boxes over time. Raccoon depredations caused declines in wood duck breeding populations between 1938-1940 and 1958-1961 in Illinois, USA

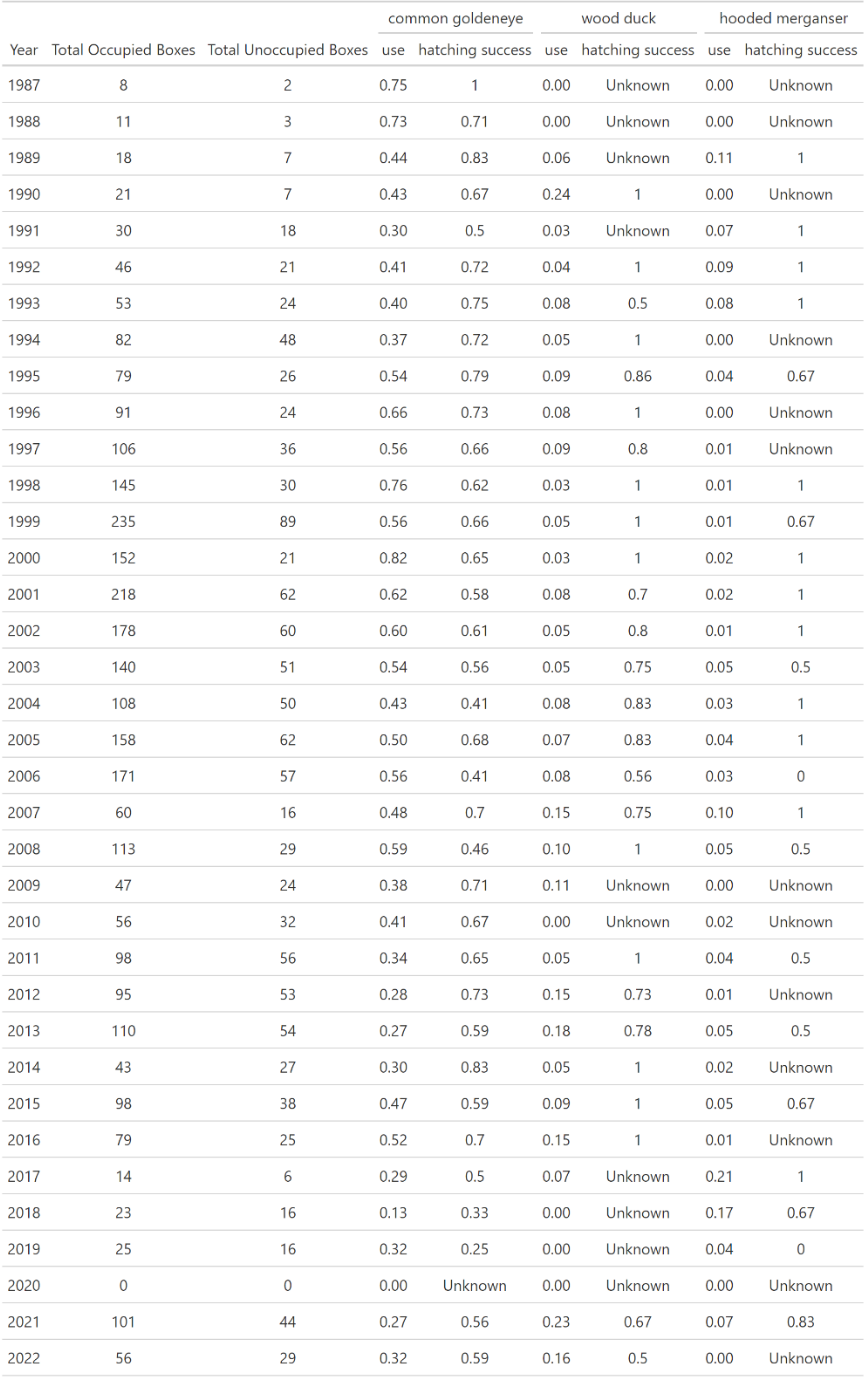
(Bellrose et al. 1964). Raccoon depredations, which I observed anecdotally in this study, may in part explain the declining usage of nest boxes by common goldeneyes over the last 36 years in the LSJR. Additionally, Dow and Fredga (1983) found that female common goldeneye who choose to nest at a different nest site in the year following a nest depredation would have lower reproductive success. Therefore, predation can increase emigration from an area and reduce box occupancy or invoke lower reproductive success in those birds that remain and select new nesting sites.

I tested for interspecific dump nesting as a possible explanation for the declining usage and hatching success of wood ducks by including a dump nest model. Parasitism of nests has been linked to decreased reproductive success (Gates and Gysel 1978, Eadie 1989, Haramis and Thompson 1995). If competition in the form of parasitism (also known as ‘dump nesting’) has increased, this may explain some of the patterns observed here. However, I found no support that interspecific parasitism was autocorrelated over time and therefore is unlikely the cause of the observed temporal pattern. I was unable to assess intraspecific nest parasitism among breeding females, although this is most often found as nesting densities increase (Gates and Gysel 1978, Eadie 1989, Haramis and Thompson 1995).

My approach of analyzing a 36-year dataset on hatching success of cavity nesting waterfowl within the LSJR provides some of the first long-term insights to changes in common goldeneye reproduction at the landscape level. I found common goldeneye exhibit autocorrelative patterns in their reproductive success with negative trends indicating previous years usage and reproductive performance influences that of the current year. While the dataset does not include the causes of nest failure, or identify the true source of DD patterns, I suspect that predation, not the number of nest boxes is limiting to common goldeneye breeding populations. Therefore, efforts to reduce predation through improved nest box placement (e.g., placing nest boxes higher in the canopy, in a more randomly distributed manner, and/or at lower densities), and use of predator guards or metal poles where possible, would be beneficial for improved management. Additionally, more work is needed to understand the relative use and availability of natural cavities within the LSJR and how forest management could be improved to provide better natural nesting opportunities for common goldeneyes and other cavity-nesting waterfowl.

Tables and Figures

Table 2.1 Nesting records from 1987-2022 within the lower Saint John River. Includes proportion of occupied/unoccupied boxes along with usage and hatching success of all three cavity nesting species



			common goldeneye		wood duck		hooded merganser	
Year	Total Occupied Boxes	Total Unoccupied Boxes	use	hatching success	use	hatching success	use	hatching success
1987	8	2	0.75	1	0.00	Unknown	0.00	Unknown
1988	11	3	0.73	0.71	0.00	Unknown	0.00	Unknown
1989	18	7	0.44	0.83	0.06	Unknown	0.11	1
1990	21	7	0.43	0.67	0.24	1	0.00	Unknown
1991	30	18	0.30	0.5	0.03	Unknown	0.07	1
1992	46	21	0.41	0.72	0.04	1	0.09	1
1993	53	24	0.40	0.75	0.08	0.5	0.08	1
1994	82	48	0.37	0.72	0.05	1	0.00	Unknown
1995	79	26	0.54	0.79	0.09	0.86	0.04	0.67
1996	91	24	0.66	0.73	0.08	1	0.00	Unknown
1997	106	36	0.56	0.66	0.09	0.8	0.01	Unknown
1998	145	30	0.76	0.62	0.03	1	0.01	1
1999	235	89	0.56	0.66	0.05	1	0.01	0.67
2000	152	21	0.82	0.65	0.03	1	0.02	1
2001	218	62	0.62	0.58	0.08	0.7	0.02	1
2002	178	60	0.60	0.61	0.05	0.8	0.01	1
2003	140	51	0.54	0.56	0.05	0.75	0.05	0.5
2004	108	50	0.43	0.41	0.08	0.83	0.03	1
2005	158	62	0.50	0.68	0.07	0.83	0.04	1
2006	171	57	0.56	0.41	0.08	0.56	0.03	0
2007	60	16	0.48	0.7	0.15	0.75	0.10	1
2008	113	29	0.59	0.46	0.10	1	0.05	0.5
2009	47	24	0.38	0.71	0.11	Unknown	0.00	Unknown
2010	56	32	0.41	0.67	0.00	Unknown	0.02	Unknown
2011	98	56	0.34	0.65	0.05	1	0.04	0.5
2012	95	53	0.28	0.73	0.15	0.73	0.01	Unknown
2013	110	54	0.27	0.59	0.18	0.78	0.05	0.5
2014	43	27	0.30	0.83	0.05	1	0.02	Unknown
2015	98	38	0.47	0.59	0.09	1	0.05	0.67
2016	79	25	0.52	0.7	0.15	1	0.01	Unknown
2017	14	6	0.29	0.5	0.07	Unknown	0.21	1
2018	23	16	0.13	0.33	0.00	Unknown	0.17	0.67
2019	25	16	0.32	0.25	0.00	Unknown	0.04	0
2020	0	0	0.00	Unknown	0.00	Unknown	0.00	Unknown
2021	101	44	0.27	0.56	0.23	0.67	0.07	0.83
2022	56	29	0.32	0.59	0.16	0.5	0.00	Unknown

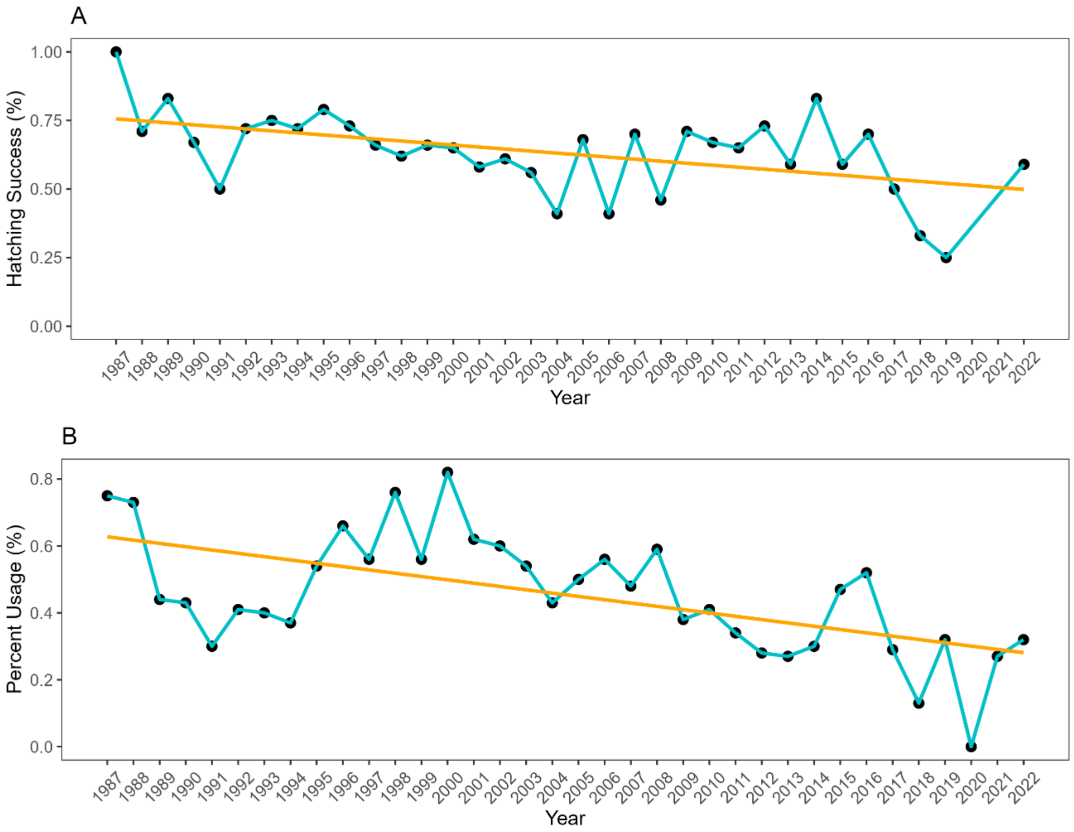


Figure 2.1. Annual hatching success (A) and usage (B) of nest boxes by common goldeneye from 1987 to 2022 based on nest box records collected in the LSJR, NB. Years with no usage of boxes, or where hatching success of clutches was unsure were omitted from the dataset.

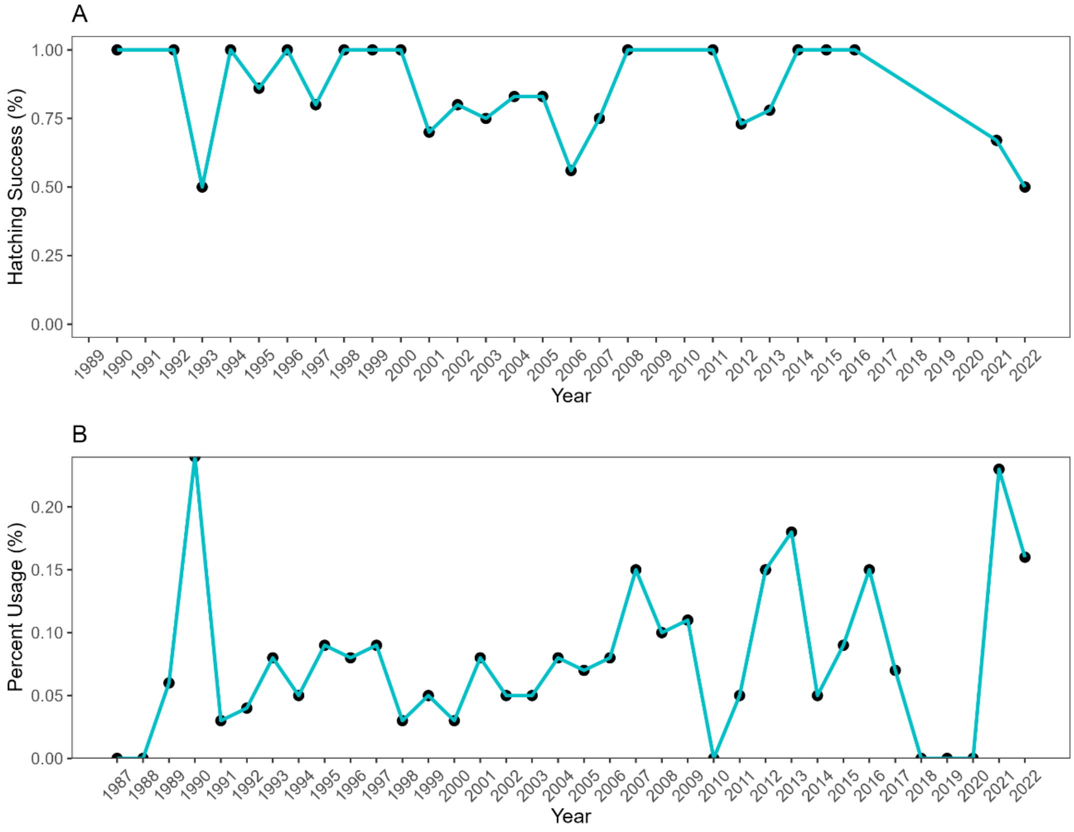


Figure 2.2. Annual hatching success (A) and usage (B) of nest boxes by wood ducks from 1989 (A) and 1987 (B) to 2022 based on nest box records collected in the LSJR, NB. Years with no usage of boxes, or where hatching success of clutches was unsure were omitted from the dataset.

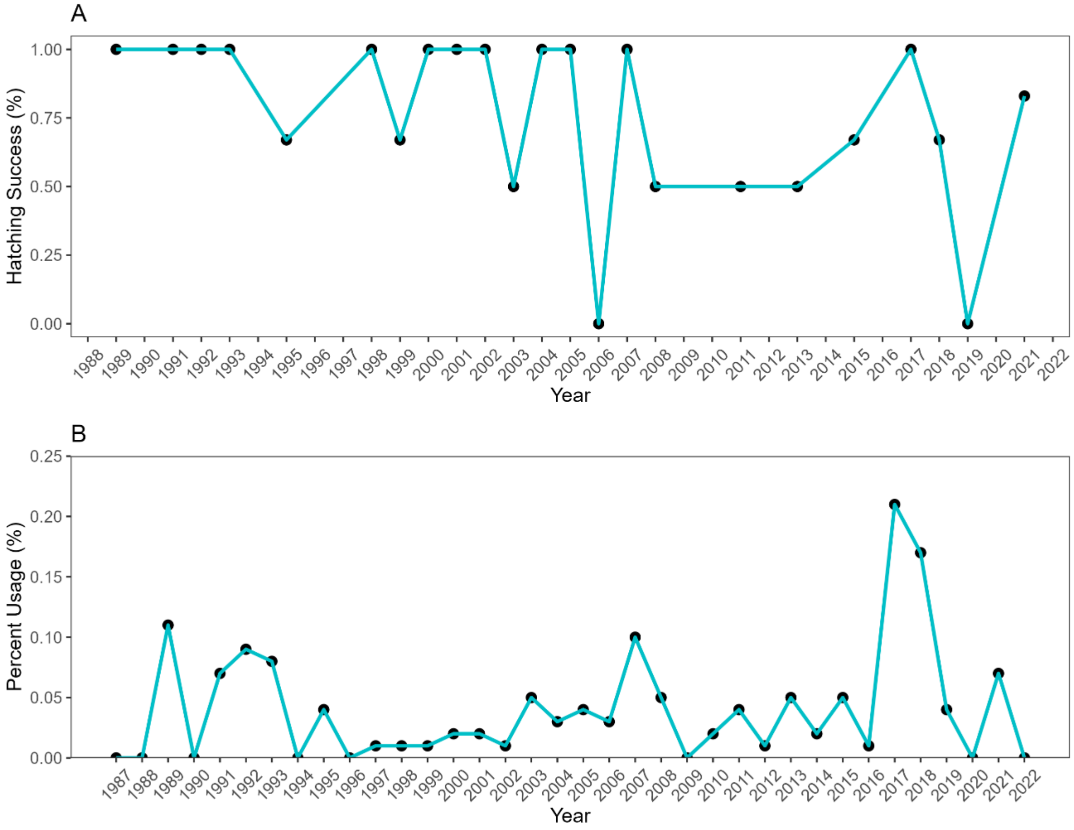


Figure 2.3. Annual hatching success (A) and usage (B) of nest boxes by hooded mergansers from 1988 (A) and 1987 (B) to 2022 based on nest box records collected in the LSJR, NB. Years with no usage of boxes, or where hatching success of clutches was unsure were omitted from the dataset.

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Chapter 3. Age and region-specific survival of wood ducks in the Atlantic Flyway and migratory movements of juveniles to non-breeding grounds

Abstract

Breeding season dynamics and the survival of duckling and nesting female wood ducks (*Aix sponsa*) have been studied extensively as nest boxes provide convenient capture-recapture locations. However, other periods such as migration and survival at non-breeding ground have received considerably less attention. Here I use two data types: band recovery data, and locations from radio-tracked birds to estimate fall and winter survival of juvenile wood ducks within the Atlantic flyway. Band recovery data suggests differences between southern resident and northern migratory populations, and effects of age and sex on survival. Data from six tracked birds indicates that juvenile birds migrating south commonly over-winter in the Chesapeake Bay, USA region. Banding results provides the first robust survival estimates for juvenile wood ducks, which can be used to inform future management of the species.

Introduction

Robust survival estimates are fundamental to population ecology, and especially for game species, are core to wildlife management (Lebreton et al. 1992). For many bird species in North America, especially waterfowl, the survival of breeding females and post-fledgling ducklings has been studied extensively (McGilvery 1969, Ball et al. 1975, Davis et al. 2001, Dyson et al. 2018). Survival estimates are commonly calculated for cavity-nesting waterfowl on their breeding grounds, as nest boxes make the location of breeding females and post fledgling juveniles relatively easy to find (Zicus and Hennes 1987, Katzner et al. 2005). While breeding ground studies provide important demographic information, there has been increased interest in conditions during the non-breeding period as population declines were detected among many migratory birds (Holms et al. 1986, Robbins et al. 1989, Faaborg et al. 2010). Studies focusing on non-breeding periods have shown within-winter survival to be relatively high, although migratory periods pose a significantly higher mortality risk (Sillett and Holmes 2002, Latta 2003, Johnson et al. 2006, Faaborg et al. 2010). In raptors, survival during migration was found to be six times lower than at the breeding and non-breeding grounds (Klaassen et al. 2014). Low survival during migratory periods may significantly limit bird populations (Sherry and Holmes 1995), for instance through low recruitment due to high juvenile mortality (Drummond et al. 2019).

Wood ducks (*Aix sponsa*) are a cavity-nesting species ranging from southern Canada to northern Mexico. Northern populations are fully migratory whereas southern populations are partially migratory or non-migratory (Bellrose and Holm 1994). Among European robins (*Erithacus rubecula*), migratory populations experience higher mortality rates when compared to southern resident or partially migratory populations (Andriaensen and Dhondt 1990). A similar pattern was found in two populations of Egyptian vultures (*Neophron percnopterus*); island resident populations, and in particular juveniles, had higher survival from mark-resight studies when compared to migratory populations (Sanz-Aguilar et al. 2015). Therefore, northern migratory wood ducks may experience higher mortality than southern resident or partially migratory populations.

