Population Monitoring Options for American Black Bears in the Northeastern United States and Eastern Canada

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Executive Summary

INTRODUCTION

Population monitoring is essential for effective black bear management, but uncertainty about applicability, strengths, and weaknesses of different techniques makes it difficult for wildlife managers to decide which methods are most appropriate to accomplish their objectives. Consequently, the Northeast Black Bear Technical Committee (NEBBTC), a group of bear managers charged by the Northeastern Wildlife Administrators Association to share data and ideas across jurisdictions, commissioned this technical report to investigate black bear population monitoring options for the northeastearn United States and Eastern Canada. The NEBBTC is comprised of black bear managers from 13 U.S. states and 6 Canadian provinces: Connecticut, Delaware, Massachusetts, Maryland, Maine, New Brunswick, Newfoundland, New Hampshire, New Jersey, New York, Nova Scotia, Ontario, Pennsylvania, Prince Edward Island, Quebec, Rhode Island, Vermont, Virginia, and West Virginia.

The objectives of this report were to (1) provide an overview of current status and management of American black bears in the region, (2) produce an in-depth review of reliable techniques to estimate population parameters and monitor population trends, and (3) provide guidance to agencies to better enable them to choose monitoring tools that best meet their management objectives given available resources. We surveyed NEBBTC managers to get a better understanding of their management needs and resource limitations. We used computer simulations to evaluate different data types (e.g., hair traps, radiotelemetry) and statistical models

(e.g., capture-recapture, spatial capture-recapture). The monitoring options we present are necessarily general and are not meant to be critical of current practices of any jurisdictions.

BIOLOGY

Black bears inhabit a variety of habitats in the Northeast, including boreal and hardwood forests, coastal wetlands, and even barren ground. The variety of habitats and extensive geographic range of the 19 NEBBTC jurisdictions encompass a landscape that is highly variable in productivity. Consequently, population parameters (e.g., density, survival, population growth rate) vary widely for black bears in this region. A number of research projects have been conducted on black bears in the Northeast so the range of population growth and vital rates is relatively well-documented.

MANAGEMENT

Black bear populations are generally increasing throughout NEBBTC jurisdictions and the greater number of bears often leads to human-bear conflicts, particularly in areas with relatively high human densities. This situation puts pressure on both bears and people, making it difficult for managers to find effective management strategies. Our survey of NEBBTC managers revealed a number of interesting perspectives on management objectives and population monitoring methods. Harvest is the primary tool used by managers to affect change in Northeast black bear populations. Almost all NEBBTC managers would like to find cost-effective monitoring techniques that produce accurate and precise estimates of population parameters to efficiently assess the success of management objectives. The results of our manager survey helped guide the focus and scope of this technical report.

MONITORING TECHNIQUES

We identified many black bear population monitoring options for managers in the Northeast, but there is no single best method for all bear populations and management objectives. One of the greatest dilemmas for bear managers today is that the most expensive monitoring methods (i.e., radiotelemetry, DNA-based mark-recapture) are also the methods that provide the most accurate and precise estimates of population parameters. However, less expensive methods are ultimately poor investments because the data often provide only limited inference with high uncertainty, thereby restricting the ability of managers to predict responses to changes in habitat and management.

Key Findings:

- Management objectives, spatial scale, and existing data should be considered when determining the methods to monitor population parameters.

- Indices can be useful for decision making when strong correlation with abundance can be demonstrated, but this has yet to be accomplished for black bears.

- Indices are most useful for examining fluctuations in long-term population dynamics.

- Estimates are more useful than indices to wildlife managers for making management decisions because they provide numbers with associated measures of precision.

- A number of marking techniques can be used in capture-recapture models, with ear tagging and DNA-based methods being common and effective marking methods.

- Large study areas can be difficult to adequately sample when home ranges are small, but managers may benefit from economies of scale compared with smaller studies, which often require the same minimum investments in labor and equipment.

- Multiple sampling methods (e.g., hair traps, bear rubs, scat surveys, harvest) should be strongly considered because the use of independent methods increases precision and reduces bias of parameter estimates.

Abundance and Density

Our evaluation suggests that the most cost-effective and reliable technique to estimate black bear population abundance and density is DNA-based mark-recapture, for which a number of sampling methods exist. Newer models such as spatially explicit capture-recapture (SECR) use information on spatial locations of detectors such as hair traps and bear rubs, making them more suitable for estimating density when sampling is adequate (e.g., given typical home-range sizes). Although SECR models are a rapidly developing area of research, when precise estimates of abundance are needed, managers should still consider non-spatial models as they have a longer history in bear research and management. Fortunately, spatial and non-spatial estimators can usually be used in tandem. Regardless of the model type, the use of multiple sampling methods (e.g., hair traps, harvest) should be considered to reduce bias and increase precision of estimates.

Key Findings:

- Closed mark-recapture models represent one of the best techniques to estimate abundance for black bears because of a long history of use and demonstrated reliability.

- Density often is a more relevant parameter than abundance to wildlife managers, and is a particularly useful parameter to evaluate habitat quality within and between studies.

- Newly emerging estimation techniques based on spatially-explicit capture-recapture models show great promise to account for the spatial location of sampled animals in relation to the sampling grid to adjust density estimates accordingly.

- Our simulations suggest that managers considering spatially-explicit capture-recapture models should be aware that unbiased and precise density estimates require that the sampling area is sufficiently large and detectors spaced close enough to adequately characterize home ranges.

Survival and Reproduction

One of the most reliable ways to estimate survival and reproductive parameters for black bears is capture, radiomarking, and monitoring. Although radiotelemetry can be expensive, it provides the most useful and reliable information on survival, reproduction, and emigration. Monitoring costs are greatly reduced if GPS collars are used compared with VHF collars, although intitial costs are higher. Radiotelemetry-based methods also provide valuable information about causes of mortality and factors affecting reproduction. Furthermore, telemetry data can be used for other purposes such as habitat evaluation, movement dynamics, and estimating dispersal. Some open population mark-recapture models (e.g., robust design) can estimate apparent survival, but not true survival. DNA-based mark-recapture methods are a viable second option for estimating survival and reproduction.

Key Findings:

- Among the most effective ways to accurately and precisely estimate survival for black bears is to radiocollar a sufficiently large sample of the population and track their fates; providing robust estimates of survival and information on causes of mortality, important information to adjust management of populations with negative trajectories.

- Mark-recapture methods can be used to estimate apparent survival, but cannot account for loss through emigration.

- The most effective way to accurately and precisely estimate reproductive parameters for black bears is to radiocollar a representative sample of the female population and monitor their reproductive output over time. An added benefit is that radiotelemetry can be used to also estimate adult survival and other variables (e.g., home-range size, habitat use).

- Reproductive success can be measured by monitoring animals radio-marked during den visits and through direct observations.

Population Growth

A diversity of methods can be used to monitor black bear population growth in the Northeast, but we suggest managers focus on two proven methods, radiotelemetry (i.e., demographic analysis using survival and recruitment rates) and DNA-based mark-recapture. NEBBTC managers indicated the importance of understanding the relationship between population trends and management efforts. Thus, we believe that one method, radiotelemetry, stands out because of its proven track record and potential to detect different causes of population changes. Conversely, DNA-based mark-recapture methods are powerful because they can produce precise estimates of growth rates in relatively short time periods and because the use of covariates allows testing of hypotheses for potential causes of population change. Although DNA-based methods can be expensive, costs can be reduced with subsampling, using multiple sampling methods concurrently, or skipping years.

Key Findings:

- Population growth reflects the cumulative changes to population abundance due to birth, immigration, death, and emigration and may be one of the most effective measures of population performance for making management decisions.

- Useful methods for estimating population growth require that they have sufficient power to detect changes in abundance as a direct result of management and habitat changes, if such changes occurred.

- Interpretation and extrapolation of population growth rates should be done carefully, particularly when monitoring only portions of a larger population.

- Population projection models can be reliable but are data intensive. Estimates of the standing age distribution are important but difficult to estimate. The method requires extensive use of radiotelemetry, but animals can be distributed over a large area.

- Traditional radiotelemetry methods may provide more detail regarding cause-specific mortality, but may suffer from issues of geographic closure violation more so than do more recently developed mark-recapture approaches.

- Mark-recapture techniques such as Pradel models can produce precise estimates of population growth in short time periods relative to known-fate methods and are less affected by most capture biases.

CONCLUSIONS

Choosing Monitoring Technique

There is much variability in bear and human demographics, environmental conditions, and social perspectives among NEBBTC jurisdictions, which makes it difficult to establish region-wide standards for black bear research and management. Black bear populations are increasing throughout the Northeast whereas wildlife conservation revenue from hunting revenues has diminished. Shrinking revenues combined with increasing challenges and expectations make it

more important than ever for managers to choose the most appropriate, reliable, and cost-effective population monitoring techniques to meet management objectives.

Estimates of black bear population parameters are generally more valuable to managers than indices that do not provide estimates of sampling or process error. Because many reliable monitoring methods are available and there is an almost universal desire by NEBBTC managers for accuracy and precision, we suggest that estimation methods be given preference over indices. Use of indices for black bear population monitoring may be justified when budgets are severely constrained and the index has well-known properties and data are already recorded for other purposes (i.e., harvest), but they will generally not be suitable for most management objectives.

Population Monitoring Scenarios

Black bear population monitoring studies should be designed to quantify the response of a population to management or other actions, but choosing an appropriate study design can be challenging for managers given the number of methods available. Ideally, a study design should be both effective at providing the desired information and efficient at collecting the required data. To aid managers in selecting the most appropriate monitoring methods and study design, we used computer simulations to evaluate a range of population scenarios with several classes of population parameter estimation models, including mark-resight, open, closed, and spatially-explicit mark-recapture, and demographic analyses. In this report we provide generalized study design recommendations for monitoring black bear populations based on the results of our computer simulation analyses and collective experience.

For simplicity, we focused our evaluation of black bear monitoring options on the most suitable methods currently available for estimating the two population parameters identified as

being most important to NEBBTC black bear managers (i.e., abundance and population growth). We present our recommendations for 6 scenarios, ranging from small ($N \leq 500$) to large ($N > 2{,}500$) population sizes and from declining to stable or increasing population trends. Acknowledging financial limitations, we urge managers to always strive to use the best monitoring techniques available, particularly those methods that inform adaptive management and function across jurisdiction boundaries.

Future Work

Methods for monitoring bear populations are evolving rapidly. Not only has technology changed (e.g., DNA analysis, GPS collars), but the methods for analyzing such data (e.g., Bayesian statistics, hierarchical methods) have changed as well. One area that we think holds much promise is the integrated population analysis methods that combine harvest and auxiliary data. Another area that holds promise is the integration of mark-recapture methods and occupancy estimation. Our report is a current view of the state of the art of population monitoring and we urge managers to monitor the published literature so that they may take advantage of these anticipated advances. More research will be required to evaluate the performance of these integrated population analysis approaches and to develop software programs for application by bear managers.

Section I. Introduction

Chapter 1. Background and Goals

1.1 BACKGROUND

American black bears (*Ursus americanus*) have been hunted since colonial times for their fur, meat, and fat and to reduce or eliminate conflicts with humans. By the early to middle part of the twentieth century, black bear populations in many areas in North America had substantially declined from historic levels because of excessive killing by humans. Eventually, black bears were recognized for their ecological and cultural value, which resulted in their classification as a game species in most jurisdictions. Consequently, population recovery from overexploitation was an important management goal in the 1960s through early 1980s. Bear populations rebounded as a result of habitat recovery and conservative hunting regulations. As populations increased, hunting emerged as what many view as a safe, effective, and responsible mechanism to reach and maintain population objectives for black bears in a cost-effective manner. In recent decades, black bear populations throughout North America have increased in abundance (Garshelis and Hristienko 2006) and distribution (Figure 1.1). Forty states in the U.S. and all Canadian Provinces, except Prince Edward Island, have resident populations of black bears. In a survey of states and provinces (hereafter, jurisdictions) with black bear populations in 1988 and 2001, 32 jurisdictions reported population increases during that time period, 10 jurisdictions reported stable populations, and 2 reported declines (Hristienko and McDonald 2007).

Although the American black bear has been researched more than any other bear species, Garshelis and Hristienko (2006) found that population trends reported by agencies were not always based on actual population estimates ($\leq$57% of jurisdictions analyzed based on 1988–

2001 period). They argued for more rigorous estimation of population size for states and

provinces. However, they noted that such efforts may be cost-prohibitive and of limited use

because measurement error and annual variation make trend interpretation difficult. Therefore,

other approaches should be considered for monitoring bear populations. Concomitant with

increasing and expanding bear populations, bear conflicts have increased in recent years with 34

jurisdictions reporting increasing human-bear conflicts (Hristienko and McDonald 2007).

Accordingly, black bear management in many portions of North America has gradually
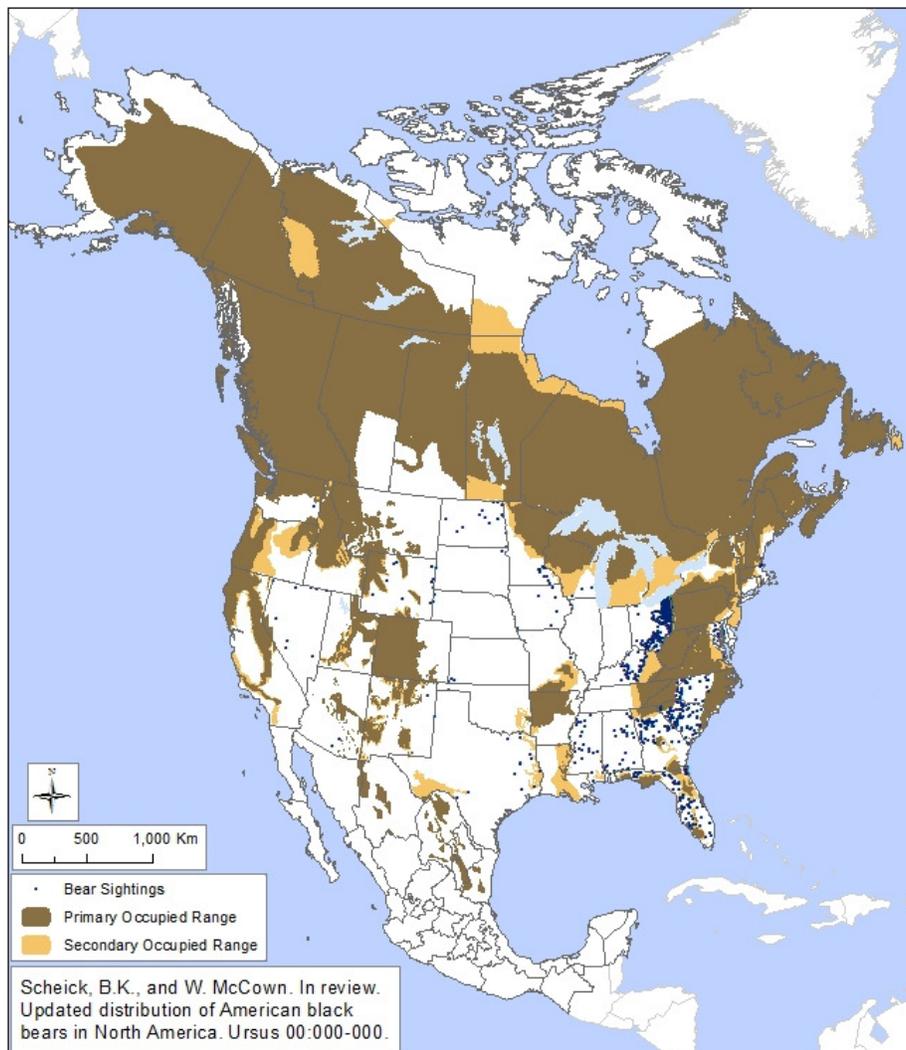
Figure 1.1: Black bear distribution in North America. From Scheick et al. (2012). Bear distributions were mapped by state and provincial biologists using 36-km[2] hexagonal grid cells to identify primary and secondary occupied range.

shifted from population recovery to enhancing harvest opportunities and reduction of human-bear conflicts. Clearly, wildlife management agencies need reliable information and tools to effectively respond to changing management circumstances without putting long-term viability of bear populations at risk. More specifically, agencies faced with increasing bear populations need information on how best to use hunting to reduce human-bear conflicts and they need monitoring methods that allow them to determine if harvest is having the desired effect without risking persistence of the bear population.

1.2 NORTHEAST BLACK BEAR TECHNICAL COMMITTEE

Because of a desire for regional collaboration and perspective on bear population management and monitoring techniques, wildlife agency directors from the northeastern U.S. and eastern Canada created the Northeast Black Bear Technical Committee (NEBBTC). The NEBBTC meets annually to address a dynamic list of charges from the Northeastern Wildlife Administrators Association. The committee is comprised of black bear managers from each of the following 13 U.S. states and 6 Canadian provinces: Connecticut, Delaware, Massachusetts, Maryland, Maine, New Brunswick, Newfoundland, New Hampshire, New Jersey, New York, Nova Scotia, Ontario, Pennsylvania, Prince Edward Island, Quebec, Rhode Island, Vermont, Virginia, and West Virginia. In April 2010, the NEBBTC was charged by the Northeastern Wildlife Administrators Association to review population estimation and monitoring techniques for black bears in the Northeast and to develop a technical publication to help bear managers select appropriate monitoring methods and make informed study design choices. NEBBTC members were asked to cooperate by sharing data, ideas, and opinions.

1.3 OBJECTIVES

The NEBBTC's first charge, to review population estimation and monitoring techniques for black bears in the Northeast, represents an important information need for black bear managers. There are many rapidly evolving techniques for monitoring but uncertainty about applicability and efficacy of each technique makes it difficult for wildlife managers to decide which methods are most appropriate to accomplish their management objectives. Consequently, NEBBTC commissioned this technical report to investigate black bear management and monitoring options for jurisdictions in the northeastern U.S. and eastern Canada.

Jurisdictions of the NEBBTC have a long history of using research to guide their management efforts and a variety of monitoring methods have been used. For example, Maine has estimated bear population parameters using radiotelemetry for >30 years, longer than any state in the U.S. other than Minnesota (Barrett 2011). In Pennsylvania, state wildlife managers have marked bears since 1980 to estimate population abundance and trend (Ternent 2006). Biologists from New York have used age reconstruction from harvest data to estimate population growth rates (New York State Department of Environmental Conservation 2007). Recent black bear population inventory and monitoring efforts in the Northeast have focused on estimating abundance using noninvasive DNA-based capture-mark-recapture methods developed by Woods et al. (1999). Genetic assignment tests using samples from these studies have revealed considerable inter-jurisdictional movement (State Department of Environmental Conservation New York 2007, Huffman 2008). Also, collaboration among jurisdictions takes advantage of certain economies of scale. For example, lower capture rates can be used to reliably estimate large populations compared with small ones. Thus, cooperative regional research and monitoring programs for black bears in the Northeast are appealing for many different reasons.

Some population monitoring methods lack sufficient precision to detect small but meaningful changes in population parameters, may not be feasible at spatial and temporal scales most beneficial to managers, or may provide little information regarding underlying population processes. Financial and logistical constraints often determine the scope of ecological monitoring programs and the techniques considered (Caughlan et al. 2001). Many ecological monitoring programs lack well-defined objectives and neglect sources of variation or error (Yoccoz et al. 2001). Further, populations often cross geopolitical boundaries, and data collection and management objectives may differ substantially among jurisdictions. Regional research and monitoring approaches may better meet the shared objective of sustainable bear populations in the Northeast, but coordinating efforts across wildlife programs benefits from uniform procedures to ensure data compatibility (Lindenmayer and Likens 2010). As such, the primary objective of this report is to assess available methods and provide recommendations that incorporate statistical rigor and precision, feasibility, and cost-effectiveness for a range of population and management objectives. Our objectives were to (1) provide an overview of current status and management of American black bears in the region, (2) produce an in-depth review of reliable techniques to estimate population parameters and monitor population trends, and (3) provide guidance to agencies to better enable them to choose monitoring tools that best meet their management objectives and available resources. Our recommendations are necessarily general and are not meant to be critical of current practices of any jurisdictions. To clarify our use of terminology in this report, we have provided a glossary of terms for managers (Appendix A).

Chapter 2. Study Region

2.1 GEOGRAPHY, CLIMATE, AND VEGETATION

Northeastern North America is characterized by a diverse array of geography, climatic conditions, and black bear habitats. NEBBTC jurisdictions (hereafter, Northeast; Figure 2.1) encompass Atlantic Coastal Wetlands in the east and north to Interior Highlands in the west (Alexander 1967). The granitic Appalachian Mountains dominate much of the region, reaching their highest point at Mt. Washington, New Hampshire (1,917 m). The Atlantic Ocean serves as the eastern and northern borders for many NEBBTC jurisdictions. The region also contains some of the largest freshwater (i.e., Lake Erie, Lake Ontario) and saltwater (i.e., Chesapeake Bay, Hudson Bay) bodies in North America.

The climate of the 13 U.S. states within the study area is classified as humid mid-latitude, with cold winters, warm summers, and distinct autumn and spring seasons (Alexander 1967). The climate of the 6 Canadian Provinces is much colder with shorter summers and more days of lingering snow, so forests typically produce less hard and soft mast. However, daylight is much longer in summer so lowland habitats can be quite productive for black bear foods.

Forests in Connecticut, Delaware, Maryland, New Jersey, Pennsylvania, Rhode Island, Virginia, and West Virginia, are predominated by oak (*Quercus* spp.) and hickory (*Carya* spp.), which provide abundant hard mast for black bears in autumn (Ryan et al. 2004). The forests of Maine, New Hampshire, New York, Ontario, Quebec, and Vermont primarily consist of maples (*Acer* spp.), American beech (*Fagus grandifolia*), and paper birch (*Betula papyrifera*), with only beech providing a valuable, but variable, source of hard mast (McLaughlin et al. 1994).

Massachusetts is a transition zone between those two major forest types. Southern

Newfoundland, Nova Scotia, and Prince Edward Island are dominated by relatively unproductive

boreal spruce-fir forests, which transition to barren ground near the Atlantic Coast and above the

Arctic Circle. Forest regeneration after urbanization and suburbanization over the last century

has resulted in reestablishment of abundant hard mast producing trees, which may have been a

contributing factor to increasing black bear populations in the Northeast.

Figure 2.1: *Northeast study region of 13 U.S. states and 6 Canadian provinces comprising the jurisdictions of the Northeast Black Bear Technical Committee.*

2.2 HUMAN POPULATION TRENDS

The Northeast is one of the most densely populated areas in North America because of its agricultural productivity, proximity to waterways, and early history of settlement. The region is home to almost 100 million people with 72 million people inhabiting the Northeastern U.S. and 23 million residing in eastern Canada (Figure 2.2). The sizes of jurisdictions vary widely: Rhode Island and Prince Edward Island each are <6,000 km^2 whereas Quebec is >240-fold larger in area, with nearly 1.4 million km^2. The Northeast also has some of the sparsest human populations in North America. The lowest density of humans in the Northeast occurs in
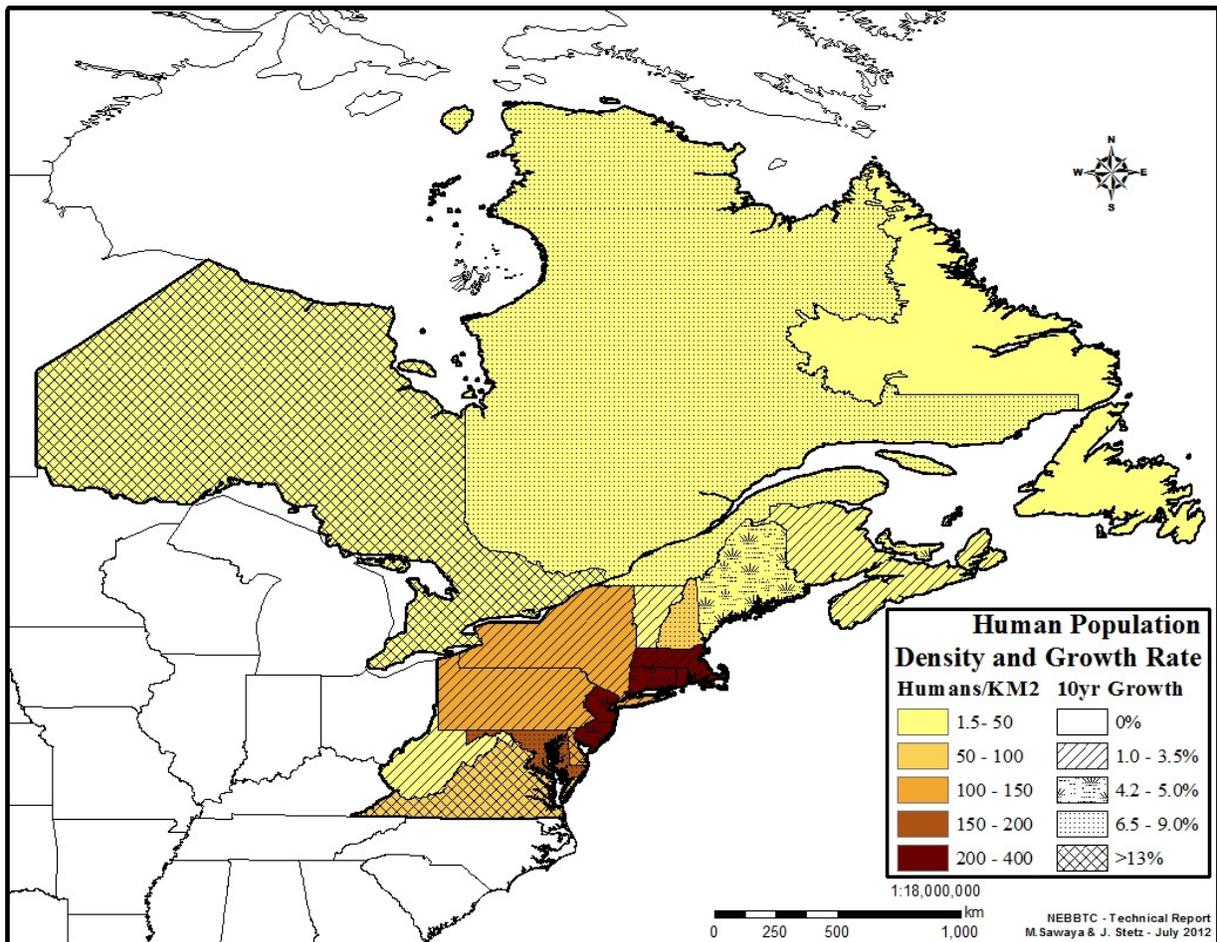
Figure 2.2: *Human population density and growth rates by jurisdiction in the northeastern U.S. and eastern Canada.*

Newfoundland (Figure 2.2), whereas the greatest densities are concentrated near New York City and Toronto, Ontario. According to the 2010 U.S. Census, the average human density in the northeastern U.S. was 105 people/km^2 compared with 32 people/km^2 countrywide (U.S. Census Bureau 2010). In fact, the 13 NEBBTC member states contain almost 25% of the entire U.S. population of >311 million people, but constitute only 7% of the total land area (681,748 km^2). Almost 70% of the entire Canadian population of >33 million people resides in the 6 NEBBTC provinces, while occupying only 32% of the total land area of Canada.

Human populations in the Northeast are experiencing dramatic variation in growth rates among jurisdictions. Between 2000 and 2010, the U.S. population grew by 9.7%, but the average population growth in the 13 Northeast states was only 4.7% (U.S. Census Bureau 2010). Between 2006 and 2011, Canada's population grew by 5.9%, whereas eastern Canadian provinces grew by 11.4% (Stat Canada 2011). Ontario and Delaware, which differ considerably in size and density, experienced approximately 15% growth from 2000 to 2010 (Figure 2.2). Other jurisdictions recorded less growth but only Newfoundland reported a decrease in the human population.

2.3 BLACK BEAR DISTRIBUTION

Approximately 232,000 black bears (Noyce 2012) inhabit the 3.6 million km^2 encompassed by the NEBBTC jurisdictions. Historically, the entire Northeast was occupied by black bears (Hall 1981). At present, no resident black bear populations exist in Delaware or Prince Edward Island (Figure 2.3), but during the past 2 decades bear range has expanded in Connecticut, Massachusetts, New Jersey, New York, Pennsylvania, Rhode Island, Virginia, and West Virginia, and sightings have been reported recently in Delaware (Scheick et al. 2012).

2.4 BEARS, HUMANS, AND MANAGEMENT

High and increasing human densities in the Northeast, combined with increasing bear numbers and close proximity of bears to humans in many areas, have humans and bears on a collision course. Twelve of 17 jurisdictions in the Northeast occupied by black bears reported increasing human-bear conflicts between 2000 and 2010 (Noyce 2012). Black bears are omnivorous and are often drawn to anthropogenic foods such as garbage, birdfeeders, and agricultural crops. Although black bear attacks on humans are rare, incidents do occur and have resulted in injuries or even death, costly litigation, and negative perceptions of bears by the public. Conflicts
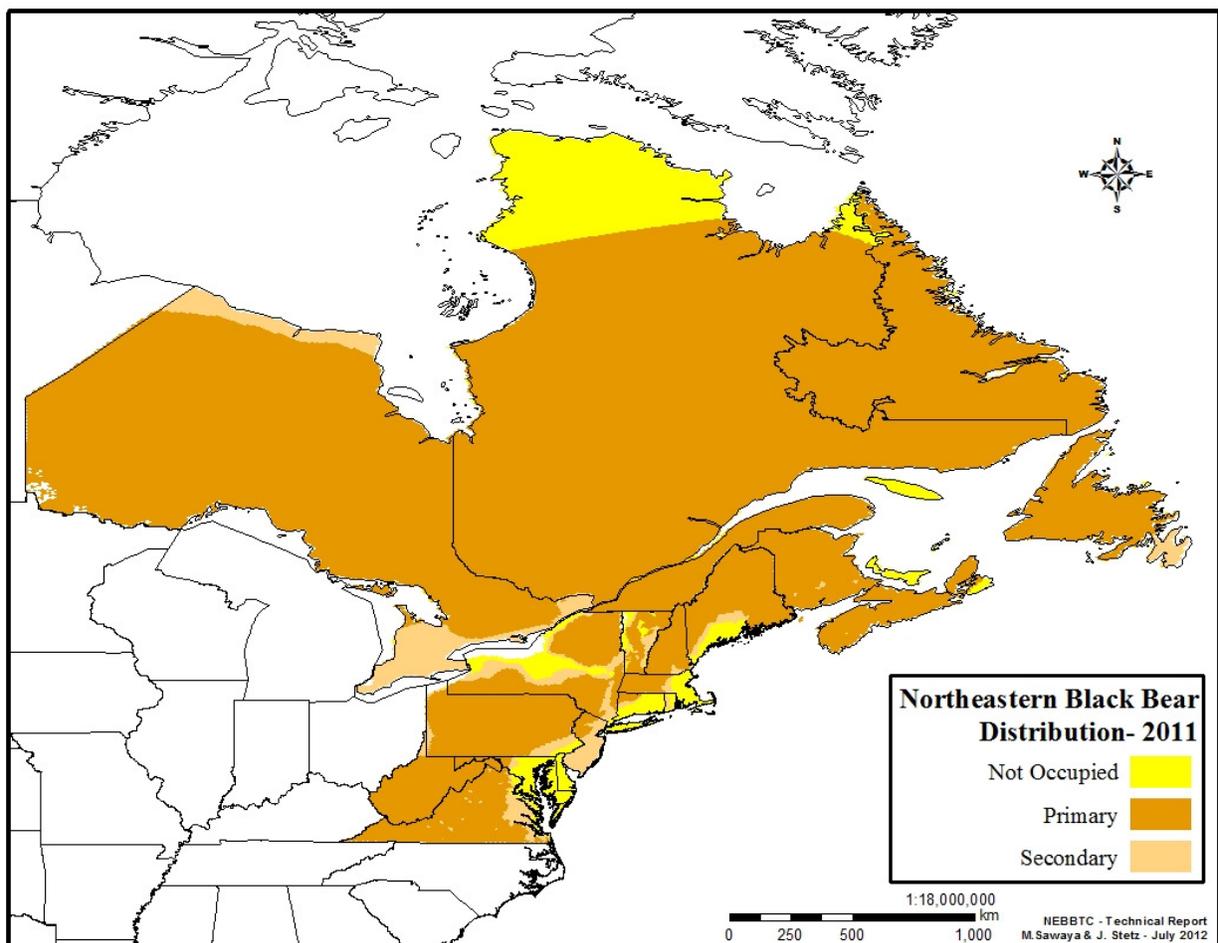
Figure 2.3: *Primary and secondary black bear range in northeastern U.S. and eastern Canada. From Scheick et al. (2012).*

between humans and bears are inevitable where their activities overlap. Also, vacation or retirement properties owned by urban residents are becoming more commonplace in the Northeast and they often have little experience dealing with wild animals. Such changing dynamics present many challenges to black bear managers in the Northeast.

Finding socially-acceptable management solutions to bear-human conflicts is a major priority for black bear managers in North America. Bear hunting has a long history in the Northeast and is a traditional and established form of population management. Hunting is viewed by many as a safe, effective, and responsible mechanism to reach and maintain population objectives for black bears. Of the 40 U.S. and 12 Canadian provinces and territories in North America that have black bear populations, 42 rely on hunting as their primary management tool (Hristienko and McDonald 2007). Bear hunting in some form is permitted in 14 of the 17 NEBBTC jurisdictions with bears (Noyce 2012). Other population management tools have been proposed (e.g., immunocontraception, relocation) but few, if any, are deemed adequately successful or economically feasible at the spatial and temporal scales of concern to management agencies.

Hunting can be a significant source of funding for wildlife conservation and management activities in NEBBTC jurisdictions (Figure 2.4). In Pennsylvania for example, residents are required to purchase bear hunting licenses that total more than $2 million in revenue per year. Also, conservation funds are generated in the U.S. from a tax on sporting arms and ammunition (i.e., Federal Aid in Wildlife Restoration Act (16 U.S.C. 669-669i; 50 Stat. 917) of September 2, 1937, commonly called the "Pittman-Robertson Act."). These funds may be applied toward habitat management, land acquisition, conflict reimbursement programs, research, or other activities to foster wildlife management and conservation.

Although hunting remains the primary mechanism for bear management in most areas, hunting participation and revenues are declining across most statesin the region (Figure 2.5), as across most of North America (U.S. Department of Commerce 2006). From 1996 to 2006, hunter numbers declined by 3.7% in the U.S. (U.S. Census Bureau 2010), leading many states to increase license fees. Such trends have been most distinct in Maine, Massachusetts, and Rhode Island with a 35–62% decline in hunting revenues and a concomitant net loss of over 300,000 hunters. These trends are even more striking considering that, on average, human populations in the Northeast grew by approximately 6% during this period (U.S. Census Bureau 2010).
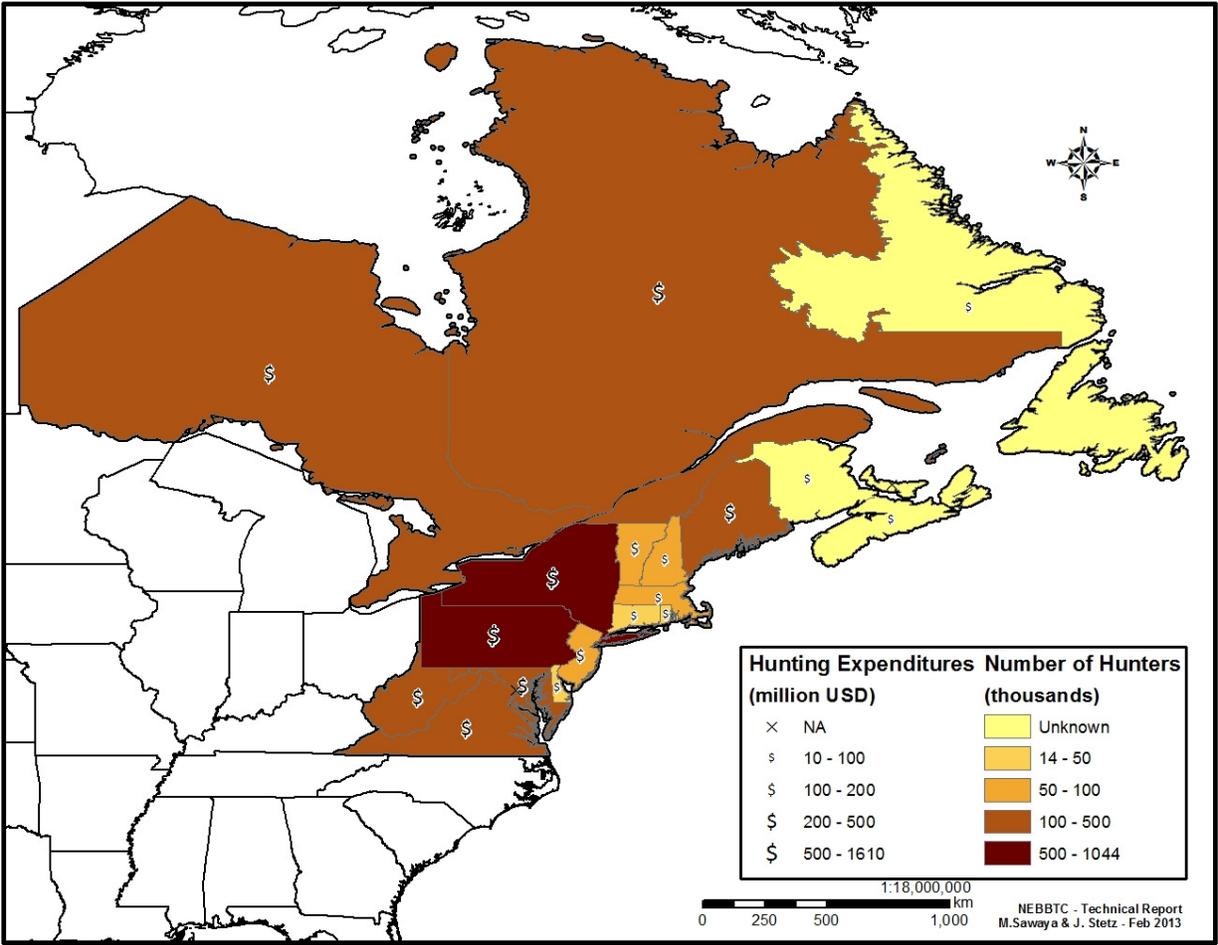
Figure 2.4: *Summary of hunting expenditures (U.S. $) and number of hunters in northeastern U.S. and eastern Canada.*

Over 70% of NEBBTC bear managers stated that hunting is the most important management tool available (Chapter 6). However, increasing and expanding black bear populations in NEBBTC jurisdictions, combined with decreasing hunter participation and revenue, present substantial challenges to successful black bear management in the Northeast. These uncertainties lead to increased urgency to identify and apply scientifically reliable and cost-effective monitoring techniques to meet black bear management objectives. Ultimately, managers need to be able to link population parameter estimates with the drivers of population change to effectively implement adaptive management (Nichols and Williams 2006).
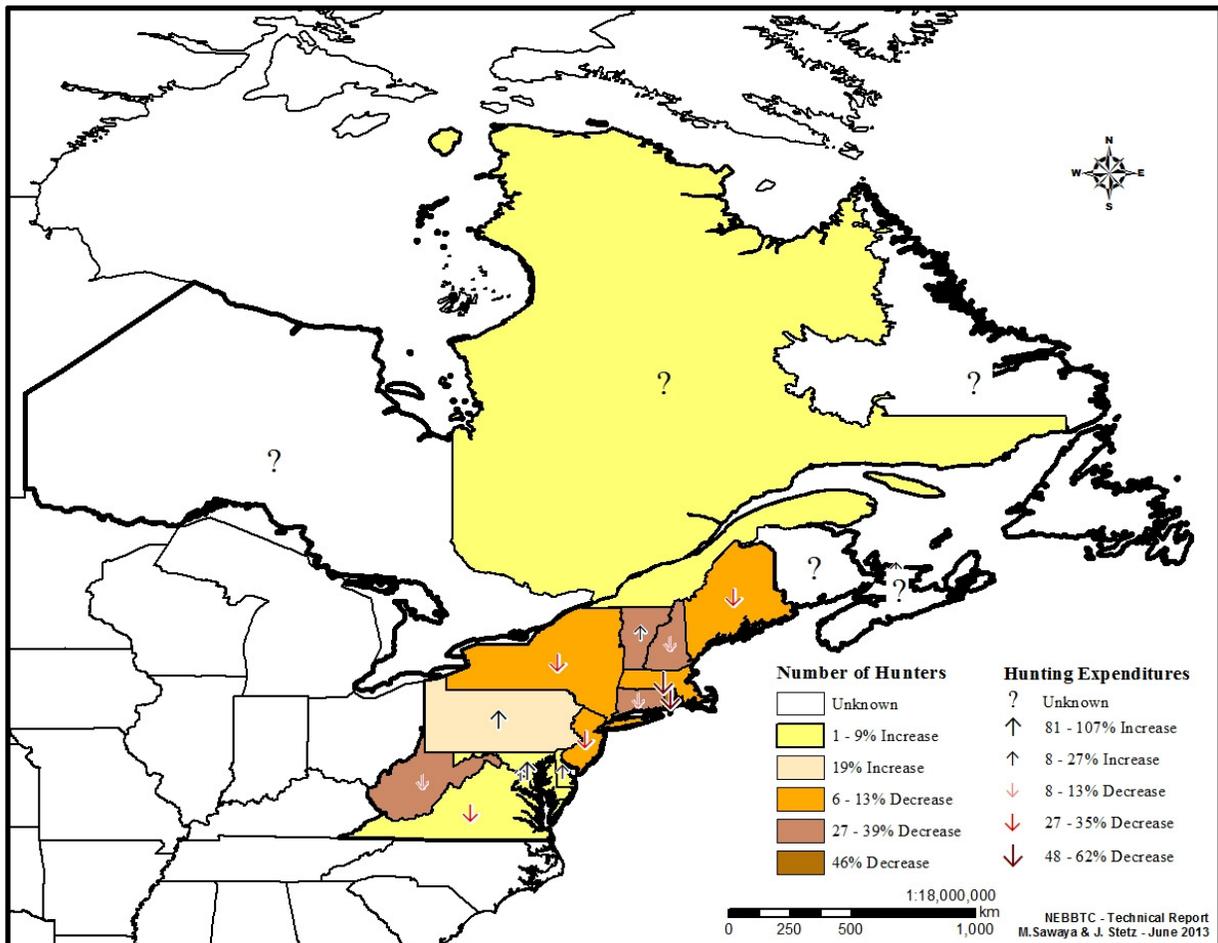
Figure 2.5: *Summary of trends in hunting expenditures (U.S. $) and number of hunters in the northeastern U.S. and eastern Canada.*

Section II. Biology

Chapter 3. Phylogeny and Description

3.1 NOMENCLATURE

Common name: American black bear (English), Ours noir (French), Oso negro (Spanish)

Scientific name: *Ursus americanus* Pallas, 1780

3.2 PHYLOGENY

Recently resolved genetic phylogenies suggest a rapid expansion of bear species, along with many other taxa, during a time of dramatic global climatic change approximately 6 million years ago at the Miocene-Pliocene boundary (Krause et al. 2008). During this radiation, the common ancestor of the Asiatic black bear (*U. thibethanus*), American black bear, and the sun bear (*Helarctos malayanus*) separated from the group containing the sloth bear (*Melursus ursinus*), brown bear (*U. arctos*), polar bear (*U. maritimus*), and cave bear (*U. spelaeus*). Genetic evidence suggests that the Asiatic and American black bears split approximately 4.1 million years ago and are considered sister taxa, although archeological data and other predictions place this divergence prior to the first opening of the Bering Strait around 5.3 million years ago (Krause et al. 2008). Following the extinction of cave bears, American black bears were likely the only bear species occupying North America until brown bears presumably traversed the Bering Strait approximately 12,000 years ago. This genetic legacy and the rapid radiation that led to modern species is evidence of the success of Ursids in general and highlights the sometimes cryptic differences between species that go deeper than their names suggest.

