Diving-duck Productivity: Effects of Predator Management on Nest Success and Investigating Sightability-adjusted Brood-pair Ratios

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Abstract

Nest success, a major component of productivity, is often used as the metric to measure the effectiveness of various management efforts aimed at increasing waterfowl productivity. Although numerous studies have proven predator reduction increases nest success for upland nesting waterfowl, less is known about its effects on the over-water nesting guild (i.e. diving ducks), which there exists no current management specifically for over-water nesting ducks. From 2015-2017 in the Prairie-Parkland Region of Manitoba, we assessed daily-survival rates of over-water duck nests in areas where efforts to reduce the local predator community were being coordinated and compared them to nearby areas where no targeted predator management occurred. Given the challenges in locating over-water nests, we also investigated an alternative method to estimate productivity using multiple rounds of surveys to derive brood-pair ratios. Brood-pair ratios have been widely used to index productivity, but biases associated with detection probabilities (the probability a pair or brood is seen during a survey) can result in underestimating abundances, especially for broods. We conducted replicate surveys to estimate detection probabilities of broods hatched from over-water nests to include in brood-pair ratios and compared productivity estimates derived from adjusted brood-pair ratios with estimates calculated from nest success on the same sites. We located and monitored 1,673 over-water nests from a variety of duck species to derive daily-survival rates and nest success estimates using Shaffer's logistic-exposure methods and included a variety of covariates hypothesized to influence the probability a nest was successful. Nest success ranged from 14-48% across trapped and control sites, yet no overall trapping effect was observed despite numerous predators being removed from the landscape. Temporal effects such as nest-age and initiation date were influential predictors of daily-survival rates, which increased with nest-age and as the nesting season progressed. Detection probabilities for broods were estimated from 1,915 unique encounter histories using Huggin’s closed-capture methodology, which also incorporated covariates hypothesized to influence detectability. Detection probabilities were >50% for broods during all survey rounds and most heavily influenced by the percentage of the inundated wetland unobstructed for viewing broods. Sightability-adjusted brood-pair ratios for single-species were weakly correlated with nest success, however, combining all diving-duck species resulted in strong correlations between sightability-adjusted brood-pair ratios and nest success on each site-year ($n = 18$, $R^2 = 50\%$, $P = 0.0005$). Therefore, sightability-adjusted brood-pair ratios when multiple species are combined provide a useful alternative to index local diving-duck productivity when estimating nest success is unfeasible.

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Assessment of Predator Reduction on Nest Success in Overwater Nesting Ducks

***Abstract* -**Although numerous studies have demonstrated that lethal predator management improves nesting success of upland nesting waterfowl, less is known about its potential effect on overwater nesting species. We assessed daily survival rates of overwater nesting duck nests in the Prairie-Parkland region of southwestern Manitoba during 2015-2017. Professional trappers removed known waterfowl nest predators such as skunk (*Mephitis mephitis*), red fox (*Vulpes fulva*), and badger (*Taxidea taxus*) in addition to focused efforts targeting raccoon (*Procyon lotor*), American mink (*Neovison vison*), and corvids (*Corvus corax*, *Corvus brachyrhynchos, Pica hudsonia*), between the months of March and July when local breeding occurs. Each year, 3 65-km^2 study sites were subjected to intensive trapping whereas 3 equally-sized but untrapped sites served as controls. We located and monitored 504 canvasback (*Aythya valiseneria*) and 422 American redhead (*Aythya americana*) nests, as well as 747 overwater nests of other species, including lesser scaup (*Aythya affinnis*), ring-necked duck (*Aythya collaris*), ruddy duck (*Oxyura jamaicensis*), and mallard (*Anas platyrhynchos*). We compared daily survival rates, estimated using Shaffer's logistic-exposure method, between trapped and control sites to assess the effect of predator reduction on nest success. We found no positive effect of trapping in nest success models. Daily survival rates were positively influenced by temporal effects such as initiation date and nest age, and the presence of infrared trail cameras at nest-sites to identify predators also had a positive effect. Failure to document a treatment effect on nest success was unexpected and indicates that in comparison to upland nests, targeted trapping at this scale was ineffective at increasing nest success for the overwater nesting guild.

Introduction

Waterfowl populations are stochastic and influenced by climactic events on a regional scale driven by the abundance of water (Leitch 1964). Regions boasting the highest densities of breeding waterfowl contain a diversity of wetlands that vary in size and seasonality, providing high-quality systems where adequate resources exist throughout the waterfowl nesting season. The Prairie Pothole Region (PPR) has been identified as the most important region of the United States and Canada that provides habitat for North American upland nesting ducks (Bellrose 1980; Batt et al. 1989). Agricultural practices throughout the PPR have changed the landscape considerably, resulting in reduced waterfowl nesting habitat due to grassland conversion and wetland drainage (SOCE 1991; Watmough and Schmoll 2007). Large scale federal policies in the United States, like the Conservation Reserve Program and Wetland Reserve Program, aim to mitigate waterfowl habitat loss through subsidy payments to private landowners willing to convert cultivated land to perennial cover, or restore wetlands systems. Similar policies, however, do not exist in much of Canada (Kennedy and Mayer 2002), and participation in the United States has waned due to rising commodity prices and reduced area caps, justifying alternative techniques to increase waterfowl productivity be investigated (Stubbs 2014).

Nest success (the probability that a clutch of egg survives long enough for at least one to hatch) has been identified as the most important vital rate affecting waterfowl productivity (Cowardin et al. 1985; Klett et al. 1988; Hoekman et al. 2002), and predation has been reported as the most substantial cause of nest failure in areas with high densities of nesting waterfowl (Stoudt 1982; Serie et al. 1992; Greenwood et al. 1995; Sargeant and Raveling 1992; Maxson and Riggs 1996; Arnold et al. 2007). To reduce nest predation, lethal predator removal via trapping has been investigated at various experimental scales leading up to, and during the summer nesting season. Successful implementation of predator reduction when focused on upland nest success resulted in success ranging from 24% to 66%, with the average difference being 20% higher than areas experiencing no predator reduction (Garrettson et al. 1996; Hoff 1999; Garrettson and Rohwer 2001; Pieron and Rohwer 2010; Dassow 2010).

The Prairie Parkland Region of the PPR including portions of southwestern Manitoba, central Saskatchewan, and east-central Alberta is an increasingly important waterfowl breeding area (Prairie Habitat Joint Venture 2014; Barker et al. 2014). Estimates of upland nest success from investigations in this biome range from 2--16% (Arnold et al. 1993; Koons and Rotella 2003; Emery et al. 2005; Delta Waterfowl Foundation, unpubl. report), suggesting high predation rates resulting in nest success rarely meeting the 15% threshold considered necessary to maintain a stationary population trend (Cowardin and Johnson 1979). Even though predation appears high, this transitional ecozone where the Canadian Great Plains meet the Western Boreal Forest is comprised of high wetland densities, including many semi-permanent and permanent wetlands (National Wetlands Working Group 1997) that provide hydrological stability important to overwater nesting ducks for nesting and brood rearing (Bellrose 1980). This guild of waterfowl, which contains most of North America's "diving ducks", and makes up 15% of the continental duck population, dives to access food and constructs floating nests within emergent vegetation in wetlands (Bellrose 1980; USFWS 2017). Species abundance models predict densities of overwater nesting ducks in Canada to be highest in the Prairie Parkland Region (Barker et al. 2014). Since the mid-1990's, much of the Prairie Parkland Region has received above-average precipitation providing adequate nest site availability for overwater nesting ducks, however, these conditions have not resulted in increased population sizes for some species as expected (Leitch and Kaminski 1985; Prairie Habitat Joint Venture 2014). Two species within this group breeding extensively throughout the Prairie Parkland Region, canvasback (*Aythya valisineria*) and lesser scaup (*Aythya affinis*), have stationary and declining populations indicating that there may be additional productivity constraints beyond nest site availability during the breeding season (Péron and Koons 2012; Arnold et al. 2016; USFWS 2017).

Female canvasbacks exhibit some of the highest rates of philopatry among duck species, with 76% of adults and 27% of juveniles returning to the previous year's breeding site (Anderson 1986; Serie et al. 1992; Anderson et al. 1997). Previous attempts to increase productivity of canvasbacks used egg clutch manipulation to counter egg-loss caused by interspecific parasitism, and predator diversion fences to increase nest success, resulting in local population growth during following years as indicated by increases in breeding pair abundances (Anderson et al. 1997). However, there was uncertainty as to which management technique was most responsible for population growth as these treatments were confounded within individual experimental sites (Anderson et al. 1997).

Currently, there exists no targeted management efforts that specifically focus on increasing the productivity of overwater nesting ducks. Based on the effect predator reduction has had on nest success of ducks in upland systems, we hypothesized that daily survival rates of nests for overwater nesting ducks, and subsequent nest success would be higher on sites that experienced predator reduction (via trapping). We included additional covariates in our survival models that we hypothesized would influence daily survival rates to identify which nests were most vulnerable to nest mortality. Lastly, we utilized trail cameras placed at overwater nest sites to identify which species were the primary predators of overwater nests in southwestern Manitoba and hypothesized that predators tied closely to wetland systems were the primary sources of nest mortality.

Study Area

We conducted this study from 2015 through 2017 in the Prairie Parkland Region of southwestern Manitoba, Canada in the rural municipalities of Rossburn, Odanah, Shoal Lake, Hamiota, and Birtle. This area has been described in detail elsewhere, most notably by Kiel et al. (1972) and Stoudt (1982). The region is best characterized by slightly rolling topography with high densities of wetlands ranging in size from <0.2 ha to >250 ha. Land-use practices vary due to differences in soil conditions, but cereal grains, oilseed, and row-crop agriculture dominates this landscape; mixed perennial cover represented as haylands and pasture accounts for <30% of land cover. Stands of aspen (*Populus* spp.) and oak (*Quercus* spp.) grow where agriculture is not practical due to rocky soil, sloped topography, or high-water levels. Land ownership is ~97% private in these municipalities with Ducks Unlimited Canada, The Nature Conservancy, and Manitoba Habitat and Heritage Corporation as primary owners of the remaining ~3%.

The predator community inhabiting this region is diverse and includes predators of the Western Boreal Forest and Prairie Ecozones. Both coyote (*Canis latrans*) and red foxe (*Vulpes fulva*) occur in the study area along with other known mammalian waterfowl predators such as American mink (*Neovison vison*), striped skunk (*mephitis mephitis*), Franklin’s ground squirrel (*Poliocitellus franklinii*), and American badger (*Taxidea taxus*). Potential avian predators of waterfowl and nests include American crow (*Corvus brachyrynchos*), black billed magpie (*Pica hudsonia),* red-tailed hawk (*Buteo jamaicensis*), northern harrier (*Circus cyaneus*), Swainson’s hawk (*Buteo swainsoni*), and great horned owl (*Bubo virginianus*).

Methods

We selected 6 study sites encompassing 64.75 km² each (25 sq. miles); 2 near the town of Minnedosa, Manitoba (50°12’ N, 99°47’ W) and 4 in proximity to Shoal Lake, Manitoba (50°26’ N, 100°34’ W), where high densities of breeding, overwater nesting ducks were known to occur.

We assessed overwater daily survival rates and nest success at these sites from 2015 through 2017. Wetlands near Minnedosa were mostly Class II, III, and IV wetlands (Stewart and Kantrud 1971), remaining mostly inundated throughout the breeding season, ranging from <2--10 ha in size with vegetation buffers 10--30 meters wide of cattail (*Typha* spp.) and bulrush (*Scirpus* spp.) around the edges. The 4 remaining sites near Shoal Lake also contained Class II, III, and IV wetlands, but wetland complexes were more variable in size (<2--250 ha) and the landscape had lower relief. The low-relief landscape resulted in higher connectivity of wetland basins promoting moving water with less uniform buffers, eliminating the patchwork "pothole" assemblage common to the Minnedosa portion of our study area.

Nest-Predator Trapping

Delta Waterfowl Foundation hired 3 professional trappers to target American mink, raccoon, American crow, common raven, and black-billed magpie on 3 sites (1 near Minnedosa and 2 near Shoal Lake). Trapping occurred each year between 15 March and 15 July, during the primary nesting season for overwater nesting ducks on our study sites. Trappers deployed their first traps in locations accessible by vehicle, typically along roadsides or near abandoned farmsteads. They subsequently deployed additional traps farther from roads following spring snow melt. Trapping methodology and effort changed each year based on preliminary nest success results from the previous year to meet new objectives set by Delta Waterfowl Foundation (M. Buxton, personal communication). During 2015, 1 of the contracted trappers had ≥15+ years' trapping experience with Delta Waterfowl Foundation and mentored new trappers to maintain consistent trapping effort with what had been found to increase nest success in previous upland nesting investigations (M. Buxton, personal communication). Trappers used a variety of trap types, including conibear box sets, live-traps, and dog-proof "coon-cuffs", and dispatched all targeted predators captured. Changes for 2016 included trap-count benchmarks to be met by trappers (150 sets by 7 April, 300 sets by 7 May), adjusted placement of traps, and an increase in the diversity of traps used. Trappers were encouraged to extend effort into remote areas that may have functioned as refugia for predators. Observations of ravens removing eggs from overwater duck nests in 2015 and the location of a raven egg-cache containing >70 eggs from overwater nests in 2016 provided evidence to support diversifying trap types to heavily target corvids. Trappers added ladder-style and conical wire-cage avian traps in 2016, and in 2017 Delta Waterfowl Foundation employed a fourth trapper to solely target corvids on all 3 trapped sites. No predator trapping occurred on the 3 remaining sites, which served as controls for comparative analysis. Trapped and control sites remained consistent for the duration of the study, however, 1 trapped site was moved 35 km northeast due to access being withdrawn by landowners.

Wetland Inundation

Observers recorded observations of wetland inundation levels during scheduled indicated breeding pair and brood abundance surveys conducted on all plots during each year of the study as part of related research at these same sites. Timing of these surveys corresponded to breeding phenology of canvasbacks, the earliest initiating overwatering nesting species (31 April - 15 May). On each survey plot ($n = 150$), observers recorded an ordinal estimate of the proportion of the wetland inundated ranging from totally dry (0) to water extending outside the basin (5), at 24% intervals (i.e., 1 = 1-25% inundated, 2 = 26-50%, 3 = 51-75%, 4 = 76-100%).

Nest Searching and Monitoring

A ground crew of observers searched for nests of overwater nesting ducks on a portion of all sites. Each site was divided into 2.59-km² (1 sq. mile) sections following the Dominion Land Survey System of rural Canada, resulting in 25 sections per site. Given the length of the nesting season and the time required to locate overwater nests, we randomly selected a subsample of quarter-sections (64.75 ha/160 acres) to be search plots because not all wetlands on a site could be searched in a year with the available effort. Plots cumulatively accounted for 25% of the area within each site and remained unchanged across years unless landowner permission to access a plot was withdrawn. In this event, we randomly chose a new plot from the remaining quarter-sections within the section, to maintain a similar distribution of sampling effort within each site. Nest searching occurred 1-2 times on these plots between 9 May and 30 June each year to maintain consistent effort across all sites.

Observers commonly worked in teams of 2, systematically searching the inundated vegetative buffers of wetlands for overwater duck nests. Observers monitored all overwater duck nests located regardless of species, with a requirement that water depth be ≥10 cm at the edge of the nest bowl. Upon locating a nest, observers recorded a GPS point and determined the incubating species by either identification of the female, egg characteristics, or feathers present in the nest. During each visit, observers candled all eggs to determine nest age, initiation date, and nest viability (Weller 1956). Once all data were collected, observers covered eggs with nesting material and left the nest in the direction of open water to reduce trails possible trails used by predators. In some cases, especially late in the nesting season (mid-June through early July), new growth of vegetation made relocating nests challenging so observers placed a small piece of flagging tape on nearby vegetation in a random direction and distance from the nest to aid subsequent nest relocation. Observers monitored nests at 7--10 day intervals until eggs hatched or a nesting attempt failed, and the cause of nest failure was determined by assessing nest contents and timing nest visits to occur following projected hatch dates as suggested by Lariviere (1999). An intact nest bowl including eggshell caps or membranes indicated a successful hatch, whereas many small (1 cm) eggshell fragments, the presence of blood or yolk in the nest bowl, or a physically destroyed nest, were all signs that a nest had been depredated. Nest fates were assigned as successful (≥ 1 egg hatched), abandoned (no advancement in incubation stage as determined by candling since the previous visit), or depredated, during the final visit.