In northern portions of the Atlantic flyway, wood duck survival appears highest among adult males, then adult females, followed by hatch year females and lastly hatch year males, indicating juvenile survival is lower than other age classes (Johnson et al. 1986). Results from Johnson et al. (1986) contrasts with Shirkey and Gates (2020) who found that although sex best explained differences in survival, there were differences in survival by age with hatch year females having the lowest survival, followed by adult females, adult males, then hatch year males. Higher mortality of juvenile birds has been found in other waterfowl species, for instance in a 3-year study of radio-marked northern pintails (*Anas acuta*), mortality rates of adults were 71.4% compared to 55% for immature birds between October and February (Cox et al. 1998). Migoya and Baldassarre (1995) showed that, when hunting pressure was low and habitat abundant, there were no differences in survival rates of pintails between adult and immature birds. Therefore, mortality rates of juveniles may fluctuate more with increased hunting pressure and lower body condition during the hunting season when compared to adults (Hepp et al. 1986, Krementz et al. 1988).

Studying birds during non-breeding periods such as migration remains difficult due to limitations of certain technologies (Robinson et al. 2010). For instance, while geolocators can track birds across long distances, they can have coarse spatial resolution and animals must often be re-caught to obtain data (Burger and Shaffer 2008, Robinson et al. 2010). Some tracking devices are too heavy for smaller species and the cost of certain technologies can be a limiting factor (Berthold et al. 2004, Thorup et al. 2010, Robinson et al. 2010). Another approach for tracking birds during the non-breeding period is with automated radio telemetry (such as Motus: Taylor et al. 2017) which uses constantly operating towers to automatically detect tags. While no published studies to date have used the MOTUS system to track waterfowl migration, some studies have used it to estimate survival rates for other bird species. For instance, the survival of post-fledgling barn swallows (*Hirundo rustica*) in Ontario, Canada was estimated using the Motus wildlife tracking system within a mark-recapture framework (Evans et al. 2020).

Here, I sought to estimate wood duck survival during the non-breeding period, which in turn will improve our understanding of wood duck populations. To do this I used two separate data types. First, I attached VHF (very high frequency) transmitters to juvenile wood ducks that were tracked using Motus, to estimate survival over the entire non-breeding period. Second, I used banding data from the USGS bird banding laboratory collected over the last 63 years to estimate hunting-related mortality and any specific differences among age classes and northern versus southern populations. I predict that hunting-related mortality for juvenile wood ducks and individuals from northern populations will be higher than for adults and southern populations, respectively. I also predict that survival estimates generated from MOTUS data will be lower than those generated from banding data as it will represent all sources of mortality, not just hunting, and can therefore be used to understand recruitment rates into the following year's breeding population.

Methods

Site selection and trap construction

I captured wood ducks throughout the month of August in 2021 and 2022. I selected trapping sites based on annual usage of various managed impoundments and ponds. I used four sites in 2021 and five sites in 2022 with 12 traps in total each year. Trapping areas (Figure 3.1) within these sites were selected based on 1) water levels, which could fluctuate throughout August and cause some areas to be too deep or shallow (ideal water depth for most traps was between 0.5-1.5m), 2) the location within the wetland, and 3) areas where wood ducks were known to be in high densities. At each site, I removed vegetation such as wild rice (*Zizania aquatica*), bulrush (*Scirpus atrovirens*), or pickerelweed (*Pontederia cordata*) to make room for the traps. Once I cleared sections of approximately 5x5m, I baited the area with cracked corn (used as both a food attractant and visual stimulus) on floating boards and in the surrounding water. Sites were checked periodically to determine if waterfowl were present, as indicated by feathers, visual observations, or other evidence. Once I determined that sites were visited regularly, I closed the traps. Traps were constructed from a roll of 2.54x2.54 cm welded metal wire. This roll of wire was left for a day or two to allow the feeding waterfowl time to adjust to this new object in the area. The roll was then extended into a semi-circular configuration, with 1.9 cm diameter poles used to support the wire. One bait board was placed within the trap while a second was placed in front, outside of the trap. This configuration was left for several days to allow waterfowl time to adjust. Finally, the trap was closed into its final configuration, leaving a funnel opening at the front for ducks to enter. Mesh was placed on top of the trap which kept waterfowl from escaping. Both boards were baited with cracked corn and additional corn was spread within the trap and funnel. Traps were left over night and checked daily from the 9 August until 4 September in 2021 and from 4 August until 4 September in 2022.

Capturing and banding

Once wood ducks were trapped, I placed any newly caught individual (i.e., not re-traps) in holding cages until they could be transported back to shore. Once all wildlife had been removed, I again baited traps to be left until the next day. I occasionally removed traps in both years from an area (or opened again to the semi-circular configuration) if they were continually empty or had signs of damage from predators such as raccoon (*Procyon lotor*) and/or black bear (*Ursus americanus*). Once on shore, I sexed, aged, and fitted a USGS leg band to all individuals. At the time of most trapping activities (early to mid August) male wood ducks were in their eclipse plumage and therefore breeding plumage could not be used to sex individuals. Male wood ducks were identified by the presence of red eyes, white colored tear-drop shapes on tips of secondaries, and a cloacal check. Female wood ducks were sexed by the lack of red eyes, a straight white bar on the tip of secondaries, and a cloacal check. I aged all wood ducks by the presence of notched tail feathers which are only present on birds hatched in that year, as these notches develop through the process of moulting into their adult plumage from their downy feathers. Older individuals who have undergone at least one molt (were hatched at least in the year prior) will not have these notches present. If individuals were hatched in that year, I classified them as hatch year (HY) unless they had more than four blood feathers present in which case they were classified as local (L). I assigned all other birds with no notched tail feathers to an after-hatch year (AHY) category.

VHF tags

I fitted all HY and L wood ducks with Lotek radio transmitters at the frequency 166.38 Hz (standard for MOTUS system). HY wood ducks were tagged from 7-22 August in 2021 (n = 42; 28 female, 14 male) and from 4-10 August in 2022 (n = 33; 18 female, 15 male). I attached tags to HY or L wood ducks using glue to fix them to cut dorsal feathers which allows for the tags to fall off during the next moulting cycle. Feathers in this region of the back are retained from the time where birds receive their juvenile plumage until >144 days (>4 months) (Bellrose and Holm 1994) and therefore are likely to last the entire study period.

Some L birds had not yet moulted into their juvenal plumage and only pin feathers were present on the interscapular area. In such cases I did not tag L birds to ensure that the tag did not hamper the individual's proper plumage growth and to avoid complications from gluing pin feathers in place. The "clip-and-glue" method I used consists of clipping a patch (approximately 2-3cm in diameter) of interscapular dorsal feathers where the tag will be placed. The tag was then activated using a Lotek infrared tag activator and checked to ensure it is transmitting using a Lotek SRX800 telemetry receiver. Polyacrylamide glue was placed on the bottom surface of the tag which is fitted to the clipped patch on the bird and held for ~1 minute depending on weather conditions. The tag was checked to ensure that neither the device or the antenna interfered with the natural movement and pose no additional risk of catching vegetation. Once the tag's glue had dried, I released birds immediately to the appropriate wetland from which they were caught. I re-checked tags on re-captured birds to ensure it was still activated and had not become dislodged from the cut patch of feathers. In one instance a tag had partially been removed, and in this case, I used additional polyacrylamide glue to re-attach the tag to the appropriate location.

Survival models

I tracked radio-tagged juvenile wood ducks using the MOTUS automated radio telemetry array (Taylor et al. 2017; http://motus.org). Data were collated with R (v 4.2.1; R Core Team 2023) using package *motus* (v 6.0.1; Birds Canada 2022). Data were filtered to only include detections between August and March of 2021/2022 and 2022/2023, latitudes of 36.312499º to 46.601597º, and longitudes of -77.7650066º to -66.567559º. Few detections occurred with run lengths >4 (standard filter used by *motus*), and therefore observations were included if Burstslop was >0.20 s and SDfreq was >0.08 (Crewe et al. 2019, Bliss 2020). Post-filtering, some detections remained that were inconsistent with wood duck migration (e.g., detected in another continent or western North America), and were therefore removed. Additionally, all detections from two towers (Newtowne Neck State Park, MD, and Lynnfield, MA), were chronologically, or geographically inconsistent and were therefore excluded due to high likelihood of being false detections. The low detection rates in this study precluded an analysis using a mark-recapture model.

I estimated survival during the hunting season using banding data from the USGS bird banding laboratory (USGS Bird Banding Laboratory 2023) from all available years (1960-2022) within the Atlantic Flyway. I only used hunting-related mortality for two reasons: first, there were few band recoveries over the 63 years that were not hunting-related, and secondly, comparing the two values from MOTUS and banding provides information on possible rates of non-hunting related mortality, which is often unobservable. I used a Burnham joint live encounter-dead recovery model which is a generalization of both the Cormack Jolly-Seber model and ring recovery model, allowing for the analysis of both recapture and dead recoveries to improve precision in parameter estimates (Burnham 1993, Chastant et al. 2014). For instance, the use of a Burnham joint model decreased standard errors from survival rates of mallards (*A. platyrhynchos*) by 12% for females over the Brownie model (Brownie 1985) through the inclusion of extra information (live recaptures) (Barker and White 2001). I chose to use the Burnham model over the Barker model (Barker 1997) because the Barker model requires live re-sightings between capture occasions (i.e., banding seasons) (Kendall et al. 2013). Within this dataset the number of live birds observed was low and would likely provide no additional information. The four parameters in the joint live encounter-dead recovery model are S ('true' survival probability), F (site fidelity), r (reporting probability of dead birds), and p (recapture probability). Site fidelity and recapture probability were fixed, however, reporting probability was allowed to vary with time. This was to reduce model complexity and total number of parameters used. It should also be noted that due to inability to account for each individual, the estimate is best termed 'apparent' survival. However, the Burnham model assumes 'true' survival is known, therefore from here on, it will be referred to simply as survival.

I encoded encounter histories in a "Live-Dead" format (LD) where the first value indicated a live observation whereas the second value indicated if a bird was reported dead during that year. Therefore, a bird banded or observed in a given year but not found dead would be assigned an encounter history of "1 0" while a bird found dead in a given year, but not seen alive was assigned an encounter history of "0 1". In certain cases – especially juveniles – where a bird was banded in a given year and found dead that same year, I assigned an encounter history of "1 1". I conducted statistical analyses in R (v 4.2.1; R Core Team 2023) using package *RMark* (v 3.0.0; Laake 2017) to construct a model set (n = 15 models) with four separate covariates: age, sex, time, and region. The two age classes used in this analysis were juveniles (those hatched during the most recent breeding season, and therefore <1 year of age) and adults (>1 year of age). Region was also used as a covariate due to differences in the life history strategies of northern and southern wood duck populations. In Johnson (1986) banding records were split into three regions: New York-Eastern Canada, Northeastern, and Southeastern. Due to little difference in survival New York-Eastern Canada and Northeastern regions (Johnson 1986), they were combined for this analysis. Northern birds were classified as being banded North of West Virginia, and Maryland, while southern birds were those banded in Virginia and south (Figure 3.2). Northern birds are considered fully migratory compared to southern populations who are partially migratory or residents. These differences in migratory strategies have been tied to differences in survival with northern populations experiencing lower survival when compared to southern populations (Nichols and Johnson 1990, Hepp and Hines 1991). I retained southern birds in this analysis, as some northern breeding birds migrate to southern non-breeding areas where they can be recovered (either found dead or re-caught). Time was also used as a covariate as hunting pressure and associated regulations have likely changed over time and therefore may influence the level of hunting related mortality. The 63 years of data were split into three terms with the first spanning from 1960 – 1980, the second from 1981 – 2001, and the third from 2002 – 2022. This was done to create equal sized bins for easy comparison. I ranked candidate models using Akaike's information criterion corrected for small sample size (AIC_c) (Hurvich and Tsai 1987) and Akaike weights (Akaike 1978) to find the model with the most support. I chose AIC_c due to the relatively large number of parameters (n = 27 in global model).

Results

MOTUS Movements

Six tags provided enough data for post-filtering, two tags (Tag1, and Tag2) were birds I tagged in 2021, while the other four (Tag3, Tag4, Tag5, and Tag6) I tagged in 2022. Tags 1, 3, 4, and 6 were on females (66%), while tags 2 and 5 were on males (33%). Tag 1 had six detections spanning from 16 August 2021 to 16 February 2022 (Appendix A.1A). Tag 2 had six detections spanning from 17 October 2021, until 4 January 2022 (Appendix A.2A). Tag 3 had seven detections spanning from 12 October 2022, until 12 September 2022 (Appendix A.3A). Tag 4 had three detections spanning from 21 November 2022, until 2 February 2023 (Appendix A.4A). Tag 5 had a single detection on 19 February 2023 (Appendix A.5A). Tag 6 had three detections spanning from 22 October 2022, until 10 February 2023 (Appendix A.6A).

The birds with Tags 2 and 6 moved along coastal routes to New Jersey, USA and Delaware, USA respectively. Tag 2 data showed movements from Maine, USA in

October to the Chesapeake Bay region in November (Appendix A.2B). Tag 6 data showed movements from Maine, USA in October, to New Jersey USA in January (Appendix A.6B). Tag 1 data showed movements inland from Maine, USA in August to Pennsylvania, USA in January (Tag 3B). Later in the season Tags 4 and 5 were detected in the Chesapeake Bay, USA region (Appendix A.4B, Appendix A.5B), while Tag 3 was only detected early in the study within Maine, USA (Appendix A.3B). Tags 1 and 2 were the only tags that showed any northerly movements later in the season (Appendix A.1B, Appendix A.2B). Tag 1 was detected in Pennsylvania, USA in December, before moving to Ontario in January, then moving to Massachusetts, USA towards the end of January with its last detection being recorded during February in New York, USA (Appendix A.1B). Tag 2 after being detected near the Chesapeake Bay, USA region was then detected during January in Ontario before moving east into Vermont, USA where it was last detected (Appendix A.2B).

Survival Analysis

Survival estimation with a Burnham Joint Live-Dead recovery model showed the best fitting model included the effects of sex, age, region, and year (Table 3.1). The highest ranked model included 27 parameters and had substantial support over other models ($W_i = 1$). For northern birds during the first period (1960-1980) (Table 3.2, Figure 3.4), survival was nearly identical among adult males (S = 0.59, CI = 0.58-0.60) and adult females (S = 0.59, CI = 0.58-0.61), followed by juvenile males (S = 0.46, CI = 0.44-0.48), and juvenile females (S = 0.45, CI = 0.42-0.47). During the second period (1981-2001) (Figure 3.4), survival was highest among adult males (S = 0.64, CI = 0.63-

0.65), followed by the nearly identical survival rates of adult females (S = 0.59, CI = 0.58-0.60), and juvenile males (S = 0.59, CI = 0.57-0.61), with juvenile females (S = 0.50, CI = 0.47-0.52) again having the lowest survival. For the final period (2002-2022) (Figure 3.4), survival was the highest and nearly identical among adult males (S = 0.61, CI = 0.60 – 0.62), and juvenile males (S = 0.61, CI = 0.59-0.62), followed by adult females (S = 0.56, CI = 0.54-0.57), and lastly juvenile females (S = 0.55, CI = 0.53-0.57).

For southern birds during the first period (1960-1980) (Table 3.3, Figure 3.4), survival was highest among juvenile males (S = 0.65, CI = 0.62-0.68), followed by similar survival rates of juvenile females (S = 0.61, CI = 0.57-0.64) and adult males (S = 0.61, CI = 0.60-0.62), with adult females (S = 0.58, CI = 0.57-0.60) having the lowest survival. During the second period (1981-2001) (Figure 3.4), survival was highest among juvenile males (S = 0.67, CI = 0.65-0.69), followed by nearly identical survival rates between adult males (S = 0.66, CI = 0.65-0.67) and adult females (S = 0.66, CI = 0.65-0.67), with juvenile females (S = 0.57, CI = 0.54-0.60) having the lowest survival. For the final period (2002-2022) (Figure 3.4), survival was highest among juvenile males (S = 0.69, CI = 0.68 – 0.71), followed by similar survival rates of juvenile females (S = 0.61, CI = 0.60-0.63) and adult males (S = 0.61, CI = 0.61-0.62), with adult females (S = 0.59, CI = 0.57-0.60) having the lowest survival.

Recovery rates differed slightly between periods (Table 3.3). Recovery rates were highest in the third period (2002-2022) (r = 0.16, CI = 0.16-0.16), followed by the first period (1960-1980) (r = 0.10, CI = 0.09-0.10), and finally second period (1981-2001) (r = 0.08, CI = 0.08-0.08).

Discussion

My analysis of wood duck survival over the last 63 years suggests there are age, sex, and regional differences in survival probability. I found substantial support that wood ducks from the northern Atlantic flyway have lower survival than those from the southern Atlantic flyway, which parallels the findings of Johnson et al. (1986) over a much shorter time period (1964-1983) that survival in northern populations was highest among adult males and lowest among juvenile females. My results differ with respect to juvenile males that appeared to have higher survival during 1981-2001 than was found previously. My results from 1981-2001 support Shirkey and Gates (2020) finding that survival was highest among juvenile and adult males, followed by adult females, with juvenile females having the lowest survival. Similarities in the results to both studies are likely the result of overlap in banding records, which provides support for the results found here. Differences from our results and these studies may be the result of differences in chosen time periods, banding regions selected or models used.

Wood ducks from the southern Atlantic flyway have higher survival than those from the northern Atlantic flyway (Figure 3.3). This was consistent between the three periods analyzed and may be the result of lower body condition in northern wood duck populations because of their longer migration (Nichols and Johnson 1990 as cited in Hepp and Hines 1991). This finding is consistent with the harvest “condition bias hypothesis”, where individuals in lower body condition tend to be less risk-averse than those in better condition, increasing rates of hunting-related mortality (Weatherhead and Greenwood 1981, Greenwood et al.1986, Reinecke and Shaiffer 1988, Dufour et al. 1993, Pace and Afton 1999, Fowler et al. 2020).

I found that differences in survival among juvenile and adult birds were related to which region they occupied (Figure 3.4). Northern juveniles experienced lower survival compared to adults, while southern adults had lower survival compared to juveniles. Juveniles may differ in the migratory efficiency or timing on their way to the wintering grounds (Rotics et al. 2016, Bontekoe et al. 2023). Among northern pintails (*A. acuta*), juveniles often arrived to wintering grounds later and with less time to forage for resources prior to the hunting season (Fleskes et al. 2002). Juveniles were also less efficient at gathering resources prior to the hunting season, spending more time in search of food, while adults gathered local information on habitat type (sanctuary vs. hunter occupied) (Fleskes et al. 2002). Additionally, lacking prior knowledge of hunting threats may result in increased juvenile mortality (Francis et al. 1992, Cox et al. 1998, Cox and Afton 1997). Body condition and feeding efficiency likely explains both regional differences (northern birds migrate further and therefore expend more energy than southern birds), and age differences (juveniles may have lower body conditions at the start of the hunting season) in the harvest rates of wood ducks.

I detected differences in survival rates between sexes, with females consistently having lower survival than males, other than in the first period (1960-1980) where northern juvenile males had the lowest survival and in the southern region where adult male survival was slightly lower than for females (Figure 3.4). Juvenile females had the overall lowest survival in the northern region although it appears to have been increasing over time (Figure 3.4). There have been few studies examining sex specific differences in harvest mortality (Cooch et al. 2014); my findings indicate a pattern on which further work could focus to better understand whether low female survival is the result of hunting and determine the drivers of the increasing survival of juvenile female wood ducks in the northern Atlantic flyway.

