2022

POPULATION MONITORING OF WHITE-TAILED DEER IN RHODE ISLAND

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POPULATION MONITORING OF WHITE-TAILED DEER

IN RHODE ISLAND

BY

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A THESIS SUBMITTED IN PARTIAL FULFILLMENT OF THE

REQUIREMENTS FOR THE DEGREE OF

MASTER OF SCEINCE

IN

BIOLOGICAL AND ENVIRONMENTAL SCIENCES

UNIVERSITY OF RHODE ISLAND

2022

MASTER OF SCIENCE THESIS

OF

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UNIVERSITY OF RHODE ISLAND
2022

ABSTRACT

Wildlife conservation and management occurs across the world through many different mechanisms and underlying principles. North America has developed a unique and successful process coined the North American Model of Wildlife Conservation. A key outcome of this model is that wildlife science informs management decisions, which are made by government officials in the public's trust. If a species undergoes some form of legal take, managers are often required to ensure it is done responsibly with empirical evidence and consideration of ecological and societal objectives. Recent research suggests that 60% of wildlife management systems in Canada and the United States were not using science to guide their decisions, as they contained fewer than half of what they referred to as four "fundamental hallmarks of science": measurable objectives, evidence, transparency, and independent review. We borrow from their framework and expand on it by evaluating whether white-tailed deer (*Odocoileus virginianus*) management in the northeastern United States includes the essential elements of a structured decision-making process. Our aim is to evaluate the regional management of a species that receives considerable focus to better understand whether the ideals of the North American Model of Wildlife Conservation are being implemented by way of a logical, transparent, and science-based decision-making process. Of the 11 states evaluated, seven had published a white-tailed deer management plan. Of these seven, we found that the "hallmarks" and most structured decision-making components were present, and the information collected was being used to inform decisions. Our findings indicate four main ways white-tailed deer management may be improved in the northeast United States: 1) states without a management plan should develop one, 2) states should incorporate an external review process, 3) states could consider alternative actions for each measurable objective and their consequences, and 4) states need to consider tradeoffs among multiple and possibly conflicting objectives. Our recommendations should lead to increased management transparency and build public support.

Additionally, a key principle of The North America model of Wildlife Conservation is that science is the proper tool for discharging wildlife policy. Using science to understand population abundances and dynamics is especially critical in managing harvested wildlife. Tracking population changes allows resource managers to adapt regulations to ensure populations are maintained. In Rhode Island, USA white-tailed deer (Odocoileus virginianus) are annually harvested, but there is no systematic annual population estimation to track changes, which may put the population and forest ecosystem at risk. Our objective was to evaluate the utility of statistical population reconstruction (SPR) to monitor white-tailed deer in Rhode Island by estimating annual deer abundance, harvest probabilities, and recruitment for males and females, separately. To do so, we used age-at-harvest data collected from hunter harvested deer from state operated check stations (2011-2020) and online/phone reporting, hunter effort derived from annually reported deer harvest, and natural mortality probabilities from the literature. Without a reliable measure of reporting rate, we considered three possible reporting rates (25%, 50%, and 75%). As not all deer reported were aged, we used random forest models to predict the age of 19,277 deer reported via mail-

in/online/phone using age, weight, sex and antler beam measurements of deer checked by staff. The out-of-sample prediction accuracy was between 85-99% with most over 90%. We estimated male abundance with a 75% reporting rate to range from a low of 9,503 (SE, 1,291) in 2017 to a high of 15,767 (SE, 2,183) in 2011, with the most current estimate at 10,054 (SE, 1,325) in 2020. Using a 50% reporting rate, male abundances were higher, ranging from a low of 13,730 (SE, 1,753) in 2017 to a high of 22,271 (SE, 2,912) in 2011, with the most current estimate at 14,031 (SE, 1,745) in 2020. Using a 25% reporting rate, male abundances were the lowest, ranging from a low of 9,310 (SE, 362) in 2015 to a high of 10,766 (SE, 369) in 2019, with the most current estimate at 10,525 (SE, 362) in 2020. Depending on the reporting rate, the male population between 2011-2020 was estimated to be either slightly increasing or decreasing. The SPR failed to produce realistic estimates for females with estimated harvest probabilities near or at zero, which inflated abundance estimates to unreasonable values (>1 million). Overall, SPR appears to be a useful methodology for monitoring deer populations in Rhode Island. However, to rely on it as part of management policy will require several improvements over the current implementation. Foremost, it is recommended that hunter effort, reporting rate and survival probability are determined in Rhode Island via additional research, such as hunter surveys and survival studies.

ACKNOWLEDGMENTS

I would like to thank hunters and sport shooters who provide funds through the Pittman Robertson Act, which place an excise tax on firearms, ammunition, and other hunting equipment. This allows the United States Fish and Wildlife Service the capability to provide funding for research like this. Without them or those who came before them who passed Pittman Robertson Act in 1937, funds may not have been available to fund this research.

In addition, I would like to thank the Rhode Island Department of Environmental Management, Division of Fish and Wildlife for allocating funds to this project. Specifically, Jay Osenkowski for providing support to further my education and career. This research is critical to the development of my knowledge and to better the deer program in Rhode Island on behalf of the public. In addition, I owe thanks to Leland Mello, Charles Brown, Josh Beuth and the remainder of Rhode Island Division of Fish and Wildlife staff who have assisted me throughout my career.

I would like to thank the University of Rhode Island for overseeing this project, especially my major professor Dr. Brian Gerber for his patience with me as a full-time student and full-time employee.

Last, but not least, thank you to my family and friends who have supported me through this process. Its been a long journey, I know based on the routine question, “wait, your still in school?” but it was worth it.

PREFACE

The two chapters of this thesis have been provided in the manuscript format of the respective journal they were submitted to. Manuscript 1 (includes Tables, Figures, and References A) and Manuscript 2 (includes Tables, Figures, and References B) follows the Journal of Wildlife Management journal guidelines. The end of each chapter contains references.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

MANUSCRIPT 1

1 April 2022

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RH: Ferreira and Gerber ● Deer Management

White-tailed Deer Management Review in the Northeast United States with Respect to Decision Science

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ABSTRACT Wildlife conservation and management occurs across the world through many different mechanisms and underlying principles. North America has developed a unique and successful process coined the North American Model of Wildlife Conservation. A key outcome of this model is that wildlife science informs management decisions, which are made by government officials in the public's trust. If a species undergoes some

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form of legal take, managers are often required to ensure it is done responsibly with empirical evidence and consideration of ecological and societal objectives. Recent research suggests that 60% of wildlife management systems in Canada and the United States were not using science to guide their decisions, as they contained fewer than half of what they referred to as four “fundamental hallmarks of science”: measurable objectives, evidence, transparency, and independent review. We borrow from their framework and expand on it by evaluating whether white-tailed deer (*Odocoileus virginianus*) management in the northeastern United States includes the essential elements of a structured decision-making process. Our aim is to evaluate the regional management of a species that receives considerable focus to better understand whether the ideals of the North American Model of Wildlife Conservation are being implemented by way of a logical, transparent, and science-based decision-making process. Of the 11 states evaluated, seven had published a white-tailed deer management plan. Of these seven, we found that the “hallmarks” and most structured decision-making components were present, and the information collected was being used to inform decisions. Our findings indicate four main ways white-tailed deer management may be improved in the northeast United States: 1) states without a management plan should develop one, 2) states should incorporate an external review process, 3) states could consider alternative actions for each measurable objective and their consequences, and 4) states need to consider tradeoffs among multiple and possibly conflicting objectives. Our recommendations should lead to increased management transparency and build public support.

KEY WORDS deer, management plan, Northeast, *Odocoileus virginianus,* structured decision making, white-tailed deer.

There is no single model or set of principles that is globally applied to the management and conservation of wild animals. In North America, the North American Model of Wildlife Conservation (NAMWC) is used, which is governed by seven fundamental principles: wildlife as a public trust resource, elimination of markets for wildlife, allocation of wildlife by law, wildlife can only be killed for a legitimate purpose, wildlife is considered an international resource, science is the proper tool for discharging wildlife policy, and democracy of hunting. A key outcome of these principles is that wildlife is a public resource in which science is used to inform management decisions, which are made by government officials on behalf of the public (stakeholders). In most cases, regulations are set by wildlife managers or legislation is proposed to the legislature (Wildlife Society 2010). The Public Trust Doctrine, which the Supreme Court decided, Martin v. Waddell, 41 U.S. 367 (1842) outlines how certain resources (e.g., wildlife) cannot be taken into private ownership and must be managed as a public resource (Organ et al. 2012). States are held responsible for wildlife on behalf of the stakeholders so it is critical that wildlife is managed following the NAMWC to ensure sustainable use and transparent government processes based on empirical science. This is often a difficult and challenging task for wildlife management agencies, as they must manage wild animal populations in the face of competing interest (e.g., social and environmental) and short- and long-term objectives as well as stochastic environmental events.

This is the case with white-tailed deer (*Odocoileus virginianus*; hereafter deer) in the United States, in which population management objectives will often include social factors, such as hunter participation and satisfaction, impacts to the general (non-hunting) public (e.g., deer vehicle collisions, crop damage, property damage) and environmental factors that include forest and deer health. In some instances, maximizing harvest can meet short and long-term objectives as increased harvest can reduce negative impacts of high deer densities, increase hunter satisfaction, and maintain a healthy forest and deer population in perpetuity. However, challenges arise as a healthy forest and deer population often are products of lower deer densities, while high hunter participation and satisfaction are often products of higher deer densities.

To make effective science-based population decisions to meet short- and long-term objectives, it is critical to have: an understanding of the ecology of the species, a thorough knowledge of the population dynamics, a monitoring strategy that relates population measurements to management objectives, and a clear understanding of stakeholders' interests and engagement with the species (Williams et al. 2002, Nichole and Williams 2006, Conroy and Peterson 2013). Achieving these criteria in a perspicuous and transparent manner is challenging. Namely, because the decision-making process for many natural resource problems is complex. One approach management agencies can take is to employ structured decision making (SDM), which provides a logical framework to a complex decision process. The SDM framework guides the decision maker to formulate the problem within the environmental and social context, determine the desired objectives, compile the alternative sets of actions (management decisions in this case), evaluate the consequences of those actions, and compare the tradeoffs among actions in terms of meeting multiple objectives (Robinson et al. 2016). Using an SDM framework assists managers to understand the species and population by evaluating evidence to determine effective management actions to reach desired objectives and goals (e.g., reduce deer vehicle collisions, increasing harvest, etc.).

One of many positive outcomes of following the SDM framework is that the entire decision process can be transparent (Martin et al. 2009), which builds trust between the stakeholders and decision-makers (McCool and Guthrie 2001). Other outcomes are justification for management actions which may reduce political interference or justify funding needs. An example of an SDM application in wildlife management was its use by Montana, Fish, Wildlife and Parks in providing a transparent and state-wide decision-making framework to minimize risks of pneumonia epizootics for bighorn sheep (*Ovis canadensis*). This occurred based on a clearly defined decision context, rather than being limited to reactive measures following an outbreak (Sells et al. 2016).

Conversely, in the Fraser River of southwest British Columbia, the Department of Fisheries and Oceans management of sockeye salmon (*Oncorhynchus nerka*) relied on an informal process of bargaining and trading among the key interests; quantitative analyses were conducted to estimate returning adults, but little was done in terms of formal or structured analyses of management alternatives. This significantly limited the capacity for making key tradeoffs in a defensible and transparent manner (Gregory and Long 2009). When managers are not transparent, purposefully, or not, conflict or tension can arise between stakeholder's and managers (Irwin et al. 2011). Documenting the decision-making process and its essential elements makes it readily available for stakeholder and independent/external review, as well as decision maker reference over the tenure of the document. The need of stakeholder input and a form of independent review is critical in assessing management plans for transparency, shortcomings, errors, and rigor (Artelle et al. 2018).

While most wildlife management plans may touch on some of the elements of an SDM process, they may not do so completely. A review by Artelle et al. (2018) found that 60% of wildlife management systems (n = 667) in Canada and the United States contained fewer than half of the “four fundamental hallmarks of science”, which were defined as: measurable objectives, evidence, transparency, and independent review. Failing to have all hallmarks could lead to real or perceived mismanagement. For example, failing to explicitly state management objectives make it difficult to evaluate improvement, success, or failure. Thus, a deer monitoring program may estimate a 10% annual increase in deer density, but without a clearly articulated objective, it is unclear whether this is good, intended, or whether current management is working.

Artelle et al. (2018) also found that big game taxa (versus other taxa) and jurisdiction at increasing latitude tend to achieve the four fundamental hallmarks of science. A likely reason is that big game (particularly deer) is typically in high demand from hunters and in some instances, the number of animals that can be sustainably harvested is less than the number of hunters. Therefore, the management agency must be certain that the realized harvest does not exceed maximum sustainable harvest. The level of certainty to accurately allocate permits often requires more extensive research to determine the ecology and population dynamics of the species, thus leading to increased rigor and likelihood of achieving the four hallmarks. Furthermore, there is also incentive for the agency to prevent mismanagement that would result in under or overharvest and potential population declines, as this could impact hunting license and permit sales, thereby reducing available funds which are used for a variety of state-wide wildlife conservation programs (e.g., species of concern, land purchases, habitat restoration, etc.).

In addition to managing responsibly for the sake of wildlife, public comments toward wildlife management agencies often largely come from hunters, regarding an upcoming hunting season. To answer stakeholders' questions preemptively, management plans should be published in an appropriate language level, aimed to ensure stakeholders understand that there is a clear framework to the species management, specifically annual harvest decision making and population monitoring. Using the SDM (or similar) process explains management information and decisions clearly, increasing transparency and trust. Stakeholder trust is important to agencies as they rely on stakeholders as a tool in wildlife management. In the case of managing deer and other game species, it is critically important to maintain population levels at biological and cultural carrying capacity to maintain healthy wildlife populations, satisfy stakeholders, and provide a source for funding wildlife conservation.

Here, we are interested in understanding whether deer in the northeast United States are managed following the NAMWC, specifically as it relates to the elements of a SDM process. As deer are one of the most hunted big game species in the United States and are the only native big game species hunted in several northeast states, except ME, NH and VT (where moose hunting occurs), there is significant stakeholder interest in sustainably managing harvest, as well as promoting healthy forests. Following the findings from Artelle et al. (2018), we expected state management agencies would achieve the four hallmarks and essential elements of the SDM process in the management of deer in the northeast. We reviewed northeastern state's deer management plans (DMP) and specifically evaluated them in reference to SDM elements, 1) identifying management objectives (goals/measurable objectives), 2) considering alternative actions (actions/alternative actions) and their consequences, 3) evaluating actions relative to objectives (monitoring), and 4) identifying the optimal action considering tradeoffs among multiple objectives (tradeoffs). We further included the additional elements from Artelle et al. (2018) that are not explicitly included in the SDM framework: 1) transparency, and 2) review (Table 1). Secondly, we established whether criterion were logically linked together. Lastly, we compared our findings with Artelle et al. (2018), looking specifically at the four hallmarks of science. We hope this work will offer guidance on how states could improve their management of game species to follow the NAMWC, which will ideally lead to increased rigor in managing populations with trust from stakeholders. This should also help states build more cohesive regional management strategies.