Trail Cameras

Observers assessed each nest upon initial location as a candidate for trail camera deployment to identify nest predators. Ideal candidates were nests where eggs were >7 days into incubation to minimalize the likelihood of abandonment, and a clear line of site was evident through minimal disturbance to vegetation in the immediate vicinity of the nest to not compromise nest concealment (personal obs., Caldwell et al. 2013). We re-evaluated whether nests were candidates for trail camera deployment during subsequent nest checks and observers deployed either a Bushnell Trophy Cam or Bushnell Ninja Cam (Bushnell Outdoor Products, Overland Park, KS, USA) 1-3 m from nests suitable for camera placement programmed to take photos at 1-min intervals. Observers replaced batteries and digital memory cards in cameras during nest checks on the same nest monitoring interval of 7--10 days. We reviewed photographs to identify predators if there was evidence that a predator had visited the nest (i.e., the nest was destroyed, yolk or blood was present in the nest bowl, or there was a change in clutch size). In the event multiple species of predators contributed to a single nest failure, we assigned the depredation event to the species most responsible for contributing to the destruction or eventual abandonment of the nest.

We conducted nest monitoring under scientific permit granted by the Canadian Wildlife Service (permit number 15-MB-SC001 issued April 2015 and amended May 2017). Our Animal Care and Use protocol for 2015-2016 was approved by Louisiana State University (permit number A2014-03), and the University of Minnesota in 2017 (permit number 1704-34715A). Predators were trapped and removed in accordance with Manitoba Department of Sustainable Development permits WB17115 (2015), WB18524 (2016), and WB20141 (2017).

Statistical Analysis

During this study, we documented and monitored all suspected nesting attempts at least once. Locating an active overwater nest does not rely on the physical presence of an incubating female, and therefore may require ≥1 revisit to confirm nest status. Finding overwater nests in the laying stage is common and nests during this period often appear disheveled or unmaintained, much like an abandoned or failed nest. Nest abandonment occurs for many reasons, and community dump-nesting was suspected in instances where brood parasitism was high and no eggs were destroyed; therefore, we removed nests from our analyses with no evidence of an incubating female after ≥1 revisit even when the nest had not been depredated. Using trail cameras, we documented numerous events where muskrats (*Ondatra zibethicus*), Canada geese (*Branta canadensis*), and American coots (*Fulica americana*) were the cause of nest failure or abandonment. We considered that these nests failed due to "displacement", which we defined as the destruction or subsequent abandonment of a nest caused by an organism with no intentions of consuming the eggs or the incubating female. When either nest failure or nest abandonment was suspected to be caused by the observer or due to displacement, we censored the final nest exposure interval in our survival analysis as these causes of nest failure were not related to predation.

We used the logistic-exposure method (Shaffer 2004) implemented in Program R (R Development Core Team 2008) to estimate daily survival rates (DSR) of duck nests, then extrapolated DSR to the entire 35-day nesting period (Baldassarre 2014) to obtain period nest survival estimates, hereafter "nest success" (Klett et al. 1986; Arnold et al. 2007) for trapped and control sites. We used generalized linear mixed models with a binomially distributed response variable (nest exposure interval survived or failed) to account for both random and fixed effects because our data were comprised of numerous types (e.g., categorical, continuous, integer, factor), and we treated each nest exposure interval as an independent observation.

We first constructed 2 candidate models that would serve as our starting point for assessing the effectiveness of predator reduction; a treatment-year model where all nests on sites under the same treatment application were pooled each year, and a site-year model which assessed nest success on each site each year and accounted for unmeasured variation across study sites. We identified a list of covariates to include in models of DSR, which were identified based on existing literature related to predator reduction or we hypothesized would be related to our treatment and design constraints of the study. We included year (2015, 2016, or 2017), treatment (control vs. trapped), and species (mallard [*Anas platyrynchos*], American redhead [*Aythya americana*], ring-necked duck [*Aythya collaris*], ruddy duck [*Oxyura jamaicensis*], canvasback, and lesser scaup), and presence or absence of a camera. We included initiation date, nest age, and Julian date as continuous covariates; however, only 2 of these temporal covariates were included in any one model. Nest initiation date was estimated by observers by backdating the oldest egg in a clutch via candling and adding the length of the laying period, assuming 1 day for each egg when applicable. In nests where parasitism was evident (>50% of nests), determining the full clutch size and duration of the laying period was impossible due to an unknown amount of eggs being displaced during parasitism events. We chose to use a laying period of 9 days representing a full clutch of 9 eggs which is consistent with the average clutch size for many overwater nesting ducks (Baldassarre 2014). Nest age was interval-based and calculated as the age of the nest since nest initiation (Shaffer 2004). We also included quadratic terms for nest age, initiation date, and Julian date to account for the possibility of non-linear temporal effects. We included interactions between pairs of covariates for which there were a priori reasons to suspect such interactions would be related to DSR.

We first chose a saturated model including all covariates (including quadratic terms) and interactions. We then reduced the number of covariates using backwards step-wise elimination (Agresti 1996) until we identified the model with the lowest Akaike's Information Criterion adjusted for sample size (AIC_c, Burnham and Anderson 2002). To generate DSR estimates, we used coefficient estimates from the best-supported model and incorporated them into a logistic regression formula;

$$Survival = \frac{exp(\beta_0 + \ \beta_1\chi_1 + \cdots \beta_k\chi_k)}{1 + exp(\beta_0 + \ \beta_1\chi_1 + \cdots \beta_k\chi_k)}$$

where *Survival* = estimated DSR, $B_0 \ldots B_k$ is a series of logistic regression coefficients, and $x_1 \ldots x_k$ represent observed covariate values from the data (Shaffer et al. 2007).

Bootstrap Analysis

We used data-bootstrap analysis to generate estimates of nest success and evaluate overdispersion, allowing DSR to vary by nest age. Each bootstrap analysis sampled the original dataset with replacement, and we chose to resample unique nest records rather than individual nest monitoring intervals as successful nests are likely to contain more monitoring intervals than failed nests. Therefore, the resampled bootstrapped data contained the same unique number of nests as the original data; however, the total length of observations differed slightly because of differences in the number of nest monitoring intervals per nest record. We then used our best-supported model to predict estimates of DSR, holding the camera covariate at 0 and initiation date at the mean for all years, allowing nest age to vary from 1 to 35 days, and adjusting treatment level and year to derive desired DSR. We conducted 6 different bootstrap analysis for each combination of treatment and year resulting in 5,000 individual estimates of DSR for each day of the nesting period (1--35). The product of DSRs from 1--35 was then calculated along with associated standard errors, and the geometric mean from all simulations returned 5,000 estimates of nest success for each year and treatment application. For estimates of model overdispersion, we compared standard deviations from bootstrapped beta estimates with the associated standard error of the same betas in the best-supported model and calculated the ratio of these 2 variances (i.e., $[\mathrm{SD}(ß_1)]^2/[\mathrm{SE}(ß_1)]^2$; Bishop et al. 2008).

Results

Over the course of 3 years, trappers removed 2,104 predators from trapped sites. Although trappers focused removal efforts on raccoons and corvids, skunks accounted for 41% of the total catch. Raccoons and ravens made up 25 and 15%, respectively, of all predators removed, followed by crows and magpies (15%), canids (4%), and mink (1%). Increased effort following 2015 resulted in increases of 26% and 35% more predators being removed in 2016 and 2017, compared to 2015, which was largely the result of increased raccoon and raven catches. American mink, skunk, and canid captures declined from 2016 to 2017 by 50%, 13%, and 11%, respectively (Table 1).

We included 5,614 nest exposure intervals from 1,673 overwater nests in survival analysis, totaling 25,294 exposure days. Nests on sites where predator reduction was being implemented represented 56% of the total nests used in analysis. We located overwater nests of 10 duck species, some not considered to be overwater nesting ducks. Species composition of the nests were canvasback (30.1%), American redhead (25.2%), mallard (17.0%), ruddy duck (14.5%), lesser scaup (6.3%), ring-necked duck (5.8%), and miscellaneous dabblers collectively accounting for 1% (blue-winged teal [*Anas discors*], gadwall [*A. strepera*], northern pintail [*A. acuta*], and northern shoveler [*A. clypeata*]). Species composition was similar among years. Among the nests used for survival analysis, 55.3% were successful and 44.7% failed. Predation accounted for the most nest failures (80.8%), and abandonment was the second leading cause

(16.9%) of nest failure. The remaining 2.3% of failed nests were due to unknown causes, such as the inability to relocate a nest.

The best-supported model of DSR included covariates year, treatment, initiation date, presence of a camera, nest age, and interactions between year and treatment, initiation date and treatment, and nest age and the presence of a camera (Pearson χ^2 = 3,629.53, dispersion = 0.924; Table 2).

We identified 2 competitive models to assess predator reduction at the treatment-year scale ($k = 6$), as well as the site-year scale ($k = 18$). The less parameterized model including an interaction between treatment and year omitted site-specific variations and indicated that predator reduction did not result in higher DSR of overwater nests on sites where trapping was being implemented. Mean DSRs from bootstrap analysis on control sites were 0.976 (SE±0.003) in 2015, 0.964 (SE±0.004) in 2016, and 0.943 (SE±0.005) in 2017. Mean DSRs from trapped sites were 0.976 (SE±0.003) in 2015, 0.962 (SE±0.003) in 2016, and 0.952 (SE±0.004) in 2017. Corresponding bootstrap analysis estimates of nest success on control and trapped sites were 0.433 (90% CI: 0.355 - 0.517) and 0.431 (90% CI: 0.361 - 0.499) in 2015, 0.282 (90% CI: 0.229 - 0.339) and .259 (90% CI: 0.212 - 0.307) in 2016, 0.129 (90% CI: 0.097 - 0.169) and 0.183 (90% CI: 0.144 - 0.226) in 2017, respectively (Fig. 1). Overdispersion estimates calculated by the ratio of variances between bootstrapped beta coefficient's standard deviations and the best-supported model's standard errors were negligible (Table 3). Year was negatively associated with DSR (mean difference in beta estimates each year = -0.4232) and nest estimates between control and trapped were most different in 2017, with means differing by 0.052.

The model including a site-year covariate allowed us to assess DSR differences on individual sites each year by including an interaction between site and year. The site-year model revealed drastic differences in DSR among sites during our 3-year study, with resulting nest success estimates ranging from 0.107--0.605. The 2 models performed similarly, but to draw inferences regarding the broader effectiveness of predator reduction, we chose to use the treatment-year model.

Among covariates related to characteristics of nests, nest age had a strong positive relationship with DSR every year, initially low at nest initiation and increasing throughout egg-laying until hatch (DSR range 0.898 - 0.991, $p < 0.001$; Fig. 3). Daily survival rate was positively related to initiation date of overwater nests, with nests initiated early in the nesting season having higher mortality than nests initiated later. Nests on trapped sites began with higher DSR in all years; however, any indication of a treatment effect in DSR disappeared by mean initiation date (Julian day 143) in 2015 and 2016, and entirely by the end of the season (Julian day 181) in all years (Fig. 4). Nest initiation dates ranged from 15 April - 30 June (Julian day 105 -- 181), with 80% of all nests being initiated between 2 May - 9 June (Julian day 122 -- 160) and the mean being 23 May (Julian day 143, SD = 15.09). Mean initiation for canvasbacks, the earliest nesting species in our study was 9 May (Julian day 129, SD = 10.6) and peak initiation spanned a range up to 6.2 days among years. Nest initiation of other species of overwater nesting ducks peaked 2-3 weeks after canvasbacks; American redhead (mean = 25 May, Julian day 145, SD = 12.4), ring-necked duck (mean = 29 May, Julian day 149, SD = 9.6), lesser scaup (mean = 31 May, Julian day 151, SD = 12.4), and ruddy duck (mean = 6 June, Julian day 157, SD = 9.1; Fig. 2). Species was not included the best-supported model for DSR despite relative differences in nest-site-specific characteristics (microhabitat) and nesting chronology.

The presence of a camera at a nest was positively related to DSR and represented 17.5% of nest exposure intervals used for survival analysis. Mean nest age for camera intervals was 23 days (SD = 8.2), as compared with non-camera intervals, which was 19 days (SD = 9.3). Cameras at nests captured approximately 1,940,000 photos from 252 individual duck nests, most of which were canvasback (*n* = 199; 79%) and American redhead (*n* = 29; 8%) nests. Resulting photographs provided positive identification for 85% of nest failures, with camera malfunction (i.e., dead batteries, no available card space, washed out photo) accounting for the remaining 15%. We captured 38 nest failures due to predation in photographs and meso-carnivores accounted for 76% of these failures. Of all identified events; 18 were of raccoons, 9 were of avian predators, 7 were of canids, 3 were of mink, and 1 was of a skunk. Further breakdown of instances where avian predators were the primary cause of nest failure included 4 events of red-tailed hawk predation, 2 where the incubating female duck was killed at the nest. Crows and magpies were observed in 4 instances, and we captured only 1 photo of a raven at a nest during all 3 years of our study. There were no differences between trapped and control sites in the number of predation events or species responsible for nest failures as captured by trail cameras.

Discussion

We evaluated the effectiveness of predator reduction as a tool to increase DSR of overwater duck nests in southwestern Manitoba, Canada. During the 3-year study, we did not detect a positive treatment effect as indicated by DSR that would justify the use of predator reduction as a management tool to increase overwater nest success in this landscape. Predation was the primary source of nest mortality on both trapped and control sites, and it appears that in our study, predator abundance, especially that of raccoons, on trapped sites was not reduced enough to influence DSR. In 2017, when overall nest success of overwater nests was lowest, there was a slight indication that predator reduction had a positive effect and increased nest success by 0.05. That effect was minimal and requires more investigation, however, it does present several questions as to why there was no reduction in predation until possibly the final year of the study. Our results suggest that unlike for upland nesting ducks, efforts aimed at decreasing predation pressures via trapping are less beneficial to the overwater nesting guild and habitat factors coinciding with the timing of the nesting season likely play a more pivotal role in the probability that an overwater duck nest survives to hatch as indicated by the positive effect of initiation date.

Trapping and the Predator Community

The effort allocated towards reducing predators on each trapped site in our study approached the upper bounds possible by a single trapper on 65 km^2. Unfortunately, beyond assessing effects on nest success, our ability to make inferences regarding the extent to which predator reduction was effective at reducing predator abundances is limited. Our baseline of knowledge regarding the predator community comes from general observations of predators on sites, images of predators captured via cameras at nest sites, and published literature assessing the interactions among predators, prey, and the landscape. Considering this information, we make several suppositions regarding the role each predator species played at the time of this study.

American mink that inhabit prairie landscapes in southern Canada rely on aquatic systems for foraging and given the absence of fish and crustaceans in many prairie wetlands, mink diets are largely comprised of waterfowl during the spring and summer (Eberhardt and Sargeant 1977; Arnold 1986). Building nests within inundated vegetation increases exposure to mink, exacerbating possible effects of mink to affect nest mortality, female mortality, and duckling mortality of overwater nesting ducks (Sargeant and Arnold 1984; Korschgen et al. 1996; Krapu et al. 2004). Despite 3 instances of mink being captured in photographs depredating nests or female ducks, trappers believed mink densities were low during our study, and little additional evidence suggested substantial mink predation of nests. Thus, mink were not likely a main cause of nest predation in our study. Similarly, canids accounted for a small percentage of the trappers' total catch and comparatively few photographs from trail cameras were of coyotes or foxes depredating overwater nests. Red foxes have been found to prey heavily on nesting waterfowl in upland systems (Sargeant et al. 1987) and coyotes are more likely to enter wetlands than foxes

(Young and Jackson 1951), but neither are known to spend significant time foraging in wetlands and it is unlikely that canids were a primary cause of overwater nest mortality in our study (Azevedo et al. 2006).

Ravens, the largest of the corvids found on our study sites, can rely on avian prey in areas with high densities of breeding waterfowl (Stiehl 1978). Stiehl (1978) reported that bird eggs and feathers were the second and third most frequent remains (combined frequency of 40%) in raven pellets following small mammals in Harney Basin, Oregon. Ravens were mostly absent in southwestern Manitoba prior to 1990 (Sargeant et al. 1993), but Breeding Bird Survey data indicate an annual increase of 15% beginning in the mid-1990s (Sauer et al. 2017). The impact of ravens on overwater nest success in prairie-parkland ecosystems is largely unknown (Madden et al. 2015). However, trapping efforts targeted at ravens in our study during 2017 may have influenced predation dynamics and contributed to slightly higher nest success on trapped sites in 2017 (Newton 1998; Ellis-Fellege et al. 2012).