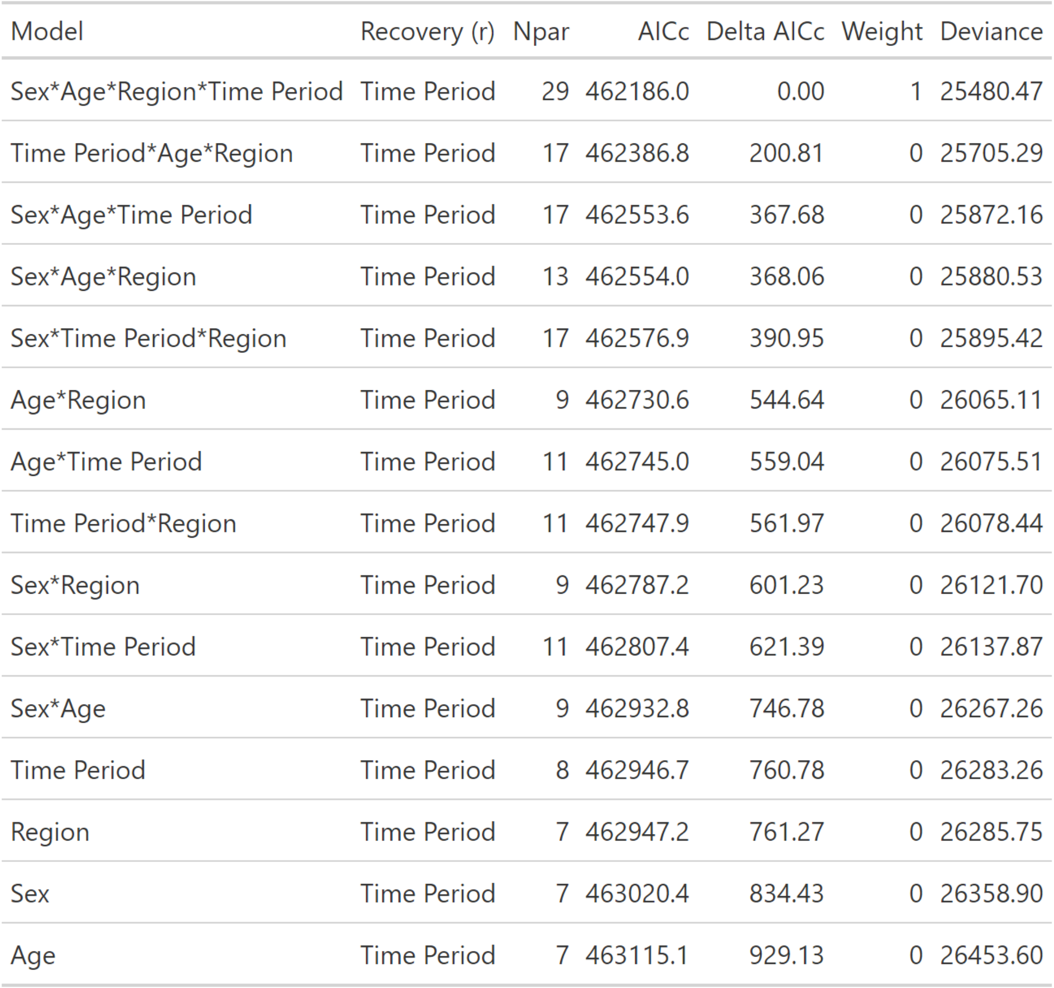
Six radio-tagged birds were detected using MOTUS towers along eastern North America, of which three were detected within the Chesapeake Bay, USA region (Appendix A.2, 4, and 5). Due to the low number of detections, it is uncertain if these birds remained in this region for prolonged periods of time and if other tagged birds were also inhabiting this region but were not detected. The low number of detections could have resulted from tag retention issues and/or human error. Wheeler (1991) studied blue-winged teal (*Spatula discors*) that were captured and reared in pens with tags attached using various attachment methods. The glue method as used on the 74 birds tagged here performed poorly on all ducks. Feathers were pruned and pulled off rapidly after tagging (1-7 days). This may explain why a large proportion of the birds tagged were never detected on the MOTUS system throughout 2021, 2022 or 2023. Tag loss did not explain the high number of false detections received throughout North America. It is plausible that some false detections were in fact true, however, it is nearly impossible to determine exactly how many. Due to the low number of birds with probable detections and the low number of detections per tagged bird, these results should be interpreted with caution. While these detections provide some insight into the movements of juveniles to the non-breeding grounds, results were based on my interpretation of probable movements and timing. However, detections from New Brunswick to and within the Chesapeake Bay region, often had the highest support based on methods described above. Therefore, these detections provide some evidence of migratory and non-breeding movements but are likely unfit for analysis of survival or migration.

The Chesapeake Bay has long been recognized as an important region for waterfowl (Perry and Deller 1996). However, due to anthropogenic development, the abundance of submerged aquatic vegetation declined throughout the 1960's and 1970's likely reducing the quality of the habitat for waterfowl (Chesapeake Bay program 1992, Perry and Deller 1996). There is a high likelihood of northern juveniles experiencing increased harvest mortality resulting from poor body condition in the fall. Therefore, ensuring wintering habitat is suitable may increase survival, resulting in additional recruitment into the spring breeding population. My study demonstrates that northern wood ducks experience higher hunting-related mortality than southern ducks. More work is likely needed to understand the role of body condition and how it differs by region, age, and sex. Additionally, waterfowl managers including Ducks Unlimited, the Eastern Habitat Joint Venture, and federal agencies (e.g., Environment and Climate Change Canada, US Fish and Wildlife Service) should work to ensure that the Chesapeake Bay remain productive for juvenile wood ducks.

Depending on the goals of local management, different approaches could be taken with waterfowl hunting regulations. Shirkey and Gates (2020) demonstrated that reducing bag limits from three to two led to a small reduction in mortality for all age classes. Additionally, delaying the onset of hunting season for wood ducks may provide more time for juveniles to acquire resources and local information. This mirrors the suggestion of Bellrose and Holm (1994) that earlier regulatory seasons may contribute to increased hunting-related mortality for northern wood ducks. However, this may only be applicable in situations where recruitment is low. Therefore, management options to improve survival estimates for northern wood ducks may be feasible by altering harvest regulations. It should be noted that more work would need to be done to understand the effect that these lower survival estimates have on population sizes in the northern portion of their range.

Tables and Figures

Table 3.1. Models included in survival analysis, including number of parameters (Npar), change in Akaike’s Information Criterion adjusted for small sample size (AICc), model weight, and deviance.



Model	Recovery (r)	Npar	AICc	Delta AICc	Weight	Deviance
Sex*Age*Region*Time Period	Time Period	29	462186.0	0.00	1	25480.47
Time Period*Age*Region	Time Period	17	462386.8	200.81	0	25705.29
Sex*Age*Time Period	Time Period	17	462553.6	367.68	0	25872.16
Sex*Age*Region	Time Period	13	462554.0	368.06	0	25880.53
Sex*Time Period*Region	Time Period	17	462576.9	390.95	0	25895.42
Age*Region	Time Period	9	462730.6	544.64	0	26065.11
Age*Time Period	Time Period	11	462745.0	559.04	0	26075.51
Time Period*Region	Time Period	11	462747.9	561.97	0	26078.44
Sex*Region	Time Period	9	462787.2	601.23	0	26121.70
Sex*Time Period	Time Period	11	462807.4	621.39	0	26137.87
Sex*Age	Time Period	9	462932.8	746.78	0	26267.26
Time Period	Time Period	8	462946.7	760.78	0	26283.26
Region	Time Period	7	462947.2	761.27	0	26285.75
Sex	Time Period	7	463020.4	834.43	0	26358.90
Age	Time Period	7	463115.1	929.13	0	26453.60

Table 3.2. Survival of northern and southern wood ducks within the Atlantic Flyway from 1960-2022, derived from a Burnham Joint Live-Dead model, showing the survival rate, standard error (se), and lower (lcl) and upper confidence limits (ucl).

	Northern				Southern			
	Survival	se	lcl	ucl	Survival	se	lcl	ucl
2002-2022								
Adult Female	0.56	0.007	0.54	0.57	0.59	0.006	0.57	0.60
Adult Male	0.61	0.004	0.60	0.62	0.61	0.004	0.61	0.62
Juvenile Female	0.55	0.010	0.53	0.57	0.61	0.009	0.60	0.63
Juvenile Male	0.61	0.007	0.59	0.62	0.69	0.007	0.68	0.71
1981-2001								
Adult Female	0.59	0.007	0.58	0.60	0.66	0.005	0.65	0.67
Adult Male	0.64	0.004	0.63	0.65	0.66	0.004	0.65	0.67
Juvenile Female	0.50	0.012	0.47	0.52	0.57	0.014	0.54	0.60
Juvenile Male	0.59	0.009	0.57	0.61	0.67	0.009	0.65	0.69
1960-1980								
Adult Female	0.59	0.007	0.58	0.61	0.58	0.008	0.57	0.60
Adult Male	0.59	0.006	0.58	0.60	0.61	0.006	0.60	0.62
Juvenile Female	0.45	0.011	0.42	0.47	0.61	0.017	0.57	0.64
Juvenile Male	0.46	0.010	0.44	0.48	0.65	0.013	0.62	0.68

Table 3.3. Recapture (p), reporting (r), and site fidelity (F) probabilities from Burnham Joint Live-Dead model showing the standard error (se), and lower (lcl) and upper confidence limits (ucl).

Parameter	Estimate	se	lcl	ucl
P(.)	0.01	0.0003	0.01	0.01
F(.)	0.74	0.0101	0.72	0.76
r(1960-1980)	0.10	0.0009	0.09	0.10
r(1981-2001)	0.08	0.0006	0.08	0.08
r(2002-2022)	0.16	0.0010	0.16	0.16

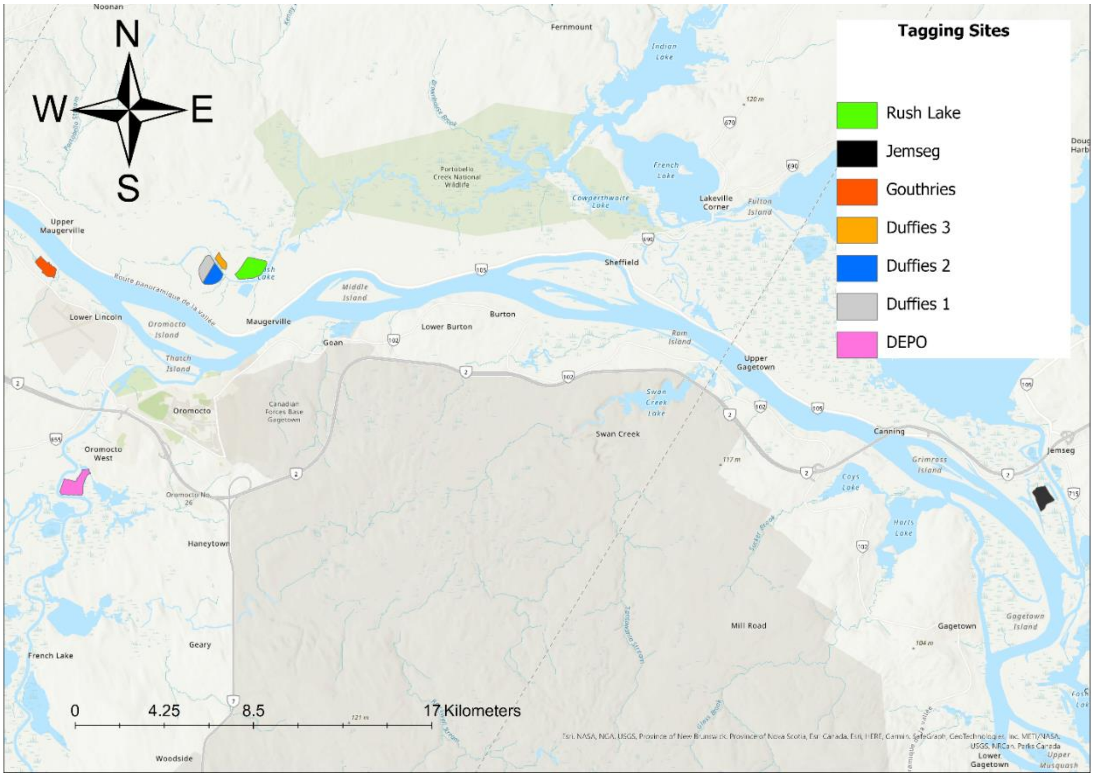


Figure 3.1. Sites where wood ducks were captured using funnel traps during August of 2021 and 2022. Rush Lake and Jemseg were the only two sites not used in both years for trapping, due to construction, and access, respectively.

Figure 3.2. Northern and southern regions for wood ducks within the Atlantic flyway as defined in Johnson et al (1986). Ontario and Quebec, while traditionally part of the Mississippi flyway, are included because southern regions of these provinces have wood duck populations which may migrate into the Atlantic flyway.

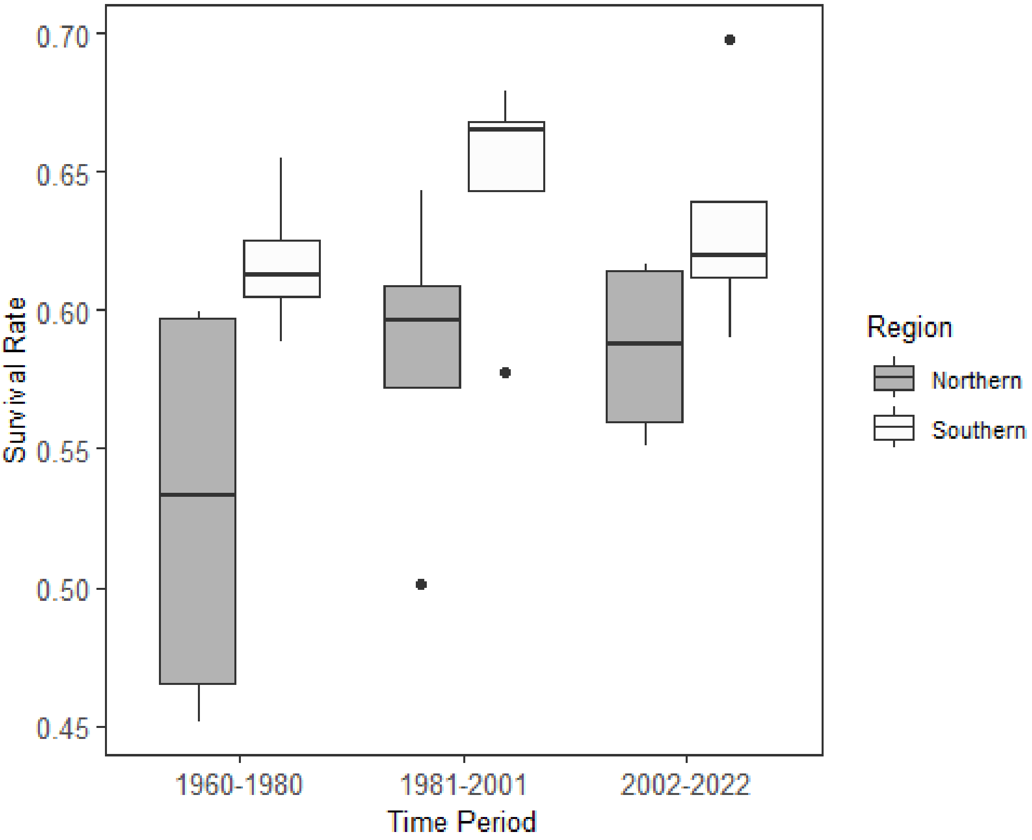


Figure 3.3. Difference in survival rates between northern and southern wood duck populations during three time periods spanning from 1960-2022 based on hunter returns and capture-recapture data.

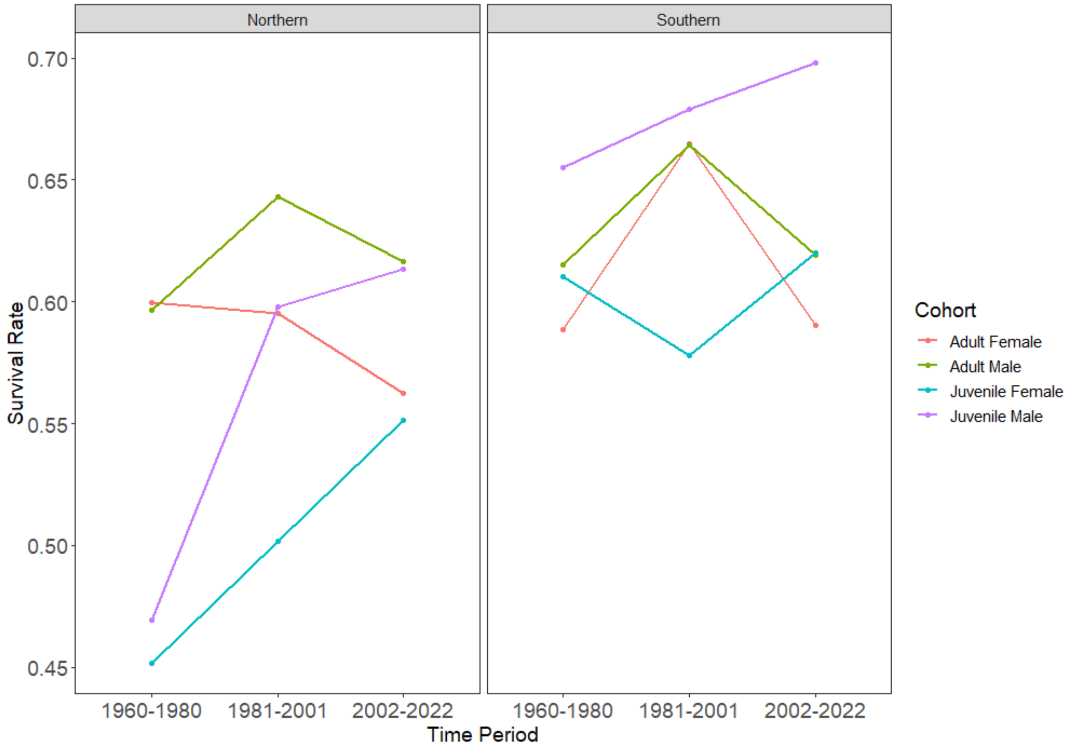


Figure 3.4. Changes in survival rates over three time periods for adult females, adult males, juvenile females, and juvenile males in two regions of the Atlantic flyway (Northern and Southern). Data from hunter returns and capture-recapture data.

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Chapter 4. Identification of limiting factors for population growth of wood ducks and common goldeneye through population viability and sensitivity analyses

Abstract

To improve management, especially of game species, population viability analyses (PVA) and sensitivity tests can be focussed on population vital rates of interest. Age 0-1 survival of wood ducks (*Aix sponsa*) and common goldeneyes (*Bucephala clangula*) are important determinants of population growth and decline. Here, I use updated survival estimates for wood duck from the northern Atlantic flyway, and new observations that reproduction of common goldeneyes can be positively density-dependent, to update PVA model parameterization and subsequent sensitivity analyses. I found that age 0-1 survival for wood ducks remains a limiting factor, although the updated model predicts lower (and possible negative) growth rates, along with potential extinction at higher mortality values. For common goldeneyes, I found little evidence that incorporating positive density dependent effects leads to significant differences in model outcomes. These results suggest that increasing survival of 0-1 year-old wood ducks, especially through enhanced hatching success and duckling survival, would be best for targeted management.

Introduction

Understanding vital rates are important for game management when setting annual hunting regulations or informing conservation decisions (Rockwell et al. 1997, Hoekman et al 2002, Aubry et al. 2010, Hepp et al. 2020). For instance, harvest of most waterfowl species is managed within flyways based on an adaptive harvest management approach which requires information on breeding, mortality, and density dependence to create predictive models (Nichols et al. 2007). While these rates have been identified based on nesting studies and banding data, there has been comparatively little research quantifying the importance of these rates on overall population dynamics. For waterfowl in particular, an understanding of limiting factors for population growth is important for management through the North American Waterfowl Management Plan (NAWMP) and associated Joint Ventures. Wood ducks (*Aix sponsa*) and common goldeneyes (*Bucephala clangula*) are listed as priority species under the NAWMP for the Canadian provinces of Ontario, Quebec, and New Brunswick (EHJV 2017) where population growth is targeted.

The natural nesting sites of wood ducks and common goldeneyes are in tree cavities. Between the 1960's and 1990's New Brunswick saw dramatic declines in natural cavity abundance due to forestry, agriculture, and Dutch elm disease (Prince 1968, Johnsen et al. 1994). As a result, New Brunswick currently attempts to improve cavity-nesting duck populations through the provisioning of nest boxes. There has been considerable focus on attaining higher reproductive output through nest boxes to increase population sizes in the region (Ducks Unlimited Canada 2000). Whether these boxes have resulted in population-level outcomes has been difficult to assess as there is a lack of information regarding other parts of their annual cycle, such as estimating mortality rates at different life-stages. A previous study in New Brunswick (Harding 2021) determined that survival of age 0-1 wood ducks and common goldeneye had the largest influence on overall population size and would likely be limiting to population growth. Conversely, the number of nest boxes seemed to have little effect on local population size, indicating that adding more nest boxes would likely provide few population-level benefits (Harding 2021). Although these analyses provided useful management information, there remains a lack of regional survival rates for specific age classes such as juvenile birds. An analysis of banding data using a Burnham Joint Live Dead model (Chapter 3) provides new information to improve a population viability analysis (PVA) for wood ducks and thus better predict future population size and the influence of vital rates.

PVAs are stochastic models that simulate future population change through knowledge of current demographics and state variables. While useful in making management decisions, such as in species-at-risk assessments, the current quality and use of PVAs in many research applications has been questioned (Beissinger and Westphal 1998, Pe'er et al. 2013). Only 18.1% of 160 published PVAs from 1990-2017 were considered high quality (Chaudhary and Oli 2020). Often attempts to identify reliable extinction probabilities or future population sizes are restricted due to poor quality data or lack thereof, especially in those that are spatially explicit (Reed et al 2002). However, an extension of a PVA called a sensitivity analysis has been widely used to examine the influence of changing vital rates on population dynamics (McCarthy et al. 1995, Benton and Grant 1999, Mills and Lindberg 2002, Manlik et al. 2016, Manlik et al. 2018). Sensitivity analysis allows examination of different management scenarios (e.g., changing the number of nest boxes or influencing survival rates) on long term population outcomes (Mills and Lindberg 2002). These comparative predictions are likely less sensitive to variation or inaccuracy in model parameters and construction than attempting to determine exact probabilities through a PVA alone.

