ENHANCING FURBEARER MANAGEMENT IN NEW YORK

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Management of furbearers presents numerous challenges due to their often elusive nature, relatively low population densities, and limited distributions. Improved knowledge of harvests, and harvest impacts on populations, can lead to enhanced management strategies that provide opportunities for sustainable use while conserving populations.

We implemented a management action to explore the feasibility of providing fisher harvest opportunities beyond the traditional trapping season closure of December 10th. An experimental management action was implemented to address the question of whether fisher-trapping effort, or capture vulnerability, would vary in a season extension. We collected over 100,000 trap nights of data and found no significant change in capture vulnerability, but that trapping effort, both in terms of number of active trappers and mean individual effort, was significantly lower during the extended portion of the seasons. Thus, we found that the addition of fisher harvest opportunities beyond December 10th does not increase cumulative harvest proportionally.

We also developed a population model for a previously unexploited bobcat population. Survivorship parameters were estimated from harvest age structure using a Bayesian approach that allowed for the incorporation of external data and parameter updating as new data was made available. The posterior estimates of survivorship were incorporated into simulations of projection matrix models. The distribution of projected growth rates produced from the matrix model simulations showed decreasing variation in model projections as the survivorship information was refined. Ultimately, as more information was obtained, we refined our annual survivorship

estimates to 0.81 (σ^2=.006). Consequently, our estimated projected growth rates changed from λ=0.93 (σ^2=0.28) to λ=1.14 (σ^2=0.014).

Finally, we propose that understanding the population status of furbearers, and the impacts of management actions, is crucial for wildlife management agencies in fulfilling their obligations to society. We recommend what data are needed to gain this understanding and how these data can be collected from harvest-dependent sources. In this, we hope to demonstrate that the challenges of furbearer management are not insurmountable and to encourage agencies to developing strong, data-driven furbearer conservation programs that improve the management and stewardship of this resource.

BIOGRAPHICAL SKETCH

Nathan M. Roberts was born to the Rev. Michael and Linda Roberts. From an early age, Nathan has been fascinated with the natural world. As an adolescent in the Ozark Mountains of Missouri, he witnessed the importance of wildlife resources to rural cultures. He quickly realized he wanted to be part of ensuring that these resources were managed in way that is scientifically sound, equitable, and sustainable. He believes that the stewardship of our natural resources is not only a service to society, but a charge of the Creator who has blessed us with His creations.

Nathan earned his undergraduate degree at the University of Missouri. He worked, briefly, for the Missouri Department of Conservation before returning to the University of Missouri to earn his Masters degree while studying river otters. After the completion of his graduate degree, Nathan was employed as a wildlife biologist for the United States Department of Agriculture's National Wildlife Research Center. His research interest in furbearer management and quantitative biology brought him to Cornell University to conduct his doctoral research under the guidance of Dr. Patrick Sullivan.

This dissertation is dedicated to David A. Hamilton (1955-2007);

scientist, mentor and friend.

You have impacted my life, and many others, beyond measure.

Words cannot express my gratitude.

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This work could not have been a success without the combined effort of many partners. I would first like to thank my major advisor, Dr. Pat Sullivan, for his support, guidance, and instruction. Dr. Sullivan challenged me and exposed me to new methods of inquiry and investigation. He allowed the freedom and flexibility to explore my interests and mature as a scientist. I am indebted to Dr. Evan Cooch for introducing me to the concept of Adaptive Harvest Management, and the fact that this term can be more than simply a buzz word. Dr. Milo Richmond was a fountain of knowledge. He imparted wisdom to me on subjects ranging from wildlife ecology to the importance of cooperation between agencies. Dr. Richmond graciously provided office space, vehicle, and other logistical support from the USGS Cooperative Fish and Wildlife Research Unit. Dr. Dan Decker always challenged me to look at the bigger picture of my work.

The New York State Department of Environmental Conservation provided the funding for this work. More significantly, they provided advice and support whenever requested. Gordon Batcheller has been a mentor on this dissertation, and many other issues. I am grateful for his support, friendship, and the many opportunities he provided. Thanks are also due to Patty Riexinger for her support and encouragement. The Furbearer Management Team welcomed me from the start and provided assistance whenever needed. I especially thank Andrew MacDuff and William Sharick for their collaboration on the experiments described within this dissertation. They took a risk, for which I am grateful.

My fellow graduate students were always a source of inspiration. The expertise of the graduate students at Cornell is outstanding. I am especially grateful to my fellow quantitative and ecology focused peers: Amielle DeWan, Willie Fetzer, Elise Zipkin, Beth Gardner, and Chris Janelle. Numerous undergraduate students also assisted with mailings, data entry, necropsies and dissections.

I appreciate the advice of my colleagues in the Northeast Fur Resources Technical Committee, the US Furbearer Conservation Technical Working Group, and the US C.I.T.E.S. Working Group. The ability to reach out to biologists across the country, and beyond, for advice and suggestions has been invaluable.

Finally, I would like to thank the trappers of New York. They provided the data necessary to make this project a success. They did this out of their love for the resource and a genuine interest in the scientifically sound management of the resource they cherish. My interactions with the American trapper remind me of why I was first drawn to this field, and reaffirms my dedication to the stewardship of the resources we treasure.

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Using experimental trapping seasons to explore fisher harvest management uncertainties: Is timing everything?

Abstract

An experimental trapping season was used to address structural and partial controllability uncertainties related to the timing and duration structure of a fisher (Martes pennanti) trapping season in northern New York. Management actions were implemented to explore the impact of permitting fisher harvest beyond the traditional closing date of December 10th. Uncertainties regarding capture vulnerability and harvest effort have thwarted serious consideration of altering the season structure in northern New York. An experimental season was implemented in a portion of northern New York to explore these uncertainties. We found that capture vulnerability of fisher did not change between the traditional portion of the season (October 25 – Dec 10) and the experimental season extension (Dec 11–Jan 10). Trapping effort was significantly less during the extended portion of the season, both in terms of number of active trappers and mean individual effort. Thus, changes in the cumulative harvest appear to be driven by changes in harvest effort, rather than changes in capture vulnerability. Increasing harvest opportunity in this system does not appear to proportionally increase cumulative harvest.

Keywords: Adaptive resource management, fisher, Martes pennanti, furbearer, mustelid, mustelidae, trapping, fur harvest

Introduction

Several species of wildlife in New York are hunted or trapped primarily for their fur. The State of New York regulates the harvests of several furbearer species, including fisher (*Martes pennanti*). The New York State Department of Environmental Conservation (NYSDEC) is the regulatory agency charged with managing New York's furbearer resources and has the ability to regulate harvests to achieve desired objectives. The primary objective of furbearer management in New York is to maximize the positive impacts of the wildlife resource for the public (MacDuff et al. 2009). The long-term, sustainable, consumptive use of wildlife is recognized as producing positive impacts and is one of the goals of the management program. Harvest is regulated by hunting and trapping seasons that define the timing and duration of harvest opportunity, regulate the method of take (e.g., devices used), and establish individual harvest quotas (daily or seasonal).

Fisher management in New York, specifically the St. Lawrence River valley of Northern New York, provides an ideal opportunity for implementing an experimental regulatory action to explore management uncertainties and improve fisher harvest management. Currently, the fisher trapping season in New York is October 25th to December 10th. Trappers have requested that the NYSDEC reevaluate the fisher trapping season structure. The impetus of this request is the perception among trappers that a season closing date of December 10th is too early and that fisher pelts are not as valuable during this time period as they would be later in the winter when the fur is of higher quality (A. MacDuff, NYSDEC, pers. comm.). Maintaining the duration of the current trapping season by opening on a later date and closing on a later date is not feasible because the raccoon (*Procycon lotor*) and skunk (*Mephitis mephitis*) season would still need to open by October 25th to provide trappers an opportunity to take these species before they become dormant. If trapping is permitted for raccoon, skunk, and fox; trappers will likely catch fisher while targeting these species. The management dilemma is as follows:

Extending the closing date of the fisher trapping season would increase the overall fisher trapping season and thus harvest opportunity. It is unknown if, and how, an increase in harvest opportunity would affect actual harvest. It is possible that increased opportunity will greatly increase overall harvest. It is also possible that an extended season might only produce a limited increase in harvest. Factors influencing this process include trapper motivation, trapper time availability, trapper access to trapping locations, and seasonal changes in fisher behavior. This uncertainty can be categorized as an issue of "partial controllability" for managers (Williams 1997).

It is also unknown if fishers are more vulnerable to trapping mortality later in the winter. This uncertainty can be categorized as an issue of "structural uncertainty" (Williams 1997). Male river otters (*Lontra canadensis*) in New York and Massachusetts appear to be more vulnerable to harvest as the harvest season progresses into late winter. This observation has been attributed to seasonal behavioral changes in males (Chilelli et al. 1996). Male fishers are known to experience seasonal behavioral changes similar to male river otters (Arthur et al. 1989). Whether this affects harvest vulnerability is unknown. Similarly, changes in behavior related to increases in metabolic demands associated with decreased ambient temperature are unknown. If such behavior changes occur, it would be reasonable to assume that capture vulnerability would also increase.

Management actions could be postponed until further information is provided through research such as controlled trap grid studies to evaluate temporal trends in capture vulnerability, or trapper surveys to predict responses in trapping effort to regulatory changes. Such investigations would be costly. In the absence of research-derived data, the management options are to (1) leave the current season structure in place or (2) extend the trapping season by extending the closing date. However, key uncertainties affect the ability of the NYSDEC to proceed with an informed management policy. The limiting uncertainty is about the effect extending the trapping season has on fisher harvests.

To understand this relationship, we must explore the elements that contribute to the fisher catch. Catch is a function of the rate of capture given a unit of effort. The measure *Catch per Unit Effort* is assumed to be proportional to average population size as follows:

$$\frac{\#Captures}{Effort} \propto \overline{N}$$

The number of captures is a function of (1) the total population size, (2) the probability of capture given a unit of effort, and (3) the total capture effort. Effort, as it relates to fisher harvest, can be measured as trap nights, where a trap night represents one trap, capable of capturing a fisher, deployed for one night. To predict the catch that would occur as fisher harvest opportunities are extended later into the winter, we need to understand the temporal relationship of effort and capture vulnerability during the time periods under consideration. In the situation presented, however, there are limiting uncertainties that prevent this. Specifically, these uncertainties can be characterized in the following questions:

1) Does an increase in harvest opportunity result in an equivalent increase in harvest effort?

2) Does fisher vulnerability to harvest differ during late winter (Dec 10 – Jan 10) in comparison to the traditional season (Oct 25 – Dec 10)?

Both questions contribute to the larger uncertainty regarding response in cumulative harvest to management actions. Under the traditional management routine (e.g., harvest permitted from October 25 – December 10), the cumulative harvest can be viewed as the product of the cumulative harvest effort during this period and the probability of capture given a unit of effort. Furthermore, the cumulative effort is the product of the mean individual effort of a trapper and the total number of active trappers. Under the alternative management scenario, these functional relationships continue to persist. Should the value of any of these variables (mean individual effort, number of active trappers, or mean probability of capture) differ between the traditional season (October 25 – December 10) and the additional harvest opportunity provided by the management action (December 11 – January 10), the function that composes the total cumulative harvest could be altered. Therefore, to examine the uncertainty regarding what effect a management action that provides additional harvest opportunities has on actual harvest, we must examine each of the variables that contribute to the cumulative harvest.

Methods

The key uncertainties identified as impediments to fisher management in northern New York were characterized as model statements incorporated into sets of competing hypotheses. The three hypotheses sets are as follows:

Hypotheses set one:

$$HO: \tau_{\mathrm{t}1Y_i} = \tau_{t2Y_i}$$

$$HA: \tau_{t1Y_i} \neq \tau_{t2Y_i}$$

where $\tau_{\mathrm{ti}Y_i}$ is the average number of active trappers, per six-day period, during time interval ti, and nested within year Y_i. t1 is defined as the traditional trapping season between October 25th and December 10th, and t2 is defined as the extended portion of the trapping season between December 11th and January 10th.

Hypotheses set two:

$$HO: \xi_{t1_{Yi}} = \xi_{t2_{Yi}}$$

$$HA: \xi_{t1_{Yi}} \neq \xi_{t2_{Yi}}$$

where $\xi_{ti_{Yi}}$ is the average individual trapping effort, per six-day period, during time interval ti, and nested within year Y_i. t1 is defined as the traditional trapping season, and t2 is defined as the extended portion of the trapping season.

Hypotheses set three:

$$HO: \theta_{t1_{Yi}} = \theta_{t2_{Yi}}$$

$$HA: \theta_{t1_{Yi}} \neq \theta_{t2_{Yi}}$$

where $\theta_{ti_{Yi}}$ is the average catch per unit effort (e.g. CPUE= (# fisher captured / total trap nights)*1000)), per six-day period, during time interval ti, and nested within year Y_i. t1 is defined as the traditional trapping season, and t2 is defined as the extended portion of the trapping season.

A regulatory change was implemented for the 2006-2007 trapping season (New York Codes, Rules, and Regulations Section 6.4 ENV-35-06-00010-A). This action established a temporary (three-year) experimental, 31-day extension to the existing trapping season in 6,625 km² of northern New York, resulting in an experimental season extension. This area represents less than 25% of the entire area in which fisher harvest occurs in New York. Additional conditions of the management action included: (1) all participating trappers must acquire a special additional permit, thus allowing for precise enumeration of participation rates; (2) all participating trappers must maintain a daily trapping log to record trapping effort and associated catch, this log must be provided to the state agency at the conclusion of each season; (3) the skinned head, or jaw, of each trapped fisher must be surrendered to the state agency, thus enabling the collection of age-structure data; and (4) the regulation expires after the third season, effectively forcing a readdress of this management issue.