Raccoons have been presumed to be a significant overwater nest predator in several studies (Stoudt 1982; Sargeant et al. 1993; Maxson and Riggs 1996), and during our study, raccoons were the second-most-frequent species removed by trappers, but accounted for over half the depredated nests as determined from photographs. From 2010-2013, Minnedosa was the focus of a predator reduction study conducted by Delta Waterfowl Foundation assessing nest success of upland nesting waterfowl in response to predator removal (Delta Waterfowl Foundation, unpubl. data). Trapping efforts during that study focused on reducing skunk and raccoon densities, and as in our study, skunk and raccoon were the 2 species most frequently removed and resulting nest success was on trapped sites was 24%, compared to 2% where no trapping was being implemented (Delta Waterfowl Foundation, unpubl. report). This difference between the effect of predator removal in our study compared to increased nest success where predator removal has been implemented to benefit upland nesting waterfowl, coupled with evidence from photographs at nests in our study, provides additional support that raccoon predation has a larger impact on overwater nests than upland nests. In the agricultural-dominated landscape of east-central North Dakota, an area where raccoon densities and diets are like those in Minnedosa (Pitt 2006), Greenwood (1982) reported that 62% of raccoon foraging sites were wetlands, most likely resulting in high predation pressure by established communities of lone males and family groups. Similar movement data recorded from male raccoons in Minnedosa found resident males to be more social, even aggregating in groups potentially in response to food availability (Fritzell 1978; Pitt 2006). Socially tolerant males may contribute to higher raccoon densities, which decreases the likelihood that removing an individual will lead to unoccupied territory. Higher raccoon densities, a less fragmented landscape, and ample food availability for raccoons in wetlands, make reducing raccoon abundances to achieve higher overwater nest success extremely challenging.

Trail Cameras

Photographs taken at overwater nests indicated raccoons were the primary source of nest failure and are consistent with assumptions from previous overwater nesting studies (Stoudt 1982; Sargeant et al. 1993; Maxson and Riggs 1996). Daily survival rate was higher at nests where we placed cameras, consistent with literature summarized by Richardson et al. (2009), where in all but 1 instance in 21 comparisons showed the presence of a camera reduced nest mortality or had no statistically significant effect on DSR of nests. There are 2 possible explanations for the positive effect of the presence of a camera on DSR in our study; the presence of a camera deterred predators, or the sample of nest exposure intervals we used in our analysis when a camera was present was biased.

Although observers attempt to make cameras placed near nests as inconspicuous as possible, cameras are difficult to situate in an overwater setting where they are not visible, especially from above, due to sparse vegetation at nest sites. Corvids use visual clues to locate food and are likely to detect cameras not well concealed by vegetation. Additionally, corvids also exhibit an aversion or relative neophobia when encountering unfamiliar objects (Heinrich et al. 1995; Thompson et al. 1999). Similar neophobic behaviors have also been exhibited by canids, especially coyotes, when camera equipment or human activity are encountered (Hernandez 1997; Herranz et al. 2002; Sequin et al. 2003). Even though raccoons have not been documented exhibiting neophobic tendencies, cameras situated near nests may have deterred raven and canid predators contributing to higher daily survival rates.

Alternatively, we may have selectively placed cameras at nests farther along in incubation, resulting in a non-random sample of nests which may have introduced a positive bias for camera presence associated with nests more likely to be successful (Renfrew and Ribic 2003). Camera deployment typically occurred after a nest was >7 days into incubation (day 16), determined by candling, as disturbance leading to nest abandonment is less likely with increased parental investment (personal obs., Richardson et al. 2009; Caldwell et al. 2013). Placing cameras at nests >7 days into incubation precluded us from observing nest failure during the period when nest mortality is highest, and, the probability a clutch survives to hatch can be 10-13% greater on day 16 than on day 1 (Fig. 3). Additionally, nest failure is most likely during exposure intervals early during the period when a nest is active and these intervals from nests where we deployed cameras were underrepresented in our sample resulting in a further positive bias.

Temporal effects on DSR

Temporal effects of nest age and nest initiation heavily impacted overwater nest success and supports trends found in other avian nesting studies where survival increases with nest age and throughout the season (Klett and Johnson 1982; Garrettson and Rohwer 2001; Pieron and Rohwer 2010; Fig. 4). Nest-site selection and microhabitat may pre-dispose an overwater nest to risks of predation as found in upland nesting waterfowl (Klett and Johnson 1982; Borgo and Conover 2016). However, female behavior may attract predators to nest-site locations especially in the overwater nesting guild (Grant et al. 2005, Grand et al. 2006). The construction of an overwater nest is more involved than the simple dirt-scrape common to upland nests early in the laying stage, with the construction of a floating platform starting before the first egg is laid and continuing until the clutch is completed. Residual vegetation found in the immediate vicinity of the nest-site (1-2 m) is utilized for nest construction, potentially reducing concealment. Changes in activity levels and response to disturbance by the incubating female near hatch is best explained as “risk-taking” by Forbes et al. (1994), where females late in incubation remain on nests longer in the presence of disturbance, thereby reducing potential detection of the nest. Therefore, explanations for age-related survival in overwater nesting ducks are confounded with nest site selection and incubating female behavior.

Daily survival rates for nests initiated early in the nesting season were consistently lower on control sites than trapped sites, possibly as a result from early trapping efforts targeting locally over-wintering predators. During the last week of May (Julian date = 150), when peak initiation for all species except canvasback occurs, nest success on control sites surpasses success on trapped sites negating the impact of trapping from earlier in the season (Fig. 4). At the beginning of the nesting season, few other food resources are available and concealment cover is lacking when the first females begin initiating nests, resulting in those nests becoming a primary food source for predators. Nest success mid-season was relatively consistent across our study sites and can be attributed to emerging food sources in wetland systems contributing to predator swamping (Flint and Grand 1996).The continued increase in nest success until the end of the nesting season aligns with seasonal movement patterns for meso-carnivores, where home ranges shrink due to focalized movements in response to abundant resources (Greenwood 1982; Crabtree and Wolfe 1988; Ackerman 2002).

Habitat Conditions and Predator Reduction

Our estimates of nest success declined each year across our study sites suggesting there were differences in habitat conditions each year (Fig. 5). Waterfowl Habitat and Breeding Pair Survey data show an increase each year in the number of May ponds from the region where our study sites were from 2015 to 2017 (+14% in 2016, +24% in 2017). Observations of pond inundation levels recorded by our crew on our study sites made in early May reflect a different trend, with 36% of wetlands being recorded drier in 2016 than 2015, and 21% being drier in 2017 than 2015 (n = 1,398). Studies in southwestern Manitoba throughout the 1960s, 1970s, and 1980s, show drastic differences in nest success estimates for canvasbacks, corresponding closely with water levels (Olsen 1964; Stoudt 1982; Anderson 1986). Slight changes in water levels can have great implications for overwater nesting ducks as nesting cover becomes limited, and small nesting ponds preferred by most species are the first to be compromised when water levels are low at the onset of the breeding season (Watmough and Schmoll 2007). It is possible that the effectiveness of predator reduction at increasing overwater nest success is dependent on habitat conditions and resulting locations of overwater nests. In years when abundant nesting cover exists, overwater nests are found at low-densities and resulting nest success is high for ducks, due to the reduced foraging capabilities of predators (Lariviere and Messier 2001; Gunnarsson and Elmberg 2008). Conversely, drought-like conditions lead to low nest success for overwater nests (Olsen 1964; Rogers 1964; Serie et al. 1992). In 2017, after 2 consecutive years of declining local water conditions and declining nest success, there was slight indication that trapping potentially had a positive influence on sites where trapping was implemented. When poor water conditions reduce nest site availability, overwater nests are likely to be initiated in higher densities resulting in a clustered assemblage on the landscape. Ringelman et al. (2012, 2014) found that nest success for upland nests was higher when nests were more densely clustered, as nests shared fates directly related to their nearest neighbor. Although overwater nesting ducks show little evidence of density dependence relating to nesting and duckling survival, the application of trapping may impact a higher proportion of nests when nests are more densely clustered (Gunnarsson et al. 2013). We propose that poor habitat conditions could alter overwater nesting densities enough to allow predator reduction to sustain nest success above the 15% threshold.

Management implications

Based on our results, predator management is not a tool that should be used by managers to increase overwater nest success in southwestern Manitoba, however, some indication exists that it may have an effect during drier years. Managers aiming to increase success of overwater duck nests should focus efforts on nests early in incubation or nests early in the year when DSR are lowest. Even though cameras provide useful information regarding causes of nest failures, some species may be more aware of the presence of a camera than others and conclusions made from captured photographs should mention the possibility of unrepresented species due to neophobic tendencies. Investigations into density dependent drivers of overwater nesting ducks should be further investigated during the nesting season and the development of a conservation strategy specifically for the overwater nesting guild should be a priority of waterfowl managers given areas of high breeding densities are fairly limited in North America.

Tables

Table 1.1. Total predators removed from 65 sq.km (8.1 km x 8.1 km) study sites between 15 March and 15 July as part of a predator reduction effort to increase nest success of overwater nesting ducks in southwestern Manitoba, Canada, 2015-2017. A variety of trap types were used for removal including live cage traps, conibear box sets, and dog-proof land sets for mammals, as well as conical wire-cage traps, adapted ladder-style traps, and decoying for avian predators.

Site	Year	Raccoon	Mink	Skunk	Red Fox/Coyote	Raven	Crow/Magpie	Total
2[a]	2015	30	2	81	1	8	32	154
7[a]	2016	89	7	118	13	23	32	282
7[a]	2017	73	2	81	19	66	53	294
3	2015	41	0	85	5	3	15	149
3	2016	44	1	63	18	80	17	223
3	2017	69	1	49	5	38	64	226
6	2015	36	5	124	7	16	35	223
6	2016	68	3	124	8	22	27	252
6	2017	76	3	133	8	51	30	301
Total		526	24	858	84	307	305	2,104

[a] Sites 2 and 7 were not subjected to trapping during all 3 years of the study. Due to access limitations, site 2 was removed and was replaced with site 7, 35 km northeast in 2016. Despite the relocation of the site, the professional trapper on these 2 sites was the same in 2016 and 2017.

Table 1.2. Coefficient estimates from the best-supported model used to estimate daily survival rates of overwater duck nests in response to intensive predator reduction efforts in southwestern Manitoba, Canada, 2015-2017. We used generalized linear mixed models with a binomially distributed response variable and then applied logistic-exposure methods (Shaffer 2004) treating each nest exposure interval as an independent observation to derive estimates of daily survival rates.

Survival = *Year*treatment + initiation*treat+ camera*nest age*

		95% Conf. Int.				
Covariate	ß	Lower	Upper	Std. Error	z value	Pr(>|z|)
Intercept	1.259	0.584	2.464	0.611	2.060	0.039
Treatment[a]	1.210	-0.321	2.741	0.780	1.550	0.121
Year						
2015	-	-		-	-	-
2016	-0.423	-0.745	-0.111	0.161	-2.617	0.008
2017	-0.913	-1.222	-0.616	0.154	-5.924	3.13*e*-09
Year x Treatment						
2016 x Treatment	-0.061	-0.484	0.362	0.215	-0.284	0.776
2017 x Treatment	0.189	-0.221	0.602	0.209	0.904	0.365
Initiation Date[b]	0.012	0.005	0.020	0.003	3.171	0.001
Camera[c]	1.573	-0.911	2.286	0.349	4.506	6.6*e*-06
Nest Age[d]	0.041	0.032	0.051	0.004	8.641	< 2.0*e*-16
Initiation Date x Treatment	-0.008	-0.019	0.0017	0.005	-1.631	0.102
Camera x Nest age	-0.042	-0.074	-0.0118	0.015	-2.717	0.0065

[a] Treatment applied as either 0 or 1.

[b] Initiation date represented by Julian values, ranging from 107-178 with mean = 142.9.

[c] Presence of a camera, either 0 or 1.

[d] Nest age estimated as the number of days since initiation, ranging from 1-46 with mean = 19.6.

Table 1.3. Overdispersion estimates ($\widehat{\boldsymbol{C}}$) calculated as a ratio of the variances between the best-supported daily survival rate model (Orig. Model Betas and SE) and the bootstrapped betas calculated from 5,000 simulations (Mean Bootstrap Betas and SD). Bootstrap analysis was conducted by sampling nests randomly from the original dataset with replacement and treating each nest exposure interval as an independent observation.

Coefficient	**Orig. Model Betas**	**SE**	**Mean Bootstrap Betas**	**SD**	$\widehat{\boldsymbol{C}}$
Intercept	1.2593	0.6113	1.2745	0.60617	0.98303
Year2016	-0.4232	0.1617	-0.4256	0.16597	1.05353
Year2017	-0.9131	0.1541	-0.9182	0.15637	1.02937
Treatment	1.21	0.7804	1.2211	0.75814	0.94355
Initiation	0.01255	0.0039	0.01252	0.003908	0.97486
Camera	1.573774	0.3492	1.58593	0.3648	1.09124
Nest Age	0.04163	0.0048	0.04144	0.00489	1.03139
Year2016*Treat	-0.061384	0.2157	-0.06156	0.2222	1.0611
Year2017*Treat	0.18934	0.2093	0.19052	0.213	1.03506
Initiation*Treat	-0.008501	0.0052	-0.0085899	0.00506	0.94479
Camera*Nest Age	0.0427	0.0157	-0.042792	0.0161	1.05536

Figures

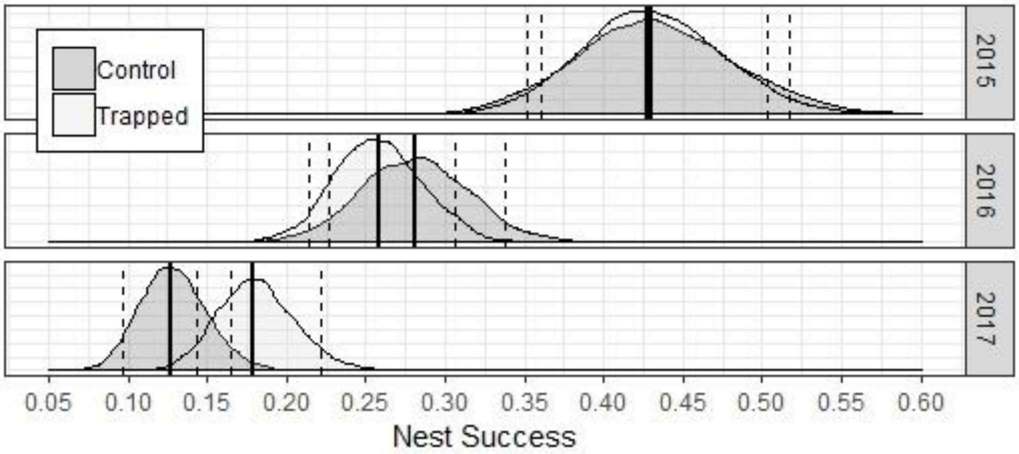

Figure 1.1. Model-based nest success estimates for overwater duck nests from bootstrap-analysis including 5,000 simulations at each year and treatment level, in southwestern Manitoba, Canada, 2015-2017. Geometric mean is indicated by solid vertical lines, 90% confidence intervals are indicated by dashed vertical lines. Covariate estimates for camera presence were set to 0, nest age and initiation date was set equal to their means for all 3 years (19.6 and 143, respectively).

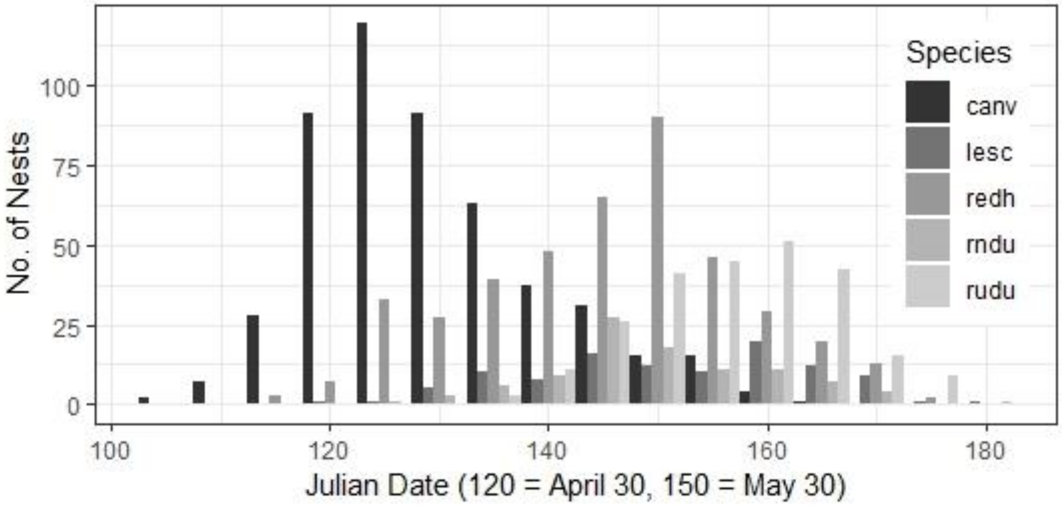

Figure 1.2. Bar plot of initiation dates for overwater duck nests by species, as estimated by adding the clutch size to the backdated age of the oldest egg determined by candling (Weller 1956). Nest included were found during years 2015-2017, in southwestern Manitoba, Canada. Peak initiation date averaged over all years for canvasback (*Aythya valisineria*) was 4 May (Julian date = 125), and average initiation date for all nests was 23 May (Julian date = 143).