STUDY AREA

This research area was the northeastern Unites States, including Connecticut (CT), Delaware (DE), Maine (ME), Maryland (MD), Massachusetts (MA), New Hampshire (NH), New Jersey (NJ), New York (NY), Pennsylvania (PA), Rhode Island (RI), and Vermont (VT) (Figure 1). The states were selected to include all the northeastern states with the addition of MD and DE as they are states with the highest human density outside of the northeast, resembling RI's populations as this research will directly be used to develop RI's DMP.

METHODS

We attempted to acquire DMPs for the broad northeastern region by searching the official website for each respective state wildlife agency; if a DMP was not available online, we contacted the state's deer program lead, requesting the plan.

We evaluated each DMP in terms of the criterion; goals, measurable objectives, monitoring, actions/alternative actions, consequences, tradeoffs, transparency, and review (Table 1). First, we evaluated whether each criterion was present. We did not assume DMPs would use the exact SDM language or structure, such that we attempted to identify language that outlined the essential ideas of each criterion of interest (Table 1). Second, if a plan identified goals, measurable objectives, and actions, we evaluated whether these were logically linked together. For example, once a goal has been established, there must be a way to determine if the goal is being achieved. To determine this, objectives not only need to be measurable, but they must be directly linked to the goal. The same is the case for actions and monitoring strategies being related to the objective. These linkages provide an efficient and effective way to determine whether states are following the SDM process (Neckles et al. 2015). To quantify linkages, if the relationship was direct, it was given a score of one. If the relationship was not direct, it was given a score of zero. This was done for all possible criterion links; goals with a directly related measurable objective, measurable objectives with at least one directly related monitoring strategy, measurable objectives with at least one action, and measurable objectives with alternative actions. We report the percentage of direct links for each criterion and summarize the states that did or did not meet a minimum threshold of 80% for a criterion. We chose an 80% threshold to evaluate states to indicate whether most criterion are being achieved.

In addition to those criteria, we also evaluated DMPs for the presence of consequences, tradeoffs, transparency, and review. Consequences were evaluated based on the presence of a DMP stating the potential outcomes based on actions (e.g., increasing the hunting season by x days will increase hunter satisfaction by y percentage). We differentiate consequences from tradeoffs in that; tradeoffs are specifically about the objectives and consequences are about the effects of actions. Tradeoffs were evaluated based on the presence of a DMP stating how multiple objectives would be positively or negatively impacted by a certain action (e.g., increasing deer density via certain actions will increase the likelihood of meeting a hunter satisfaction objective, but jeopardize an objective on maintaining forest health). Transparency was evaluated based on whether the DMP was publicly available (e.g., accessible on the state wildlife agency's official website) and recorded whether it had been externally/independently reviewed. An internal review did not satisfy the criterion; most, if not all, the DMPs would have gone through an internal review process. Lastly, we summarized results by criterion in two different ways, by averaging across all states and only states with a DMP.

We compared our findings from evaluating 11 northeastern states for their presence of the four hallmarks: measurable objectives, evidence, transparency, and independent review with the results from Artelle et al. (2018) for all states. If a state satisfied at least one part of a criterion we considered that criterion present (e.g., only one of the following needed to be present for evidence: report quantitative information about populations, report uncertainty in population parameter estimates or estimate realized hunting rates).

RESULTS

We were able to acquire DMPs from seven of the 11 states: Delaware, Maine, Maryland, New Hampshire, New York, Pennsylvania, and Vermont (see Appendix 1 for DMP).

New Jersey had a policy document that was developed to specifically address crop damage caused by deer. This document provided some insight to deer management but was not equivalent to a holistic DMP. MA, CT, and RI did not have publicly available DMPs and it was confirmed with each state's deer project leader that there was no documented DMP. However, per conversation with each state's deer project leader, MA stated they have an ongoing verbal plan that has never been recorded on paper and has annual meetings with their Fish and Wildlife Board to review their "plan" (D. Stainbrook, Massachusetts Department of Fish and Game, personal communication). Connecticut's project leader outlined that their goal is to manage deer, so they are in balance with both biological and cultural carrying capacity (H. Kilpatrick, Connecticut Fish and Wildlife, personal communication). Rhode Island's deer project leader stated they currently manage deer for a healthy ecosystem, healthy deer population, and maintaining sustainable harvest while minimizing negative impacts from deer (first author). All seven states with a DMP identified their goals, measurable objectives, action(s), and monitoring strategies in some way, while eight out of the 11 states did not conduct an independent review process (Table 2).

Of the 11 states reviewed, goals for deer management comprised managing deer and their habitat sustainably with considerations to cultural carrying capacity and hunter satisfaction. Measurable objectives for each goal varied among states, as did the actions considered. For example, the first goal of NY's DMP was "Population Management: Manage deer populations at levels that are appropriate for humans and ecological concerns." Measurable objectives based on this goal were to "assess and monitor deer population size and condition using best available techniques." Furthermore, their monitoring strategy is to "annually collect sex, age, antler measurements, and other biological data as needed to monitor trends in deer condition and population dynamics by WMU [wildlife management unit] Aggregate." This tiered approach lays out the process needed to achieve the main goal. All states with a DMP formatted and linked the criteria in a logical fashion. For example, in PA's DMP, their first goal was to "Manage deer for a healthy and sustainable deer herd" followed by an objective, "Maintain reproduction at or above 1.50 embryos per adult doe" followed by strategies to collect such data "Annually collect reproductive data from road-killed deer."

All states with a DMP (n=7) had goals and measurable objectives present. Of those seven states, on average, 92% of the measurable objectives were directly linked to their goals. Individual state's measurable objectives were directly linked to their goals 67-100% of the time (Tables 2, 3). Six states with a DMP, had at least 80% of their measurable objectives directly related to their goals. We found all states with a DMP directly related their measurable objectives to at least one monitoring strategy; on average, 82% of the time and ranging from 17-100%. Again, six states with a DMP had over 80% of their monitoring strategies directly related to the objectives. We found all states with a DMP had measurable objectives with at least one linked action occurring on average 83% of the time and ranging from 17-100%. Again, six states had at least 80% of their objectives with at least one action. We found that each state's DMP listed alternative actions on average 59% of the time and ranging from 17-84%. Stating what actions were currently occurring/going to occur to achieve objectives was not clear for most DMPs. Only one of the seven states (PA) had over 80% of their objectives with alternative actions. The remaining states included fewer alternative actions in their DMP with the lowest states being VT at 33% and NH at 17%.

We found that no state explicitly considered a process to evaluate consequences among actions. When states did list actions they did not state when they would be put into place. Likewise, no state explicitly stated how they made decisions while considering tradeoffs among objectives. However, some states did discuss that management actions or decisions will benefit their goal but may impact other objectives negatively (e.g., NY stating, "Further, determination of an acceptable impact threshold will invariably involve trade-offs between desired levels of deer abundance and ideal forest composition"). Here, the desired level of deer abundance for hunters may be above what is desired to maintain an ideal forest composition. We found all seven states with a DMP to be transparent (since they published a DMP). However, we found only three states used an external reviewer to evaluate their DMP (DE, MD, and PA).

In examining the data from Artelle et al. (2018) for the 11 states considered here (Table 4), we found that 100% of these states had at least half of the four hallmarks (Table 5). This exceeds the 60% found to achieve this for all North American wildlife management systems they surveyed. Specifically, seven states (CT, DE, MD, MA, NY, RI, and NJ) had two of four of the criteria, while one state had three of four (VT) and three states (ME, NH, and PA) had all four criteria (Table 5). We found all seven states with a DMP identified more than half of the hallmarks described in Artelle et al. 2018 (Table 2). Focusing only on the SDM criterion, all states that published a DMP had at least five of eight (63%) criteria listed.

DISCUSSION

Managing animal populations without a clearly defined and approved plan can lead to haphazard and unconnected decisions over time. If the plan is not written in a transparent decision process it may also jeopardize the public's perception of government, ongoing management actions, and their trust in the public doctrine. Further, it is difficult to impossible to maintain a long-term, consistent, decision-making process with staff turnover when there is no guidance document (e.g., management plan). The SDM process is valuable because it is proactive, rather than reactive. Forethought can be placed into decisions that must be made before they arise, preventing a reactive decision that may be inconsistent, and have potential consequences to the species or stakeholders. Additionally, it provides the necessary information, (e.g., goals, measurable objectives, actions, and monitoring strategies) to write a management plan which can assist in the event staff turnover occurs and provides additional benefits to a program. Employing SDM results in a rigorous, transparent, value driven process with an understanding of the problem and the effects of potential management actions on stakeholder values, as well as how key uncertainties can affect the decision (Robinson et al. 2016).

We found northeastern states that developed a DMP were largely successful at including most essential elements of an SDM framework. Based on Artelle et al. 2018 and our findings, deer are largely being managed following the principles of the NAMWC. However, the criterion that needed the most improvement were evaluating consequences, tradeoffs, and conducting an external review. Outlining a process to evaluate consequences for alternative actions and tradeoffs among objectives was absent from all states. These are important parts of the decision process, as they connect empirical data or expert knowledge to evaluate how decisions affect current or future system dynamics which can be achieved using statistical models (Gerber et al. 2018). One possibility that management agencies could consider when incorporating tradeoffs would be to weight their objectives. For example, objective A could be given a weight of 80% and objective B could be given a weight of 20%. This would imply that objective A is four times more important than objective B. The weights could be considered in the context of the number of stakeholders that would benefit. However, this could be controversial because some stakeholders' objectives will be weighed less than others. A solution could be to evaluate the sensitivity in the optimal action by evaluating all possible combinations of weights (Gerber et al. 2018); it may be that the acceptable weights for each objective by stakeholder leads to the same optimal action.

While we assumed internal reviews were already being completed by all states, internal and external reviews are not equal. An internal review performed by agency employees or the person(s) drafting the plan may have inherent biases, as the reviewers may have drafted or assisted in drafting the plan. However, they are likely to have a good understanding of the agency structure, social, economic, and ecological issues, and how the DMP works operationally. An external reviewer may have fewer biases as they can more easily remain independent and objective. Although, it is less likely that they have "insider" knowledge on how the agency works or the local implication of a plan's strategy. And while they may provide pertinent and science driven comments, suggestions from reviewers may be infeasible to enact given the structure of the government organization. We determined DE, MD and PA were the only DMPs that stated they went through an external review process and appeared valid. Delaware's external review was conducted by Dr. Jacob L. Bowman, Associate Professor with the University of Delaware. Maryland's external review was conducted by the Wildlife Advisory Council, the deer plan stakeholder group, deer experts external to MD, and an outside professional. Pennsylvania had components of their plan externally reviewed, such as the performance of its deer harvest estimating procedures. This evaluation was submitted to a scientific journal for an independent, scientific review by professional biologists and statisticians. There could be several reasons for why states did not conduct an external review of their DMP, such as time or funding constraints. We encourage agencies to strive for both internal and external reviews on a consistent basis (e.g., when DMPs are updated) given their importance to the public to whom agencies are managing on their behalf.

We also found that no state logically linked criterion all the time. The link between criterion is needed as it provides support for why each criterion (monitoring strategies or actions) are present. This may be because of our interpretation of the language used in the management plans, or because the authors did not adequately present all the links. One solution that would ensure all criterion are logically linked together is to adopt a formal SDM framework and language. This would ensure each goal has the necessary criterion, completing the SDM process. These points of clarity regarding linked actions and goals could be easily uncovered through a thorough, external review.

In evaluating a state's transparency, an agency could still be transparent without a DMP by publishing its information in an alternative way, but for this review, failing to have a DMP resulted in being categorized as not transparent. The four states that did not have an available DMP do provide annual summaries (CT, MA, and RI) or an information document (NJ), but do not include the same information as a DMP. It should be noted that if states were not transparent in their DMP, criterion that was not detected may be available in guidance documents that were not publicly available. Future avenues of research to consider may include bringing together working groups of deer project leaders from all states to discover all documentation more fully.

Artelle et al. (2018) stated that 60% of management systems contained fewer than half of the four hallmarks (measurable objectives, evidence, transparency, and independent review). Thus, we might expect similar results for northeastern states deer management plans to include less than half of the hallmarks. However, we found that all northeastern states with a DMP did in fact have at least half of the four hallmarks present, and three states (DE, MD, and PA) had all four hallmarks present. As found by Artelle et al. (2018), big game species often had more than half of the hallmarks detected, which aligned with our results.

There are important differences to consider between our results and those of Artelle et al. (2018). Foremost is that Artelle et al. (2018) did not separate states that did and did not have a management plan. The lack of a management plan reduces the ability to determine the presence of hallmarks. It would be interesting to consider what percentage of systems with a management plan, not "publicly available management information" (what Artelle et al. (2018) evaluated for), had at least half of the hallmarks. If all states with a DMP had the hallmarks present in our review and the states without a DMP were scored as if they were absent, it may explain the finding that big game species were more likely to achieve the hallmarks. Whereas species without management plans may have the hallmarks, but they have yet to be fully outlined in a guidance document, as is possible for the states without DMPs, such as MA, CT, and RI.

Another difference between our results and Artelle et al. (2018), was that we identified criterion in DMPs that were missed in their surveys. For example, Artelle et al.

(2018) stated NY did not provide measurable objective. Yet, we identified measurable objectives in their DMP, e.g., “improve hunter access to public and private lands” and “increase deer harvest in areas with generally overabundant deer by establishing Deer Management Focus Areas by regulation with intensified use of traditional hunting.” In another instance, Artelle et al. (2018) stated MD did not provide measurable objectives. However, our review concluded that MD did in fact provide measurable objectives, e.g., “assist community groups or other organizations in managing specific deer populations and provide staff support to accomplish shared goals when appropriate.” Artelle et al. (2018) may have overlooked some of these objectives as they do not have specific, measurable strategies, but this does not mean that are not measurable. For instance, in MD, the agency could track how many community group meetings are held to assist in meeting deer management goals. In essence, the measurable objective in this case is something the state wants to achieve that will help them attain their goal, whereas the monitoring is the information used to track the objective.