Here, I will use a combination of PVA and sensitivity analysis to improve an existing model for wood ducks and common goldeneyes in the lower Saint John River (LSJR) area of New Brunswick (Harding 2021). Increased precision in these models will come from the incorporation of density dependent reproduction parameters for common goldeneyes, and improved survival estimates for wood ducks within the northern Atlantic Flyway. I predict that being able to specify density dependent effects for common goldeneye but not wood ducks will alter the outcomes of previous PVAs and may further support the idea that nest box density is not limiting population growth. I also predict that more precise wood duck survival estimates, especially for age 0-1 individuals, will allow me to confirm juvenile survival as a rate-limiting parameter for population growth.

Methods

PVA Parameterization

My goal is to increase the precision of PVAs created by Harding (2021; hereafter "the original models") based on newly collected data (density dependence in nesting and age 0-1 survival of wood ducks). Therefore, to create a set of PVA models that could easily be compared, I used VORTEX (V 10.5.6.0; Lacy and Pollack 2023) and started with the input values from Harding (2021). I then updated the parameters for wood duck survival and density dependence in common goldeneye. Most values in the starting PVA were obtained through a literature review for wood ducks and common goldeneye (Table 4.1, 4.2), with a focus on selecting values from studies with the closest regional proximity (Harding 2021). Some values for the LSJR such as initial population size, mean/SD in brood size, and carrying capacity were calculated by Harding (2021); these were used to create five different scenarios. Each scenario corresponds to different levels of nest site availability using carrying capacity as a proxy. Model 1 assumed only nest boxes were available within the LSJR, model 2 (low density) and 3 (high density) are natural-cavity only models based on indexes calculated from Service NB GeoNB "Forest" layer, and New Brunswick Hydrographic Network layer (Harding 2021). Models 4 and 5 combine available nest boxes with cavity abundances used in models 2 and 3.

Based on my finding that reproduction was positively density dependent for common goldeneyes but not wood ducks (Chapter 2), I updated the original model by incorporating density dependence within the reproductive system where at P(0) 80% of the population would be breeding (all females, assuming equal sex ratio), while at P(K) 93% of the females would be breeding. P(0) and P(K) represent percent of females breeding under low and high densities within a density dependent system, respectively. The original model (Harding 2021) was non density-dependent and set 80% of females as available for breeding. To obtain P(K), I used values from Pöysä and Pöysä (2002) who experimentally increased the number of nest boxes at a site. They found that before introducing nest boxes (what I will assume is their P(0)) there were an average 27.5 breeding pairs that produced 17 broods. After adding boxes, the number of breeding pairs increased to an average of 38.4, although the number of broods only increased to 19. The population in the Pöysä and Pöysä (2002) study was density-dependent, therefore I can assume that after adding the nest boxes, the percent of breeding females would correspond to P(K). By assuming one brood per female, the percent of females breeding at P(0) and P(K) can be calculated. I found that at P(0), 62% of females were breeding while at P(K) 49% of females were breeding. Therefore, the difference in percent females breeding at P(0) vs P(K) is 13%. Since my study population of common goldeneyes appears to display positive density-dependence, at P(K) the number of breeding females should be 93% (P(0) of 80% + 13%).

To estimate adult wood duck mortality, I used the inverse of my survival estimates calculated using banding data in Chapter 3. I calculated a monthly survival estimate, based on data in Chapter 3, for age 0-1 males and females. I assumed a "starting" survival rate of 69% based on estimated hatching success of wood ducks in New Brunswick calculated by Harding (2021). I then applied a 53% survival rate based on literature values for ducklings to the flight stage (McGilvrey 1969). I then calculated an 11-month survival estimate based on the 12-month estimate calculated in Chapter 3, assuming that survival was consistently distributed throughout time. I used VORTEX to build the population-based models with 100-time steps and 500 iterations (Brook et al. 1999, Lindenmayer et al. 2000, Harding 2021).

Sensitivity Test

I ran sensitivity tests for wood ducks and common goldeneye in VORTEX. For wood ducks, I conducted sensitivity tests on age 0-1 mortality, adult (>1) mortality, and percent of females breeding. For common goldeneye, I ran sensitivity tests on P(0), P(K), and percent of females breeding. Percent females breeding within the common goldeneye model represents breeding under a non density-dependent system (original model). I created maximum and minimum values representing ±25% of the base value for all tested parameters. I tested all mortality rates and breeding parameters in incremental values of 2% (as used in Harding (2021)) and followed these with sensitivity tests run as a single-factor analysis with 100 samples.

Results

The improved survival estimates for juvenile male and female wood ducks led to differences from the original model in the predicted population size over a 100-year period (Figure 4.1); the time to reach a stable population size increased and intrinsic rate of increase (r) decreased substantially. The per capita rate of increase in the original model varied from 0.5218-0.5257 while the updated mortality estimates reduced per capita rate of increase to 0.1925-0.1970 (Table 4.3). Additionally, models with the updated parameterization never reached the same stable population size as those with the original values (Table 4.2; Figure 4.1), although there was major overlap in confidence intervals suggesting similarity.

Updating the common goldeneye PVA by including positive density-dependent reproduction (where differences in the percent of females breeding changed with population density) had little effect on growth rates or population size (Table 4.2). The only instance where the newly created PVA differed from the previous version was in model-1 (nest box only model), in which density-dependent processes appear to have resulted in population decline over time (Figure 4.2A). It also remains as the only model with a probability of extinction >0 (Table 4.2.).

Sensitivity analysis on the wood duck models revealed differences in population growth rates and population size of several tested parameters. The updated survival rates resulted in male and female age 0-1 mortality, percent females breeding, and adult female mortality all differing substantially from the original model. Increases in age 0-1 mortality resulted in the population size declining more rapidly and becoming extinct prior to reaching +25% of the base values (Appendix B.1-5). Perturbations of this age 0-1 mortality showed negative growth rates at higher tested values, which was not found by varying any other parameter (Appendix B.6-10). In both the original and updated model, the overall trends appear the same, although the original model used lower mortality rates resulting in initially higher population size and growth rates than the updated model. These results differ when comparing the female adult survival parameters; where tested values were nearly identical, although the trends differed (Appendix B.6-10). The original model showed that as mortality increases, there are minimal effects to growth rates and population size. However, within the updated model, as mortality increased both population size and growth rates began to decline (Appendix B.1-10), although never reaching a population size of 0, nor showing negative growth rates. For adult males, tested values and trends were similar between models, although population size and growth rates were lower in the updated compared to the original model. The updated model sets appear to also place an increased importance on the percent of females breeding. When compared to the original models, increasing female breeding results in larger population sizes and increased growth rates (Appendix B.6-10).

Similar to the results from the PVA model, sensitivity tests for common goldeneye revealed similar population sizes and growth rates regardless of variations in the parameters of: percent females breeding under a non density-dependent system, density-dependent system at low densities, and a density dependent system at high densities (Appendix B.11-12).

Discussion

Here, I used newly available data to update the PVA models of Harding (2021) for wood ducks and common goldeneye and found support for my prediction that improving survival estimates for wood ducks would lead to differences in model outcomes. The updated models showed lower population growth rates and overall population sizes (Table 4.3). Higher mortality rates for the updated wood duck PVA led to a lower stable equilibrium and lower population growth rates (Figure 4.1). However, contrary to my prediction for common goldeneye, the inclusion of density-dependent breeding had only minor effects on model outcomes with no substantial change to growth rates or population size other than the “nest box only” model (Table 4.4; Figure 4.2).

I ran sensitivity tests on five wood duck parameters: adult female mortality, adult male mortality, age 0-1 female mortality, age 0-1 male mortality, and percent of females breeding where base values were allowed to vary by ±25%. Increasing age 0-1 mortality by 25% led to negative growth rates and a reduction of the population size to 0. Unlike the updated model, increasing age 0-1 mortality in the original model by 25% led to a reduction in population growth rates.. This supports the findings of Harding (2021) that age 0-1 survival is important for population growth, although higher mortality estimates used in the updated model show that negative growth rates and population-level extinction are possible. Age 0-1 survival (especially age 0-1 females) being the most important vital rate is supported by Hepp et al. (2020) who used a perturbation analysis and life-stage simulation analysis to examine vital rates and their effect on wood duck populations. Importantly, however, Hepp et al. (2020) deconstructed various life-stages further than in my analysis. My models defined juveniles as those from ages 0-1, while Hepp et al. (2020) could test the effects of nest success and female recruitment separately. While both parameters were important, female recruitment had a larger influence on growth rates, suggesting that increasing female recruitment through improved duckling survival would be the optimal management strategy.

The addition of density-dependent effects to the common goldeneye models appears to have had negligible impacts on model outcomes for the updated PVA or sensitivity test. Results for both percent of females breeding at P(0) and P(K) remain nearly identical to results from Harding (2021). However, these results support my findings from Chapter 2 in terms of positive density-dependent effects; as percent of breeding females increases within the density-dependent models, population size and growth rates continue to increase (Appendix B.11-12).

While the PVA and sensitivity analysis conducted here provide information on influential vital rates for wood ducks, the results should be interpreted cautiously, as there remains considerable gaps in knowledge of cavity-nesting duck ecology, particularly at regional scales. As is typical for PVAs, high levels of uncertainty and potential data inaccuracies should be considered before using these results to make management decisions. However, given that my updated model and those of Harding (2021) both show that 0-1 age individuals are most important for population growth rate, management should, until we know more, be focused there. Secondly, as highlighted by Manlik et al. (2018), management actions targeting specific vital rates may not always be feasible. For example, Manlik et al. (2018) showed that a sensitivity analysis for the endangered lesser kestrel (*Falco naumanni*) found adult survival to have the largest influence on population growth and decline. However, survival estimates in this endangered population were higher than those reported for stable populations, meaning increases to survival had little feasibility (Hiraldo et al. 1996).

Survival of age 0-1 birds is comprised of three separate rates, which would each require distinct management actions of variable achievability. Hatching success and duckling survival are the two mast manageable rates. Increasing the number of nest boxes does not necessarily increase hatching success for wood duck (see Chapter 2). However, reducing predation of nest boxes (Bellrose and Johnson 1964) and intra- and inter-specific competition (Jones and Leopold 1967, McGilvery and Uhler 1971) may allow for increased hatching success. The duckling stage is also a critical life-stage where habitat quality and predation are often the largest mortality factors (Bellrose and

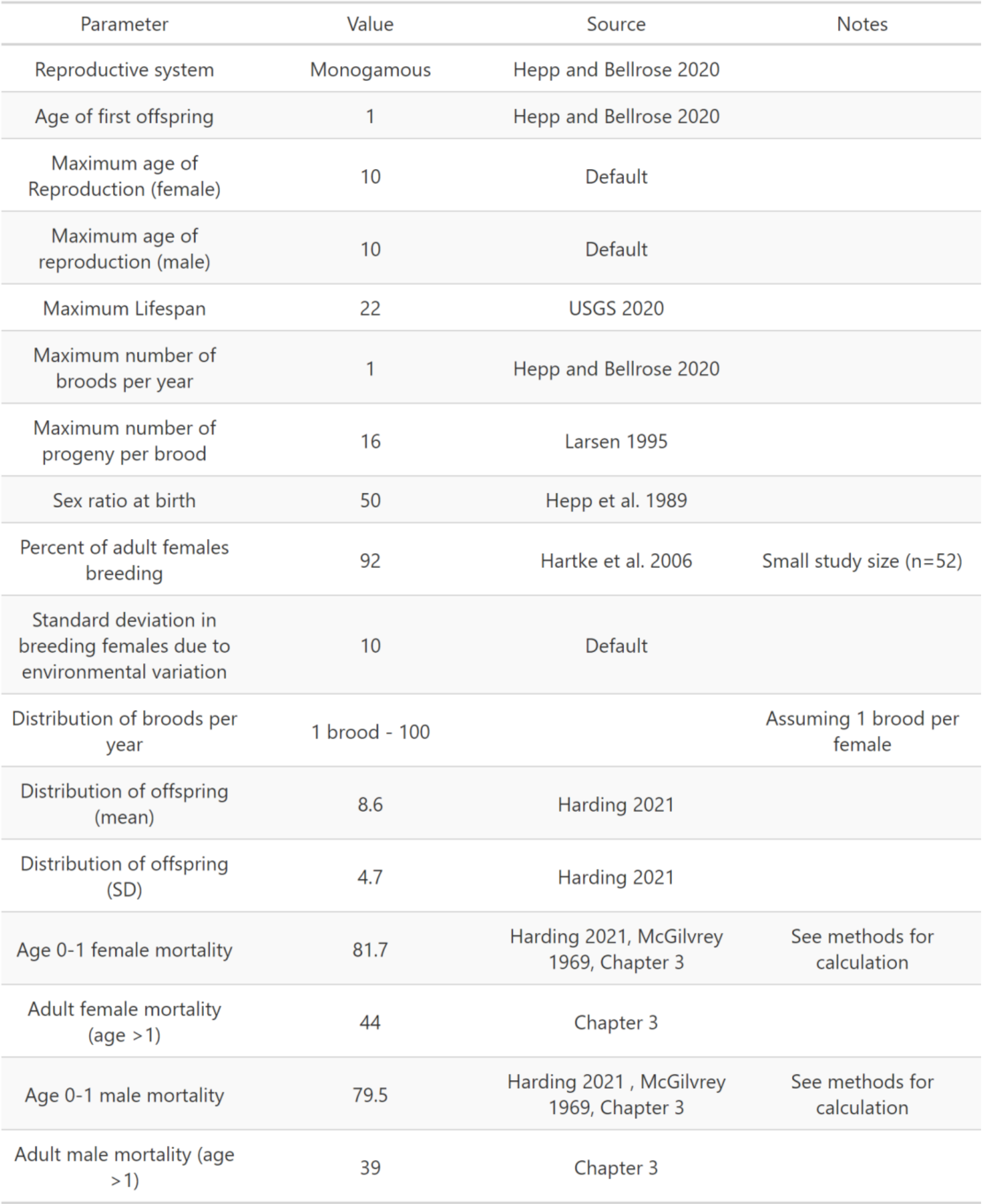
Holm 1994, Davis et al. 2007, Davis et al. 2009). Therefore, increasing habitat quality and reducing predation would likely reduce limitations on population growth Lastly, while the survival of age 0-1 and adult female wood ducks in my study is not high (see Chapter 3), management actions to decrease juvenile mortality may not be feasible despite their importance to population growth and decline. Decreasing mortality rates would likely need to come from changes to waterfowl harvest regulations. However, such changes would need to be highly nuanced; it would be an unreasonable expectation that a hunter could easily distinguish between adult and juvenile ducks mid-flight.

Breeding Bird Survey and Atlantic Flyway Breeding Waterfowl Survey data indicated wood duck populations have increased from 1.20 million to 1.37 million from 1993-2013 (Zimmerman et al. 2015). During this same time period, juvenile female survival increased from 50% to 55% within the Northern Atlantic Flyway (Chapter 3). While difficult to identify the exact cause for population increases at such a large scale, these complimentary trends may suggest an underlying relationship. However, most juvenile mortality (especially in the fall) is likely linked to harvest (Bellrose and Holm 1994). Wood duck bag limits and season lengths are regulated through an adaptive harvest management approach, which does facilitate age or sex specific distinctions in regulations. Further, it is also very unlikely a hunter could discern a juvenile from an adult in flight if an age restricted harvest were put in place Therefore, the feasibility of management actions targeting hatching success and duckling survival is high, however, far less likely for juveniles despite their importance.

This study justifies further investigation into specific life history periods and vital rates of cavity-nesting waterfowl. For instance, while I developed a survival estimate for the entire northern Atlantic Flyway based on banding data, there is not yet an estimate for juvenile wood ducks within New Brunswick. As this vital rate largely influences the growth and persistence of wood duck populations in my study and a study by Harding (2021), local or regional estimates should be obtained to make appropriate management decisions. Common goldeneyes and wood ducks are both harvested recreationally and are priority species under the Eastern Habitat Joint Venture implementation plan ((EHJV 2017). Despite their economic and social importance, few efforts have successfully evaluated the vital rates and demographic parameters important for population growth (but see Hepp et al. 2020, and Lawson et al. 2017). My results suggest that a focus on nesting and duckling survival provides the best compromise between feasibility of management and overall likelihood to reduce limitations for wood duck population growth.

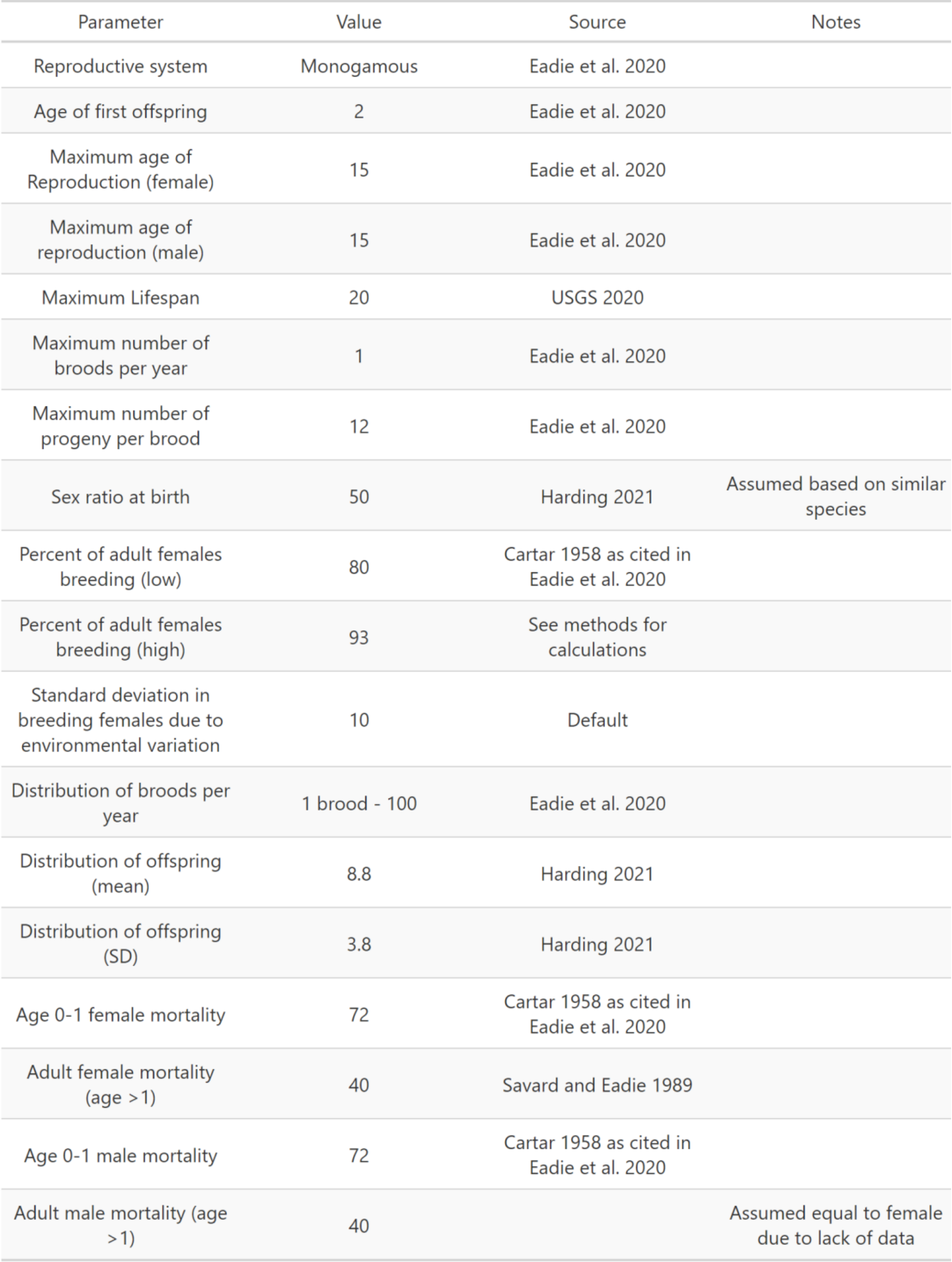
Tables and Figures

Table 4.1. Parameter values and source publications for wood duck population viability analysis in VORTEX.