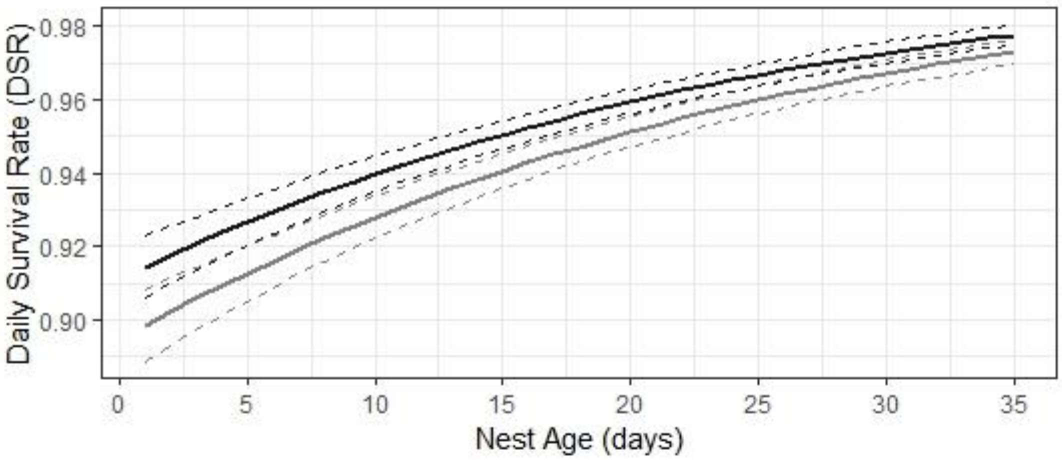

Figure 1.3. Daily survival rates (DSR) corresponding to the age of an overwater nest since initiation on control and trapped sites in 2017 from southwestern Manitoba, Canada, 2015-2017. The grey lines represent DSR for a nest on control sites, and the black lines represent DSR for a nest on trapped sites. Cumulative estimates of nest success between pooled control and trapped sites were 12.7% and 17.9%, respectively, in 2017, and was the only year where a treatment effect was apparent. Nest age (days) can be referenced as initiation date = 1, incubation = 9, hatch = 35. Dashed lines represent 95% CI for respective daily survival rates.

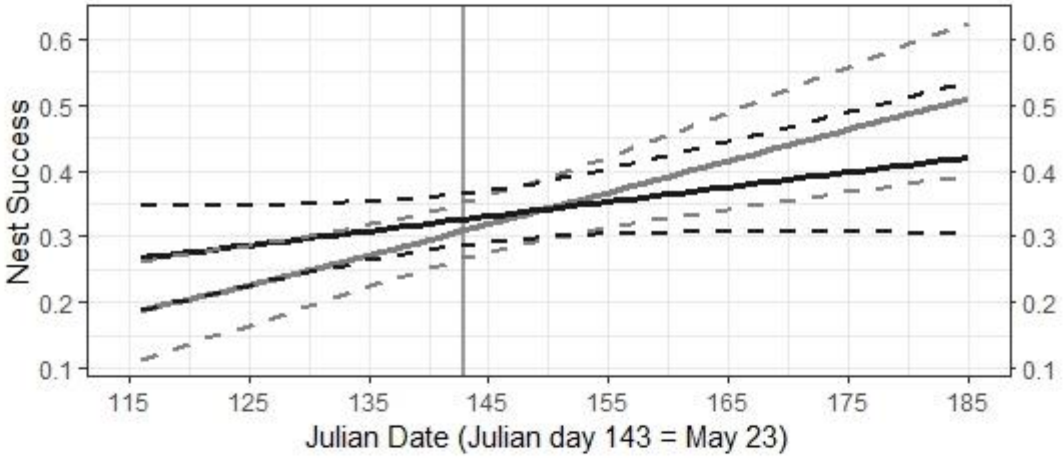

Figure 1.4. Model-based nest success estimates for overwater duck nests predicted by Julian initiation date from control and trapped sites in southwestern Manitoba, Canada, from 2015-2017. The grey lines represent survival of a nest on control sites, and the black lines represent survival of a nest on trapped sites. We pooled sites under the same treatment application, weighted years equally, set the camera covariate to 0, and set nest age equal to its mean (19.6 days). Solid lines are the predicted survival rates, dashed lines represent 95% confidence intervals calculated using the Delta Method, and the solid vertical line indicates mean initiation date for all nests (Powell 2007).

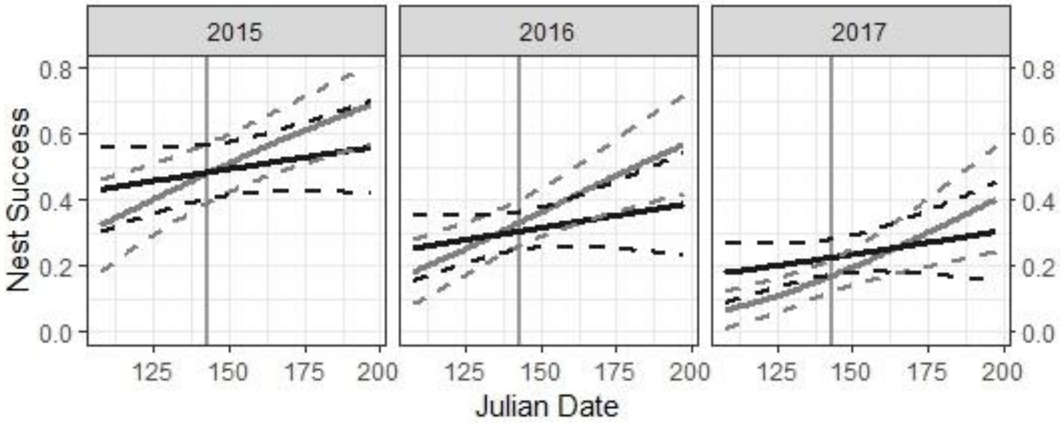

Figure 1.5. Predicted overwater nest success estimates for nests initiated throughout the peak breeding season on control and trapped sites in southwestern Manitoba, Canada, from 2015-2017. Grey lines represent predicted survival of nests on control sites and black lines represent predicted survival of nests on trapped sites. We pooled nests from study sites under the same treatment application, set the camera covariate to 0, and set nest age equal to its mean. The solid lines represent nest success estimates, dashed lines represent 95% confidence intervals, and the vertical line represents mean initiation date for all nests during the study (Julian date = 143).

Chapter 2:

Estimating Diving Duck Productivity using Sightability-Adjusted Brood-Pair Ratios

***Abstract*-** Surveys for waterfowl breeding pairs and broods have been widely used to index productivity in lieu of traditional methods that require estimates of both nest and brood survival. Although both approaches have inherent biases, developments in survey methodology and design have addressed a few of the key components to mitigate bias and make survey estimates more precise. Estimating detection probabilities through replicate surveys provides an adjustment for individuals that go unseen or are not detected (i.e., sightability-adjusted counts). Although adult pairs of breeding waterfowl tend to have high detection probabilities, ducklings are more difficult to observe and counts of broods are typically biased due to low detection probabilities. By directly estimating the detection probability of broods, an adjustment can be applied to estimate total brood abundance when not all broods on the landscape are observed. During 2015-2017, we conducted pair and brood surveys for diving ducks to compare with estimates of overwater nest success on 7 sites in southwestern Manitoba where intensive nest monitoring efforts were ongoing. We estimated detection probabilities for broods by conducting replicate ground-based surveys and used mark-recapture methods to analyze 1,915 unique 2-occasion encounter-histories of diving duck broods. Cumulative detection probabilities (i.e., $P*$, the probability of seeing a brood during at least 1 survey) for diving duck broods were ≥ 0.70 for all species during 5 of the 6 survey rounds. Among site-years, SA-BPRs were 0.053 - 1.275 for canvasbacks (*Aythya valisineria*) and 0.118 - 0.726 for all species of combined diving ducks. We estimated nest success for overwater nests (n = 1,673 nests), which ranged from 0.07-0.54. When comparing estimates of SA-BPRs to nest success estimates using a linear model, adjusted R^2 values indicated 19% and 50% of the variation in SA-BPR was explained by nest success for canvasback and all species of diving ducks combined, respectively. Although SA-BPR estimates for canvasbacks

appear less correlated with nest success, combining multiple species resulted in strong correlations for diving ducks and could be used to index productivity of the local breeding population at a 65 km^2 scale.

Introduction

Regional estimates of both guild- and species-specific breeding waterfowl populations are important for informing harvest regulations and for designing and implementing management or conservation practices (NAWMP 1986). The realization that large-scale standardized population counts for waterfowl were needed was recognized as early as 1935, and since 1955 waterfowl counts have been conducted by various agencies to estimate breeding population sizes (Hawkins 1984; USFWS 2017). To better understand the biological and ecological mechanisms influencing waterfowl populations, efforts at the local scale have focused on quantifying important vital rates that drive productivity (Cowardin and Johnson 1979). Waterfowl productivity can be influenced by a variety of factors and low nest success can have a substantial influence on productivity (Bouffard 1988; Hoekman et al. 2002). However, hatchability (the proportion of eggs that hatch) and duckling and female mortality have also been identified as key metrics limiting local population growth (Talent 1983; Cowardin et al. 1985; Klett et al. 1986; Maxson and Riggs 1996). Estimating vital rates can be expensive, often requiring intensive ground efforts to obtain adequate sample sizes and those estimates are limited in overall applicability, both spatially and temporally.

Among waterfowl vital rates, hen success is the probability an individual female hatches at least 1 egg during any nesting attempt (≥1) in a single breeding season and is a product of nest success and renesting propensity, both of which are dependent on species, locale, and seasonal conditions (Cowardin and Johnson 1979; Doty et al. 1984). Hen success has been most widely used to index productivity (i.e., recruitment rates) for mallards (*Anas platyrynchos*) and requires locating and monitoring nests multiple times. Due to low densities and imperfect detection rates of nests, deriving estimates of hen success for diving ducks requires intensive effort.

Alternatively, brood-pair ratios can be used to estimate productivity using surveys conducted before and after the peak nesting season to determine breeding pair and brood abundances and ultimately, what percentage of breeding pairs produce a brood (Hammond and Johnson 1984). Such surveys conducted from the air or ground, however, are subject to biases resulting from imperfect detection probabilities ($p < 1$; Nichols et al. 2000; Anderson 2001, 2003; Rosenstock et al. 2002). Pagano (2007) showed that ground counts failing to account for detection probabilities < 1 can result in dramatic underestimates of population size, especially for broods. Corrected productivity estimates that incorporated detection probabilities (sightability-adjusted brood-pair ratios; SA-BPR) were moderately correlated with estimates of hen success for upland nesting dabbling ducks (Pagano 2007).

Pagano (2007) also observed a small sample of diving ducks and reported that detection probabilities for diving ducks were higher than dabblers likely due to differences in their responses to disturbance; diving duck broods often creche in open water and dive when disturbed, whereas dabblers seek the cover of emergent vegetation. If that is generally the case, higher detection probabilities of pairs and broods of diving ducks could increase precision in estimates of productivity using pair and brood counts. Furthermore, estimating productivity based on pair and brood counts requires considerably less effort than estimating productivity based on hen success and could be used to inform management (Sugden and Butler 1980; Cowardin and Blohm 1992).

Our primary objective was to evaluate using brood and pair counts of diving ducks to estimate productivity and compare estimates of productivity derived from counts with those derived from hen success. We focused on accurately estimating brood abundances by deriving brood detection probabilities including multiple covariates thought to influence brood detection and applied resulting detection probabilities derived from the best-supported models each year to site-specific brood observations. Sightability-adjusted brood abundances were combined with indicated breeding pair observations from that year on the same survey sites to derive sightability-adjusted brood-pair ratios. Based on the small sample of diving ducks observed by Pagano (2007), we expected strong correlations between estimates of productivity derived using SA-BPR and hen success which would advocate for a less intensive alternative to estimate diving duck productivity.

Study Area

We collected data from 7 sites in the Prairie Parkland Region of southwestern Manitoba; 2 near Minnedosa (50°12’N, 99°47’ W), and 5 near the town of Shoal Lake (50°26’N, 100°34’ W). Each site encompassed 8.1 x 8.1 km and was divided into 25 legal quarter sections (1.6 x 1.6 km). Each section thereafter, was subdivided into 4 potential survey-plots (64.57 ha) and we randomly chose 1 plot from each section where access was granted by the landowner to serve as a focal plot (n = 25/site, n = 150/year; Fig 1.). On selected plots, ground crews conducted pair counts, nest searches, and brood counts for 5 species of diving ducks (canvasback [*Aythya valisineria*], lesser scaup [*Aythya affinis*], American redhead [*Aythya americana*], ring-necked duck [*Aythya collaris*], and ruddy duck [*Oxyura jamaicensis*]).

Methods

Breeding Pairs

Timing of counts was based on breeding chronology of canvasbacks and therefore, variations in migration chronology influenced start date for pair counts each year. Upon the arrival of canvasbacks to the study sites, we conducted roadside transects every 1-2 days until ratios between lone males and lone pairs neared 1:1 for canvasbacks, suggesting half the breeding females had started nesting (Dzubin 1969). Once this ratio was met, we presumed that all breeding canvasbacks had reached the study sites and migrants had departed (Dzubin 1969; Anderson 1985; Serie and Cowardin 1990). Pair counts took 6-10 days to complete across all 150 plots we surveyed (9 May 2015 - 15 May 2015, 30 April 2016 - 7 May 2016, 4 May 2017 - 14 May 2017), and were conducted by walking to every wetland within the boundaries of the plot. Upon arriving at a wetland, observers determined the proportion of the wetland inundated and recorded the corresponding level; 0 = dry, 1 = 1-25%, 2 = 26-50%, 3 = 51-75%, 4 = 76-100%, 5 = >100%. Observers used binoculars to detect and record the number of individuals and associated social classifications of all 5 diving duck species, as outlined by Dzubin (1969). Observers conducted counts independently and each observer had ≥1 week of waterfowl identification training. We considered observations of lone males, lone pairs, grouped pairs, groups of 2 males and 1 female, and grouped males comprised of <5 individuals, to be observations of indicated breeding pairs (Dzubin 1969). For small wetlands (<0.5 ha) straddling plot boundaries, we assigned the wetland to the plot in which the centroid of the wetland occurred. For wetlands >0.5 ha, ducks found on the wetland within the boundaries of the plot were included in counts for that plot. Ducks that were seen flying over the ponds, or ducks that swam in or landed within the plot boundaries during surveys were excluded (Hammond 1969). Observer intentions were not to flush ducks, so, notes of accidental flushes and potential landing spots were revisited after the survey was completed to reduce the potential for double-counting. Observations at each wetland typically lasted >1 minute, but observers could spend any length of time necessary to count and classify observations of diving duck pairs (Pagano and Arnold 2009*a*). Surveys began at sunrise and concluded by 1400 (CST) when weather conditions were favorable (wind speeds <40 km/h, no more than moderate rainfall). In 2016 and 2017, we randomly resampled 10 plots on each site 2-6 days following the initial survey, to estimate standard errors associated with breeding pair abundances.

Brood Counts

We conducted 2 rounds of brood surveys each year; timing of the first round was approximately 3 weeks after estimated peak hatch of canvasbacks based on nest monitoring data (6 July - 10 July 2015, 4 July - 9 July 2016, 7 July - 12 July 2017). The second round followed 18-20 days after conclusion of the first round, to observe broods from successful re-nests and from later-nesting species (30 July - 4 August 2015, 27 July - 2 August 2016, 30 July - 4 August 2017). Brood counts began at sunrise and extended throughout the day when weather was favorable (wind speeds <40 km/h, no more than moderate rainfall). Ground crews completed 2 replicate surveys on each plot within the same day, allowing at least 2 hours to elapse between the end of the first survey and the start of the replicate. To reduce false-positive errors related to brood mis-identification, each replicate survey was completed by the same observer. During each survey, observers recorded start and end times, wind speed using Beaufort Scale (Simpson 1926), relative cloud cover (0-3; clear, 1-33%, 34-66%, 67-100%), and precipitation (0-3; none, fog/drizzle, light or moderate rain, heavy rain/snow). At each wetland, observers estimated wetland inundation following the same protocol as in pair counts. Observers also visually estimated the percentage of the basin inundated and not obscured by vegetation capable of concealing broods. We identified re-sighted broods during replicate surveys based on a combination of criteria including location, species, age class (≤1 sub-class difference, Gollop and Marshall 1954), and brood size (≤3 duckling difference). We surveyed all inundated wetlands within plots when the water surface was not completely obscured. We coordinated survey rounds so that no early hatching broods would fledge before the conclusion of the first round, no broods could hatch and fledge in between rounds, and most late hatching broods hatched before the second round. Observations at each wetland typically lasted >1 minute, but observers could spend any length of time necessary to accurately identify broods (Pagano and Arnold 2009*b*).