One reason for states not completing or publishing a DMP is that they may be able to maintain more flexibility in how they manage, as they are not bound by a management plan. However, as stated previously, a repercussion is that stakeholders are unclear how management is being conducted and whether agencies are meeting their goals. Stakeholders are left to their own personal experience and thoughts to determine how populations or species are faring. It should be noted that interactions between stakeholders and the species may not come solely from the species but also with the habitats species use (e.g., urban setting and forests). More importantly, many of the negative interactions may be perceived threats or problems that likely will never be realized. A proper management plan states a species status, desired status, and how the species is going to achieve that status. This can help stakeholders understand the full considerations required to assess a species' status.

MANAGEMENT IMPLICATIONS

Largely, we found the NAMWC to be a cornerstone to the management of deer in the northeastern United States. Management plans are important guidance documents for agencies and for the public to understand how resources are being managed. However, improvements can always be made. This study will directly aid in the development of Rhode Island's Deer Management Plan that will better the management of deer through time in the state for sustainable use by hunters and to minimize conflicts where they occur. It will also allow and prompt increased transparency by RI's resource agency as they aim to create a scientifically driven management process that includes stakeholder involvement. In addition to individual states using our findings to refine their own management plan, it may also assist states in developing guidance for regional management and other species-specific plans (e.g., wild turkey, black bear, trout). When goals, objectives and data align between states, the possibility to combine data to increase rigor could benefit many state agencies. This could especially benefit disease management, specifically in cases where disease transmission between states could occur (e.g., chronic wasting disease [CWD]). Data sharing could also increase disease management response time and efforts leading to a more efficient and effective response.

More broadly, this work will hopefully guide agencies or regional organizations outside the northeast to evaluate their management process or management plan (if present) to better manage a species. A current evaluation of management and development of a species-specific management plan using the SDM process will likely result in better science driven, species management leading to species persistence through time. In addition, if the species is used in some way (consumptive or non-consumptive) by stakeholders, this will ensure all stakeholders voices are heard and the species will achieve short- and long-term objectives.

ACKNOWLEDGMENTS

We thank hunters and sport shooters who provide funds through the Pittman Robertson Act through an excise tax that places on firearms, ammunition, and other hunting equipment. This allows the United States Fish and Wildlife Service the capability to provide funding for this research. In addition, we thank and the University of Rhode Island for overseeing this project and the Rhode Island Department of Environmental Management, Division of Fish and Wildlife for allocating funds to this project. In addition, thank you to all the anonymous reviewers and the associate editor of this journal.

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Figure 1. Map showing states reviewed for white-tailed deer management plans. States shown above and years of their DMP; Connecticut (N. A.), Delaware (2010-2019), Maine (2017-2027), Maryland (209-2018), Massachusetts (N.A.), New Hampshire (2016-2026), New Jersey (1999), New York (2012-2016), Pennsylvania (2009-2018), Rhode Island (N.A.), and Vermont (2010-2020).

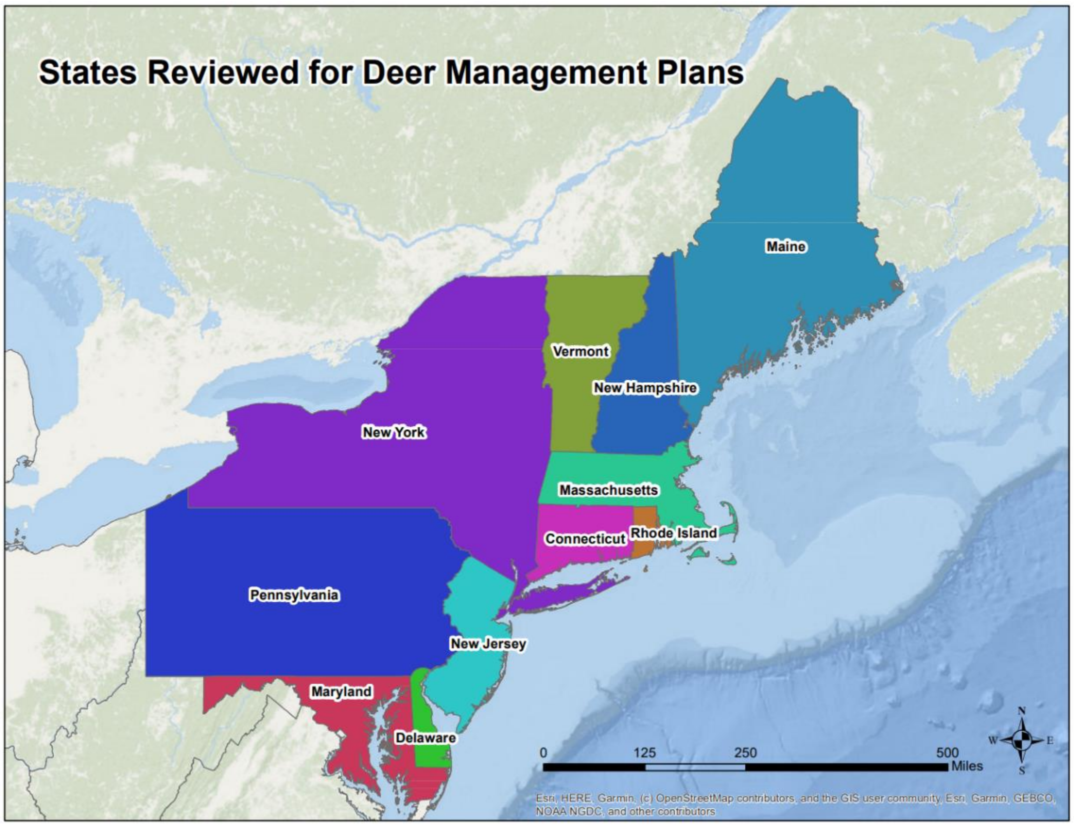

Table 1. Descriptions and examples of structured decision-making criterion used to evaluate Northeast United States white-tailed deer management plans.

Criteria	Description	Example
Goal	An overarching desire that addresses something of importance.	Maintain a healthy deer population
Measurable objective	Quantifiable information that provides a benchmark or threshold to determine if the goal is being met.	Maintain deer reproduction at or above 1.5 embryos per adult doe.
Monitoring	The process by which empirical information is gathered from a system to estimate measurable objectives.	Does the DMP have an effective and reliable technique to monitor embryos per adult female? Inspect road killed adult does for fetuses
Actions/alternative actions	Feasible actions that are believed to impact the system to meet an objective.	Maintain harvest regulations, increase the season length for harvesting deer by shotgun, or purchase land to increase hunter access to lands
Consequences	A description of the expected outcomes from making a specific decision.	To increase embryos per female should we alter harvest or increase habitat quality? Explain the feasibility, expected outcomes and what other unintended consequences each action may have.
Tradeoffs	Actions may be beneficial in meeting some objectives, while worse for other objectives.	Increasing deer density will raise hunter satisfaction but may lower forest health.
Transparency	The ability for the public (stakeholders) to review and understand the decision process.	The state has an up-to-date deer management plan that is publicly available.
Review	A third-party (independent or external) review to evaluate a management plan.	The state requests a university to review the DMP for scientific rigor

Table 2. Summary of whether evaluated Northeast United States white-tailed deer management plans (DMP) had fundamental elements of a structured decision-making framework.

States	Goals	Measurable Objectives	Monitoring	Actions	Consequences	Tradeoffs	Transparent	Review
Connecticut*	**Yes**	No	No	No	No	No	No	No
Delaware	**Yes**	**Yes**	**Yes**	**Yes**	No	No	**Yes**	**Yes**
Maine	**Yes**	**Yes**	**Yes**	**Yes**	No	No	**Yes**	No
Maryland	**Yes**	**Yes**	**Yes**	**Yes**	No	No	**Yes**	**Yes**
Massachusetts*	No	No	No	No	No	No	No	No
New Hampshire	**Yes**	**Yes**	**Yes**	**Yes**	No	No	**Yes**	No
New Jersey*	**Yes**	**Yes**	**Yes**	**Yes**	No	No	No	No
New York	**Yes**	**Yes**	**Yes**	**Yes**	No	No	**Yes**	No
Pennsylvania	**Yes**	**Yes**	**Yes**	**Yes**	No	No	**Yes**	**Yes**
Rhode Island*	**Yes**	No	No	No	No	No	No	No
Vermont	**Yes**	**Yes**	**Yes**	**Yes**	No	No	**Yes**	No

*DMP not available

Table 3. Evaluation of Northeast United States white-tailed deer management plans (DMP) in terms of structured decision-making criterion: goals, measurable objectives and a review was present and if states were transparent.

States	Goals present	Measurable Objectives present	Measurable Objectives that directly relate to Goals	Measurable Objective with at least one directly related monitoring strategy	Measurable Objective with at least one action	Alternative Actions per Measurable Objective	Consequences considered	Tradeoffs considered	Transparent (Publicly available DMP)	External review
Connecticut	**Yes**	No	0%	0%	0%	0%	No	No	No	No
Delaware	**Yes**	**Yes**	100%	97%	97%	74%	No	No	**Yes**	**Yes**
Maine	**Yes**	**Yes**	80%	80%	93%	60%	No	No	**Yes**	No
Maryland	**Yes**	**Yes**	100%	96%	100%	79%	No	No	**Yes**	**Yes**
Massachusetts	No	No	0%	0%	0%	0%	No	No	No	No
New Hampshire	**Yes**	**Yes**	67%	17%	17%	17%	No	No	**Yes**	No
New Jersey	**Yes**	**Yes**	100%	100%	67%	67%	No	No	No	No
New York	**Yes**	**Yes**	100%	95%	100%	67%	No	No	**Yes**	No
Pennsylvania	**Yes**	**Yes**	100%	100%	84%	84%	No	No	**Yes**	**Yes**
Rhode Island	**Yes**	No	0%	0%	0%	0%	No	No	No	No
Vermont	**Yes**	**Yes**	100%	89%	89%	33%	No	No	**Yes**	No
Average	91%	73%	75%	67%	65%	48%	0%	0%	64%	27%
Average (excluding CT, MA, NJ, and RI)	100%	100%	92%	82%	83%	59%	0%	0%	100%	43%

Table 4. Results from Artelle et al. (2018) in terms of components per hallmark (measurable objective, evidence, transparency, and review).

States	Measurable Objective	Evidence				Transparency				Review		% of Components present
		Report quantitative information about populations	Report uncertainty in population parameter estimates	Estimate realized hunting rates	Explain technique for setting hunting quotas	Explain how population parameters are estimated	Explain how realized hunting rates are estimated	Provide publicly available management information	Respond to public inquiry	Subject management plans to any review	Subject management plans to external review	
Connecticut	No	**Yes**	No	**Yes**	No	**Yes**	**Yes**	**Yes**	**Yes**	No	No	55%
Delaware	No	**Yes**	No	**Yes**	No	**Yes**	**Yes**	**Yes**	No	**Yes**	**Yes**	64%
Maine	**Yes**	**Yes**	No	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	No	No	73%
Maryland	No	**Yes**	No	**Yes**	No	**Yes**	**Yes**	**Yes**	**Yes**	No	No	55%
Massachusetts	No	No	No	**Yes**	No	No	**Yes**	**Yes**	No	No	No	27%
New Hampshire	**Yes**	**Yes**	No	**Yes**	No	**Yes**	**Yes**	**Yes**	No	**Yes**	No	64%
New Jersey	No	No	No	**Yes**	No	No	No	**Yes**	**Yes**	No	No	27%
New York	No	No	No	**Yes**	No	**Yes**	**Yes**	**Yes**	**Yes**	No	No	45%
Pennsylvania	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	No	91%
Rhode Island	No	**Yes**	No	**Yes**	No	**Yes**	**Yes**	**Yes**	No	No	No	45%
Vermont	**Yes**	**Yes**	**Yes**	**Yes**	No	**Yes**	**Yes**	**Yes**	**Yes**	No	No	73%
% Present	36%	73%	18%	100%	18%	82%	91%	100%	64%	27%	9%	56%
% Present (excluding CT, MA, NJ, RI)	57%	86%	29%	100%	29%	100%	100%	100%	71%	43%	14%	66%

Table 5. A summary of Artelle et al. (2018) data determining if at least one part of each hallmark was present.

States	Measurable Objective	Evidence	Transparency	Review	% of Components present
Connecticut	No	**Yes**	**Yes**	No	50%
Delaware	No	**Yes**	**Yes**	No	50%
Maine	**Yes**	**Yes**	**Yes**	**Yes**	100%
Maryland	No	**Yes**	**Yes**	No	50%
Massachusetts	No	**Yes**	**Yes**	No	50%
New Hampshire	**Yes**	**Yes**	**Yes**	**Yes**	100%
New Jersey	No	**Yes**	**Yes**	No	50%
New York	No	**Yes**	**Yes**	No	50%
Pennsylvania	**Yes**	**Yes**	**Yes**	**Yes**	100%
Rhode Island	No	**Yes**	**Yes**	No	50%
Vermont	**Yes**	**Yes**	**Yes**	No	75%
% Present	36%	100%	100%	27%	66%
% Present (excluding CT, MA, NJ, RI)	57%	100%	100%	43%	75%

APPENDIX A. DEER MANAGEMENT PLANS

Connecticut: N.A.

Delaware: http://www.dnrec.delaware.gov/fw/Hunting/Documents/Deer%20Plan%20-%20FINAL%2005212010.pdf

Maine: https://www.maine.gov/ifw/docs/18-MDIFW-03-Big-Game-Management.pdfhttps://www.maine.gov/ifw/docs/18-MDIFW-03-Big-Game-Management.pdf

Maryland: Maryland's DMP was updated after the research was completed. Here is a link to the most recent DMP. https://dnr.maryland.gov/wildlife/Documents/2020-2034MarylandWTDeerPlan.pdf

Massachusetts: N.A.

New Hampshire: https://www.wildlife.state.nh.us/hunting/documents/game-mgt-plan.pdf

New Jersey: https://www.state.nj.us/dep/fgw/pdf/govdrrpt.pdf

New York: https://www.dec.ny.gov/docs/wildlife_pdf/deerplan2012.pdf

Pennsylvania: https://www.pgc.pa.gov/Wildlife/WildlifeSpecies/White-tailedDeer/Documents/2009-2018%20PGC%20DEER%20MGMT%20PLAN%20-%20FINAL%20VERSION.pdf

Rhode Island: N.A.