3.3 DESCRIPTION

The American black bear varies in color across its range, from black in much of the eastern U.S. and Canada (Figure 3.1), to cinnamon or blonde in southwestern states, to bluish in parts of Alaska, and the striking, nearly white pelage of Kermode bears of British Columbia (Pelton 2003). Several color phases can be found within the same regions. In some areas, black bears are difficult to distinguish from sympatric grizzly bears.

The American black bear is the smallest of the 3 extant bear species in North America, but shows substantial variation in size, reflecting regional differences in habitat quality. Adult bears are 120–200 cm in length and 70–105 cm shoulder height. The average mass of adult male black bears is 68–158 kg, although males with access to abundant hard mast or agricultural or anthropogenic foods may exceed 365 kg. Black bears are sexually dimorphic, with females approximately two-thirds the size of males and rarely exceeding 175 kg. Seasonal variation in abundance and quality of food resources also impacts bears, with body mass in autumn approximately 30% greater than in spring.

Figure 3.1: *American black bear.*
Photo credit: Maine Department of Inland Fisheries and Wildlife

Chapter 4. Life History and Population Dynamics

4.1 DENNING

Black bears typically begin denning in October or November, depending on when adequate fat reserves are obtained and how long food remains available. Hellgren and Vaughan (1989) reported that pregnant females in the Great Dismal Swamp of Virginia and North Carolina denned for 119 days, which was among the shortest of recorded denning periods for areas with similar climates and likely reflected habitat quality of this area. Black bears are known to temporarily leave their dens during winter in search of food or to not den at all when in poor condition (Hamilton and Marchinton 1980). Alternately, if adequate food is available, denning is not necessary other than to give birth. Adult males are usually the last to enter and first to exit their dens, a trait that has implications for how spring and fall hunts are structured in areas where female mortality is a concern (Weaver and Pelton 1994).

Figure 4.1: *American black bear family in winter den.*
Photo credit: New York Department of Environmental Conservation

Black bear den selection can vary by region, sex, individual selection, and reproductive status. Bears use many types of dens including tree cavities, caves, small depressions, culverts, excavations, brush piles, open ground nests, or similar locations (Figure 4.1). Black bears exhibit a specialized form of hibernation, enabling them to remain dormant for long periods of time without eating, drinking, urinating, or defecating (Lundberg et al. 1976, Nelson et al. 1983). Compared with other hibernating species, black bears are specialized because they are able to recycle nitrogen waste from metabolic processes into proteins, which aids in the prevention of muscle and bone loss that occurs with most animals during long sedentary periods (Harlow 2012). Depending in large part on their reproductive status, bears may lose 25–40% of their body weight during hibernation, due partly to the fact that bears maintain constant body temperature around 35°C, unlike "true" hibernators. The heart rate of hibernating bears drops from an average of 40–50 beats/minute to approximately 8 beats/minute. Because black bears remain relatively alert, predation during the denning period is limited but has been observed (e.g., Rogers and Mech 1981). Denning presents researchers with a unique opportunity to capture black bears with limited risk of injury to the bears or humans, and provides the best opportunity to observe true litter sizes from which cub survival can be reliably estimated (Vashon et al. 2003).

4.2 REPRODUCTION

Black bear females reach sexual maturity between 2 and 7 years of age and enter into estrus during summer, with the peak occurring in late June and July (Pelton 2003). Black bears are promiscuous, meaning that a single male can mate with multiple females and vice versa. Consequently, adult females are the primary drivers of population growth (Lariviere 2001).

Females can mate during the same summer that their cubs disperse, potentially sooner if the litter is lost. As with most components of black bear life history, age of sexual maturity (i.e., primiparity) differs by habitat quality and density (Czetwertynski et al. 2007). Generally, bears in eastern North America mature earlier, with age at primiparity averaging 4.46 years (95% credible interval 4.02–4.96) versus 5.58 years (95% credible interval 5.06–6.07) in the west (Beston 2011). Further, reproductive synchrony (i.e., birth pulses) may occur because of hard mast crop failures or similar environmental events, as has been documented in the Allegheny Mountains of western Virginia (Bridges et al. 2011).

Black bears, like all bears with the possible exception of the sun bear, exhibit delayed implantation, a physiological adaptation that suspends development of the fertilized egg (blastocyst) for several months. If during this time the female is not able to accrue necessary fat reserves, the blastocyst is reabsorbed or cubs may be born but consumed by the mother (M. Vaughan, U.S. Geological Survey, personal communication). While in the den, adult female

Figure 4.2: American black bear cubs of year.
Photo credit: Maine Department of Inland Fisheries and Wildlife

bears give birth to 1–6 cubs (Figures 4.2 and 4.3), typically during late January–early February, after a gestation period of 60–70 days. In the northeastern part of their range, mean litter sizes range from 2.3 to 3.0 cubs (Bridges et al. 2011).

Cubs are born blind, nearly hairless, and weigh approximately 360 g. Black bear milk averages 33% fat which is about 10 times that found in human milk. Cubs are dependent on milk for at least 22 weeks, but nurse for 30 weeks on average. Upon exiting the den, bears will wander in search of food, although it is often initially scarce, and they therefore may occasionally return to the den. Cubs grow to 18–27 kg within their first 6 months, although about half will not survive their first year, with male cubs having greater mortality rates than females (Elowe and Dodge 1989). Cubs remain with their mother for approximately 1.5 years after birth, during which time they learn how to obtain food, interact with other bears, and, in many parts of black bear distribution, how to coexist with humans. This is a critical phase that sets the stage for a bear's life and eventual fitness.

4.3 POPULATION DYNAMICS

Fall hard mast failure influences milk fat content in spring, but not cub survival because females can switch to alternative food sources and still produce milk of sufficient quality to nourish cubs (McLaughlin et al. 1994, McDonald and Fuller 2005). Primary mortality sources for cubs are starvation, predation, and human causes, such as vehicle collisions and management removals. Although controversial, the sexually selected infanticide hypothesis has been supported in some populations after experimental removals of adult males. Czetwertynski et al. (2007) reported older age of first reproduction and lower cub survival (66%) in higher-density areas that were not hunted compared with lower-density areas where hunting occurred (83%).

Hunting is an obvious driver of survival rates in hunted populations (e.g., Hellgren and Vaughan 1989, Schwartz and Franzmann 1991, Beringer et al. 1998, Koehler and Pierce 2005, Czetwertynski et al. 2007) but other vital rates can be affected as well. For example, hunted populations can shift to younger ages of sexual maturity and greater investments in reproduction that compensate for elevated mortality (Czetwertynski et al. 2007, Beston 2011). Such shifts, including those driven by non-harvest mortality, have important implications for long-term management because adult female survival has been a primary focus of conservation and management efforts, including more conservative harvest quotas for females (Beecham 1983, Mitchell et al. 2009).

The availability of food resources is also a critical determinant of abundance and density of bear populations. Clearly, areas with abundant foods can support a greater density of bears

Figure 4.3: *Female American black bear with 5 cubs of year.*
Photo credit: T. Sears

than those where food production is poor. Variation in biological carrying capacity of a region can be inferred from the range of population densities observed in black bears. In the western part of their range, densities as low as 0.8 bears/100 km^2 exist in areas of low productivity that also contain grizzly bears (Hebblewhite et al. 2003). Conversely, higher-quality habitats, although still with resident grizzlies, may support 45.0 bears/100 km^2 (Jonkel and Cowan 1971). In the East, however, densities range from 10.5 bears/100 km^2 in Maryland (Bittner et al. 2013) to 201 bears/100 km^2 in small areas in eastern North Carolina (van Manen et al. 2012).

Life-stage simulation analyses and other methods have been used to estimate the relative effects that individual vital rates have on population growth (i.e., elasticity; Wisdom et al. 2000). Like most large mammals (Heppell et al. 2000), population growth rates of black bears are driven primarily by adult female survival (Beston 2011). The strength of this relationship, however, may vary across populations, as can the drivers of the vital rates themselves. For example, Beston (2011) found an 18% difference in elasticity estimates of adult female black bear survival between western and eastern populations (0.67 and 0.55, respectively). Conversely, cub and yearling survival, along with fecundity, had greater elasticities in eastern populations (Beston 2011). Such differences are likely caused by more abundant and higher-quality foods in the East, allowing females to produce more offspring beginning at an earlier age, but may be confounded by greater human-caused mortality (Czetwertynski et al. 2007). The meta-analysis of Beston (2011) using data from 1959–2007 suggested that 55% of eastern black bear populations were increasing compared with 34% of western populations. With existing high densities of bears, increasing numbers of bears, and increasing human populations, challenges to bear and bear-human conflict management in the East will likely continue to increase as well.

Section III. Management

Chapter 5. Population Management Strategies

5.1 MANAGEMENT GOALS

Goals for black bear management in North America range from protection of imperiled

populations, to maximization of sustainable yield, to population reductions for nuisance bear

control (Miller 1990). Harvest is the primary tool for agencies to manage black bear populations

(Figure 5.1), with sustainable population objectives and quotas playing prominent roles in bear

management plans (Hristienko and McDonald 2007). Public input is often considered along with

biological data when setting quotas because most black bear populations are managed based on

social, rather than biological, carrying capacity. State and provincial governments in North

America are usually responsible for drafting legislation to regulate bear hunting. Wildlife

agencies in the Northeast typically use a best-available-science approach along with adaptive

Figure 5.1: *Harvested black bear at game check station.*
Photo credit: Pennsylvania Game Commission

management to achieve population targets. Although black bear management is a high priority for many wildlife agencies, constraints to bear population management include inadequate habitat protection, political pressures, technological limitations of available population management techniques, and inadequate financial support for management (Miller 1990). Consequently, many bear management plans are driven as much by financial and logistical considerations as they are by biology.

Setting meaningful objectives for population parameters (e.g., abundance, survival, and population growth) is fundamental to effective black bear management. In the past two decades, adaptive resource management has gained widespread recognition as a flexible and rigorous way to ensure management objectives are met (Williams et al. 2002). Adaptive management relies on setting population objectives and developing a set of monitoring and management programs that iteratively adapt to changing population dynamics. The first major step for managers in this process is to establish scientifically-defensible population objectives based on social, financial, demographic, and habitat-based criteria. Although population targets are often difficult to achieve because of uncertainty of population status and harvest rates, clearly defined targets provide important perspective for long-term management efforts and can be used to objectively evaluate progress being made with management plans.

5.2 HARVEST MANAGEMENT

Harvest is the primary tool used by most bear managers to reduce human-bear conflicts in the Northeast (Chapter 7). However, establishing harvest regulations for black bears is complicated because multiple agencies are often involved in the management of this wide-ranging species. Legal authority for black bear management varies among jurisdictions and can be vested at many

levels (i.e., community, state or province, national). Bears inhabiting public lands are managed

by agencies within the Department of the Interior (e.g., National Park Service, U.S. Fish and

Wildlife Service, Bureau of Land Management) or Department of Agriculture (U.S. Forest

Service) in cooperation with state and provincial management agencies. Most wildlife

regulations passed by state legislatures or independent commissions apply to all lands within

those jurisdictions, including federal, state, and private lands. Although public and private

landowners can restrict hunting access, they cannot implement more liberal regulations than the

states. Harvest may also be limited near human-dominated areas due to firearms restrictions.
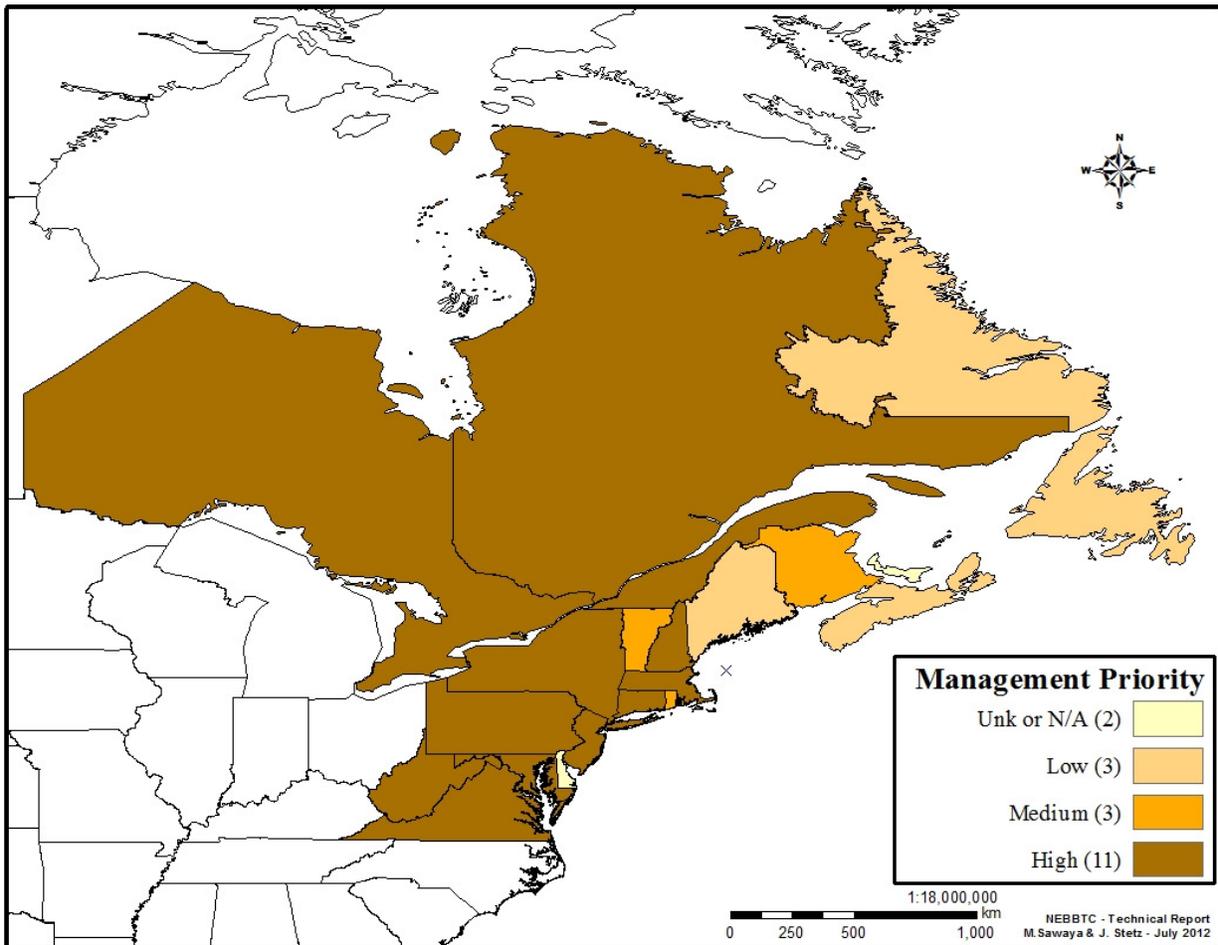
Figure 5.2: *Northeast human-bear conflict management priority by jurisdiction (Noyce 2012).*

Human-bear conflict management is a high priority in 12 out of 19 NEBBTC jurisdictions (Figure 5.2; Appendix B) and bears are managed by wildlife departments within state or provincial natural resource agencies region wide. Hunting regulations influence what, how, when, and where wild game may be taken. Because the vulnerability of black bears to harvest is heavily influenced by sex, time of year, harvest techniques (e.g., hounds, baiting, archery), and the length of hunting seasons, hunting regulations can have dramatic effects on bear population dynamics. Choosing appropriate harvest techniques and establishing sustainable harvest rates are two crucial steps to effective black bear management and conservation.
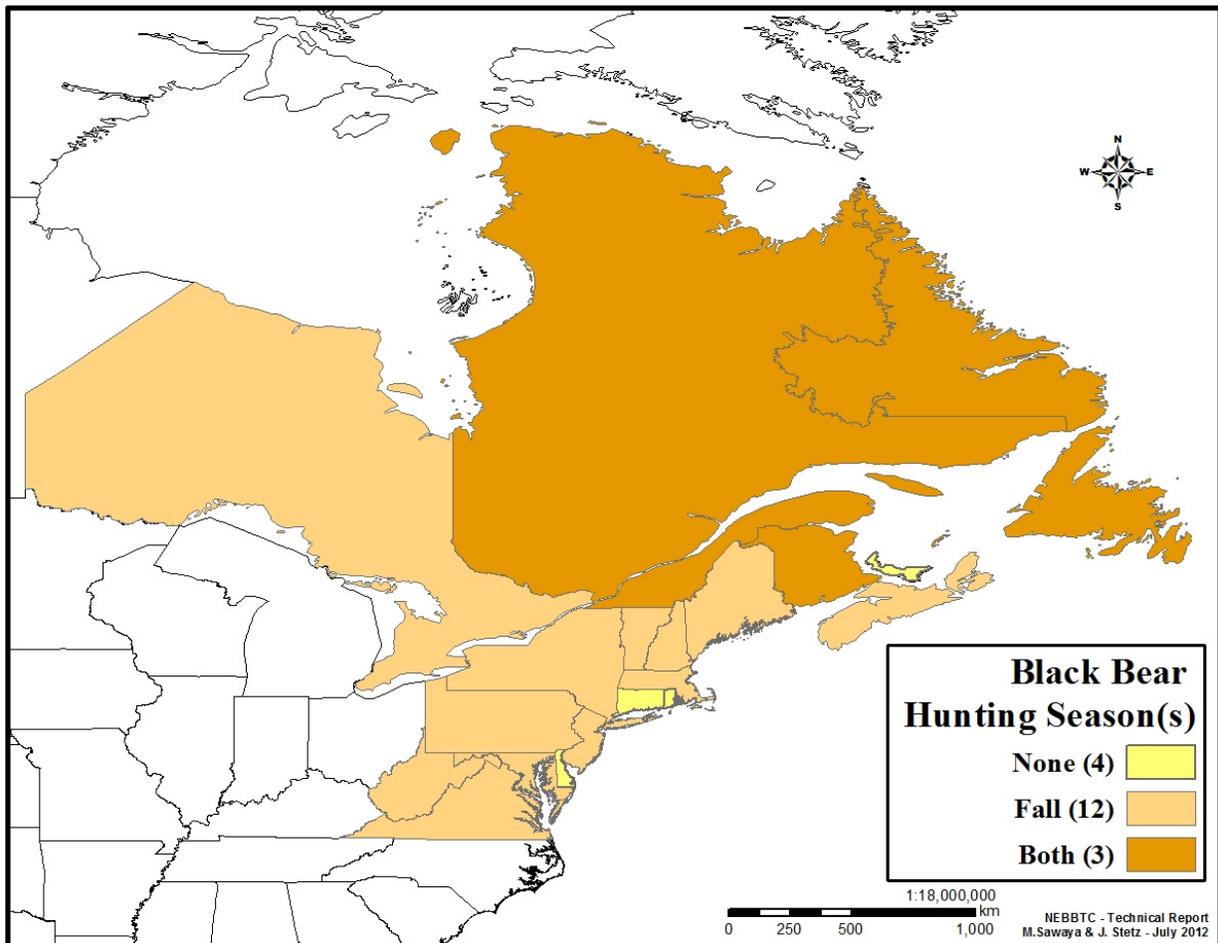
Figure 5.3: *Northeast jurisdictions that permit spring or fall bear hunting seasons (Noyce 2012).*

5.3 HARVEST REGULATIONS

5.3.1 Timing of Season

Throughout its range, American black bear harvest generally consists of spring hunts, fall hunts, or both. Spring hunts typically provide better quality pelts and more palatable meat, whereas fall hunts have lower hunter success but yield much larger bears (Ternent 2006). The timing of hunting seasons can also be used to control the ratio of males to females in the harvest or to reduce the susceptibility of females with cubs, by capitalizing on differences in the duration of denning between sexes. Of the 19 NEBBTC jurisdictions, 4 do not allow harvest, 12 have a fall hunting season, and 3 have both spring and fall hunting seasons (Figure 5.3).

5.3.2 Season Length

Duration of season affects harvest by exposing bears to varying hunting pressure and effort. Season length varies greatly among jurisdictions, lasting from 13 weeks in Maine to just a few days in Pennsylvania. Individual state and provincial regulations may also influence actual hunter days even when season lengths are equal. For example, 8 of the 11 states in the Northeast do not permit black bear hunting on Sundays so hunters may only be in the field 6 days per week. Northeast states that prohibit all Sunday hunting, with few exceptions, are Connecticut, Delaware, Maine, Maryland, Massachusetts, New Jersey, Pennsylvania, and Virginia.

5.3.3 Harvest Method

Various methods exist for harvesting American black bears in the Northeast. Method of take can have vastly different effects on hunter success and harvest selectivity. The use of baits to attract bears can increase hunter success and enable hunters to be selective. Baiting bears,

however, is controversial because of social concerns about animal welfare (i.e., feeding wildlife), environmental concerns (i.e., bait sites resembling garbage dumps), and the potential to habituate bears, which could lead to increased human-bear conflicts. Almost half (47%) of NEBBTC jurisdictions allow the use of bait to hunt black bears (Figure 5.4).

Another method for harvesting black bears is using trained hounds to chase bears until they climb trees or bay on the ground, where they may then be taken by hunters. The use of hounds increases hunter success because hounds can detect and track bears through dense vegetative cover with greater efficiency than humans. Hunting bears with hounds can allow
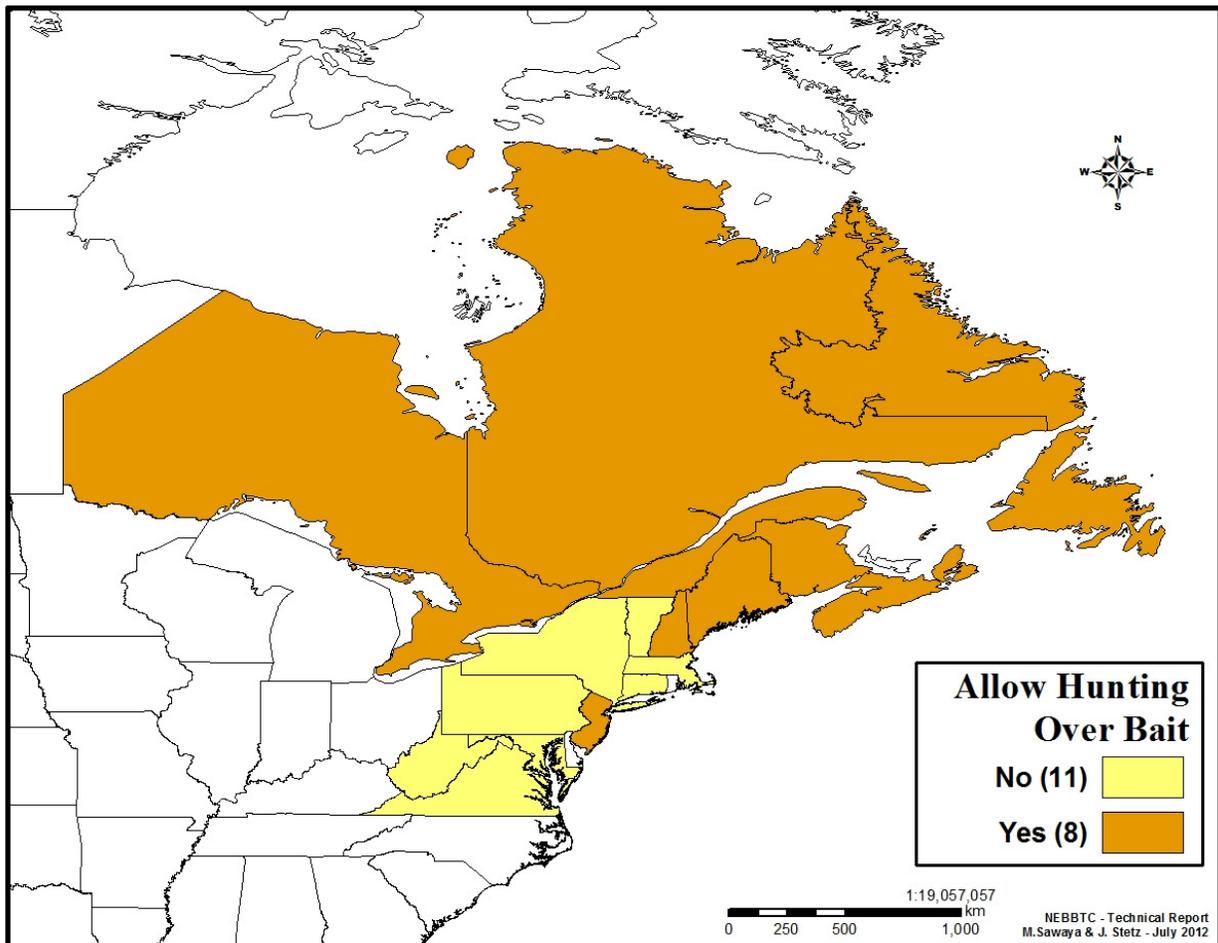
Figure 5.4: *Northeast jurisdictions that permit the use of bait for hunting black bears (Noyce 2012).*

for greater selectivity, like the use of baits, because hunters may have multiple opportunities to determine whether to harvest a particular animal. Hunting bears with hounds is also controversial because of concerns about fair chase and animal welfare (e.g., exhaustive pursuits, injuries from jumping out of trees). Additionally, hound hunting generally comes with a dog training season and there is controversy as to whether these training seasons negatively affect bears, particularly cubs. Almost 1/3 of NEBBTC jurisdictions allow the use of hounds for hunting (Figure 5.5).
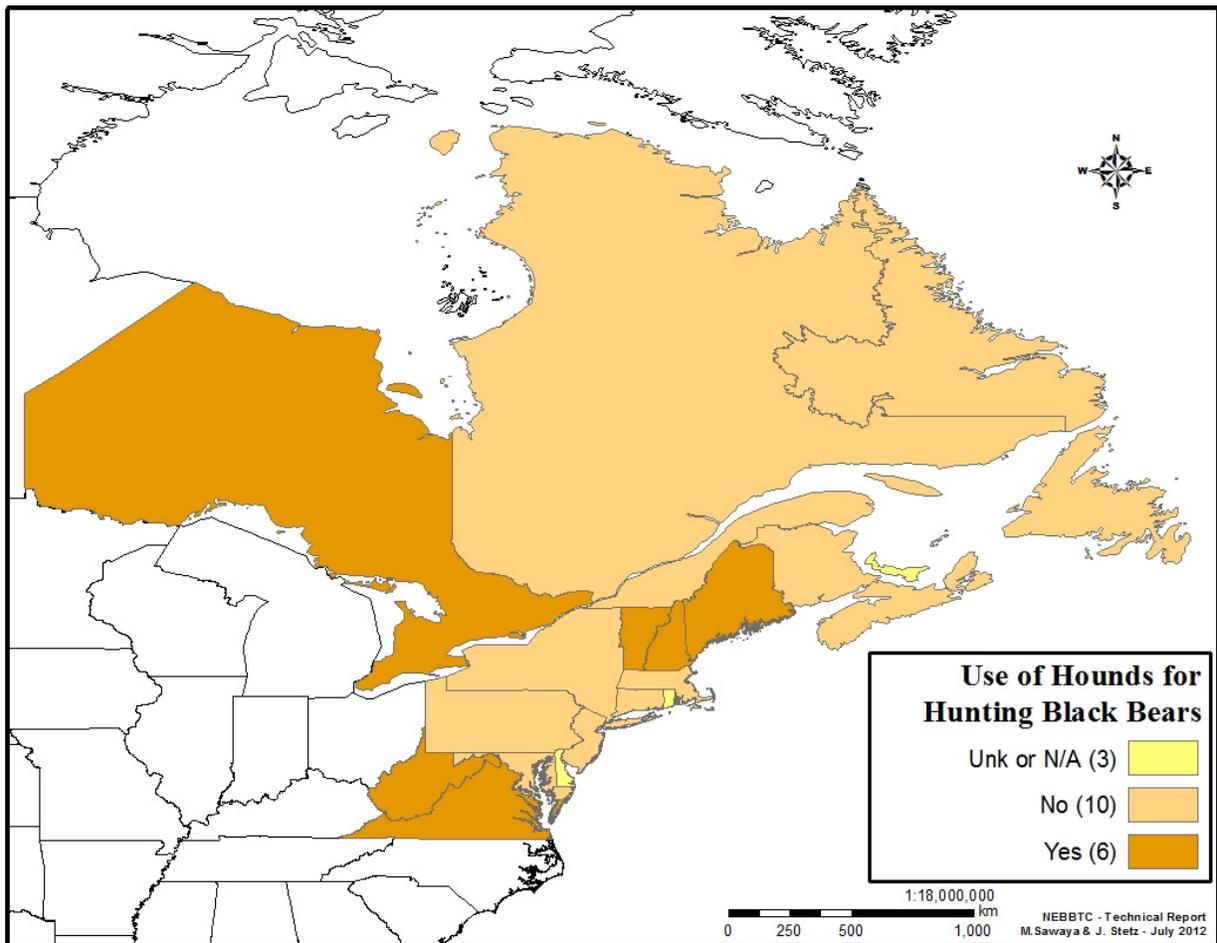
Figure 5.5: *Northeast jurisdictions that permit the use of hounds for black bear harvest (Noyce 2012).*

5.3.4 Bag Limit

Bag limits establish the number of animals a hunter may harvest, which is typically 1 bear per hunter per hunting season among NEBBTC jurisdictions. Although this has some effect on the total number of bears harvested, bag limits have the broader effect of distributing the harvest among a greater number of hunters. Bag limits also determine the allowable sizes (i.e., age classes) and sexes that may be harvested. In the Northeast, most bag limits permit the taking of any bear of either sex older than a cub of the year, although most jurisdictions prohibit the taking of adult females accompanied with young of any age. Some states (e.g., Pennsylvania, Massachusetts) do not have bag limit distinctions based on age or sex.

5.3.5 Permit System

Black bear harvest regulations in the Northeast are implemented through different types of hunting permits. The total number of animals allowed to be harvested may be set as a fixed number or percentage (i.e., quota) of the estimated population and hunting opportunities may be allocated through licenses or special permits. Many wildlife management agencies require anyone hunting bears to purchase a black bear license costing $5–50, depending on the jurisdiction. Some states also require a conservation or general hunting license as a prerequisite to buying a black bear license. Multi-species licenses are offered in some jurisdictions, usually combination packages for deer (*Odocoileus virginianus*), turkey (*Meleagris gallopavo*), and black bear. Combining opportunities for harvesting game species usually results in discounted license fees, thus increasing overall license sales and harvest rates. Baiting does not require additional licenses to be purchased, but some jurisdictions that allow the use of hounds also require a special hound hunting license in addition to a bear tag.

Chapter 6. Black Bear Status by Jurisdiction

6.1 MANAGEMENT STATUS

The history of black bear recovery, habitat quality, and human demographics are diverse across the Northeast so bear management status varies widely. Two NEBBTC jurisdictions (i.e., Delaware, Prince Edward Island) have no resident bear populations, whereas 16 of 19 currently classify black bears as game species (Noyce 2012). Large states such as New York and Pennsylvania have played particularly important roles as reserves and source populations for surrounding states. Smaller states that contain remnant bear habitat and high densities of humans (i.e., Connecticut, Delaware, Rhode Island), have correspondingly small bear populations and relatively few conflicts, whereas medium-sized states with larger bear populations and higher human densities (e.g., New Jersey, Massachusetts) tend to have many conflicts. The southern portions of Canadian provinces tend to have far greater numbers of bear-human conflicts than those in the north because human densities (and probably bear densities) are typically much greater along the border with the U.S.

The following jurisdiction-by-jurisdiction summary of bear management status is based on a recent summary of a survey of wildlife management agencies by Noyce (2012), who adapted methods by Hristienko and McDonald (2007). Standardized surveys improve inferences across diverse jurisdictions where information may be collected or reported in different formats. We calculated bear density (no. bears/100 km^2) by dividing the estimated number of bears for each jurisdiction as reported in Noyce (2012) by total occupied area as determined by Scheick et al. (2012).

6.1.1 Connecticut

The Department of Energy and Environmental Protection is responsible for black bear management in the state of Connecticut. No management plan currently exists for black bears in Connecticut (Noyce 2012). Only northwest Connecticut is primary black bear range and the population is contiguous with southwest Massachusetts and eastern New York. Bear range is expected to expand across much of the state (Figure 6.1). Research on the growing bear population has been conducted for 10 years. No black bear hunting is currently allowed in Connecticut.
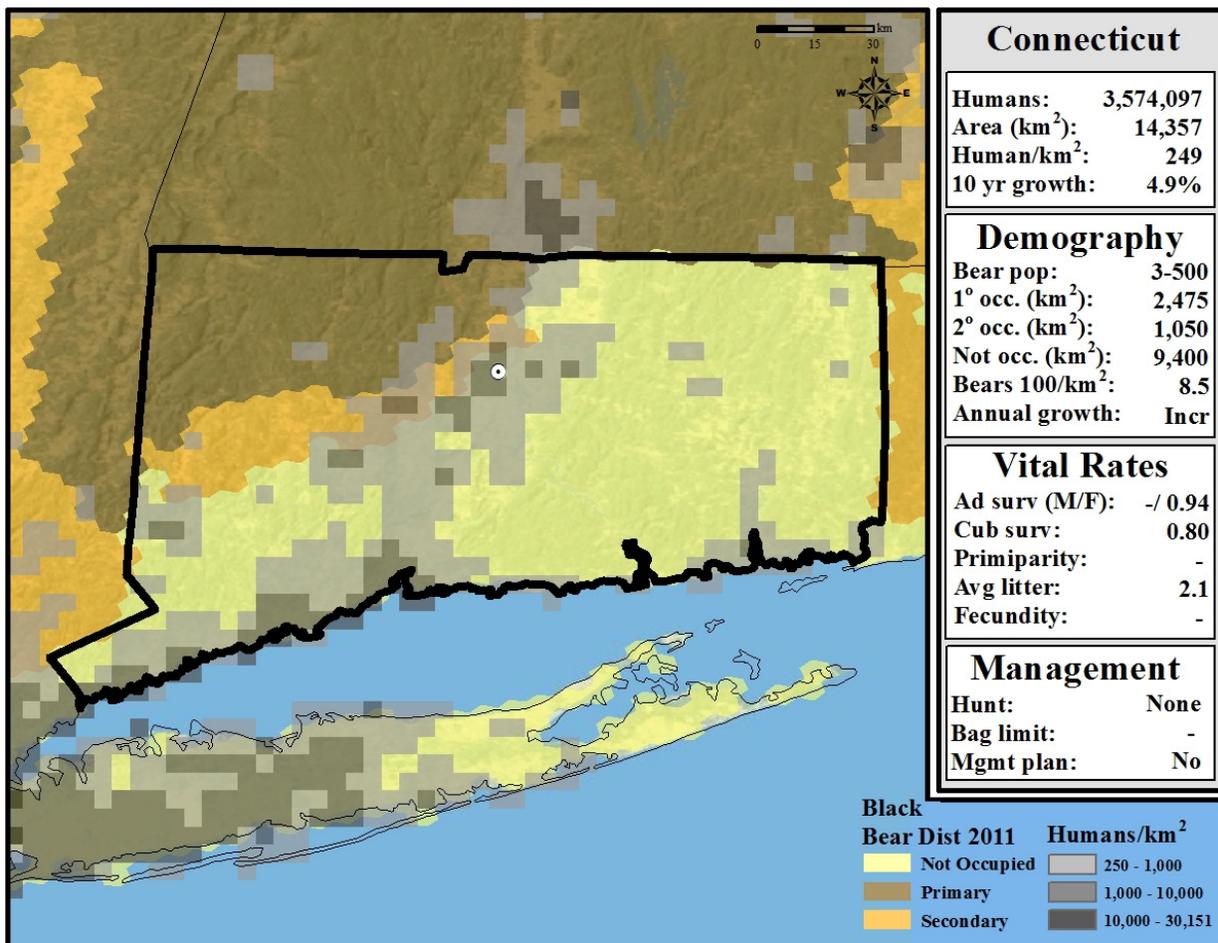
Figure 6.1: *Summary information for Connecticut's human and black bear populations. Black bear primary and secondary range from Scheick et al. (2012).*

6.1.2 Delaware

The Division of Fish & Wildlife is responsible for black bear management in the state of Delaware. None of Delaware is primary or secondary black bear range because of high human densities. The Philadelphia metropolitan area and its suburbs most likely restrict immigration from southern New Jersey (Figure 6.2). Delaware has no established bear population and no black bear research has been conducted in the state to date. Consequently, Delaware does not have a long-term bear management plan (Noyce 2012) and no black bear hunting is allowed.