Each trapper was provided with a logbook to record information, including the date traps were set, the Wildlife Management Unit location of each trap, the total number of traps set each day, and the associated catch in each trap. The mandatory logbooks also allowed enumeration of the number of active trappers each day, their trapping effort (measured as trap nights), and the catch associated with these efforts. These data were pooled into thirteen six-day periods for analysis. These data were collected during both the 2006/2007 and 2007/2008 trapping seasons.

The final hypotheses set relies on the underlying assumptions inherent of catch-effort models including (Skalski et al. 2005, Williams et al. 2002, Quinn and Deriso 1999):

1) The population is closed with the exception of harvest.
2) Harvest and effort are accurately reported.
3) Each animal has an equal and independent probability of capture.
4) Capture vulnerability is constant within a sampling period.

It is reasonable to assume that the fisher population is approximately closed. Fisher have relatively small home ranges (19 to 79 km^2 for males) compared to the overall study area of 6,625 km^2 (Kelly 1977, Jones 1991), minimizing the influence of movement into or out of the study population. In addition, parturition in fisher occurs in later spring, hence, immigration in the form of births does not occur during the time in which harvest is occurring (Powell 1993).

The assumption of accurate harvest and effort reporting is reasonable given that all trappers were provided a standardized logbook, with instructions, to report these data. While we cannot ensure that each trapper reported these data with complete accuracy, we assume that any bias this introduces is consistent across years.

We can reasonably assume that all animals have an equal probability of capture regardless of sex or age. A recent survey of trappers in the Northeastern United States found that 78% of trappers use lethal, body-gripping traps to harvest fisher (AWFA 2005). Over 97% of fisher that encounter these devices are killed (AFWA 2009). Thus, individuals do not have an opportunity to learn trap avoidance or affinity behaviors. Similarly, trappers cannot release smaller or less valuable animals as all animals caught in these traps are killed. Furthermore, according to pelt-sealing databases in New York, sex ratios for fisher are fairly constant through the duration of the season, indicating no trends in capture vulnerability as a function of sex (New York State Department of Environmental Conservation 2006).

The final assumption asserts that capture vulnerability is constant within a sampling period. This issue is the basis of the third hypotheses set and, as such, we further clarify this assumption to be that capture vulnerability within the traditional trapping season is constant, and across both years, is constant. Similarly, we assume that capture vulnerability within the extended portion of the trapping season, and across both years, is constant. We make no assumption regarding equal capture vulnerability between the traditional and extended portions of the trapping season, leading to the hypotheses presented. Rather, we assume that within a year, population size is constant and that any difference in the catch-per-unit-effort ratio between season portions is a function of unequal capture vulnerability.

All hypotheses were tested using a two-sided F-test (α=0.05). Annual variation was controlled for by treating t_i (traditional portion of season or extended portion of season) as a nested variable within the variable Y_i (year). All models were constructed in SAS (SAS Institute, Carey, IN, USA) using the PROC GLM procedure.

Results

The experimental management action was first implemented in the 2006/2007 trapping season. During this season, the number of active trappers in each six-day pooled interval ranged from 68 to 105 trappers (Figure 1.1).

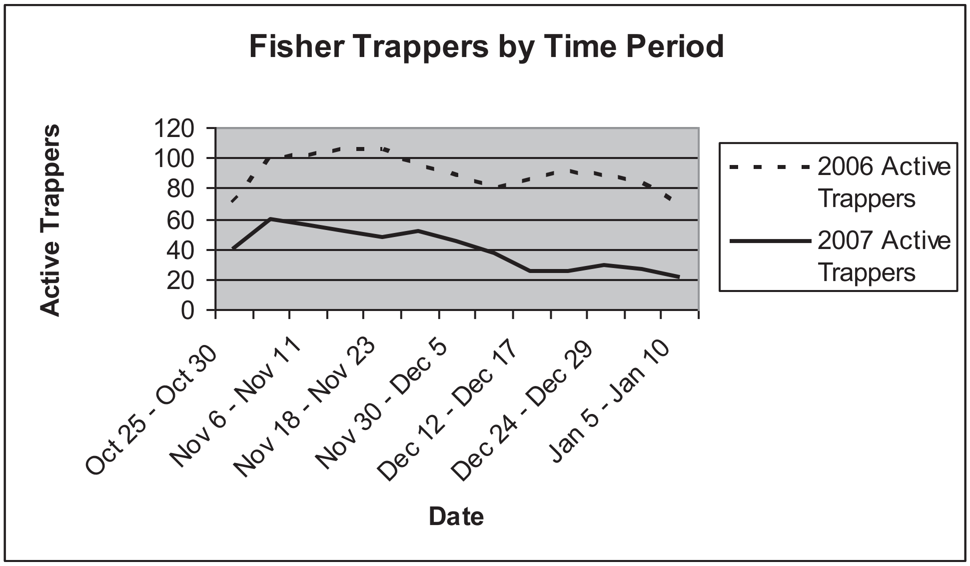

Figure 1.1 Number of active trappers during the 2006/2007 and 2007/2008 fisher trapping seasons by six-day interval.

During the 2007/2008 trapping season, the number of active trappers in each interval ranged from 21 to 60 trappers (Figure 1.1). The mean number of active trappers was significantly higher ($p = 0.003$) during the traditional portion of these seasons than during the extended portion of these seasons.

During the 2006/2007 trapping season, the mean individual trapping effort in each interval ranged from 48.2 to 77.5 trap nights (Figure 1.2).

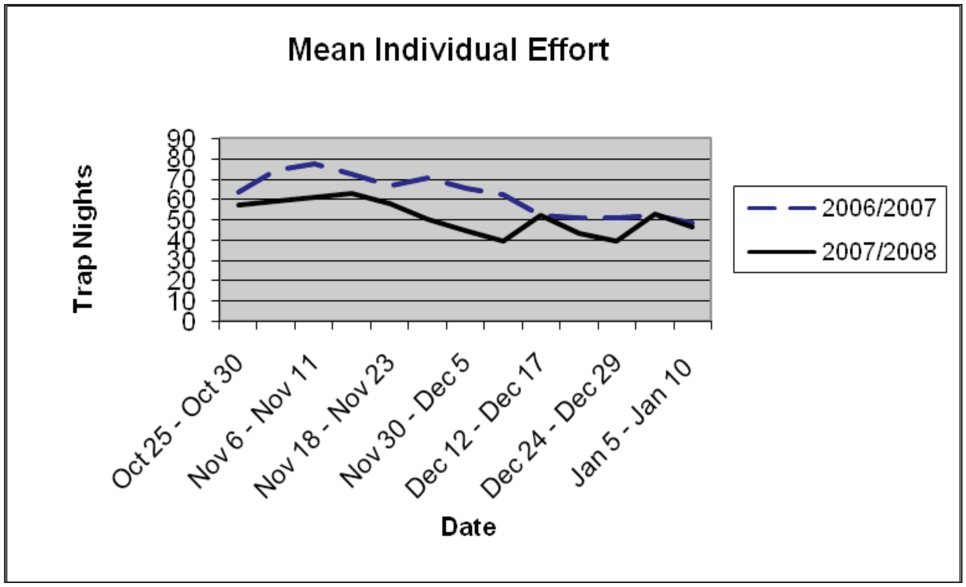

Figure 1.2 Mean individual trapping effort, in trap nights, during the 2006/2007 and 2007/2008 trapping seasons.

During the 2007/2008 trapping season, the mean individual effort in interval ranged from 39.4 to 63 trap nights (Figure 1.2). The mean individual trapping effort was significantly higher ($p < 0.0001$) during the traditional portion of both seasons than during the extended portion of these seasons.

During the 2006/2007 trapping season, the catch per unit effort in each interval ranged from 0.52 to 1.53 fishers per 1,000 trap nights (Figure 1.3).

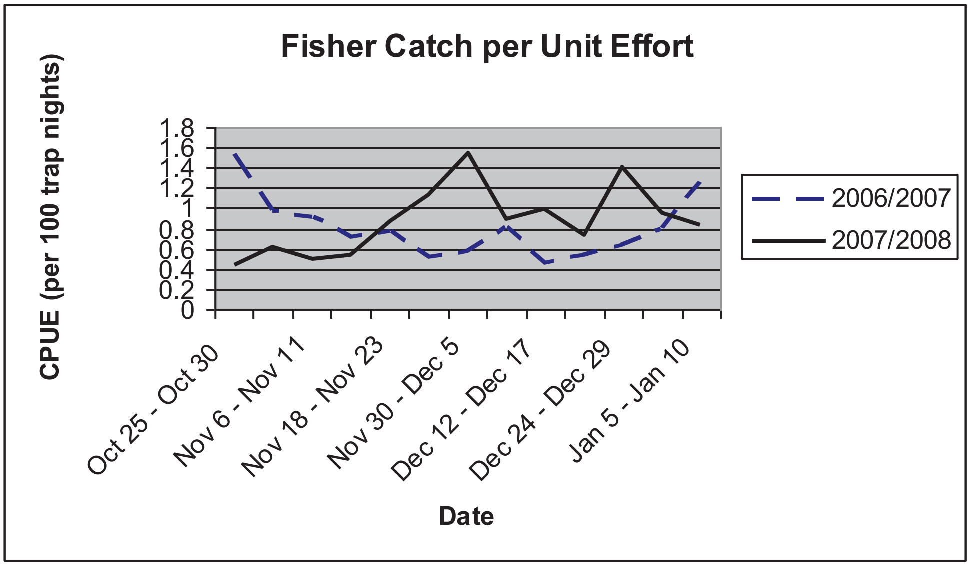

Figure 1.3 Fisher catch per unit effort during the 2006/2007 and 2007/2008 trapping seasons.

During the 2007/2008 trapping season, the catch per unit effort in each interval ranged from 0.44 to 1.55 fishers per 1,000 trap nights (Figure 1.3). There was no significant difference (p=0.5562) in mean catch per unit effort between the traditional portion of the seasons (October 25 to December 10) and the extended portion of the seasons (December 11 to January 10). Similarly, there was no significant difference (p=0.4264) between the mean catch per unit effort during the 2006/2007 season and the 2007/2008 season.

Discussion

The use of the experimental season process and the associated record-keeping requirements facilitated the investigation of management uncertainties in an efficacious and robust manner. While the implemented management action was specified as a three-year action, we focused our analysis on data yielded from the first two years due to time constraints. We found that capture vulnerability did not change within each season. The mean catch per unit effort was not significantly different during the extended portion of these seasons (December 11 to January 10). In Manitoba, Canada, fisher activity and movement was found to increase significantly during the breeding season (Leonard 1980, Leonard 1986). Breeding activity in Manitoba, Canada, peaked in the month of March (Leonard 1986). Similarly, in Maine, USA, breeding activity, as determined from carcass examinations, was recorded between late February and mid-April (Coulter 1966, Wright and Coulter 1967). While temporal patterns in breeding behavior have not been investigated in New York, parturition dates have been estimated for New York fisher and these suggest that reproductive cycles in New York are similar to these other regions (Eadie and Hamilton 1958). Given that the harvests in this study ceased by January 10th, it is likely that behavioral changes attributable to breeding behavior had not begun. Similarly, if changes in metabolic demands related to ambient temperature produce behavioral changes, these changes did not appear to affect capture vulnerability. A similar mustelid of the same genus, the short-tailed weasel (*Mustela erminea*), was found to produce a sufficient amount of surplus heat from typical movements to compensate for the energetic costs of thermoregulation in winter (Sandell 1989). While the energetics of the fisher have not been examined in detail, it is possible that this species exhibits similar patterns. Regardless of the underlying causes, we were successful in addressing our limiting structural uncertainty regarding capture vulnerability relevant to additional harvest opportunities later in the winter. Our results suggest that the extension of harvest opportunities until January 10th will not result in increased fisher capture vulnerability.

The number of active trappers, and the mean individual harvest effort expended by these trappers, was significantly less during the extended portion of both seasons. There are many factors known to motivate trapper effort including available time (e.g., vacation, work schedules, etc.), weather, access to trapping lands (e.g., some roads are not maintained during the winter, some territory might only be accessible with snowmobiles or skis after snowfall, difficulty obtaining permission to access private land), and economic returns (Siemer et al. 1994). In the present situation, increased access to remote areas following consistent snowfall was suggested as a potential catalyst for increased trapping pressure. Ultimately, we found that trappers did take advantage of the increased harvest opportunity. This opportunity was met with decreased relative effort, both in terms of participating trappers and actual trapping effort per trapper, than during the traditional harvest season. This information directly addresses the limiting uncertainty of partial controllability relating to whether increased harvest opportunity would result in a proportional increase in actual harvest effort.

Catch-per-unit-effort data is appealing due to the ease of which these data can be collected. However, the underlying assumptions of catch per unit effort to infer population status should not be overlooked. The assumption of a closed population could be violated if a trapping season extended into the birthing season or if the study area was sufficiently small to allow significant immigration or emigration movements. The addition of individuals through births or immigration would result in a positive bias in the index. The loss of individuals through emigration out of the study area could lead to a negative bias in the index, suggesting a population-level decline that is nonexistent.

Errors in effort or harvest reporting could also compromise the utility of the index. Under-reporting of harvest could occur if trappers were hesitant to report their successes, leading to an artificially low catch-per-unit-effort value. Over-reporting is not likely as all fisher pelts must be sealed prior to sale in New York and seals are only issued after a catch is made. However, other furbearer species do not require pelt sealing, such as raccoon and muskrat (*Ondatra zibethicus*), and thus, this scenario should not be disregarded for these species.