Vermont: https://vtfishandwildlife.com/sites/fishandwildlife/files/documents/Learn%20More/Library/REPORTS%20AND%20DOCUMENTS/HUNTING/BIG%20GAME%20MANAGEMENT%20PLAN%20-%202010/BIG%20GAME%20MANAGEMENT%20PLAN-COMPLETE.pdf

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1 April 2022

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Statistical population reconstruction of white-tailed deer in Rhode Island

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Abstract

A key principle of The North America model of Wildlife Conservation is that science is the proper tool for discharging wildlife policy. Using science to understand population abundances and dynamics is especially critical in managing harvested wildlife. Tracking population changes allows resource managers to adapt regulations to ensure populations are maintained. In Rhode Island, USA white-tailed deer (*Odocoileus virginianus*) are annually harvested, but there is no systematic annual population estimation to track

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changes, which may put the population and forest ecosystem at risk. Our objective was to evaluate the utility of statistical population reconstruction (SPR) to monitor white-tailed deer in Rhode Island by estimating annual deer abundance, harvest probabilities, and recruitment for males and females, separately. To do so, we used age-at-harvest data collected from hunter harvested deer from state operated check stations (2011-2020) and online/phone reporting, hunter effort derived from annually reported deer harvest, and natural mortality probabilities from the literature. Without a reliable measure of reporting rate, we considered three possible reporting rates (25%, 50%, and 75%). As not all deer reported were aged, we used random forest models to predict the age of 19,277 deer reported via mail-in/online/phone using age, weight, sex and antler beam measurements of deer checked by staff. The out-of-sample prediction accuracy was between 85-99% with most over 90%. We estimated male abundance with a 75% reporting rate to range from a low of 9,503 (SE, 1,291) in 2017 to a high of 15,767 (SE, 2,183) in 2011, with the most current estimate at 10,054 (SE, 1,325) in 2020. Using a 50% reporting rate, male abundances were higher, ranging from a low of 13,730 (SE, 1,753) in 2017 to a high of 22,271 (SE, 2,912) in 2011, with the most current estimate at 14,031 (SE, 1,745) in 2020. Using a 25% reporting rate, male abundances were the lowest, ranging from a low of 9,310 (SE, 362) in 2015 to a high of 10,766 (SE, 369) in 2019, with the most current estimate at 10,525 (SE, 362) in 2020. Depending on the reporting rate, the male population between 2011-2020 was estimated to be either slightly increasing or decreasing. The SPR failed to produce realistic estimates for females with estimated harvest probabilities near or at zero, which inflated abundance estimates to unreasonable values (>1 million). Overall, SPR appears to be a useful methodology for monitoring deer populations in Rhode Island. However, to rely on it as part of management policy will require several improvements over the current implementation. Foremost, it is recommended that hunter effort, reporting rate and survival probability are determined in Rhode Island via additional research, such as hunter surveys and survival studies.

Key Words: deer harvest, PopRecon, Rhode Island, statistical population reconstruction, white tailed deer.

Monitoring harvested wildlife populations is a cornerstone to making informed and logical harvest policy (Williams et al. 2002, Nichole and Williams, 2006, Conroy and Peterson, 2013). Tracking population abundance through time is especially useful to evaluate population trends and the impact of policy changes (Clawson et al. 2017 and Decker et al. 2014). However, obtaining the data to track populations accurately is often costly and time consuming (Clawson et al. 2017 and White et al. 1989). Natural resource management agencies often do not have the resources to conduct state-wide annual empirical population studies, such as large-scale mark-recapture studies (Clawson 2015). However, agencies do regularly collect biological data on hunter harvested animals, which includes the animals age, sex, and health. This information is useful for several reasons, including disease monitoring, age and sex ratio estimates, and herd health monitoring (Cretois et al. 2020, Norton et al. 2021). Tracking the age-at-harvest of a population over time may also be useful in estimating annual population abundance to monitor population changes in a cost-effective manner. One-way agencies can estimate statewide populations with only a fraction of the total age-at-harvest data is through statistical modeling, specifically statistical population reconstruction (SPR), which accounts for the probability a harvest is reported.

The SPR model (Gove et al. 2002) is an integrated data model that combines age-at-harvest information and additional data to estimate population demographics. By itself, the age-at-harvest matrix composed of annual harvest and age classes cannot be used solely, as model parameters are not identifiable (Gove et al. 2002). SPR requires hunter effort, reporting rate, and auxiliary information on abundance, survival, or harvest rates to allow parameters to be identifiable and model the transitions in expected harvest counts from age class i in year j to the subsequent harvest count in age class $i + 1$ in year $j + 1$, etc. For many harvested wildlife species, empirical estimates of these auxiliary parameters exist from additional studies. As such, SPR may be a cost-effective strategy to monitor wildlife population dynamics using commonly collected data from natural resource management agencies. Despite its potential and the many statistical developments made using SPR (Broms et al. 2010, Clawson et al. 2013 and Gove et al. 2002), applications still appear to be uncommon (but see Hatter et al. 2018 and Howard et al. 2018). The SPR model may be especially useful for monitoring populations of cervids, such as white-tailed deer (*Odocoileus virginianus*; hereafter, ‘deer') because state agencies routinely collect age-at-harvest data and other related information. More so, white-tailed deer are commonly a species of interest for stakeholders, residents and state resource agencies.

Rhode Island’s deer population impacts many aspects of the state’s culture, economy, public safety, and environment. Often, deer population levels can have positive and negative effects simultaneously, which require a balancing act by natural resource agencies to maintain an appropriate population level. Deer hunting has occurred in Rhode Island since before European colonization and is a long-lasting tradition today. Motivations may be different now, but people still participate for recreation while providing wild meat for family and friends. Hunting also provides a source of revenue to the local economy as hunters purchase equipment, fuel, food, and travel expenditures. In addition to local revenue, hunting provides state and federal funding through license and permit sales that support wildlife conservation of deer, other game and non-game species in Rhode Island. Without deer, deer hunters and their benefits would cease to exist (Rhode Island Department of Environmental Management, Division of Fish and Wildlife, n.d.).

While hunters may see higher deer populations positively, for others it is a cause for concern. Deer cause vehicle collisions, which leads to property damage, personal injury, and even death. Overabundant deer populations can cause the forest composition to change from browsing pressure that could have long lasting effects on other native wildlife and plant species (Rooney 2001). Also, wildlife diseases, such as Chronic Wasting Disease (CWD), can have major population impacts (Edmunds et al. 2016) and need to be surveyed for to maintain a healthy and sustainable population. It is critical that the deer population is monitored accurately within budgetary constraints so agencies can detect changes in the populations that may cause significant negative effects to Rhode Island's culture, economy, public safety and environment. Monitoring the population provides agencies the information needed to make regulatory changes to reduce negative effects and to ensure the species persists through time.

There are no recent population estimates of deer in Rhode Island despite an annual harvest, depredation permits issued, deer vehicle collisions, and registered complaints of too many and too few deer (Rhode Island Department of Environmental Management, Division of Fish and Wildlife, 2021). Here, we investigate the utility of the SPR model to estimate deer abundance and harvest probabilities in Rhode Island. This information will be used to better track and monitor the deer population and will provide scientific evidence required to make management decisions, such as changes in bag limits or season lengths. Our specific objectives are to provide the first empirical estimates of white-tailed deer abundance in Rhode Island for male and females using SPR, and to evaluate the sensitivity of parameter estimates to reporting rate, quantity of data, and natural survival probability.

Study Area

Our study area is the entire state of Rhode Island, United States (Figure 1), located in the New England region. It is the smallest state by area at 1,214 sq. miles (3,144 sq. km), but is the second-most densely populated state at 1,061.4 people/sq. mile (2020 U.S. Census Bureau). Rhode Island has an eastern broadleaf forest with most hardwood forests consisting of mainly oaks and maples with smaller patches of softwoods across the state (Butler et al. 2012) with contiguous patches of forest shared between Connecticut and Massachusetts. Rhode Island consists of approximately 750 sq. miles (1,207 sq. km) of deer habitat which excludes all water bodies and highly developed areas (RIGIS Forest Habitat 2010, 2012). Rhode Island is divided into four deer management zones (Figure 1). Deer management zone 1 consists of the coastal and more urban areas of the state. Deer management zone 2 is the western and eastern part of the state with larger tracts of forest and less urban areas. Deer management zone 3 is Patience and Prudence Islands located within the Narraganset Bay. Deer management zone 4 is Block Island (the entirety of the town of New Shoreham), an island located approximately 10 miles off the southern coast. Here, we use data collected from all deer management zones.

Methods

Statistical Population Reconstruction using Pop Recon 2.0

We fit SPR models using the software PopRecon 2.0 (Clawson et al. 2017), which was developed to be a user-friendly way to analyze single sex of either full-age or pooled-age at harvest data. For all analyses, we chose to arrange data into three age classes for each sex and pool ages of the last group: fawn (age=0.5), yearling (age=1.5), and adult (age=≥2.5). In the following subsections, we describe the necessary data and considerations needed to fit SPR models for harvested deer in Rhode Island.

Harvest data

Age-at-harvest is the main source of data used in SPR models. Simply, it is the count of harvested individuals in each age class in each year. However, data collection is not created equally for all deer harvest across time and space. In Rhode Island, harvest data is collected via mandatory in-person check stations, online/phone reports, or mail-in harvest report cards prior to 2018. Mandatory check stations were operated by Division of Fish and Wildlife staff during the first four days (Saturday – Tuesday) of the muzzleloader deer season, starting on the first Saturday of November and require all deer harvested in zones 1 and 2 (regardless of method of take) to be physically checked at a deer check station. The data collected at deer check stations includes age, sex, antler points, weight, male antler beam measurements (only yearlings prior to 2017 as an indicator of overall health), and disease surveillance. The age of individual deer was determined by analyzing deer tooth wear and replacement (Gee et al. 2002) by Division of Fish and Wildlife staff, as well as by analyzing front incisors via cementum analysis conducted by Matson Lab (Gilbert 1996) during 2017, 2018, and 2019. Deer harvested in zones 3 and 4 are not required to be brought to a deer check station. These deer and all other deer harvested outside of the check station period across all zones are reported online, over the phone, or with mail-in report cards (prior to 2018). Harvest data from 2011-2020 (Figure 2) were combined for all deer management zones using three age classes (0.5, 1.5, ≥2.5; Table 1). To reconcile missing age-at-harvest information from deer not checked by biologists, predictive modeling was used.

Predictive modeling of age-at-harvest

The SPR model requires all harvested individuals to be classified into age classes. Harvested deer reported via the online reporting system or the mail-in harvest report cards were used to obtain hunter harvested deer's sex, estimated weight, antler points and in some instances, age class (fawn/adult). However, much of these data lack age classifications, or if they do not, their accuracy may be suspect as hunters can often misidentify larger fawns and smaller yearling females as each other. From 2011 to 2020, there were 19,277 (males: 9,589 and females: 9,688) deer reported by hunters without age classification (Table 2). To include harvest samples in the SPR modeling, we required a predictive model to assign each deer to an age class.

We fit predictive models trained using the ages of deer determined by biologists observed at check stations and those based on tooth cementum analysis. In comparing ages based on cementum analysis and those determined by biologists, we found they largely agreed when classifying deer into three age groups (Table 3). For each deer reported online or over the phone, we considered all available biological data collected as potential covariates to predict age class. These variables included, sex, antler beam, estimated weight, and hunter age classification. We fit these data using Random Forest Models optimized for within-sample predictive accuracy (Cutler et al. 2007) using the R package “randomForest”. We evaluated out-of-sample predictive accuracy by withholding 10% of the check station data and calculating a multi-class value of Area Under the Curve (AUC; Hand and Till 2001) using the R package “pROC” (Robin et al. 2011); a multi-class AUC value of 1 indicates perfect prediction.

Reporting rate

The reporting rate is the percentage of deer harvested that were reported relative to all deer harvested (reported and not reported). Rhode Island has not completed a study to estimate harvest reporting rate among Rhode Island hunters, but we assume not at all deer harvested within Rhode Island are reported (physically checked or reported online/over the phone). Therefore, we looked to recent literature to determine a reporting rate. A hunter survey issued by the Connecticut Department of Energy and Environmental Protection (CT DEEP) estimated reporting rate at 39%. This survey determined that hunters reported almost 2.5 times more deer in the hunter survey than were reported using the legal reporting method (archery report kill card; Kilpatrick et al. 2005). A similar study was completed in Pennsylvania where harvest reporting rate varied from 36-60% depending on year, hunting season, and type of deer harvested (Rosenberry et al. 2004). Reporting rate has also been estimated via hunter surveys in Virginia, where they found it to range from 60-80% from 1993 through 2004. In 2004, they mandated electronic checking and noticed an increased reporting rate at ~80% ranging from 70-90% (W. Knox, Virginia Department of Wildlife Resources, personal communication). In the absence of a Rhode Island specific reporting rate, we assume Rhode Island's reporting rate is 50%, similar to Connecticut's reporting rate. This is due to its close geographic location and similar harvest reporting methods.

Sensitivity to changes in reporting rates, data quantity, and age classes

Since Rhode Island does not have an empirically based hunter reporting rate and reporting rates from other states are highly variable, we thus cannot defend a single reporting rate to be used. We chose to evaluate the sensitivity of our estimated parameters to changes in harvest reporting rate by considering two additional scenarios of harvest reporting, 25%and 75% of the male (Table 4) and female (Table 5) harvest. We expect as reporting rate decreases, abundance increases (if x is the number of observed aged individuals, abundance (N) for an age class in a given year is $N = x/(r*h)$ where r is the reporting rate and h is the harvest probability).

We also evaluated the sensitivity of our model to data quantity (Skalski et al. 2012). This was completed by shortening the time-series of age-at-harvest to 2016-2020 (Table 6) and comparing estimates with those using the full 10-year period. It is recommended to conduct a data deletion technique to evaluate the sensitivity of model results to data removal (Skalski et al. 2012). If model results change substantially due to the removal of a few years of data, results using the full data should be viewed with caution

(Skalski et al. 2012). Lastly, we examined the sensitivity of model results to the number of age classes used (two classes vs three classes). Both male and female SPR models were fit using two and three age classes.

Additional forms of deer harvest

In addition to unreported hunter harvested deer, hunters inevitably wound deer that are never recovered, of which some succumb to their injuries. Wounding loss is defined as deer fatally wounded by hunters and not recovered. Rhode Island does not have empirically based wounding loss rates, therefore we researched literature to determine wounding loss. Wounding loss rates have been estimated at approximately 20% of the total harvest, but could reach as high as 27% (Stormer et al. 1979). In a more recent study during the 1989–2006 hunting seasons, 104 bowhunters failed to recover 162 of 908 deer hit by arrows or crossbow bolts, corresponding to an 18% wounding rate (Pederson 2008). This wounding rate considers all deer hit and not recovered, which will be higher than the wounding loss as some wounded deer survive. Wounding loss is something that is likely out of the Division of Fish and Wildlife's control, but it needs to be estimated to correctly estimate total harvest. Wounding loss rate was assumed to be 15%, which is slightly lower than previous research by Pederson (2008). It was reasoned that wounding loss should be lower because Rhode Island's harvest is dominated by more accurate harvest methods, such as muzzleloader and crossbow which often prevent hunters from shooting at running deer, while shotgun hunting with buckshot is prohibited which was used primarily to shoot at running deer. Also, hunting with parties greater than five is prohibited which is typically conducted to move deer toward hunters which results in hunters shooting at running deer. Wounding loss was incorporated into our models by including it in the total harvest by multiplying 0.15 by the sum of reported and non-reported harvest.