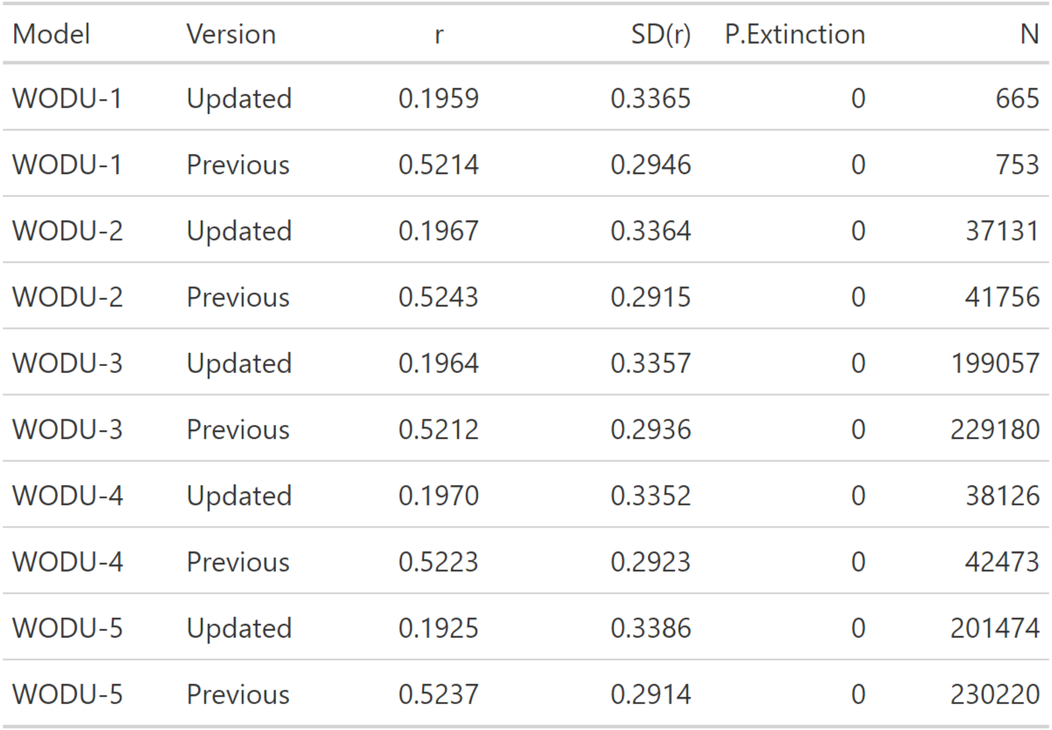
Parameter	Value	Source	Notes
Reproductive system	Monogamous	Hepp and Bellrose 2020	
Age of first offspring	1	Hepp and Bellrose 2020	
Maximum age of Reproduction (female)	10	Default	
Maximum age of reproduction (male)	10	Default	
Maximum Lifespan	22	USGS 2020	
Maximum number of broods per year	1	Hepp and Bellrose 2020	
Maximum number of progeny per brood	16	Larsen 1995	
Sex ratio at birth	50	Hepp et al. 1989	
Percent of adult females breeding	92	Hartke et al. 2006	Small study size (n=52)
Standard deviation in breeding females due to environmental variation	10	Default	
Distribution of broods per year	1 brood - 100		Assuming 1 brood per female
Distribution of offspring (mean)	8.6	Harding 2021	
Distribution of offspring (SD)	4.7	Harding 2021	
Age 0-1 female mortality	81.7	Harding 2021, McGilvrey 1969, Chapter 3	See methods for calculation
Adult female mortality (age >1)	44	Chapter 3	
Age 0-1 male mortality	79.5	Harding 2021 , McGilvrey 1969, Chapter 3	See methods for calculation
Adult male mortality (age >1)	39	Chapter 3	

Table 4.2. Parameter values and source publications for common goldeneye population viability analysis in VORTEX.



Parameter	Value	Source	Notes
Reproductive system	Monogamous	Eadie et al. 2020	
Age of first offspring	2	Eadie et al. 2020	
Maximum age of Reproduction (female)	15	Eadie et al. 2020	
Maximum age of reproduction (male)	15	Eadie et al. 2020	
Maximum Lifespan	20	USGS 2020	
Maximum number of broods per year	1	Eadie et al. 2020	
Maximum number of progeny per brood	12	Eadie et al. 2020	
Sex ratio at birth	50	Harding 2021	Assumed based on similar species
Percent of adult females breeding (low)	80	Cartar 1958 as cited in Eadie et al. 2020	
Percent of adult females breeding (high)	93	See methods for calculations	
Standard deviation in breeding females due to environmental variation	10	Default	
Distribution of broods per year	1 brood - 100	Eadie et al. 2020	
Distribution of offspring (mean)	8.8	Harding 2021	
Distribution of offspring (SD)	3.8	Harding 2021	
Age 0-1 female mortality	72	Cartar 1958 as cited in Eadie et al. 2020	
Adult female mortality (age >1)	40	Savard and Eadie 1989	
Age 0-1 male mortality	72	Cartar 1958 as cited in Eadie et al. 2020	
Adult male mortality (age >1)	40		Assumed equal to female due to lack of data

Table 4.3. Wood duck population viability analysis model outputs from both previous and updated parameterization. Includes growth rate (r), standard deviation in the growth rate (SD(r)), probability of extinction (P.Extinction) and final population size (N).



Model	Version	r	SD(r)	P.Extinction	N
WODU-1	Updated	0.1959	0.3365	0	665
WODU-1	Previous	0.5214	0.2946	0	753
WODU-2	Updated	0.1967	0.3364	0	37131
WODU-2	Previous	0.5243	0.2915	0	41756
WODU-3	Updated	0.1964	0.3357	0	199057
WODU-3	Previous	0.5212	0.2936	0	229180
WODU-4	Updated	0.1970	0.3352	0	38126
WODU-4	Previous	0.5223	0.2923	0	42473
WODU-5	Updated	0.1925	0.3386	0	201474
WODU-5	Previous	0.5237	0.2914	0	230220

Table 4.4. Common Goldeneye population viability analysis model outputs from both previous and updated parameterization. Includes growth rate (r), standard deviation in the growth rate (SD(r)), probability of extinction (P.Extinction) and final population size (N).

Model	Version	r	SD(r)	P.Extinction	N
COGO-1	Updated	0.1219	0.2429	0.014	1102
COGO-1	Previous	0.1086	0.2494	0.000	1255
COGO-2	Updated	0.1419	0.2494	0.000	55413
COGO-2	Previous	0.1096	0.2481	0.000	54262
COGO-3	Updated	0.1328	0.2503	0.000	303960
COGO-3	Previous	0.1109	0.2499	0.000	297051
COGO-4	Updated	0.1405	0.2495	0.000	56838
COGO-4	Previous	0.1105	0.2475	0.000	55380
COGO-5	Updated	0.1342	0.2488	0.000	306665
COGO-5	Previous	0.1087	0.2472	0.000	299026

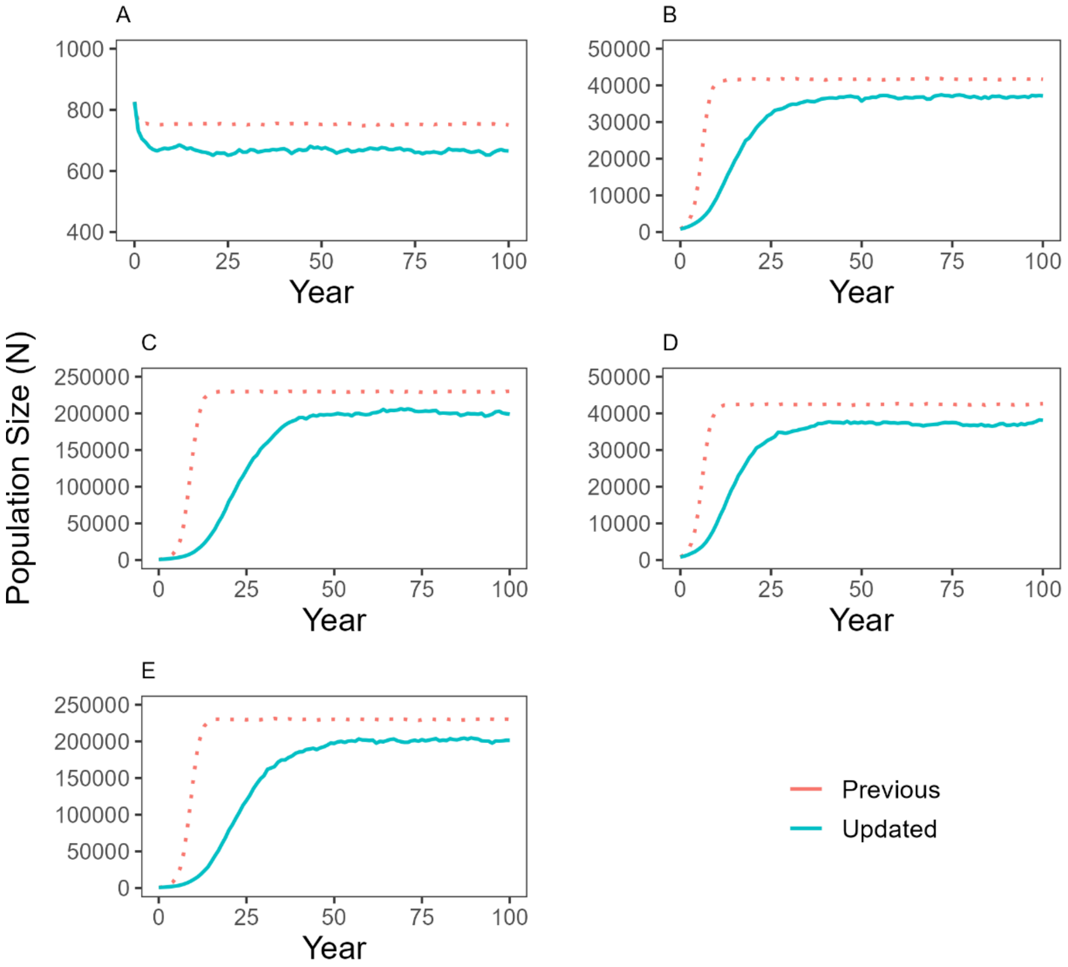


Figure 4.1. Population viability analysis of wood ducks under previous and updated parameterization for all 5 models. A) nest box only model, B) low cavity density model, C) high cavity density model, D) low cavity density, and nest box model, and E) high cavity density, and nest box model. Differences in models are expressed through estimates of carrying capacity (K).

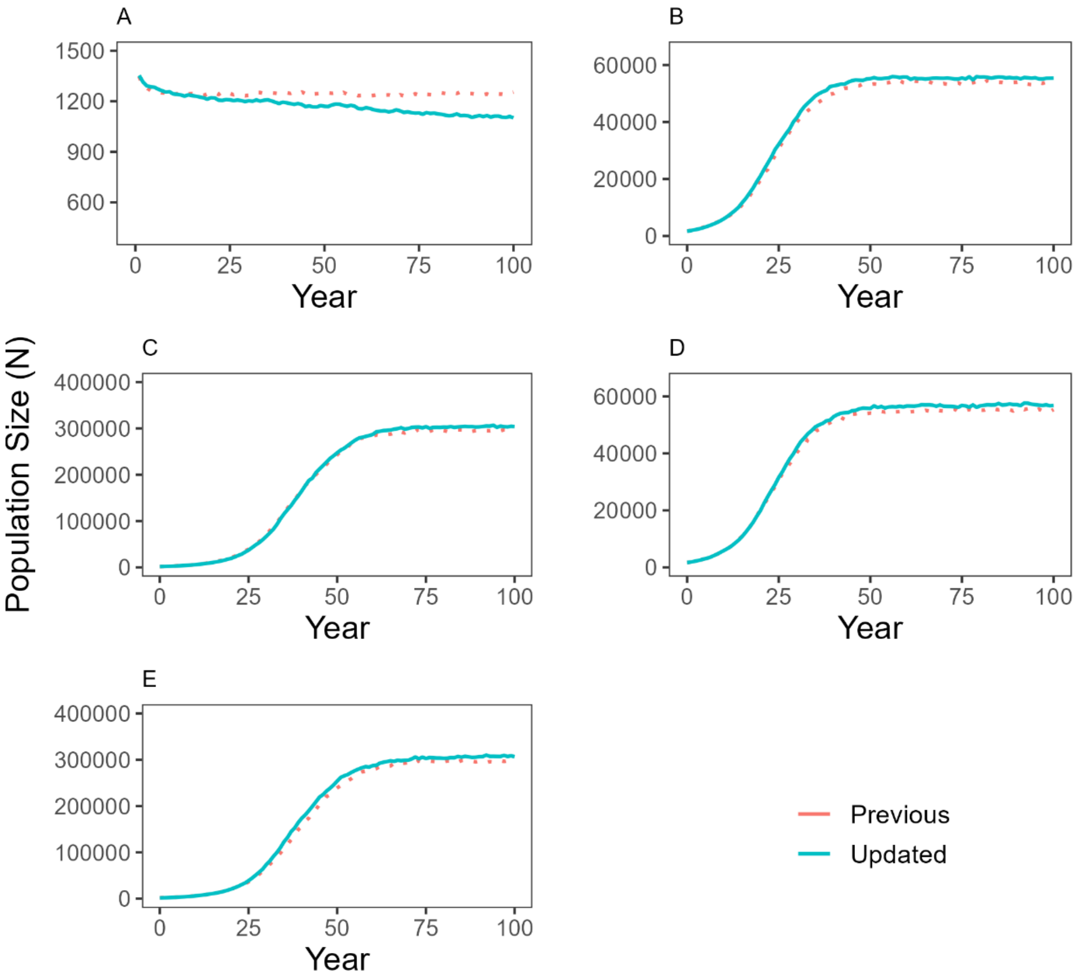


Figure 4.2. Population viability analysis of common goldeneye under previous and updated parameterization for all 5 models. A) nest box only model, B) low cavity density model, C) high cavity density model, D) low cavity density, and nest box model, and E) high cavity density, and nest box model. Differences in models are expressed through estimates of carrying capacity (K).

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Chapter 5. General Conclusion

This thesis examined the population dynamics of cavity-nesting waterfowl during different life-history stages. This research was generated by management information needs of the Eastern Habitat Joint Venture, for which wood ducks (*Aix sponsa*), common goldeneyes (*Bucephala clangula*), and hooded mergansers (*Lophodytes cucullatus*) are priority species for conservation (EHJV 2017). I focussed primarily on the life-history stages of nesting and juvenile age classes, which correspond to vital rates where more information is needed to understand limitations on population growth (Hepp et al. 2020, Lawson et al. 2017). This thesis built upon findings from Harding (2021) where population viability analyses were created for five scenarios (nest box only, natural cavity high vs. low, and all nests high vs. low) and showed that perturbations in age 0-1 survival were highly influential.

In chapter 2, I hypothesized that common goldeneyes and hooded mergansers, but not wood ducks, would display density-dependence. I predicted that if nesting in artificial boxes was density-dependent, then I would expect usage rates and hatching success to be autocorrelated over time and display opposing trends. To test this prediction, I used 36 years of nesting records for wood ducks, common goldeneyes, and hooded mergansers, which were obtained from Bill Cooper, a prolific long-term Ducks Unlimited Canada (DUC) nest-box steward. I then used an autoregressive integrated moving average (ARIMA) model to test these data for density-dependence. I demonstrated that common goldeneyes, but not wood ducks or hooded mergansers display first-order autocorrelation in their use of nest boxes and hatching success. The results suggest that common goldeneye hatching success is higher when a larger proportion of boxes are being used (i.e., positive density-dependence). This was also supported by a positive (although weak) linear relationship between hatching success and usage rates in nest boxes.

A common cause for nest failure was depredation by raccoons (*Procyon lotor*) and female abandonment of nests. I was unable to determine the cause of abandoned nests, although in Central Illinois, nearly 33.6% of raccoon depredated nests resulted in female mortality (Bellrose and Holm 1994). Raccoon encounters may also increase rates of flushing by females from nests. When examining wood duck nest boxes between 1958 and 1965, a decline in flushing rates were found to be related to declines in raccoon depredation attempts (usually with a single-year time lag) (Bellrose and Holm 1994). It is unlikely that raccoon populations are higher in my study area (the area of the lower Saint John River; LSJR) than in other areas where density dependence has not been found. However, unlike in many areas where predator guards or metal poles are used to reduce predation, nest boxes in New Brunswick are commonly mounted on trees, and at high densities. This likely makes them easily accessible for some predators. The use of predator reduction devices (e.g., metal poles, predator guards) and strategic placement of nest boxes have been suggested to reduce depredation of nests (Strange et al. 1971, Haramis and Thompson 1985, Bellrose and Holm 1994). This should therefore be a simple yet effective way to improve nest box programs in the LSJR.

High rates of nest failure have been shown to reduce the site fidelity of females. In American robins (*Turdus migratorisu*) and brown thrashers (*Toxostoma rufum*), females who nested successfully were more likely to nest in the same location the following year; unsuccessful females were more likely to select a different breeding site the following year (Haas 1998). The reduction in site fidelity was based on females using a “decision rule” based on previous breeding experience (Haas 1998). Therefore, the trend I observed in common goldeneye could result from some degree of avoidance of LSJR nest boxes. Instead, females could be favouring areas where nest boxes are placed at lower densities with predator protection or making a switch to natural cavities. Natural cavity availability in New Brunswick declined dramatically between the 1960s and 1990s due to forestry activities and Dutch elm disease (Prince 1968, Johnsen et al. 1994). The lowest natural cavity availability was in the 1990s, when Johnsen et al. (1994) found only a single cavity per hectare, compared with 5.5 cavities per hectare in 1963-64 (Prince 1968). This period of low natural cavity availability corresponds approximately with the beginning (1987) of the nesting dataset I analyzed, and the period with the highest nest box usage by common goldeneyes. If natural cavity availability has since increased, as predicted to occur within the northeastern United States of America (Denton et al. 2012), then this may explain the reduction of nest box usage by common goldeneye. The resulting decline in hatching success may be due to partitioning of resources by high- and low-quality, or experienced and inexperienced breeders. If natural cavity availability was more limited when LSJR nesting records began, I would expect more high-quality females, or more experienced breeders, were nesting in boxes. However, if natural cavities are more abundant, and/or the quality of nest boxes has declined, then I predict that LSJR nest boxes are occupied by a greater proportion of low quality or inexperienced breeders. Differences in reproductive success between experienced and non-experienced breeders have been shown (Dow and Fredga 1984, Gauthier 1989, Hepp and Kennamer 1993), and may present opportunities for future research within the LSJR.

One limitation of the data used in Chapter 2 is that nest box placement, maintenance, and observations may have been inconsistent over time. I used boxes placed and maintained by a DUC steward, who added, removed, and altered boxes at various points over the last 36 years. This is why I used the proportion (amount used by a species/total used) rather than total abundance, as boxes could be added or removed at any point. An additional limitation is that the total number of boxes within the LSJR is unknown and only estimated. Most of the land in the LSJR region is privately owned, and individuals could place nest boxes within or near my study area. In addition, we lack data on of the true usage rates or availability of natural cavities in the region. Attempts were made by Harding (2021) and me to quantify the number of suitable cavities and locate active nests. This proved to be challenging, and no active nests were found in either study. It is certain that natural cavities are being used. I personally observed a female nesting in a large-toothed aspen (*Populus grandidentata*) on the shores of Grand Lake, NB. In communications with others, I have also learned that nesting in natural cavities is still occurring within New Brunswick, although quantifying this within my analysis would not have been possible.

In chapter 3, I hypothesized that the survival rate of juvenile wood ducks is likely lower than that of adult wood ducks due to inexperience with hunters and migration. I predicted that if juvenile wood ducks have increased risks associated with hunting and migration, then survival rates should be lower. While wood duck survival estimates exist within the Atlantic Flyway, others have been done at a comparably small spatial scales, such as a single state (Shirkey and Gates 2020) or are outdated (Johnson et al. 1986), which is problematic as harvest regulations and environmental conditions vary spatially and temporally. In this chapter, I obtained two years of telemetry data and 63 years of band-recovery data to create a survival estimate for wood ducks within the Atlantic flyway. The telemetry (via MOTUS) data from 2021 and 2022 had a high number of false detections, and due to birds migrating when the tags were activated, discerning these from valid detections proved challenging. I therefore sought to augment these data and obtained banding data from the USGS bird banding laboratory to estimate survival in a mark-recapture framework. Estimates of survival appear to differ for all age and sex cohorts between the northern and southern Atlantic Flyway, with northern birds having lower annual survival. This may be attributed to the northern wood ducks being fully migratory compared to the southern wood ducks which are partial migrants (Bellrose and Holm 1994), which may cause them to be more susceptible to harvest. Additionally, in the southern portions of their range, food likely remains more available into the fall when compared with northern regions due to colder temperatures.

In the third time period of my analysis (2002-2022) within the northern portion of their range (which likely approximates that of current wood ducks in New Brunswick, Canada), survival is highest among adult males, then juvenile males, followed by adult females and juvenile females. As age and sex were both covariates within the best fitting model, it is clear that age-specific differences exist, but more pronounced are the sex-specific differences (Figure 3.10). Females appear to have much lower survival rates compared to males, which is consistent with results from Shirkey and Gates (2020) under both a two- and three-bird daily harvest limit. This analysis provides one of the most comprehensive survival estimates within the Atlantic Flyway allowing for the examination of differences in survival over three time periods ranging from 1960 to 2022, and among the northern and southern portions of the flyway.

In chapter 4, I hypothesized that I would observe similar PVA results as Harding (2021), although by adding the results of Chapters 2 and 3, the precision of the results should improve our ability to make inferences. In this chapter, I took the updated survival estimates for wood ducks and the understanding that nesting was positively density dependent for common goldeneyes to create an updated PVA and sensitivity tests. For common goldeneyes, outcomes from the population viability analysis and sensitivity tests showed no discernible difference between reproduction under a density-dependent and non-density-dependent system other than model 1. Model 1 was the nest box only model and therefore constrained the population at a very low (likely unrealistic) stable equilibrium. For wood ducks, the population viability analysis revealed that with the lower survival estimates created using banding data, the updated model under all five scenarios had a lower growth rate and took longer to reach a stable equilibrium. Sensitivity tests confirmed the findings of Harding (2021) that age 0-1 survival was important for population growth, although the updated model also demonstrated the importance of adult female survival and the percent of females breeding.