The assumption of equal probability of capture among individuals can be violated for furbearer species. While fisher are fully grown by their first winter (Powell 1993), this is not consistent among furbearer species. For example, beaver (*Castor canadensis*) continue to grow until 4 or 5 years of age (Hill 1982). Because of this, during any given trapping season, beaver sizes may range from 5 kg to >30kg (Hill 1982). Beaver traps can be made to select larger animals (AWFA 2008). The effect of removing those individuals that are easily caught is that the remaining population will be composed of individuals that are more difficult to capture. This could lead to a decrease in catch per unit effort that is simply an artifact of the remnant population being composed of those individuals less likely to be captured, as opposed to a true decline in the overall population.

The assumption of constant capture vulnerability could be violated if seasonal behavioral changes were present within the sampling period. This issue was the impetus of our final hypotheses set. We made the assumption that capture vulnerability was constant within the defined sub-periods of the traditional and extended portions of the trapping seasons to facilitate a comparison of capture vulnerability between these sub-periods. However, this introduces another assumption that the overall population size is constant within a trapping season, and that any change in catch per unit effort within a trapping season between the sub-periods of the season is a function of capture vulnerability. Gould and Pollock (1997) demonstrate, through simulation, that a violation of an assumption of the constant capture vulnerability can bias the catch per unit effort estimates. In fact, this potential source of bias is one of the arguments the authors use to promote the robust design framework for catch-effort parameter estimation as this approach makes no assumption of constant capture vulnerability (Gould and Pollock 1997).

Capture vulnerability could also vary between years for species that are cyclic, or dependent upon cyclic prey species. While fisher are considered a generalist species not subject to population cycles, other furbearers do experience this phenomenon. Canada Lynx (*Lynx canadensis*) populations are known to cycle in response to fluctuations in snowshoe hare (*Lepus americanus*) densities in part of their range (Anderson and Lovallo 2003). It is reasonable to assume that during periods of low prey abundance, Canada lynx would be more vulnerable to capture. Jakubas et al. (2005) suggested that capture vulnerability of American Marten (*Martes americana*) is strongly influenced by American Beech (*Fagus grandifolia)* mast failures and the effects of these failures on the small mammal populations that constitute the prey base of the species. The authors did note that the generalist nature of the fisher likely prevented this trend for this larger species. If capture vulnerability did vary between seasons, then the resulting catch per unit effort estimate would be biased accordingly.

Though the primary objectives of this study were to evaluate an issue of partial controllability related to increasing harvest opportunity later into the winter and an issue of structural uncertainty related to capture vulnerability later into the winter, the data collected can also be used to monitor relative abundance given the assumptions discussed above are maintained. Catch per unit effort is frequently used to monitor furbearer population trends (Dixon 1981). Strickland (1994) found catch per unit effort to be a reasonable indicator of population changes in fisher in Ontario. Similarly, a strong relationship between a CPUE survey and marten density has been demonstrated in Québec (Fortin and Cantin 1990). Chilelli et al. (1996) suggested that catch-per-unit-effort surveys have utility in measuring river otter population trends in the Northeast United States and Roberts et al. (2008) found catch per unit effort, coupled with bridge-sign surveys, to be a useful measure of river otter population trends in the Midwest United States. The pelt sealing requirement for fisher in New York facilitates an accurate enumeration of the harvests. This would facilitate the use of more robust methods, such as the removal methods that utilize cumulative catch estimates (Pollock et al. 1984, Bishir and Lancia 1996, and Gould and Pollock 1997). However, the sample-sizes we were able to acquire provided for high statistical power, using relatively simple techniques.

The experimental season framework allows for an empirical evaluation of response in trapping effort to changes in harvest opportunities. Furthermore, this investigative framework permits the simultaneous evaluation of both the structural uncertainty, a biological issue, and the partial controllability uncertainty, a social/human-behavior issue. These management uncertainties had limited the ability of the NYSDEC to maximize the positive impacts of the wildlife resource to the public. Hesitation to extend harvest opportunities was embedded in concern about how extensive the contribution of an extended season would be to the overall cumulative harvest and the relative composition of this contribution.

These data enhanced our understanding of fisher harvest management in northern New York. We focused on three key variables that influence the overall cumulative harvest of fisher: the number of individuals pursuing fisher, the amount of harvest effort expended by these individuals, and the rate of harvest relative to effort. Uncertainty regarding how these variables would respond to an increased harvest opportunity later in winter had limited viable management options. Addressing these issues via a traditional field research paradigm would have been extremely difficult. The question regarding capture vulnerability could have been investigated using field studies designed to imitate harvest efforts and associated captures. However, it is very unlikely that a similar sample size, 100,219 trap nights, could have been obtained. By integrating management actions with research objectives, we were able to acquire a more representative and substantial data set comprised of authentic harvest effort expended by actual trappers.

We learned that fisher do not appear to be more vulnerable to capture during the extended portions of the seasons than during the traditional portions. We also learned that harvest effort, both in terms of active trappers and mean individual effort, was significantly less during the extended portions of the New York seasons than during the traditional portions. Although we found no change in catch per unit effort between years, an indication that the relative abundance of fisher did not decline, these findings provide indications as to what management and regulatory actions may be most effective in controlling harvests if this should be desired in the future. Within the time period examined (October 25 – January 10), capture vulnerability can be considered relatively constant. Other factors may influence the temporal placement harvest opportunities, such as the simultaneous harvest of other furbearers, but our results suggest that consideration of capture vulnerability is not essential. Thus, managers should consider actions that will likely influence trapper effort, rather than redirecting effort, if the intended outcome is to impact cumulative harvest. While the duration of harvest opportunities can certainly influence the cumulative harvest, management should not assume that changes in opportunity will result in proportional increases in effort. Our investigations focused on providing additional opportunities, it is unknown if a similar response would be realized with a reduction of harvest opportunities. Similarly, it is unknown if a similar response can be expected in other regions of New York state with markedly different weather and land-use patterns. Regions of southern New York experience much less snowfall than northern New York. It is possible that this could influence harvest effort by influencing the means of access to trapping territories (e.g., motor vehicle or snowmobile).

Our findings suggest that fisher harvest opportunities in northern New York could be extended beyond the traditional closing date of December 10th, possibly as late as January 10. A monitoring effort, such as the catch-per-unit-effort logbooks, should be implemented to monitor the long-term effect of additional harvest on this population. In the event of population trend concern, it will be important to understand how to most effectively reduce harvest pressure. Therefore, we also suggest that the response in harvest effort to decreased harvest opportunity be examined in a limited area. Finally, we recommend that experimental seasons, coupled with appropriate sampling protocols, be utilized to examine similar management uncertainties for other furbearer species. This approach is well suited to address both biological and social uncertainties, both of which are plentiful in furbearer management.

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CHAPTER 2

Reducing Parametric Uncertainty in a Bobcat Population Model: A Bayesian Perspective.

Abstract

We wanted to learn about the population demographics of a previously unexploited bobcat (*Lynx rufus*) population in order to construct a projection matrix model useful for predicting population growth rates and the effects of varying mortality rates on these predicted growth rates. However, acquiring reliable demographic parameter estimates for elusive carnivores, such as bobcat, can be problematic due to difficulty in obtaining adequate sample sizes. A management action was designed and implemented to facilitate the collection of age-structure data to directly assess survivorship rates. The survivorship parameters were estimated using a Bayesian framework that allowed the incorporation of external data and for updating of the parameter as additional data became available. The posterior estimates of survivorship were sampled from and incorporated into simulations of a projection matrix model. The distribution of projected growth rates determined from the matrix model simulations showed decreasing variation in projected growth rates as parametric uncertainty was reduced. Through this Bayesian framework for updating the parameter estimates, we refined our estimate of the projected population growth rate from $\lambda=0.93$ ($\sigma^2=0.28$) to $\lambda=1.14$ ($\sigma^2=0.014$).

Keywords: age structure, estimation, Bayesian, matrix models, survivorship

Introduction

Bobcats (*Lynx rufus*) are one of 14 mammal species managed as furbearers in New York. Management of furbearer species, with the exception of federally endangered species, is the mandated responsibility of the New York State Department of Environmental Conservation (NYSDEC). The objectives of furbearer management in New York are to (1) maintain sustainable populations, (2) allow sustainable harvests in areas with secure populations and (3) foster range expansion into all suitable habitats (MacDuff 2009).

Bobcat harvests had not occurred in the Oswego-Delaware Hills region of southern New York for over 23 years by 2006. Wildlife managers had considered reinstituting a harvest component into the long-term management program in this area. The NYSDEC was hesitant to take this action, primarily because of several biological and management uncertainties. The key uncertainty was whether this population was growing, given the realized rates of reproduction and survivorship, sufficiently to sustain additional mortality in the form of harvest.

A population model could be a useful tool to synthesize demographic information, characterize biological uncertainties, and explore projected population growth rates under differing parameter values. A model could be used to explore the predicted effect of various management actions on the population, such as increasing mortality through regulatory changes that facilitate increased harvests. Matrix population models have been frequently used to characterize the population dynamics of wildlife species (Caswell 2001). These models are particularly well suited for species that exhibit age or life stage structured population dynamics. Uncertainty can be incorporated into model predictions by simulating multiple realizations of the model where parameter values are drawn from a distribution that reflects the parameter estimate and variance. Because the parameter values vary with each iteration, the resulting model output (i.e. λ, representing the change in relative population abundance) will also vary; producing a distribution of projected output values. The magnitude and degree of variation in model outputs can be used when considering the confidence placed in the model performance and the likely consequences of management actions. For example, the risk of population decline, or of a population breeching a numeric threshold, can be viewed in terms of probabilities (i.e., there is an X% probability that the population will decline under a given management scenario). Thus, under this approach, the degree of variation in model predictions can help characterize the degree of uncertainty likely to be encountered when management actions are employed. Therefore, in order to enhance the utility of models, it is desirable to reduce uncertainties that have management consequences, thus reducing variation in model predictions.

Harvest dependent data, such as the age structure of the population, fecundity counts, or catch per unit effort were not available at the beginning of this study for this bobcat population. The elusive nature of bobcats prohibits the effective use of traditional visual observational studies such as those used for assessing waterfowl or ungulate populations (Skalski et al. 2005). In the absence of complex field studies, the development of a useful population model for bobcats will necessitate some form of harvest-dependent data. This presents a dilemma, as the information needed to explore harvest effects on the population cannot be obtained without initiating a harvest. A limited, short-term experimental harvest may facilitate the collection of data useful in constructing a population model. However, even after harvest-dependent data are made available, parameter estimation for carnivores, such as bobcats, can be challenging due to limited sample sizes. While it is possible to obtain estimates of age-specific reproduction and mortality, the associated variances of these estimates may be quite wide when sample sizes are limited (i.e. using only those data obtained from the specific population under study). These large variances, when propagated in a simulation model, lead to high uncertainty in the model output. Therefore, reducing parametric uncertainty in the model is desired to reduce to subsequent uncertainty in model outputs.

Bayesian inference provides a formal mechanism for combining new information with existing information, thus maximizing the utility of the available data. In effect, the parameter estimate derived from existing information, the prior estimate of the parameter, is updated with the inclusion of new data to define the posterior distribution. This ability to augment sparse data sets with external information has proven useful for estimating the demographic parameters of other elusive species, such as salamanders and bears (Brooks et al. 2000, McDonald and Fuller 2001, Lindstöm et al. 2010). Our objective is to create a population model for the bobcat population that incorporates biological uncertainty as parametric uncertainty. We intend to use Bayesian inference to update parameter estimates by utilizing both existing and empirical data sources. In this, we hope to reduce parametric uncertainty in our population model and create a process that will lead to a reduction in uncertainty in the system and promote continued learning and understanding of bobcat biology and management practices.

Methods

We initially developed a Leslie matrix projection model using parameters estimated from the literature (Leslie 1945, Leslie 1948, Caswell 2002). Bobcat populations are structured by age, as fecundity rates, and possibly survivorship, differ by age class (Leonard 1986, Rolley 1987). Bobcat mortality rates are fairly constant for adult animals (Litvaitis et al. 1987, Chamberlain et al. 1999). Reproductive rates for bobcats are lower for yearlings, but constant once bobcats reach the adult age class in their second year (Crowe 1975). Thus, the bobcat life cycle can be characterized as stage specific (Figure 2.1).

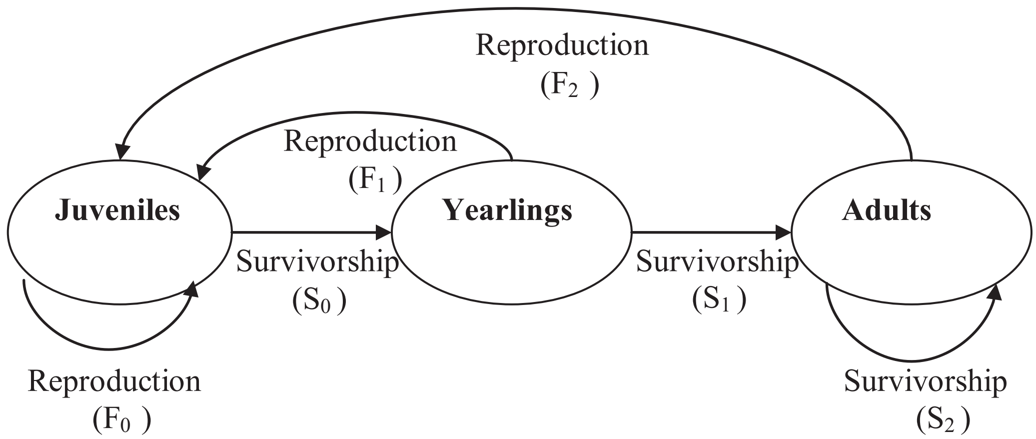

Figure 2.1: Stage-specific life-cycle diagram for the bobcat.