Another form of harvest or additional “take” includes non-seasonal take and road-killed deer. Take, refers to the removal of deer from the landscape regardless of purpose or intention. Non-seasonal take refers to deer that are harvested under deer damage permits or scientific collectors permits. This includes but is not limited to deer taken by farmers suffering from crop loss or airports removing deer causing safety concerns. All non-seasonal take is assumed to be reported as this is highly regulated. Non-seasonal take is approximately 10-50 per sex annually and occurs throughout the year so it is being excluded from the data analysis.

Deer are also regularly struck by vehicles and succumb to their injuries. Many of these deer are reported to the RIDEM Division of Law Enforcement which issues the Division of Fish and Wildlife an annual report. We assume not all roadkill mortality is detected or reported, as not all accidents may be reported and deer that do not succumb to their injuries near the road may succumb to them out of sight from the roadway with a lower probability of being detected and reported. Some data suggests that female roadkill is higher than male roadkill at 2:1, respectively (Allen and McCullough 1976); however, male roadkill does increase significantly during the breeding season. Additional research showed a less evident split where 58% of roadkill were female (Bellis and Graves 1971). Rhode Island roadkill data from 2000-2015 (excluding 2009 and 2010) were missing 45% of sex identification. From 2016-2020 all roadkill sex is unknown. Therefore, road-killed deer were not added to the abundance estimates as the sex ratio for the majority of road-killed deer is unknown.

Hunter Effort

Hunter effort is the amount of effort hunters spend attempting to harvest deer in a single year. This is a fundamental component of SPR models to estimate annual harvest probabilities (Gove et al. 2002), which is done using a catch-effort parameterization (Seber 1982), where annual harvest probability in year t (p_t) is a function of a coefficient of vulnerability (c) and hunter effort (f_i), such that $p_t = 1 - e^{-c*f_t}$. Hunter days have been used as hunter effort broadly across the literature (Clawson et al. 2017), but in the absence of it, we determined annual hunter effort for each sex by using the annual reported harvest (2011-2020), standardized to the average over the same time (Table 1).

Survival and harvest probability

Survival probability is defined as the probability of surviving to the next year excluding hunting mortality. In the absence of Rhode Island specific survival data, these data were determined by a literature review for all age classes (Table 7). Fawn survival has been estimated at 14% - 87% across eastern North America (Dion 2018), with the closest to Rhode Island and most recent from Connecticut in 2018 at 0.36 for a period of 90 days (Kilburn 2018). Once fawns reach 90 days, there is little to no reduction in annual survival based on studies of white-tailed deer fawn survival from the eastern United States and Canada from 1996-2017 (Dion 2018). With that, and the large range in estimated fawn survival (Dion 2018), we used fawn survival of 0.36 (SE, 0.1) from Connecticut as they are the closest to Rhode Island and likely have similar factors influencing survival, such as the presence of coyotes and bobcats. However, Connecticut does have black bears, but it appears they are not reducing survival greater than other predators or natural causes (Dion et al. 2020).

We found adult survival varied widely across the literature. In Michigan's upper peninsula, survival rates were estimated at 0.81 (SE, 0.09) for adult females, 1.0 (SE, 0.05) for adult males and yearling females, and 0.84 (SE, 0.07) for yearling males (Van Deelen et al. 1997). Additional research in Oklahoma estimated adult male survival at 0.86 (SE, 0.07) (Ditchkoff et al. 2001) and in New Brunswick, adults were estimated at 0.66 for males and 0.89 for females (Whitlaw 1998). We chose to use survival of 0.90 (SE, 0.1) for yearlings and adults males and 0.98 (SE, 0.001) for yearlings and adults females (Table 7). These values were chosen as RI has milder winter conditions than Michigan and New Brunswick.

When entering survival into PopRecon 2.0, it requires the user to select the last distinct age class to determine how survival probability varies by each age class. We chose to use fawn specific survival probabilities (age class 0.5) and yearling and adult specific survival probabilities (age class 1.5 and ≥2.5) for males and females. Therefore, the last distinct age for survival probability was set to one in PopRecon 2.0. The last distinct age for harvest probability was set to one for males and zero for females in PopRecon 2.0.

Initial values

Model parameters, survival and harvest probability, are estimated from Pop-Recon 2.0, but require initial values. These values are simply to allow the algorithm to begin the maximization optimization of the model's log-likelihood. We used initial values for male survival probability at 0.85 - 0.95 and females at 0.98 – 0.99. We used initial values for male harvest probability at 0.1 - 0.5 and females at 0.1 - 0.3. These estimates were chosen based off literature (Ditchkoff et al. 2001, Van Deelen et al. 1997, Whitlaw et al. 1998), expert knowledge and what would allow the models to fit the data.

Results

Male and female harvest data and predictive modeling of age-at-harvest

The harvest data consists of the total reported deer harvest (n=21,669) within Rhode Island from 2011-2020 (Table 1). Of all deer reported, biological data (sex, precise weight, age-at-harvest, antler points and antler beam measurements) were collected by Division of Fish and Wildlife staff at check stations for 2,392 deer (11% of all harvested deer) and used in developing predictive models that would estimate age-at-harvest when age was not collected by Division of Fish and Wildlife staff for 19,277 deer (89%). Of the 2,392 deer checked and aged, n=1,625 (68%) were males and n=767 (32%) were females (Table 2). Of the 482 deer that were aged by staff and had cementum analysis completed, 404 (84%) were aged into the three age classes correctly (Table 3). When classifying them into two age classes, 474 (98%) were aged correctly.

In total, we used 17 different trained predictive models for each combination of types of available data for each deer. The out-of-sample prediction accuracy was between

85% and 99%; most were predicted with an accuracy of over 90%. Table 8 is a concordance table between predictions (0.5, 1.5, 2.5) and staff member classification. They largely agree, but there is still error.

Hunter effort for male and female harvest

Hunter effort from male data ranged from 0.82 – 1.19 while female data resulted in a range from 0.87 – 1.25 (Table 1). Hunter effort was also calculated from 2016-2020 to determine the sensitivity of the model with minimal data from 2016-2020. We found hunter effort for males to range from 0.80 – 1.2 and for females from 0.90 – 1.1 (Table 6).

Male statistical population reconstruction using Pop Recon 2.0

Using a 50% reporting rate, male abundances ranged from a low of 13,730 (SE, 1,753) in 2017 to a high of 22,271 (SE, 2,912) in 2011, with the most current estimate at 14,031 (SE, 1,745) in 2020 (Table 9). Abundance estimates produced with 25% and 75% reporting rates are discussed below when evaluating sensitivity to reporting rates.

Female statistical population reconstruction using Pop Recon 2.0

We were unable to produce realistic estimates using the female data for many of our scenarios. We found that female total abundance estimates varied greatly through the 10-year span across 75% and 50% reporting rates (Table 10). The 25% reporting rate did not estimate total abundance as the model failed to fit. Estimates between 75% and 50% reporting rates ranged from a low total abundance estimate of 1,729 to a high of 230,777 with an outlier of 536,014,025,990. Female harvest probability estimates also varied from a low of 0.006 to a high of 0.971 through the 10-year span across both reporting rates for all ages. Female survival probability estimates did not vary across the two reporting rates as age=0.5 was 0.358 or 0.36 and age=1+ was 0.98 for both reporting rates. The 75% reporting rate estimated total abundance from 225,285 to 230,777 with standard error estimates greater than abundance estimates. The 50% reporting rate estimated total abundance from 1,729 in year 10 to 536,014,025,990 in year 1. However, excluding year 1, the highest total abundance estimate was 4,474 in year 3. It's unclear why year 1 had significantly higher abundance estimates than the following years with similar harvest probabilities and the same survival probabilities. Both the reported harvest and the hunter effort was neither the highest nor lowest amongst all years.

Sensitivity to changes in male data quantity

In the male harvest, when 2011-2015 data was removed and if reporting rate was 75% or 50%, total abundance estimates were ~10% higher on average across all estimates (Table 11). This was mainly impacted by the level of recruitment (age=0.5) increasing ~1,000 - 4,000 across all years (~31%). Whereas for age=1.5 and ages=≥2.5, abundance estimates decreased in almost every instance by ~5%, ranging from a 1% increase to a 10% decrease. The harvest probability slightly decreased for age=0.5, from a range of 0.033 - 0.046 to a range of 0.025 – 0.038. In ages=1+ they ranged from 0.208 – 0.291 and increased slightly to 0.217 – 0.314. Survival probabilities had no significant change. When the 5 years of data was removed and there was a 25% reporting rate, there was a significant change in total abundance estimates as they increased from 9,467 – 10,766 to 29,734

– 46,953. Age=0.5 had the largest increase ranging from 9,373 – 30,337 (1,820% increase on average). The harvest probability range and survival probability decreased from 0.275 – 0.362 to 0.026 – 0.039 and 0.878 to 0.359 respectively.

Sensitivity to changes in reporting rates

We found SPR estimates were sensitive to the assumed reporting rate (25%, 50% and 75%; Figure 3). We found male total annual abundance estimates varied considerably from 2011-2020 across all reporting rates, ranging from a low total abundance estimate of 9,310 to a high of 22,271. Male harvest probability estimates also varied with age=0.5 ranging from a low of 0.032 to a high of 0.362 and age = ≥1.5 ranging from a low of 0.201 to a high of 0.304. Male survival probability estimates varied across all reporting rates; age=0.5 ranged from a low of 0.348 to a high of 0.878 and age=1+ ranged from a low of 0.854 to a high of 0.901.

We found little variability in harvest and survival probabilities between the 75% and 50% reporting rates. For the harvest probability, age = 0.5 was only different by 7.7% and age = ≥1.5 was only different by 1.5% (Table 9). This was calculated by taking the difference of the averages of the 75% and 50% harvest probability. The same was completed for survival probabilities, where age = 0.5 was different by 1.1% and age = ≥1.5 was different by 1.4% (Table 9). As such, abundance estimates did not differ. The 75% reporting rate estimated total abundance from 9,503 to 15,767 and the 50% reporting rate estimated total abundance from 13,730 to 22,271 (Table 9). The 50% reporting rate abundance estimates increase from 75% reporting rate as expected. However, the 25% reporting rate abundance estimates did not increase as expected. The total abundance estimates with a 25% reporting rate ranged from 9,310 to 10,766. This is likely due to the change in harvest and survival probability estimates. Specifically, the harvest probability estimate increased in age=0.5 from ~0.040 (average between 75% and 50%) to ~0.304, a 650% increase. Alternatively, the survival probability of age=0.5, increased from 0.35 (average of 75% and 50% reporting rate) to 0.88, a 141% increase. The percent increase (650%) in harvest probability superseded the percent increase (141%) in survival probability, which lead to the abundance estimate from the 25% reporting rate being comparable to the abundance estimate at the 75% reporting rate.

The same comparison was attempted with female harvest; however, a comparison could not be completed between 25% reporting rates as models failed to fit. Reporting rates at 75% and 50% produced abundance estimates ranging from 130 million to 245 million, on average a 71,944% increase in the case of a 75% reporting rate and 2,057,561% increase in the case of a 50% reporting rate. The harvest probability was estimated at zero for all three reporting rates with data from 2016 - 2020, compared to 0.007 – 0.006 for 75% reporting rate, and 0.915 – 0.936 for 50% reporting rate with 2011 – 2020 data. Survival probabilities showed little change between the two estimates. Data from 2011-2020 estimated survival probabilities for age=0.5 at 0.36 and age=1+ at 0.98 for both 75% and 50% reporting rates. When only data from 2016-2020 was used at 75% and 50% reporting rate, survival probabilities increased 0.09 to 0.45 for age=0.5 and decreased 0.06 for age=1+ to 0.93.

Evaluating sensitivity to quantity of male and female age classes

When age classes were reduced from three to two, the male SPR model produced estimates for 75% and 25% reporting rates, but failed to fit for 50% reporting rate (Table 12).

Overall, the abundance estimates were much higher, ranging from 138,378 (75% reporting) to 409,908,882 (25% reporting). Both survival probabilities remained constant at 0.36 for age=0.5 and 0.919 and 0.918 for age=1. The reason for the increased abundance estimates is likely due to the harvest probability estimate, as it was zero across all years and ages for both reporting rates. The female SPR model was able to estimate abundance for all reporting rates (Table 13), but total abundance estimates increased from ~227,000 to ~415,000 (~50% increase) for the 75% reporting rate and increased from ~3,000 to ~618,000 (20,500% increase) for 50% reporting rate. The 25% reporting rate estimated total abundance around 6,000,000. Based on both SPR models, when ages classes are reduced to two, harvest probabilities decreased to near zero, and abundance estimates increase significantly.

Discussion

Empirical estimates of annual deer abundance are important to properly manage deer through time as there are many stakeholders affected by the deer population and its impacts. Often, it can be a topic of contention as certain stakeholder groups may want opposing actions and results (Curtis 2020). It is critical to manage deer properly as a sufficient population provides hunters with recreation and a local sustainable food source that they share with family and friends. Deer hunting also creates revenue for wildlife management (including non-game species) through legal regulated hunting license sales. Conversely, overpopulation can have negative forest impacts, damage farmers agriculture, homeowner decorative plants, and increase deer vehicle collisions (Rooney 2001, Curtis 2020). Having a current and local understanding of population demographics will allow for proper management maintaining a healthy self-sustaining population that supports recreational hunting, a wild food source, reduced human-deer conflicts, and generates revenue to fund management of many species across Rhode Island (Oran et al. 2012, RI DEM DFW Annual Report 2020, 2022).

We found that important information for effectively using SPR modeling to monitor the Rhode Island deer population are missing. This included Rhode Island specific reporting rate, wounding rate, and survival probability. As such, it required using estimates from the literature that may not be accurate for Rhode Island. Our results demonstrated how important understanding reporting rates are when estimating population size. For male abundance estimates, when reporting rate differed (25%, 50%, and 75%) in year six, abundance estimates ranged from ~9,500 with 25% reporting, ~20,000 with 50% reporting and ~14,000 with 75% reporting. This ~11,000 range provides a clear need for an accurate reporting rate. Given the size of Rhode Island (1,214 sq. miles), male density estimates range from 7-18 males/sq. mile. Given that the males likely make up less than half of the population, this is expected as current deer density (males and females combined) appears above ideal density of 10-20 deer/sq. mile (DeCalesta 2017).