Nest Searching and Monitoring

Nest searching for overwater nests began immediately following the conclusion of pair surveys and extended until approximately 3 weeks after peak hatch of canvasbacks (9 May - 30 June). Nest searching centered around searching selected plots, ensuring 1-2 complete searches each year. Observers were encouraged to also search any areas of the site pending landowner permission after all plots had been searched. Observers typically worked in teams of 2, navigating inundated vegetation buffers of wetlands where overwater nests are likely to be found. Observers recorded all overwater duck nests found in ≥10 cm of water and we monitored nests on a 7--10 day cycle until the eggs hatched or the nesting attempt failed. When a nest was found, observers recorded the location using a GPS to aid in relocating the nest. Data collected at the nest for success analysis included species, clutch size, nest age (via candling; Weller 1956), water depth (cm), distance to dry edge (m), and distance to open water (m). During the final visit, we determined the fate of the nest as successful (≥ 1 egg hatched), abandoned (no advancement in incubation stage as determined by candling since the previous visit), or depredated (presence of eggshell fragments or yolk, destroyed nest bowl, presence of dead female). See Johnson, Ch. 1 for more details.

Data Analysis

Sightability- Adjusted Brood-Pair Ratios

We used bootstrapped sampling to estimate the average number of pairs per plot for each species, on each study site. To accomplish this, we randomly selected with replacement from the 25 plots on each site, in each year, and calculated a mean from the bootstrapped samples along with their associated standard deviations for each species. Pairs from replicate surveys during 2016 and 2017 could be included in the bootstrap sampling, however, if a plot where replicate surveys occurred was selected, we randomly chose either the original survey or the replicate survey to be used so that replicate plots and non-replicate plots had an equal likelihood of being selected. To derive site-level estimates of pair abundance, we multiplied the average number of pairs per plot by the number of plots on each site (n = 25). We derived confidence intervals using the Delta method (Powell 2007).

For broods, we created a 2-occasion encounter-history for each observed brood and estimated detection probabilities using maximum-likelihood estimators calculated using Program MARK (White and Burnham 1999). The constructed encounter-histories included occasions of 0 or 1 indicating a brood's absence (0) or presence (1) during the survey, and resulted in 3 possible encounter-histories when a brood was detected during ≥1 survey: 10, 01, and 11. We used Huggin's closed-capture models (Huggins 1989, 1991) in Program MARK, because the inherent heterogeneity associated with broods and timing of surveys required capture probabilities to be modelled to include observable covariates. Huggin's model is also a linear-logistic model conditional on ≥1observation during the survey round. Using the derived detection probabilities, we estimated cumulative detection probability (P*) as:

$$P^{*} = 1 - (1 - p)^{k};$$

where the probability of detecting a brood during ≥1 survey is equal to *1* minus the probability of failing to see a brood during each of the k = 2 survey occasions.

During brood surveys, observers also collected data on additional covariates that we hypothesized would influence detection probabilities (Ringelman and Flake 1980; Giudice 2001; Pagano 2007). In preparation for analysis in Program MARK, we organized detection data for each individual year to include 7 attribute groups (i.e., the 5 target species and 2 rounds per year), and 10 covariates including duckling age class (Gollop and Marshall 1954), maximum brood size (the maximum number of ducklings observed when an individual brood was sighted), time start for each survey, categorical averages for precipitation, cloud cover, and wind speed between each replicate, wetland inundation level, estimated wetland visibility (estimated percent of the inundated wetland not obscured by vegetation; 5% intervals), and break duration (elapsed time between the end of the initial survey and the start of the replicate survey). In Program MARK, we assumed that each encounter was an independent encounter and allowed capture probabilities to vary for each survey. Furthermore, we constrained the parameters recapture probability (c) and capture probability (p), to equal one another ($c = p$), so that all encounters were considered independent. Thus, the probability of detecting a brood during the second survey wasn't dependent on previous detections. We used the logit link function in Program MARK for analyses (White and Burnham 1999), which assumes the log odds ratios varies as a linear function of covariates and constrains detection probabilities within the parameter space of 0 – 1. Using the sample-size adjusted extension of Akaike's Information Criterion (AIC_c; Burnham and Anderson 2002), we determined the best-supported model indicated by the lowest AIC score. For each year, we used a forward-stepwise covariate approach adding and substituting 1 covariate at a time until AIC_c values no longer decreased indicating any additional covariates reduced support for the model (Table 1).

Using the best-supported model for each year, we inserted estimated detection probabilities of broods for each species and for all target species derived from Program MARK into the cumulative detection probability equation, returning the sightability adjustment necessary to determine total brood abundance. We calculated a sightability-adjusted brood abundance estimate for each species at each site-year ($\widehat{N}$), by dividing the total number of individual broods observed (n) by the species-corrected cumulative detection probability:

$$(\widehat{N} = \frac{n}{P*}).$$

Confidence intervals were calculated using the Delta method (Powell 2007).

Using indicated breeding pair abundances and sightability-adjusted brood abundances, we calculated SA-BPR for each site-year as the ratio of brood abundance divided by the estimated abundance of pairs on the same site, for each species and for all target species combined. To mitigate the chance of a brood being counted during both rounds, we excluded broods older than (how many weeks?) (sub-class 2A; Gollop and Marshall 1954) from the second round of surveys as these broods were old enough to have been counted during the first round of surveys (Gammonley and Fredrickson 1998). Lastly, I amalgamated all 3 years of brood data and included the top 10 best-supported models from yearly analysis adding the covariate year to identify the best-supported model for brood detectability overall.

Definition of Assumptions

Inherent to these surveys are a variety of assumptions, which in turn affect the accuracy of estimates and should be addressed. (1) *No false positives.* Observers were trained prior to the beginning of surveys to ensure non-ducks or dabbling ducks are not mistaken for diving ducks. (2) *Samples are independent*. We relied on our criteria to individually “mark” broods allowing for recognition of re-sighted broods during independent surveys. (3) *Closure assumption.* The closure assumption for double observer counts is met by conducting surveys within a short period to reduce opportunities for broods to immigrate or emigrate from surveyed wetlands between survey replicates. (4) *All breeding pairs are counted, and all broods come from previously recorded pairs.* Diving duck pairs have high detectability and surveys are conducted on the same wetlands for both pairs and broods.

Traditional Productivity

We generated single-specie SA-BPR estimates for canvasbacks, and for combined diving ducks including and excluding canvasback survey data to compare with hen success, as calculated by Cowardin and Johnson (1979; Appendix 1, page 33):

$$H = \alpha_1 S e^{\alpha_2 (1-S)^2}.$$

This equation, originally designed using mallard data, suggests hen success (*H*) is a function of the probability a clutch survives to hatch, (*S*), and the proportion of females who initiate nests as it applies to initial attempts (α_1) and renesting attempts (α_2). No nesting seasons during this study were under drought conditions where initial nesting attempts would have been affected (T. W. Arnold, personal comm.). Therefore, we presumed all females attempted ≥1 nest and set α_1 equal to 1 for all hen success estimates. Numerous factors influence the probability a female will renest, such as age of the nest when terminated, timing of failure within the nesting season, and experience of the female (Doty et al. 1984). Additionally, it is likely that hydrological conditions at the time of nest failure play a more pivotal role in overwater renesting than in upland nesting species. Doty et al.'s (1984) sample taken near Minnedosa reported 14 of 24 (58%) nasal-marked canvasbacks renested over a 2-year period. Ring-necked ducks have been found to renest at various levels, but most reports suggest between 30-50% (Mendell 1958; Coulter and Miller 1968). Ruddy ducks are believed to renest rarely, likely due to late nest initiation coupled with the energy required to produce a second clutch (Baldassarre 2014). Renesting in American redheads is low, likely due to their ability to utilize intra- and interspecific brood parasitism allowing them to increase fecundity without renesting (Weller 1959). Because renesting in overwater nesting ducks is more variable than for upland nesting ducks, we assessed hen success at various levels of renesting and reduced the exponent credited to renesting under 2 different scenarios in addition to Cowardin and Johnson's (1979) formula:

$$H = P \text{ - } \textit{no re-nests} \ (\alpha = 0)$$

$$H = Se^{0.5(1-S)^2}$$ - *50% of failed nests initiate re-nests* ($\alpha = 0.5$)

$$H = Se^{(1-S)^2}$$ - *original formula* ($\alpha = 1$)

To generate nest success estimates (S), we used the logistic-exposure method (Shaffer 2004) implemented in Program R (R Development Core Team 2008) to derive daily survival rates (DSR) of overwater duck nests. We then extrapolated DSRs to the entire 35-day nesting period (Baldassarre 2014) to obtain period nest success estimates for each site-year (Klett et al. 1986, Arnold et al. 2007). We used generalized linear mixed models with a binomially distributed response variable of survival to account for both random and fixed effects given the data were comprised of various types (e.g., categorical, continuous, integer, factor) and each nest was treated as an independent observation.

We chose a list of covariates to include in daily survival models, which have been identified in the literature as possible factors influencing nest success. We included year (2015, 2016, 2017), site-year (n = 18), nest-age (via candling; Weller 1956), initiation date (backdated from candling), and calendar date (Julian) along with their quadratic forms as continuous covariates; however, any 1 model never included >2 of these temporal covariates.

Our model selection approach began with including all covariates, quadratic terms, and their interactions. We used a backwards step-wise approach to identify the best-supported model using Akaike's Information Criterion adjusted for sample size (AIC_c, Agresti 1996, Burnham and Anderson 2002). We generated DSRs using coefficient estimates from the best-supported model incorporated into a logistic regression formula (Shaffer et al. 2007). Finally, we used a data-bootstrap analysis using 1,000 simulations where nest-age could vary, and all other covariates were held at their mean (see Johnson Ch 1 for details). We used period nest survival estimates as the basis for our hen success estimates each site-year.

We used linear and quadratic models to measure correlations between hen success under 3 renesting scenarios and SA-BPR for each target species, combined diving ducks, and combined diving ducks excluding canvasbacks. The amount of variation explained was indicated using adjusted R^2 values for each linear or quadratic comparison where hen success was the independent variable and SA-BPR was the response variable.

Results

Pair Counts

The number of canvasback pairs on each site varied each year and abundances ranged from 23.1 (SD = 1.3) pairs to 67.5 (SD = 2.4) pairs (3-fold), with a mean overall density of 2.5 pairs/km^2. Counts of combined diving duck pairs also varied, and estimated site abundances ranged from 113.6 (SD = 4.9) pairs to 455.9 (SD = 11.3) pairs (4-fold), with a mean overall density of 17.5 pairs/km^2 (Table 2).

Brood Counts

Ground crews surveyed 1,831 unique ponds each year, and brood abundances were highly variable across site-years. Canvasback brood abundances varied by up to 12-fold, ranging from 3.4 (SD = 0.26) to 39.5 (SD = 2.24), and averaging 17.0 across all site-years. Combined diving duck brood abundances varied up to 8-fold, ranging from 23.2 (SD = 0.9) to 199.5 (SD = 5.27), and averaging 92.5 across all site-years (Table 3). Covariates affecting the probability of detecting a brood varied each year, but survey round, wetland visibility, and brood size were consistently included in the best-supported models (Table 4). Detection probabilities for diving duck broods were 0.49 (95% CI: 0.457-0.527) in 2015, 0.57 (95% CI: 0.533-0.622) in 2016, and 0.63 (95% CI: 0.585-0.675) in 2017. Mean detection probabilities by species were 0.60 (95% CI: 0.53-0.67) for canvasbacks, 0.54 (95% CI: 0.46-0.62) for lesser scaup, 0.51 (95% CI: 0.44-0.57) for American redheads, 0.56 (95% CI: 0.47-0.64) for ring-necked ducks, and 0.56 (95% CI: 0.51-0.61) for ruddy ducks. Overall, detection probability increased with wetland visibility, brood size, brood age, and year, and decreased with survey round, hours since dawn/hours until dusk, and wind speed (Table 5).

Using the conditional probability expression, cumulative detection probabilities for diving duck broods increased each year; 0.771 (95% CI: 0.725-0.812) in 2015, 0.812 (95% CI: 0.761-0.855) in 2016, and 0.836 (95% CI: 0.785-0.877) in 2017. The same trend was evident at the species level (Fig. 2).

Traditional Productivity

We used estimates of period nest survival from an ongoing predator reduction project overlapping the same study sites and years (Johnson, ch1.) to calculate hen success. The number of nests located and monitored each year varied, but the species composition of nests was similar. Nest totals varied from 7-54 (mean = 28) across sites for canvasbacks, and 63-136 (mean = 93) for combined diving ducks (Table 3). Period nest survival estimates for all overwater nests among site-years ranged from 0.07 (90% CI: 0.03 - 0.13) to 0.54 (90% CI: 0.42 - 0.68), with an average success rate of 0.27 (Table 6). We estimated hen success under 3 renesting scenarios at each site-year ($n = 18$) to compare with estimates of SA-BPR. SA-BPR explained 19% and 42% of the variation in hen success for canvasbacks and combined diving ducks, respectively, and was best fit by linear models at $\alpha = 0$ (canvasbacks: p-value = 0.038, combined diving ducks: p-value = 0.002; Table 7, Fig. 3). When canvasback SA-BPR data were excluded from the combined diving duck data set, SA-BPR explained 50% of hen success at $\alpha = 0$ and was best fit by a linear model (p-value = 0.0005; Fig. 3). Adjusted R^2 values for the remaining target species varied, but American redhead and lesser scaup showed a minimal amount of variation explained by SA-BPR (Table 7, Fig. 4).

Discussion

Overall, diving duck productivity estimated using sightability-adjusted brood-pair ratios (SA-BPR) and traditional productivity methods showed similar correlations to that of other guild-specific investigations (Pagano et al. 2014). The covariates included in best-supported models of brood detectability varied each year, however, survey round, brood size, and wetland visibility were consistently included in the best-supported models. To our surprise, species was not included in the best-supported model except for 1 year, however, we retained this covariate to derive annual brood abundance estimates for each species. The best-supported model for overall brood detectability using data from all 3 years of surveys included year, brood size, brood age, time of day, wind, and wetland visibility. High detection probabilities of diving duck broods resulted in moderate to strong correlations between SA-BPR and hen success productivity estimates when renesting was lowest. Contrary to our predictions, hen success and SA-BPR estimates for canvasbacks were weakly correlated despite specific attention paid to timing of surveys and local canvasback breeding phenology. Furthermore, combined diving duck SA-BPR estimates that excluded canvasback data showed higher adjusted R^2 values then when canvasback data was included. Although inferences from surveys of canvasbacks were inconclusive, guild-specific estimates for diving ducks were comparable with other investigations of dabbling ducks accepted by the scientific community and could be utilized alongside those surveys. We tested a variety of covariates known to influence detection probabilities of dabbling duck broods that have never been investigated for diving ducks and addressed a variety of assumptions specific to surveys where the study species have high detection probabilities, but home ranges and seasonal movements are unknown.

To generate estimates of productivity using both techniques for this study, substantial ground effort was required. SA-BPR surveys took approximately 231 person-days to complete each year, whereas monitoring nests to collect data used to derive nest success estimates required approximately 539 person-days. SA-BPRs incorporate aspects of nest success, nesting effort (nesting propensity and renesting effort), and brood survival to derive productivity estimates in less time and effort than traditional nest monitoring methods (Cowardin and Johnson 1979). Despite the logistical benefits, using SA-BPR's would not be justified unless comparable or stronger inferences could be drawn from resulting estimates. Because we used multiple data sets to derive SA-BPR, there exists more opportunities for bias to be introduced to SA-BPR estimates. Opportunities for survey-based biases exist at nearly all tiers of the data collection process including survey timing, experience of observers, and detection probability modelling (Dzubin 1969; Burnham and Anderson 2004; Pagano and Arnold 2009a, 2009b; Pagano et al. 2014). Additionally, assumptions associated with survey methods are often difficult to meet in biological systems and must be taken into consideration when designing survey protocol (Otis et al. 1978; Williams et al. 2002).