Delaware

Humans:	897,934
Area (km^2):	6,447
Human/km^2:	139
10 yr growth:	14.6%

Demography

Bear pop:	0
1° occ. (km^2):	0
2° occ. (km^2):	0
Not occ. (km^2):	5,125
Bears 100/km^2:	0
Annual growth:	N/A

Vital Rates

Ad surv (M/F):	-
Cub surv:	-
Primiparity:	-
Avg litter:	-
Fecundity:	-

Management

Hunt:	No
Bag limit:	N/A
Mgmt plan:	No

Black Bear Dist 2011

Not Occupied		Humans/km^2
Not Occupied		250 - 1,000
Primary		1,000 - 10,000
Secondary		10,000 - 30,151
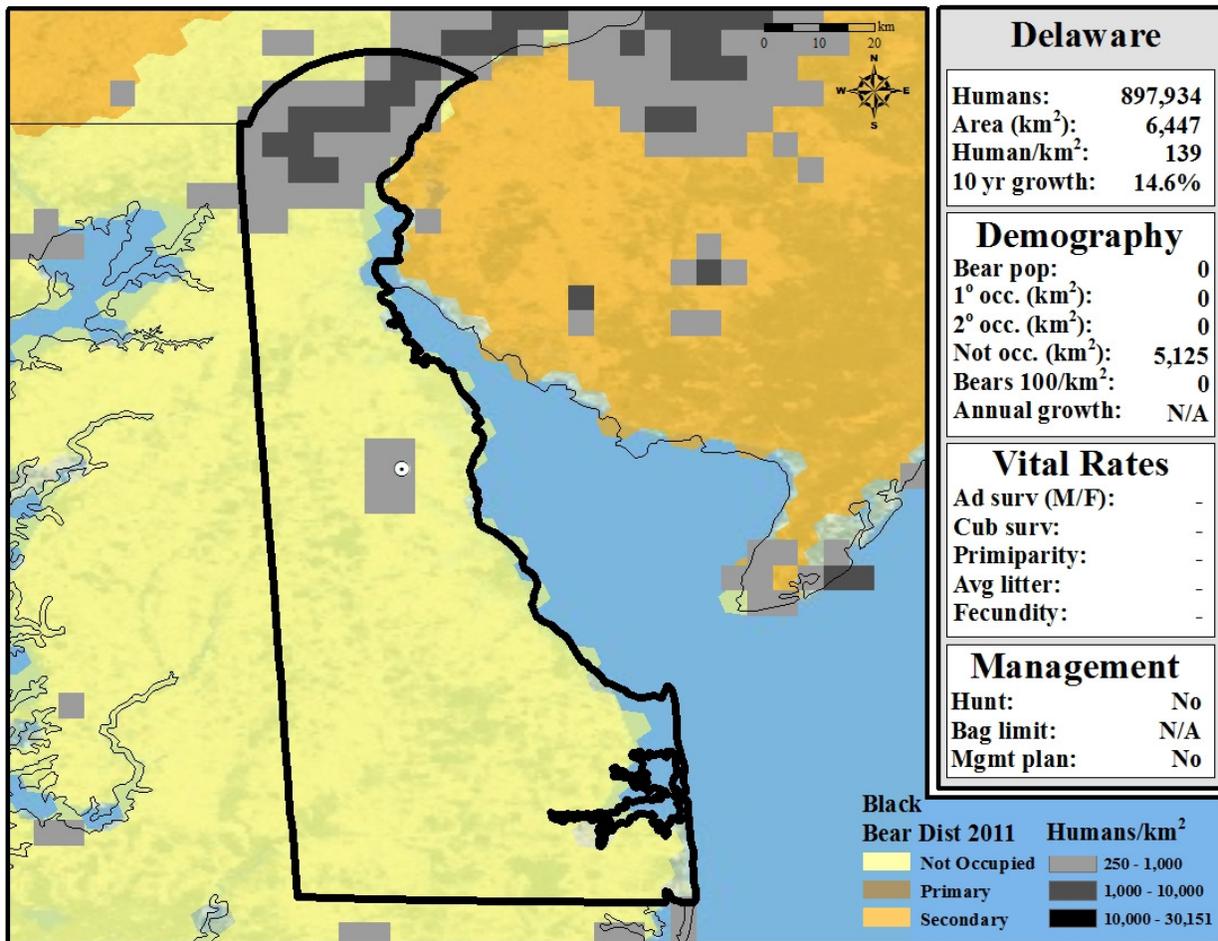
Figure 6.2: *Summary information for Delaware's human and black bear populations. Black bear primary and secondary range from Scheick et al. (2012).*

6.1.3 Maine

The Department of Inland Fisheries and Wildlife is responsible for black bear management in the state of Maine. Black bear management in Maine is guided by a long-term plan. Almost all of Maine is primary black bear range, with only the southern coastal portion of the state unoccupied (Figure 6.3). Maine has been the site of ongoing black bear research using radiotelemetry since 1975 (Barrett 2011). Hunters in Maine are allowed to harvest 1 bear/person in the fall. With >30,000 black bears, Maine has more bears than any state in the Northeast and populations continue to increase (Noyce 2012).

Maine

Humans:	1,328,361
Area (km^2):	91,646
Human/km^2:	15
10 yr growth:	4.2%

Demography

Bear pop:	31,000
1° occ. (km^2):	68,175
2° occ. (km^2):	5,675
Not occ. (km^2):	9,975
Bears 100/km^2:	42.0
Annual growth:	Incr

Vital Rates

Ad surv (M/F):	-/0.86
Cub surv:	0.59-0.79
Primiparity:	4.5
Avg litter:	2.2
Fecundity:	0.71

Management

Hunt:	Fall
Bag limit:	2
Mgmt plan:	Yes

Black Bear Dist 2011
- Not Occupied
- Primary
- Secondary

Humans/km^2
- 250 - 1,000
- 1,000 - 10,000
- 10,000 - 30,151

Figure 6.3: Summary information for Maine's human and black bear populations. Black bear primary and secondary range from Scheick et al. (2012).

6.1.4 Maryland

The Wildlife and Heritage Service is responsible for black bear management in the state of Maryland. Most of western Maryland is primary black bear range, except for areas encompassing the metropolitan areas of Baltimore and Washington and the Delaware-Maryland-Virginia peninsula (Figure 6.4). Maryland has been the site of telemetry- and DNA-based black bear research. Maryland's population of 600 bears is increasing (Noyce 2012) and black bear management is guided by a long-term plan. Hunters in Maryland are allowed to harvest 1 bear/person during a fall hunt.
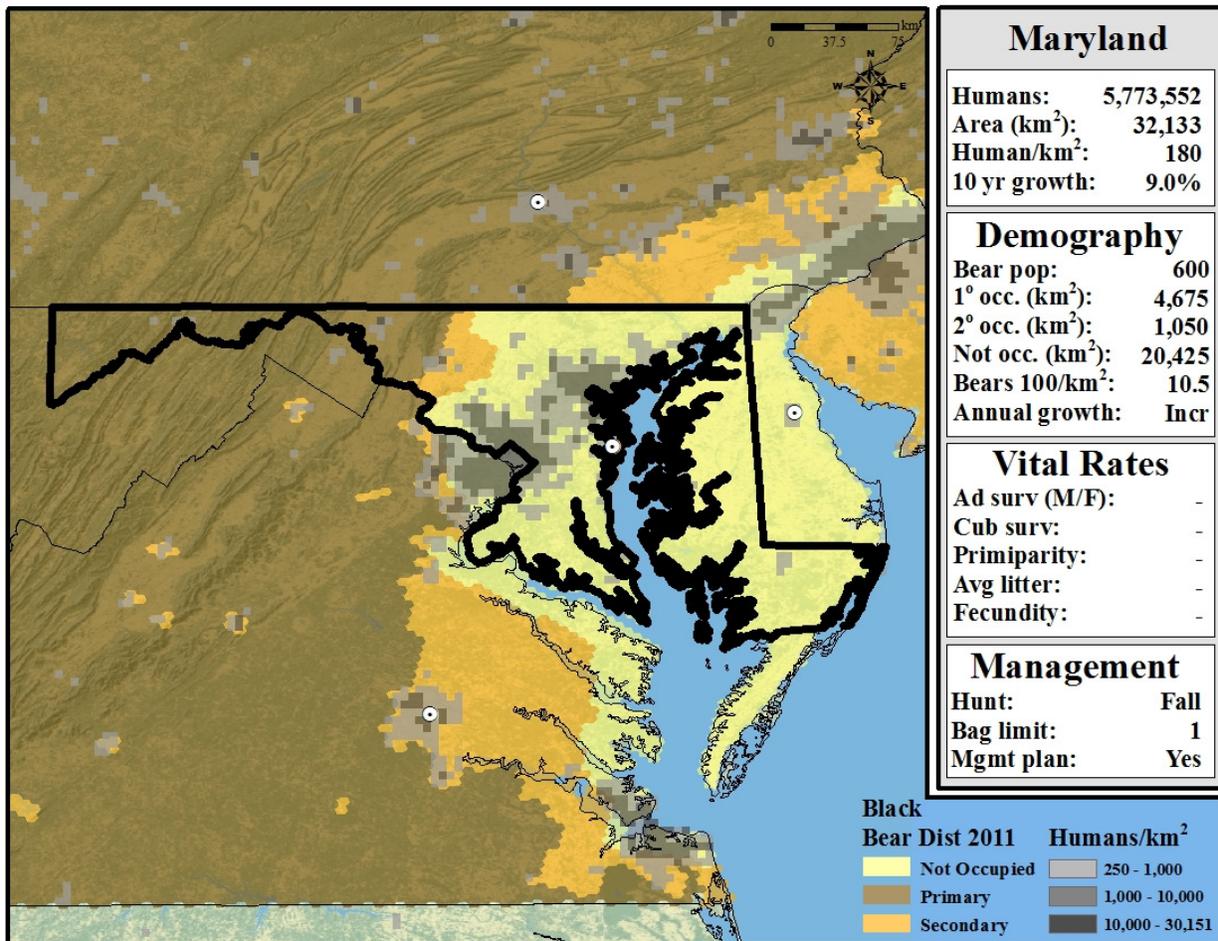
Figure 6.4: *Summary information for Maryland's human and black bear populations. Black bear primary and secondary range from Scheick et al. (2012).*

6.1.5 Massachusetts

The Division of Fisheries and Wildlife is responsible for black bear management in the state of Massachusetts, which is guided by a long-term plan (Noyce 2012). Almost all of western Massachusetts is primary black bear range with only the eastern portion of the state surrounding Boston being unoccupied (Figure 6.5). Black bear populations in Massachusetts are contiguous with Connecticut, New Hampshire, New York, and Vermont. Massachusetts has been the site of ongoing black bear research for >20 years. Hunters are allowed to harvest 1 black bear/person during a fall hunt. Massachusetts bear populations are increasing (Noyce et al. 2012).

Massachusetts

Humans:	6,547,629
Area (km^2):	27,336
Human/km^2:	240
10 yr growth:	3.1%

Demography

Bear pop:	3,000
1° occ. (km^2):	10,425
2° occ. (km^2):	1,625
Not occ. (km^2):	8,725
Bears 100/km^2:	24.9
Annual growth:	Incr

Vital Rates

Ad surv (M/F):	-/0.87
Cub surv:	0.53-0.74
Primiparity:	4.0
Avg litter:	2.4
Fecundity:	-

Management

Hunt:	Fall
Bag limit:	1
Mgmt plan:	Yes

Black Bear Dist 2011 / Humans/km^2

Not Occupied	250 - 1,000
Primary	1,000 - 10,000
Secondary	10,000 - 30,151

Figure 6.5: *Summary information for Massachusetts's human and black bear populations. Black bear primary and secondary range from Scheick et al. (2012).*

6.1.6 New Brunswick

The Fish and Wildlife Branch is responsible for black bear management in the province of New Brunswick. Black bear management is not guided by a long-term plan (Noyce 2012). Almost all of New Brunswick is primary occupied black bear range with small pockets of secondary range interspersed (Figure 6.6). No black bear research has been conducted in New Brunswick. Hunters in New Brunswick are allowed to harvest 1 bear/person during a spring or fall hunt season. New Brunswick bear populations are increasing (Noyce 2012).
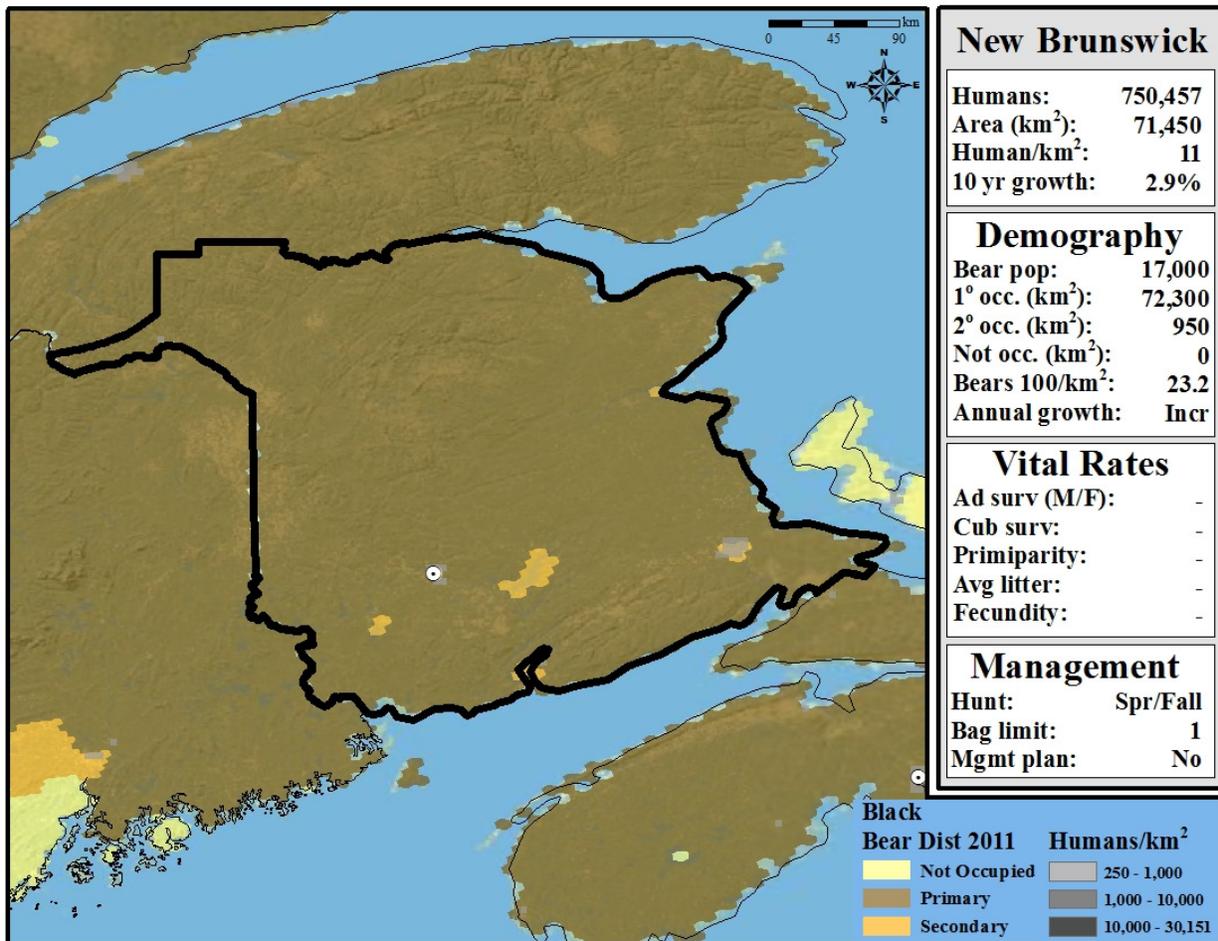
Figure 6.6: *Summary information for New Brunswick's human and black bear populations. Black bear primary and secondary range from Scheick et al. (2012).*

6.1.7 Newfoundland

The Department of Environment and Conservation is responsible for black bear management in the province of Newfoundland and bear management is guided by a plan (Noyce 2012). Almost all of Newfoundland is primary occupied black bear range, with only the southeastern tip considered secondary range (Figure 6.7). No research has been conducted on black bears in Newfoundland. Hunters in Newfoundland may harvest 2 bears/person/year; 1 during a spring and 1 during a fall hunt. Newfoundland's bear population is increasing (Noyce 2012).
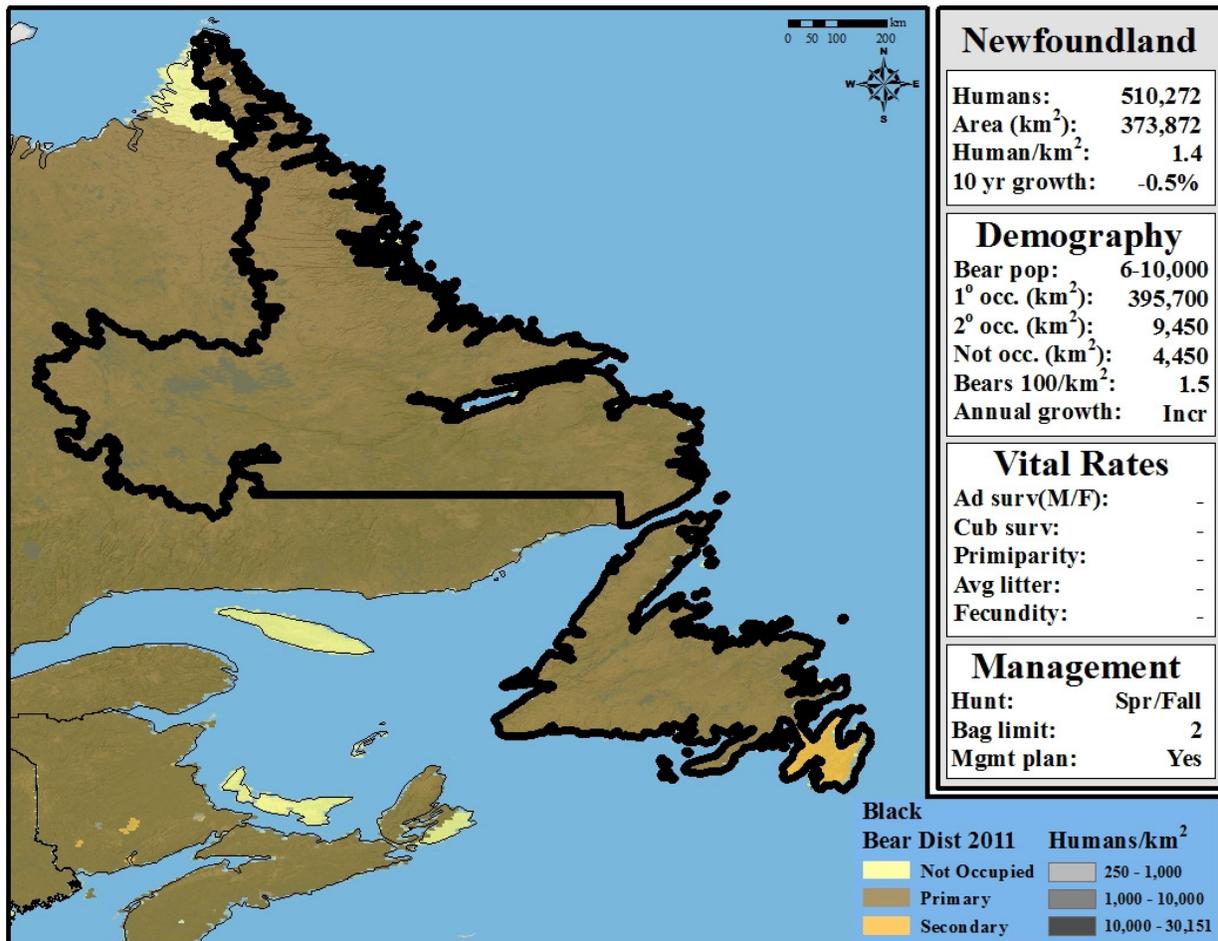
Figure 6.7: *Summary information for Newfoundland's human and black bear populations. Black bear primary and secondary range from Scheick et al. (2012).*

6.1.8 New Hampshire

The Fish and Game Department is responsible for black bear management in the state of New Hampshire. Management is guided by a long-term plan. Almost all of New Hampshire is primary black bear range and only a small area surrounding Manchester is considered secondary range (Figure 6.8). Black bear research has been ongoing for almost 20 years. Hunters in New Hampshire are allowed to harvest 1 bear/person during a fall hunt. New Hampshire's bear population remains stable (Noyce 2012).

New Hampshire

Humans:	1,316,470
Area (km^2):	24,216
Human/km^2:	54
10 yr growth:	6.5%

Demography

Bear pop:	5,000
1° occ. (km^2):	22,525
2° occ. (km^2):	1,525
Not occ. (km^2):	25
Bears 100/km^2:	21.0
Annual growth:	Stable

Vital Rates

Ad surv (M/F):	-/0.87
Cub surv:	0.77
Primiparity:	3.5 - 5.5
Avg litter:	2.0
Fecundity:	0.50

Management

Hunt:	Fall
Bag limit:	1
Mgmt plan:	Yes

Black Bear Dist 2011

	Humans/km^2	
Not Occupied		250 - 1,000
Primary		1,000 - 10,000
Secondary		10,000 - 30,151

Figure 6.8: *Summary information for New Hampshire's human and black bear populations. Black bear primary and secondary range from Scheick et al. (2012).*

6.1.9 New Jersey

The Division of Fish and Wildlife is responsible for black bear management in the state of New Jersey. Almost all of northern New Jersey is primary occupied black bear range, whereas most of southern New Jersey is secondary range (Figure 6.9). Management of black bears in New Jersey is guided by a long-term plan and research has been ongoing for almost 20 years. Hunters in New Jersey are allowed to harvest 1 bear/person during a fall hunt. In northern New Jersey, where human densities are high, bear populations are increasing substantially because of access to anthropogenic food sources, resulting in frequent bear-human conflicts.

New Jersey

Humans:	8,791,894
Area (km^2):	22,588
Human/km^2:	389
10 yr growth:	4.5%

Demography

Bear pop:	3,400
1° occ. (km^2):	5,100
2° occ. (km^2):	14,325
Not occ. (km^2):	25
Bears 100/km^2:	17.5
Annual growth:	Incr

Vital Rates

Ad surv (M/F):	- /0.94
Cub surv:	0.70
Primiparity:	3.0
Avg litter:	3.0
Fecundity:	~0.6

Management

Hunt:	Fall
Bag limit:	1
Mgmt plan:	Yes

Black Bear Dist 2011

| Bear Dist	Humans/km^2
Not Occupied	250 - 1,000
Primary	1,000 - 10,000
Secondary	10,000 - 30,151

Figure 6.9: *Summary information for New Jersey's human and black bear populations. Black bear primary and secondary range from Scheick et al. (2012).*

6.1.10 New York

The Bureau of Wildlife in the Department of Environmental Conservation is responsible for black bear management in the state of New York. Almost all of New York is primary occupied black bear range, with 2 main populations subdivided by unoccupied range along the I-90 transportation corridor that bisects the state from east to west (Figure 6.10). New York has been the site of ongoing black bear research for almost 30 years. Hunters in New York are allowed to harvest 1 bear/person during a fall hunting season. Bear populations in New York are increasing (Noyce 2012).
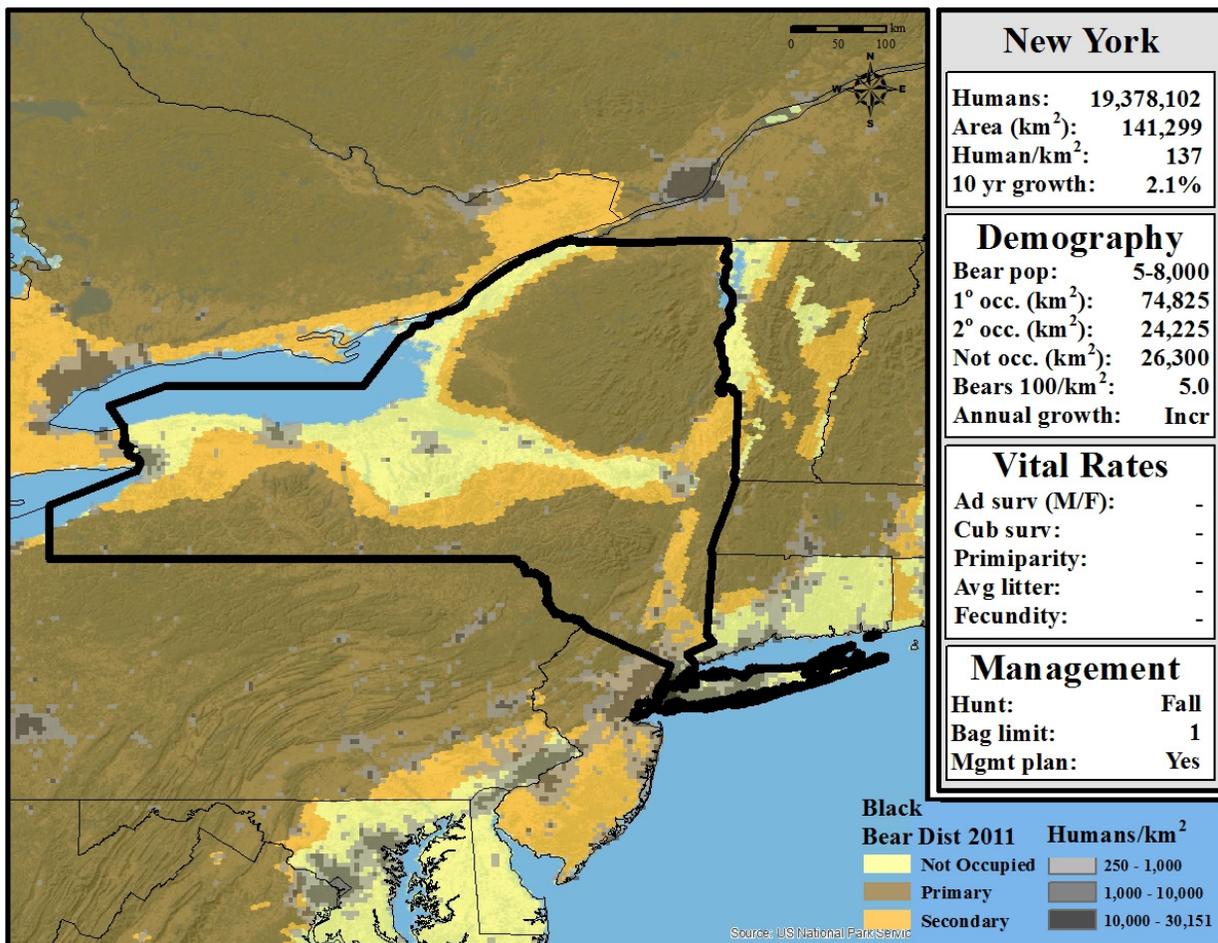

Figure 6.10: *Summary information for New York's human and black bear populations. Black bear primary and secondary range from Scheick et al. (2012).*

6.1.11 Nova Scotia

The Wildlife Division is responsible for black bear management in the province of Nova Scotia. Black bear management is not guided by a long-term plan (Noyce 2012). Almost all of Nova Scotia is primary occupied black bear range, with only the northern island and the northeast coast considered to be unoccupied (Figure 6.11). We are not aware of research conducted on black bears in Nova Scotia. Hunters in Nova Scotia may harvest 1 bear/person during a fall hunting season. Nova Scotia's black bear population is increasing (Noyce 2012).
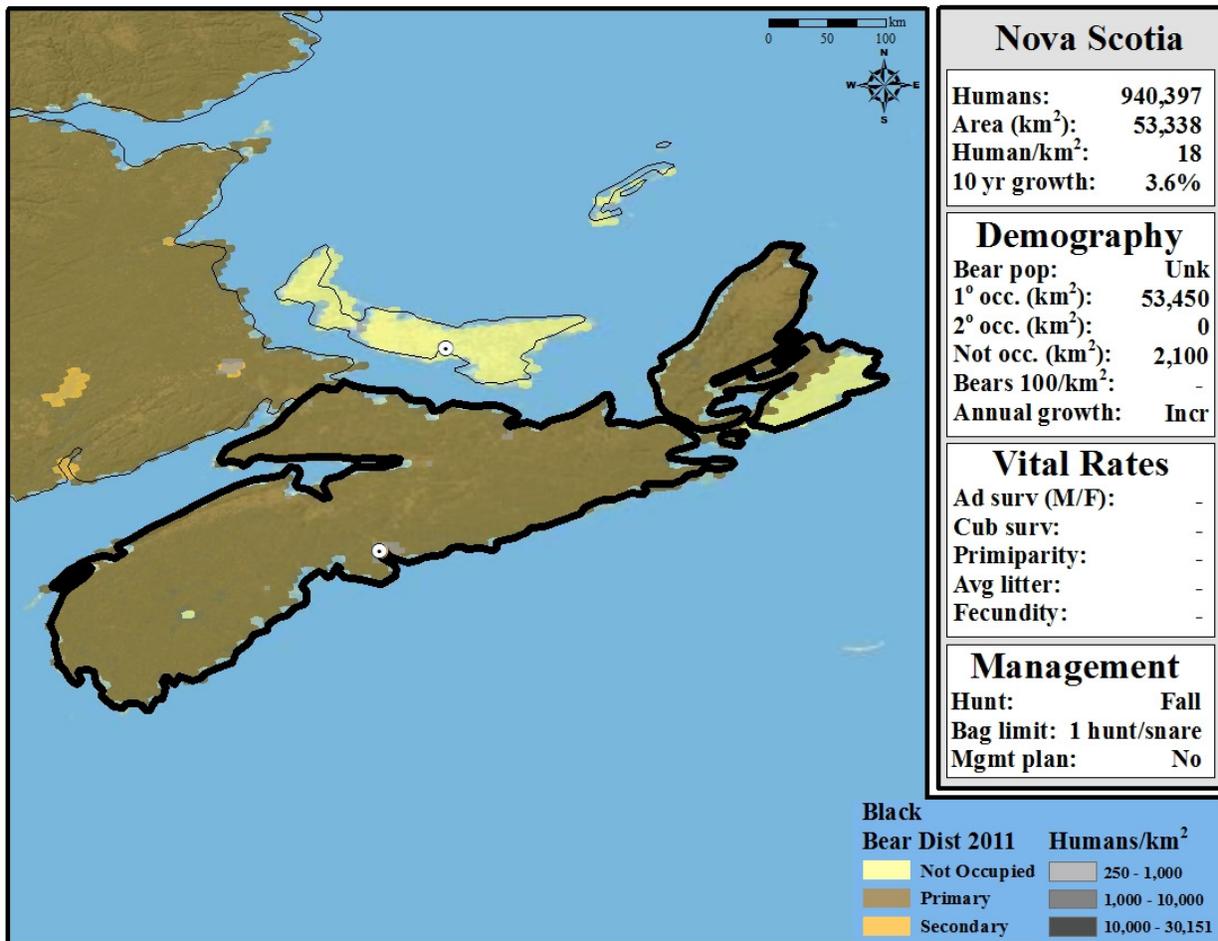

Figure 6.11: *Summary information for Nova Scotia's human and black bear populations. Black bear primary and secondary range from Scheick et al. (2012).*

6.1.12 Ontario

The Fish and Wildlife Branch of the Ministry of Natural Resources is responsible for black bear management in the province of Ontario, which is guided by a long-term plan (Noyce 2012). Almost all of Ontario is primary occupied black bear range, with only the northern edge and the southern tip surrounding Toronto considered to be secondary range (Figure 6.12). Ontario has been the site of ongoing black bear research for almost 30 years (M. Obbard, Ministry of Natural Resources). Hunters in Ontario may harvest 1–2 bears/person during a fall hunting season, depending on the management unit. Ontario's bear population remains stable (Noyce 2012).
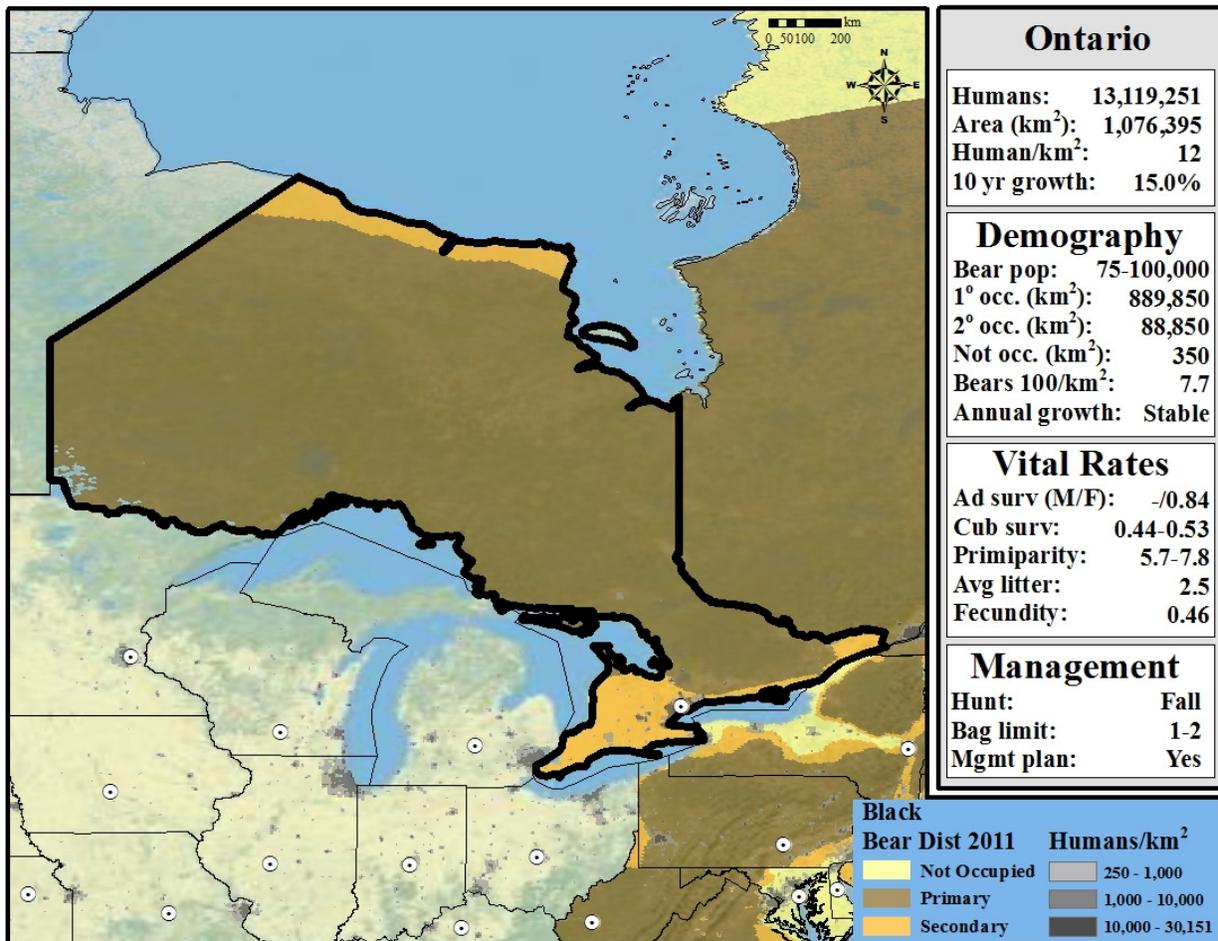

Figure 6.12: *Summary information for Ontario's human and black bear populations. Black bear primary and secondary range from Scheick et al. (2012).*

6.1.13 Pennsylvania

The Pennsylvania Game Commission is responsible for black bear management in the state of Pennsylvania. Management is guided by a long-term plan. Almost all of Pennsylvania is primary black bear range, with only the southeast corner surrounding Philadelphia being unoccupied (Figure 6.13). Pennsylvania has been the site of ongoing black bear research for almost 40 years (Alt 1976). Hunters in Pennsylvania annually harvest approximately 20% of the estimated 18,000 black bears in the state, with bear populations thought to be stable (Ternent 2006).

Pennsylvania

Humans:	1,2702,379
Area (km^2):	119,283
Human/km^2:	107
10 yr growth:	3.4%

Demography

Bear pop:	18,000
1° occ. (km^2):	106,725
2° occ. (km^2):	8,100
Not occ. (km^2):	2,500
Bears 100/km^2:	16.5
Annual growth:	Stable

Vital Rates

Ad surv (M/F):	-/0.59
Cub surv:	0.84
Primiparity:	3.2-3.5
Avg litter:	3.0
Fecundity:	0.62

Management

Hunt:	Fall
Bag limit:	1
Mgmt plan:	Yes

Black Bear Dist 2011 / Humans/km^2

| Black Bear Dist 2011	Humans/km^2
Not Occupied	250 - 1,000
Primary	1,000 - 10,000
Secondary	10,000 - 30,151

Figure 6.13: *Summary information for Pennsylvania's human and black bear populations. Black bear primary and secondary range from Scheick et al. (2012).*

6.1.14 Prince Edward Island

The Fish and Wildlife Division is responsible for black bear management in the province of

Prince Edward Island. All of Prince Edward Island is unoccupied black bear range (Figure 6.14).

No research that we are aware of has been conducted on black bears on Prince Edward Island.

No resident black bears exist on Prince Edward Island, therefore no harvest is allowed.
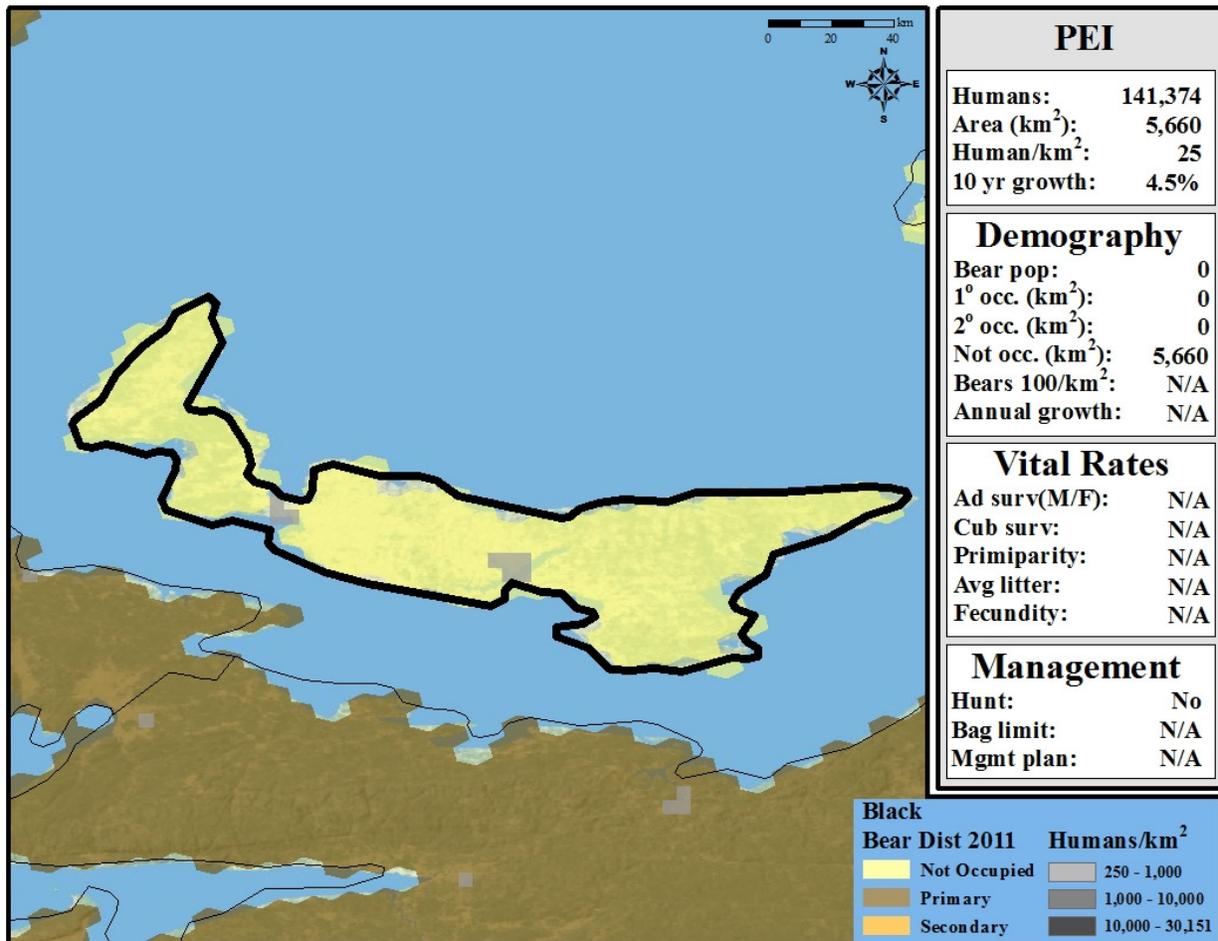
Figure 6.14: *Summary information for Prince Edward Island's human and black bear populations. Black bear primary and secondary range from Scheick et al. (2012).*

6.1.15 Quebec

The Ministry of Sustainable Development, Environment, Wildlife and Parks is responsible for black

bear management in the province of Quebec. Almost all of Quebec is primary occupied black

bear range, with only the northwestern portion north of the Arctic Circle considered unoccupied

(Figure 6.15). Quebec has been the site of ongoing black bear research for almost 20 years.