Importantly, we also found that we were unable to produce reliable abundance estimates of female deer when fitting the full time series of data using SPR models. One major difference between male and female data that may have contributed to this issue is the number of female deer aged by staff. There were over twice as many males aged by staff than females (1,625 compared to 767, respectively, Figure 4). As such, there may have been higher error in predicting female ages than male ages. This issue may have been exacerbated because there are two variables for males that females don't have, antler points and antler beam measurements. Moving forward, it will be important to obtain more data on accurately aged females, as well as additional variables to be used in predictive modeling that are useful to predict female ages.

Another piece of information critical for SPR modeling that could be improved is hunter effort. Foremost is that hunter effort should be derived annually to produce reliable estimates (Rosenberry et al. 2004). A typical and effective way to estimate hunter effort is through hunter effort surveys. Hunter effort surveys have been completed in Rhode Island in the past via questionnaires on harvest report cards asking hunters how long they hunted and how many deer they observed. However, we were unable to use these results as they are biased towards only successful hunters (approximately 25% of all hunters). Future surveys will require collecting information from hunters that are not successful. Other possible methods to track hunter effort are hunting licenses and deer permits sold. License and permits receipts have been hand counted prior to 2018, which resulted in unreliable data. Fortunately, since 2018, a new online licensing system, https://rio.ri.gov/ gives the Division of Fish and Wildlife accurate/reliable license and permit data which will allow them to obtain trends in hunter effort via license and permit sales in the future. However, a caveat that needs to be resolved is separating out those that purchased a license and those who actively hunted in a given year.

In our implementation of SPR modeling, we chose to aggregate all harvest data across the four management zones to ensure there was sufficient data to produce reliable estimates. However, it should be further investigated to see how each zone can be estimated independently as each deer management zone is composed of different land mass, habitat types, human population, and hunting regulations. For example, deer management zone 2 is comprised of approximately 500 square miles of rural, less developed lands and has an antlerless season bag limit of 2 with an annual reported harvest of around 1,200 deer. In contrast, deer management zone 4 is the entirety of Block Island, a 9 square mile island where the deer densities have been estimated upwards of 75 deer/sq. mile and the season bag limit for antlerless deer is unlimited and there is a $150 "bounty" on deer on the island with an annual total reported harvest of around 250 deer. Ideally, population monitoring should be done within each management zone separately to inform future harvest policy.

Fitting integrated data models, such as SPR, is computationally challenging. We chose to use PopRecon 2.0 as the software to conduct SPR modeling due to its user-friendly interface. After using PopRecon 2.0 extensively, a couple of things should be noted. While the software was generally intuitive to use in most cases, several issues arose. When the model failed to run it would force the program to quit, requiring the user to re-enter the entire set of data with some change hoping the model would run. There were no diagnostics reported or even a simple error message. Lastly, once auxiliary data were entered, it could not be edited without exiting the program. While PopRecon 2.0 is incredibly useful when there are no issues, it has some major limitations when there are problems. Given the issues we found using it for Rhode Island deer population monitoring, future research should investigate alternative model fitting algorithms and software to implement SPR models.

Management Implications

Rhode Island specific estimates of reporting rate, wounding loss, and hunter effort are needed for accurate population estimates from SPR models. Rhode Island should consider conducting research, such as mark-recapture (Goode et al. 2014 and Marescot et al. 2015) studies and annual hunter surveys to estimate annual reporting rate. In this research, three scenarios (25%, 50%, and 75% reporting rate) were presented, assuming reporting rate was the same between sexes, which may not be true. Therefore, Rhode Island Division of Fish and Wildlife should attempt to increase reporting rate as high as possible or determine, with confidence, what the actual reporting rate is prior to including SPR results as part of the official monitoring strategy.

Additionally, hunter surveys and survival studies should be conducted to better estimate wounding loss in Rhode Island as the current rate is not supported by Rhode Island specific data and rates per method are likely different for the three methods that can be used: archery, muzzleloader, and shotgun. Given the archery harvest exceeded the muzzleloader harvest for the first time, the rate may also be changing through time. Ideally a mark-recapture study with collared deer would be used during the same time frame as hunter surveys to determine the accuracy of hunter surveys, potentially reducing costs of future research. Regarding hunter effort, hunter-days may be a better measure of effort than simply the total number of hunting licenses sold (Clawson et al. 2017) or standardized harvest. A hunter survey could be used to obtain an estimate of hunter-days.

To address the concerns of reliably aging females, one option that is already in regulation for the 2021-2022 hunting season is the change in check station days. This will hopefully result in an increased number of females aged by staff. In addition, if staff could collect more information/characteristics on females that were accurately aged by staff that are also able to be recorded on online/phone reports, this could improve the predictive model's accuracy. Additional options could include allowing photographs to be submitted with the online harvest reports; adding check station days; or collecting additional data from butchers or taxidermists that process deer for hunters. If no additional characteristics are reliable, it may result in only using two age classes, fawn and adult.

Lastly, determining sex of road-killed deer is critical as Pop Recon 2.0 operates on a single sex estimation and road-killed deer could exceed 50% of the total reported harvest. Therefore, when possible, sex should be identified when collecting roadkill data so that data can be used. This will produce more accurate estimates when adding total roadkill's to abundance estimates. This may be accomplished by collaborating with the RIDEM Division of Law Enforcement to obtain more accurate roadkill sex data.

Acknowledgements

We thank hunters and sport shooters who provide funds through to the Pittman Robertson Act through an excise tax placed on firearms, ammunition, and other hunting equipment. This Act allows the United States Fish and Wildlife Service to provide funding for this research. In addition, we thank and the University of Rhode Island for overseeing this project and the Rhode Island Department of Environmental Management, Division of Fish and Wildlife.

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Table 1. Rhode Island white-tailed deer age-at-harvest data collected from 2011-2020. Male and female were separated as PopRecon 2.0 only allows for single sex estimation. Each sex was pooled into three age classes: age = 0.5 (fawns), age = 1.5 (yearlings), and age = ≥2.5 (adults). Hunter effort was derived from annual hunter effort for each sex by using the annual reported harvest (2011-2020), standardized to the average over the same time.

Year	Male				Female			
	Age 0.5	Age 1.5	Age ≥2.5	Hunter Effort	Age 0.5	Age 1.5	Age ≥2.5	Hunter Effort
2011	269	254	657	1.05	19	17	1,146	1.13
2012	168	348	700	1.08	46	19	941	0.96
2013	214	287	645	1.02	57	11	1,241	1.25
2014	163	303	568	0.92	35	16	1,083	1.08
2015	172	250	493	0.82	8	8	953	0.93
2016	190	222	538	0.85	16	5	965	0.94
2017	51	203	715	0.86	79	12	823	0.87
2018	126	397	665	1.06	85	45	807	0.90
2019	116	292	875	1.14	65	25	912	0.96
2020	112	451	770	1.19	100	57	859	0.97
Total	1,581	3,007	6,626	-	510	215	9,730	-

Table 2. Rhode Island deer aged vs. not aged for each specific sex. Deer that were aged were done so at state operated check stationed by Division of Fish and Wildlife staff. Deer "not aged" were assigned an age by using a predictive model.

Year	**Males**		**Females**	
	Aged	**Not Aged**	**Aged**	**Not Aged**
2011	130	1,050	81	1,101
2012	238	978	108	898
2013	117	1,029	83	1,226
2014	220	814	90	1,044
2015	181	734	61	908
2016	164	786	87	899
2017	95	874	65	849
2018	155	1,033	55	882
2019	123	1,160	81	921
2020	202	1,131	56	960
Total	1,625	9,589	767	9,688

Table 3. Hunter harvested deer aged via Matson's Laboratory via cementum analysis compared to results of aging deer at check stations aged by staff via tooth wear.

Check Station Ages	**Lab Ages**		
	0.5	**1.5**	**≥2.5**
0.5	35	1	2
1.5	3	111	18
≥2.5	2	59	251

Table 4. Rhode Island male white-tailed deer total harvest under 25%, 50% and 75% reporting rate estimates.

Year	Total	Reported Harvest	Non-Reported Harvest	Wounding Loss
		25% Reporting		
2011	5,428	1,180	3,540	708
2012	5,594	1,216	3,648	730
2013	5,272	1,146	3,438	688
2014	4,756	1,034	3,102	620
2015	4,209	915	2,745	549
2016	4,370	950	2,850	570
2017	4,457	969	2,907	581
2018	5,465	1,188	3,564	713
2019	5,902	1,283	3,849	770
2020	6,132	1,333	3,999	800
Year		**50% Reporting**		
2011	2,714	1,180	1,180	354
2012	2,797	1,216	1,216	365
2013	2,636	1,146	1,146	344
2014	2,378	1,034	1,034	310
2015	2,105	915	915	275
2016	2,185	950	950	285
2017	2,229	969	969	291
2018	2,732	1,188	1,188	356
2019	2,951	1,283	1,283	385
2020	3,066	1,333	1,333	400
Year		**75% Reporting**		
2011	1,809	1,180	393	236
2012	1,865	1,216	405	243
2013	1,757	1,146	382	229
2014	1,585	1,034	345	207
2015	1,403	915	305	183
2016	1,457	950	317	190
2017	1,486	969	323	194
2018	1,822	1,188	396	238
2019	1,967	1,283	428	257
2020	2,044	1,333	444	267

Table 5. Rhode Island female white-tailed deer total harvest under 25%, 50% and 75% reporting rate estimates.

Year	Total	Reported Harvest	Non-Reported Harvest	Wounding Loss
		25% Reporting		
2011	5,437	1,182	3,546	709
2012	4,628	1,006	3,018	604
2013	6,021	1,309	3,927	785
2014	5,216	1,134	3,402	680
2015	4,457	969	2,907	581
2016	4,536	986	2,958	592
2017	4,204	914	2,742	548
2018	4,310	937	2,811	562
2019	4,609	1,002	3,006	601
2020	4,674	1,016	3,048	610
Year		**50% Reporting**		
2011	2,719	1,182	1,182	355
2012	2,314	1,006	1,006	302
2013	3,011	1,309	1,309	393
2014	2,608	1,134	1,134	340
2015	2,229	969	969	291
2016	2,268	986	986	296
2017	2,102	914	914	274
2018	2,155	937	937	281
2019	2,305	1,002	1,002	301
2020	2,337	1,016	1,016	305
Year		**75% Reporting**		
2011	1,812	1,182	394	236
2012	1,543	1,006	335	201
2013	2,007	1,309	436	262
2014	1,739	1,134	378	227
2015	1,486	969	323	194
2016	1,512	986	329	197
2017	1,401	914	305	183
2018	1,437	937	312	187
2019	1,536	1,002	334	200
2020	1,558	1,016	339	203

Table 6. Rhode Island white-tailed deer age-at-harvest data collected from 2016-2020. Male and female were separated as PopRecon 2.0 only allows for single sex estimation. Each sex was pooled into three age classes: age = 0.5 (fawns), age = 1.5 (yearlings), and age = ≥2 (adults). Hunter effort was derived from annual hunter effort for each sex by using the annual reported harvest (2016-2020), standardized to the average over the same time.

Year	**Male**				**Female**			
	Age 0.5	**Age 1.5**	**Age ≥2.5**	**Hunter Effort**	**Age 0.5**	**Age 1.5**	**Age ≥2.5**	**Hunter Effort**
2016	190	222	538	0.80	16	5	965	1.00
2017	51	203	715	0.80	79	12	823	0.90
2018	126	397	665	1.00	85	45	807	1.00
2019	116	292	875	1.10	65	25	912	1.00
2020	112	451	770	1.20	100	57	859	1.10

Table 7. Survival probability for male and female white-tailed deer.

Year	**Age**	**Male**	**Standard Error**
1-10	0.5	0.36	0.1
1-10	≥1.5	0.9	0.1
Year	**Age**	**Female**	**Standard Error**
1-10	0.5	0.36	0.001
1-10	≥1.5	0.98	0.001

Table 8. A concordance table between predictions and staff age classifications using the predictive model.

Age	**Fawn**	**Yearling**	**Adult**
0.5	1,545	59	467
1.5	202	716	2,226
≥2.5	982	144	15,054

Table 9. PopRecon 2.0 results from male data collected from 2011-2020. The data was arranged in three age classes, age ≥2.5 being pooled. Hunter effort was determined with male harvest (Table 1). For harvest probabilities, last distinct age was set at one, and range was set from 0.1 - 0.5. Auxiliary data and random effects were not used. Survival probabilities last distinct age was set at one, and range was set from 0.85 – 0.95. Estimates with standard errors were used as auxiliary data. Age = 0.5 years, 1-10 were 0.36 S.E. = 0.1 and age=1, years 1-10 were 0.9 S.E. = 0.1 (Table 7). Random effects were not used. Results are calculated for all reporting rates 25%, 50% and 75% (Table 4).