Pair Counts

Logistical constraints inhibited our ability to use double-observer approaches, resulting in estimating pair abundances using single-observer methods (Cowardin and Blohm 1992). Single-observer methods fail to quantify detection and have been shown to bias pair abundance estimates, however, to a lesser extent in diving ducks than in upland nesting species (Diem and Lu 1960; Nichols et al. 2000; Pagano and Arnold 2009*a*). Standard errors surrounding pair abundances from the subsample of repeat pair counts were lowest for canvasbacks, indicating that our timing of pair counts was best matched for canvasbacks and the remaining species we counted may have not yet settled onto breeding ponds and migrants were still present within the count area (Dzubin 1969). Pagano (2007) reported single-observer estimates captured ~90% of American redhead and lesser scaup breeding pairs but missed ~25% of ruddy duck pairs. It is likely that phenology and detection affected estimates of pair abundance for ruddy ducks, whereas phenology alone influenced American redhead, ring-necked duck, and lesser scaup estimates. Had ground crews not needed to nest search immediately following pair counts, conducting a second round of pair counts likely would have reduced bias in breeding pair abundance estimates for all species.

Brood Counts

Our results demonstrate that considerable bias can be removed from brood abundance estimates when >1 survey is conducted. Our estimates of detection probabilities for combined diving duck broods were nearly identical to what Pagano (2007) found in North Dakota when only 1 survey was conducted, and cumulative detection was 9-15% lower under our 2-survey protocol when compared with Pagano and Arnold's (2009*b*) 3-survey protocol. Sightability estimates among species were similar, with canvasbacks having the highest detection probability each year. We didn't identify any single species as having consistently low detection probabilities, which may be why species failed to be included in most of the best-supported models. Also, like Pagano and Arnold (2009*b*), our results suggest assessing detectability for the guild rather than individual species resulted in better-supported models for diving duck brood detectability.

Overall, detection probabilities increased slightly each year (Fig. 5), but the cause of this increase is unknown. Pagano and Arnold (2009*b*) demonstrated that more experienced observers had higher rates of detection than novice observers. In our study, observation crews had similar levels of experience each year conducting brood surveys, but we did not assess individual observer's rates of detection. Conversely, brood detection probabilities within each year declined between the first and second rounds of brood surveys even though all observers had accrued at least 1 round of experience conducting the first round of brood surveys (Fig. 5). Ringelman and

Flake (1980) reported a positive relationship between Julian date and detection probabilities for mallard and teal broods. That effect however, was confounded by 2 factors; brood age and seasonal changes of wetlands (declining water levels and the resulting decrease in the amount of flooded emergent cover for broods to hide), both thought to positively influence brood detection probabilities. Rounds of surveys in our study were separated by 18-21 days each year, and vegetation surrounding wetlands likely grew during that period affecting the observers' ability to detect broods. Our walk-up survey approach is more prone to impacts caused by growing vegetation than roadside surveys, and we consider the reduction in visible area of a wetland to be the primary source reducing detection probabilities of broods between survey rounds.

Brood size had a positive relationship with detectability, with broods having higher numbers of ducklings being more likely to be detected. Mean brood size for diving duck broods was 5.6 ducklings, where $p = 0.55$ and increased by 0.02-0.03 with each additional duckling (Fig. 6). Similarly, brood age also had a positive relationship with detectability, where $p = 0.50$ for class 1A ducklings and increased by ~0.04 per age class (Fig. 7). The positive relationship of these 2 covariates and detectability is consistent with investigations of brood size and brood age separately (Bartonek and Hickey 1969; Ringelman and Flake 1980; Pagano and Arnold 2009b; but see Rumble and Flake 1982 and Giudice 2001). Although this may lead us to conclude that large, older broods have the highest detection probabilities, broods meeting this description are uncommon and are more likely to be multiple cresched broods. Duckling mortality is highest during the first 2 weeks after hatch, resulting in an overall decrease in brood size as ducklings get older and advance age classes (Korschgen et al. 1996; Leonard et al. 1996). Therefore, younger broods tend to be larger broods. Conversely, we observed that adult females were present for 60-70% of observations of class 1A ducklings, but only 22-46% of observations for class 2C ducklings suggesting differences in parental care exist dependent on the age of the brood (De

Sobrino 1995; Pagano and Arnold 2009*b*). An attending female potentially increased detection probabilities of young broods as these same broods often seek refuge or hide when the female is not present (Bartonek and Hickey 1969). Conducting multiple surveys to include all age classes is recommended to assess detection probabilities of broods without individual brood bias due to factors such as parental care.

Time of day influenced brood detectability and was consistent with other studies suggesting surveys taking place in the early mornings and late afternoons have the highest detection probabilities (Diem and Lu 1960; Ringelman and Flake 1980; Pagano and Arnold 2009*b*). In our study on diving ducks, only broods observed during surveys conducted between 1050 and 1230 had resulting detection probabilities < 50%. Pagano and Arnold (2009*b*) reported detection probabilities > 50% for diving duck broods throughout the entire day and increasing to > 60% during the evening. Ringelman and Flake (1980) estimated detection probability of blue-winged teal and mallard broods were < 50% within 180 minutes of sunrise and remained < 50% until at least 800 minutes (13.3 hrs) after sunrise. Higher detection probabilities stemming from increased mid-day activity of diving duck broods may relate to foraging and resting behavior exhibited by this group (Mendall 1958; Joyner 1977; Maxson and Pace 1992). Dabbling duck broods have been observed feeding on invertebrate prey extensively at night (Swanson and Sargeant 1972) and even though direct observations of nighttime feeding by diving duck broods hasn’t been recorded, it likely occurs and has been used to explain mid-morning lulls in diving duck brood feeding activity (Joyner 1977). Feeding activity for both dabbling and diving duck broods is highest in the early morning and near dusk (Diem and Lu 1960; Joyner 1977; Ringelman and Flake 1980), however, time of day appears to have a smaller effect on diving duck brood detection probabilities compared to dabbling duck broods.

The covariate with the strongest effect on diving duck brood detection probability was wetland visibility (the estimated percentage of the inundated wetland unobscured to the observer). Hammond (1970) claimed that brood detection probability was approximately equal to the percent open water in a wetland, and therefore, have a predisposed predictability unrelated to the brood-specific descriptions such as brood size, brood age, and species (Austin et al. 2000; Yerkes 2000). Giudice (2001) also found that brood detection probabilities for mallards were best described by covariates representing visual obstruction, but that additional covariates describing brood descriptions were influential as well indicating heterogeneous detection probabilities for broods. Even though brood detection probabilities were similar to mean percent visibility in our study, models including only wetland visibility ("vis") were not competitive with the top-supported models that have additional descriptive covariates for broods (Ringelman and Flake 1980; Giudice 2001; Pagano et al. 2014; Fig. 8).

Of the survey condition covariates included, only wind proved to influence brood detection probabilities. Surveys with higher average winds had a negative effect on detection probability possibly due to a reduction in the efficiency of ducklings to forage when windy conditions exist. This has been found to cause duck broods to seek refuge in protected areas such as secluded bays or among emergent vegetation where visibility is severely compromised (Ringelman and Flake 1980; Giudice 2001). Cloud cover and precipitation had a negligible effect on detection probabilities, however, we did not conduct surveys during periods of steady rainfall.

Assumptions of surveys

The closure assumption (i.e. no emigration or immigration during the study period) is the most challenging assumption of the double-observer closed capture methodology to meet (Otis et al. 1978; Williams et al. 2002). Although it is not possible to know whether this assumption is met in field studies, abundance estimates are unbiased if both immigration and emigration are random (Kendall 1999). To minimize opportunities of violating this assumption, sampling occasions were completed 3.5 hours apart on average to reduce the possibility of inter-wetland brood movements between occasions and it is unlikely that these movements occurred (Evans et al. 1952; Beard 1964; Rotella and Ratti 1992; but see Smith 1971 for dabbling ducks). Regarding the closure assumption in our overall brood-pair ratios, we chose to combine all broods observed on a study site for comparison with pair observations due to low sample sizes. In previous studies (e.g., Pagano et al. 2014), only broods counted on the same wetlands where pairs had been seen were included in the final analysis. The remaining assumptions were met by relying on our brood marking criteria and completing the surveys quickly and consistently. Similar criteria were used by Pagano et al. (2014) during their investigations into dabbling duck brood detectability.

Comparing Methods

Nest success has been identified as the most important vital rate influencing recruitment for dabbling ducks and has taken precedence as the metric measured to index local waterfowl productivity (Cowardin and Johnson 1979; Klett et al. 1988; Hoekman et al. 2002). Because SA-BPR's include renesting effort, we incorporated renesting propensity into estimates of hen success for comparisons with SA-BPR (Cowardin and Johnson 1979). Adjusting α affected hen success estimates substantially, increasing diving duck productivity by 0.04-0.08 at $\alpha = 0.5$, and by 0.10-0.19 at $\alpha = 1.0$, yet correlations indicated by adjusted R^2 values differed negligibly between the 2 methods (Table 7). Hen success at $\alpha = 0$ and SA-BPR had the highest adjusted R^2 values when compared to hen success at other renesting rates, indicating the predicted relationship between these 2 estimates fit best under the presumption that no renesting occurred. Even though some level of renesting does occur in all but drought years, adjusting α simply resulted in non-linear transformations of nest success that failed to increase correlations (Pagano 2007).

In this investigation of diving duck productivity, we expected to find highly correlated estimates of productivity derived from hen success and SA-BPR given high detection probabilities of diving duck broods were found and causes of mortality for broods is similar to causes of mortality for diving duck nests (Bouffard 1988; De Sobrino 1995; Korschgen et al. 1996). Our results were consistent with a similar study of upland nesting duck productivity and surveys to evaluate diving duck productivity could be implemented where in addition to those. Like Pagano et al. (2014) reported with upland nesting ducks, we found that nest success and SA-BPRs were strongly correlated when multiple species were combined to generate 1 overall estimate for each site. Small sample sizes likely contributed to a lack of correlation in comparisons of canvasbacks, reducing precision of pair and brood abundance estimates which would ultimately skew SA-BPR's (Pagano et al. 2014).

Nearly 25% of the wetlands we surveyed for broods were completely obscured by vegetation so any broods occupying these wetlands had no probability of being detected. Additionally, pairs were likely misrepresented for 4 of the 5 diving duck species we surveyed for based on the early timing of pair counts relative to their nesting phenology. If canvasback productivity estimates derived from hen success and SA-BPR showed moderate levels of correlation, the importance of survey timing could justify weak correlations between estimates for the other species of diving ducks. However, canvasback productivity estimates were weakly correlated and furthermore, removing canvasback data from combined SA-BPR estimates increased the amount of variation in hen success explained by SA-BPR suggesting canvasback data add unexplained variation to the linear model.

Canvasback SA-BPR estimates were more variable than combined SA-BPR estimates but typically t remained realistic, only exceeding 1.00 on a single occasion. SA-BPR estimates >1.00 would suggest there were more broods than pairs on the landscape which is biologically impossible unless females hatch >1 brood. Canvasback productivity is highly influenced by brood survival and has been shown to vary drastically on a local scale, like our SA-BPR estimates, even in hydrologically stable systems (Anderson et al. 1997). Two explanations for widespread low productivity in canvasbacks include poor habitat affecting nest success, and inclement weather affecting brood mortality (Korschgen et al. 1996; Anderson et al. 1997). Results from this study, however, show canvasbacks being more productive when estimated by SA-BPR as compared to corresponding nest success estimates. This could mean canvasback nest success is higher than other species of overwater nests, or our surveys underestimated the number of pairs on our sites, or our surveys overestimated brood abundances on our sites. Even though pair counts were timed based on recommendations in the literature, home range sizes for canvasback females change drastically depending on the stage of nesting cycle they are in, and underestimated pair abundances would bias SA-BPR high. Anderson (1984) estimated home ranges for post-arrival pairs to be 73-ha, pre-laying pairs to be 150-ha, and laying pairs to be about 25-ha. Therefore, one ~65-ha survey plot could be part of multiple female home ranges during post-arrival and pre-laying, yet be absent of canvasbacks once laying has begun. This problem is exacerbated when considering the stage of pairs within a population varies and are unknown to observers, so estimates are subject to uncertainty given the locations of survey plots amongst the everchanging assemblage of home ranges on a study site.

Management Implications

Managers may consider conducting initial surveys to determine relative productivity at low costs, justifying more intensive investigations into specific vital rates such as nest success, brood survival, or other potential limitations to productivity when necessary. Surveys designed to estimate brood abundance for ducks need not take guild specific protocol into consideration, as all covariates investigated have the same effect on detectability on diving ducks as dabbling ducks. Imprecision associated with SA-BPR of canvasbacks should be further explored, with specific attention paid to home range sizes and seasonal movements.

Tables

Table 2.1. Description of covariates measured and included in variety of diving duck brood detection models analyzed using Huggin's closed capture methodology in Program MARK. Covariates included were measured from 1,915 diving duck brood observations in southwestern Manitoba, Canada, 2015-2017.

Covariate	Definition	Factor Type	Abbr.	Class	Range	Mean
Cloud Cover	Average cloud cover during survey.	Environmental	cloud	categorical	0-3	0.8
Precipitation	Average precipitation during survey.	Environmental	precip	categorical	0-3	0.02
Visibility	Percent of inundated wetland unobstructed to observer.	Environmental	vis	continuous	0-1	0.66
Water	Relative wetland inundation.	Environmental	water	categorical	0-5	3.96
Wind	Average wind speed during survey on Beaufort Scale (Simpson 1926).	Environmental	wind	categorical	0-7	2.4
Brood Age	Estimated ageclass of brood as defined by Gollop and Marshall 1954.	Group	ageclass	categorical	1A - 2C	2.4
Brood Size	Number of ducklings in brood.	Group	maxBS	continuous	1 - 15	5.6
Species	Species of duckling brood.	Group	spp	categorical	N/A	N/A
Year	Year	Group	year	categorical	2015 - 2017	N/A
Break	Elapsed time between end of survey 1 and beginning of survey 2.	Temporal	break	continuous	1.92 - 7.76	3.5
Round	Round of survey.	Temporal	round	categorical	1 or 2	N/A
Time Start 1	Time at which first survey effort began.	Temporal	time.1	continuous	6.18 - 14.51	9.24
Time Start 2	Time at which second survey effort began.	Temporal	time.2	continuous	9.42 - 19.83	14.14

Table 2.2. Breeding pair densities per square kilometer estimated and extrapolated from survey plot observations on 7 study sites, each 25 mi^2 (645 ha), in southwestern Manitoba, Canada, from 2015-2017. Four- letter codes represent the 5 target species of diving ducks (CANV = canvasback, LESC = lesser scaup, REDH = American redhead, RNDU = ring-necked duck, RUDU = ruddy duck). Surveys were conducted on sites where intense predator management efforts were being implemented (trapped sites), and sites where no organized effort was being allocated (control sites). Standard deviations are included and were calculated using the Delta method (Powell 2007).

Diving Duck Pair Densities (Pairs/sq.km)- SW Manitoba 2015-2017

		2015		2016		2017				2015		2016		2017	
Control Sites		Density	SD	Density	SD	Density	SD	Trapped Sites		Density	SD	Density	SD	Density	SD
Arrow Creek								Kelloe/Elphinstone[a]							
	CANV	3.0	0.3	2.2	0.3	3.1	0.3		CANV	1.5	0.2	2.0	0.3	1.9	0.3
	LESC	2.5	0.4	2.5	0.3	6.8	0.6		LESC	1.6	0.2	1.4	0.2	0.9	0.2
	REDH	8.8	0.7	4.4	0.4	8.5	0.7		REDH	5.9	0.5	2.5	0.3	1.6	0.2
	RNDU	2.2	0.3	2.5	0.3	2.9	0.4		RNDU	2.5	0.3	2.9	0.4	1.9	0.3
	RUDU	4.6	0.4	2.9	0.3	6.9	0.6		RUDU	3.3	0.5	0.5	0.1	0.7	0.2
Odanah								Minnedosa							
	CANV	2.6	0.3	2.2	0.2	1.5	0.2		CANV	2.5	0.3	4.2	0.4	3.0	0.3
	LESC	3.7	0.5	2.9	0.5	4.1	0.7		LESC	5.3	0.6	6.5	0.8	10.0	1.2
	REDH	7.5	0.5	6.8	0.8	5.9	0.5		REDH	4.9	0.4	5.9	0.4	6.2	0.4
	RNDU	1.3	0.2	2.0	0.3	2.0	0.4		RNDU	1.0	0.1	4.1	0.5	2.2	0.3
	RUDU	6.3	0.5	4.0	0.4	3.8	0.5		RUDU	3.4	0.4	2.8	0.4	3.3	0.5
Raven Lake								Shoal Lake							
	CANV	1.4	0.2	2.4	0.3	2.6	0.3		CANV	1.9	0.2	3.1	0.3	3.7	0.3
	LESC	1.2	0.3	0.8	0.2	1.3	0.2		LESC	3.9	0.6	4.7	0.5	4.3	0.6
	REDH	4.2	0.5	3.9	0.3	5.4	0.6		REDH	6.5	0.6	5.9	0.6	6.8	0.5
	RNDU	2.5	0.3	1.7	0.2	1.4	0.3		RNDU	2.0	0.2	2.0	0.3	3.8	0.5
	RUDU	2.4	0.3	1.4	0.2	1.9	0.2		RUDU	4.7	0.5	4.3	0.4	5.5	0.6

[a] Kelloe site was removed after 2015 and replaced by Elphinstone site for 2016-2017.