Hunters in Quebec may harvest 1 or 2 bears/person during a spring or fall hunting season (fall

season in select zones only). With an estimated 70,000 black bears, Quebec has one of the most

robust populations of black bears. Quebec's bear population is increasing (Noyce 2012).
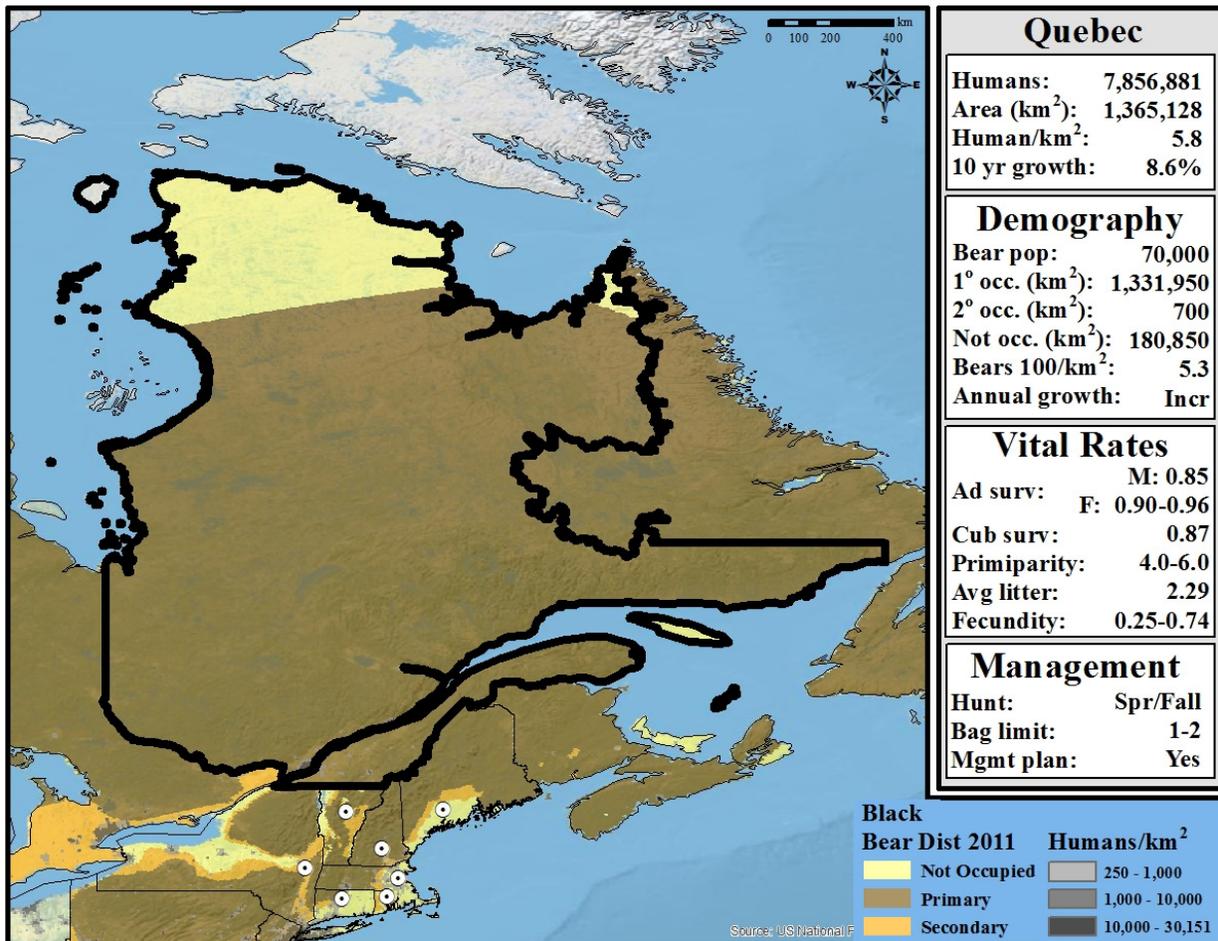
Figure 6.15: *Summary information for Quebec's human and black bear populations. Black bear primary and secondary range from Scheick et al. (2012).*

6.1.16 Rhode Island

The Division of Fish and Wildlife is responsible for black bear management in the state of Rhode Island. Almost all of Rhode Island, much of which surrounds the city of Providence, is unoccupied black bear range, with only the western region considered secondary occupied range (Figure 6.16). No formal black bear research that we are aware of has ever been conducted in Rhode Island. No black bear hunting is allowed in Rhode Island. Although few black bears inhabit Rhode Island, the population is increasing (Noyce 2012). The bear population increase is most likely a result of immigration from Massachusetts.
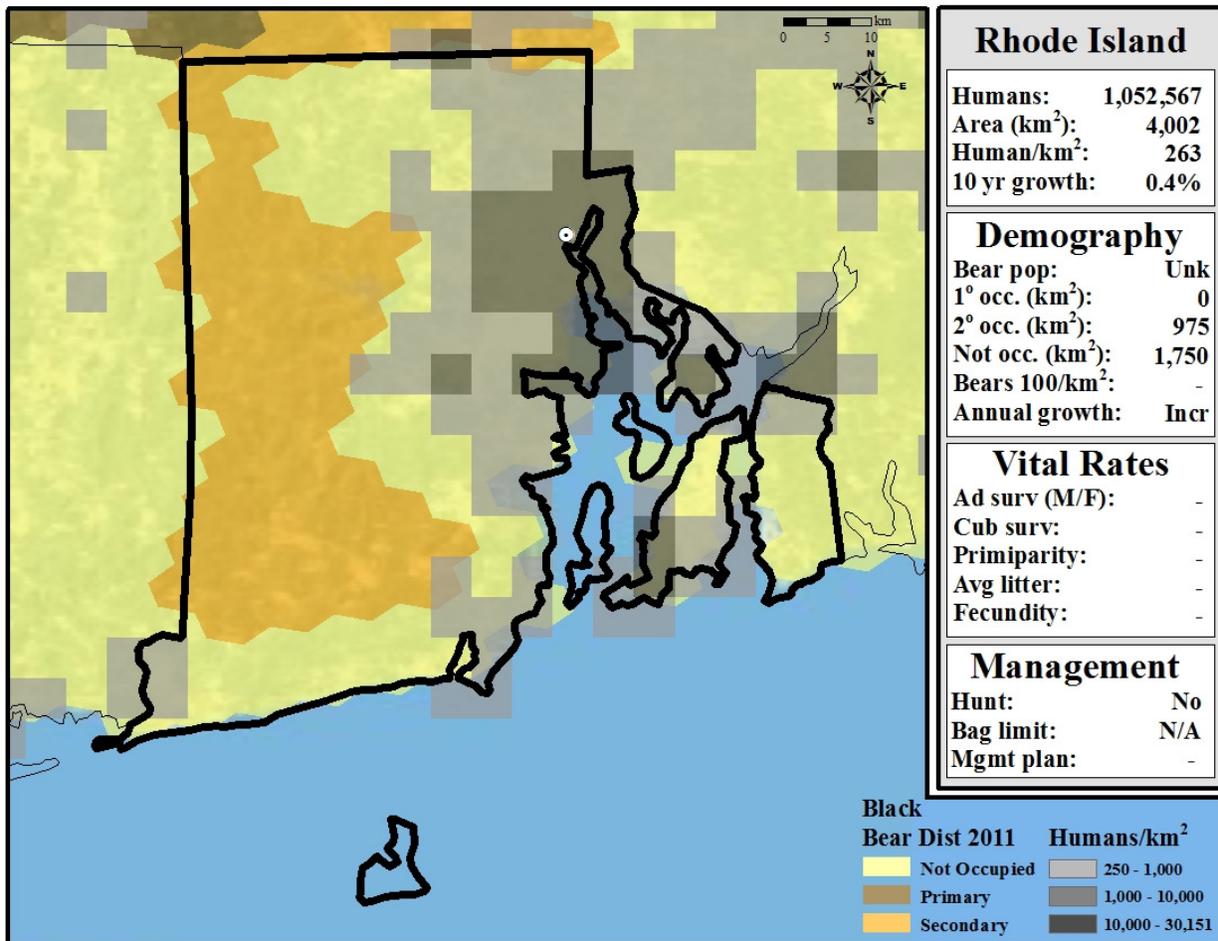

Figure 6.16: *Summary information for Rhode Island's human and black bear populations. Black bear primary and secondary range from Scheick et al. (2012).*

6.1.17 Virginia

The Department of Game and Inland Fisheries is responsible for black bear management in the state of Virginia. Black bear management in Virginia is guided by a long-term plan. Almost all of Virginia is primary occupied black bear range, with only the eastern edge of the state near the Atlantic Ocean being unoccupied (Figure 6.17). Virginia has been the site of ongoing black bear research for over 30 years. Hunters in Virginia are allowed to harvest 1 bear/person during a fall hunting season. Virginia's bear population is increasing (Noyce 2012).

Virginia	
Humans:	8,001,024
Area (km^2):	110,785
Human/km^2:	72
10 yr growth:	13.0%

Demography	
Bear pop:	16,000
1° occ. (km^2):	83,850
2° occ. (km^2):	13,475
Not occ. (km^2):	7,100
Bears 100/km^2:	17.6
Annual growth:	Incr

Vital Rates	
Ad surv:	M: 0.89
	F: 0.91
Cub surv:	0.87
Primiparity:	3.7
Avg litter:	2.49
Fecundity:	0.69

Management	
Hunt:	Fall
Bag limit:	1
Mgmt plan:	Yes

Black Bear Dist 2011 — Humans/km^2

| Not Occupied | 250 - 1,000 |
| Primary | 1,000 - 10,000 |
| Secondary | 10,000 - 30,151 |

Figure 6.17: *Summary information for Virginia's human and black bear populations. Black bear primary and secondary range from Scheick et al. (2012).*

6.7.18 Vermont

The Department of Fish and Wildlife is responsible for black bear management in the state of
Vermont. Black bear management is guided by a long-term plan. Much of Vermont,
particularly the Green Mountains, is primary black bear range, with unoccupied range on the
western and secondary range along the eastern borders of the state (Figure 6.18). Vermont has
been the site of ongoing black bear research for almost 20 years. Hunters in Vermont are
allowed to harvest 1 bear/person during a fall hunting season. Vermont's black bear population
is increasing (Noyce 2012).

Vermont

Humans:	625,741
Area (km^2):	24,901
Human/km^2:	25
10 yr growth:	2.8%

Demography

Bear pop:	4,500-6,000
1° occ. (km^2):	14,425
2° occ. (km^2):	4,850
Not occ. (km^2):	4,900
Bears 100/km^2:	23.3
Annual growth:	Incr

Vital Rates

Ad surv (M/F):	-
Cub surv:	0.26
Primiparity:	5.33
Avg litter:	-
Fecundity:	0.87

Management

Hunt:	Fall
Bag limit:	1
Mgmt plan:	Yes

Black Bear Dist 2011

|	Humans/km^2
Not Occupied	250 - 1,000
Primary	1,000 - 10,000
Secondary	10,000 - 30,151
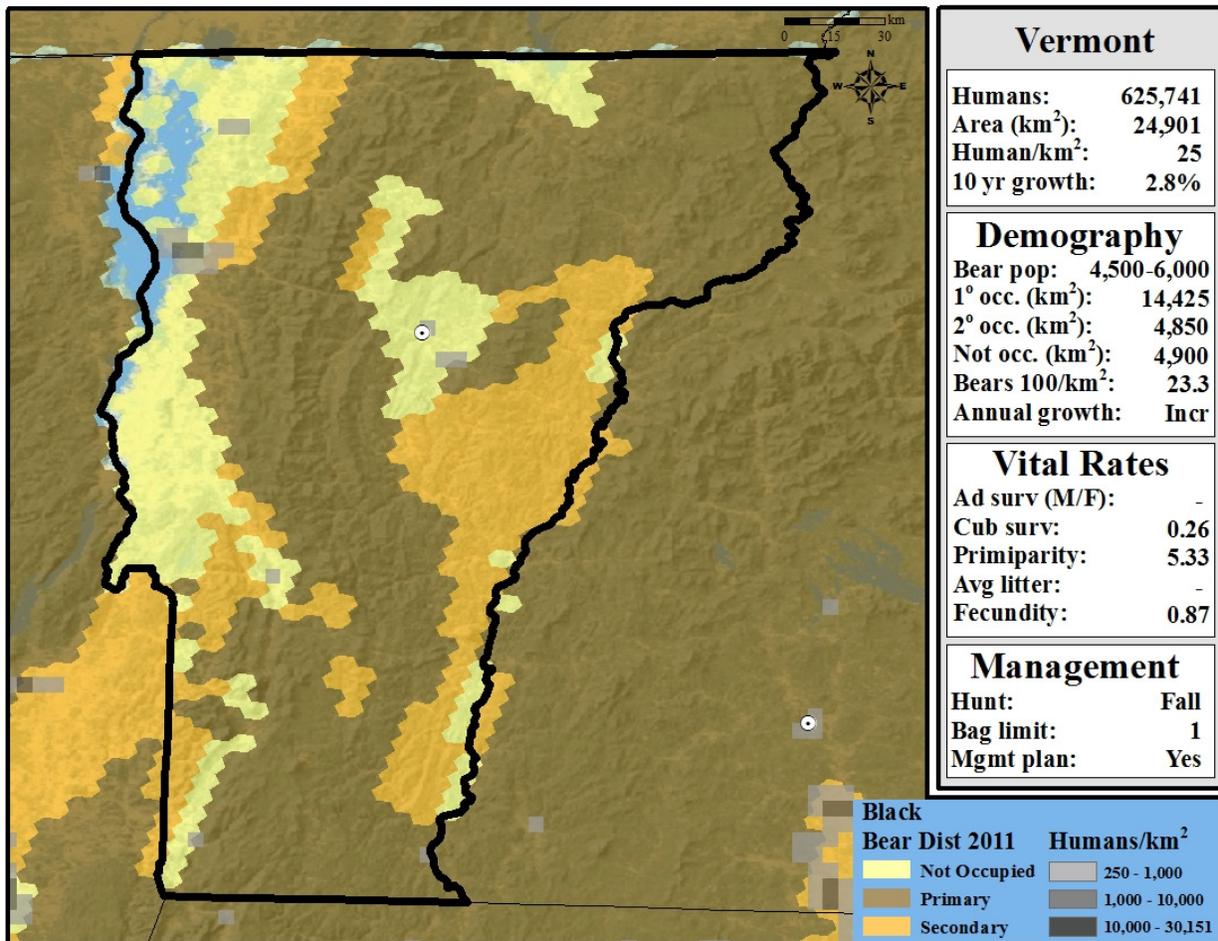
Figure 6.18: Summary information for Vermont's human and black bear populations. Black
bear primary and secondary range from Scheick et al. (2012).

The Division of Natural Resources is responsible for black bear management in the state of West Virginia. Most of West Virginia is primary occupied black bear range, with only the northern panhandle considered unoccupied (Figure 6.19). West Virginia has been the site of ongoing black bear research since 1972 (Ryan et al. 2004) and bears are managed according to an operational plan that is updated every 5 years. Hunters in West Virginia are allowed to harvest 1 or 2 bears/person during a fall hunting season. West Virginia's bear population is increasing (Noyce 2012).
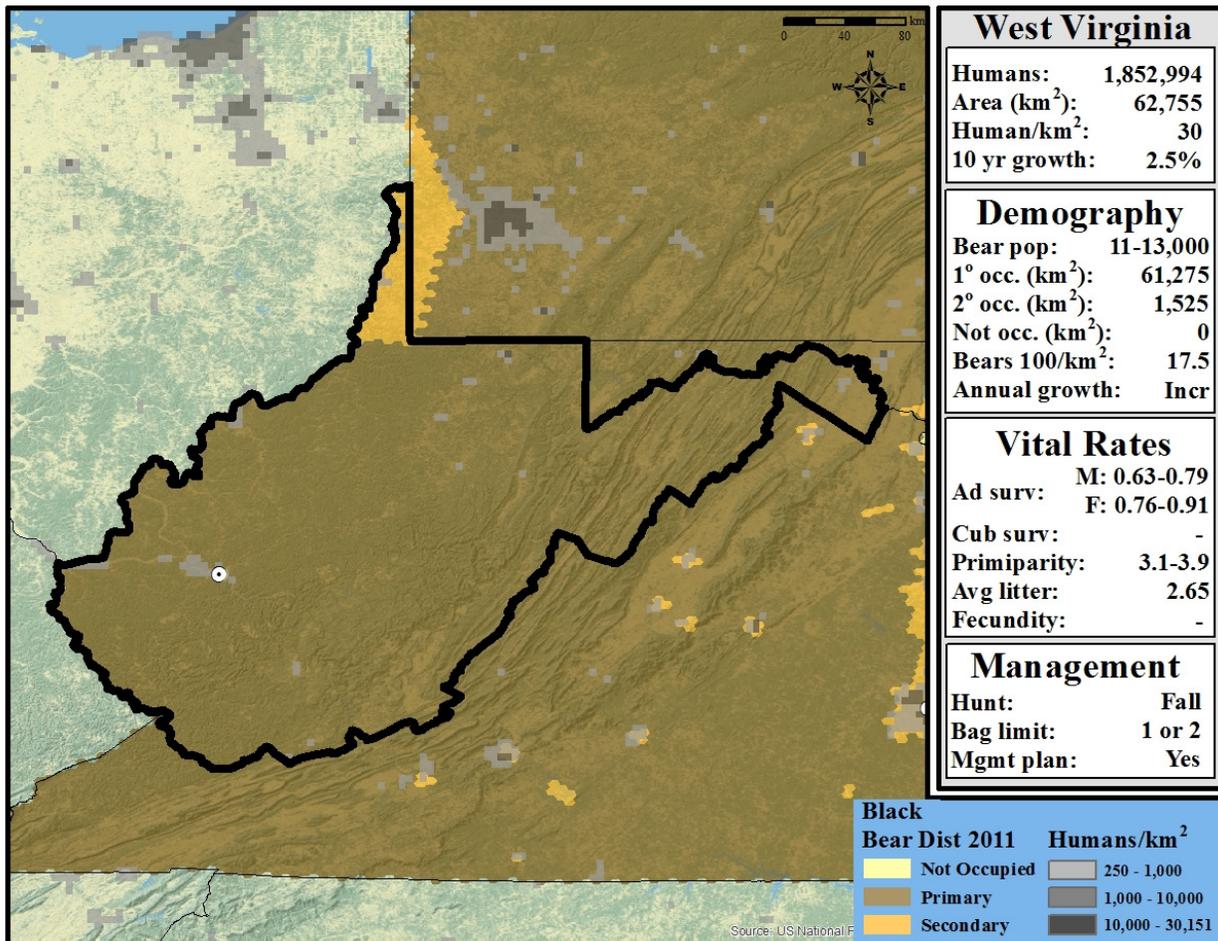

**Figure 6.19:** *Summary information for West Virginia's human and black bear populations. Black bear primary and secondary range from Scheick et al. (2012).*

Chapter 7. Manager Survey

7.1 BACKGROUND AND METHODS

As we have illustrated in the preceding sections, there are a many differences among jurisdictions in black bear population characteristics such as density and growth rates reflecting, and sometimes driving, the underlying variation in their biology and management. Our literature review was based on peer-reviewed articles, theses and dissertations, and agency reports. It produced a substantial amount of information and data on black bear populations from the past several decades. Clearly, however, research and monitoring efforts within any given jurisdiction rarely are as extensive as managers would like, nor are they consistent across jurisdictions. Despite the relative importance of black bear management across the Northeast, substantial gaps will likely always exist due to budgetary limitations

In an attempt to fill in some of the information gaps from the literature, we developed a survey specifically for black bear managers in the Northeast. Our interests were three-fold. First, we wanted to capture information about black bear populations that was not available in the literature from those individuals most knowledgeable about these populations. Second, we wanted to learn how policies, communication, and management collectively function across this large, diverse region. Finally, we wished to better understand managers' perceptions about science and how they use research results in decision making.

We consulted survey design experts when building our questions to ensure clarity and to minimize response bias (S. Clark, Yale University; M. Patterson, University of Montana; S. Wilson, Blackfoot Challenge). Consistently, experts suggested we use the Likert scale (Likert

1932), where respondents select one of five answers ranked from strongly disagree to strongly agree. This is a very powerful and rapid way of clustering beliefs to identify patterns across a sample of people. Questions are often written as the extreme position relative to a specific concept. For example, "Population growth rates are the most important parameter for management decision making." In theory, this approach forces each respondent to rank their beliefs within a given topic or cluster of questions. As responses are not enforced across questions, however, it is possible for more than one question in a cluster to have the same rank. Therefore, we used strategic follow-up questions to more fully understand certain topics when the Likert scale may have been inadequate to extract the desired information.

To further ensure accurate responses, we provided definitions of terms that could be interpreted differently by respondents. This was particularly important for geographical extent, because what one person considers a local extent may be considered regional by someone else. We therefore established that all questions pertained to the entire jurisdiction, i.e., the state or province of the respondent. This definition likely decreased the resolution of information in some situations such as a single jurisdiction having multiple bear populations of differing status. For almost all of our primary questions, however, it provided an appropriate resolution and reflected the respondents' professional perceptions.

Instead of contacting every person involved with black bear management in the region, we chose to contact the primary representative to the NEBBTC for each jurisdiction. We recognized that this approach would not capture the variation in responses as well as a more comprehensive list. However, it reduced redundancy that could bias interpretation of questions for areas where multiple personnel were involved in bear management. Although we collected the name, agency, and jurisdiction for each respondent for tracking purposes, we assured

respondents that responses to sensitive questions would remain anonymous. Responses were collected over a three-month period in early 2012.

Generally, the survey can be divided into several broad topics, including management objectives, management tools, the role of the public, role of science, and trans-jurisdictional issues. Many of our questions, however, related to >1 topic, requiring somewhat holistic, qualitative interpretations. The survey, including the full list of questions and responses, is provided at the end of this report (Appendix B).

7.2 RESULTS AND DISCUSSION

Fifteen of the 19 NEBBTC jurisdictions responded to the survey. Of the 6 Canadian provinces, Newfoundland, Ontario, and Quebec responded. Of the 13 states, only Delaware did not respond, which is not surprising as they do not have a resident bear population (Chapter 6). Our discussion of survey results is limited to those jurisdictions that responded and must therefore be interpreted accordingly.

As expected from such a diverse group of wildlife managers, responses varied considerably across jurisdictions. One of the few exceptions to this was that decisions are made at the level of the state or provincial management agency for all NEBBTC members other than Ontario, where decisions were made at the local level. There were several comments, however, that decisions can be made by the legislature without regard to data and without managers' involvement. Most respondents believed that decisions are made at the most effective level within a jurisdiction, although they can at times be influenced by political and financial concerns.

Seventy-three percent of respondents agreed there is good communication regarding bear population management objectives and status across jurisdictions, at least within the domain of

the NEBBTC. Most also agreed that it is important to establish region-wide monitoring standards, although most responded that these currently do not exist. The significance of communication was clear given how bear population dynamics in one jurisdiction can impact management decisions of their neighbors, particularly in light of the rapid growth of some bear populations in the Northeast.

Within jurisdictions, 73% of respondents reported that their agency has explicit bear management objectives and a formal bear management plan (Figure 7.1). Perhaps reflecting the rapidly changing status of bear populations in the Northeast, only one respondent indicated that
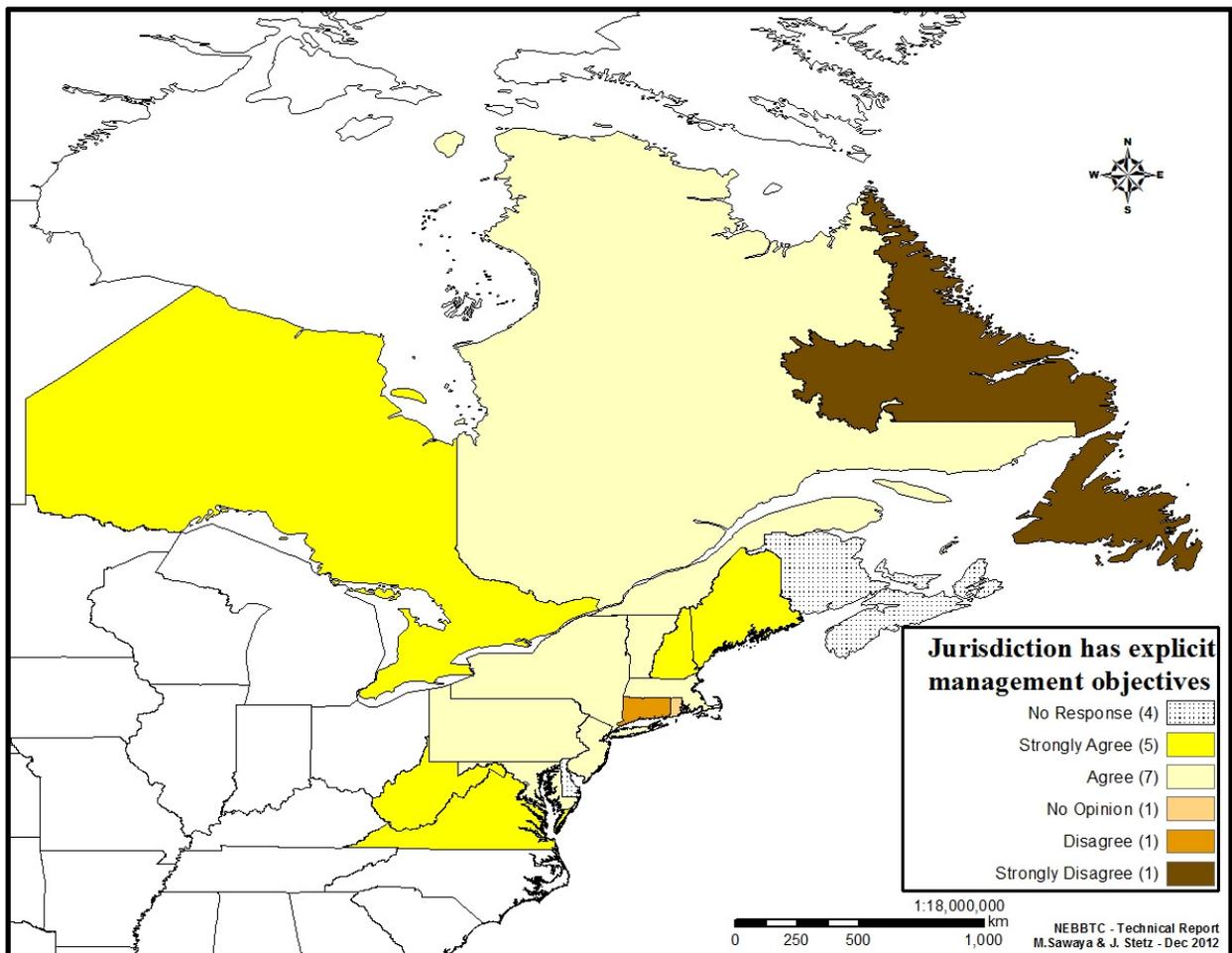
Figure 7.1: *Survey responses regarding the existence of explicit black bear management objectives by NEBBTC jurisdiction.*

>80% of their population objectives have been met, with 53% of respondents reporting that objectives have been met for <40% of their jurisdiction (Figure 7.2). Nearly every jurisdiction in the Northeast reported increasing black bear populations based on the survey by Noyce (2012) and, accordingly, respondents stated that minimizing human-bear conflicts is among the most important investment they can make to reach and maintain management objectives. Several respondents noted, however, that these efforts are directly linked with education, which confounded interpretation across this cluster of questions to some degree. Regardless, the role of education in reducing conflicts was a clear priority for most jurisdictions, which was reflected in
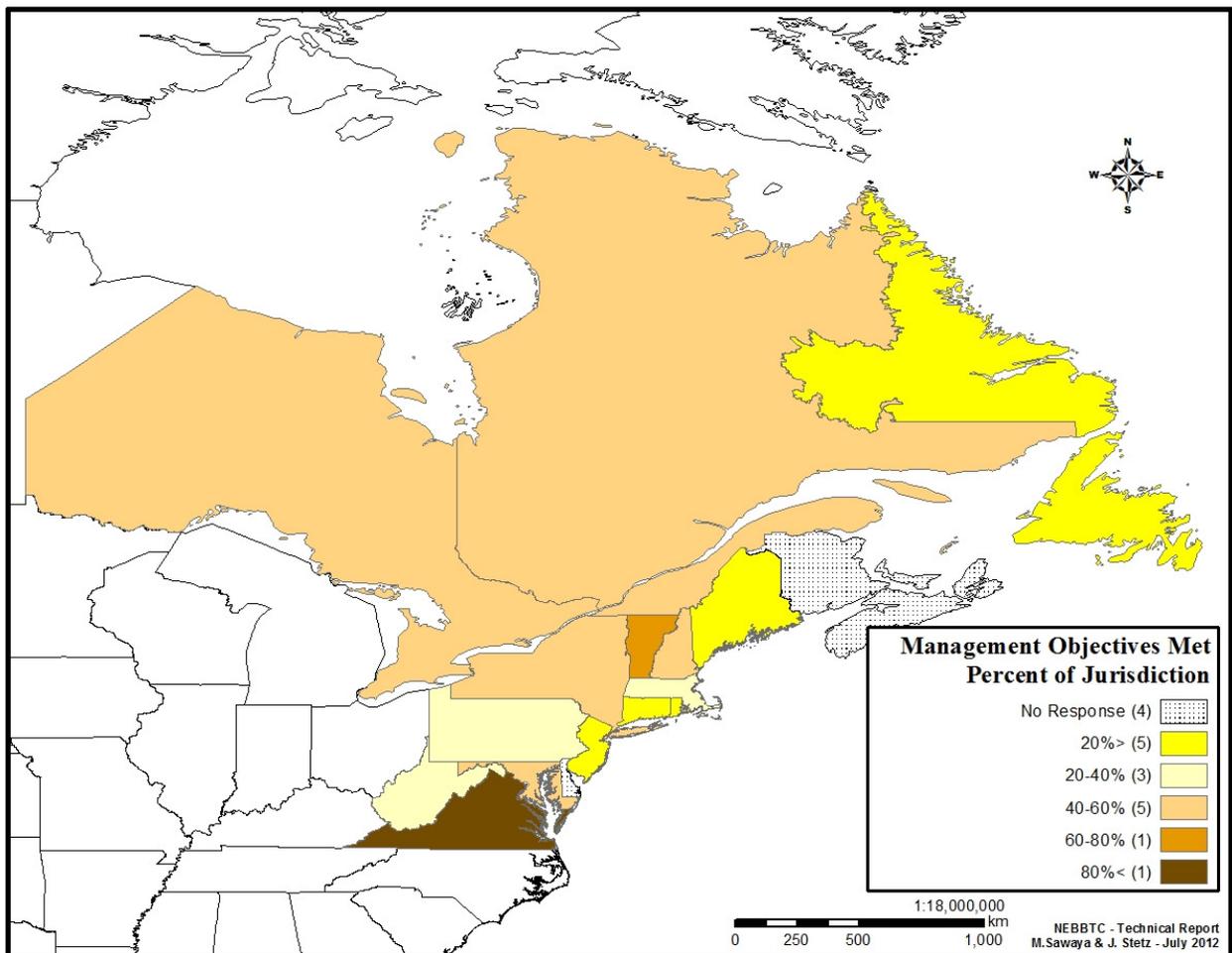
Figure 7.2: Survey responses regarding whether black bear management objectives had been met by NEBBTC jurisdiction.

our finding that 58% of jurisdictions have formal education programs in place about bears. Further, minimizing conflicts, along with maintaining sustainable harvests, were typically the primary objectives of management plans. Accordingly, adjusting harvest levels was considered to be the most important management tool for every jurisdiction where hunting occurred. Responses about the role of enforcement of game, feeding, and sanitation laws were among the most consistent, but few respondents regarded this as important, relative to other factors, in reaching population objectives.

Regardless of the specific mechanisms, 47% of respondents claimed that current efforts are adequate for reaching or maintaining management objectives in their jurisdiction. Similarly, 53% of respondents stated that adaptive management principles play an important role in their agency, although use of adaptive management did not always correspond with having adequate management efforts in place. Noyce (2012) reported that 53% of NEBBTC jurisdictions had experienced an increase in human-bear conflicts over the preceding 10 years, with no jurisdictions reporting a decrease. Despite the rapid growth in bear populations and in human-bear conflicts reported in Noyce (2012), 67% of our survey respondents indicated that <40% of their populations were at or beyond social carrying capacity (Figure 7.3).

A more detailed understanding of population dynamics was considered important for improving management by all but one respondent. Similarly, there was consensus that monitoring bear population demography is important to successful management. However, we submit that our question was poorly phrased because some respondents referred to human populations rather than bear populations. Regardless, the role of obtaining reliable data on bear population dynamics, and improved understanding of the drivers, was clear. This is also evident in the apparent trend toward using more robust research techniques, with fewer jurisdictions

relying on density extrapolation, population reconstruction, and biomarking, and more jurisdictions using genetic studies and mark-recapture methods (Noyce 2012).

Given their increased use, we asked for specific perceptions on the reliability and feasibility of using DNA-based mark-recapture for monitoring bear populations. Generally, respondents regarded DNA-based methods as being very well suited for certain objectives, such as estimating population abundance in small areas, but viewed them as prohibitively expensive for large-scale questions. Several respondents appreciated the greater precision of parameter estimates from genetic studies, particularly given the typically short study durations.
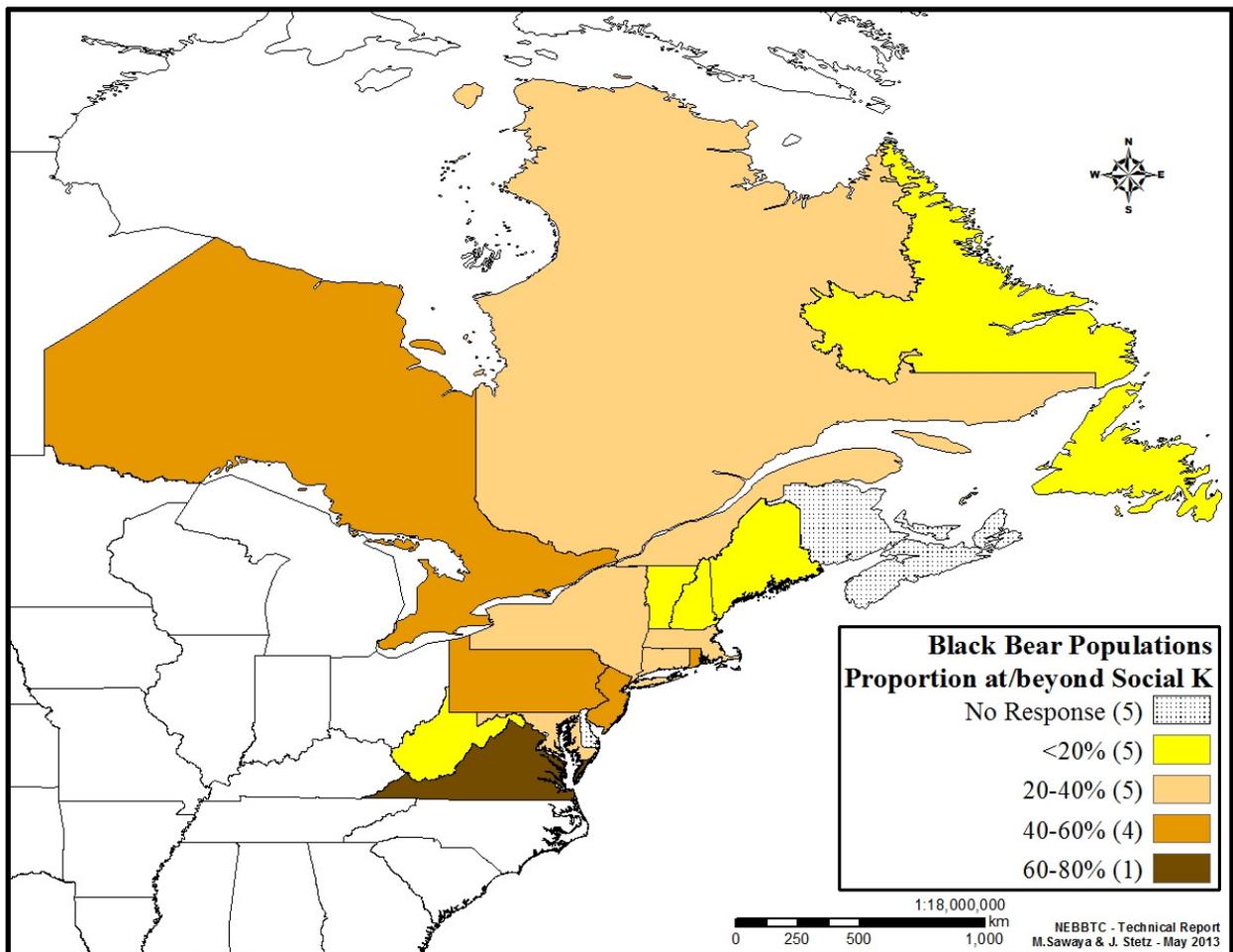
Figure 7.3: *Survey responses regarding the proportion of black bear populations at or beyond social carrying capacity by jurisdiction.*

This was consistent with the 73% of respondents who indicated the precision of demographic estimates influences how they apply research results to decision making, although only 40% of respondents had thresholds of precision below which they discounted research results.

Given the broad range of population objectives and status, and personal history of respondents, an overall synthesis of the survey would be of limited utility. Our manager's survey was successful, however, in providing regionally relevant information not available through other sources (e.g., reports, published literature). The variation of responses is of itself extremely valuable, clearly indicating that there are no one-size-fits-all answers to management of black bears in the Northeast. Further, certain insights, such as the support for monitoring standards, suggest that broader discussions across jurisdictional boundaries may be important in reaching regional objectives of sustainable harvests while minimizing human-bear conflicts.

Section IV. Monitoring Techniques

IV.1 INTRODUCTION TO MONITORING TECHNIQUES

To some, monitoring implies a passive approach to wildlife management consisting simply of following trends in population size or other parameters with little understanding of what is driving the trend and even less understanding about how to modify it. The context that we use for monitoring in this document includes tracking trend in vital rates, but it also involves learning what is causing the changes, how confident we are that the trend is real, and assessing population response to management changes. In a broader sense, we use the term monitoring to describe the estimation of population parameters that may be useful for management.

Population parameters such as abundance and survival are often impossible to directly measure and difficult to reliably estimate for wide-ranging, elusive species such as black bears. Obtaining estimates of demographic parameters for black bears requires intensive and often expensive study designs to achieve reasonable levels of accuracy and precision (Harris et al. 2011). Precision of estimates is generally considered acceptable for management decision making when coefficients of variation are <20% (Pollock et al. 1990). As we reported in Chapter 7 and Appendix B, however, managers often desire greater precision to evaluate outcomes of management actions.