The associated projection matrix model is as follows:

$$\begin{pmatrix} n_{0,t+1} \\ n_{1,t+1} \\ n_{3,t+1} \end{pmatrix} = \begin{pmatrix} F_0 & F_1 & F_2 \\ S_0 & 0 & 0 \\ 0 & S_1 & S_2 \end{pmatrix} \begin{pmatrix} n_{0,t} \\ n_{1,t} \\ n_{3,t} \end{pmatrix}$$

where F_j is the fecundity of stage-class j, S_j is the survivorship of stage age class j, N is the number of individuals at time t, and t is year since the inception of the model.

Mean litter size for adult bobcats has been estimated to range from 2.5 in Nova Scotia to 3.5 in Utah (Parker and Smith 1983, Gashwiler et al. 1961). Anderson (1987) surveyed 21 bobcat studies and found an average litter size to be 2.7 (SE = 0.09) kittens per litter. While yearling reproduction may occur (estimates of litter size have been based on corpora lutea counts), it is estimated to be 15-19% less than reproduction in adults (Parker and Smith 1983, Rolley 1985). However, there is debate as to whether these counts provide an accurate estimate of fecundity in yearlings (Knick 1990). Stys and Leopold (1993) question whether yearling females possess the necessary hormonal secretions to maintain pregnancy. If a yearling lacked this ability, it is possible that corpora lutea counts would still be present from ovulation, despite unsuccessful gestation. Wanting to maintain a conservative estimate of population growth, we assumed that successful reproduction did not occur until after the second year. Given that mean fecundity is measured as a positive, continuous value, we defined the fecundity parameters as gamma distributed ($\Gamma(\alpha, \beta)$, $\mu = \frac{\alpha}{\beta}$, $\sigma^2 = \left(\frac{\alpha}{\beta^2}\right)$) with hyper-parameters α and β where α is the shape parameter and β is the inverse scale parameter of the gamma probability distribution function $p(\theta) = \frac{\beta^{\alpha}}{\Gamma(\alpha)} \theta^{(\alpha+1)} e^{-\beta/\theta}$, $\theta > 0$, $\alpha > 0$, $\beta > 0$. We calculated the α and β parameters of the gamma distribution by solving for these parameters given $\mu = 2.7$ (SE = 0.09) as determined from existing literature. However, projection matrixes characterize only the female component of the population. Thus, in the model, F_0 and F_1 will be fixed and set equal to 0, while F_2 will be characterized as 50% of the value of a random variable defined by a gamma distribution with hyper-parameters α=4.05 and β=3 and denoted as $\Gamma(\alpha=4.05, \beta=3)$.

Survival rates for bobcats vary considerably regionally, often reflecting variation in harvest pressure. An un-harvested population in Illinois was estimated to experience an 86% annual survival rate (Nielson and Woolf 2002). Similarly, an un-harvested population in Idaho was estimated to experience an adult survival rate of 97% (Crowe 1975). Conversely, Fuller et al. (1995) found that annual survival in a heavily exploited population in Massachusetts was as low as 49%. Juvenile mortality rates may not vary significantly from adults (Knick 1990, Parker and Smith 1983). However, Rolley (1985) found that juvenile survival rates in Oklahoma were only 30% whereas adult survival rates were 53-66%. Given the wide range of variability in survivorship estimates, we initially defined the survivorship for year one to be uniformly distributed and bound by the extreme values found in the literature. Thus, survivorship in the model is defined as being S = U(0.30,0.97).

The resulting matrix model may now be characterized as follows:

$$\begin{pmatrix} n_{0,t+1} \\ n_{1,t+1} \\ n_{2,t+1} \end{pmatrix} = \begin{pmatrix} 0 & 0 & (\Gamma(\alpha = 4.5, \beta = 3)) * 0.5 \\ U(0.3,0.97) & 0 & 0 \\ 0 & U(0.3,0.97) & U(0.3,0.97) \end{pmatrix} \begin{pmatrix} n_0, t \\ n_1, t \\ n_2, t \end{pmatrix}$$

This model is considered reflective of our knowledge of this bobcat population prior to any investigations as it only comprises information extracted from the literature. The predictions from this model are examined by performing ten thousand simulations of the model where, for each simulation, the reproductive or survivorship parameter values were randomly drawn from a distribution specified by the hyper-parameters. The associated dominant eigenvalue, termed lambda (λ), was recorded for each simulation of the model. The λ values represent the projected population growth rate for the specific simulation of the model and the distribution of the λ values produced by all the simulations is reflective of the uncertainty in the underlying model parameters.

A sensitivity analysis (Caswell 1996, 2001), using Tuljapurkar's approximation for stochastic elements (Tuljapurkar 1990), reveals that the variation in the survivorship parameters has the greatest influence on the variation in model outputs (Figure 2.2).

$$\begin{Bmatrix} F_0 & F_1 & F_2 \\ S_0 & 0 & 0 \\ 0 & S_1 & S_2 \end{Bmatrix} \Rightarrow \begin{Bmatrix} 0 & 0 & 0.228 \\ 0.297 & 0 & 0 \\ 0 & 0.35 & 0.604 \end{Bmatrix}$$

Figure 2.2: Stochastic sensitivities of initial projection matrix model.

Additional data on bobcat fecundity is not likely to reduce uncertainty significantly beyond what is indicated in published accounts as this rate is fairly constant across regions (Parker and Smith 1983, Rolley 1985). However, additional information on the survivorship parameters could be useful as these rates vary considerably in literature accounts as reflected in our initial parameter estimates. Thus, refinement of the survivorship parameter estimates to better reflect the rates realized in this population is necessary to reduce uncertainty in these parameters, represent the dynamics of this particular population, and reduce the uncertainty of projections produced by models constructed with these parameters.

To refine the model parameters, we designed a short-term experimental, regulatory action to facilitate the collection of harvest age structure data. This unique regulatory action initiated an experimental trapping season during 2006-2007 (New York Codes, Rules, and Regulations Section 6.4 ENV-35-06-00010-A). This action established an experimental trapping season in a ~4,900 km² area of southern New York, from October 25th to February 15th, for three years. Additional conditions of this management action included: (1) all participating trappers must obtain a special additional permit, thus allowing for precise enumeration of participation rates, (2) all participating trappers must maintain a daily trapping log to record trapping effort and associated catch, this log was required to be provided to the state agency at the conclusion of each season, (3) the carcasses of each trapped bobcat must be surrendered to the state agency, and (4) the regulation was scheduled to expire after the third season, effectively forcing the agency to readdress this management issue upon completion of the experimental season.

The age structure of harvested bobcats was determined using cementum annuli counts of the canine tooth (Crowe 1975). All aging analysis was conducted by Matson's Laboratory (Matson's Laboratory, Milltown, Montana, USA). We assumed that the demographics of the harvest approximated the demographics of the population and assumed no age-specific harvest bias.

Survivorship was estimated from the age structure of the harvest. By assuming that recruitment (n_0) and survivorship are constant, the expected harvest in age-class x (H_x) can be written as $E(H_x) = pS^x n_{x=0}$ where p is the probability of being harvested and S is the probability of an individual surviving the interval from one age class to the next (Skalski et al. 2005). Instantaneous survivorship is classically estimated using linear regression applied to the model $\log(H_x) = \acute{\eta} + \ln(S)\, x$, where ή is treated as a nuisance parameter that contains both p and N_0 as $\log(pN_0)$. However, we estimate S using a Bayesian framework and assign a prior to both the ή and the S parameters. An uninformative prior was assigned to the ή parameters for both year one, and year two, as this parameters is treated as a nuisance parameter. An informative prior was utilized for the S parameter as explained below.

The survivorship parameter (S) is estimated using a Bayesian framework at year one given the data available at that time. The posterior estimate is then incorporated into the projection matrix by drawing values from this distribution to serve as the parameter estimate for S for each simulation of the projection matrix. This survivorship parameter is again estimated at year two, given the additional information made available, and the updated estimate of this parameter is incorporated into the projection matrix in an identical manner, producing a series of simulations utilizing the updated parameter estimates.

For year one, the prior distribution of the S parameter was specified as a uniform distribution bound by the extreme values found in the literature. This prior distribution was reflective of our knowledge of this parameter given the available data and is informative because it reduces the parameter space from all possible values, i.e. 0.0 to 1.0, to 0.3 to 0.97. The age structure of the harvest at year one formed the likelihood used to update this prior and estimate the posterior distribution of S. The posterior distribution was estimated using Markov Chain Monte Carlo algorithms utilizing the Gibb's sampler. A single Markov chain was initiated and allowed to run 1,000,000 iterations. The first 500,000 values of the chain were discarded and the remaining chain was thinned by selecting one value in fifty. The Markov Chain Monte Carlo algorithms were performed in WinBUGS through program R™.

For the second year, the posterior distribution of the S parameter was estimated using an informative prior distribution derived from the posterior distribution of this parameter at the conclusion of year one. This prior was specified as a beta distribution $p(\theta) = \frac{\Gamma(\alpha+\beta)}{\Gamma(\alpha)\Gamma(\beta)}\theta^{\alpha-1}(1-\theta)^{\beta-1}$, $\theta \in [0,1]$, where $E(\theta) = \frac{\alpha}{(\alpha+\beta)}$ and $\sigma^2 = \frac{\alpha}{(\alpha+\beta)^2(\alpha+\beta+1)}$. The age structure of the harvest at year two formed the likelihood used to update this prior and estimate the posterior distribution of S for year two. The posterior distribution was also estimated using Markov Chain Monte Carlo algorithms utilizing the Gibb's sampler and were truncated and thinned as described previously.

Projection matrixes were constructed at three intervals: (1) prior to collecting any empirical information, (2) after updating the survivorship parameter with the data made available after the first year of experimental management action, and (3) after updating the survivorship parameter with the additional data made available following the second year of the management action.

The posterior distribution of S at year one was sampled from to incorporate into a matrix projection model for year one. This projection matrix model was considered reflective of our knowledge of the population dynamics of this population at this time and was characterized as

$$\begin{pmatrix} n_{0,t+1} \\ n_{1,t+1} \\ n_{2,t+1} \end{pmatrix} = \begin{pmatrix} 0 & 0 & (\Gamma(\alpha = 4.5, \beta = 3)) * 0.5) \\ \psi_{t=1} & 0 & 0 \\ 0 & \psi_{t=1} & \psi_{t=1} \end{pmatrix} \begin{pmatrix} n_0, t \\ n_1, t \\ n_2, t \end{pmatrix}$$

where each $\psi_{t=1}$ is a random value drawn from the posterior distribution of S at time t=1 and $\Gamma(\alpha=4.05, \beta=3)$ is a random value drawn from a gamma distribution with parameters $\alpha=4.05$ and $\beta=3$ as no additional information was gathered on reproductive potential.

The posterior distribution of S at year two was sampled from to incorporate into a matrix projection model for year two. This model was considered reflective of our knowledge of the population dynamics of this population at this time and was characterized as

$$\begin{pmatrix} n_{0,t+1} \\ n_{1,t+1} \\ n_{2,t+1} \end{pmatrix} = \begin{pmatrix} 0 & 0 & (\Gamma(\alpha = 4.5, \beta = 3) * 0.5) \\ \psi_{t=2} & 0 & 0 \\ 0 & \psi_{t=2} & \psi_{t=2} \end{pmatrix} \begin{pmatrix} n_0, t \\ n_1, t \\ n_2, t \end{pmatrix}$$

where each $\psi_{t=2}$ is a random value drawn from the posterior distribution of S at time t=2 and $\Gamma(\alpha=4.05, \beta=3)$ is a random value drawn from a gamma distribution with parameters $\alpha=4.05$ and $\beta=3$.

Ten thousand simulations of each projection matrix model were run and the associated dominant eigenvalue (λ) was recorded for each simulation. The resulting distribution of simulated projections was recorded for each model.

Results

During the 2006-2007 trapping season, 25 bobcat carcasses were collected from the experimental trapping zone. These bobcats ranged in age from 0 to 12 years (Figure 2.3).

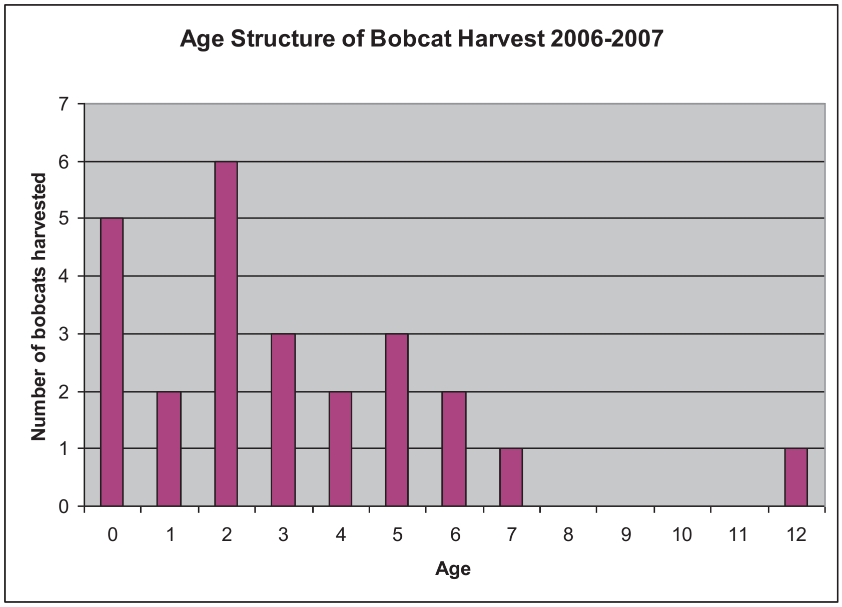

Figure 2.3: Age structure of bobcats collected in 2006-2007

During the 2007-2008 trapping season, 15 bobcat carcasses were collected from the experimental trapping zone. These bobcats ranged in age from 0 to 9 years (Figure 2.4).