Reporting Rate	Year	Abundance Estimate				Harvest Probability Estimate		Recruitment Estimate	Survival Probability Estimate			Model Metrics	
		Age 0.5	Age 1.5	Age ≥2.5	Total Annual	Age 0	Ages ≥1.5	Age 0	Years	Age 0.5	Ages ≥1.5	# parameters	14
25%	1	2,020.10	2,281.80	5,902.10	10,204.0 (345.418)	0.328 (0.0067)	0.274 (0.0059)	2,020.1 (125.106)	1-10	0.878 (0.0085)	0.901 (0.0058)	Log-likelihood	-6271
	2	1,242.00	3,074.60	6,184.60	10,501.2 (366.822)	0.335 (0.0068)	0.280 (0.0060)	1,242.0 (94.731)				AIC	12570
	3	1,621.70	2,604.40	5,853.00	10,079.1 (359.350)	0.320 (0.0066)	0.267 (0.0058)	1,621.7 (111.391)					
	4	1,326.20	2,961.60	5,551.70	9,839.5 (367.316)	0.294 (0.0062)	0.244 (0.0053)	1,326.2 (103.526)					
	5	1,500.10	2,628.10	5,182.60	9,310.8 (362.162)	0.266 (0.0057)	0.221 (0.0049)	1,500.1 (115.090)					
	6	1,627.70	2,290.00	5,549.70	9,467.4 (362.136)	0.275 (0.0058)	0.228 (0.0050)	1,627.7 (119.470)					
	7	438	2,098.70	7,392.00	9,928.7 (395.994)	0.277 (0.0059)	0.231 (0.0051)	438.0 (58.937)					
	8	919.10	3,463.30	5,801.20	10,183.6 (362.151)	0.330 (0.0067)	0.276 (0.0059)	919.1 (79.608)					
	9	828.2	2,486.70	7,451.60	10,766.5 (369.935)	0.350 (0.0070)	0.293 (0.0062)	828.2 (73.837)					
	10	752.7	3,609.60	6,162.80	10,525.1 (362.325)	0.362 (0.0072)	0.304 (0.0064)	752.7 (68.337)					
50%	1	14,279.60	2,228.20	5,763.50	22,271.2 (2919.222)	0.043 (0.0061)	0.261 (0.0305)	14,279.6 (2191.827)	1-10	0.352 (0.0316)	0.866 (0.0317)	# parameters	14
	2	8,893.50	3,055.00	6,145.10	18,093.7 (2314.844)	0.044 (0.0063)	0.268 (0.0311)	8,893.5 (1439.084)				Log-likelihood	-332.311
	3	11,433.00	2,525.00	5,674.70	19,632.8 (2587.078)	0.042 (0.0059)	0.255 (0.0299)	11,433.0 (1808.676)				AIC	692.622
	4	9,854.80	2,981.80	5,589.60	18,426.2 (2438.636)	0.038 (0.0054)	0.233 (0.0278)	9,854.8 (1609.519)					
	5	11,253.10	2,631.20	5,188.80	19,073.1 (2577.047)	0.034 (0.0048)	0.211 (0.0255)	11,253.1 (1826.492)					
	6	12,562.60	2,369.70	5,742.70	20,675.0 (2785.131)	0.035 (0.0050)	0.218 (0.0262)	12,562.6 (2014.645)					
	7	3,512.10	2,259.50	7,958.40	13,730.0 (1753.561)	0.036 (0.0050)	0.220 (0.0264)	3,512.1 (704.643)					
	8	6,251.00	3,258.60	5,458.40	14,968.0 (1892.498)	0.044 (0.0061)	0.264 (0.0307)	6,251.0 (1047.870)					
	9	6,528.20	2,744.10	8,222.90	17,495.2 (2158.837)	0.047 (0.0066)	0.280 (0.0323)	6,528.2 (1102.213)					
	10	4,956.50	3,352.10	5,723.20	14,031.8 (1745.679)	0.049 (0.0069)	0.291 (0.0332)	4,956.5 (850.787)					
75%	1	10,198.30	1,552.80	4,016.40	15,767.5 (2183.784)	0.040 (0.0059)	0.250 (0.0316)	10,198.3 (1619.284)	1-10	0.348 (0.0316)	0.854 (0.0319)	# parameters	14
	2	6,316.80	2,117.00	4,258.20	12,692.1 (1719.393)	0.041 (0.0061)	0.256 (0.0323)	6,316.8 (1051.316)				Log-likelihood	-345.201
	3	8,227.60	1,773.30	3,985.40	13,986.3 (1940.907)	0.039 (0.0058)	0.244 (0.0310)	8,227.6 (1340.001)				AIC	718.402
	4	7,060.20	2,085.80	3,910.10	13,056.1 (1820.996)	0.035 (0.0052)	0.223 (0.0287)	7,060.2 (1184.566)					
	5	8,138.30	1,859.00	3,666.00	13,663.3 (1941.725)	0.032 (0.0047)	0.201 (0.0263)	8,138.3 (1357.999)					
	6	8,944.00	1,647.90	3,993.60	14,585.6 (2067.083)	0.033 (0.0048)	0.208 (0.0270)	8,944.0 (1475.952)					
	7	2,473.80	1,554.50	5,475.10	9,503.3 (1291.853)	0.033 (0.0049)	0.210 (0.0273)	2,473.8 (505.000)					
	8	4,547.00	2,312.70	3,874.00	10,733.8 (1440.716)	0.041 (0.0060)	0.252 (0.0318)	4,547.0 (782.372)					
	9	4,515.00	1,851.00	5,546.70	11,912.8 (1562.691)	0.044 (0.0064)	0.268 (0.0335)	4,515.0 (781.691)					
	10	3,609.80	2,380.50	4,064.20	10,054.5 (1325.869)	0.046 (0.0067)	0.278 (0.0345)	3,609.8 (633.418)					

Table 10. PopRecon 2.0 results from female data collected from 2011-2020. The data was arranged in three age classes, age ≥2.5 being pooled. Hunter effort was determined with female harvest (Table 1). For harvest probabilities, last distinct age was set at zero, and range was set from 0.1 - 0.3. Auxiliary data and random effects were not used. Survival probabilities last distinct age was set at one, and range was set from 0.98 – 0.99. Estimates with standard errors were used as auxiliary data. Age = 0.5, years 1-10 were 0.36 S.E. = 0.0001 and age=1, years 1-10 were 0.98 S.E. = 0.0001 (Table 7). Random effects were not used. Results are calculated for all reporting rates 25%, 50% and 75% (Table 5).

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Reporting Rate	Year	Abundance Estimate				Harvest Probability Estimate	Recruitment Estimate	Survival Probability Estimate			Model Metrics	
		Age 0.5	Age 1.5	Age ≥2.5	Total Annual	All Ages	Age 0.5	Years	Age 0.5	Ages ≥1.5		
25%		Does not run										
50%	1	8,616,130,705.40	7,709,169,578.50	519,688,725,706.40	536,014,025,990.3 (118898972995.224)	0.959 (0.0010)	8,616,130,705.4 (2814938749.481)	1-10	0.358 (0.0003)	0.980 (0.0003)	# parameters	13
	2	148.30	61.20	3,032.70	3,242.2 (120.904)	0.934 (0.0014)	148.3 (18.640)				Log-likelihood	-28131.6
	3	194.80	37.60	4,241.70	4,474.2 (148.742)	0.971 (0.0008)	194.8 (22.252)				AIC	56289.2
	4	84.30	38.50	2,607.10	2,729.8 (85.483)	0.953 (0.0011)	84.3 (11.056)					
	5	16.20	16.20	1,931.80	1,964.3 (59.422)	0.929 (0.0014)	16.2 (4.095)					
	6	55.60	17.40	3,350.80	3,423.7 (136.871)	0.931 (0.0014)	55.6 (11.845)					
	7	260.30	39.50	2,712.10	3,011.9 (116.417)	0.915 (0.0016)	260.3 (25.663)					
	8	188.60	99.80	1,790.60	2,079.0 (64.578)	0.922 (0.0015)	188.6 (15.742)					
	9	151.50	58.30	2,125.10	2,334.8 (69.143)	0.934 (0.0014)	151.5 (14.492)					
	10	170.20	97.00	1,461.90	1,729.1 (40.127)	0.936 (0.0013)	170.2 (11.168)					
75%	1	3,667.10	3,281.10	221,181.90	228,130.0 (264327.155)	0.008 (0.0093)	3,667.1 (4435.648)	1-10	0.36 (0.0003)	0.98 (0.0003)	# parameters	13
	2	10,343.80	4,272.40	211,597.50	226,213.7 (263435.466)	0.007 (0.0079)	10,343.8 (12259.933)				Log-likelihood	-131.192
	3	9,981.00	1,926.20	217,305.10	229,212.3 (266605.654)	0.009 (0.0102)	9,981.0 (11776.473)				AIC	288.384
	4	7,025.80	3,211.80	217,398.60	227,636.3 (265025.108)	0.008 (0.0089)	7,025.8 (8373.223)					
	5	1,870.50	1,870.50	222,818.40	226,559.3 (264024.599)	0.007 (0.0076)	1,870.5 (2401.824)					
	6	3,699.90	1,156.20	223,147.30	228,003.3 (265685.533)	0.007 (0.0077)	3,699.9 (4534.503)					
	7	19,799.50	3,007.50	206,266.00	229,073.1 (266985.919)	0.006 (0.0071)	19,799.5 (23306.392)					
	8	20,450.70	10,826.90	194,161.70	225,439.3 (262682.641)	0.006 (0.0074)	20,450.7 (24049.213)					
	9	14,970.60	5,757.90	210,048.60	230,777.0 (269089.459)	0.007 (0.0079)	14,970.6 (17671.716)					
	10	22,173.80	12,639.00	190,472.50	225,285.3 (262792.104)	0.007 (0.0080)	22,173.8 (26066.588)					

Table 11. PopRecon 2.0 results from male data collected from 2016-2020. The data was arranged in three age classes, age ≥2.5 being pooled. Hunter effort was determined with male harvest (Table 6). For harvest probabilities, last distinct age was set at one, and range was set from 0.1 - 0.5. Auxiliary data and random effects were not used. Survival probabilities last distinct age was set at one, and range was set from 0.85 – 0.95. Estimates with standard errors were used as auxiliary data. Age = 0.5, years 1-5 were 0.36 S.E. = 0.1 and age=1, years 1-5 were 0.9 S.E. = 0.1 (Table 7). Random effects were not used. Results are calculated for all reporting rates 25%, 50% and 75% (Table 4).

Reporting Rate	Year	Abundance Estimate				Harvest Probability Estimate		Recruitment Estimate	Survival Probability Estimate			Model Metrics	
		Age 0.5	Age 1.5	Age ≥2.5	Total Annual	Age 0.5	Ages ≥1.5	Age 0.5	Years	Age 0.5	Ages ≥1.5	# parameters	9
25%	6	31,965.50	4,378.00	10,609.60	46,953.1 (8242.472)	0.026 (0.0051)	0.225 (0.0342)	31,965.5 (6535.528)	6-10	0.359 (0.0447)	0.899 (0.0443)	Log-likelihood	-169.949
	7	9,811.30	4,577.60	16,123.20	30,512.1 (4972.238)	0.026 (0.0051)	0.225 (0.0342)	9,811.3 (2352.835)				AIC	343.898
	8	16,074.80	6,099.40	10,216.90	32,391.1 (5346.130)	0.033 (0.0063)	0.273 (0.0401)	16,074.8 (3407.948)					
	9	16,268.80	4,998.20	14,977.50	36,244.5 (5794.404)	0.036 (0.0069)	0.296 (0.0427)	16,268.8 (3449.995)					
	10	12,661.80	6,306.10	10,766.60	29,734.5 (4825.494)	0.039 (0.0075)	0.318 (0.0451)	12,661.8 (2757.077)					
50%	6	16,479.70	2,230.80	5,406.20	24,116.7 (4289.061)	0.026 (0.0050)	0.222 (0.0344)	16,479.7 (3388.874)	6-10	0.358 (0.0447)	0.895 (0.0443)	# parameters	9
	7	4,971.10	2,292.40	8,074.30	15,337.9 (2535.788)	0.026 (0.0050)	0.222 (0.0344)	4,971.1 (1194.390)				Log-likelihood	-168.938
	8	8,393.20	3,146.50	5,270.60	16,810.3 (2809.478)	0.032 (0.0061)	0.269 (0.0403)	8,393.2 (1784.979)				AIC	355.876
	9	8,230.90	2,498.00	7,485.40	18,214.3 (2951.165)	0.035 (0.0067)	0.292 (0.0430)	8,230.9 (1751.354)					
	10	6,532.60	3,213.40	5,486.30	15,232.2 (2489.123)	0.038 (0.0073)	0.314 (0.0455)	6,532.6 (1419.628)					
75%	6	11,493.40	1,531.30	3,711.00	16,735.7 (3040.633)	0.025 (0.0048)	0.217 (0.0347)	11,493.4 (2390.671)	6-10	0.357 (0.0447)	0.889 (0.0445)	# parameters	9
	7	3,375.20	1,531.90	5,395.70	10,302.7 (1746.178)	0.025 (0.0048)	0.217 (0.0347)	3,375.2 (815.756)				Log-likelihood	-177.367
	8	5,941.90	2,191.20	3,670.40	11,803.5 (2017.395)	0.031 (0.0060)	0.264 (0.0408)	5,941.9 (1274.512)				AIC	372.734
	9	5,582.60	1,666.10	4,992.70	12,241.5 (2032.889)	0.034 (0.0066)	0.286 (0.0436)	5,582.6 (1198.991)					
	10	4,555.60	2,203.10	3,761.40	10,520.0 (1749.158)	0.037 (0.0072)	0.307 (0.0461)	4,555.6 (993.110)					

Table 12. PopRecon 2.0 results from male data collected from 2011-2020. The data was arranged in two age classes, age ≥1.5 being pooled. Hunter effort was determined with male harvest (Table 1). For harvest probabilities, last distinct age was set at one, and range was set from 0.1 - 0.5. Auxiliary data and random effects were not used. Survival probabilities last distinct age was set at one, and range was set from 0.85 – 0.95. Estimates with standard errors were used as auxiliary data. Age = 0.5, years 1-10 were 0.36 S.E. = 0.1 and age=1, years 1-10 were 0.9 S.E. = 0.1 (Table 7). Random effects were not used. Results are calculated for all reporting rates 25%, 50% and 75% (Table 4).

Reporting Rate	Year	Abundance Estimate			Harvest Probability Estimate		Recruitment Estimate	Survival Probability Estimate			Model Metrics	
		Age 0.5	Age ≥1.5	Total Annual	Age 0.5	Ages ≥1.5	Age 0.5	Years	Age 0.5	Ages ≥1.5	# parameters	14
25%	1	122,529,777.10	287,379,105.80	409,908,882.9 (1459453602.493)	0.000 (0.0000)	0.000 (0.0001)	122,529,777.1 (435653374.599)	1-10	0.360 (0.0316)	0.918 (0.0278)	Log-likelihood	-82.9572
	2	74,109,170.50	320,163,213.20	394,272,383.8 (1404595903.260)	0.000 (0.0000)	0.000 (0.0001)	74,109,170.5 (263791285.776)				AIC	193.9144
	3	100,662,758.50	303,611,618.90	404,274,377.4 (1439668004.023)	0.000 (0.0000)	0.000 (0.0001)	100,662,758.5 (358064327.485)					
	4	85,051,984.00	314,747,528.00	399,799,512.0 (1424136937.553)	0.000 (0.0000)	0.000 (0.0000)	85,051,984.0 (302772341.987)					
	5	101,320,126.20	303,112,136.10	404,432,262.3 (1440364475.105)	0.000 (0.0000)	0.000 (0.0000)	101,320,126.2 (360623830.930)					
	6	108,601,551.40	300,845,336.80	409,446,888.2 (1458012935.140)	0.000 (0.0000)	0.000 (0.0000)	108,601,551.4 (386423585.738)					
	7	28,357,454.10	353,498,044.70	381,855,498.8 (1361645106.917)	0.000 (0.0000)	0.000 (0.0000)	28,357,454.1 (101674058.347)					
	8	56,983,373.10	332,620,937.20	389,604,310.3 (1388424500.431)	0.000 (0.0000)	0.000 (0.0001)	56,983,373.1 (203047867.065)					
	9	48,612,047.90	338,691,514.30	387,303,562.1 (1380415458.355)	0.000 (0.0000)	0.000 (0.0001)	48,612,047.9 (173281777.500)					
	10	44,883,998.10	338,872,834.10	383,756,832.2 (1367865273.931)	0.000 (0.0000)	0.000 (0.0001)	44,883,998.1 (160019162.815)					
50%		Does not run										
75%	1	44,036,904.90	104,173,180.10	148,210,084.9 (481795413.310)	0.000 (0.0000)	0.000 (0.0000)	44,036,904.9 (143388390.186)	1-10	0.360 (0.0316)	0.919 (0.0275)	# parameters	14
	2	26,698,524.80	116,335,518.40	143,034,043.1 (465117112.386)	0.000 (0.0000)	0.000 (0.0000)	26,698,524.8 (87042184.966)				Log-likelihood	-79.0273
	3	36,117,899.30	109,874,682.90	145,992,582.2 (474655946.236)	0.000 (0.0000)	0.000 (0.0000)	36,117,899.3 (117668821.826)				AIC	186.0546
	4	30,505,230.10	113,861,731.90	144,366,962.0 (469449223.556)	0.000 (0.0000)	0.000 (0.0000)	30,505,230.1 (99463058.590)					
	5	36,229,745.90	109,319,717.50	145,549,463.3 (473270876.516)	0.000 (0.0000)	0.000 (0.0000)	36,229,745.9 (118107603.115)					
	6	38,723,258.60	108,194,394.00	146,917,652.6 (477685053.885)	0.000 (0.0000)	0.000 (0.0000)	38,723,258.6 (126199037.807)					
	7	10,194,984.70	128,183,451.70	138,378,436.4 (450237422.592)	0.000 (0.0000)	0.000 (0.0000)	10,194,984.7 (33483877.679)					
	8	20,461,640.70	120,466,826.90	140,928,467.6 (458350905.309)	0.000 (0.0000)	0.000 (0.0000)	20,461,640.7 (66780571.555)					
	9	17,481,806.50	122,849,187.90	140,330,994.4 (456431806.138)	0.000 (0.0000)	0.000 (0.0000)	17,481,806.5 (57076223.222)					
	10	16,159,479.90	123,054,691.10	139,214,171.0 (452812472.739)	0.000 (0.0000)	0.000 (0.0000)	16,159,479.9 (52767818.454)					