Parameter estimates fall into 4 general combinations of accuracy and precision (Figure IV.1): accurate and precise, accurate but imprecise, precise but inaccurate, and imprecise and inaccurate. Estimates that are accurate (low bias) and precise (low uncertainty) are the most beneficial to wildlife management. Perhaps the most dangerous are estimates that are precise but inaccurate (Figure IV.1), as they may mislead managers to have false confidence in reliability of estimates. Black bear managers in most NEBBTC jurisdictions are interested in obtaining the most accurate and precise population parameter estimates possible (Chapter 7; Appendix B).
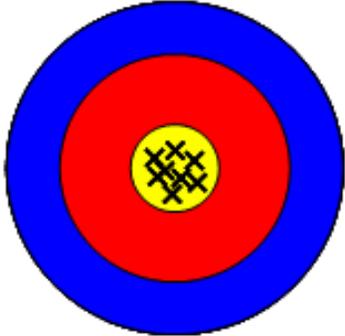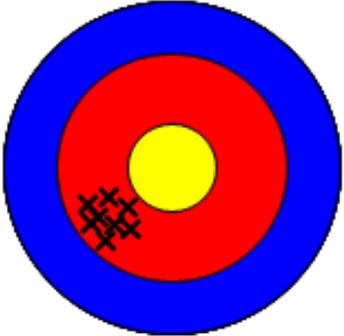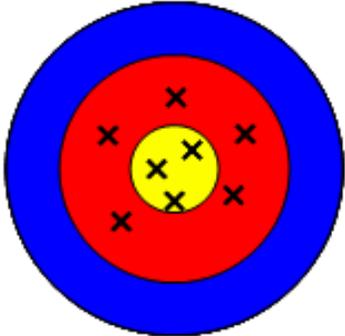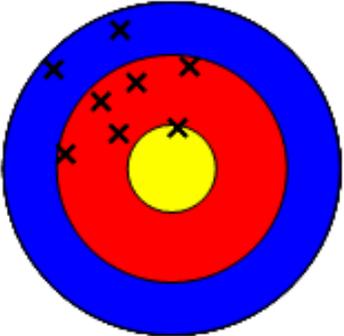
Figure IV.1: *Estimates of population parameters fall into four general categories of accuracy and precision.*

Chapter 8. Abundance and Density

8.1 ABUNDANCE

Black bear population abundance (i.e., number of animals in an area of interest at a given time) in the Northeast is generally increasing (Chapter 6). Abundance remains one of the most important parameters to wildlife managers (Nichols and Hines 2002, Schwartz et al. 2007), including black bear managers in the Northeast (Chapter 7). Abundance is used for setting harvest quotas, monitoring population changes, and understanding population dynamics. Generally, precision of abundance estimates increases with greater sampling intensity, which usually means greater cost. Thus, most managers in NEBBTC jurisdictions consider the tradeoffs between sampling intensity (i.e., cost) and reliability (i.e., accuracy and precision) as the primary consideration when choosing methods to estimate black bear abundance (Chapter 7).
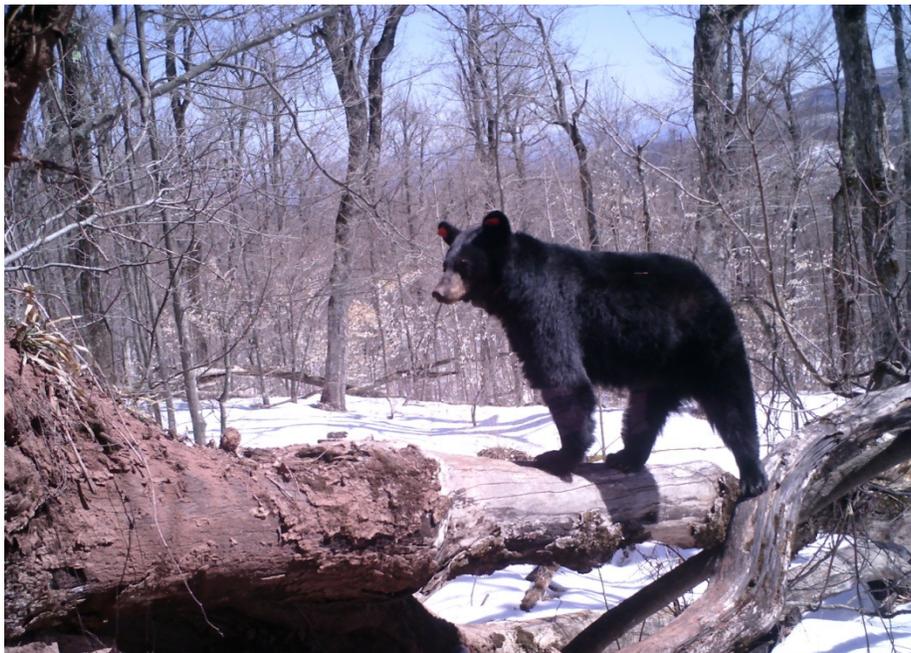
Figure 8.1: *Remote photograph of black bear with uniquely numbered ear tags. Photo credit: New York Department of Environmental Conservation*

8.1.1 Indirect Estimates of Abundance

Abundance is usually thought of as a discrete number of animals inhabiting a particular space, but indirect measures of abundance, or indices, can often be useful depending on the objective (Caughley 1977, Lancia et al. 1996). Reliable indices can provide estimates of population trend in response to perturbations, which may be all that is needed for certain jurisdictions, and can cost far less than a population estimate. The best indices are those that have a linear and positive correlation with population size. Non-linear relationships can be useful as well, if the curvilinear form can be quantified. Unfortunately, the strength and shape of the relationships for most indices of black bear abundance have not been investigated. Following is a discussion of several types of commonly used indices to black bear abundance.

8.1.1.1 Bait-station Index

Bait-stations have been used by ≥15 wildlife management agencies in North America (Garshelis 1990), including Maryland (Figure 8.2). The method evolved from pre-baiting for black bear trapping (Johnson and Pelton 1980), and involves establishing a series of bait-station routes in bear habitat, often along roads or trails. Bait, often opened cans of sardines or bakery products, is suspended by a string from a tree branch about 3 m above the ground at each of a series of sampling sites. A bear visiting the site will generally climb the tree to obtain the bait, leaving claw marks as an indication that the site was visited. Baits are usually checked after 5–7 days and the proportion of visited bait sites is used as an index of abundance. Bait-station surveys are usually conducted annually to monitor bear population trends.

Several potential problems exist with bait-station surveys. First, a non-visit does not mean that bears are not in the area of the bait; 20–30% of bait sites are not visited even where

bear densities are high. Therefore, detection is not perfect (i.e., <100%) and can vary by factors

not associated with population abundance, such as fluctuations in natural foods eaten by bears.

Further, the relationship is likely curvilinear and asymptotic because bear populations may

continue to increase even when the bait-station index has reached 100%. Another issue with this

method is that bears may become food-conditioned and less wary of humans, similar to effects of

trapping with bait (Ternent and Garshelis 1999, Brongo et al. 2005).

The only rigorous evaluation of bait-station surveys and population trend was performed

by Clark et al. (2005) and Rice et al. (2001). Clark et al. (2005) found that bait-station indices
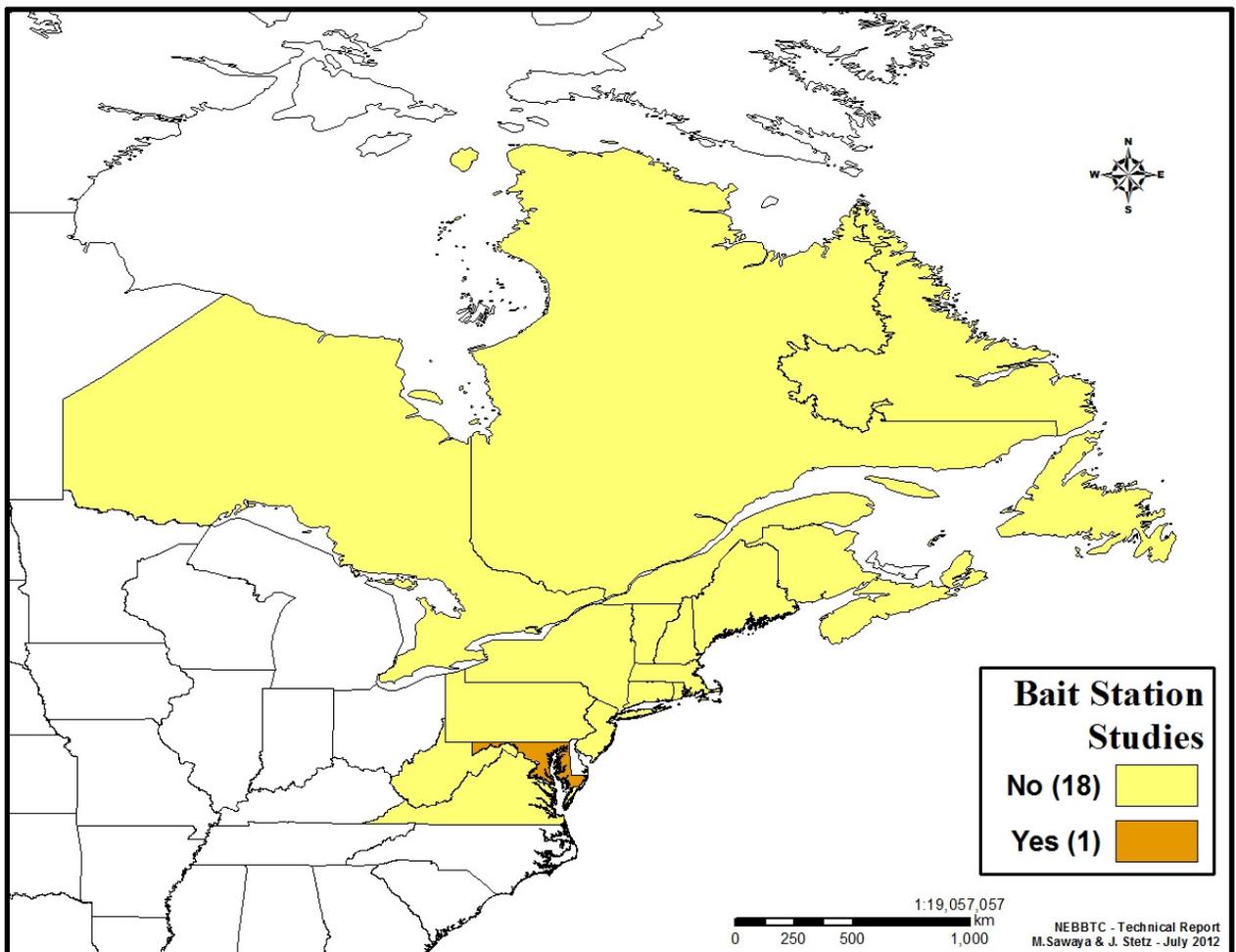
Figure 8.2: *Use of bait station studies by NEBBTC jurisdiction (Noyce 2012).*

were not a good predictor of population growth within a 330-km^2 study area within Great Smoky Mountains National Park, Tennessee. Bait-station indices were correlated, however, with indices of acorn abundance, suggesting that the availability of natural foods affected visitation rates. Rice et al. (2001) used a power analysis and concluded that bait-station surveys in Idaho and Washington lacked adequate power to detect even gross population declines. We note, however, that surveys across a broader geographical region (southern Appalchians; Southern Appalachian Black Bear Study Group, unpublished data) clearly reflected the general increase in bear numbers over the past 3 decades. Year-to-year fluctuations in bait-station indices are likely affected by sampling error and extraneous factors such as natural food availability, so we view bait stations as a method most useful for detecting gross population trends over a long period of time (i.e., decades) in a broad geographical context (i.e., multi-state or province wide). Occupancy estimation methods may be used with detections based on bait-station data if the sites were surveyed repeatedly within a short period of time (MacKenzie et al. 2006). Given such data, Royle et al. (2005) described methods for estimating abundance, but this has not been attempted with bears.

8.1.1.2 Observations

We define observations as any attempt to record, in a standardized way, bear-human encounters, either by design or incidental. For example, many jurisdictions record numbers of nuisance bear complaints received from the public in a standardized manner, so those numbers can be compared each year as general trend in bear abundance. Other jurisdictions have recorded observations of bears by the public, usually for small, re-establishing populations. These types of observational data are affected by factors other than population abundance (e.g., mast failures

usually coincide with increasing nuisance bear complaints or road kills), so these data should only be used as a general measure of population trend. Additionally, road kill data are affected by traffic volume, which has been steadily climbing for decades throughout eastern North America (van Manen et al. 2012).

Observational air or ground surveys for black bears are sometimes feasible in areas where cover is sparse and bears are easily detected (Schwartz et al. 2002), but except in more northerly units of some Canadian provinces, this method is usually not feasible. Even in relatively open areas, sightability is strongly dependent on habitat types (e.g., meadows, alpine). Observations may be easier to obtain at known feeding sites (e.g., garbage dumps, berry patches), but the same assumptions and complexities with using nuisance complaint data probably apply. Remote cameras hold some possibility as a monitoring tool if differences in trap success among camera types can be overcome (Kelly and Holub 2008). Tiger (*Panthera tigris*) abundance has been estimated with remote cameras (Royle et al. 2009) but this requires that individual animals be identified from photographs. Although potentially feasible for Asiatic black bears and sun bears based on chest markings (Ngoprasert et al. 2012), this would not be reliable for American black bears without supplemental markings (e.g., Mace et al. 1994).

8.1.1.3 Harvest Data

All NEBBTC jurisdictions with hunting seasons monitor annual harvest and many require physical checking of harvested bears to obtain data on sex, weight, and age (i.e., cementum annuli analysis of teeth) and other information (Figure 8.3). Stable harvest trends can be used to illustrate that the bear population is not declining dramatically, and that the pattern suggests an overall sustainable harvest rate given some knowledge of hunter effort. Of course, an important

assumption is that harvest opportunities and reporting levels are constant. For example, declining populations may sometimes show stable trends in harvest for a period of years because more hunting effort (i.e., more hunters afield, more hunter days) may be expended. Finally, harvest indices can be sensitive to sampling variation (Diefenbach et al. 2004). Consequently, harvest trend data are insensitive to population changes at best and may be misleading at worst. If effort can be quantified (e.g., number of hunter days), the harvest per unit effort (sometimes referred to as catch per unit effort or CPUE) could be calculated and used as an index of
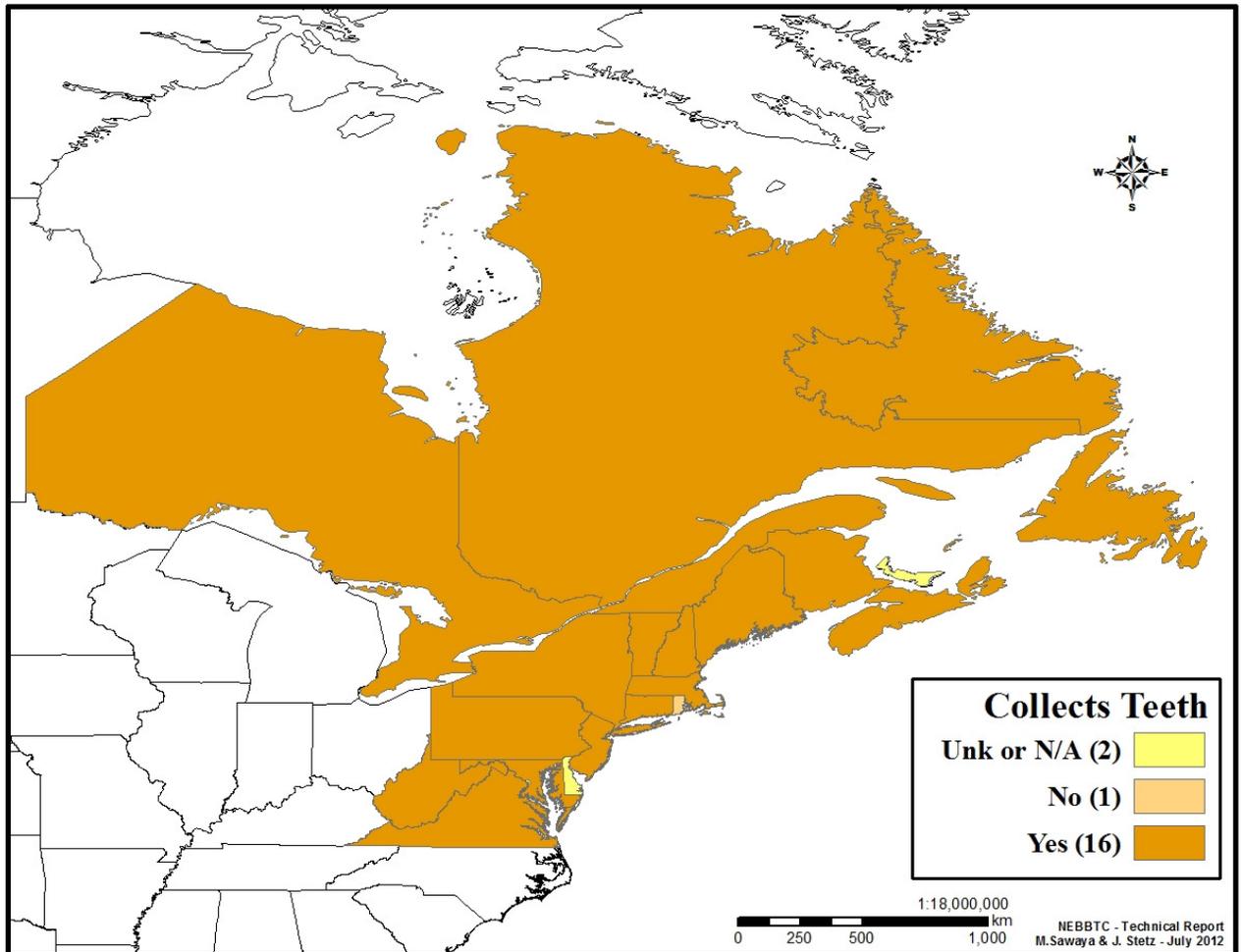
Figure 8.3: *Collection of black bear teeth for cementum annuli analysis by NEBBTC jurisdiction (Noyce 2012).*

abundance, as it is for other species. Variables other than population abundance affect hunter

success (e.g., weather, duration of hunt, methods allowed) and would have to be included in any

CPUE models, but positive relationships have been demonstrated for moose (Schmidt et al.

2005). However, analyses of fishery data indicate CPUE can remain stable while abundance

declines (Harley et al. 2001). Furthermore, even under the best conditions, precision of CPUE

indices of abundance is generally low compared with other estimators of abundance (Harley et

al. 2001).

8.1.2 Direct Estimates of Abundance

Although indices may be suitable for some purposes, direct enumeration of population size may

be more suitable to support management objectives. For example, maximum sustainable yield

harvesting requires an estimate of population size (N). The simplest population enumeration

concept is a census or total count, whereby every animal in the population can be observed and

counted. In that special case, the detection or capture probability (p) is 1, sometimes referred to

as perfect detection. In most instances, however, detection probabilities are not perfect ($p < 1$)

and only a proportion of the population is captured or detected (C). In those cases, population

size can be estimated if that proportion (p) is known, using the basic relationship $N = C/p$.

Consequently, almost all population estimation methods focus on the estimation of p.

8.1.2.1 Mark-recapture

Mark-recapture is used to estimate p, enabling the estimation of a number of population

parameters including abundance, density, population growth, and apparent survival (true survival

times probability of animal remaining in the population). For estimating black bear abundance

(Figure 8.4), this technique begins with first taking a sample from the population, marking the

animal in a unique way (e.g., ear tags, biomarking, radio collars, DNA) and later recapturing

those animals, or recovering them in the case of harvest samples, with the same or different

detection methods. In its simplest form, the proportion of marked animals recovered in the

recaptured sample is *p*.

Mark-recapture models are based on a number of assumptions, including that the

population is closed to additions or deletions, marks are not lost and are read correctly, and all

animals have the same probability of capture. Biases may be difficult to discern, but can be
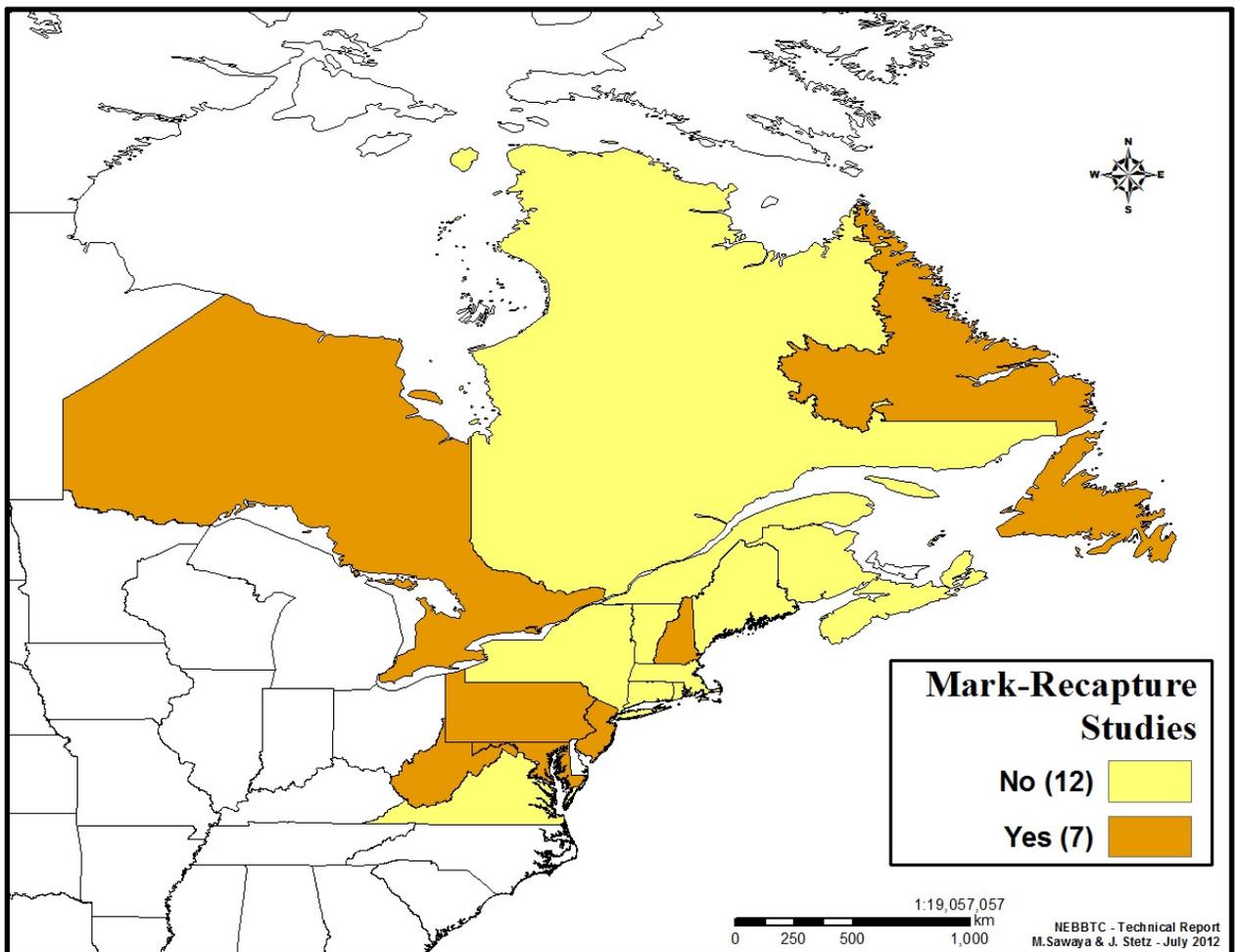
Figure 8.4: *Use of mark-recapture methods to estimate black bear abundance by NEBBTC jurisdiction (Noyce 2012).*

prevalent in even large scale mark-recapture black bear studies (Garshelis and Noyce 2006).

Violation of some or all of these assumptions is common in bear studies (e.g., ear tag loss, mortality, trap shyness) and sophisticated methods have been developed to estimate or avoid such biases. For example, open population estimators have been developed when geographic (i.e., immigration and emigration) or demographic (births and deaths) closure violations occur between sampling occasions (Jolly 1964, Seber 1965).

A variety of techniques have been used to mark black bears for mark-recapture purposes. Live capture is a common form of initial marking and recapturing (Figure 8.5). Live capture is

Figure 8.5: *Marked black bear live-captured in culvert trap.*
Photo credit: West Virginia Division of Natural Resources

relatively expensive but immobilized bears can be fitted with tattoos, ear tags (Figure 8.6), and

radio collars and age and sex data can be collected along with other individual attributes that can

be useful for estimating capture probabilities or for other purposes. Live-capture studies for

bears generally mean small sample sizes and limited geographic extent. However, an exception

is Pennsylvania, where a range-wide mark-recapture effort has been ongoing since 1980

involving >800 marked (i.e., ear-tagged) animals per year. Pennsylvania is 1 of 7 NEBBTC

jurisdictions that have conducted mark-recapture studies to estimate black bear population

parameters (Figure 8.4).

Figure 8.6: *Marking black bear with uniquely numbered ear tag.*
Photo credit: Pennsylvania Game Commission

Biomarkers such as tetracycline and radioisotopes hold promise for mark-recapture studies. Radioisotopes are effective markers but, where bears are hunted, there is concern about health risks posed by consuming meat of marked bears. Tetracycline, which fluoresces in bone tissue under ultraviolet light, offers a non-hazardous alternative. For example, tetracycline baits have been used in Minnesota and Michigan to estimate statewide bear populations (Garshelis and Visser 1997, Belant at al. 2011). Tooth or rib samples were obtained from hunter-killed bears and examined under a microscope to detect the tetracycline. Cautions include the failure of tetracycline to fluoresce in some tooth samples (because of inadequate dosage or slow growth in some seasons and in old animals), markers fading over time in bone samples, and bears emigrating from the sampled area, all of which positively biases estimates of population size. Further, if non-target species take a significant proportion of the baits, population estimates will be positively biased. Wide spacing between baits is necessary to ensure that individual bears do not consume >1 bait. Also, animals that are more prone to consume tetracycline baits may also be more prone to harvest, thereby introducing bias (Garshelis and Noyce 2006). The method is attractive because almost all NEBBTC jurisdictions occupied by black bears allow hunting (Figure 5.3), which would enable easy access to recapture samples. Unfortunately, biomarkers do not enable researchers to individually identify animals, which limit the choice of population estimators that can be used. Biomarker projects in New Hampshire and New York were unsuccessful because of low bait consumption and insufficient marking of bears (A. Timmins, New Hampshire Fish and Game Department; J. Hurst, New York Department of Environmental Conservation; personal communication). Biomarkers are not currently used by any NEBBTC jurisdictions to study bear populations.

Another method for marking bears is the use of visually observable marks, the most common being colored streamers attached to the ears of the bear (Mace et al. 1994, Martorello et al. 2001). Live-captured bears are marked and later observed at camera stations, with capture histories being generated from the photos. Advantages of the technique are that the remote cameras are relatively inexpensive to operate and it does not result in the avoidance behavior associated with live trapping. Drawbacks include the inability to identify individuals because of poor picture quality or the head position of the bear, streamers can break or fall out, and the method may raise ethical issues about encumbering an animal with such tags and the undesirable aesthetics to wildlife observers. Observations of bears by airplane have also been used with mark-resight techniques to estimate bear abundance (Miller et al. 1998), but that technique is largely infeasible in the NEBBTC region because of a predominantly closed forest.

Mark-recapture methods based on DNA extracted from bear hair or scat samples have received widespread use in recent years for estimating population abundance throughout North America (Woods et al. 1999, Mowat and Strobeck 2000, Boersen et al. 2003), including 7 NEBBTC jurisdictions (Figure 8.7). This is largely due to technical breakthroughs in the 1990s based on polymerase chain reaction, enabling small amounts of DNA from hair or scat to be amplified and then genotyped (Foran et al. 1997). Microsatellites have been the most commontype of marker used to identify individual black bears for DNA-based mark-recapture studies. Woods et al. (1999) devised a hair trap, also known as snares or corrals, by stringing barbed wire around a series of trees to form an enclosure around a baited center (Figure 8.8). This type of sampling is often referred to as noninvasive genetic sampling because, following medical terminology, biological samples are obtained without breaking the skin.

Noninvasive genetic sampling, however, is not without its pitfalls. Hair and scat samples are of poor quantity and quality compared with blood or tissue samples, thus they may be prone to genotyping errors (allelic dropout, false alleles) that can lead to animals losing their "marks" and overestimating the number of individuals from incorrect genotypes (Taberlet et al. 1996). Also, if marker power (i.e., genetic diversity) is insufficient, individuals may not have unique genotypes, leading to underestimates of abundance (Mills et al. 2000). Methods have been developed to identify and remove genotyping errors from datasets (Taberlet et al. 1996, Paetkau 2003), substantially reducing the influence of these errors on population estimates.
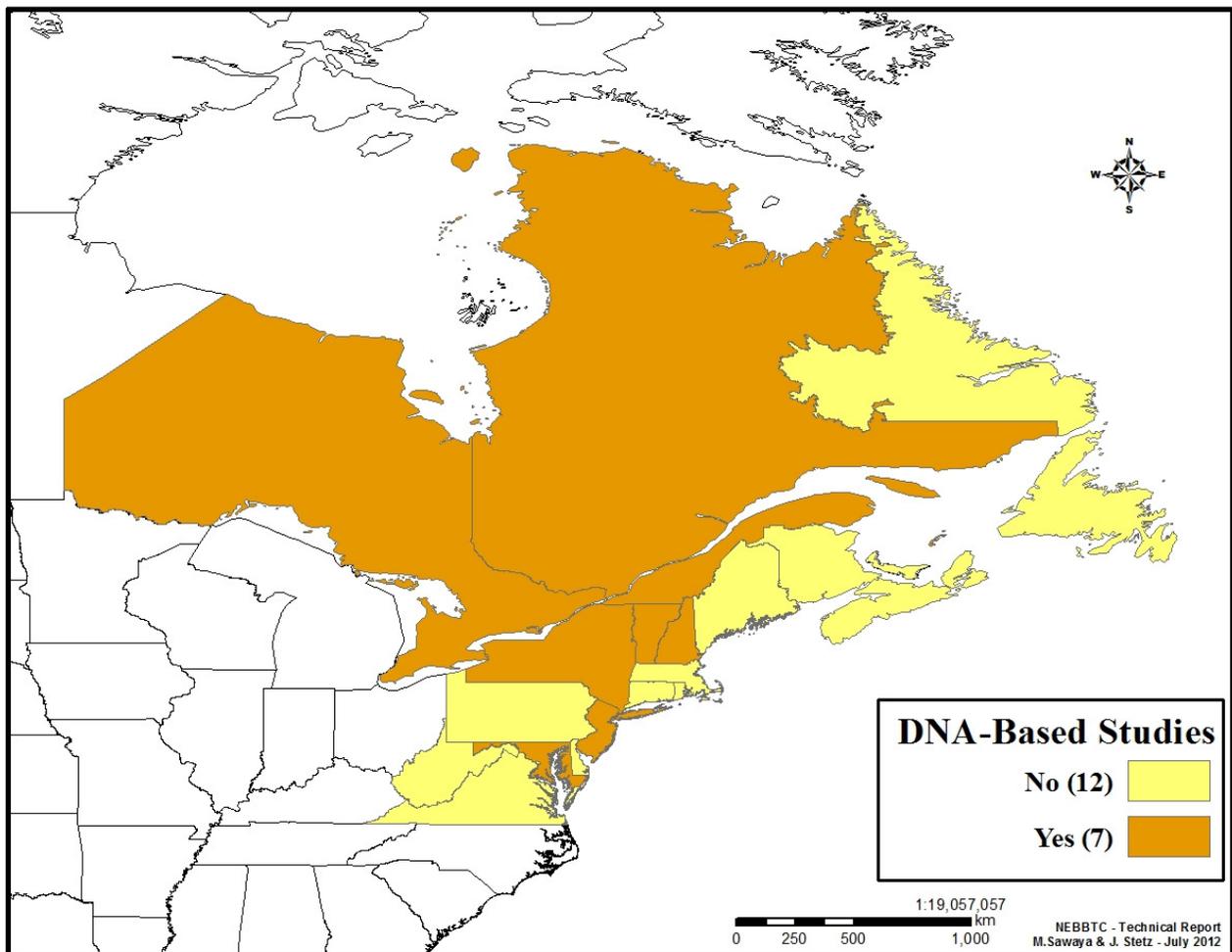
Figure 8.7: *Use of DNA-based mark-recapture to estimate black bear abundance by NEBBTC jurisdiction (Noyce 2012).*

Laboratory issues aside, insufficient sampling and capture biases can lead to erroneous or imprecise estimates of population parameters (Waits and Leberg 2000; Boulanger and McLellan 2001; Boulanger et al. 2004*a,b*; Settlage et al. 2008, Laufenberg et al. 2013). Black bears are particularly prone to some capture and sampling biases and these issues should be taken seriously when designing mark-recapture studies. Clearly, the use of simple Lincoln-Peterson estimators of bear abundance is no longer justified because of the inability to cope with sampling biases. Pilot genetic sampling studies should be conducted to ensure the desired genotypic discrimination can be achieved (Kalinowski et al. 2006, Settlage et al. 2008). Wildlife managers

Figure 8.8: *Bear hair trap used to collect hair for genetic analysis and DNA-based mark-recapture estimates. Photo credit: M. Sawaya*

may be reluctant to embrace genetic monitoring methods because unfamiliarity with methods and models (Schwartz et al. 2007). Stetz et al. (2011) developed an online resource for managers to bridge this barrier (http://alaska.fws.gov/gem/mainPage_1.htm).

Although hair snagging (sometimes also referred to as snaring) with barbed wire has become the standard DNA collection method for black bear studies (Long et al. 2008), bear scat can provide a source of DNA from shed epithelial cells as well. Studies show the use of scat detection dogs greatly improves efficiency of scat surveys (Long et al. 2007, Wasser et al. 2007), but low microsatellite amplification rates can still severely limit detection probabilities. Considerable effort has been directed to identifying the best methods for scat collection (e.g., swab of epithelial cells from surface of the scat) and storage (Murphy et al. 2007), but with current technologies capture probabilities typically remain too low to use scat alone for abundance estimation. Consequently, genetic analyses are more efficient and reliable (i.e., fewer genotyping errors) for hair compared with scat (Roon et al. 2003).

DNA-based mark-recapture studies on black bears using hair collection have been conducted throughout North America, including several in the Northeast (Tredick et al. 2007, Obbard et al. 2010, Wegan et al. 2012; see Appendix C for summary). Clevenger and Sawaya (2010) used barbed wire strung across wildlife crossing structures to collect hair from traveling bears and this method could be adapted to any types of known crossing location. Hirth et al. (2002) found ample black bear hair for genetic analysis on bark and broken twigs of crab apple trees (*Malus pumila*) when bears were climbing trees to eat ripening fruit in fall. They suggested that in the Northeast sampling in apple orchards could potentially replace or augment DNA collection from hair traps.

One potential concern with hair sampling studies is that hair samples from >1 bear can become mixed on barbed wire (Roon et al. 2005). As part of a large laboratory test, Kendall et al. (2009) submitted >800 blind samples, including 115 intentionally mixed samples consisting of hair from closely related (i.e., full siblings) bears. Their results were conclusive in that no genotyping errors were made. Further, black bear hair samples are abundant enough that mixed samples can be discarded or replaced by another sample. Mixed samples should not constitute a major problem as long as standard laboratory protocols and error checking procedures are used (Paetkau 2003). A hair sampling device to sample only 1 bear was developed by Immell and Anthony (2008) but, to our knowledge, has not been used beyond that study.

Similar to other bear species, American black bears rub on trees, posts, and other objects (Figure 8.9). Although this behavior is not fully understood, it provides an opportunity to collect high-quality hair samples for use in mark-recapture studies. The first large-scale application of this sampling method used rub trees to collect grizzly bear hair in Glacier National Park, Montana (Kendall et al. 2008, 2009). Researchers also used bear rubs to estimate grizzly and black bear abundance in Banff National Park, Alberta (Sawaya et al. 2012) but found very low detection rates for black bears relative to grizzlies. Considerably greater detection rates were documented, however, for black bears with bear rubs in Glacier National Park (J. Stetz, unpublished data). However, sampling biases, if any, associated with DNA collected from rub trees compared with samples obtained from more systematic sampling methods such as hair snares remain poorly understood.

Although hair traps have become the standard method for hair collection, variation in study design is common and may affect parameter estimates. We note that many of these issues are concerns for any mark-recapture study, not just genetic sampling. For example, black bear

researchers have used both rewarding (e.g., pastries, donuts) and non-rewarding lures (i.e., scents) to entice bears to enter hair traps. Rewarding lures have commonly been used in eastern North America to attract bears to sites, but bears may exhibit a positive behavioral response, which can result in negatively biased abundance estimates if not modeled appropriately. Researchers in the western U.S. and Canada have used a mixture of rotten cattle blood and decomposed fish with success, but recapture probabilities are lower than rewarding lures, which can lead to problems in modeling capture heterogeneity (i.e., differences in capture probability among individuals not related to previous capture). Behavioral biases caused by lures likely are more easily modeled than the more insidious heterogeneity biases that can go undetected if

Figure 8.9: *Female black bear with cubs using a bear rub in northeastern Pennsylvania. Photo credit: B. Higgins*

capture probabilities are low. Regardless, the ability to recapture animals is central to mark-recapture studies and the effects of lure or bait on detections should be further explored.

Researchers may select different heights of wire depending on the physical characteristics (i.e., body size) of bears in the sampled population. Studies targeting grizzly bears have almost universally set the wire at 50 cm to allow cubs to pass under the wire, but still allow for high detection rates of adults. However, research has shown that hair samples from even cubs of the year are also collected with hair traps (Kendall et al. 2009). Ideally, every adult bear that enters a hair trap would leave hair as they pass over or under the wire, but a number of studies have documented lower detection rates for males than females. One contributing factor for this difference may be that males have different molting patterns or schedules or males may be tall enough to step over the wire without leaving hair. Because male detection probabilities are often lower than females at hair traps, some researchers have used two strands of barbed wire set at approximately 40 and 60 cm in an attempt to capture more males (Drewry et al. 2013; J. Clark, U.S. Geological Survey, unpublished data). As noted previously, it is vital to know what segments of a population are being sampled to ensure estimates are properly interpreted.

The density of hair-sampling sites on the landscape can also affect detection probabilities. One assumption for mark-recapture studies is that all animals have the same probability of capture. This is rarely realistic, of course, but studies to estimate population size should be designed to ensure that all animals have at least some opportunity to be detected. Thus, at a minimum, trap spacing should be no greater than the smallest seasonal home-range diameter of bears within the sampled area (Boulanger et al. 2004, 2006). Summer home ranges of females are thus useful for guidance because black bear home ranges are smaller for females than males and because most hair sampling surveys take place in summer. In eastern North America, where

bear home ranges are relatively small, this means that hair traps often need to be only a few kilometers apart. Additionally, because bear densities in the East can be high, obtaining adequate capture probabilities requires a high density of hair traps (Settlage et al. 2008). In high-density areas, the number of sampling sites that need to be maintained can be overwhelming. Also, the number of hair samples collected may be high and overtax the budget for DNA analysis. In those cases, DNA analysis of only a subset of the total number of hair samples collected may be an option (Tredick et al. 2007, Settlage et al. 2008, Dreher et al. 2009). However, subsampling may reduce capture probabilities so determining an appropriate level of subsampling is important (Laufenberg et al. 2013). To reduce the probability of obtaining duplicate samples during a sampling interval, many researchers have used subsampling methods whereby only 1 sample from a site-week combination was used (e.g., Settlage et al. 2008). Such subsampling is statistically sound in theory but may result in biases against bears travelling in family groups (i.e., females with cubs or yearlings), and should be acknowledged. Also, subsampling might make it more difficult to model behavioral biases because the first occasion that an animal is captured may actually be a recapture of an animal whose hair was previously collected but not genotyped (Laufenberg et al. 2013).

Another consideration is whether or not to move hair traps between sampling sessions. Leaving sites in place and rebaiting them takes considerably less work than moving them, but capture probabilities are generally greater when sites are moved (Boulanger et al. 2006). The tradeoff between labor costs and detection probabilities is complicated by the type of lure used. Sites with rewarding lures may have greater detection rates when they are not moved as a positive trap response bias may increase over time.

Many other variables may influence hair trap capture probabilities, including weather conditions that can affect sample quality and lab standards for genetic analysis. Previous live captures can negatively affect capture probabilities with hair traps as previously live-captured bears may develop wariness of similar sites (Boulanger et al. 2008, Kendall et al. 2009). Although study design and environmental factors may pose significant problems for mark-recapture studies, sampling variation in estimates of black bear abundance can be distinguished from the real stochastic changes in population level (i.e., process variation) with careful study design and statistical analysis.

A wide variety of mark-recapture models currently exist to estimate black bear population abundance and each have their own strengths and weaknesses (Appendix D). Many methods use procedures to account for the biases prevalent with sampling bears (e.g., trap response bias, unequal capture probabilities, lack of population closure). Other models have been developed to provide more precise estimates of abundance (e.g., multi-method models, models that allow individual covariates). The choice of which method to use depends on the objectives, budget, and biological realities of the study and, therefore, should be determined prior to initiating fieldwork. Because of relatively large home ranges of bears, one of the greatest challenges in using closed population models is violation of the assumptions of geographic closure (Boulanger et al. 2001). Sampling large study areas can reduce such violations but that can mean an intensive sampling effort. Use of open population models that relax the assumption of closure, such as Jolly-Seber (Jolly 1965, Seber 1965) or Pradel (1996) models, can be effective as well (e.g., Clark and Eastridge 2006).