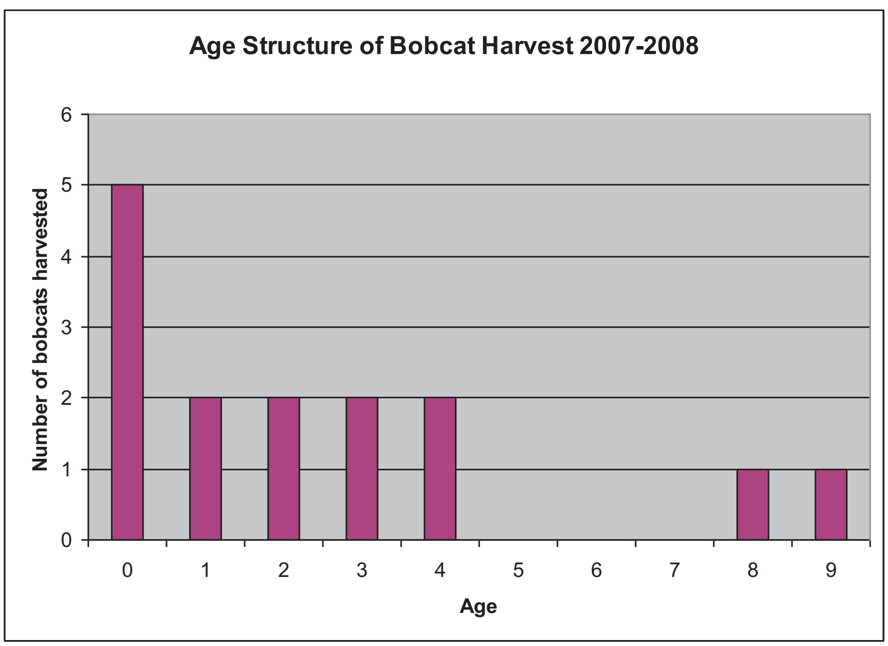

Figure 2.4: Age structure of bobcats collected in 2007-2008.

For both years, the fecundity parameters were drawn from a fixed distribution and thus, no updating of this parameter occurred.

For year one, the prior distribution of the survivorship parameter consisted of a semi-informative uniform distribution U(0.3,0.97) constructed from the range of plausible values reported in the literature. The posterior survivorship estimate, derived by combining this prior with the empirically derived likelihood from the age structure developed from captured animals for year one, yielded a Bayesian derived posterior estimate of survivorship for year one with notably less variation than the prior distribution (Figure 2.5).

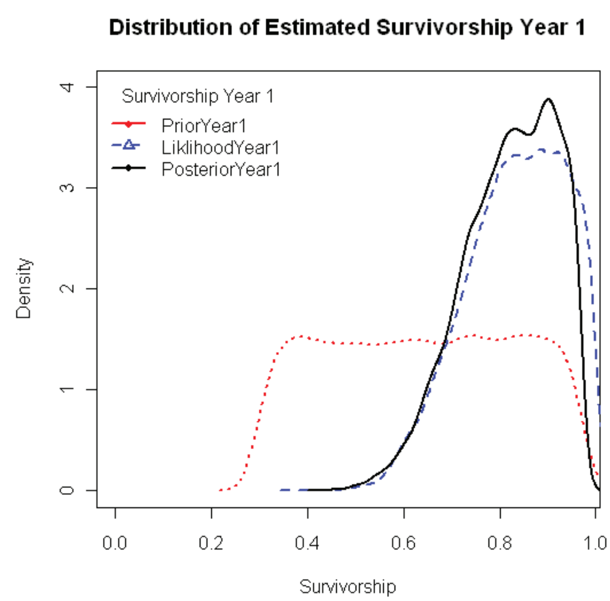

Figure 2.5: Distribution of estimated survivorship values for year one.

A distribution was needed to form the prior distribution of the survivorship parameter at year two. Thus, the posterior distribution of the survivorship parameter at year one was approximated by a beta distribution, with parameters a= 9.588 and b=2.158, to form the prior distribution for this parameter at year two. Because this distribution is an approximation of the posterior estimate of S at year one, the posterior estimate of S at year one and the prior estimate of S at year two are similar, but not exact. The posterior survivorship estimate, derived by combining this prior with the empirically derived likelihood from the age structure for year two, yielded a Bayesian derived posterior estimate of survivorship for year two with only slightly less variability than the prior distribution (Figure 2.6).

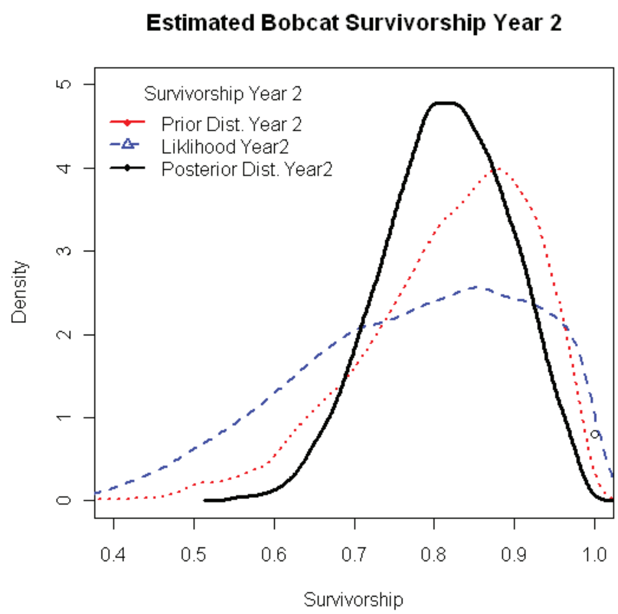

Figure 2.6: Distribution of estimated survivorship probabilities for year two.

Simulations of the initial projection matrix model, in which the survivorship hyper-parameters were estimated solely from the literature, produced a distribution of lambda values that were approximately normal with mean λ=0.93 and variance $\sigma_\lambda^2 = 0.028$ (Figure 2.7).

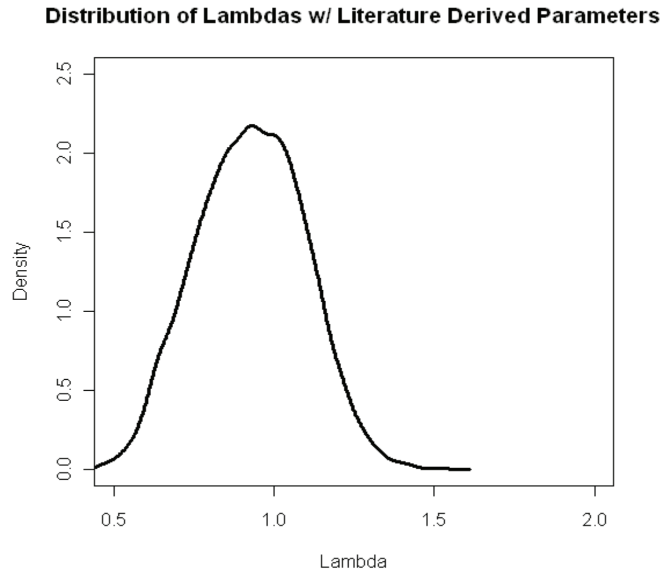

Figure 2.7: Frequency of λ values produced from model simulations with literature derived parameter values only.

The variation associated with the input parameter estimates in this model reflects the degree of uncertainty that exists given the available data prior to collecting any additional information. An estimated 64.9% of model simulations under prior parameter specifications resulted in negative growth.

Simulations of the projection matrix model for year one, in which the survivorship parameter values were drawn from the posterior estimate of survivorship at year one, produced a distribution of lambda values with a mean value of λ=1.14 and σ^2_λ =0.017 (Figure 2.8). Only 13.5% the models produced predicted negative population growth.

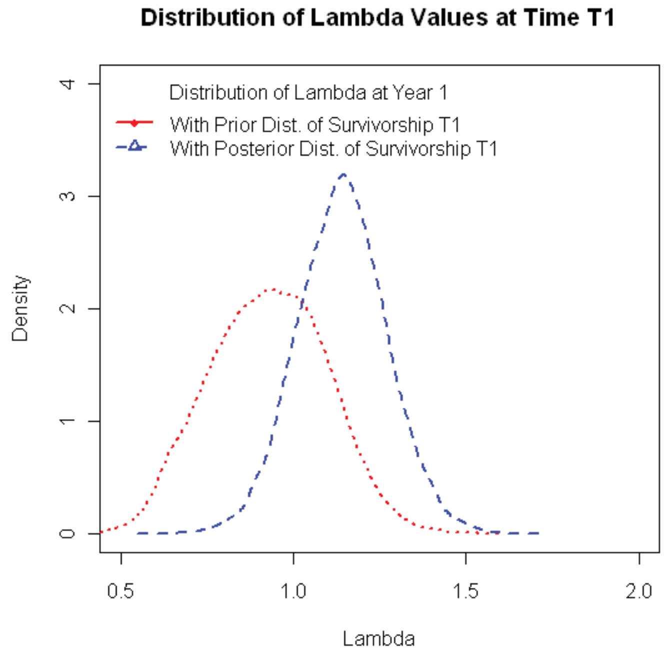

Figure 2.8: Distribution of λ values produced from model simulations with the prior and posterior estimates of survivorship at year one.

Simulations of the projection matrix model for year two, in which the survivorship parameter values were drawn from the posterior distribution of survivorship at year two, produced a distribution of lambda values with a mean value of λ=1.14 and σ^2_λ =0.014 (Figure 2.9). Only 12.2% the models produced predicted negative population growth.

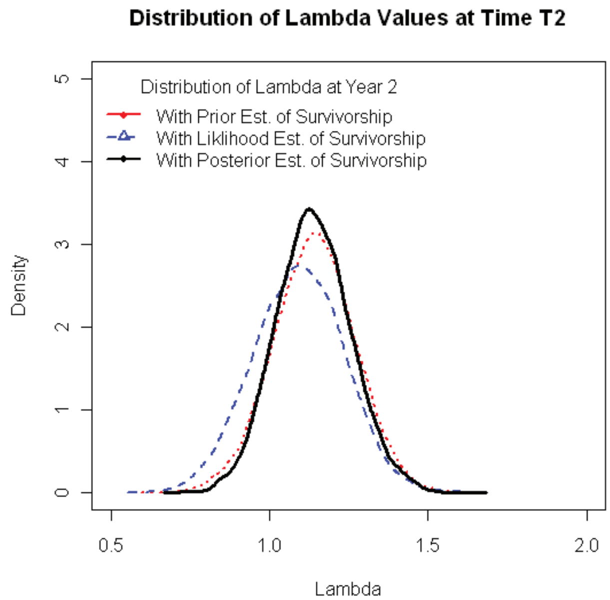

Figure 2.9: Distribution of λ values produced from model simulations with the prior and posterior estimates of survivorship at year two.

Discussion

Bayesian inference provided a framework for incorporating existing and new information to create models to reflect our improved understanding of the system state. Our models suggest a positive growth rate for this population. The final model predicted a mean λ of 1.14 (S.D. = 0.12). These projections indicate that some increased mortality could be sustained by this population and introduced into the long-term management strategy.

This bobcat population provided a unique opportunity to develop a framework for collecting information from an unexploited population, given that this area had been not experienced any harvest for over two decades prior to our experiment, allowing us to make reasonable assumptions regarding a stable-age distribution reflective of natural mortality. This allowed us to estimate survivorship from the age structure of the initial harvests and incorporate it into a model that can be used to predict the subsequent consequences of management actions and determine the best pathways for improving understanding. A management action, in the form of a short-term experimental harvest, was designed and implemented to facilitate the collection of harvest-dependent data. Bayesian inference, by combining existing information, in the form of an informative prior, and newly gathered information, in the form of the likelihood, allowed for the maximum utility of sparse data sets that otherwise may be of limited value (Durban et al. 2000, Dixon et al. 2005, Royle and Dubovsky 2001). We found that the Bayesian approach worked well for updating our model to reflect our increased knowledge of this population. The precision of our survivorship estimates increased with each time step. Consequently, the variation in the projection matrix model simulations, assessed as variation in projected population growth rates, decreased at each time step. This effect is most dramatic at year one, where empirical data are first introduced and combined with the literature derived data to refine the survivorship parameters. This resulted in increased confidence in model performance and increased the utility of the population model as a useful instrument to explore potential population responses to management actions. The predicted responses in projected growth rate, with the refined model, to changes in parameter values that reflect potential management actions (e.g. increasing mortality X percent) are less variable and, therefore, allow for a greater degree of confidence for the manager. In addition, the development of this population model provides a foundation from which a more complete model can be developed as information is gathered.