Table 2.3. Indicated breeding pairs, number of nests used in survival analysis, and sightability-adjusted brood abundances for overwater nesting ducks on 7 study sites, in southwestern Manitoba, Canada, from 2015-2017. Indicated breeding pairs and SA-brood abundances are from surveys on 25 survey plots (16.4 km^2) from each site, whereas nests used in survival analysis were from any location within the site boundaries where observers had been granted access.

	Pairs			Nests			Broods		
Site[a]	2015	2016	2017	2015	2016	2017	2015	2016	2017
Canvasbacks									
Arrow Creek	48	37	50	14	20	19	36	27	26
Odanah	42	35	24	31	22	23	13	2	13
Raven Lake	23	40	42	20	20	33	15	12	4
Kelloe[b]	24	-	-	7	-	-	22	-	-
Minnedosa[b]	41	68	48	24	39	30	24	16	12
Shoal Lake[b]	31	51	59	29	34	33	40	18	9
Elphinstone[b]	-	33	30	-	52	54	-	3	15
Combined Diving Ducks									
Arrow Creek	341	234	456	68	109	86	203	133	128
Odanah	346	290	280	71	68	104	111	38	65
Raven Lake	191	164	204	71	81	85	82	60	24
Kelloe[b]	240	-	-	70	-	-	132	-	-
Minnedosa[b]	279	378	401	63	129	109	106	97	86
Shoal Lake[b]	308	323	390	99	136	93	176	110	57
Elphinstone[b]	-	150	114	-	115	116	-	33	40

a Kelloe site was only surveyed in 2015. Elphinstone was not surveyed in 2015.

b Sites where predator reduction was being implemented.

Table 2.4. Best-supported Huggin's closed-capture models (as ranked by AIC_c) for estimating detection probabilities of diving duck broods using a 2-occasion encounter history in southwestern Manitoba, Canada, from 2015-2017. In total, 1,915 broods were observed on nearly 11,000 wetlands surveyed during 2 survey rounds (early and late). The best-supported model was selected for use in deriving sightability-adjusted brood abundances and was indicated by the model with the lowest AIC_c. Also included is the null model for reference.

	Model	**AICc**	**Δ AICc**	w_i	**Model Likelihood**	***K***	**Deviance**
2015	{p = round+vis+maxBS+ageclass+cloud}	3987.044	0.000	0.660	1.000	7	3973.013
	{p = spp+round+vis+maxBS+ageclass+cloud}	3990.684	3.640	0.107	0.162	11	3968.612
	{p = round+vis+maxBS+ageclass}	3991.825	4.782	0.060	0.092	6	3979.802
	{p = spp+round+vis+maxBS+ageclass+cloud+wind}	3992.653	5.609	0.040	0.061	12	3968.568
	{p(null)}	4058.060	71.017	0.000	0.000	1	4056.059
2016	{p = spp+round+vis+maxBS+ageclass+break+cloud+wind}	2193.420	0.000	0.406	1.000	13	2167.246
	{p = spp+round+vis+maxBS+ageclass+cloud+wind}	2194.123	0.703	0.285	0.704	12	2169.974
	{p = spp+round+vis+maxBS+ageclass+time.1+break+cloud+wind}	2195.338	1.918	0.155	0.383	14	2167.137
	{p = spp+round+vis+maxBS+ageclass+wind}	2195.428	2.008	0.149	0.366	11	2173.302
	{p(null)}	2284.765	91.345	0.000	0.000	1	2282.763
2017	{p = round+vis+maxBS+time.1+time.st2+break}	1895.262	0.000	0.395	1.000	8	1879.185
	{p = spp+vis+maxBS+time.1+time.st2+break}	1895.809	0.547	0.301	0.761	11	1873.667
	{p = spp+round+vis+maxBS+time.1+time.st2+break}	1896.907	1.644	0.174	0.440	12	1872.738
	{p = round+vis+time.1+time.st2+break}	1898.517	3.255	0.078	0.196	7	1884.457
	{p(null)}	1980.005	84.742	0.000	0.000	1	1978.003
Overall Brood Detectability							
	{p = year+vis+maxBS+ageclass+time.st1+time.st2+wind}	5446.787	0.000	0.367	1.000	9	5428.752
	{p = year+vis+maxBS+ageclass+break+time.st1+time.st2+wind}	5449.435	2.649	0.097	0.266	11	5427.384
	{p = year+vis+maxBS+ageclass+break+time.st1+time.st2+water+wind}	5450.157	3.369	0.068	0.185	12	5426.096
	{p = spp+vis+maxBS+ageclass+time.st1+time.st2+wind}	5451.060	4.272	0.043	0.118	12	5427.000
	{p(null)}	5631.585	184.797	0.000	0.000	1	5629.584

Table 2.5. Logit-link function beta values for parameters included in best performing overall brood detectability model for diving duck broods, including all brood observations from 2015-2017, analyzed using Program MARK. The equation structure for back-transforming the beta estimates to real estimates is as follows: $logit(p) = \beta_0 + \beta_1 + \beta_x \dots$

Parameter	Beta	SE	95% LCI	95% UCI
Intercept	-2.697	0.412	-3.505	-1.889
2016	-2.618	0.416	-3.435	-1.801
2017	-2.496	0.425	-3.330	-1.663
Visibility	2.120	0.250	1.629	2.611
Brood Size	0.113	0.150	0.830	0.144
Brood Age	0.160	0.036	0.089	0.232
Time Start 1	-0.162	0.039	-0.240	-0.084
Time Start 2	0.167	0.039	0.090	0.244
Wind	-0.171	0.039	-0.249	-0.092

Table 2.6. Nest success estimates for overwater nesting ducks on each site-year estimated using Shaffer's logistic-exposure method and bootstrapped 1,000 times to derive 95% confidence intervals. Overwater nesting ducks included canvasback, lesser scaup, American redhead, ring-necked duck, ruddy duck, and mallard in southwestern Manitoba from 2015-2017. Some sites were subject to intensive predator trapping but no treatment effects were found (Johnson, ch1, unpublished).

	2015			2016			2017		
Site	*n*	*S*	95% CI	*n*	*S*	95% CI	*n*	*S*	95% CI
Arrow Creek	68	0.492	(0.354 - 0.646)	109	0.239	(0.163 - 0.332)	86	0.115	(0.072 - 0.180)
Kelloe[a]	70	0.542	(0.415 - 0.687)	-	-	-	-	-	-
Minnedosa[a]	63	0.307	(0.200 - 0.450)	129	0.127	(0.084 - 0.184)	109	0.154	(0.101 - 0.226)
Odanah	71	0.431	(0.299 - 0.589)	68	0.238	(0.149 - 0.354)	104	0.146	(0.093 - 0.217)
Raven Lake	71	0.277	(0.180 - 0.419)	81	0.298	(0.205 - 0.411)	85	0.073	(0.036 - 0.129)
Shoal Lake[a]	99	0.371	(0.270 - 0.488)	136	0.309	(0.223 - 0.405)	93	0.168	(0.105 - 0.250)
Elphinstone[a]	-	-	-	115	0.335	(0.244 - 0.439)	116	0.164	(0.112 - 0.234)

[a] Sites where targeted efforts to reduce predators were being implemented.

Table 2.7. Results from comparisons of SA-BPR and hen success estimate using a linear model from 7 study sites in southwestern Manitoba, Canada, from 2015-2017. Nest success estimates were generated from a sample of overwater nests located and monitored on the same study sites, during the same years. Hen success estimates were calculated using Cowardin and Johnson's (1979) formula for productivity, adjusted to represent 3 different renesting propensity scenarios: HS (α = 0): no failed nesting attempts are re-initiated by females; HS (α = 0.5): 50% of failed nesting attempts are re-initiate females; HS (α = 1): original formula built for mallards (*Anas platyrynchos*). Sightability-adjusted brood-pair ratios were compared with hen success estimates for each species, and 2 additional data sets: "Combined" included breeding pairs and brood abundances of all diving duck species and "Combined (no canv)" included pair and brood abundances for all species except canvasbacks. The comparison which explaining the most amount of variation was indicated by the highest adjusted *R*2 value and is bolded for each species or group if it's p-value was statistically significant.

	HS (α = 0) ~ SA-BPR			HS (α = 0.5) ~ SA-BPR			HS (α = 1) ~ SA-BPR		
	p-value	Mult. R^2	Adj. R^2	p-value	Mult. R^2	Adj. R^2	p-value	Mult. R^2	Adj. R^2
Canvasback	**0.03841**	**0.2414**	**0.1939**	0.04064	0.2367	0.1889	0.04541	0.2274	0.1791
Lesser Scaup	0.1394	0.1314	0.07707	0.1574	0.1209	0.06596	0.1846	0.1073	0.0514
Redhead	0.07676	0.1828	0.1317	0.06859	0.1924	0.142	0.06002	0.2038	0.154
Ring-necked Duck	**0.00033**	**0.5635**	**0.5363**	0.00044	0.5481	0.5199	0.00076	0.518	0.4879
Ruddy Duck	0.007262	0.3713	0.332	0.00539	0.3927	0.3547	**0.004179**	**0.4104**	**0.3735**
Combined	**0.00206**	**0.4574**	**0.4234**	0.00209	0.4557	0.4228	0.00219	0.4531	0.419
Combined (no canv)	**0.000573**	**0.5339**	**0.5048**	0.01517	0.3161	0.2733	0.01551	0.3134	0.2714

Figures

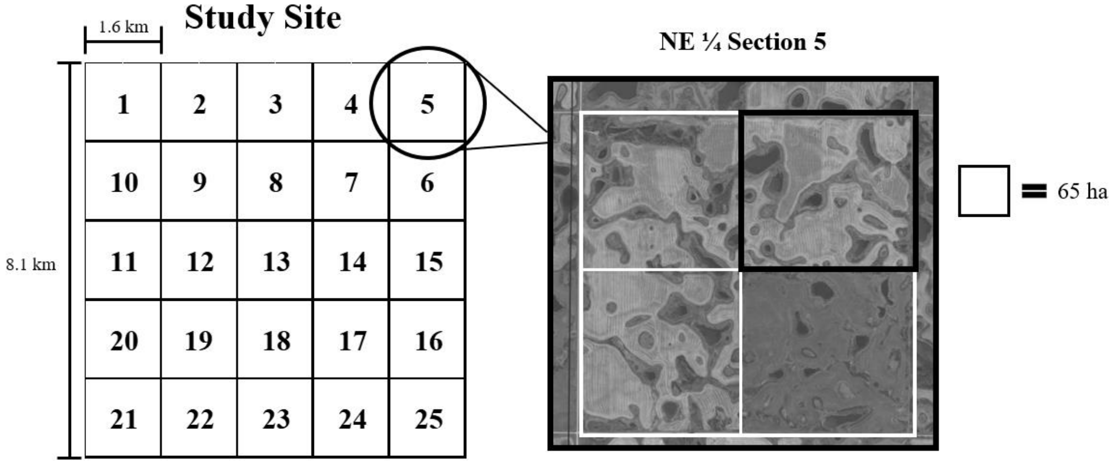

Figure 2.1. Illustrated description of survey plots within a legal section. In each legal section (numbered 1-25 in left inset), one plot was randomly selected from the four available (outlined black in right inset) for pair and brood surveys. This plot selection process occurred on all 6 study sites resulting in 150 survey plots each year for 2015-2017. The same survey plots were used each year unless permission to access a plot was withdrawn, in which case a substitute plot was selected at random from those still available. Study sites occurred in southwestern Manitoba, Canada, near the towns of Minnedosa and Shoal Lake.

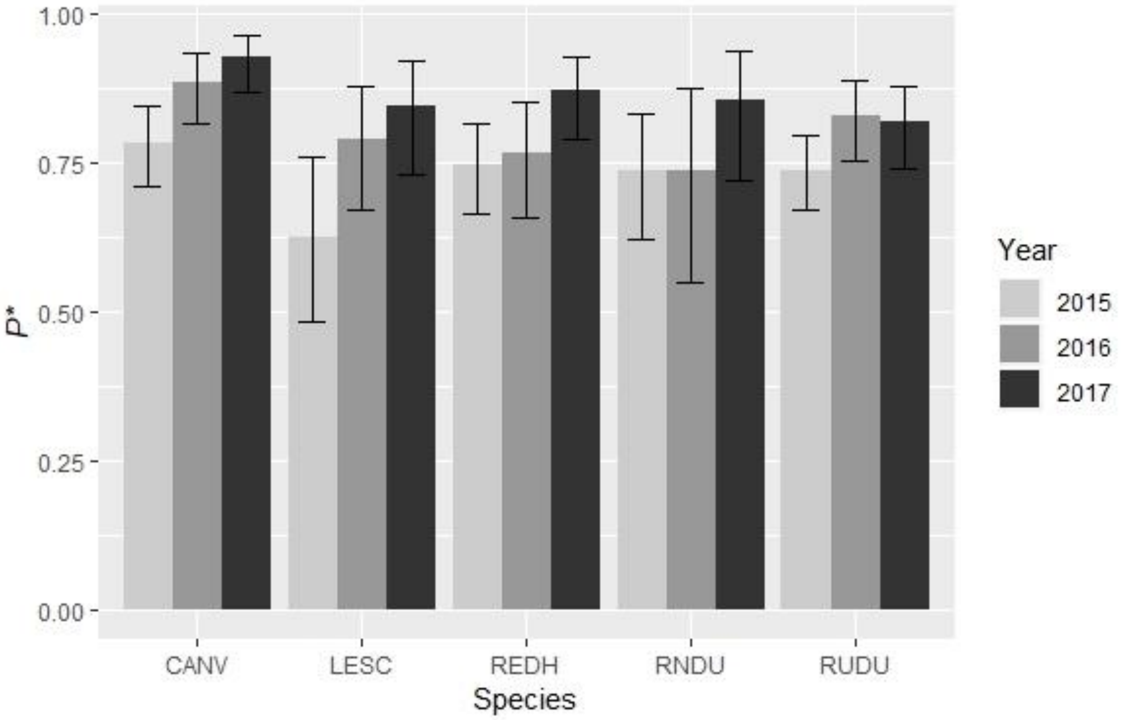

Figure 2.2. Species-specific cumulative detection probabilities (*P**) for 5 species of diving ducks (CANV = canvasback, LESC = lesser scaup, REDH = American redhead, RNDU = ring-necked duck, RUDU = ruddy duck) in southwestern Manitoba, Canada, from 2015-2017. Estimates are generated from 1,915 unique 2-occasion encounter histories using the Huggin's closed-capture approach in Program MARK. Error bars represent 95% confidence intervals.