Capture heterogeneity is a major concern with mark-recapture abundance estimates for bears because it is prevalent and difficult to estimate (Boulanger et al. 2004b). For example,

larger bears may be able to step over the barbed wire at a hair snare, resulting in lower capture

probabilities than smaller bears which, if not detected, would produce an abundance estimate that

is biased low. A variety of methods have been developed to detect and account for this bias

(e.g., Pledger mixture models, Huggins individual heterogeneity models, Jackknife models) but

they may not perform well when capture probabilities are low (Huggins 1991, Pledger 2000,

Boulanger et al. 2004, Laufenberg et al. 2013). The effect of capture heterogeneity can also be

reduced with the use of multiple sampling methods (Dreher et al. 2007, Boulanger et al. 2008).

For example, samples from hair traps have been used in conjunction with hair collected at bear

rub trees and wildlife crossing structures (Sawaya et al. 2012), from nuisance bears (Kendall et

al. 2009), and from harvest samples (Dreher et al. 2007). Samples from harvested bears might be

a readily available method for recapturing previously marked bears for some jurisdictions.

Individual covariates should be considered whenever possible (i.e., Huggins models) as

they also improve precision and reduce bias of abundance estimates. For example, it is now

common to use covariates such as each individual's average distance to the edge of the sampling

grid (Boulanger and McLellan 2001), history of previous live capture (Boulanger et al. 2004,

2008; Kendall et al. 2009; van Manen et al. 2012), and time-varying sampling effort (Kendall et

al. 2009, Sawaya et al. 2012) to improve model performance.

Mark-resight models have seen rapid improvements recently with more powerful and

flexible maximum likelihood methods being readily accessible to researchers and managers (e.g.,

Program MARK, McClintock and White 2011). These new methods can make use of detections

of animals that are unmarked, marked, individually marked, and combinations of the three.

Beyond abundance, mark-resight methods can estimate apparent survival and transition

probabilities (e.g., on or off the study area) within a robust design framework (Section 9.1.2.).

In addition to substantial developments in mark-recapture analysis techniques in recent decades, software programs have improved accordingly and facilitated application of newly emerging techniques to estimate population parameters (Appendix E). Most notably, Program MARK offers a user-friendly interface to develop models based on maximum likelihood estimation methods (White and Burnham 1999). Program MARK uses information-theoretic methods based on Akaike's Information Criterion (AIC) that enable users to compare models, test hypotheses, and average parameter estimates across multiple models (Burnham and Anderson 2002). Specialized packages for Program R are being developed rapidly (e.g., secr, RMARK) that may provide more power and flexibility. However, most R packages are command-driven and less user-friendly, although exceptions exist, such as WiSP (Zucchini et al. 2007), which uses dropdown menus.

8.2 DENSITY

Density is simply population abundance (N) divided by the area sampled (A). Despite the simplicity of the concept, determining the actual area sampled by the traps or other capture devices is difficult because animals spend time outside the immediate sampling area, particularly those that reside on the edge of the study area. This area has historically been estimated by buffering the sampled area by the mean maximum distance moved (MMDM) or ½ MMDM, by individual animals detected in the sample (Karanth and Nichols 1998). This approach, however, has been criticized as not having any true statistical foundation and, therefore, may produce biased density estimates (Efford 2004). The difficulty of reliably estimating population density is a topic that has seen much interest and development in recent years (e.g., Efford 2004, Royle and Young 2008, Foster and Harmsen 2012, Noss et al. 2012).

8.2.1 Spatially Explicit Capture-recapture

Spatially explicit capture-recapture (SECR) models have been developed that combine elements of distance sampling with mark-recapture models (Borchers et al. 2010). Non-spatial capture-recapture models are based on encounter histories that are generated for temporal sampling occasions but ignore the spatial location of trap sites. SECR models use the spatial distribution of trap sites to estimate home-range size and detectability, assuming that the probability of detection is greatest in the home range center and detectability decays as a function of distance from the center (see Borchers [2010] for a nontechnical review of SECR models). Like many
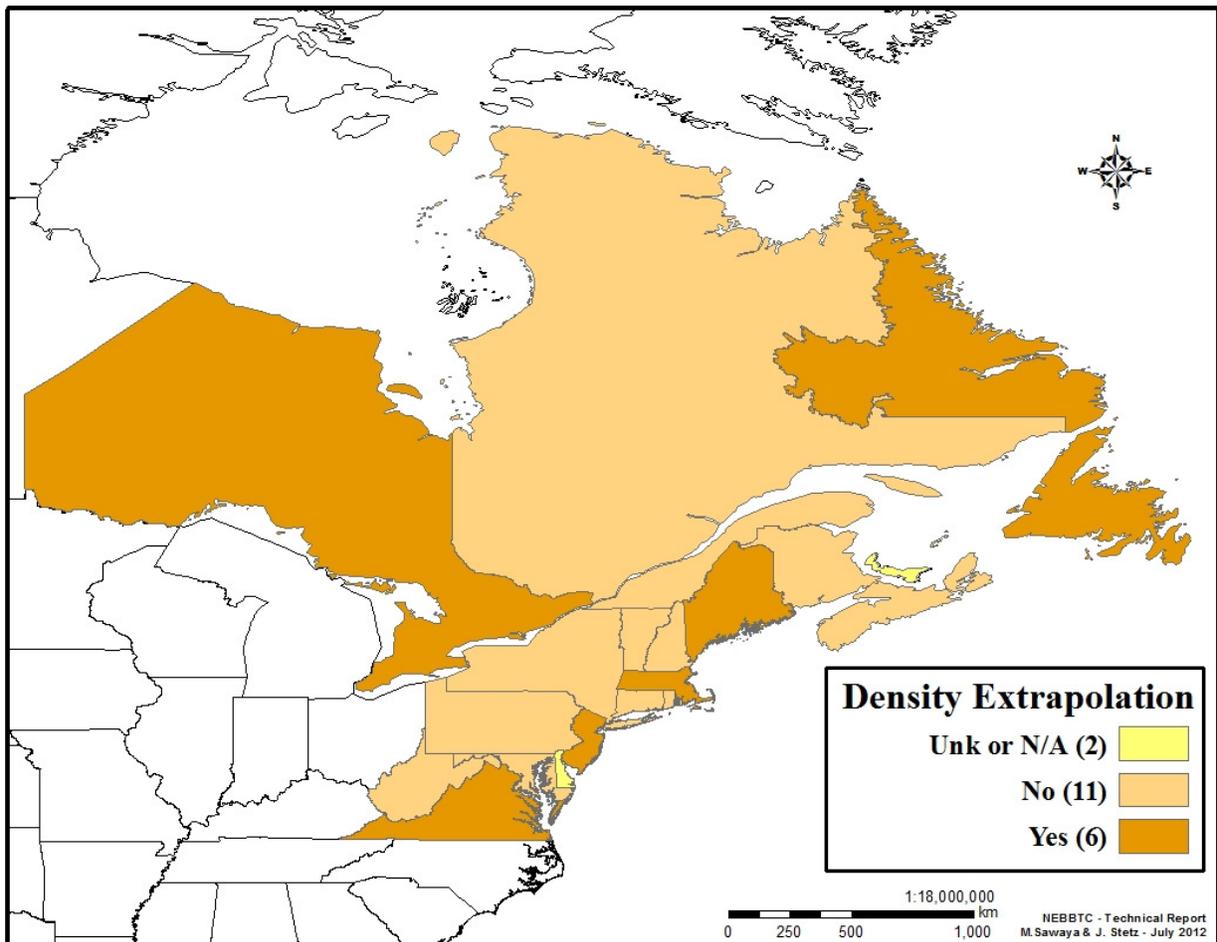
Figure 8.10: *Use of density extrapolation methods for estimating black bear abundance in NEBBTC jurisdictions.*

nonspatial population estimators, SECR models use maximum likelihood estimation methods to estimate detection probabilities. Bayesian models have recently been developed using Markov Chain Monte Carlo (MCMC) methods to estimate black bear density (Gardner et al. 2010). Hierarchical SECR models have also been used to estimate density of black bears (Royle and Young 2008, Gardner et al. 2009).   Obbard et al. (2010) conducted a comparison of density estimators for black bears in Ontario and concluded that density estimates from SECR models were lower and presumably less biased than estimates from non-spatial mark-recapture models.

About 1/3 of NEBBTC jurisdictions have used density extrapolation methods to estimate abundance (Figure 8.10). Density extrapolation methods (i.e., extrapolating density to unsampled areas based on known correlation with surrogate information such as habitat) can be used to estimate black bear abundance if the association between density and the surrogate is well-known for the area and scale of interest (Mowat et al 2005). Extrapolating density can lead to false inference, however, if the correlation with habitat is not explicitly modeled or poorly understood. One important advantage of SECR models is that the correlation of density at individual trap sites with habitat covariates can be directly integrated into the estimation process, enabling researchers to predict density in areas not sampled (Drewry et al. 2013).

8.3 ABUNDANCE AND DENSITY SIMULATIONS

It is usually impossible to discern the degree of bias in an estimate from a field study and even the most precise estimates can be severely biased (Figure IV.1). Therefore, simulations are typically used to estimate and compare the potential bias and precision of estimation methods and study designs. In such simulations, populations with known characteristics (e.g., known abundance) are created by the user, and then "sampled" according to the prospective study

design. For example, Boulanger et al. (2004*a*) used simulated detection data in Program

CAPTURE to estimate the bias of grizzly bear abundance estimates in hair trapping studies due

to the heterogeneity in cub capture probabilities. They were able to evaluate the performance of

multiple study designs (i.e., size and number of grid cells with hair traps) and thereby make

recommendations on study design that reduce this form of heterogeneity. Simulation studies

such as those have become instrumental in designing bear research and monitoring studies (e.g.,

Boulanger et al. 2008, Stetz et al. 2010, Laufenberg et al. 2013).

8.3.1 Abundance

8.3.1.1 Mark-recapture Abundance Simulations

We evaluated a number of black bear mark-recapture study designs by conducting closed-

population abundance simulations using estimates of detection probability spanning the range

found in the primary literature, focusing on studies conducted in the Northeast (Appendix C).

Using Program R (R Development Core Team 2005) package WiSP (Wildlife Simulation

Package; Zucchini et al. 2007), we simulated populations ranging in true abundance from 100 to

900 in intervals of 100 individuals (Table 8.1) within study areas of several different physical

dimensions, each with uniform density. We assumed that sampling effort was constant across k

occasions ($k = 5$, 7, or 10), depending on the particular simulation. This is reasonable as most

mark-recapture studies deploy the same number of traps each occasion, although the number and

length of occasions may vary. We used a minimum per-occasion capture and recapture

probability of 0.005 (assuming that all bears had at least some opportunity to be detected), a

maximum per-occasion value of 0.5, and we assumed no improvement in detection across

occasions (i.e., no behavioral response). All simulations used a model, M_h, to allow for the

heterogeneity in detection probabilities imposed by the simulated sampling design (Otis et al. 1978). We derived nonparametric bootstrap 95% confidence intervals with 99 runs. We calculated percent relative bias (PRB) as the difference between the estimated parameter value and truth (i.e., the value used to generate simulated data; PRB = [(estimate - truth) / truth] × 100%). To assess performance, we estimated the average relative bias and coefficient of variation across replicates and assessed CIC. Further details of these simulations can be found in the annotated R code (Appendix F).

Table 8.1. Population and sampling parameters used in closed population abundance simulations with WiSP package (Zucchini et al. 2007) in program R. Minimum and maximum capture probabilities were per occasion (k). Not every combination was run because of computational limitations.

Study area dimension (no. grid cells east-west)	Study area dimension (no. grid cells north-south)	Population abundance (N)	No. sampling occasions (k)	Min. capture probability (p)	Max. capture probability (p)
100, 200	100, 200	100–900	5, 7, 10	0.005	0.5

For those simulation scenarios with adequate data for models to converge, estimates generally followed predictions, with decreasing bias and increasing precision as true population abundance and the number of sampling occasions increased (Figure 8.11). One exception, however, was decreasing CIC due to overly precise estimates as abundance increased with the smaller study area scenario. For all but the sparsest scenarios, CV remained below 20%, and was rarely >10% for populations of ≥200 animals. These results suggest that, across a range of population sizes, the detection probabilities achieved in black bear mark-recapture studies in the Northeast have been adequate for robust abundance estimates. As expected, larger study areas

produced less biased and more precise estimates than did smaller study areas. However, near nominal coverage was achieved for the majority of scenarios and bias rarely exceeded 5% for populations of ≥300 animals.

Higher detection probabilities and more complex models would likely result in even more precise estimates; however, such data (e.g., mixture probabilities) were rare and likely too specific to a particular region or study to be useful in simulations. In general, studies should be designed to minimize closure violation and maximize detection probabilities. With this in mind, we explore factors that may determine detection probabilities later in this chapter.
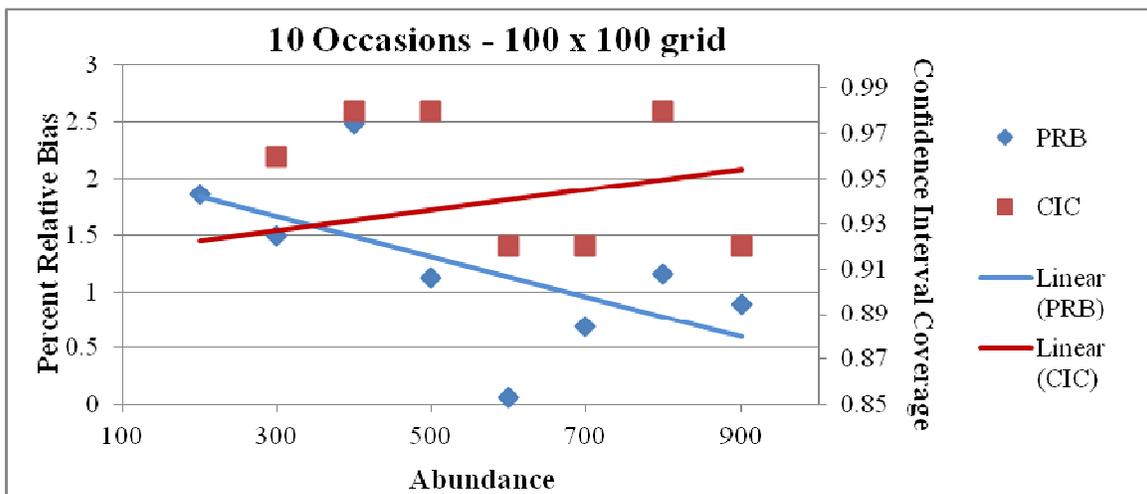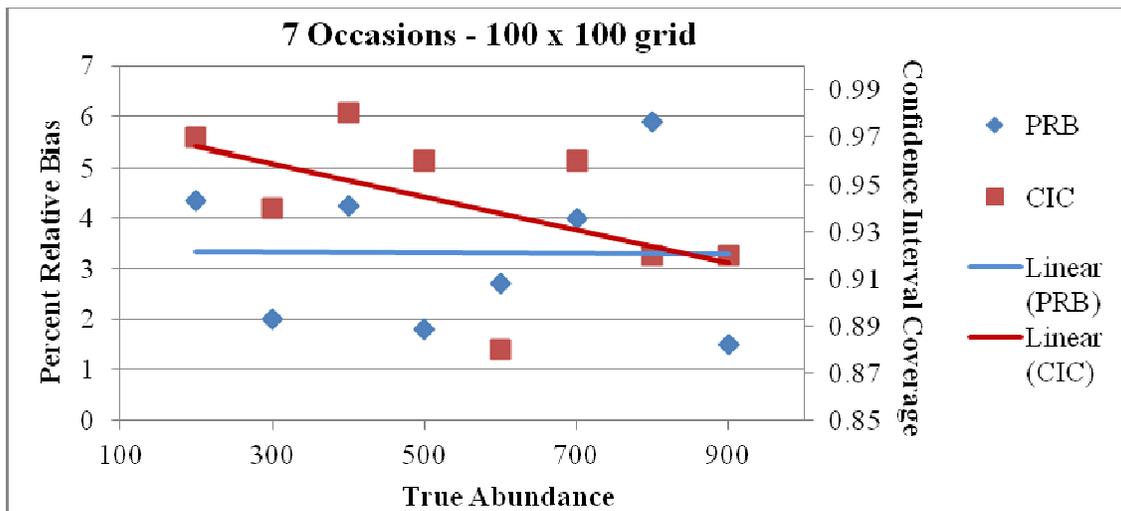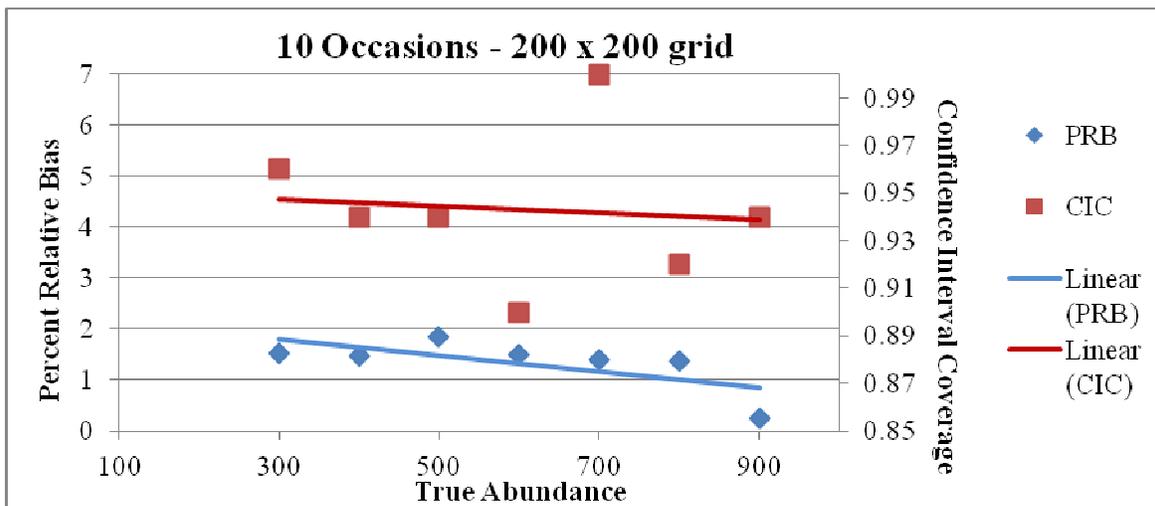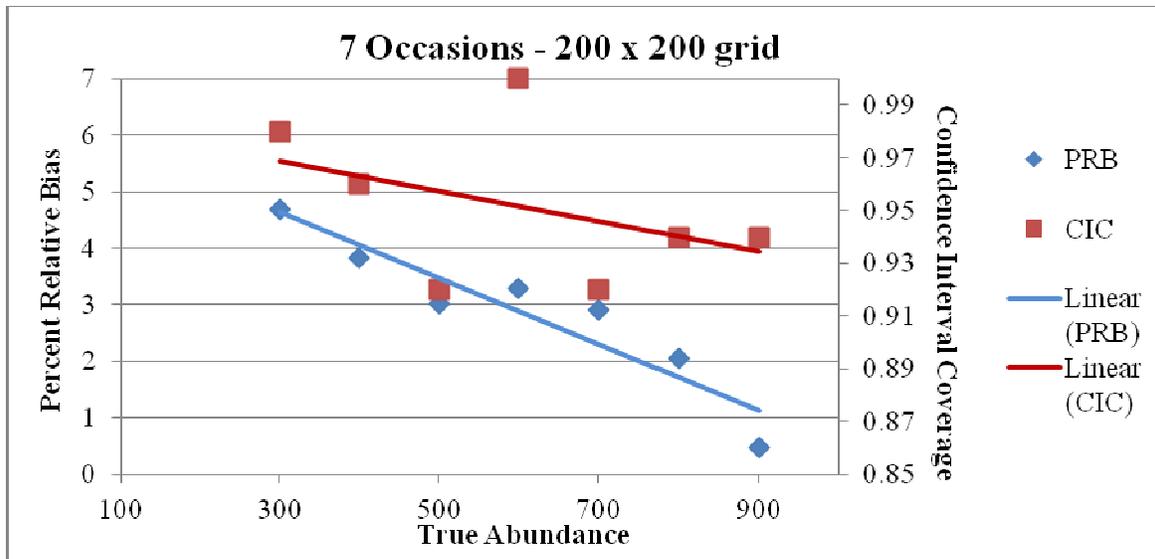
Figure 8.11: *Relative bias of abundance estimates as a function of the number of sampling occasions, true abundance, and study area dimensions. Simulations performed with WiSP package (Zucchini et al. 2007) in program R. PRB = percent relative bias; CIC = confidence interval coverage.*

8.3.1.2 Mark-resight Abundance Simulations

Because some NEBBTC jurisdictions obtain data from a large number of live-captured bears, we

explored the potential to use mark-resight with the Poisson log-normal estimator (PNE;

McClintock and White 2009). The PNE model requires individually identifiable marks, but does

not require that the number of marks be known (i.e., in case of emigration from the study area),

although the number of marks is often determined via telemetry prior to camera surveys. As

with many mark-recapture models, the assumption of geographic closure may be relaxed with

the PNE model, but abundance estimates should be interpreted cautiously. Model parameters

and results of simulations are reported in Table 8.2. Over the suite of parameters considered,

model performance was generally poor in terms of CIC because of negative bias in abundance

estimates with likely underestimated error, although performance may improve with larger

population sizes or changes in other parameters. Despite these limited results, these new classes

of mark-resight models have shown promise in a number of empirical studies (McClintock and

White 2011). They provide a major advancement over traditional mark-resight (e.g., Mace et al.

1994) by allowing multi-model inference, model averaging, and the use of covariates, in addition

to the greater flexibility as discussed previously (McClintock and White 2011).

Similar to mark-recapture models (Boulanger et al. 2008), it may be possible to combine

resighting events from multiple data sources to improve bias and precision of abundance

estimates. In addition to visible marks for use with remote cameras, genetic samples collected

during live capture may be paired with detections from hair traps or bear rubs (Chapter 11) to

create single encounter histories for use with mark-resight models. Combining detection data

from multiple sources reduces bias from any single method, and has been shown in simulation

and empirical studies to improve abundance estimates (Dreher et al. 2007, Boulanger et al. 2008, Sawaya et al. 2012).

Table 8.2. Parameter definitions and values used in mark-resight simulations in program MARK.

	Simulation inputs		Estimates from simulations		
Parameter	**Definition**	**Value**	**PRB**[a]	**CV**[b]	**CIC**[c]
N_{male}	Abundance (M)	300	-27.7%	9.1%	5.6%
N_{female}	Abundance (F)	300	-21.2%	9.2%	23.7%
n_{male}	No. known marks (M)	100			
n_{female}	No. known marks (F)	100			
Σ	Individual detection rate heterogeneity	0			
α_{male}	Mean detection rate (M)[d]	0.55			
α_{female}	Mean detection rate (F)	0.5			
U_{male}	No. unmarked individuals (M)	200			
U_{female}	No. unmarked individuals (F)	200			
Φ_{male}	Apparent survival (M)	0.85			
Φ_{female}	Apparent survival (F)	0.9			
γ''	Transition probability[e]	0.05			
γ'	Probability of not transitioning to previous state	0.5			

[a]PRB = percent relative bias ((estimate – truth) / truth) × 100%.
[b]CV = coefficient of variation.
[c]CIC = percent of simulations with confidence interval including true abundance.
[d]Mean detection rate for primary sampling occasion.
[e]For this simulation, the probability of leaving the study area (i.e., transition from observable to unobservable state).

<u>8.3.2 Density</u>

8.3.2.1 Spatially Explicit Mark-recapture Density Simulations

We used simulations in Program R to evaluate the performance of SECR methods to inform study design for black bear density estimation in the NEBBTC region. We conducted experiments covering a range of plausible sampling scenarios and population parameters based on the primary literature (Appendix C), focusing on the maximum likelihood approaches of Borchers and Efford (2008). To perform the simulations, we used the *secr* package (Efford 2012) in Program R to generate and sample populations, then derive estimates of density, from which we assessed bias and precision relative to true density. Again, we estimated bias as the average PRB across replicates and precision based on the average CV and average CIC. Further details of these simulations can be found in the annotated R code (Appendix G).

We conducted a large number of SECR simulation scenarios resulting in approximately 1,400 combinations of parameters (Table 8.3), which are available in a relational database from the authors. Results of SECR simulations were generally similar across the two grid sizes we considered, with the biggest exception being that data-rich scenarios (i.e., high density) consistently failed because of computer memory constraints for the area based on 25×25 grid cells. Similarly, for the 10×10 grid, we excluded spurious results from several low-density scenarios from our discussion.

Generally, our results indicated the greatest bias in density estimates, both positive and negative, occurred for low-density populations, with smaller home ranges (σ), and with greater spacing of sampling sites. Specifically, low-density populations were more likely to produce positively biased estimates, even with high detection rates, with large home ranges relative to site spacing. Negative bias was also associated with small home ranges in conjunction with large site

Table 8.3. Population and sampling parameters used in SECR simulations using the secr package (Efford 2012) in Program R. Not all combinations were run because of computational limitations.

Density (bears/km^2)	$g0$[a]	σ[b] (m)	No. sampling occasions (k)	Grid size	Site spacing (m)
0.1	0.05	400	5	10 × 10	1,000
0.5	0.10	1,000	7	25 × 25	2,000
1.0	0.15	2,000	10		3,000
1.5	0.20	3,000			
	0.25				

[a] $g0$ = average detection probability at the individual's activity center.
[b] σ = shape of the half-normal detection function; we converted σ into an estimate of home-range radius: $(\text{qchisq}(0.95, 2)^{0.5}) * \sigma$.

spacing, even with high detection rates and slightly higher population density. Low CIC (95%) for 10 × 10 grids also occurred with sparse data scenarios, such as low-density populations with small σ and large site spacing, with a lesser effect due to detection probability or number of sampling occasions. Only 21% of scenarios with the smallest home range achieved nominal (95%) CIC, however, these estimates were heavily biased and CIC was achieved simply because of poor precision that resulted in large confidence intervals. We found the same pattern of poor precision for 25 × 25 grids, but with only 15% of scenarios achieving 95% CIC. The number of sampling occasions seemed to be the least influential parameter overall, although more occasions did result in greater precision, particularly among sparse data scenarios. We observed the same patterns for 25 × 25 grids but associated CVs were consistently better than the 10 × 10 grid scenarios.

In summary, the 25×25 grid scenarios with the best performance in terms of bias and CIC were those with lower density and relatively large home ranges; detection probability, site spacing, and number of occasions were less important. The 10×10 grid scenarios were less consistent in terms of the influence of population density and detection rate. However, home-range size, both in absolute terms and relative to site spacing, again was the most important factor in model performance. Thus, given the level of sampling typical of black bear DNA-based mark-recapture studies, our results suggest that SECR models may produce biased and imprecise estimates for populations when home ranges are small.

Because of computer limitations, we acknowledge that the number of replicates may have been too limited for adequate inference of model performance for some scenarios. We note, however, that formal analyses of field data, as opposed to simulations, may produce better estimates by taking advantage of multiple data sources, more advanced models (e.g., mixture models with covariates), and more realistic sampling schemes based on biologists' expertise. Finally, although SECR methods hold great promise for producing more reliable density estimates than traditional mark-recapture methods, techniques are currently undergoing rapid development. Our simulations reflect the current state of this class of SECR model but may need to be revisited as models and data collection improve. For example, recent extension of SECR models to incorporate landscape resistance suggests that models using Euclidian distance between activity centers and sampling sites to estimate sigma may drastically underestimate density (Royle et al. 2013). An empirical comparison of SECR against a traditional approach using black bear genetic sampling in Glacier National Park found no difference in density estimates (Stetz et al. In Press). This was likely a function of sampling a large area (4,100 km^2; approximately 66 times larger than average male home ranges in this region), which served to

reduce edge effects. Again, these models are undergoing rapid growth and require further theoretical development as well as simulation and empirical evaluation

8.3.3 Factors Affecting Detection Probabilities

As we have shown, the combination of detection probability, sample coverage, and population characteristics themselves (e.g., abundance, density, population growth) will determine the bias and precision of parameter estimates derived from mark-recapture studies. We therefore examined the potential explanatory power of a suite of variables on detection probabilities based on estimates in the primary scientific literature (Appendix C). These included whether sites were moved between occasions, if a food reward was used (i.e., positive capture response), total abundance (or density) of the population, and total number of bears detected.

We included studies from beyond the Northeast to increase our sample size, and accounted for potential regional differences by classifying studies as Northeast, Southeast, or West. Because studies may use sampling occasions of different duration, we standardized sampling effort covariates to be a function of the number of sites deployed, the number of occasions, and the length of occasions per unit area. As our interests were qualitative (i.e., not to estimate effect sizes), we do not report beta coefficients here. We predicted that sampling effort and a random effect of each study would be among the most powerful predictors of detection rate. We used both generalized linear and mixed effects models in Program R to identify those factors (e.g., study design, population) that help explain the variation in detection probabilities obtained in black bear studies. We determined model support based on AIC_c scores for mixed effect models and significance of specific variables based on whether the 95% CI around beta coefficients included 0.

Consistent with predictions, mixed-effects models including sampling effort, the random effect of study, and whether sites were moved had the most support based on AIC_c values (Table 8.4). Support for this last factor should be interpreted with some caution, however, as the addition of a single parameter with a corresponding $\Delta AIC_c \approx 2$ does not indicate a strong effect (Arnold 2010). We found that detection probability is mostly influenced by differences among study properties beyond those associated with sampling design or properties of the population itself, although we do not to suggest these other factors are not important.

Table 8.4. Model selection results for generalized linear and mixed effect models of black bear detection probability based on hair sampling.

Model[a]	k	AIC_c[b]	ΔAIC_c	AIC_c weight	log-likelihood
Study + Effort	4	-20.86	0	0.67	15.34
Study + Effort + SitesMoved	5	-18.61	2.24	0.22	15.73
Study	3	-15.18	5.68	0.04	11.11
Effort + Density	4	-14.52	6.33	0.03	12.17
Effort	3	-13.4	7.45	0.02	10.22
Abundance + NumSites	4	-13.02	7.83	0.01	11.42
NumSites × Abundance	3	-11.54	9.32	0.01	9.29
Effort + SitesMoved	4	-11.41	9.45	0.01	10.61
Region + Effort	5	-9.46	11.4	0	11.16
Effort × Density	3	-9.01	11.85	0	8.03
Density	3	-9.00	11.85	0	8.02
Effort × Density + Region	5	-7.11	13.74	0	9.98

[a] Study = (categorical) a random effect of multiple estimates reported by the same authors; Effort = (continuous) total number of site-days per unit area; SitesMoved = (binary; yes/no) whether sites were moved between occasions; Density = (continuous) estimated density of study population; Region = (categorical) general region where study occurred (northeast, southeast, or west); NumSites = total number of sampling sites; Abundance = estimated abundance of study population. A "+" indicates an additive effect of covariates; a "*" indicates an interaction term of covariates.
[b] Akaike's Information Criterion corrected for finite sample size.

Results from fixed-effects models (i.e., removing the random effect of each study) suggested that population abundance and sampling effort were the most important predictors of

detection rate. Specifically, detection had a weak negative correlation with abundance but had a strong positive correlation with sampling effort. Results of these simulations may be used as guidelines for study design, with emphasis on ensuring adequate sampling coverage given the specific properties of the studied population. We note the capture probability principles and methods to increase detection probabilities we outlined here also apply to the mark-resight and spatially explicit capture recapture models described elsewhere in this chapter and in Chapters 9 and 10. We also note that high detection rates do not necessarily equate to robust population estimates. However, high detection rates are instrumental in identifying the most appropriate estimator, which, in turn, contributes to more robust parameter estimates.

8.5 CONCLUSIONS: ABUNDANCE AND DENSITY

- Indices can be useful for decision making when strong correlation with abundance can be demonstrated, but this has yet to be accomplished for black bears.

- Imprecision and unreliability renders indices inappropriate to support crucial decisions where errors can have substantial consequences.

- Indices are most useful for examining fluctuations in long-term population dynamics.

- Direct estimates of abundance are more useful than indices to wildlife managers for making management decisions (e.g., setting harvest quotas) because they provide estimated numbers of animals with associated measures of precision.

- Closed mark-recapture models represent one of the best techniques to estimate abundance for black bears, because of a long history of use and demonstrated reliability.

- A number of marking techniques can be used to create encounter histories for bears, with ear tagging and DNA-based methods being common and effective marking methods.

- Estimators that do not account for potential capture biases (e.g., heterogeneity, behavior) will likely produce biased estimates for black bears.

- Multiple sampling methods should be considered whenever possible as the use of different, independent methods increases precision and reduces bias of abundance estimates. In particular, strategic use of multiple sampling methods can reduce the cost of genetic sampling studies.

- Extremely large study areas can be difficult to adequately sample, particularly when home ranges are small.

- Individual covariates should be considered whenever possible (i.e., Huggins models) as they improve precision and reduce bias of abundance estimates. Managers considering these types of models should be aware that low capture probabilities will reduce precision and increase bias, so capture probabilities should be maximized whenever possible (Section 8.3.3).

- Density often is a more relevant parameter than abundance to wildlife managers, and is a particularly useful parameter to evaluate habitat quality within and between studies.

- Newly emerging estimation techniques based on spatially-explicit capture-recapture models show great promise to account for the spatial location of sampled animals in relation to the sampling grid to adjust density estimates accordingly.

- Our simulations suggest that managers considering spatially-explicit capture-recapture models should be aware that unbiased and precise density estimates require that the sampling area is sufficiently large to adequately represent home ranges.

Chapter 9. Survival and Reproduction

9.1 SURVIVAL

Population abundance is probably the most difficult population parameter to estimate but it is not always necessary for effective population management. Estimates of survival can be quite useful, because survival is essential for monitoring effects of harvest, evaluating regulation changes, and better understanding population dynamics (Sorensen 1998). Furthermore, black bear population growth is particularly sensitive to changes in adult female survival (Beston 2011), and survival is not as sensitive to the capture biases that plague abundance estimates, making robust estimation possible with less cost and effort.

Figure 9.1: Black bear cubs of year.
Photo credit: Maine Department of Inland Fisheries and Wildlife

Table 9.1. Black bear survival and reproductive rate estimates from Northeast Black Bear Technical Committee jurisdictions.

Jurisdiction	SurvAdF	SurvYngF	SurvCub	AgeFirstRepro	AvgLitSize	Fecundity	VitalRateRef
MA	0.87						Cardoza, personal communication*
MA		0.66	0.59	3.70			Elowe and Dodge (1989)*
MA			0.53-0.63				Fuller (1993)*
MA			0.74				McDonald and Fuller (2001)*
ME	0.96	0.78	0.79	4.91		0.58	McLaughlin (1998)*
ME	0.84	0.76	0.65	5.10		0.61	McLaughlin (1998)*
ME	0.96	0.71	0.59	4.47		0.58	McLaughlin (1998)*
NH	0.87		0.74				Timmins (2008)*
NJ	0.94		0.72	3.00			McConnell et al. (1997)*
NJ			0.70				New Jersey (2004) *
ON	0.87	0.78	0.46	7.81			Obbard and Howe (2008)*
ON		0.86	0.44	6.70			Obbard and Howe (2008)*
ON	0.84	0.76	0.53	6.17		0.46	Yodzis and Kolenosky (1986), Kolenosky (1990)*
PA			0.84	3.20	3.00		Alt (1980, 1981, 1989)*
PA	0.59						Diefenbach and Alt (1998)*
PA				3.53		0.62	Ternent and Sittler (2007)*
QC	0.85		0.71	6.00		0.47	Jolicoeur et al. (2006)*
QC	0.96			5.33		0.58	Jolicoeur et al. (2006)*
VA	0.93		0.70	4.00		0.50	Carney (1985)*
VA		0.78	0.72				Hellgren (1988)*
VA	0.87			4.00	2.30	0.57	Hellgren and Vaughan (1989)*
VA	0.73		0.73	3.89		0.66	Kasbohm et al. (1996)*
VA	0.81						Klenzendorf (2002)*
VA		0.87					Lee and Vaughan (2005)*
VA			0.70	2.83		0.69	Ryan (1997)*
VA	0.92	0.90	0.87	3.80	2.50		Bridges et al. (2011)
VT			0.26	5.33		0.87	Hammond (2002)*
WV	0.76	0.79		3.11	2.65		Ryan (2009)

*From Table S1 in Beston (2011)

Black bear survival can be estimated by tracking their fates with radio collars or by mark-recapture techniques, non-invasive or otherwise. Survival typically varies with age and sex and, ideally, survival would be estimated for bears of every age, sex, and reproductive status (i.e., females with and without cubs or yearlings). Fortunately, little information is usually lost by pooling into age categories (e.g., old adults [8+ yrs], young adults [3–8 yrs], juveniles [1–3 yrs], and cubs of the year [<1 yr]). Annual survival rates are calculated as the proportion of each age or sex class that survived each year.

Black bear survival and reproductive rates vary greatly throughout North America. Black bear populations in the Northeast have relatively high fecundity (i.e., no. female cubs/adult female/year) and low age of first reproduction (i.e., primiparity; Beston 2011; Table 9.1). Of the NEBBTC jurisdictions, Vermont has the greatest fecundity, whereas New Jersey, Pennsylvania, Virginia, and West Virginia have the lowest average age of first reproduction (Table 9.1).

9.1.1 Radiotelemetry

The most common method to estimate bear survival is to capture animals, radiocollar them, and monitor their signals to determine if, when, and why the animal died (Figure 9.2). Estimating survival with radiotelemetry data does not require that locations be obtained, but the status (dead, alive, unknown) must be monitored regularly and frequently, and preferably over a number of years, to estimate annual variation. Adult females are often targeted in telemetry survival studies because population growth rates are most sensitive to that population parameter (Beston 2011). Cub and yearling survival are other population parameters of interest to managers, therefore expandable radio collars that allow for substantial body growth have been developed (Vashon et al. 2003) for cubs and yearlings. Whichever age classes are monitored, it is important to

determine sample sizes needed to provide a robust estimate that will meet study objectives. For

example, if an agency desires to detect a 5% decrease in annual adult female survival, it is

important to know how many radio-collared animals would be required to meet that objective.

In general, precision of survival estimates is high even when sample sizes are moderate (e.g., 20-

30 females/year), although large samples may be required to isolate sampling variance from total

variance in parameter estimates (Harris et al. 2011, Mace et al. 2011). Although radio collars

provide valuable information on bear ecology, we note that this information represents only a

snapshot of an individual's activities and only a small subset of the total population.
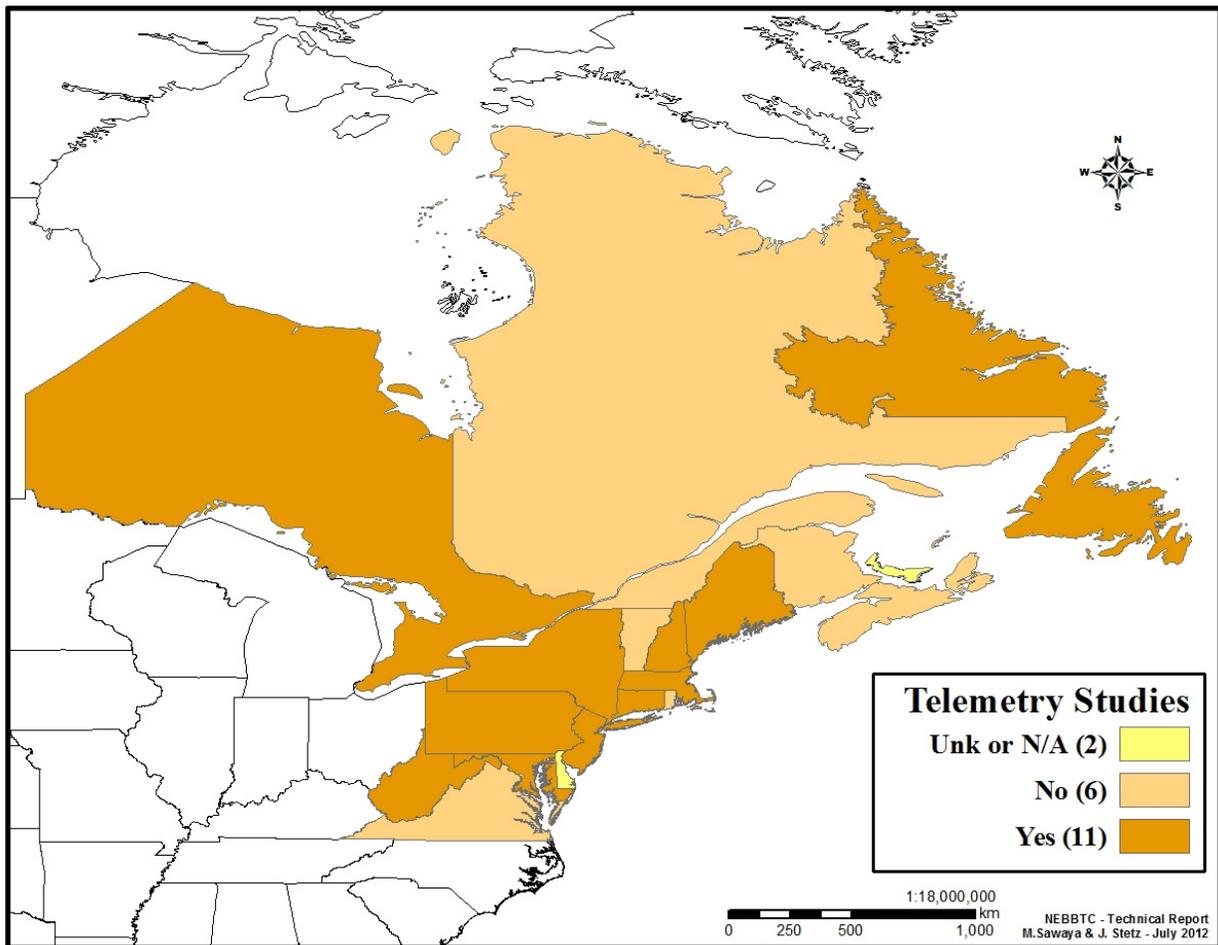
Figure 9.2: *NEBBTC jurisdictions using radiotelemetry methods to monitor black bear populations in 2011 (Noyce 2012).*

Modern VHF radio collars often have pre-programmed inactivity sensors and emit

uniquely pulsed mortality signals when animals have died (Figure 9.3). Researchers can then

quickly find animals that have died, increasing the probability that cause of death can be

determined, which is usually not possible with mark-recapture methods. Radiotelemetry

methods have greatly improved in recent years because of the integration of Global Positioning

System (GPS) technology, satellite data transfer capabilities, smaller and lighter collar designs,

and increased battery life. GPS collars have revolutionized the study of wildlife with the sheer

volume of data that may be stored on board the collar, sent to handheld receivers, or even sent

Figure 9.3: *Black bear with GPS collar.*
Photo credit: New York Department of Environmental Conservation

directly to satellites and emailed to researchers. An added advantage of radiotelemetry methods for estimating survival is that other attributes of bear ecology (e.g., movements, habitat use) can be examined as well.