Further refinement of the population model can lead to decreased variability in model predictions, thus decreasing management uncertainties when using the model. Given limited resources, careful selection of research priorities is crucial. Caswell (1996) provides a detailed description of sensitivity analysis of matrix population models. Within a matrix, each non-zero element contributes to an overall rate of change. This property permits the calculation of the partial derivatives of each non-zero element with respect to the overall rate of change, thus facilitating a comparison of the relative contribution of each non-zero element to the rate of change. Tuljapurkar's approximation presents an analogous method for matrix models with stochastic elements (Tuljapurkar 1990). This analysis, when conducted on the projection matrix that incorporated posterior estimates for survivorship at year two, reveals that changes in the variation of the adult survivorship parameter provide the greatest contribution to changes in the variation of the projected population growth rate (Figure 2.10). Therefore, efforts to reduce uncertainty in model projections for this bobcat population should focus on refining our knowledge of the adult survivorship parameter. The addition of age structure data at year two, incorporated as the likelihood at this time step, did not significantly reduce variation in the posterior distribution (Figure 2.9). The data that composed this likelihood were very sparse (n=15). To further refine this parameter, it would be beneficial to explore alternative approaches to estimating survivorship. Traditional mark-recapture studies and telemetry studies would be ideal, but the costs associated with investigations for an elusive carnivore would very likely be prohibitive. Mark-recovery studies have proved useful for estimating survivorship for some terrestrial species (Brownie 1985, Burnham 1993, Barker 1997). Given the limited harvest of bobcats in this area, marking studies would be difficult to implement due to sample size considerations. However, Bayesian approaches would facilitate the use of these limited data as well in combination with appropriate priors and combined with other data sources. Recently, Conn et al. (2008) used a Bayesian approach to combine mark-recovery data and harvest age structure data to successfully estimate abundance and survivorship of a black bear (*Ursus americanus*) population. A similar approach, utilizing both mark-recovery and age structure data in a Bayesian context, could be employed for our bobcat population to further refine the demographic parameters of the model.

$$\left\{\begin{array}{ccc} F_0 & F_1 & F_2 \\ S_0 & 0 & 0 \\ 0 & S_1 & S_2 \end{array}\right\} \Longrightarrow \left\{\begin{array}{ccc} 0 & 0 & 0.1854 \\ 0.185 & 0 & 0 \\ 0 & 0.185 & 0.446 \end{array}\right\}$$

Figure 2.10: Stochastic elasticities of projection matrix model elements at year two.

We focused only on reducing parametric uncertainty in a single, deterministic model. This assumes no density-dependence among the parameters. In addition, we assumed no successful reproduction in juveniles and yearlings, and constant survivorship across age classes. It would be useful to explore the structural uncertainties by creating multiple models, with differing degrees of interaction between the matrix elements, and compare model predictions with observations to provide evidence as to the most accurate model structure given for the system. Johnson et al. (2002) describe using the Adaptive Resource Management framework to discern between multiple models representing different hypotheses on density-dependence in mallards (*Anas platyrhynchos*). A similar approach would be valuable to further explore the structural uncertainties in bobcat population dynamics.

We also assumed that the harvest age structure was representative of the population age structure. This assumption could be violated if there an age-specific bias in the harvest due to behavioral differences in age classes that resulted unequal capture vulnerability. This assumption could likewise be violated if trapping equipment were more likely to successfully kill or restrain a specific age class. Given that bobcats reach adult size by the first winter, this scenario is unlikely (Leonard 1986, Rolley 1987). Similarly, this assumption could also be violated if trappers selectively took, or released, animals based on size or age. There were no bag-limit restrictions for trappers in these seasons that may encourage trappers to keep only larger animals. Every successful trapper in the present situation was interviewed by the NYSDEC at the conclusion of the season. There was no evidence of trappers releasing or taking captured animals based on animal size, age, or sex. If there was an age-specific bias in the harvest, than survivorship estimates based on the harvest age structure would also be biased.

Bayesian inference has been proposed as an approach to updating model probabilities in the Adaptive Resource Management process (Johnson et al. 2002, Dorazio and Johnson 2003, Prato 2005). Adaptive Resource Management provides a framework for acquiring new information to investigate key uncertainties while Bayesian inference can provides a robust and efficient methodology to combine multiple sources of information (Williams 1997, Williams and Nichols 2001, Williams 2003). Both approaches involve a component of adding additional information to further define the state of the system. Ellison (1996) noted these similarities and stated that Adaptive Management is 'precisely analogous to an iterative Bayesian learning and decision process.' While it may be an overstatement to claim the approaches are 'precisely analogous,' the conceptual similarity is evident. Integrating these investigative paradigms has great potential to advance the overall goal of better understanding the population dynamics of a given population.

The actions we took were not without risks, given that management actions were implemented despite the recognition of known uncertainties. We attempted to mediate this risk by constructing our management action with enough flexibility to halt harvest immediately if deemed necessary. There are formal mechanisms to evaluate the risks and benefits of potential management/research actions (Walters 1986, Walters and Green 1997). These processes consider the benefits of reducing uncertainties, the risk of negative consequences resulting from an action, and the probabilities of these events occurring. This information is then used to decide if the benefits outweigh the risk of a potential action. Unfortunately, quantifying the 'value' of a resource can be problematic. Commercially harvested species can be assigned economic values that reflect market values. However, this overlooks the other more intrinsic values of the species. For example, while we can quantify the value of a localized extirpation of bobcats in terms of pelt values not realized, the ecological values of the bobcat population, the inability of the public to enjoy these populations, and the loss of credibility with the public are much more difficult to quantify. The relativity simplicity of our study (e.g. we knew relativity little of this population), and the limited risk afforded by the unique conditions of our management action, precluded the need for a detailed analysis of costs and benefits.

The investigation of carnivore population dynamics and harvest management presents many challenges due to the relatively low density at which many of these wide-ranging species occur, their elusive nature, and cryptic behavior. In these situations, the use of Bayesian inference is particularly attractive as it maximizes the utility of sparse data sets by facilitating the combining and updating of multiple sources of information. The approach outlined here has merit for exploring key uncertainties for carnivore harvest management and should be considered as a research and management paradigm for these species.

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CHAPTER 3

Contemporary Furbearer Management for the 21st Century: Adapting to a Changing World.

Abstract

Wildlife management agencies have been entrusted with the responsibility of administering and managing the public's wildlife resources, including furbearers. Management of furbearers presents numerous challenges due to their often elusive nature, relatively low population densities, and limited distributions. In addition, it is often difficult to predict the actual impacts of management actions on harvest and populations.

In this chapter we present arguments on why understanding population status and the impacts of management actions if crucial to fulfilling the obligations agencies have to the public. We provide recommendations on what data are needed to gain this understanding and how these data can be collected from harvest-dependent sources. In this, we hope to demonstrate that the challenges of furbearer management are not insurmountable and to encourage agencies to developing strong, data-driven furbearer conservation programs that will improve the management and stewardship of this resource.

Keywords: furbearer, harvest, monitoring, Public Trust Doctrine, population status

The North American Model of Wildlife Conservation is a set of principles that have guided the wildlife management and conservation institute (Geist et al. 2001, Geist and Organ 2004). The Public Trust Doctrine is considered the cornerstone of North American Model of Wildlife Conservation wildlife management and is founded on the concept that wildlife is a public resource, not owned by anyone, but is instead a resource to be held in trust by the government, for the good of the public whom they serve (Sax 1970, Smith 1980, Horner 2000, Geist and Organ 2004). The term 'agency' is applied to those departments within government charged with this task and the name is entirely appropriate given that these departments are serving the role of an agent administering a resource for the benefit of others. The agent-beneficiary relationship is rooted in the assumption that the agent has specialized skills, training, and knowledge, un-possessed by the beneficiary, which can be employed for the benefit of the beneficiary (Martin v. Waddell, 41 U.S. 234).

The North American Model of Wildlife Conservation includes several components, including the positions that wildlife is a public trust resource *and* that science is the proper tool for discharging wildlife policy (Geist et al. 2001). This concept reinforces the agent-beneficiary relationship in that specialized training in scientific inquiry, and application of scientific principles, is required of the agent in order to discharge the duties of managing the public's wildlife trust.

We postulate that to fulfill the obligations of an agent, acting on behalf of the public and administering the public wildlife trust placed in their charge, agencies must (1) know what the desires and needs of the trustee are, and (2) administer the trust in a manner that is fair, equitable, sustainable, and (3) use scientific principles and relevant information to assure that these charges are adequately discharged. For the purposes of this paper, we will focus on the consumptive use aspect of furbearer management and assume that the regulated, sustainable, consumptive use of furbearers is one of the many benefits the public desires of the wildlife resource. We propose that data-driven management of furbearers is not only desirable, but essential to satisfying the obligations of the Public Trust Doctrine. We also propose that the data needed to provide a strong, scientific basis for furbearer management decisions is obtainable and should be utilized in order to embrace the North American Model of Wildlife Conservation as put into practice. We further propose that striving to increase the effectiveness of management actions to achieve goals, and our understanding of natural systems, should be a goal of all furbearer conservation programs.

The Challenges of Contemporary Furbearer Management

Furbearer management presents many challenges for wildlife management agencies. Furbearer species provide many benefits to stakeholders, including a valuable ecological role, providing opportunities for non-consumptive uses such as wildlife viewing and photography, and consumptive uses such as hunting and trapping. While our focus is on providing sustainable, consumptive use activities, even just this one aspect of furbearer management is full of challenges. Furbearer management, and particularly the consumptive use of furbearers, is a controversial branch of wildlife management. The public is often divided regarding the ethical appropriateness of fur harvest (Andelt et al. 1999). Even within the natural resource management profession, many are opposed to furbearer trapping (Muth et al. 1998). Reflective of this, over the last two decades several states have had ballot referendums that have severely restricted, or completely banned, the trapping of furbearers including, Arizona, California, Colorado, Massachusetts, and Washington (Minnis 1998). Minnis (1998) identified several reasons why these referendums occur including perceptions of animal suffering, conflicts over multiple uses of public areas, perceptions of unfair chase, and failure of the regulating agency to insure the public that the consumptive use activities were not detrimental to the long-term stability of the furbearer populations. Legal challenges to trapping programs have generally focused on the sustainability of trapping (Goedeke and Rikoon 2008, Animal Welfare Institute vs. Roland D. Martin, Commissioner of the Maine Department of Inland Fisheries and Wildlife, 2009).

Furbearer population management can be problematic due to difficulty in estimating population status and trends, and difficulty in controlling actual harvest indirectly through the adjustment of harvest opportunities. In order to have an effective furbearer harvest management program, management agencies must (1) understand the effects of management on population change and (2) be able to produce the desired effects of regulatory actions on harvests. Either of these elements alone is insufficient to effectively manage for the sustainable harvest of furbearers, or any species.

What are the Needs?

To develop an informed decision making process, we need to understand the status of the population and the effects of management actions on the population. Population status information may suggest opportunities for additional harvests or, conversely, needs to restrict harvest to ensure the long-term stability of a population. Harvest mortality can be influenced by adjusting harvest opportunities, such as the duration of a season, individual or cumulative bag limits, or influencing harvest potential through the regulation of harvest techniques and methods. While the social impacts among trappers of decreased harvest opportunities may be viewed negatively in the short term, the long-term positive impacts of maintaining the sustainably of the population should be paramount. Maintaining the health of furbearer populations can also insure the ability of other citizens to enjoy the benefits of these species. If these populations are negatively impacted, then this impacts the ability of both consumptive and non-consumptive users to enjoy the benefits these species provide in the future.

Not only is population status information is important informing management actions; these data can also be used to demonstrate that consumptive use activities are not detrimental to the long term stability of the wildlife resource. For example, in the late 1990's, Missouri's river otter management program was legally challenged three times by animal welfare organizations. These groups argued that the harvest of river otters was detrimental to the long-term stability of the population and that the court should terminate all river otter trapping in Missouri. Catch per unit effort was utilized to demonstrate that the river otter population was stable, despite the legal take of over 1,000 animals annually. Similarly, age-specific reproductive rates and population age-structure were estimated from carcass examinations and used in a population model to indicate a positive projected growth rate for this population. These data reassured the pubic that the otter trapping program was sustainable and it further provided for a successful legal defense in all three legal challenges (Goedeke and Rikoon 2008). In the absence of similar information, it would be difficult to argue that populations are stable, leaving wildlife management agencies vulnerable to legal challenges and questions of credibility. In addition, without reliable population status information, it is possible that management actions could unknowingly jeopardize populations.

Harvest management actions are intended to effect harvests by influencing hunter or trapper activities and behavior (Riley et al. 2002). For example, a reduction of harvest may be induced by reducing the length of a trapping season. But, such actions do not always result in what was initially intended. These disparities can limit the ability to control harvest and thus, render population management ineffective. A discrepancy between the intended outcome of a management action and the actual outcome of a management action is termed an issue of 'partial controllability'

(Williams 1997). Given that trapper effort is motivated by a variety of factors, including fluctuating weather and fur market conditions, controlling the harvest indirectly through controls on harvest effort is difficult (Siemer et al. 1994). Yet, in order to effectively respond to population change, maintain sustainability, and meet other specific population objectives, it is crucial to understand the actual effect management control has on harvests. Understanding what impact management actions will likely have on trapping behavior allows for the careful construction of regulations to obtain the desired impacts needed to achieve population objectives.

Acquiring the Needed Information: Is the Task Too Large?

While understanding the furbearer population status, and impacts of management actions, may seem daunting, it is not insurmountable. The needed information is not only obtainable; this information can be obtained relatively efficiently and economically.