Table 13. PopRecon 2.0 results from female data collected from 2011-2020. The data was arranged in two age classes, age ≥1.5 being pooled. Hunter effort was determined with female harvest (Table 1). For harvest probabilities, last distinct age was set at zero, and range was set from 0.1 - 0.3. Auxiliary data and random effects were not used. Survival probabilities last distinct age was set at one, and range was set from 0.98 – 0.99. Estimates with standard errors were used as auxiliary data. Age = 0.5, years 1-10 were 0.36 S.E. = 0.0001 and age=1, years 1-10 were 0.98 S.E. = 0.0001 (Table 7). Random effects were not used. Results are calculated for all reporting rates 25%, 50% and 75% (Table 5).

Reporting Rate	Year	Abundance Estimate			Harvest Probability Estimate	Recruitment Estimate	Survival Probability Estimate			Model Metrics	
		Age 0.5	Age ≥1.5	Total Annual	All Ages	Age 0.5	Years	Age 0.5	Ages ≥1.5	# parameters	13
25%	1	90,906.70	5,564,449.80	5,655,356.5 (17800908.232)	0.001 (0.0028)	90,906.7 (294175.401)	1-10	0.371 (0.0003)	0.98 (0.0003)	Log-likelihood	-674.909
	2	276,561.00	5,771,707.80	6,048,268.8 (19055301.626)	0.001 (0.0024)	276,561.0 (881199.691)				AIC	1375.818
	3	263,420.00	5,785,996.50	6,049,416.5 (19054709.263)	0.001 (0.0031)	263,420.0 (837354.004)					
	4	188,211.10	5,909,829.00	6,098,040.1 (19211443.180)	0.001 (0.0027)	188,211.1 (601905.971)					
	5	49,745.50	5,975,673.80	6,025,419.3 (18985721.975)	0.001 (0.0023)	49,745.5 (167085.401)					
	6	98,265.80	5,957,363.80	6,055,629.6 (19079585.730)	0.001 (0.0024)	98,265.8 (319901.917)					
	7	527,037.80	5,570,589.20	6,097,627.0 (19213367.253)	0.001 (0.0022)	527,037.8 (1671207.997)					
	8	554,835.10	5,561,405.70	6,116,240.7 (19275842.095)	0.001 (0.0023)	554,835.1 (1758872.339)					
	9	398,984.20	5,751,510.10	6,150,494.2 (19383684.545)	0.001 (0.0024)	398,984.2 (1267335.601)					
	10	606,843.10	5,558,683.10	6,165,526.2 (19430006.523)	0.001 (0.0024)	606,843.1 (1921869.605)					
50%	1	9,928.80	607,747.10	617,675.9 (1447398.912)	0.004 (0.0103)	9,928.8 (23965.828)	1-10	0.36 (0.0003)	0.98 (0.0003)	# parameters	13
	2	28,266.80	589,915.50	618,182.3 (1453547.014)	0.004 (0.0088)	28,266.8 (67271.742)				Log-likelihood	-73.0394
	3	26,913.70	591,157.60	618,071.3 (1452405.155)	0.005 (0.0114)	26,913.7 (63866.588)				AIC	172.0788
	4	19,105.70	599,917.40	619,023.1 (1455486.527)	0.004 (0.0099)	19,105.7 (45649.117)					
	5	5,068.90	608,898.60	613,967.5 (1444215.598)	0.004 (0.0085)	5,068.9 (12764.071)					
	6	10,036.10	608,440.40	618,476.5 (1454485.466)	0.004 (0.0086)	10,036.1 (24441.870)					
	7	53,532.00	565,813.40	619,345.5 (1456837.636)	0.003 (0.0080)	53,532.0 (126774.890)					
	8	55,718.50	558,496.10	614,214.6 (1445816.934)	0.004 (0.0082)	55,718.5 (131981.909)					
	9	39,926.90	575,561.20	615,488.1 (1448861.206)	0.004 (0.0088)	39,926.9 (94781.229)					
	10	60,862.10	557,497.30	618,359.4 (1455361.984)	0.004 (0.0089)	60,862.1 (144003.841)					
75%	1	6,710.30	410,742.30	417,452.6 (967239.758)	0.004 (0.0101)	6,710.3 (16016.606)	1-10	0.36 (0.0003)	0.98 (0.0003)	# parameters	13
	2	19,119.00	399,005.70	418,124.7 (971083.826)	0.004 (0.0086)	19,119.0 (44944.379)				Log-likelihood	-70.6129
	3	18,197.30	399,701.60	417,898.9 (969947.548)	0.005 (0.0111)	18,197.3 (42652.606)				AIC	167.2258
	4	12,925.60	405,864.50	418,790.2 (972514.728)	0.004 (0.0096)	12,925.6 (30502.848)					
	5	3,429.20	411,930.10	415,359.2 (964934.959)	0.004 (0.0083)	3,429.2 (8529.842)					
	6	6,788.00	411,520.60	418,308.6 (971628.338)	0.004 (0.0084)	6,788.0 (16329.394)					
	7	36,195.80	382,575.80	418,771.6 (972897.727)	0.003 (0.0078)	36,195.8 (84663.659)					
	8	37,682.60	377,712.40	415,395.0 (965478.375)	0.003 (0.0080)	37,682.6 (88135.502)					
	9	26,993.20	389,117.70	416,110.9 (967104.342)	0.004 (0.0086)	26,993.2 (63267.313)					
	10	41,145.60	376,893.40	418,039.0 (971461.727)	0.004 (0.0087)	41,145.6 (96124.601)					

Figure 1. Map of Rhode Island identifying the four deer management zones. Deer management zones differ by hunting regulations, including seasons and bag limits, human use, habitat type, and public acceptance of deer.

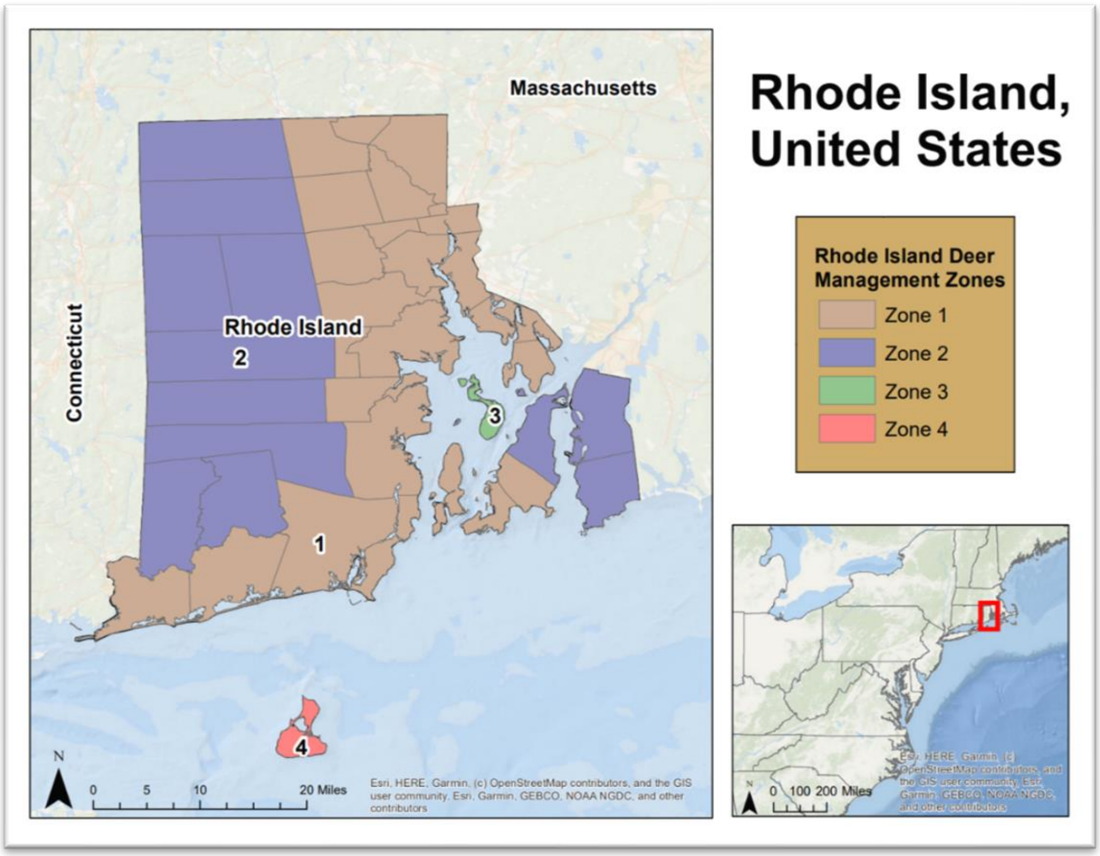

Figure 2. Rhode Island reported male and female deer hunter harvest from 2011 – 2020.

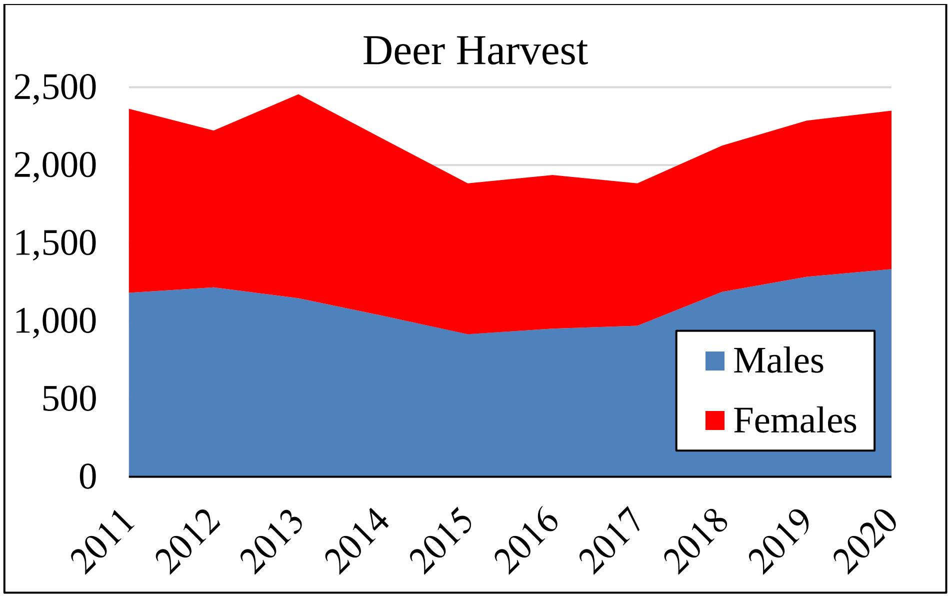

Figure 3. Male total abundance estimates for all three (25%, 50%, 75%) reporting rates. Confidence intervals (C.I.) for each abundance estimate generally increased as abundance estimates increased.

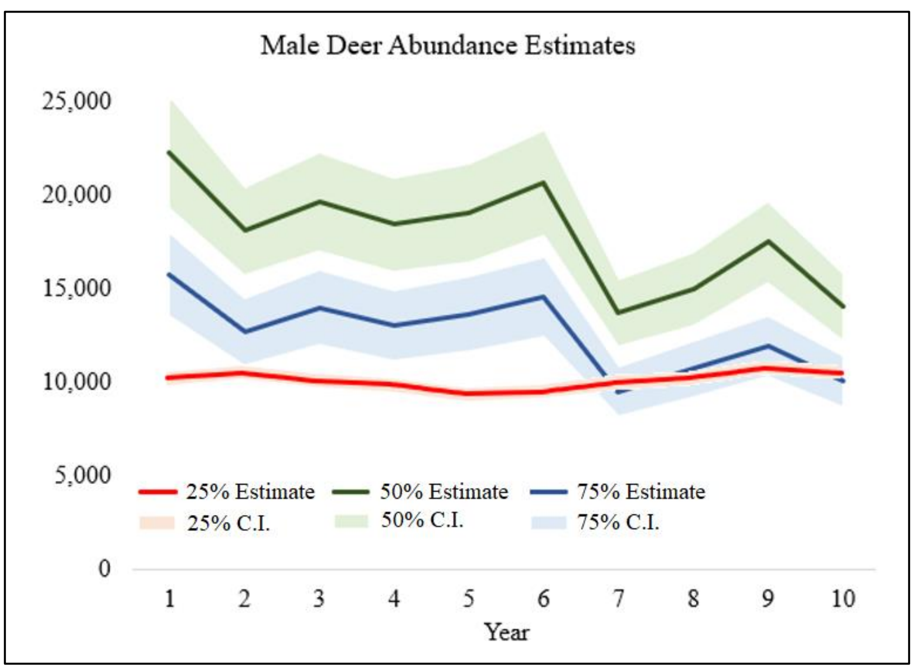

Figure 4. Rhode Island male and female deer aged via staff at state operated check stations via tooth wear.