Intuitively, it would seem that annual survival would be easy to calculate: simply divide the number of living animals after 1 year by the number originally collared. Unfortunately, that simple binomial calculation is accurate only if all the animals are collared at the same time and every animal is located on every occasion. However, if an animal is captured and collared halfway through the study period, for example, the survival estimate for the population will be biased high because that animal has already survived half the sampling season. Animals that did not survive until that time do not have the opportunity to be captured and collared. Procedures were developed to accommodate different starting dates for survival data (staggered entry design), which base survival rates on short intervals of time (e.g., 1 week) and whereby the number of surviving animals is divided by the number at risk, excluding animals whose signal was not located during that interval (Kaplan and Meier 1958, Pollock et al. 1990). The product of those weekly sampling intervals over 52 weeks produces an unbiased estimate of the annual survival rate. The method, in effect, estimates the time of death as the mid-point of the sampling interval. Thus, it is important in telemetry-based survival studies that the animals are located frequently and regularly. The Kaplan-Meier method has been implemented in Program MARK (known-fate models) to estimate survival, thus enabling users to use information-theoretic methods for model selection. If telemetry data are not collected regularly, it may still be possible to obtain a reliable estimate of survival using nest survival models, also in Program MARK. Such data are sometimes referred to as ragged entry data.

In addition to monitoring adult survival, cub and yearling survival can be monitored by approaching and visually observing radiocollared adult females with cubs throughout the non-denning period. Oftentimes, cubs can be treed and counted while the female remains nearby. Litter and cub survival can likewise be estimated using the known fate or nest survival methods.

9.1.2 Mark-recapture

Black bear survival can also be reliably estimated with mark-recapture methods. With this method, animals are captured and marked (either traditionally or using genetic sampling) and their recaptures are monitored over time (usually years). To illustrate, consider an animal that was captured during year 1 and 2, not captured in year 3, and captured again in year 4. An animal being captured is the result of the product of 2 probabilities: the probability that it was captured (p) and the probability that it was alive and still present on the study area (apparent survival, φ). The difficulty lies in not knowing whether animals that are not captured are successful at avoiding traps ($1 - p$) or dead ($1 - \varphi$). These 2 confounded terms can be separately estimated based on individuals that are trapped, not captured in subsequent year(s), and finally recaptured later in the study. For those individuals, the survival rate during the period it was not captured is known ($\varphi = 1$, as evidenced by its capture during a following year) so the odds of not being captured is solely due to the capture probability ($1 - p$). Thus, p can be estimated for that time period and the probability of survival for the animals that are never captured again can be estimated (φ is the proportion of marked animals not captured divided by p). A number of estimators using this same general premise have been developed, of which the Cormack-Jolly-Seber (CJS) method (Cormack 1964, Jolly 1965, Seber 1965) is commonly used because it estimates only 2 parameters, p and φ. Other methods such as Jolly-Seber (Jolly 1965, Seber

1965, Pradel 1996) or robust design (Pollock 1982, Kendall et al.1995) are more general because other population parameters can be estimated (e.g., abundance, population growth). The robust design model (Pollock 1982, Kendall et al. 1995) combines CJS and closed population models by sampling multiple times within each year over the course of multiple years. The within-year ("secondary") occasions allow estimation of detection probabilities and abundance, whereas across-year ("primary") occasions allow estimation of survival, immigration, and temporary emigration from the study area. These models, too, can accommodate covariates and can be extended to multi-state data types (Brownie et al. 1993) to estimate transition probabilities between different states, for example, between breeder and non-breeder status. One advantage of estimating survival with mark-recapture methods is that survival estimation is not as prone to capture biases as other parameters (e.g., N).

Finally, there may be potential to use band recovery methods, which are commonly used to estimate survival rates in birds, but have yet to be applied to bears. Brownie et al. (1985) developed a method whereby animals are tagged each year for a successive number of years and tags are recovered when those animals are found dead. The advantage of that technique is that parameter estimates are not sensitive to capture biases (particularly capture heterogeneity) in the marking process (Nichols et al. 1982, Pollock and Raveling 1982). In addition to survival, the method also provides an estimate of recovery rate, the proportion of the marked animals that are killed and retrieved by hunters and then identified as a marked animal (Mace et al. 1994). For example, if managers set up barbed-wire hair traps and obtained 400 different genotypes from 500 bear hair samples prior to each hunting season (assuming a genotyping success rate of 0.8), assuming a harvest rate of 0.1, and assuming an annual hunter kill of 400 bears, all of which were genotyped, a 10% annual difference in annual survival could be detected over a 10-year

period. If marked animals do not emigrate from the area where the samples are recovered (i.e., hunted areas), the method returns true estimates of survival (S) rather than apparent survival (φ) which included emigration. However, if a large proportion of the marked population emigrates outside the areas open for hunting, the estimate of S will be biased low. Overall, data would have to be collected over a longer period of time and at greater expense to detect a 10% difference in survival compared with some other options discussed previously. The major advantage is that relatively few sample sites would have to be established and their spatial locations would be less strict than for estimating abundance because recapture rates are not being estimated, resulting in savings in personnel time required to obtain samples.

9.2 REPRODUCTION

Reliable estimates of reproduction are important for predicting population growth and can reflect annual fluctuations in habitat conditions. Measures of female reproductive success for black bears include litter size, cub sex ratio, age of primiparity, and fecundity. Because black bear population growth rates are sensitive to changes in adult female survival and reproduction (Beston 2011), wildlife managers are often interested in obtaining accurate and precise estimates of reproductive parameters. These data are often used in tandem with survival and age structure data to project population growth using matrix or individual-based models (Chapter 10). Primiparity, litter size, and cub sex ratios for black bears are usually estimated by radio-marking and monitoring female bears in den sites. Fecundity and other reproduction parameters can be difficult to estimate because true litter sizes will seldom be known as mortality occurs immediately after (and even prior to) birth. Thus, litter size estimates depend on when the cubs are counted, which can lead to estimation errors. For example, if litter sizes are based on

placental scars and cub survival is based on radio telemetry of cubs beginning at about 2 months of age, cub recruitment will be overestimated because mortality between birth and 2 months is not accounted for. Given that caveat, black bear fecundity can be reliably estimated using a variety of techniques from inspecting reproductive tracts to observation techniques.

9.2.1 Reproductive Tracts

Female black bear reproductive tracts can be examined to count corpora lutea on ovaries and placental scars on the walls of the uterus (Figure 9.4). Corpora lutea indicate the number of eggs that were shed in the mammalian reproductive process each reproductive cycle and placental

Figure 9.4: *Black bear reproductive tract with 2 ovaries and 3 developing embryos in center. Photo credit: West Virginia Department of Inland Fisheries and Wildlife*

scars indicate the number of embryos that were implanted (Stickley 1961, Kordek and Lindzey 1980). Consequently, the average number of corpora lutea is generally greater than the number of placental scars because not all eggs will be implanted. Also, not all placental scars will become successfully birthed fetuses, and that number will typically be greater than cub counts in winter dens. Reproductive tracts can only be obtained from dead bears and are thus dependent on harvest. Although many NEBBTC jurisdictions allow bear harvest, only West Virginia collects reproductive tracts to estimate reproduction (Figure 9.5).
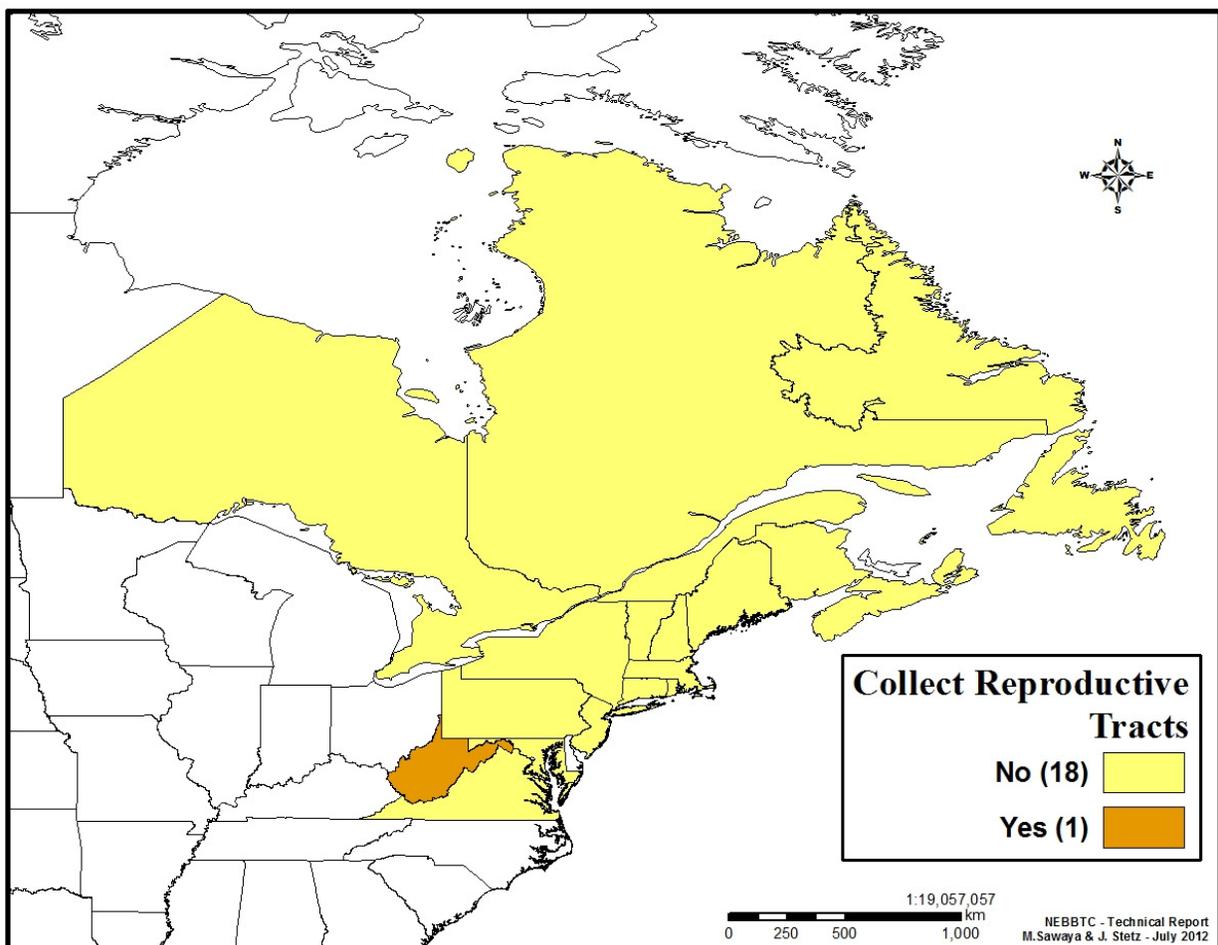
Figure 9.5: *Collection of female black bear reproductive tracts for estimating reproductive parameters by NEBBTC jurisdiction (Noyce 2012).*

9.2.2 Den Visits

Visiting black bear dens in winter can provide data on a number of reproductive parameters. Bear cubs are born in the den and may experience mortality prior to emerging in spring. Therefore, entering dens to count newborns provides a reliable estimate of litter size and sex ratio (Samson and Huot 1995, McDonald and Fuller 2001), unless mortality occurred prior to the den visit, which is difficult to confirm. The proportion of cubs observed the following year in dens as yearlings can be used to estimate cub survival. Conversely, 1-year recruitment can be directly estimated (the number of female cubs recruited into the 1-year-old age class per adult female). Research suggests that remote photography methods could be used in conjunction with den visits to improve accuracy of reproductive and cub survival estimates (Bridges et al. 2004).

9.2.3 Observations

Throughout most of their forested range, black bears generally have low sightability. Thus, observations of non-radiocollared bears for estimating reproductive parameters has limited utility in the Northeast. In open habitats or places where black bears develop strong preferences for foraging locations (e.g., berry patches, garbage bins, agricultural fields), observations may be used for estimating age of first reproduction and fecundity. If a sample of radiocollared females exists and den visits are not feasible or risk cub abandonment, researchers can use telemetry to approach them to tree cubs in the field after den emergence. This technique enables estimation of litter size, cub and yearling survival, age of primiparity, and fecundity. As with known-fate analyses, observations should be frequent to obtain sufficient precision and to reduce bias from undercounting that sometimes occurs using this method.

9.3 SURVIVAL AND REPRODUCTION SIMULATIONS

9.3.1 Survival Simulations

To explore the influence that sample size, duration of study, and population characteristics have

on the precision of survival estimates from radiotelemetry data, we conducted a suite of

simulations in MATLAB using code modified from Harris et al. (2011; modified from Doak et

al. 2005). Briefly, this simulation routine allows the user to vary a large number of parameters

related to sample design (i.e., the number of individuals per age class monitored over a

designated number of years) and population characteristics including cub, yearling, subadult, and

adult survival, and fecundity. The model also uses variances of vital rate estimates to provide a

more accurate depiction of the uncertainty in survival (or reproduction or lambda) estimates as

the number of years of monitoring increases. Parameter estimates can then be viewed in terms of

the tradeoffs between sample size and precision, thereby providing a realistic starting point for

project design. The routine also allows for defining covariance of vital rates, for example cub

and yearling survival, although we followed the suggestion of Harris et al. (2011) and did not

include such effects because of limited evidence of covariance in the literature. Another factor

not considered in our simulations is the removal of sampling variance. Because of typically

small samples, estimating and removing sampling variance is rarely done, resulting in less

precise estimates from known fate models (Harris et al. 2011). Whenever possible, however,

sampling variance should be removed prior to estimating vital rates or lambda.

    We reviewed the literature for estimates of vital rates for black bear populations in the

Northeast, drawing on the summary provided by Beston (2011). We initially considered

combinations of estimates representing either best- or worst-case scenarios (i.e., highest or

lowest vital rates from across studies), with the intent of capturing the extreme situations that managers may encounter with similar analyses. We elected, however, to use more realistic combinations of vital rates from real populations that NEBBTC managers may be familiar with (Table 9.2). As an example population with lower values for the vital rates of interest, we chose a study in east-central Ontario (Kolenosky 1990). For an example of greater vital rates, we combined estimates from 3 studies in Virginia (Table 9.2). In all cases, variance of vital rate estimates were from the same study as the vital rate estimates themselves. In addition to using estimates from different populations, we varied the number of animals and the number of years monitored for each vital rate. For a small study design, we used combinations of 10 or 30 animals monitored per age class for 3, 5, 10, and 20 years. For a large study, we used combinations of 30 or 100 animals monitored per age class for the same range of years. For more details on simulation methods, see Harris et al. (2011), Doak et al. (2005), and the annotated MATLAB code (Appendix H).

We assessed precision of estimates as their CV (Figure 9.6). For all age classes, survival estimates were least precise (largest CV) for the lower vital rate population (i.e., Ontario) with the smallest number of individuals ($n = 10$) monitored. The lowest CVs were obtained with the largest sample size ($n = 100$) for all age classes, although the influence of low or high vital rates was less consistent. Specifically, adult and subadult survival were most precise for the "low" scenario, whereas cub and yearling survival were most precise for the "high" scenario. These results likely reflect the magnitude of variance in the vital rate estimates used (Table 9.2), as cub and yearling estimates were more precise for the "high" population, and subadult and adult were more precise for the "low" population.

Table 9.2. Estimates for black bear vital rates and their variances used in demographic analysis simulations.

	Vital rate estimates					Variance estimates				
Rates	Cub	Yrlg	Subad	Adult	Fecund	Cub	Yrlg	Subad	Adult	Fecund
	S^a	S	S	S		S	S	S	S	
Low[b]	0.53	0.76	0.87	0.84	0.4570	0.00778	0.00793	0.00213	0.00063	0.00049
High	0.73[c]	0.87[d]	0.93[c]	0.93[e]	0.6875[c]	0.00493	0.00435	0.00360	0.00271	0.02000

[a] Survival estimate.
[b] Kolenosky (1990); 241 adult females monitored.
[c] Ryan (1997); 34 bears monitored (6M, 28F).
[d] Lee and Vaughn (2005); 54 yearling bears monitored (34M:20F).
[e] Carney (1985).

We obtained rapid gains in precision by increasing the number of years of monitoring, although this gain in precision slowed between 10 and 20 years (Figure 9.6). In fact, CVs for all scenarios within each age class were within ~5% after 20 years monitoring. As a general rule, going from 3 to 10 years of sampling resulted in the same improvement in precision of estimate as increasing the number of individuals monitored from 10 to 100.
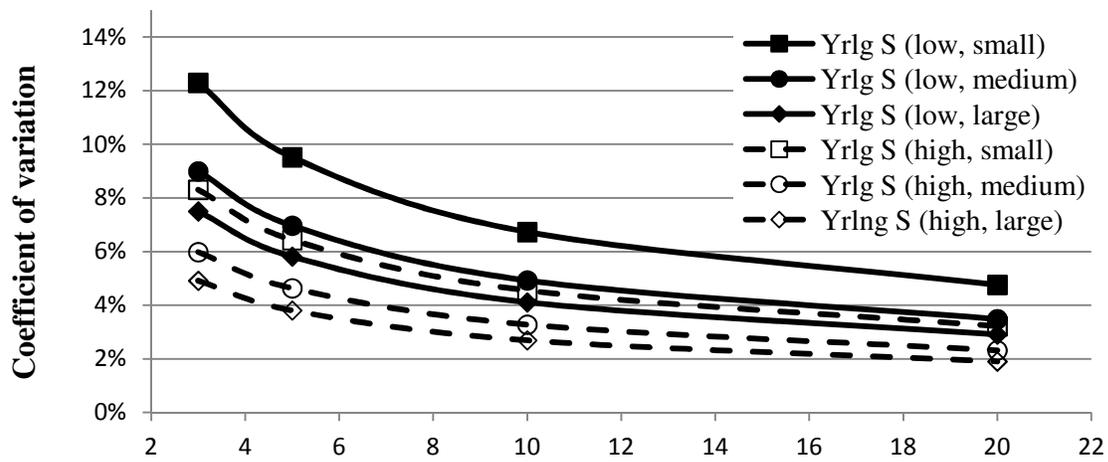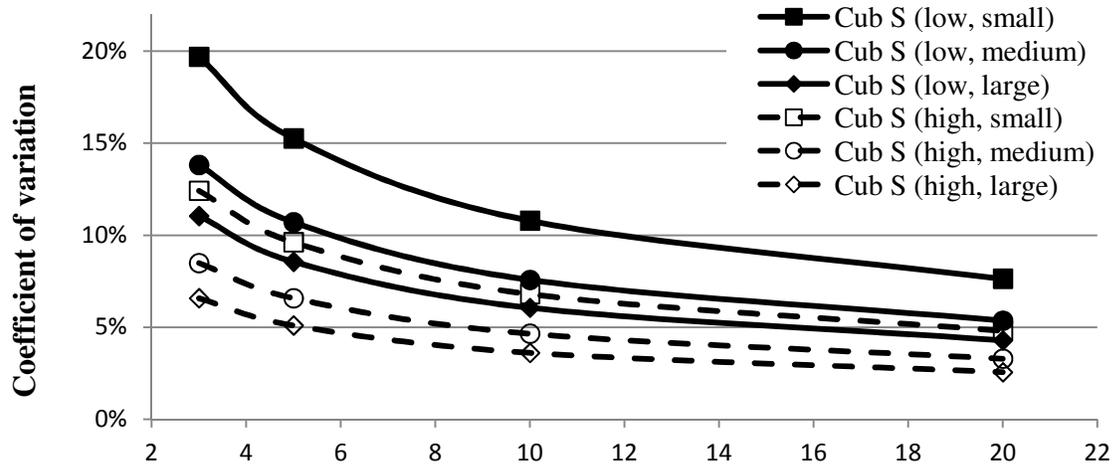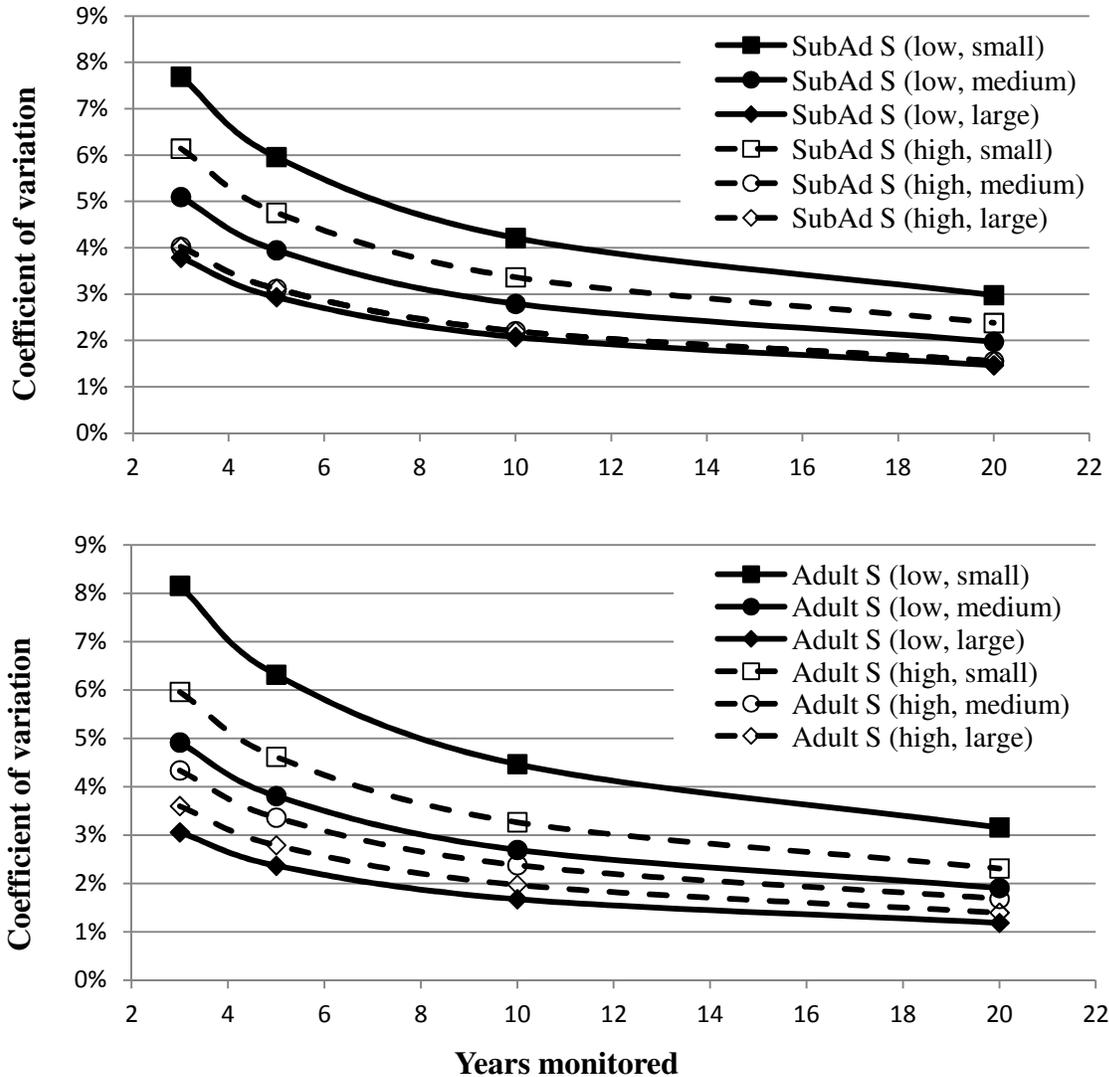
Figure 9.6: *Age class-specific estimates of precision for survival estimates from radiotelemetry data as a function of sample size and number of years monitored (3, 5, 10, or 20). Solid symbols and lines reflect "low" vital rates from Ontario (Kolenosky 1990). Open symbols with dashed lines represent "high", composite vital rates for Virginia populations (see Table 9.2). Small, medium, and large scenarios refer to 10, 30, or 100 individuals monitored, respectively, for the vital rate in question. Note the scale of y-axes may differ.*

9.3.2 Fecundity Simulations

In addition to providing estimates of precision for survival, the simulations described in section 9.3.1 allowed us to evaluate the role of sample size and variance on precision of fecundity estimates. We again used the MATLAB simulation routine of Harris et al. (2011) with the vital rate estimates presented in Table 9.2, varying the sample size and number of years of monitoring. For more details on simulation methods, see Harris et al. (2011), Doak et al. (2005), and the annotated MATLAB code (Appendix H).

Similar to the adult survival analysis, the differences between the fecundity of these populations was quite pronounced, with a considerably lower value in the Ontario population, yet with a far more precise estimate, than the composite Virginia scenario. As with adult survival, the greatest gains in precision were obtained from increasing the sample size of radio collars (Figure 9.7). We observed the same general improvement in precision as the duration of monitoring increased, with the most rapid gains occurring in early years. Again, going from 3 to 10 years of monitoring was roughly equivalent to increasing sample size from 10 to 30 or 30 to 100 (Figure 9.7).
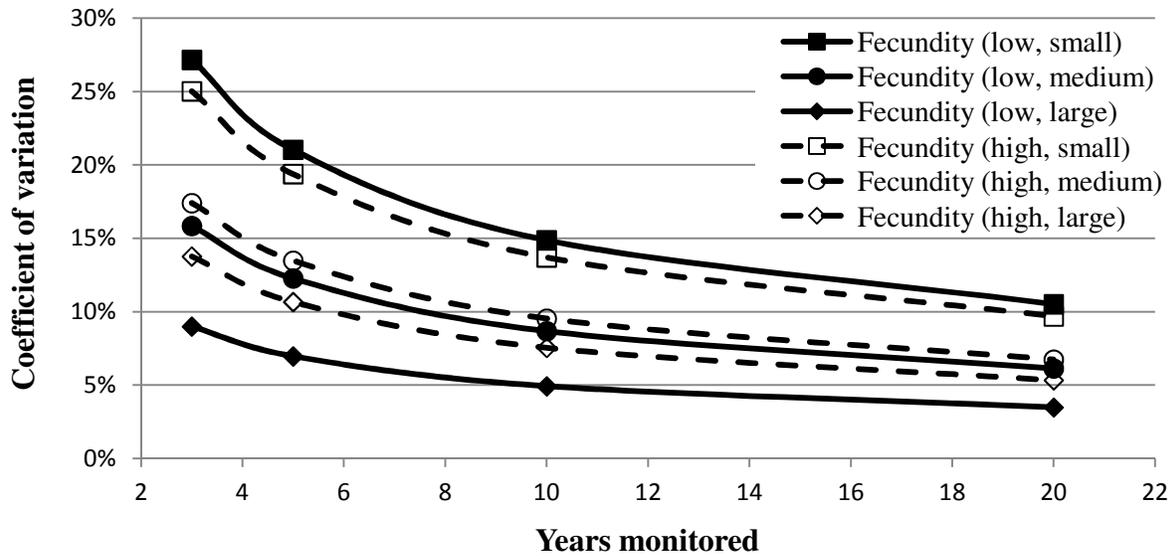
Figure 9.7: *Age class-specific estimates of precision for fecundity estimates from radiotelemetry data as a function of sample size and number of years monitored (3, 5, 10, or 20). Solid symbols and lines reflect "low" vital rates from Ontario (Kolenosky 1990); open symbols with dashed lines represent "high", composite vital rates for Virginia populations (see citations in Table 9.2). Small, medium, and large scenarios refer to 10, 30, or 100 individuals monitored, respectively, for the vital rate in question.*

9.4 CONCLUSIONS: SURVIVAL AND REPRODUCTION

- Among the most effective ways to accurately and precisely estimate survival for black bears is to radiocollar a sufficiently large sample of the population and track their fates through time.

- This method can provide robust estimates of survival and information on causes of mortality, the latter being important information to adjust management of populations with negative trajectories.

- Mark-recapture methods can be used to estimate apparent survival, but cannot account for loss through emigration.

- Band recovery methods developed for birds have potential to estimate annual survival of bears for populations that are harvested.

- The most effective way to accurately and precisely estimate reproductive parameters for black bears is to radiocollar a representative sample of the female population and monitor their reproductive output over time. An added benefit is that radiotelemetry can be used to also estimate adult survival and other variables (e.g., home-range size, habitat use).

- Reproductive success can be measured by monitoring radio-marked animals during den visits and through direct observations.

Chapter 10. Population Growth

10.1 POPULATION GROWTH

Population growth refers to changes in abundance over time, including increases, decreases, or no change, reflecting the cumulative influences of birth, death, immigration, and emigration on the demography of a population (Pollock et al. 1990). Population growth is the most important parameter to many black bear managers in NEBBTC jurisdictions (Chapter 7; Appendix B), yet it is difficult to estimate for many bear populations (Harris et al. 2011). State and provincial estimates of bear abundance over time often reveal growth patterns that differ from true trends (Garshelis and Hristienko 2006). Abundance indices can potentially be useful for measuring

Figure 10.1: Female black bear with 5 yearling cubs.
Photo Credit: T. Sears

population growth but, as discussed previously (Chapter 8), no reliable index of relative

abundance exists for black bears except for monitoring gross changes in bear populations over

extended periods of time. Therefore, we will not discuss indices further here.

In its simplest form, population growth is calculated by dividing population size during a

particular period of time by population size of the previous time period, $\lambda = N_t/N_{t-1}$ (i.e., realized

population growth). Consequently, abundance estimators have been used to estimate λ, which

offers numerous advantages over point estimates of abundance. Growth rate estimation is

generally more robust to capture heterogeneity biases than abundance (Schwarz 2001), although

Hines and Nichols (2002) found that behavioral biases were possible, particularly with short-

term data sets.

10.1.1 Demographic Analyses

Growth rate estimates from vital rate statistics in matrix or individual-based projection models

require robust estimates of population size, age- and sex-specific survival and fecundity, sex

ratios, age structure data, and age of primiparity (Clark et al. 2006, Clark et al. 2010).

Sometimes called life-table methods or demographic analyses (Harris et al. 2011), they are data

intensive, often requiring both mark-recapture and radiotelemetry techniques. If predictions of

population growth to evaluate persistence are made (i.e., projected population growth), estimates

of process variance should be included. The advantage of these methods is that the estimates can

be accurate and sensitivity analysis and population viability analyses can be used to answer a

wide array of ecological questions. Also, harvest can easily be accommodated in population

projections to evaluate different management alternatives. Like any projection, assumptions are

usually based on constant environmental conditions and variance, so these models should be

updated often. Uncertainty increases dramatically the longer the time period of the projection (Caswell 2001).

Matrix population models constitute one approach to estimating population growth. Sometimes called Leslie matrices, this method can be applied to stable or unstable age distribution data and can be used deterministically or stochastically to project population growth and variance. A number of matrix-based software tools (e.g., Poptools) have been developed that enable users to perform sensitivity and elasticity analyses. Other individual-based models have been used to model bear population growth (e.g., Riskman, GAPPS, R package demoniche). The alternate-year breeding in black bears is more easily accommodated in these individual-based models and thus have widespread appeal. In general, these tools produce reliable results given adequate estimates of vital rates.

Precision of life table-based estimates of population growth are strongly correlated with the precision of age-specific vital rate estimates. Because black bear population growth is most sensitive to changes in adult and subadult female survival and fecundity, precise estimation of those parameters is particularly important to obtain reliable estimates of projected population growth (Freedman et al. 2003, Mitchell et al. 2009). Generally, the precision and accuracy of vital rate estimates increases with the number of individuals monitored and the duration of monitoring (Harris et al. 2011). Even with long-term monitoring, many life table analyses estimate λ imprecisely with 95% confidence intervals (CIs) that often overlap 1.0, indicating the possibility of a stable, declining, or increasing population. Additionally, population age and stage distributions should be adjusted for misclassification errors to obtain more precise and less biased estimates (Conn and Diefenbach 2007). Perhaps the most difficult parameter required for many projection models is an estimate of the standing age distribution. Unless a population is

sampled almost completely, some age and sex classes are usually more susceptible to sampling, which can lead to bias. It is possible to project population growth assuming a stable age distribution, but this is probably rare for black bears because of their long lifespan and annual fluctuations in abundance of food resources.

Integrated population models can be used to estimate population growth for black bears by integrating multiple types of data. One advantage of integrated models is that they can synthesize various relevant data into a single analysis. Fieberg et al. (2010) recently used this approach to synthesize age-at-harvest data, periodic large-scale estimates of abundance, and measured covariates thought to affect black bear harvest rates. The authors concluded that integrated population models were unbiased and hold great promise for black bear population monitoring, but they recognized the assumption of age distribution being representative of the greater population may often be unreasonable.

10.1.2 Population Reconstruction

Population reconstruction has been used by NEBBTC managers to monitor bear population growth and estimate abundance, recruitment, survival, and harvest rate (Figure 10.2). The technique has been used in fisheries management for decades (Fry 1949, Pope 1972) but was popularized for wildlife management by Downing (1980), who estimated minimum population size and trend for white-tailed deer (*Odocoileus virginianus*). The technique is based on total harvest by year and a sample of ages of harvested animals to back-calculate the age distribution at the time the oldest animals were born, thus estimating minimum population size. The population size estimate is a minimum because deaths from causes other than harvest are not included. The greatest advantage of Downing reconstruction is that it requires only the total

annual harvest and a subsample of annual harvest with age data (Downing 1980). Thus, no additional costs are incurred other than cementum annuli analysis to age a subsample of the harvested population (Chapter 8). Davis et al. (2007) found that such reconstruction techniques performed best when harvest rates were high and natural mortality was low, as may be the case with bears in many NEBBTC jurisdictions. However, those authors also found that the estimates of λ could be negatively biased if harvest rates trended even moderately upward (0.01/yr) or were highly variable. This could be the case in many NEBBTC jurisdictions because bear harvests are greatly affected by mast availability and other factors, and many changes in harvest regulations have occurred in recent years.

Population reconstruction relies on the assumption that harvest and natural mortality rates do not change over time. Additionally, this method is based on the assumption of a stable age distribution and a constant harvest reporting rate. Population reconstruction relies on a number of other assumptions that are difficult to meet in many wildlife studies. Williams et al. (2002) provided a comprehensive critique of population reconstruction and identified 3 main flaws of the method: 1) survival estimates are inferred from a population model; 2) biases in the reconstruction will manifest themselves in the estimates; and 3) even if assumptions are met, estimates of sampling variation will not include the sampling error of the harvest. Also, Williams et al. (2002) suggested that population reconstruction based on age-at-harvest data alone is theoretically flawed because the method does not account for non-hunting mortality and the age and sex distribution of the harvest is probably not reflective of the sampled population. They concluded that population reconstruction should not be considered if more reliable estimation methods are available (Williams et al. 2002).

Davis et al. (2007) reported that population reconstruction held promise for black bear management but Gove et al. (2002) indicated that age-at-harvest data are insufficient to estimate the necessary demographic parameters required for population reconstruction and the method produces no estimate of sampling error. To address these and other problems, Gove et al. (2002) introduced maximum likelihood methods to estimate harvest rates and population size given auxiliary data on survival from radiotelemetry and hunter reporting rates from a telephone survey. One of the advantages of using maximum likelihood techniques for reconstruction is that various population assumptions (e.g., constant harvest, increasing harvest) can be tested

Figure 10.2: Use of population (i.e., age) reconstruction from harvest data to estimate black bear population growth rates; information compiled by (Noyce 2012).

using information-theoretic methods (Burnham and Anderson 1998) and statistical uncertainty can be measured. Recently, attention has been placed on model evaluation for statistical population reconstruction through the use of residual analyses, sensitivity analyses, and model predictions (reviewed in Skalski et al. 2012*a*). Model evaluation differs from model selection (i.e., based on AIC values) in that goodness-of-fit measures are used to identify an appropriate set of models, whereas model selection is simply the relative support for a set of models that may, in theory, all be inappropriate (Johnson and Omland 2004). Thus, model evaluation precedes model selection and averaging, but does not replace it. One suggested approach is to delete one or more consecutive year's data from the beginning or end of the series to determine model stability. If results change substantially, it is likely that inadequate data are being used. Estimability of such population reconstruction models requires auxiliary data, as even the simplest of models is over-parameterized when only age-at-harvest data are used (Skalski et al. 2012*a*). These auxiliary data can include catch per unit or harvest effort, index data, mark-recapture, or radiotelemetry data (Skalski et al. 2007, 2012a). Such combinations of data would permit an integrated analysis of data collected by many NEBBTC jurisdictions, be more statistically rigorous, and provide estimates of precision for all parameters. Further, statistical population reconstruction models appear robust to pooled age classes (i.e., when actual ages are not known), providing greater flexibility of use (Skalski et al. 2012*b*). Instead of maximum likelihood, Conn et al. (2008) used Bayesian analysis to estimate similar population parameters for black bears, again by coupling age-at-harvest data with mark-recapture data. However, the computational complexities of such an approach currently provide little utility to most wildlife managers. Clearly, more research and software development is needed on this promising technique.

<u>10.1.3 Mark-recapture</u>

Although mark-recapture methods are commonly used to estimate population size, population growth is less difficult to estimate. For example, if an estimate of abundance (*N*) is biased low because of undetected capture heterogeneity it is reasonable to assume that all subsequent estimates of *N* might be similarly biased. Assuming that, the relationship among those estimates over time should be relatively constant. Consequently, black bear population growth can be estimated using linear regression to measure the slope between a series of annual abundance estimates. The reliability of this method, of course, is dependent on the accuracy and precision of abundance estimates. Precision can be low because the regression is based on annual abundance estimates rather than the capture data that comprise those estimates.

Based on that notion, Pradel (1996) and Schwarz and Arnason (1996) developed maximum likelihood methods for estimating population growth (λ) directly from mark-recapture data without the need for estimating *N*. As discussed previously, that method also provides estimates of apparent survival (φ) and apparent fecundity (*f*), and has been added as a routine in Program MARK (White and Burnham 1999). Subsequent research has shown that the Pradel method is robust to moderate capture heterogeneity, the most difficult of all capture biases to estimate (Schwarz 2001, Hines and Nichols 2002, Marescot et al. 2011). Clark and Eastridge (2006) estimated population growth in a small population of black bears in Arkansas and adequate precision was achieved using live-capture data collected over a period of several years. Those estimates were consistent with estimates from hair-sampling and population modeling based on radiotelemetry data. The Pradel model has been used to investigate the effect of salmon availability on grizzly bear population growth in British Columbia, Canada. (Boulanger et al. 2004*c*). More recently, bear rub tree detection data were successfully used with a Pradel

model in Banff National Park, Alberta, to estimate λ for grizzly bears (Sawaya et al. 2012).