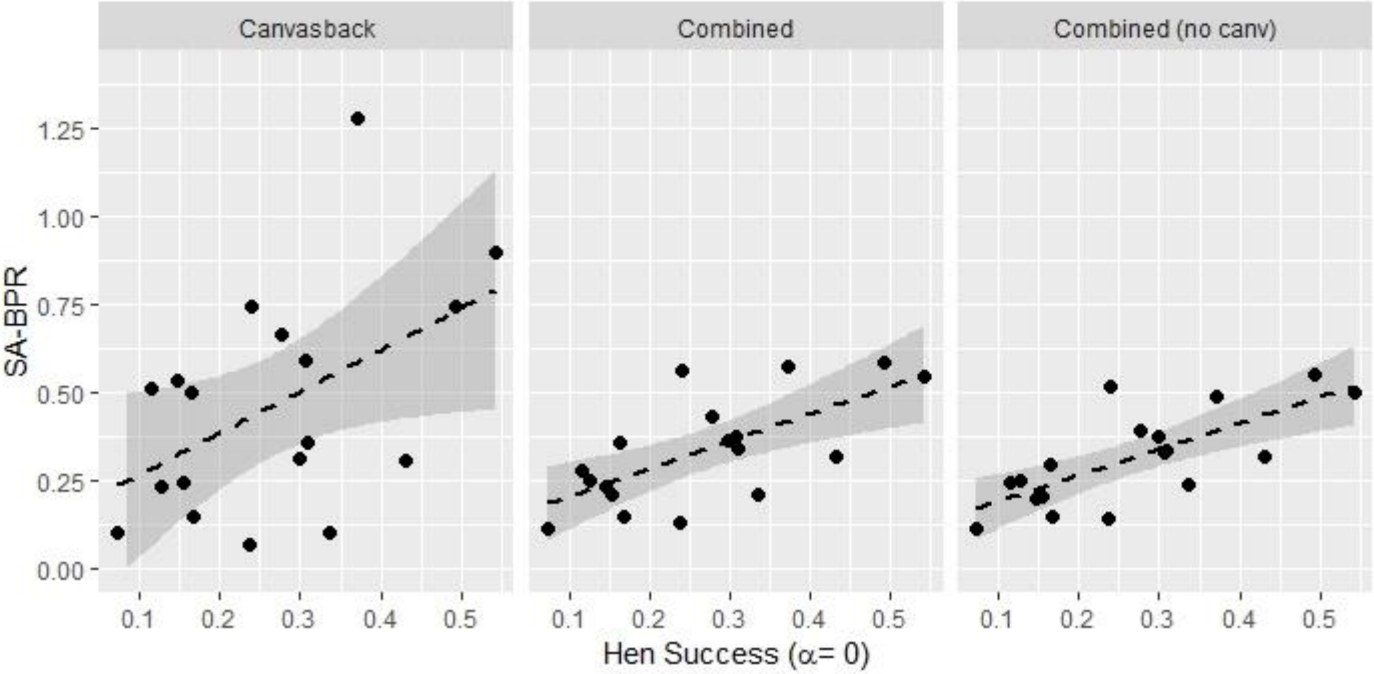

Figure 2.3. Plotted correlation between sightability-adjusted brood-pair ratios and hen success for canvasbacks, combined diving ducks, and combined (no canvasbacks) in southwestern Manitoba, Canada, from 2015-2017. SA-BPR estimates were most correlated with hen success when hen success was evaluated at $\alpha = 0$, representing no renesting, and were fit by a linear model represented by the dashed line ($n = 18$, Canvasback: $R^2 = 0.19$, $P = 0.038$; Combined: $R^2 = 0.42$, $P = 0.002$; Combined- no canvasback: $R^2 = 0.50$, $P = 0.0005$). 95% confidence intervals are shaded grey.

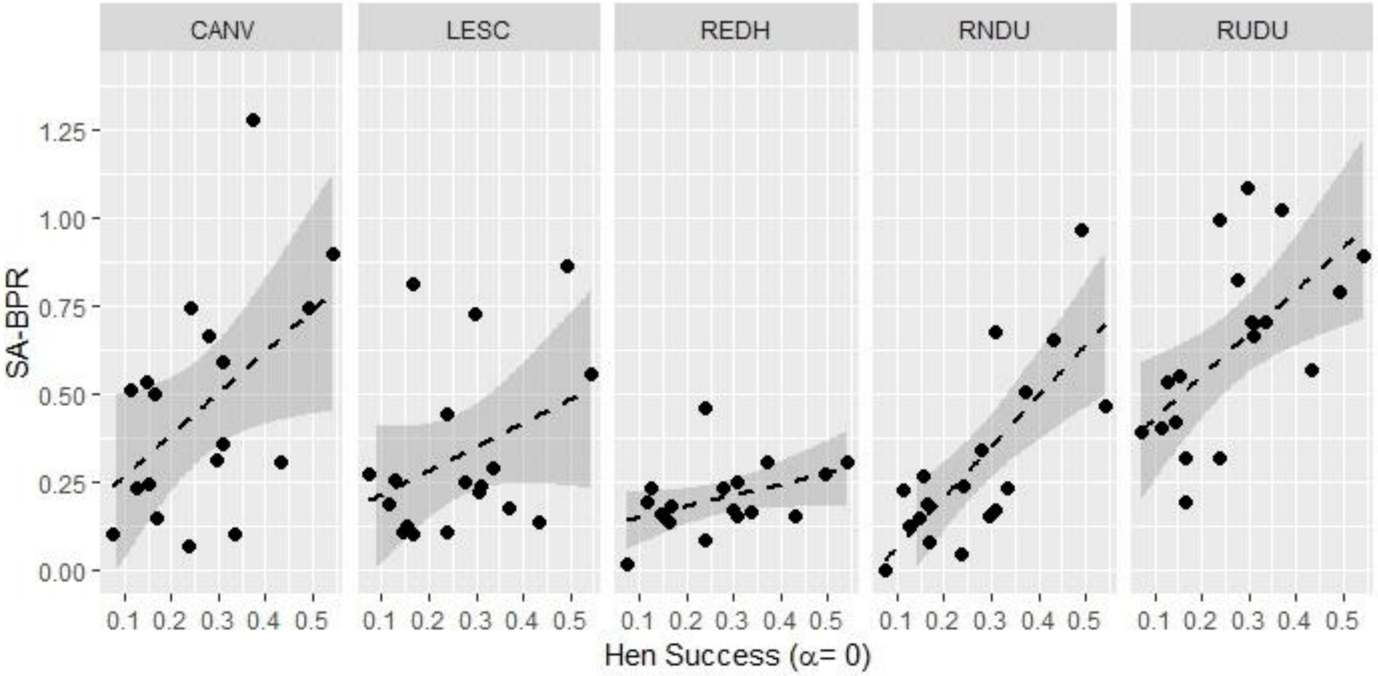

Figure 2.4. Plotted correlation between sightability-adjusted brood-pair ratios and hen success for the 5 target species of diving ducks observed (CANV = canvasback, LESC = lesser scaup, REDH = American redhead, RNDU = ring-necked duck, RUDU = ruddy duck) in southwestern Manitoba, Canada, from 2015-2017. SA-BPR estimates were most correlated with hen success when hen success was evaluated at α = 0, representing no renesting, and were fit by a linear model represented by the dashed line (*n* = 18, canvasback: R^2 = 0.19, *P* = 0.038; ring-necked duck: R^2 = 0.53, *P* = 0.0003; ruddy duck: R^2 = 0.33, *P* = 0.007). 95% confidence intervals are shaded grey.

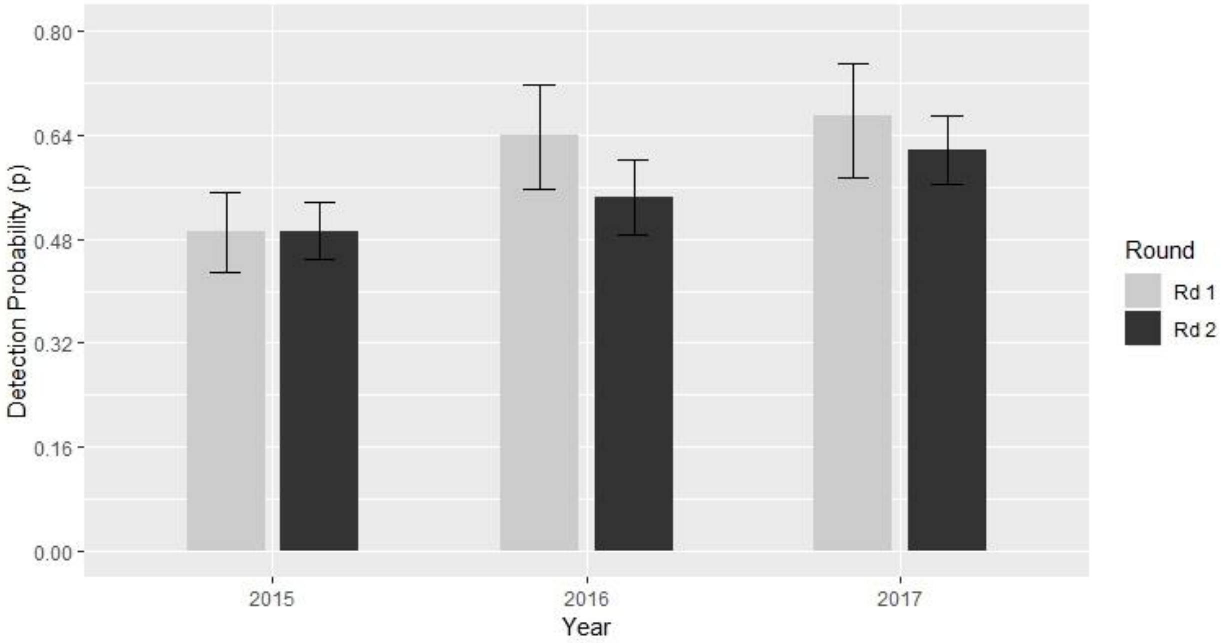

Figure 2.5. Detection probabilities for combined diving duck broods between 2 rounds of brood surveys conducted in southwestern Manitoba, Canada, from 2015-2017. Timing of the first survey round was 3 weeks after peak canvasback hatch, occurring within the first week of July, and round 2 began 18-20 days following the conclusion of round 1 to include renesting and late-initiating species. Error bars represent 95% confidence intervals.

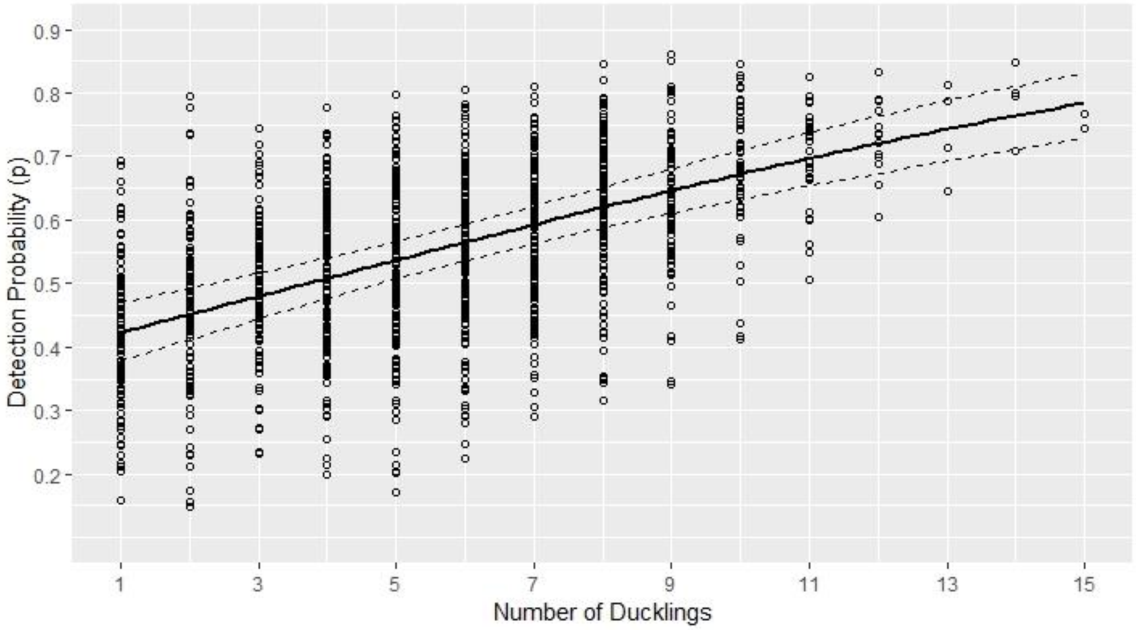

Figure 2.6. Predicted detection probabilities of diving duck broods as a function of brood size (number of ducklings observed), taken from 1,915 broods observed in southwestern Manitoba, Canada from 2015-2017. The solid black line indicates the predicted detection probability, dashed black lines are 95% confidence intervals estimated using Program MARK. The open circles represent predicted detection probabilities of individual broods observed when all other covariates are held at their means.

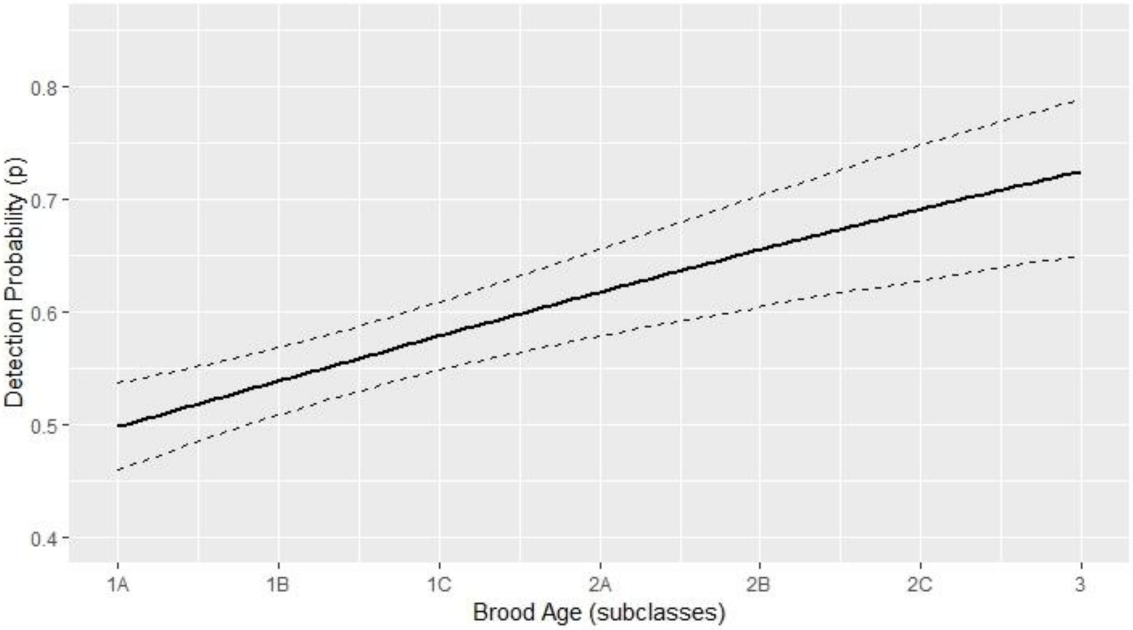

Figure 2.7. Predicted detection probability of diving duck broods as a function of brood age (age class) estimated using Program MARK. Age classes are represented on the x-axis as described by Gollop and Marshall (1954). Black dashed lines are 95% confidence intervals estimated using Program MARK.

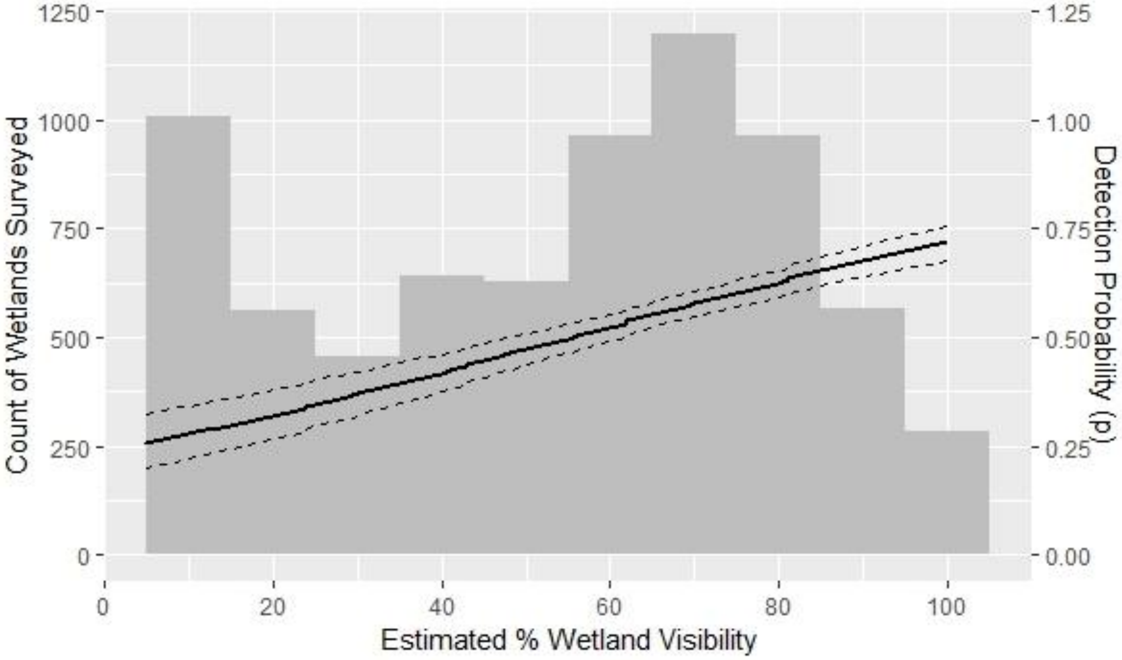

Figure 2.8. Distribution of wetlands surveyed when water surface of wetland was not completely obstructed by vegetation (>0% visibility) and corresponding predicted detection probability of diving duck broods indicated by the solid black line. Dashed lines represent 95% confidence intervals estimated using Program MARK. Brood observations during surveys were recorded in southwestern Manitoba, Canada, from 2015-2017.

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