Since juvenile survival is difficult to manage directly (as described in chapter 4), I suggest that nest boxes, natural cavities, and ducklings should be the focus of conservation and management. In Chapter 2 I found there was no density dependence in wood ducks. Therefore, increasing the number of nest boxes could increase the percent of females breeding. Further, reductions in the predation of these boxes through the use of predator guards and placing boxes on metal poles rather than trees would be beneficial. Another potential solution could be promoting trapping activities, reducing depredation, and therefore increasing hatching success. I also suggest strengthening guidelines for individuals who are given boxes to deploy by organizations such as DUC and Delta Waterfowl. When used properly, nest boxes can provide safe, secure nesting sites for waterfowl to breed, although if placed improperly (e.g., very low to the ground, on large trees, and close to residential areas where raccoons are common) they may be acting as an ecological trap, reducing the production of ducklings in an area.

Duckling survival should also be managed carefully, particularly through the development and maintenance of appropriate brood-rearing marshes. As shown by Dyson et al. (2018), emergent wetlands and those with adjacent forest and/or scrub-shrub were most selected by brood-rearing females. DUC creates and maintains many impoundments and restores damaged wetlands which aids in creating a more suitable and safer habitat for ducklings. While research on ducklings is common, no studies to date have been done in New Brunswick to evaluate their survival. This information would likely be useful within a PVA to give a more local estimate of age 0-1 survival and an understanding of whether ducklings are benefiting from the habitat enhancement and development projects.

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Appendix A:

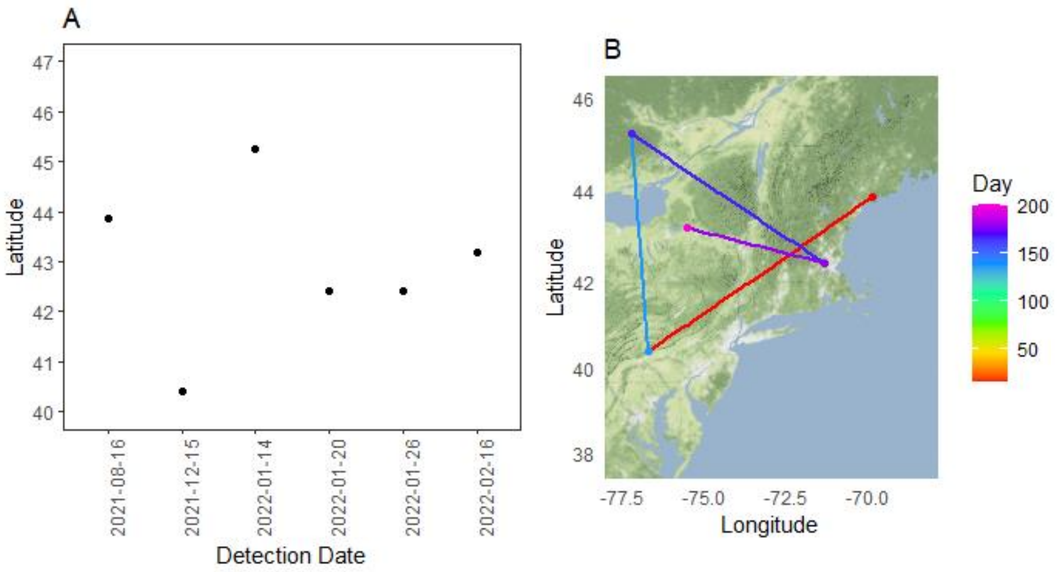


Figure A1. Detections from the MOTUS wildlife tracking system for “Tag 1” (mfgID 504) during the first year (2021-2022) of tagging waterfowl. Panel (A) shows the latitude of the MOTUS tower associated with the detection of this wood duck as a function of the date it was detected. Panel (B) is the longitudinal and latitudinal movements of this wood duck within the Atlantic flyway. Colors are representative of the number of days since the beginning of tagging efforts that year with red showing early in the season and pink indicating later season detections.

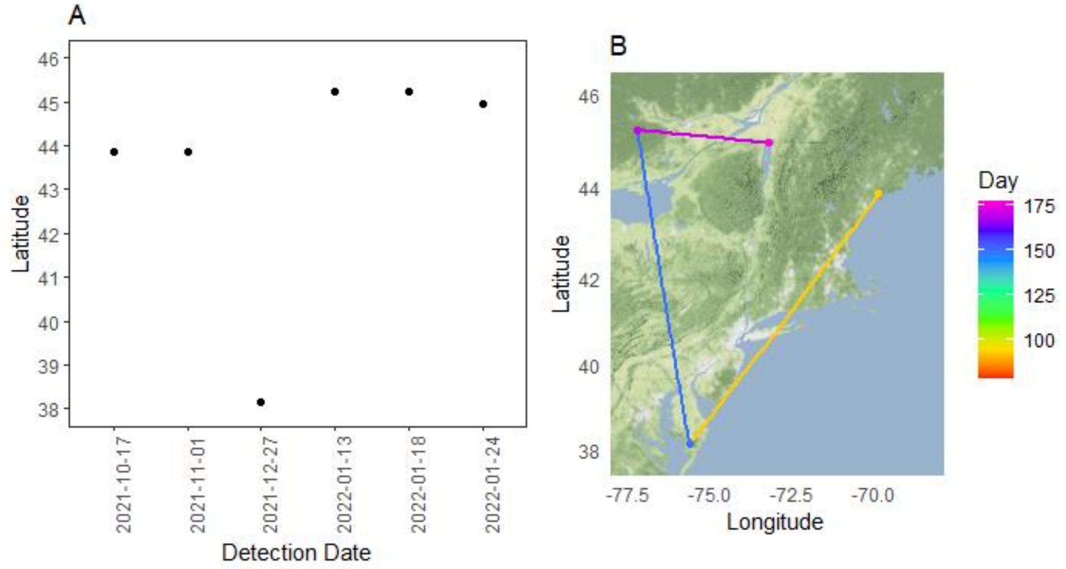


Figure A2. Detections from the MOTUS wildlife tracking system for “Tag 2” (mfID 519) during the first year (2021-2022) of tagging waterfowl. Panel (A) shows the latitude of the MOTUS tower associated with the detection of this wood duck as a function of the date it was detected. Panel (B) is the longitudinal and latitudinal movements of this wood duck within the Atlantic flyway. Colors are representative of the number of days since the beginning of tagging efforts that year with red showing early in the season and pink indicating later season detections.

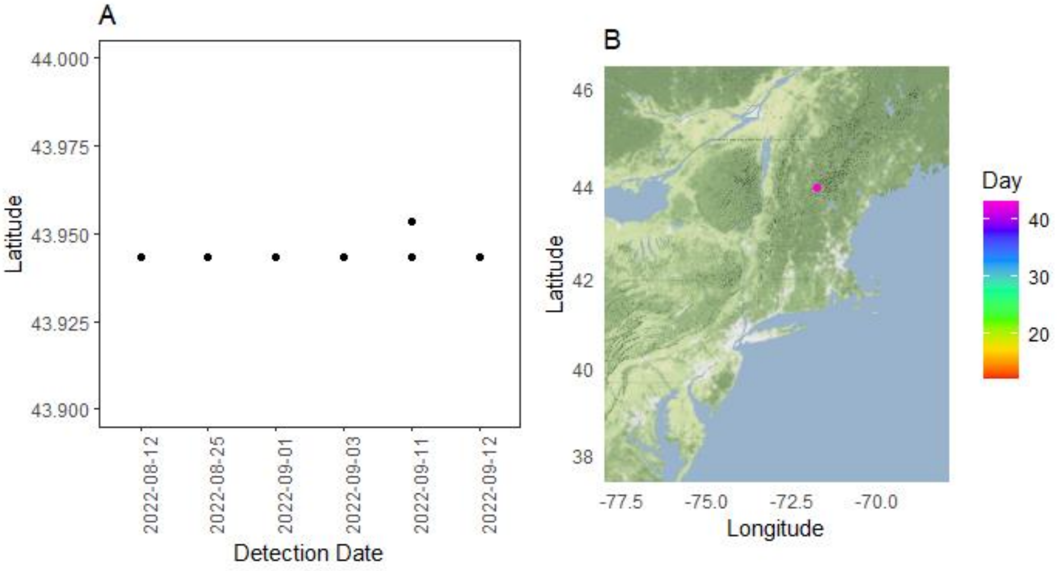


Figure A3. Detections from the MOTUS wildlife tracking system for “Tag 3” (mfID 250) during the second year (2022-2023) of tagging waterfowl. Panel (A) shows the latitude of the MOTUS tower associated with the detection of this wood duck as a function of the date it was detected. Panel (B) is the longitudinal and latitudinal movements of this wood duck within the Atlantic flyway. Colors are representative of the number of days since the beginning of tagging efforts that year with red showing early in the season and pink indicating later season detections. All detections occurred at a similar latitude and longitude and therefore, the color of panel (B) only displays the most recent detection in color (pink).

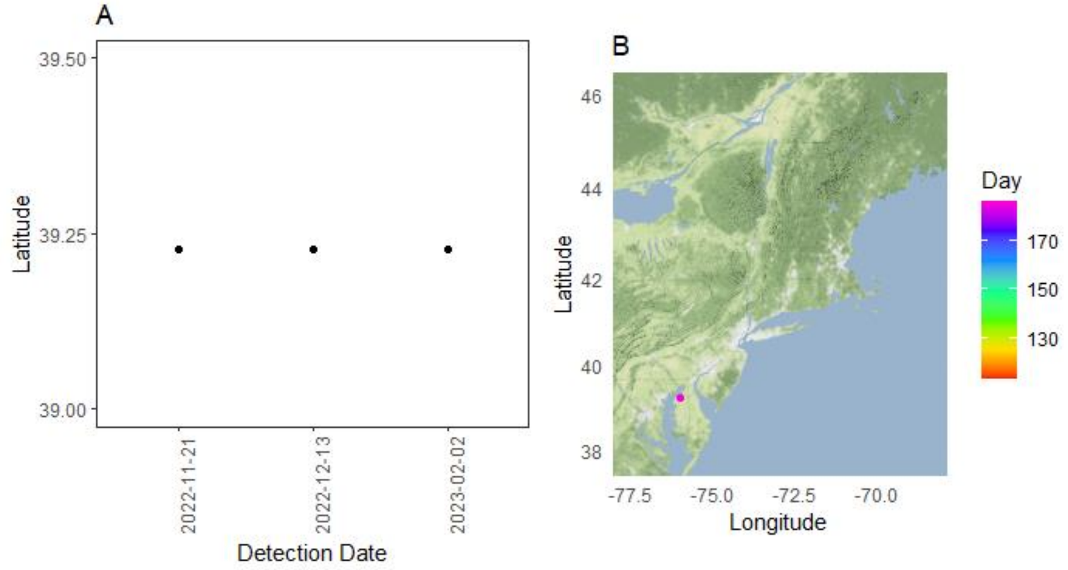


Figure A4. Detections from the MOTUS wildlife tracking system for "Tag 4" (mfID 256) during the second year (2022-2023) of tagging waterfowl. Panel (A) shows the latitude of the MOTUS tower associated with the detection of this wood duck as a function of the date it was detected. Panel (B) is the longitudinal and latitudinal movements of this wood duck within the Atlantic flyway. Colors are representative of the number of days since the beginning of tagging efforts that year with red showing early in the season and pink indicating later season detections. All detections occurred at the same latitude and longitude and therefore, the color of panel (B) only displays the most recent detection in color (pink).

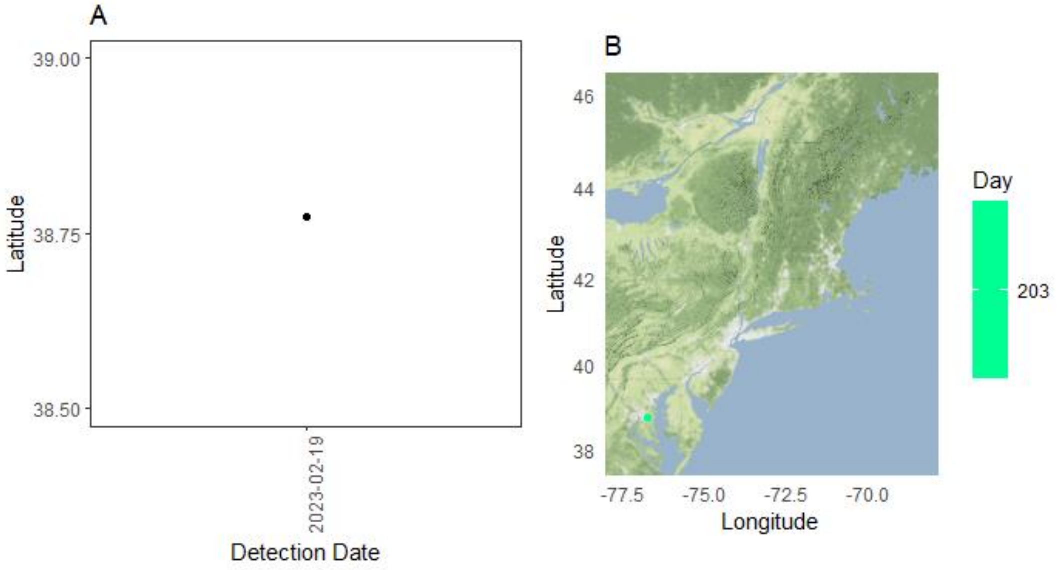


Figure A5. Detection from the MOTUS wildlife tracking system for “Tag 5” (mfID 274) during the second year (2022-2023) of tagging waterfowl. Panel (A) shows the latitude of the MOTUS tower associated with the detection of this wood duck as a function of the date it was detected. Panel (B) is the longitudinal and latitudinal movements of this wood duck within the Atlantic flyway. Colors are representative of the number of days since the beginning of tagging efforts that year with red showing early in the season and pink indicating later season detections.

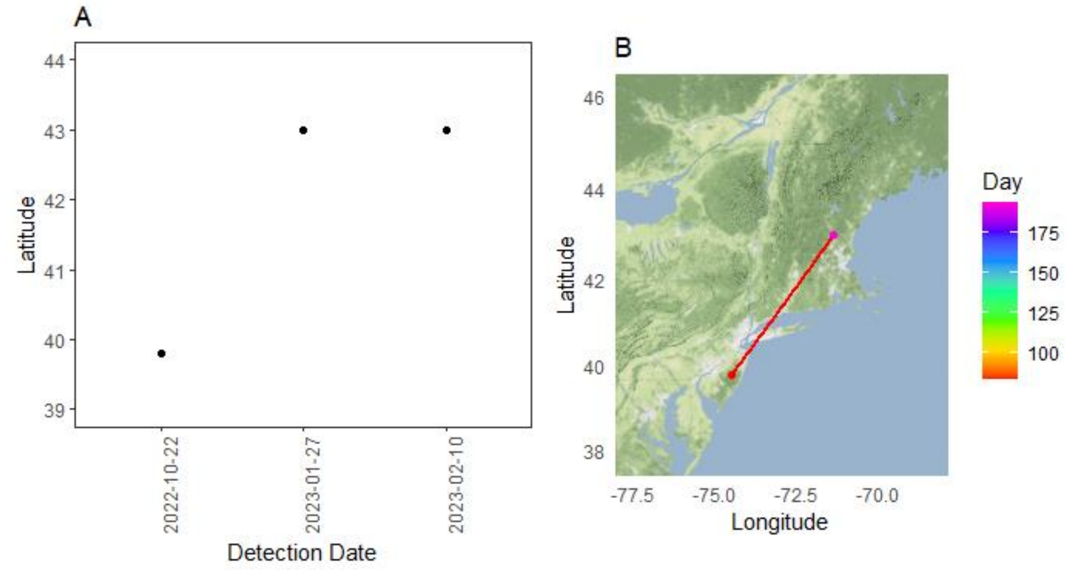


Figure A6**.** Detections from the MOTUS wildlife tracking system for "Tag 6" (mfID 280) during the second year (2022-2023) of tagging waterfowl. Panel (A) shows the latitude of the MOTUS tower associated with the detection of this wood duck as a function of the date it was detected. Panel (B) is the longitudinal and latitudinal movements of this wood duck within the Atlantic flyway. Colors are representative of the number of days since the beginning of tagging efforts that year with red showing early in the season and pink indicating later season detections.

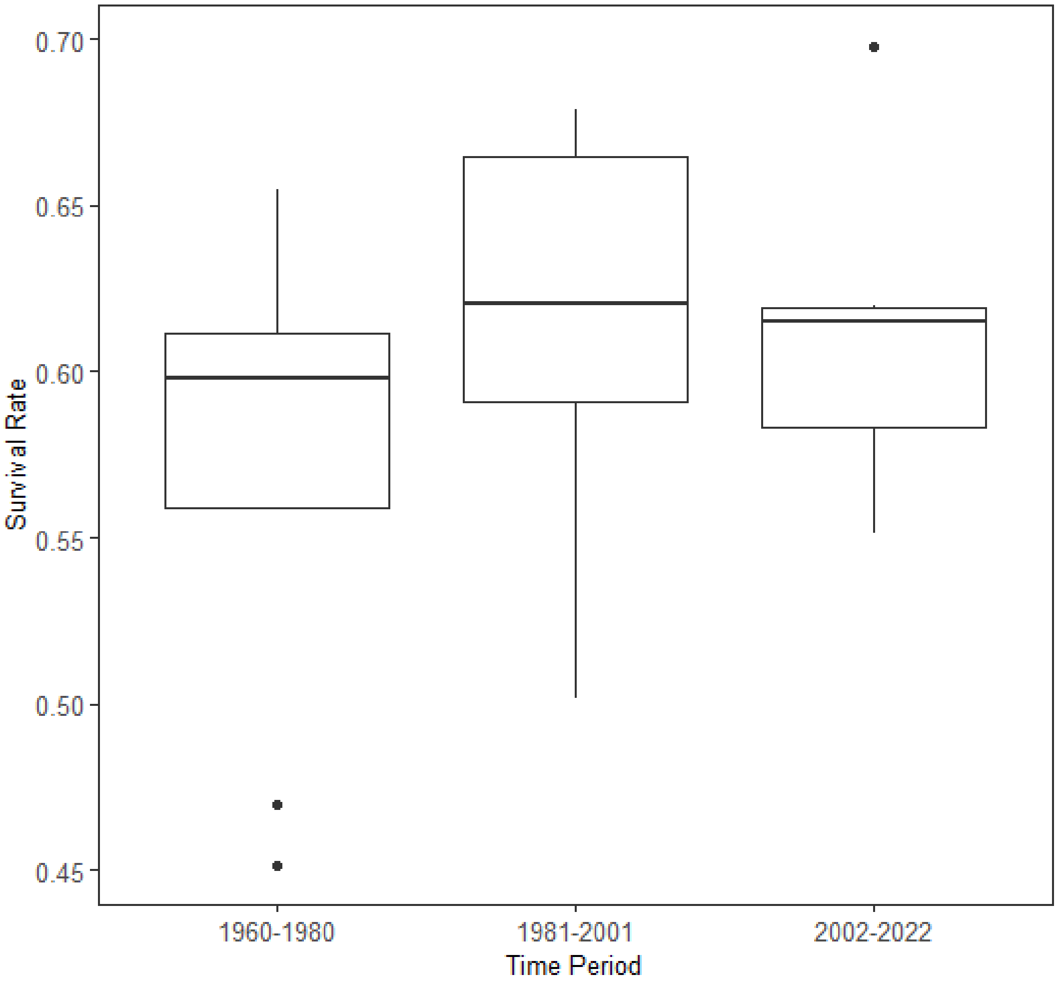
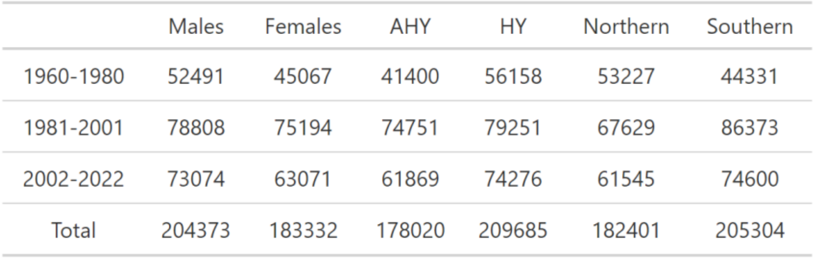


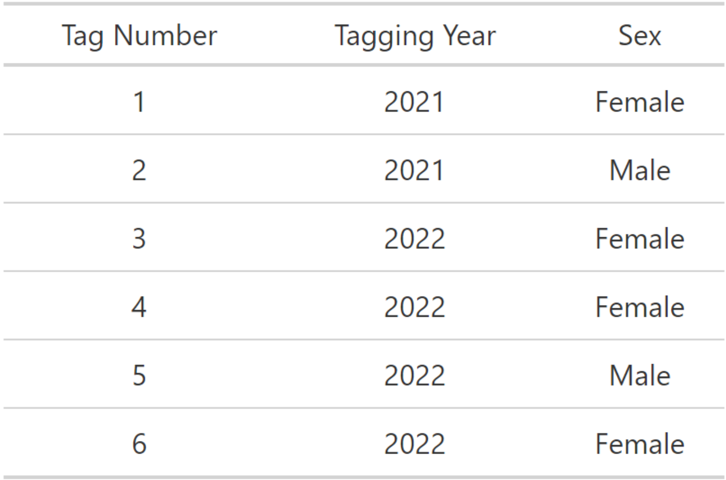
Figure A7. Difference in survival rates between three time periods spanning from 1990-2022 based on hunter returns, and capture-recapture data. Data pooled over the entire Atlantic flyway, all sex and age classes.