Large scale investigations of furbearer population status and trends can be daunting. Classic mark-recapture studies can provide robust population estimates. However, because most furbearers do not congregate, are widely distributed, and have lower densities than waterfowl or ungulates, the logistics of capturing a large number of animals make implementation on a large scale, or incorporation into long-term monitoring programs, unfeasible. Sign surveys, in which the occurrence of tracks, scats, or other evidence of furbearers is recorded, appear to be feasible for large-scale investigations of the range and distribution of some furbearers, but are of little use in assessing annual abundance indices (Roberts et al. 2008, Crimmins et al. 2009). Catch per unit effort can provide an index of abundance based on the ratio of captures given a specified amount of effort and has been identified as a useful measure of furbearer population trends (Dixon 1981, Fortin and Cantin 1990, Chilelli et al. 1996, Strickland

1994, Roberts 2008). Several jurisdictions employ catch-per-unit-effort surveys to monitor furbearer abundance including Vermont, New Hampshire, Rhode Island, Massachusetts, and Québec. Catch per unit effort relies on a number of underlying assumptions including (1) harvest and effort are accurately reported, (2) the population is closed with the exception of harvests, (3) each individual has an equal and independent probability of being captured, and (4) that capture vulnerability is constant throughout the sampling period. Violations of these assumptions can result in significant biases, thus care should be taken to assure that the assumptions are satisfied when using this index (Gould and Pollock 1996). The assumption of a closed population should be reasonable for most furbearers if the study areas are sufficiently large enough to minimize the effects of immigration and emigration, and harvest is not occurring during birthing periods. The more relevant assumption for furbearers is that catchability is constant among individuals, at least within a season. This assumption may not hold for all furbearer species. For example, canid species are captured primarily with foothold traps and cable restraint devices (AFWA 2005). There is a possibility of escape with these devices that may favor the capture of younger animals, or lead to trap avoidance in individuals that escape. Conversely, fisher (*Martes pennanti*) are captured primarily with lethal, body-gripping traps (AFWA 2005). Over 97% of fisher that encounter these devices are killed (AFWA 2009), preventing the development of trap avoidance behavior. Catchability may vary for a variety of other reasons including seasonal changes in behavior, changes in metabolic demands, and changes in capture techniques (Dixon 1981). While the use of catch per unit effort may not be a viable option for all species, or all situations, due to the difficulty in satisfying the underlying assumptions, these assumptions can be reasonably met for some species, such as fisher, providing an efficient and economical source of population status information.

Catch per unit effort, collected from trappers, is a practical index of relative abundance to use because it provides an opportunity to survey large geographic areas and gather large and relatively unbiased data sets at relatively low costs. An experiment in New York, in which catch per unit effort data were collected from fisher trappers, produced over 100,000 trap nights of information (described later as a Case Study). The ability of the catch-per-unit-effort index to detect population change is partially dependent upon sample size. To detect a 20% change, at an α level of 0.05, in New York's fisher and bobcat populations, sample sizes of 3,000 and 12,000 trap nights, respectively, are needed (Appendix 3.1). This sample size can be acquired by collecting the catch and effort information on 11 and 95 fisher and bobcat trappers respectively (Appendix 3.2) at an estimated cost of <$2,500 USD and 200 personnel hours (Appendix 3.3).

Demographic parameters, such as reproductive and survivorship rates, can also provide information helpful in understanding the population dynamics of a species (Caughley 1977, Caswell 2001, Skalski et al. 2005). Population models can be used to synthesize demographic information, estimate projected population growth rates, and explore the effects of different survivorship rates on the overall population growth. This allows a manager to explore the possible effects of manipulating survivorship, via harvest mortality, on the growth of the population. Survivorship parameters can also be examined independent of a population model to determine the actual effect of management actions on survivorship rates. Because of the utility of demographic data, this information is frequently collected by agencies from harvested animals. When carcasses are provided by trappers for examination, it is possible to gather samples from large areas at relatively low costs.

It may not be necessary to collect to collect certain demographic data if sufficient information already exists. Similarly, if the parameter of interest is unlikely to vary regionally, or in response to a management action, it may not be useful to collect these data. For example, fisher litter sizes have been reported by numerous researchers, however the mean litter size for each of 15 studies examined was between two and three offspring (Powell 1979, Powell 1993, Frost et al. 1997). Similarly, bobcat reproductive rates exhibit very little regional variation. Anderson (1987) surveyed 21 bobcat studies and found an average litter size to be 2.7 (SE = 0.09) kittens per litter. For these species, additional studies are unlikely to produce results that could not have been reasonably predicted from existing sources. Conversely, river otter reproductive rates have been found to vary significantly across their range. Chilleli et al. (1996) found that reproductive rates, and the occurrence of reproductively active river otter yearlings, varied across the Northeast United States. Roberts et al. (2010) examined fecundity rates in a recently reintroduced and expanding river otter population in Missouri, USA, and found the mean litter size, and percentage of reproductively active yearlings, to exceed other published accounts, suggesting that this parameter may be density dependent for river otter. Examining these rates empirically for river otters, and especially in novel geographic areas, is justified given the degree of variation reported in the literature.

For long-lived species with relatively low reproductive rates, such as most furbearers, survivorship generally has the greatest proportional influence on the population rate of change (Heppell et al. 2000). Given the influence that survivorship has on population trends, and that estimates cannot be readily extracted from literature sources due to the many site specific influences on this parameter, such as natural and harvest mortality, obtaining information on survivorship can lead to a better understanding of population status and trends.

Population age-structure is estimated from the examination of carcasses. To estimate these rates, the sample from which the estimate is derived must be representative of the overall population. Therefore, an important assumption is that all individuals of the population have an equal probability of being included in the sample. Another assumption is that individuals can be accurately aged. This assumption is easily met for furbearers using cementum annuli counts (Crowe 1972, Chilelli et al. 1996, Stickland 1994). If the sample consists of carcasses collected through a harvest program, the assumption that that harvest is representative of the population should be given extra consideration. Any component of the harvest regulations, or harvest techniques, which could introduce bias by disproportionately selecting a certain subcomponent of the population, will compromise the validity of the estimate. For example, consider a bobcat population where the age-structure is to be estimated from the frequency of age-classes comprising the harvest. Regulations that require the release of kittens or lactating females would bias the sample against these age-classes. The subsequent estimate of age-structure would, consequently, also be biased.

Estimating survivorship from the age-structure of the harvest can be done using a variety of methods. By assuming that recruitment (N_0) and survivorship (S) are constant, the expected harvest in age-class x (H_x) can be written as $E(H_x|x) = N_0pS^x$ where p is the probability of being harvested (Skalski et al. 2005). Thus, survivorship is classically estimated using linear regression applied to the model $\log(Hx) = \alpha + \ln(S)x$. The sample size needed to estimate a survival probability using these methods depends of the level of estimate accuracy desired (Appendix 3.4). For example, to obtain a survivorship estimate with an absolute error of 0.05%, a minimum sample size of 192 specimens should be collected for a population with a survival rate of 0.5. Similarly, to achieve a similar estimate for a population with a survival rate of 0.75, a minimum size of 72 specimens should be collected. Survival rates ≥ 0.75 are reasonable for long-lived carnivores, such as fisher (Powell 1993, Krohn et al. 1994), river otter (Melquist and Hornacker 1983, Chilleli et al. 1996, Gallagher 1999), and bobcat (Crowe 1975, Knick 1990, Nielsen and Woolf 2002). Currently, it costs about $5.00 USD to have one tooth sectioned and aged (Matson's Laboratory, Milltown, Montana, USA). Thus, if carcasses or jaws are provided by trappers, a reasonable estimate of survivorship can be obtained for <$500, plus the personnel cost of preparing the teeth for laboratory analysis (approximately 1 hour per tooth).

For some furbearer species, such as bobcat and river otter, acquiring an adequate sample size of harvested animals for a robust estimate of survivorship can be problematic given the relatively small harvests of these species. According to the New York State pelt sealing database, only one of the 11 Wildlife Management Unit Aggregates that permit bobcat harvest had more than 72 animals taken during the 2005/2006 trapping season. Similarly, only three of the 11 Wildlife Management Unit Aggregates that permit river otter harvest had more than 72 animals taken. For these species, Bayesian approaches that facilitate the integration of multiple sources of information, including external information, in the form of informative priors can be useful. This yields better estimates of demographic parameters than would be obtained using only the sparse empirical data alone. The integration of existing information, such as that which exists in the literature, with empirical information derived from the observed age structure, can maximize the utility of sparse data sets and provide an objective mechanism for identifying future information needs and incorporating future data inputs.

Predicting the actual impact of harvest-management actions on harvests may also seem difficult. Given that harvest is a function of the population size, the probability of an individual being captured given a unit of effort, and the amount of harvest effort expended, there are numerous reasons why a change in harvest opportunity may not result in a proportional change in actual harvest. Harvest effort may vary due to a trapper's available time, environmental conditions, access to trapping lands, and fur market conditions (Siemer et al. 1994). Capture vulnerability may also vary temporally, distorting the relationship between changes in harvest opportunity and actual harvest (Gould and Pollock 1996). Partial controllability can be examined by designing and implementing carefully constructed management actions, then monitoring the subsequent system response. Discrepancies between the intended effects of management actions on harvests, trapper behavior, or populations can be examined by having monitoring in place prior to initiating the management action. The system response can then be observed and compared to the intended response. Responses in trapper effort can be observed using the catch-per-unit-effort index which, by default, records trapper effort. Similarly, population level responses can also be observed through this index and in demographic parameters. These monitoring techniques, as discussed, are economically and logistically feasible for furbearers.

Building Knowledge, Enhancing Stewardship

Wildlife agencies should be constantly striving to improve their understanding of the systems they manage. Increased understanding can lead to more effective and efficient management programs. The process of Adaptive Resource Management can be used to enhance furbearer management by reducing management uncertainties (Johnson et al. 2002, Williams 2003). An active Adaptive Resource Management program requires several components for successful implementation including the abilities to (1) implement management actions intended to provide information to reduce uncertainties, (2) monitor system responses, and (3) adjust management programs to reflect new levels of understanding. New York is fortunate in that most wildlife regulation making, with the exception of those aspects directed by statutory law, is delegated to the New York State Department of Environmental Conservation. Changes in statutory law would require the action and approve of the legislative and executive branches of state government, a process that can take a significant amount of time. Changes in regulatory law, conversely, can occur within a year being proposed, assuming that no significant issues arise during the public comment period required by law. This ability to quickly adjust management actions to gain information, and in response to new information, is a critical component of Adaptive Resource Management and should be exercised.

Monitoring is an important part of the Adaptive Resource Management process. A strong monitoring program can help facilitate the Adaptive Resource Management process by identifying uncertainties, and observing system responses to management actions. The methods we presented for assessing population status can be useful in this process. Similarly, the methods to estimate trapper effort can also be used to monitor responses to management actions.

The Adaptive Resource Management framework can be adapted to provide new insights on furbearer management in New York and elsewhere. Using some of the principles of Adaptive Resource Management, we developed a population model for bobcat that demonstrated the population could sustain a harvest and continue to grow (described later as a Case Study). We found that the survivorship parameters have the greatest proportional influence on the projected growth rate for this population. However, survivorship estimates must be updated regularly as harvest mortality directly influences survivorship. If not updated, the model will cease to be valid. The Adaptive Resource Management process can be used to explore how estimates of survivorship vary under differing degrees of harvest pressure. For example, we do not know whether a bag limit for bobcat would increase survivorship by limiting harvests. In our model, we also assumed a closed population. The Adaptive Resource Management approach could be used to examine immigration from neighboring, unharvested areas, by combining classic mark-recovery studies with manipulations in regulations intended to suppress a population and create home-range vacancies.

When an active Adaptive Resource Management approach is not possible, efforts should still be taken to learn from management actions. Even if the management actions were not intended to aid investigations, information can still be gained by monitoring the system prior to, and after, the implementation of management actions. A strong monitoring program would allow for learning opportunities from most management actions, whether or not learning was the objective of the action.

In addition to improving understanding of systems and management impacts, agencies should also strive to improve their understanding of stakeholder needs and desires. The concept of defining management objectives beyond just tangible and numerical population or harvests goals, to the intangible impacts on society, is gaining momentum (Riley et al. 2003, Neck et al. 2006). Given that agencies are managing the public resource, on behalf of the public, it is logical that these agents should understand the evolving needs and wishes of their consumers.

Case studies

Herein, we present a brief synopsis of two case studies in New York where we successfully applied the principles and methods discussed. These overviews are not meant to be exhaustive reports of these studies, but rather to provide evidence that these principles can be successfully applied.

Case Study 1: Fisher harvest management

In this case study, we utilized an experimental season, and catch per unit effort data, to explore an issue of partial controllability for fisher harvest management. There was uncertainty about the effects of adjusting or increasing fisher harvest opportunities on actual fisher harvests. Fisher trapping season in New York occurs from October 25th to December 10th. Trappers had requested harvest opportunities later in the winter because it is assumed that pelts are more valuable at this time, and consistent snow conditions allow for the use of snowmobiles to access remote areas. The New York Department of Environmental Conservation (NYSDEC) was concerned that harvest opportunities later in winter may disproportionally increase harvests, potentially leading to excessive harvest. The concerns were rooted in uncertainty about how trappers would respond, in terms of effort, to increased opportunity. In addition, it was unknown if capture vulnerability would be different later in the winter. Both trapping effort and capture vulnerability contribute to cumulative catch and uncertainties regarding these elements led to uncertainty in how the actual harvests would respond to increased harvest opportunities.

To investigate these uncertainties, we designed a short-term, experimental season for a portion of northern New York. The trapping season was extended to January 10th and trappers were provided and required to maintain a daily-trapping logbook to record their effort and associated catch. This allowed us to document effort, over the course of the season, and to compare the mean effort between the traditional and extended portions of the season. This also allowed for similar examination of catch per unit effort, which, given certain assumptions, we considered a proxy for capture vulnerability.

Over three years, we collected 100,289 trap-nights of data. Using these data, we were able to determine that there were significantly less (all significance statements at $\alpha=0.05$ level) trappers active in the extended portion of the seasons, the mean individual trapping effort was significantly less during the extended portion of the trapping seasons, and that there was no significant difference in the mean catch per unit effort in the extended portion of the seasons. From these results, we concluded that trapping effort would not increase at a disproportionally high rate with increased opportunity, and that fisher were not more vulnerable to capture, between December 10 and January 10. Therefore, consideration could be given to restructuring the fisher season beyond December 10th without concern of a disproportionally high rate of harvests.