Table A1. Band summary for the wood ducks within the Atlantic flyway from 1960-2022 from the USGS bird banding laboratory.



	Males	Females	AHY	HY	Northern	Southern
1960-1980	52491	45067	41400	56158	53227	44331
1981-2001	78808	75194	74751	79251	67629	86373
2002-2022	73074	63071	61869	74276	61545	74600
Total	204373	183332	178020	209685	182401	205304

Table A2. Summary of sex composition of tagged birds detected in 2021 and 2022 from motus towers within the Atlantic flyway.



Tag Number	Tagging Year	Sex
1	2021	Female
2	2021	Male
3	2022	Female
4	2022	Female
5	2022	Male
6	2022	Female

Appendix B:

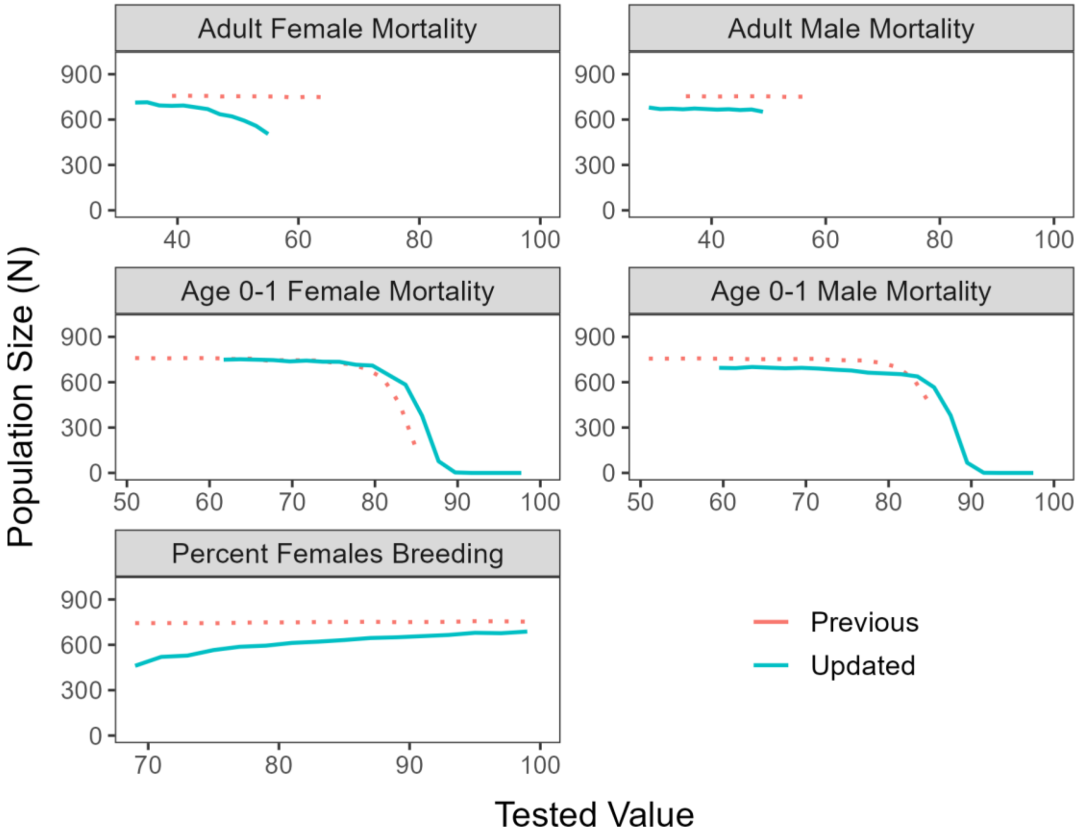


Figure B1. Sensitivity test demonstrating the effect of varying relevant wood duck model-1 parameters by ±25% of base value on population size (N) over 100 years. Includes model-1 under the updated parameterization and the previous model-1 based on Harding (2021) parameterization.

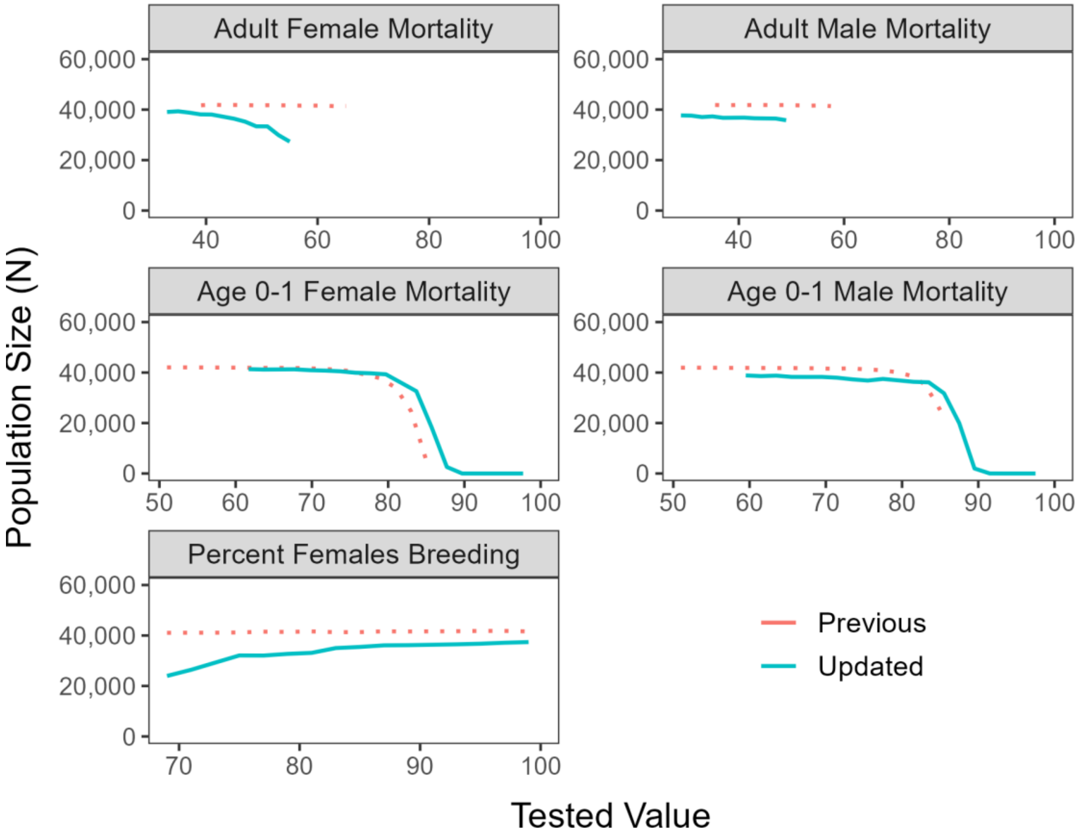


Figure B2. Sensitivity test demonstrating the effect of varying relevant wood duck model-2 parameters by ±25% of base value on population size (N) over 100 years. Includes model-2 under the updated parameterization and the previous model-2 based on Harding (2021) parameterization.

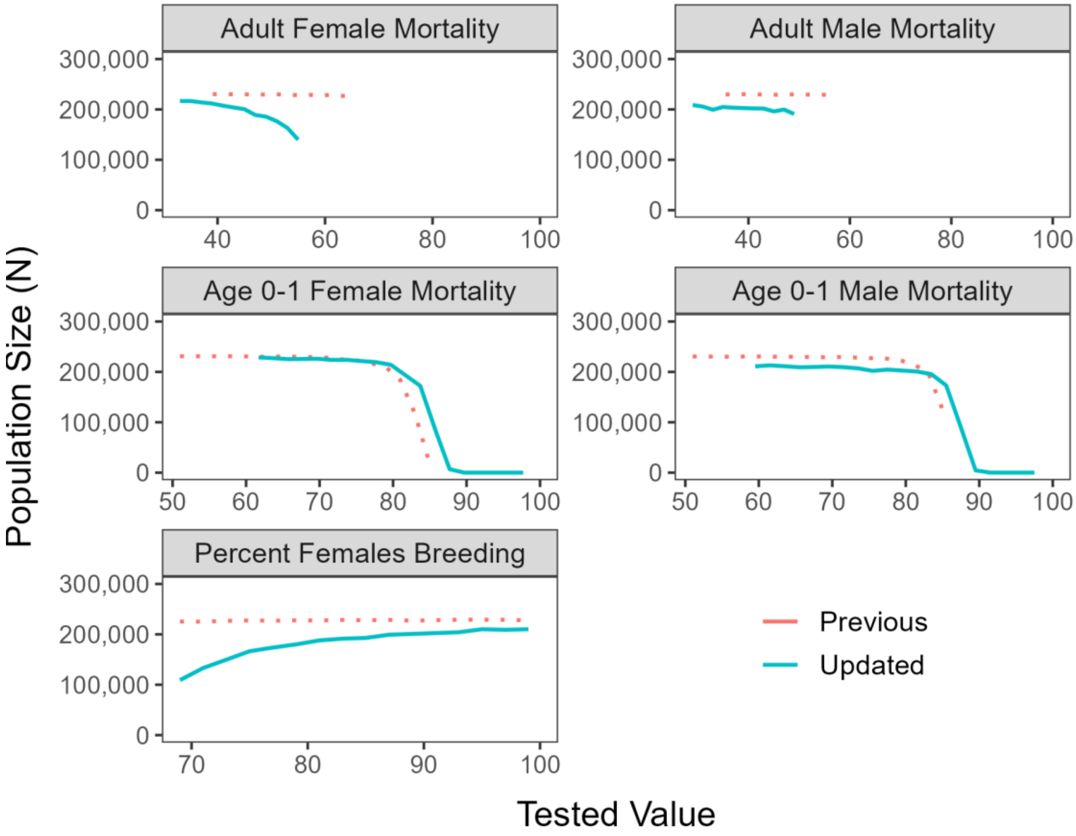


Figure B3. Sensitivity test demonstrating the effect of varying relevant wood duck model-3 parameters by ±25% of base value on population size (N) over 100 years. Includes model-3 under the updated parameterization and the previous model-3 based on Harding (2021) parameterization.

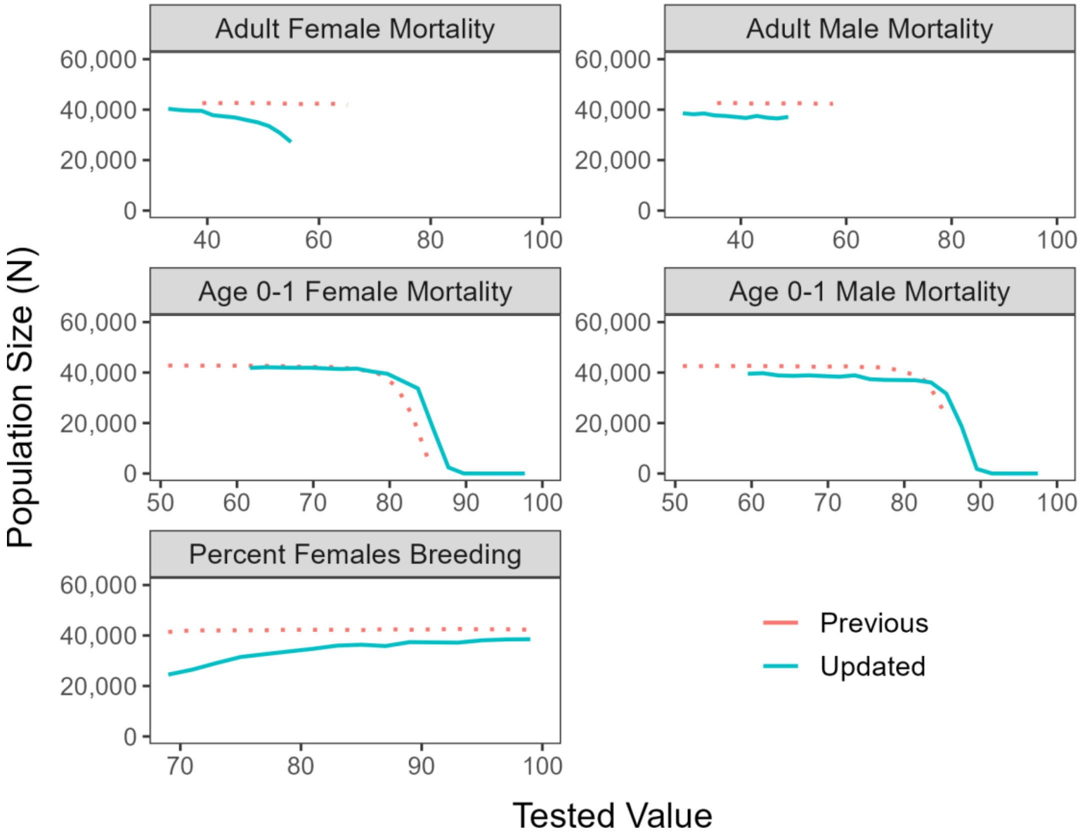


Figure B4. Sensitivity test demonstrating the effect of varying relevant wood duck model-4 parameters by ±25% of base value on population size (N) over 100 years. Includes model-4 under the updated parameterization and the previous model-4 based on Harding (2021) parameterization.

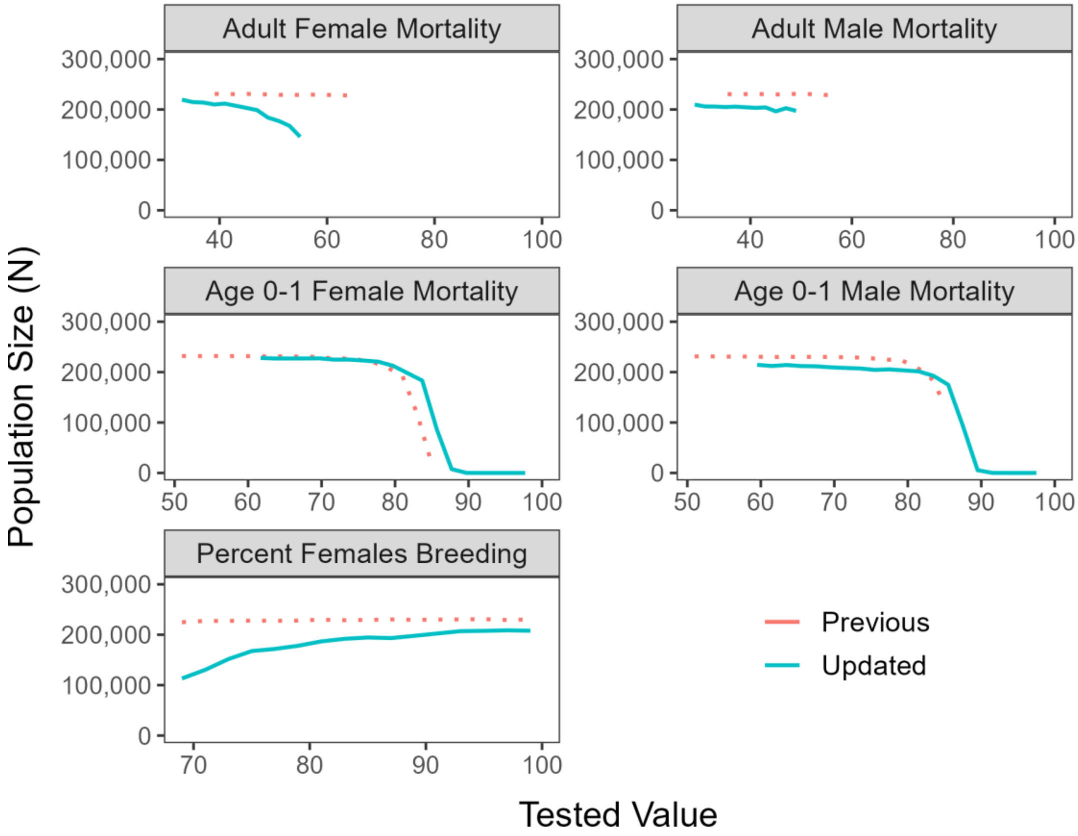


Figure B5. Sensitivity test demonstrating the effect of varying relevant wood duck model-5 parameters by ±25% of base value on population size (N) over 100 years. Includes model-5 under the updated parameterization and the previous model-5 based on Harding (2021) parameterization.

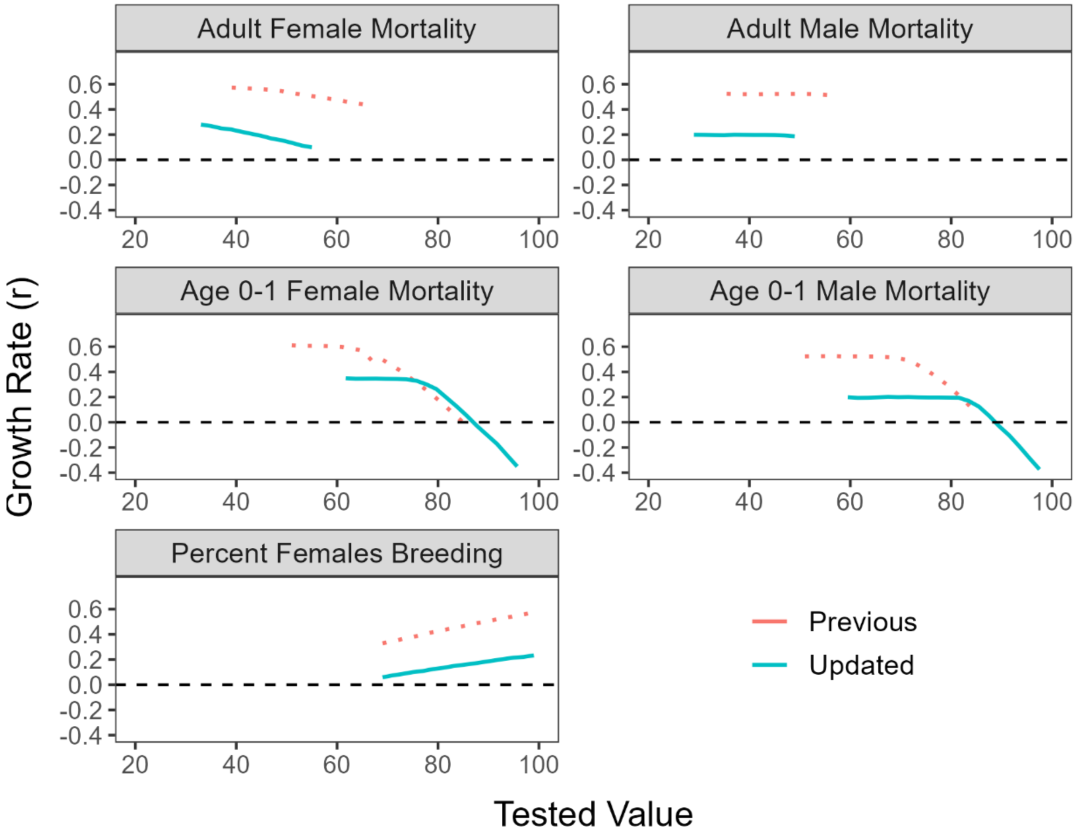


Figure B6. Sensitivity test demonstrating the effect of varying relevant wood duck model-1 parameters by ±25% of base value on growth rate (r) over 100 years. Includes model-1 under the updated parameterization and the previous model-1 based on Harding (2021) parameterization.