By implementing a management action to investigate this issue of partial controllability, we were able to obtain a large amount of information quickly and efficiently. The use of trapper-derived catch per unit effort was valuable in examining trapper responses to management actions, as well monitoring the population response. Ultimately, we were able learn about the actual effects of a management action on the system, providing information to enlighten future management actions.

Case Study 2: Bobcat population dynamics

In this case study, we used an experimental season to facilitate the collection of harvest-dependent demographic data for use in demographic parameter estimation and subsequent population model construction for bobcats. Bobcat harvests had not occurred in 22 years in the Oswego-Delaware Hills region of New York. Biologists for the NYSDEC had anecdotal information to suggest that this population had rebounded and were interested in initiating a limited harvest in 2006. Very little was known about this population and we desired to gather demographic information, estimate demographic parameters, and construct a basic population model that would be useful to predict projected population growth rates and explore the effects of manipulating parameter values on these projected growth rates.

An initial deterministic matrix model was constructed using parameter values extracted from the literature. A sensitivity and elasticity analysis on this model indicated that the survivorship parameters had a much greater proportional influence on the rate of population change than did the reproductive parameters. Thus, we focused on estimating the survivorship parameters for this population and incorporating these estimates, and their associated variances, into projection matrixes.

We initiated a three-year experimental trapping season where trappers were provided and required to keep a daily trapping logbook, similar to the fisher case study. In addition, successful trappers were required to submit skinned carcasses for examination. These requirements facilitated the collection of catch per unit effort, age-specific fecundity, and the age structure of the harvests. The age structure of the harvest was used to estimate survivorship rates. We used a Bayesian approach that allowed for the incorporation of external data, in the form of a semi-informative prior, to augment the sparse age-structure data. The Bayesian approach also creates a framework for updating the survivorship estimates as new data becomes available in the future.

Variation in projected population growth rates, as predicted by the projection matrixes, was examined using simulations. For each simulation, the parameter values were drawn from a distribution that represented the uncertainty associated with that parameter. The resulting projected growth rate was recorded for each simulation and the distribution of these values was considered reflective of the model projection uncertainly. Thus, as the parametric uncertainly decreased, the projection uncertainties should also decrease.

Ultimately, we collected 24,710 trap-nights of data. We refined our estimate of adult survivorship from essentially unknown and dependent upon only literature values, to an estimate of 0.81 (σ^2=.006). As we refined the survivorship parameters by incorporating the empirical age structure data, the projected growth rate changed from λ=0.93 (σ^2=0.28), which was completely dependent on literature values, to λ=1.14 (σ^2=0.014).

Using an experiential season, we were able to facilitate the collection of harvest-dependent demographic data. Although the data were sparse, we were able to use Bayesian inference to estimate survivorship rates from the harvest age-structure and construct a population model. This population model yielded information on projected population growth rates and the relative contribution of changes in parameter values to changes in the projected growth rate. Ultimately, we gained information on the bobcat population status and created a useful tool to explore the potential effects of management action on this population.

Conclusion

Difficulty in assessing furbearer populations, and the effects of management actions on populations, often prevents rigorous data-driven management programs for these species. Fortunately, obtaining the necessary information to make informed decisions can be done relatively efficiently and economically using harvest-dependent sources. While the use of harvest dependent data relies on numerous underlying assumptions, often these assumptions can be reasonably met through careful design and implementation of studies and management actions. By utilizing these sources of information, management programs can build knowledge, reduce uncertainties, and ultimately enhance the stewardship of these wildlife resources.

Agencies should take advantage of the wealth of information that can be provided by trappers. When using these data, it is wise to be aware of the underlying assumptions and the consequences of violating those assumptions. However, if these assumptions can be reasonably met, the resource users themselves can provide an abundance of useful data on populations and responses to management actions. Trappers can provide catch per unit effort data that yields information on population status and trapper activities. Trappers can also facilitate the collection of demographic data. These data can be used to better understand the furbearer populations, the effects of management actions on these populations, and the effects of management actions on the harvest.

Agencies should also not overlook the value of using management actions as a tool to investigate management uncertainties. Carefully designed and implemented management actions can provide large amounts of information that may not be obtainable or, feasible to obtain, otherwise. A strong monitoring program will help maximize the amount of information gained from management actions and the methods we described can be used to facilitate this monitoring. The Adaptive Resource Management approach has long been advocated for improving the understanding and effectiveness of management programs (Lancia et al. 1996). We suggest that Adaptive Resource Management is well suited for many of the challenges in furbearer management, such as partial controllability. With effective techniques to monitor system responses, such as those we presented, and a timely regulatory process that permits relatively quick implementation of management actions, Adaptive Resource Management can be adopted for furbearer management.

In the future, agencies should have in place the framework to extract the maximum amount of information possible from furbearer management programs. Developing protocols to collect and use harvest-dependent data should be instituted.

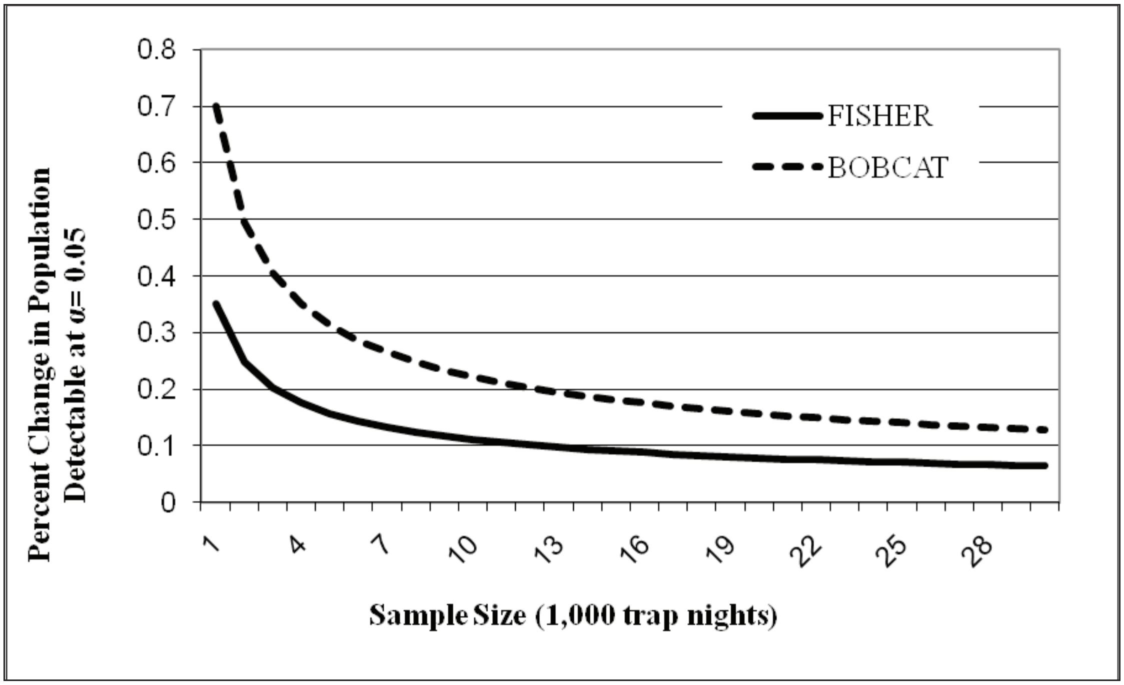
A critical component of this will be determining what information is useful. Careful consideration should be given to how the data will be used and whether the necessary underlying assumptions needed for the data to be beneficial can be met. Agencies should also have a framework for quickly adjusting management actions to respond to new information as it is gathered. This framework should also facilitate the use of management actions to investigate uncertainties, such as the use of experimental seasons and the Adaptive Resource Management process.

Agencies have an obligation to society under the Public Trust Doctrine. The public has entrusted the management of publicly-owned wildlife resources to these agencies to administer on their behalf. Wildlife management decisions, therefore, should be undertaken with the careful consideration that would be expected of an agent managing a priceless asset for a beneficiary; as this is precisely the relationship alluded to through the Public Trust Doctrine. When managing a resource as valuable as the public's wildlife resources, decisions should be based on reliable and robust information. These are obtainable for furbearers and should be utilized. Furthermore, agencies should continuously strive to improve their stewardship of this resource. Anything less than this risks deteriorating public credibility and eventual erosion of the Public Trust Doctrine foundation that has built the North American Model of Wildlife Conservation.

The challenges of furbearer management should be viewed as opportunities. By developing a strong, science-based, data-driven furbearer harvest management program, agencies will not only improve the management of this resource, they will reaffirm their role as the public stewards entrusted with this resource.

APPENDIX

Appendix 3.1: Estimated sample size, in trap nights, required to detect an X% change in the population (α=0.05) for bobcat and fisher in New York (based on catch per unit effort observed on Chapters 1 and 2).



Appendix 3.2: Estimated number of catch-per-unit-effort logbooks needed to obtain sample size of trap nights required to detect 20% in population.

The number of catch-per-unit-effort logbooks required to obtain the desired sample size varies by species. We found that during the 2008-2009 trapping season, the average fisher trapper expended 290 trap nights of effort during the season while the average bobcat trapper only expended 127 trap nights of effort during the season. Given this, to obtain 12,000 trap nights of data for bobcat, at least 95 bobcat trappers would need to record catch per unit effort in each sample unit (i.e. Wildlife Management Unit Aggregate). Similarly, to obtain 3,000 trap nights of data for fisher, at least 11 fisher trappers would need to record catch per unit effort in each sample unit.

Sample Size of Trap Nights Desired	Estimated Minimum Number of Logbooks Required
Fisher: 3,000 Trap Nights	11
Bobcat: 12,000 Trap nights	95

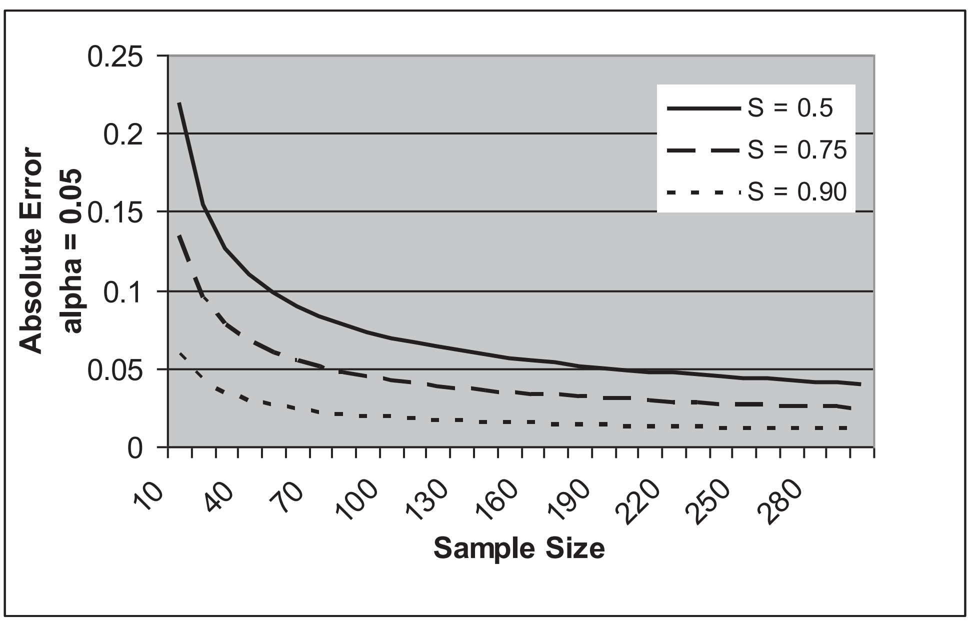
Appendix 3.3: Estimated costs to administer catch-per-unit surveys

We found that the total cost of producing a catch-per-unit effort logbook, mailing the logbook with accompanying instructional letter, mailing a reminder letter, and providing postage for the book to be returned was approximately $2.50 USD per book. Similarly, we found that a personnel commitment, in the form of temporary student employees, of 200 hours per 1,000 logbooks was needed to manage these mailings and enter the associated data into a database.

Number of Logbooks Distributed	Funding Required	Personnel Commitment
1,000	$2,500	200 hours
2,000	$5,000	400 hours
4,000	$10,000	800 hours

Appendix 3.4 Required sample size to estimate survival rates from age-structure within specified absolute error given predicted survivorship rates of 0.5, 0.75 and 0.90.

Estimating survivorship from harvest age structure can be done using a variety of methods. By assuming that recruitment (N_0) and survivorship (S) are constant, the expected harvest in age-class x (H_x) can be written as $E(H_x|x) = N_0pS^x$ where p is the probability of being harvested (Skalski et al 2005). Thus, survivorship is classically estimated using linear regression applied to the model log(Hx) = α + ln(S)x. The sample size needed to estimate survival probabilities depends of the level of estimate precision desired. With precision defined as $P(|\hat{S} - S| < \varepsilon) = 1 - \alpha$, where ε is the absolute error, we can determine the sample size needed for the estimate to have an absolute precision (ε) 1 - α percent of the time with the equation $\varepsilon = Z_{1-\frac{\alpha}{2}}\sqrt{Var(\hat{S})}$ (Chapman and Robson 1960, Robson and Chapman 1961, Skalski et al 2006). Because variance of the survival estimate is approximated as $Var(\hat{S}) \cong \frac{S(1-S)^2}{n}$, the estimated survival rates influences the sample size required to obtain the desired level of precision.



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