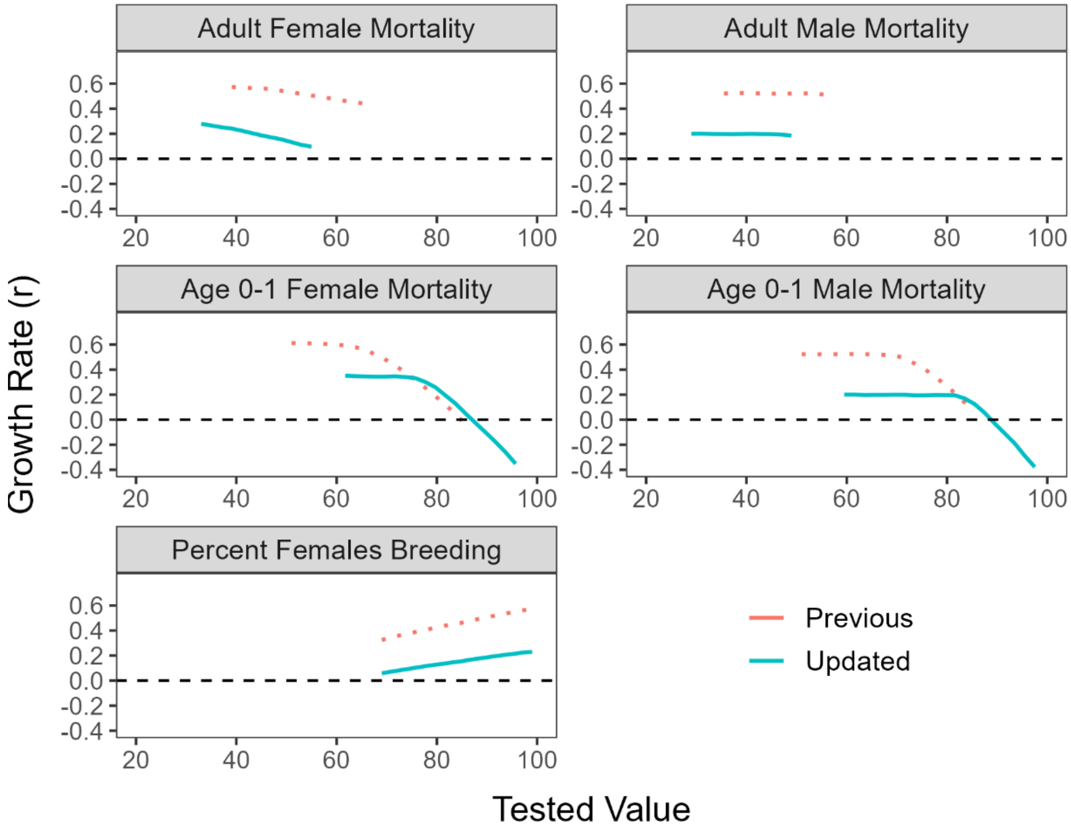


Figure B7. Sensitivity test demonstrating the effect of varying relevant wood duck model-2 parameters by ±25% of base value on growth rate (r) over 100 years. Includes model-2 under the updated parameterization and the previous model-2 based on Harding (2021) parameterization.

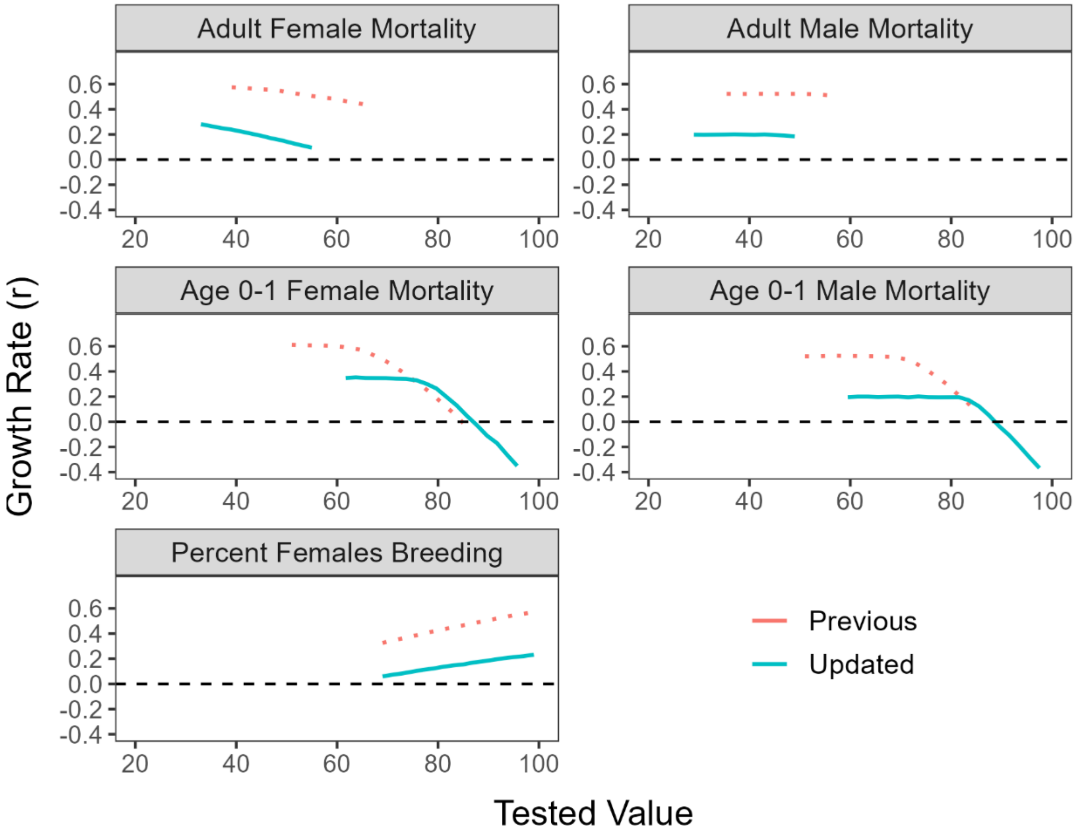


Figure B8. Sensitivity test demonstrating the effect of varying relevant wood duck model-3 parameters by ±25% of base value on growth rate (r) over 100 years. Includes model-3 under the updated parameterization and the previous model-3 based on Harding (2021) parameterization.

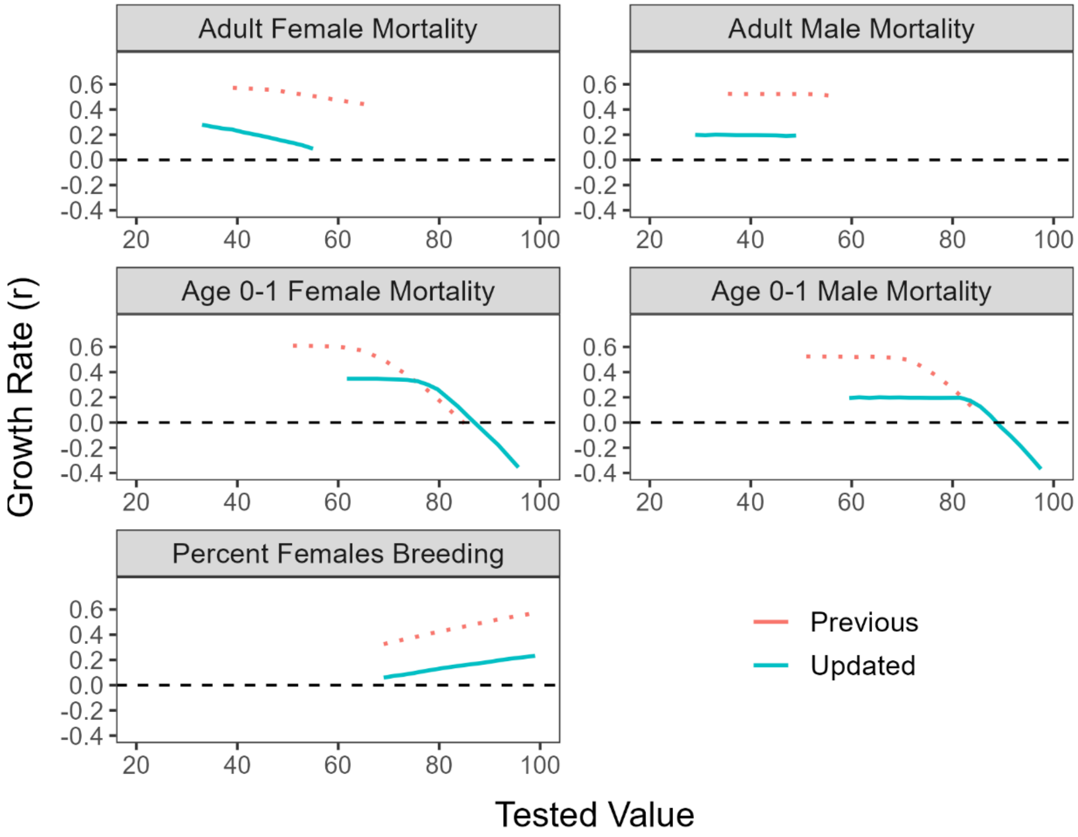


Figure B9. Sensitivity test demonstrating the effect of varying relevant wood duck model-4 parameters by ±25% of base value on growth rate (r) over 100 years. Includes model-4 under the updated parameterization and the previous model-4 based on Harding (2021) parameterization.

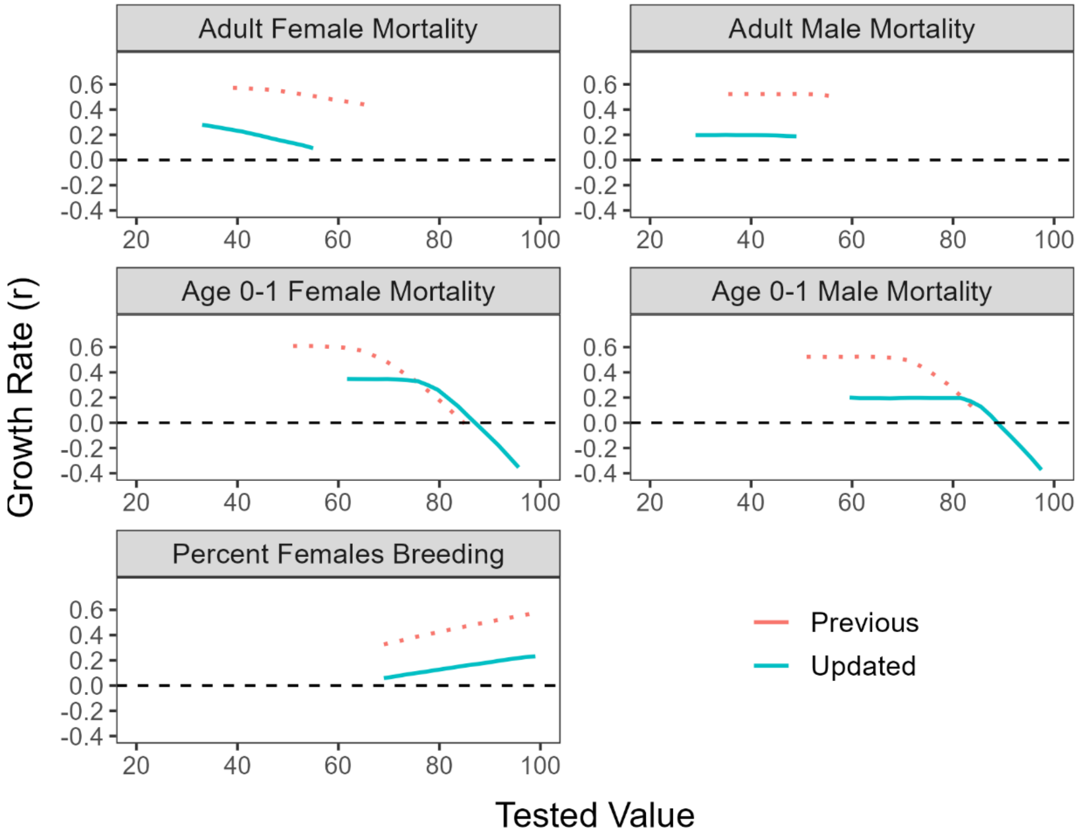


Figure B10. Sensitivity test demonstrating the effect of varying relevant wood duck model-5 parameters by ±25% of base value on growth rate (r) over 100 years. Includes model-5 under the updated parameterization and the previous model-5 based on Harding (2021) parameterization.

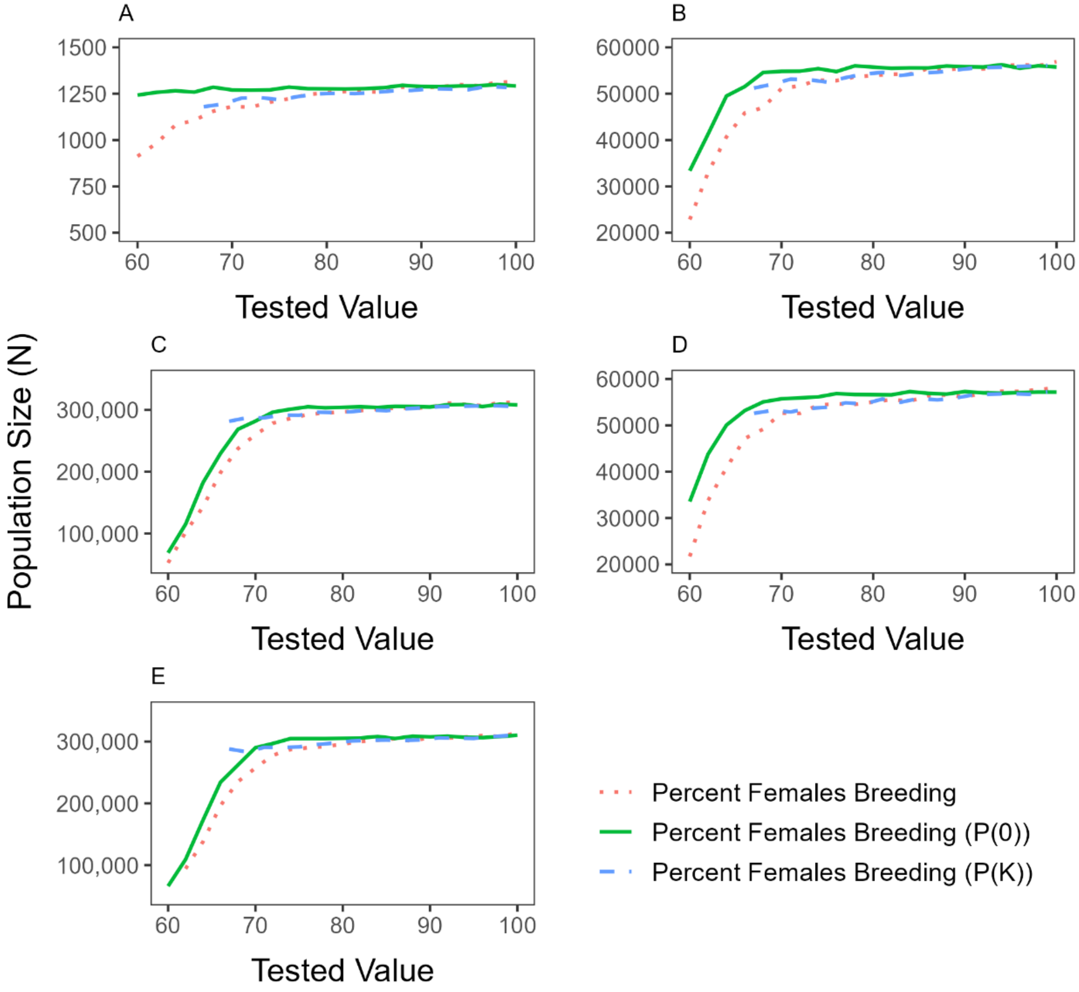


Figure B11. Sensitivity test demonstrating the effect of varying relevant common goldeneye model-1 parameters by ±25% of base value on population size (N) over 100 years. A) nest box only model, B) low cavity density model, C) high cavity density model, D) low cavity density, and nest box model, and E) high cavity density, and nest box model. Differences in models are expressed through estimates of carrying capacity (K).

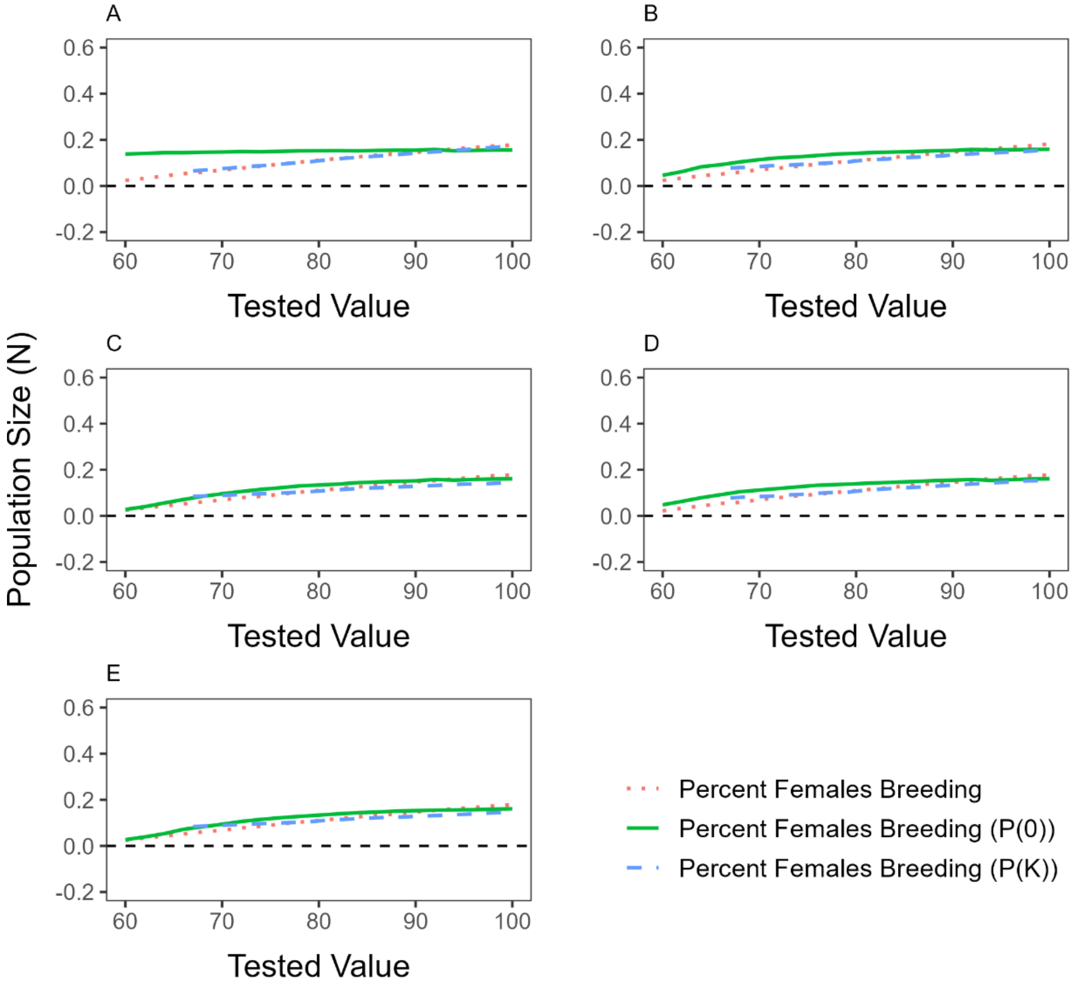


Figure B12. Sensitivity test demonstrating the effect of varying relevant common goldeneye model-5 parameters by ±25% of base value on growth rate (r) over 100 years A) nest box only model, B) low cavity density model, C) high cavity density model, D) low cavity density, and nest box model, and E) high cavity density, and nest box model. Differences in models are expressed through estimates of carrying capacity (K).

Curriculum Vitae

Candidates full name: Jared Sonnleitner

Universities attended (with dates and degrees obtained): Honours Bachelor of Science in Ecology and Environmental Biology, Thompson Rivers University (2016-2021)

Publications:

Pageau, C., J. Sonnleitner, C.M. Tonra, M. Shaikh, and M.W. Reudink. 2021. Evolution of winter molting strategies in European and North American migratory passerines. Ecology and Evolution 11(19): 13247-13258.

Prytula, E.D., M.W. Reudink, S.E. LaZerte, J. Sonnleitner, and A.E. McKellar. 2023. Shifts in breeding distribution, migration timing, and migration routes of two North American swift species. Journal of Field Ornithology 94(3): 14.

Sonnleitner, J., S.E. LaZerte, A.E. McKellar, N.J. Flood, and M.W. Reudink. 2022. Rapid shifts in migration routes and breeding latitude in North American bluebirds. Ecosphere 13(12): e4316.

Conference Presentations:

Sonnleitner, J., N. McLellan, and J. Nocera. 2022. How nesting and hatch-year survival influence population growth of wood ducks (*Aix sponsa*) in New Brunswick. Presentation at Atlantic Migratory Game Bird Technical Committee.

Sonnleitner, J., N. McLellan, and J. Nocera. 2022. Development of a PVA for wood duck (*Aix sponsa*) management in New Brunswick. Presentation at Atlantic Society of Fish and Wildlife Biologists.

Sonnleitner, J., N. McLellan, and J. Nocera. 2022. Improving cavity-nesting duck populations in New Brunswick using a PVA. Poster at the Wildlife Society Conference.

Sonnleitner, J., N. McLellan, and J. Nocera. 2023. Cavity-nesting ducks: Is nesting density-dependent? Presentation at the American Ornithological Society – Society of Canadian Ornithologists Conference.

Sonnleitner, J., N. McLellan, and J. Nocera. 2023. Survival of wood ducks (*Aix sponsa*) in the Atlantic Flyway and effects on population viability. Presentation at Atlantic Society of Fish and Wildlife Biologists.