Thus, such methods not only evaluate changes in the population over time, but enable

researchers to evaluate the proximate causes of the population trend (e.g., survival, fecundity).

Pradel models with genetic sampling based on a single annual sampling occasion,

repeated for several years was deemed efficacious for monitoring the range-wide population of

bears in Tennessee (J. Clark and F. T. van Manen, U.S. Geological Survey, unpublished data).

The number of sites would not have to be as high as required to estimate abundance with closed

or robust design methods. Given the robustness of the estimators, capture probabilities could be

lower compared with estimates of population size, which would make the technique more cost

effective. However, there are some logistical constraints. Sampling sites should be

systematically spaced so that all bears have a reasonable opportunity for detection and locations

should be changed annually to reduce behavioral bias.

10.2 POPULATION GROWTH SIMULATIONS

10.2.1 Population Projection Growth Rate Simulations

We again used a MATLAB simulation routine modified from Harris et al. (2011) to explore the influence of study design, vital rate values, and vital rate variances on estimates of population growth rate with demographic anlayses. Managers are typically most interested in the lower bound of λ estimates, so we assessed precision of λ estimates based on the width of 90% confidence intervals, which produces a 5% probability that the estimate falsely exceeds true λ, assuming accurate estimates are used with a reasonable model (Harris et al. 2011).

For all scenarios, precision improved rapidly as the number of years increased, particularly in early years, with relatively small gains between 10 and 20 years of monitoring (Figure 10.3). Consistent with Harris et al. (2011), we found the greatest improvements in precision for all scenarios by increasing the number of monitored litters (Appendix H). When considering single vital rates, however, the greatest improvement in precision for the lower vital rate scenarios (i.e., Ontario) was gained through monitoring more adults, whereas the higher vital rates scenarios (i.e., Virginia) showed the greatest improvement by monitoring fecundity more intensively. This latter finding may seem somewhat contradictory to expectations given the known value of adult female survival (Garshelis et al. 2005, Beston 2011). However, variance of adult survival estimates is typically low so gains in precision can be more easily accomplished with other vital rate estimates, as others have found (e.g., Mitchell et al. 2009).
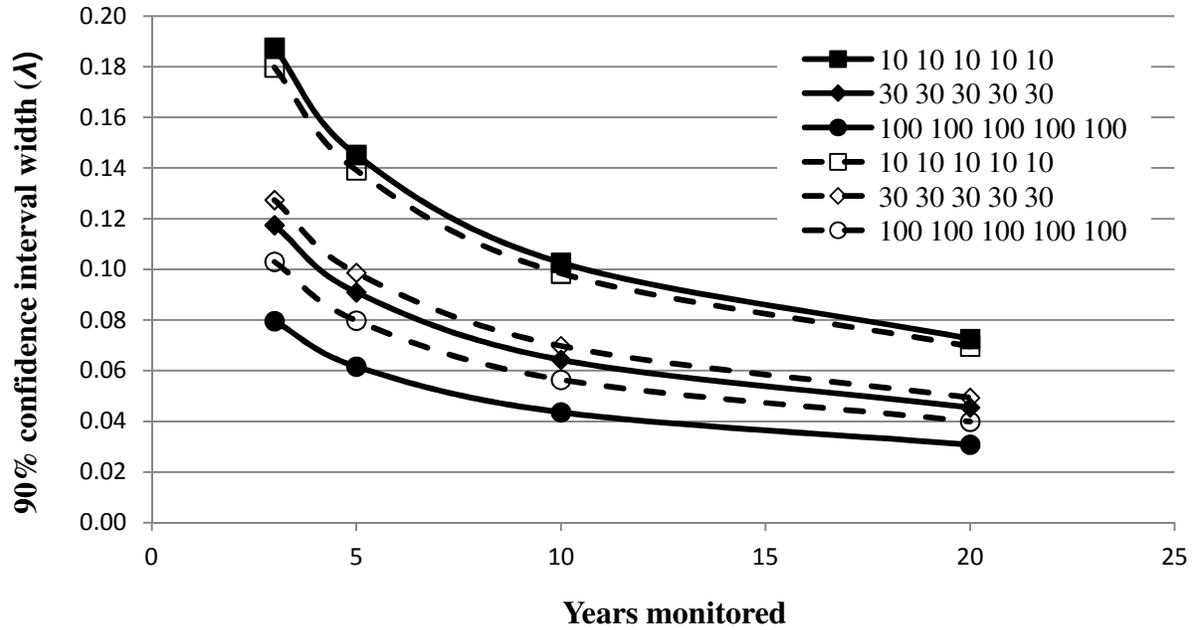
Figure 10.3: *Width of 90% confidence intervals on lambda estimates for population growth rate simulations as a function of years monitored, sample size, and vital rates. Numbers in the legend refer to age class-specific sample sizes (cub, yearling, subadult, and adult survival, and fecundity, respectively). Solid symbols and lines reflect lower vital rates from Ontario (Kolenosky 1990), whereas open symbols with dashed lines represent higher, composite vital rates of Virginia populations (see references in Table 9.1).*

10.2.2 Open Population Mark-recapture Simulations

As we discussed in Section 8.3, simulations are a powerful tool for study design when data are available on population and sampling parameters and when managers have targets of accuracy and precision. We therefore conducted a suite of open population simulations with the Pradel (1996) model in program MARK to provide reference points for estimating population growth rate in black bear populations in the Northeast. Again, we reviewed the literature and extracted estimates of population growth (λ), apparent survival (φ), and detection probabilities (p) from DNA-based mark-recapture studies (Table 10.1). We simulated populations of 100 or 500 bears, that were either monotonically increasing ($\lambda = 1.05$) or decreasing ($\lambda = 0.95$), for 5 or 10 years. We ran 250 replicates per scenario, and again assessed model performance with average PRB, CV, and CIC.

Results of our simulations were consistent with those from other analyses using the Pradel model based on DNA sampling (e.g., Stetz et al. 2010), with all considered scenarios producing unbiased (i.e., PRB < 1%) and precise (i.e., CV < 5%) estimates of population growth rate. Simulations based on larger populations and longer studies performed best. However, despite being unbiased, those scenarios had the counterintuitive result that high precision and extremely small confidence intervals led to low CIC values (Figure 10.4). Population abundance did not seem to affect CIC, but again reflecting the decreasing CV of estimates, longer studies tended to have poorer CIC than shorter studies. Greater detection probabilities did not effectively improve estimator performance: even the near doubling of female detections (from $p = 0.38$ to $p = 0.7$) resulted only in a 1.4% improvement in CV for the sparsest scenario (i.e., scenario 2, smaller and declining population with lower survival). Our results suggest that even with relatively small populations and moderate detection probabilities, robust, sex-specific

Table 10.1. Population and sampling parameters used with Pradel (1996) open population model simulations in program MARK to estimate population growth rate of black bear populations.

Scenario	N	λ	φ_M	φ_F	p_M	p_F	No. years
1	100 or 500	0.95	0.93	0.87	0.4	0.38	5 or 10
2	100 or 500	0.95	0.85	0.80	0.4	0.38	5 or 10
3	100 or 500	1.05	0.93	0.87	0.4	0.38	5 or 10
4	100 or 500	1.05	0.85	0.80	0.4	0.38	5 or 10
5	100 or 500	0.95	0.93	0.87	0.7	0.70	5 or 10
6	100 or 500	0.95	0.85	0.80	0.7	0.70	5 or 10
7	100 or 500	1.05	0.93	0.87	0.7	0.70	5 or 10
8	100 or 500	1.05	0.85	0.80	0.7	0.70	5 or 10

estimates may be obtained within 5 years of sampling. As an example, these population and sampling parameters are very similar to those of Coster et al. (2011) who conducted a study on a small area (223 km^2) with 51 grid cells of 5.2km^2. We also conducted simulations with reduced capture probabilities. Our results suggest that a 5% change in λ could be detected over a 5-year period given a capture probability of 0.05 (281 hair samples assuming a bear population estimate of ~4,500 (e.g., New Hampshire, Vermont) and a genotyping success rate of 0.8). A capture probability of 0.025 would enable detection of a 10% difference in λ. These are best-case scenarios because constant capture probabilities, survival, and population growth are assumed.

Sample sizes would have to be greater if those parameters vary, for example, by time. Regardless, it seems apparent that robust estimates can be possible with less intensive sampling than is required for estimating abundance.

These simulations were necessarily simplistic, as the data required to parameterize more complex models are not available for many areas. Tradeoffs will exist with field studies, which will almost certainly encounter heterogeneity of capture probabilities, annual variation in λ and φ, and the realities of sampling populations. However, data from field studies allow use of more complex models to accommodate those realities, including robust design or mixture models that make use of covariates and potentially multiple data types, as has been discussed elsewhere (Boulanger et al. 2004c, 2006, 2008; Stetz et al. 2010). Therefore, these rather simple scenarios should be viewed as a starting point for exploring more realistic study designs depending on the specific objectives, population characteristics, and available resources.
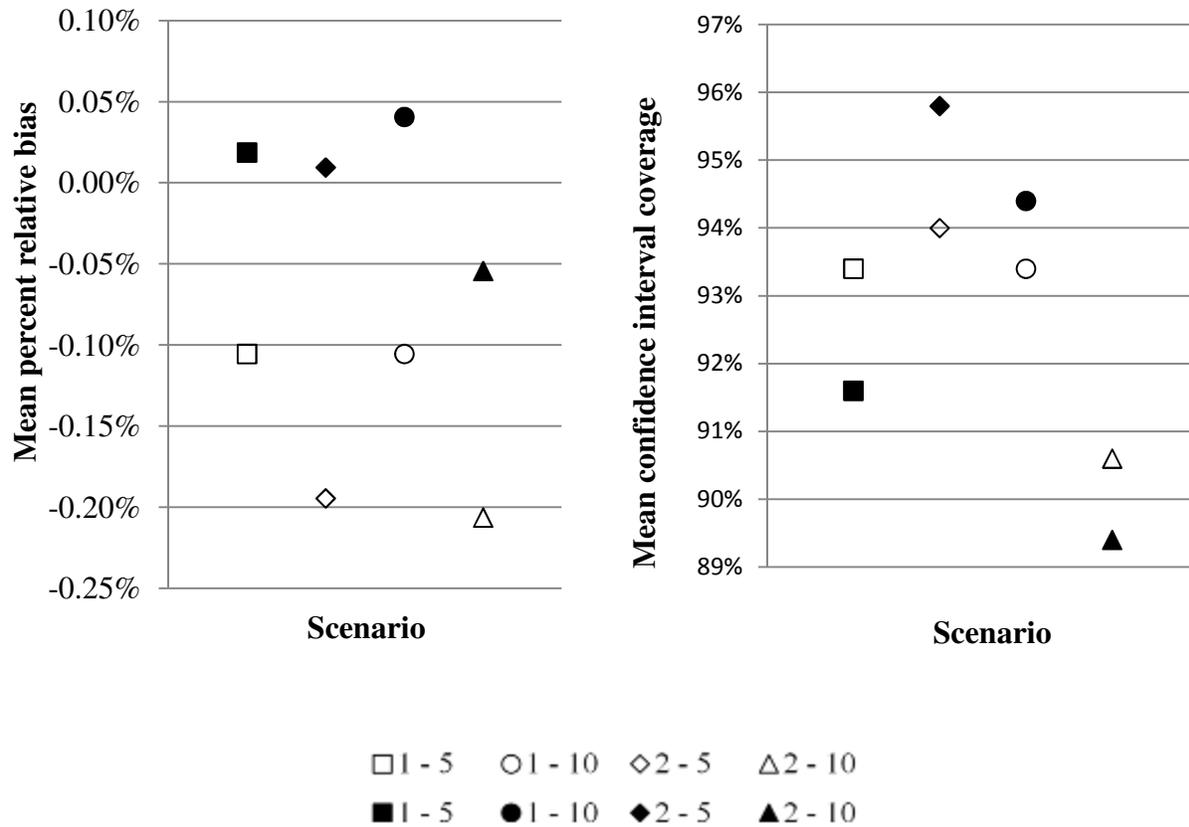
Figure 10.4: *Selected results of population growth rate simulations in program MARK. Scenario numbers are the same as in Table 10.1; e.g., "Scenario 1–5" corresponds to a declining population with high apparent survival and low detection rates with 5 years of monitoring. Open symbols represent populations of N = 100 and closed symbols N = 500.*

10.3 CONCLUSIONS: POPULATION GROWTH

- Population growth reflects the cumulative changes to population abundance due to birth, immigration, death, and emigration and may be one of the most effective measures of population performance for making management decisions.

- The most powerful methods for estimating population growth are those that have sufficient power to detect changes in abundance as a direct result of management and habitat changes, if such changes occurred.

- Population Projection can be reliable but are data intensive. Estimates of the standing age distribution are important but difficult to estimate in most situations. The method requires extensive use of radiotelemetry, but animals can be distributed over a large area. Radiotelemetry provides the ability for estimating other useful population parameters.

- Populations may act as sources for surrounding areas despite not displaying increasing local abundance.

- Interpretation of changes in population growth rates should be done carefully, particularly when monitoring only portions of a larger population.

- Methods to estimate population growth are even more varied than those for estimating abundance, and include mark-recapture, known-fate, and various indices.

- Management objectives, spatial scale, and existing data should all be considered when determining the methods to monitor population growth.

- When using indices to monitor population growth, they must reflect a reliable relationship with the processes in question; otherwise inferences about population changes may be false. For example, poor food years may result in greater movements by

bears, which could result in increased visitation to bait stations despite there being no true relationship with abundance. Because each approach has a suite of assumptions, some more difficult to address than others, use of multiple, independent methods should be considered.

• Traditional radiotelemetry methods may provide more detail regarding cause-specific mortality, but may suffer from issues of geographic closure violation more so than do more recently developed mark-recapture approaches.

• Mark-recapture techniques such as Pradel models can produce precise estimates of population growth in short time periods relative to known-fate methods and are less affected by most capture biases. Howewer, inference about ultimate drivers of growth are limited even with the use of covariates and the ability to parse out componentsof λ (e.g., f and φ).

Section V. Conclusions

Chapter 11. Monitoring Options

11.1 CHOOSING MONITORING TECHNIQUE

As the preceding chapters have illustrated, there is much variability in bear and human demographics, environmental conditions, and social perspectives among NEBBTC jurisdictions, which makes it difficult to establish region-wide standards for black bear research and management. This diversity of people, places, and bears makes bear management in the Northeast particularly complex. Black bear populations are increasing throughout the Northeast whereas wildlife conservation revenue from hunting taxes has diminished (Chapter 2). Shrinking revenues combined with increasing challenges and expectations make it more important than ever for managers to identify and apply the most appropriate, reliable, and cost-effective population monitoring techniques to meet management objectives. Over 60% of black bear managers in NEBBTC jurisdictions believe that the precision of population parameter estimates is important to consider when managing black bear populations. Given the wealth of options currently available to wildlife managers for estimating black bear population parameters, monitoring decisions are complex. Generally, the accuracy and precision of population parameter estimates is driven by cost and effort, a tradeoff that managers are forced to confront.

Human population in the Northeast is concentrated near New York and Toronto, the largest metropolitan areas in the U.S. and Canada, respectively. Much of the region is enveloped by urban development stretching 800 km (500 mile) from Boston, Massachusetts to Washington, D.C. termed the Megalopolis or "very big city" (Gottman 1961). This high human density has a pronounced effect on black bear distribution in areas of the Northeast (Figure 11.1). High human and bear densities and increasing populations challenge NEBBTC managers trying to

reduce human-bear conflicts. Bear-human conflicts have been on the rise for decades in the

Northeast and managers should be prepared for conflicts to continue to increase in response to

changing landscapes. Although the ecological plasticity of American black bears allows them to

adapt to moderate changes in habitat and availability of food resources, human land use patterns

in the Northeast are changing rapidly and the effects of climate change (Frumhoff et al. 2007)

may drastically alter the distribution of bear foods. Consequently, NEBBTC managers urgently

need effective tools to monitor black bear populations. Our primary objective was to evaluate
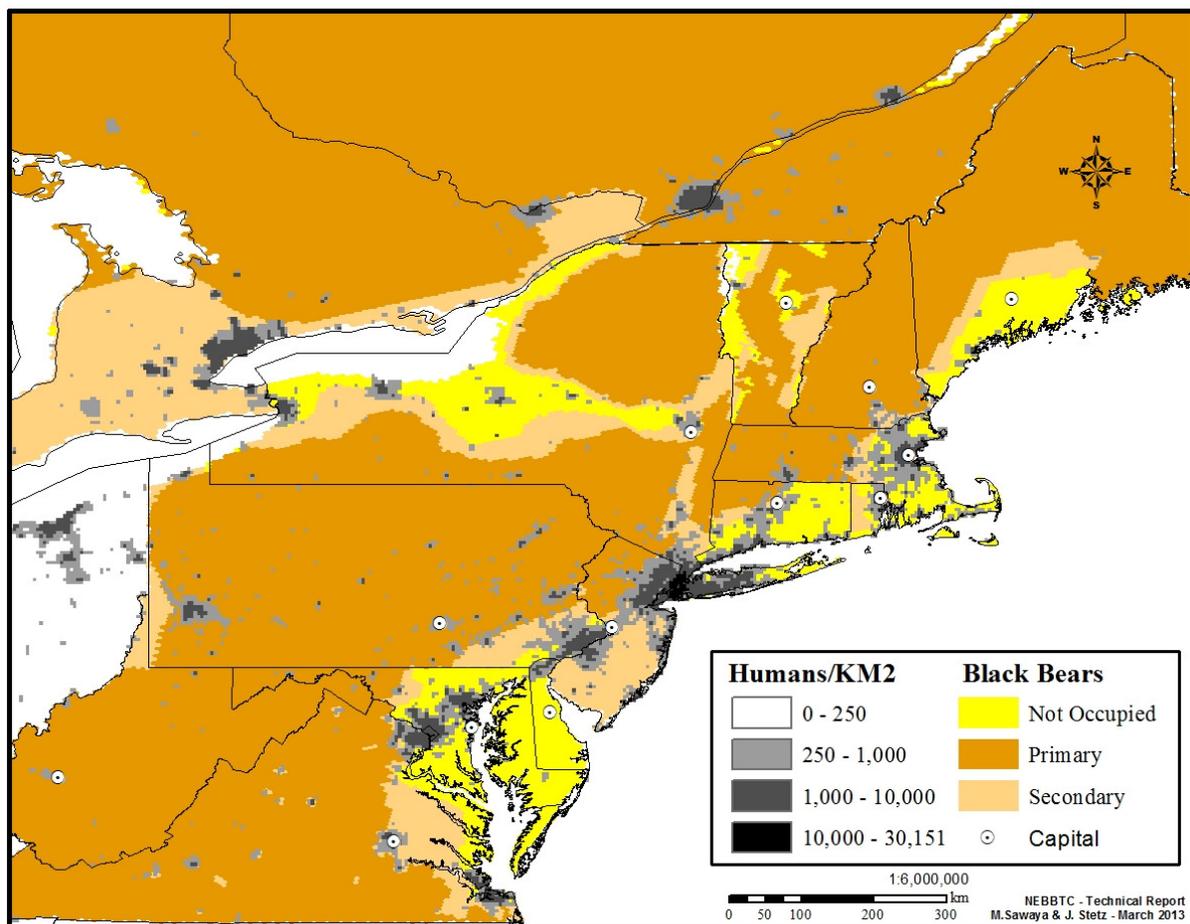
Figure 11.1: *Northeast black bear distribution in 2011 from Schieck et al. (2011) and human density in 2005.*

the effectiveness of available techniques to monitor black bear populations in the Northeast, given management objectives.

As shown in Chapters 8, 9, and 10, many suitable monitoring options exist for black bear managers in the Northeast, but there is no single appropriate method for all bear populations and management objectives (Table 11.1). Due to an abundance of monitoring methods, managers are now faced with difficult decisions about how best to select a monitoring method for their current objectives that will also be useful for future monitoring. Invariably, managers want the best methods to address their objectives, but there are always budget constraints that ultimately magnify the tradeoffs between method reliability and cost. One of the greatest dilemmas for bear managers today is that the monitoring methods that provide the most accurate and precise estimates of population parameters (i.e., radiotelemetry, DNA-based mark-recapture) are also the most expensive . However, less expensive methods are ultimately a poor investment when money and resources are used to collect data that reveals little about bear populations and provide limited inference regarding the drivers of population change. We urge managers to always strive to use the best monitoring techniques available to address their objectives, even if decisions must ultimately be made based on financial considerations.

Monitoring method selection should begin with assessing whether the method being considered can produce estimates for the parameter of interest (i.e., meet management objectives) with the manager's desired level of precision. Once all of the suitable monitoring methods have been identified, managers should then consider the potential advantages and disadvantages of each method along with any special considerations (e.g., data requirements) before selecting a monitoring method (Table 11.1).

Table 11.1. Suitability of available monitoring methods for estimating population parameters for American black bears in jurisdictions of the Northeastern Black Bear Technical Committee.

Method suitability[a]	Bait stations	Population reconstruction	Radiotelemetry[b]	Biomarking	DNA-based	Ear tags	Harvest	Observations	Remote cameras
				\<Mark-recapture\>					
Parameter of Interest									
Abundance	+	+		+	++	+	+	+	+
Density			+	+	++	+	+	+	+
Survival			++	+	+	+	+	+	+
Reproduction			++	+	+	+	+	+	+
Population Growth	+	+	++	+	++	+	+	+	+
Advantages									
Proven track record of precise estimates		+	++		++	+			
Identify individual bears[c]			+		++	+	+	+	
Determine sex ratio		+	+	+	++	+	+	+	
Provide data on multiple wildlife species									++
Can also examine genetic structure and dispersal				+	++	+	+		
Can be used with other sampling methods			+	+	+	+	+	+	+
No additional costs if harvest monitored[d]		+					+		
Can identify drivers of parameter			++		+				
Disadvantages									
Relatively expensive			-		-	-			-
Logistically difficult			--	-	--	-			-
Capture, handling, or removal required			-	--		-	-		
Baiting bears may lead to habituation	--		-	-	--	-	-		
Concern for human consumption[e]				--	-		-		
Dependent on constant harvest and mortality		--							
Cannot positively identify species (i.e., bears)	--				-				
Individual marks can be lost				-			--	-	

Table 11.1 (continued)

Method suitability[a]	Bait stations	Population reconstruction	Radiotelemetry[b]	Mark-recapture Biomarking	DNA-based	Ear tags	Harvest	Observations	Remote cameras
Special Considerations									
Provides coarse data on many individuals					++	+	+	+	
Provides fine-scale data on few individuals			++						
Best for heavily harvested populations (>20%)		++				+	+		
Best for highly visible populations						+		++	
Requires specialized lab/field equipment			+	+	+	+			+
>1 year of data collection required		++	++	+		+	+		
>5 years of data collection required		++	++						
>10 years of data collection required		++	+						

[a] "++" = most suitable or applicable, "+ = suitable or applicable, null = not suitable or applicable, "-" = disadvantageous, "--" = most disadvantageous; based on synthesis of report findings.
[b] Demographic analysis using survival and fecundity rates from radiotelemetry.
[c] DNA-based method has potential to identify greatest proportion of population.
[d] Except DNA analysis for mark-recapture or cementum analysis for population reconstruction
[e] Because tetracycline used as a biomarker or drugs used during capture.

11.1.1 Parameter of Interest

Estimates of black bear population parameters are generally more valuable to managers than indices that do not provide estimates of sampling error. Because many reliable monitoring methods are available and there is an almost universal desire by NEBBTC managers for accuracy and precision (Chapter 7), we suggest that estimation methods be given precedence over indices. Use of indices for black bear population monitoring may be justified when budgets are severely

constrained and the index has well-known properties and data are already recorded for other purposes (i.e., harvest), but they will generally not be suitable for most management objectives.

11.1.1.1 Abundance and Density

Currently, we believe the most cost-effective and reliable technique to estimate black bear population abundance is DNA-based mark-recapture (see Chapter 8; Table 11.1), for which a number of sampling methods exist. Newer models (SECR) use similar sampling methods but use information on spatial locations of detectors, making them more suitable for estimating density when sampling is adequate given the nature of the population (e.g., home-range size; Chapter 8). Although SECR models are a rapidly developing area of research (e.g., Efford and Fewster 2012), when precise estimates of abundance are needed, managers should consider non-spatial models as they have a longer history in bear research and management and they are generally more robust to violation of assumptions. Fortunately, spatial and non-spatial estimators can usually be used in tandem. Regardless of the model type, with the number of marking methods now available for bears, the use of multiple sampling methods (e.g., hair traps, bear rubs, harvest) should be considered to reduce bias and increase precision of estimates.

11.1.1.2 Survival and Reproduction

One of the most reliable ways to estimate survival and reproductive parameters for black bears is capture, radiomarking, and monitoring (see Chapter 9; Table 11.1). Although radiotelemetry can be expensive, we believe it provides the most useful and reliable information on survival and emigration. Monitoring costs (flight time, ground radio-tracking) are greatly reduced if GPS collars are used compared with VHF collars, although the intitial costs are higher. Additionally,

radiotelemetry-based methods provide valuable information about causes of mortality and factors affecting reproduction. Furthermore, telemetry data can be used for other purposes such as habitat evaluation, movement dynamics, and estimating dispersal. Some open population mark-recapture models (e.g., robust design) can estimate apparent survival, but do not estimate true survival in geographically open populations. Nonetheless, DNA-based mark-recapture methods are a viable second option for estimating survival and reproduction.

11.1.1.3 Population Growth

A diversity of methods can be used to monitor black bear population growth in the Northeast (see Chapter 10; Table 11.1), but we suggest managers focus on two proven methods, radiotelemetry (i.e., demographic analysis using survival and recruitment rates) and DNA-based mark-recapture. NEBBTC managers indicated the importance of understanding the relationship between population trends and management efforts (Chapter 7). Thus, we believe that one method, radiotelemetry, stands out because of its proven track record and potential to detect different causes of population changes (Table 11.2). Conversely, DNA-based mark-recapture methods are powerful because they can produce precise estimates of growth rates in relatively short time periods (Table 11.1) and because the use of covariates allows testing of hypotheses for potential causes of population change. Although DNA-based methods can be expensive, costs can be reduced with subsampling or skipping years. Ultimately, the decision to use radiotelemetry or DNA-based mark-recapture should be based on management objectives.

Table 11.2. Ability of population growth estimation methods for American black bears to detect different causes of trend in jurisdictions of the Northeastern Black Bear Technical Committee.

Potential causes of trend[a]	Bait stations	Population reconstruction	Radiotelemetry[b]	Mark-recapture Biomarking	DNA-based	Ear tags	Harvest	Observations	Remote cameras
Excessive human-caused mortality									
Excessive legal hunting		+	++	+	+		+		
Unreported mortality (e.g., poaching)		+	++		+				
Hunting-related mortality		+	++	+	+		+		
Conflict bears		+	++		+	+	+		
Net emigration									
Disturbance		+	++	+	+				
Decline of habitat quality		+	+	+	+				
Attractive sinks		+	++	+	+				
Density-dependent dispersal		+	++	+	+	+			
Increased road or trail density									
Vehicle collisions		+		+					
Access for hunters or poachers		+	++	+	+				
Fragmented habitats		+	+	+	+				
Intraspecific killing									
Related to high density		+	++	+	+				
Reduced carrying capacity		+	+	+	+				
Skewed sex ratio		+	+	+	++	+	+		
Hunger		+	++	+	+				
Poor reproduction									
Decline of habitat quality		+	+		+	+			
Displacement from high-quality habitat		+	++		+				
Advanced age in female cohort		++	++		+	+	+		

Table 11.2 (continued)

Potential causes of trend[a]	Bait stations	Population reconstruction	Radiotelemetry[b]	Mark-recapture					
				Biomarking	DNA-based	Ear tags	Harvest	Observations	Remote cameras
High cub mortality									
Predation			+						
Poor nutrition			++						
Disease			+						
Accidents (e.g., vehicle collisions)			++						
Orphaning			++						
Other useful analytical abilities									
Habitat modeling, fine-scale	+	++		+	+	+	+		
Occurrence modeling, coarse-scale	+	++		+	++	+	+	+	+
Core and linkage habitat prediction	+	++		+	++	+	+		
Coarse bear movement info	+	++		+	++	+	+		
Movement data, fine-scale	+	++		+	+	+	+		
Population estimation	+	+		+	++	+	+		
Condition of the bears, health, disease	+	++		+		+	+	+	+
Diet studies, fine-scale (species of food)	+	++		+	+	+	+		
Diet studies, coarse-scale (isotope)	+	++		+	++	+	+		
Fragmentation and connectivity	+	++		+	++	+	+		
Sex and age structure	++	++		+	+	+	+		
Home-range size or overlap, dispersal	+	++		+	+	+	+		

[a] "++" = most suitable or applicable, "+ = suitable or applicable, null = not suitable or applicable; based on synthesis of report findings.
[b] Demographic analysis using survival and fecundity rates from radiotelemetry.

11.2 POPULATION MONITORING SCENARIOS

Black bear population studies should be designed to meet management objectives, but choosing an appropriate study design can be challenging for managers given the number of methods available (see Chapters 8, 9, and 10; Table 11.1). Ideally, a study design should be both effective at providing the desired information and efficient at collecting the required data. Again, the most common dilemma that managers face today is not how to monitor their bear populations, but how to monitor a population to maximize data collection while minimizing project costs. Therefore, in this chapter we provide comprehensive study design recommendations for monitoring black bear populations based on the results of our simulation analyses and our collective experience.

For simplicity, we will focus our discussion and evaluation of black bear monitoring options on the most suitable methods currently available (Table 11.1) for estimating the two population parameters most important to NEBBTC managers (abundance and population growth; Chapter 7). We present our recommendations for 6 population scenarios, ranging from small to large population sizes and from declining to stable or increasing population trends. We based our population size classes of small, ($N \leq 500$), medium ($N = 500–2,500$) and large ($N > 2,500$) on the range of black bear population sizes found in NEBBTC management units. We pooled recommendations for stable and increasing populations because most managers already use those distinctions to classify population trends. Although some managers may not know the exact status of their populations, the scenarios presented in this chapter should provide guidance for study design.

11.2.1 Small, declining population ($N \leq 500$, $\lambda < 1.0$)

Small, declining black bear populations are of the greatest conservation concern to managers and advantages of monitoring should be carefully weighed against the potential disadvantages. For example, capture and handling mortality concerns are greater with smaller populations than with larger populations because management efforts are often directed to increasing adult survival. In fact, managers should consider monitoring adult survival over population growth in these situations. Accurate and precise estimates are particularly important for small, declining populations because there is little room for error. The value in monitoring smaller populations has been questioned, however, because resources could be used more effectively to secure habitat or reduce human-bear conflicts. Nonetheless, we recommend using DNA-based mark-recapture to estimate abundance and population growth for small, declining populations because this method does not involve capture or handling, is affordable at small scales, and can provide precise estimates of λ in a shorter time period compared with radiotelemetry (Table 11.1).

Because DNA-based mark-recapture abundance estimates for small populations generally have poorer precision than larger populations (Section 8.3), managers must compensate by increasing sampling effort (e.g., number of occasions, number of sampling sites per unit area) to achieve the same level of precision. For example, a large population study may achieve a CV < 20% for abundance with 4 sampling occasions, whereas a small population would require at least 7 occasions to obtain a CV < 20% (Section 8.3.1). For all population scenarios, use of multiple sampling methods (e.g., hair traps and bear rubs) is highly recommended to increase sampling intensity to improve accuracy and increase precision in a cost-effective manner. SECR or open population models should be considered because smaller areas usually can amplify capture

heterogeneity due to a high proportion of animals that have home ranges extending beyond the edge of sampling grid.

11.2.2 Small, stable or increasing population ($N \leq 500$, $\lambda \geq 1.0$)

Many small black bear populations in the Northeast are newly reestablishing populations and are not harvested, so population reconstruction is not an option (Table 11.1). If the population is very small and expanding its range, then an index such as bait-station surveys (see Section 8.1.1.1) may be sufficient to monitor abundance, occupancy, or range expansion. Managers interested in monitoring abundance or population growth should consider DNA-based mark-recapture methods. Population growth aside, these methods provide the best baseline data on population size, density, and sex ratios to use as benchmarks to gauge future population dynamics related to management actions. For this scenario, radiotelemetry could also be considered to estimate population growth, but managers should be aware that at least 5 years of monitoring will be required, or more if vital rates are highly variable, to obtain a precise estimate of population growth (see Section 10.2.1). Again, because DNA-based mark-recapture abundance estimates for small populations generally have lower precision than larger populations, managers must compensate by increasing sampling intensity (e.g., number of occasions, number of sampling sites) to achieve the same level of precision as detailed previously.

11.2.3 Medium, declining population (N = 500–2,500, $\lambda < 1.0$)

Medium, declining populations represent a unique challenge for managers because it can be more difficult to obtain financial support for monitoring compared with small, declining

populations. We suggest that managers interested in estimating abundance for medium, declining populations use DNA-based mark-recapture as these methods provide the most reliable estimates (Table 11.1). Managers interested in monitoring population growth for medium, declining populations may consider using either DNA-based mark-recapture or radiotelemetry-based methods. Radiotelemetry-based methods may be inappropriate when the age distribution of the population is believed to be non-stable or if managers cannot wait to acquire sufficient data to use projection models (Table 11.1). Although both methods are suitable for estimating population growth (Table 11.1), if capture-related mortality is not a concern, radiotelemetry is the better method to use for this scenario as it can detect the causes of trend (Table 11.2), which is a desirable attribute to NEBBTC managers (Chapter 7).

11.2.4 Medium, stable or increasing population (N = 500–2,500, $\lambda \geq 1.0$)

Stable or increasing populations of medium size are probably the most common population scenario in the Northeast. In fact, few populations in the Northeast are currently declining (Chapter 6) and many statewide and management unit population estimates fall into this size range. With larger populations and small home-range sizes, the number of sampling sites needed for DNA-based estimates of abundance can be daunting. In such cases, radiotelemetry-based methods may be a better alternative.

DNA-based mark-recapture may be a viable option for managers interested mainly in monitoring population growth because that can be accomplished with lower capture probabilities, meaning that fewer samples would have to be genotyped and sites could be sampled for fewer occasions. Harvest and rub trees, for example, can be used to augment hair trapping efforts, thereby requiring less overall sampling effort to achieve comparable accuracy

and precision of parameter estimates. When additional sampling methods are used for the recapture sample for DNA-based mark-recapture, a saturated trapping grid may not be necessary because essentially all bears marked at hair traps would be available for recapture from harvest or at rubs; i.e., the method would not be dependent on the hair-sampling grid alone for the recapture sample.

The advantage of using radiotelemetry to monitor population growth is that it would provide inference regarding potential causes of trend (Table 11.2). The advantage of using DNA-based mark-recapture to estimate annual abundance is that it would also allow for estimating population growth through regressions of population size. A more tenable alternative to population-wide DNA sampling would be to sample smaller population subset(s) and extrapolate the estimates to a wider region, given certain assumptions (e.g., similar habitat, harvest intensity, patch size). Habitat covariates based on SECR models could be used to project range-wide densities (Drewry et al. 2013). We suggest that the decision to use DNA-based or telemetry methods be based on the interest of managers to also monitor abundance or survival (Table 11.1).

11.2.5 Large, declining population ($N = 2,500$, $\lambda < 1.0$)

As described previously, sampling requirements may be too intensive to make DNA-based mark-recapture feasible for estimating abundance with large populations. Again, population projection using radiotelemetry or DNA mark-recapture to determine population growth would probably be better alternatives. Also, the economies of scale are better for radiotelemetry than for DNA-based mark-recapture because the precision of parameter estimates from known-fate models is

independent of population size (Section 10.2.1), whereas costs increase with population size for DNA-based studies (i.e., more samples collected and more bears to identify).

11.2.6 Large, stable or increasing population ($N = 2,500$, $\lambda \geq 1.0$)

Probably the most common scenario in the Northeast are large, stable or increasing bear populations. The establishment of sub-regional monitoring programs among NEBBTC jurisdictions is an important consideration, particularly where bear populations are shared among multiple jurisdictions (e.g., Pennsylvania and New York or Maryland and West Virginia). Pooling resources to collaboratively monitor populations using radiotelemetry or DNA-based mark-recapture techniques would provide long-term benefits. We suggest that managers interested in estimating abundance for large, stable or increasing populations use DNA-based mark-recapture because this method provides the most reliable estimates (Table 11.1). Again, this would most likely entail a series of estimates for population subsets, with extrapolations as suggested before. As in the previous example, use of population projection or estimation of population growth using mark-recapture methods is a more reasonable, regional or jurisdiction-wide approach.

11.3 FUTURE WORK

Methods used for monitoring bear populations are evolving rapidly as this is an active area of research. Not only has technology changed (e.g., DNA analysis, GPS telemetry), but the methods for analyzing such data (e.g., Bayesian statistics, hierarchical methods) have changed as well (Royle and Dorazio 2008). We anticipate that many of the difficulties we detail in this report will eventually be reduced or overcome. For example, SECR models have much potential to serve as a valuable population monitoring tool for bear managers but further investigations are needed, and indeed ongoing, to test model assumptions and various aspects of the estimation process. One area that we think holds much promise is the integrated population analysis methods that combine harvest and auxiliary data (e.g., Skalski et al. 2007). More research needs to be done to evaluate the performance of this technique and to develop software more readily useful to bear managers. Another area that holds promise is the integration of mark-recapture methods and occupancy estimation (Conroy et al. 2008). If detection probabilities can be coupled with site-specific densities and habitat covariates, it may be possible to monitor only detection rates in the future once that relationship is established. Our report is a current view of the state of the art of population monitoring and we urge managers to stay tuned to the published literature so that they may take advantage of these anticipated breakthroughs and advances.

Section VI. References

References

Alexander, L. M. 1967. The Northeastern United States. D. Van Nostrand Company, Ltd., Toronto, Canada.

Alt, G. L., F. W. Alt, and J. S. Lindzey. 1976. Home range and activity patterns of black bears in northeastern Pennsylvania. Transactions of the Northeastern Fish and Wildlife Conference 33:45–46.

Alt, G. L., C. R. McLaughlin, K. H. Pollock. 1985. Ear tag loss by black bears in Pennsylvania. Journal of Wildlife Management 49:316–320.

Arnold, T. W. 2010. Uninformative parameters and model selection using Akaike's Information Criterion. Journal of Wildlife Management 74:1175–1178.

Barrett, T. 2011. Bear bones: Maine's black bear biologists among nation's elite. Maine Department of Inland Fisheries and Wildlife, Augusta, Maine. http://www.maine.gov/ifw/wildlife/species/bear/bearbones.htm

Beckmann, J. P., and J. Berger. 2003. Using black bears to test ideal-free distribution models experimentally. Journal of Mammalogy 84:594–606.

Beckmann, J. P., and C. W. Lackey. 2008. Carnivores, urban landscapes, and longitudinal studies: a case history of black bears. Human-Wildlife Conflicts 2:77–83.

Beecham, J. J. 1983. Population characteristics of black bears in west central Idaho. Journal of Wildlife Management 47:405–412.

Belant, J. L., D. R. Etter, S. L. Mayhew, L. G. Visser, and P. D. Friedrich. 2011. Improving large scale mark-recapture estimates for American black bear populations. Ursus 22:9–23.

Beringer, J., S. G. Seibert, S. Reagan, A. J. Brody, M. R. Pelton, and L. D. Vangilder. 1998. The influence of a small sanctuary on survival rates of black bears in North Carolina. Journal of Wildlife Management 62:727–734.

Beston, J. A. 2011. Variation in life history and demography of the American black bear. Journal of Wildlife Management 75:1588–1596.

Bittner, S. L., T. L. King, and W. F. Harvey. 2013. Estimating population size of Maryland's black bears using hair snaring and DNA analysis. Proceedings of the Annual Conference of the Southeastern Association of Fish and Wildlife Agencies (in press).

Boersen, M. R., J. D. Clark, and T. L. King. 2003. Estimating black bear population density and genetic diversity at Tensas River, Louisiana using microsatellite DNA markers. Wildlife Society Bulletin 31:197–207.

Borchers, D. L., and M. G. Efford. 2008. Spatially explicit maximum likelihood methods for capture–recapture studies. Biometrics 64:377–385.

Borchers, D. 2010. A non-technical overview of spatially explicit capture-recapture models. Journal of Ornithology DOI: 10.1007/s10336-010-0583-z

Boulanger, J., and B. McLellan. 2001. Closure violation in DNA-based mark-recapture estimation of grizzly bear populations. Canadian Journal Zoology 79:642–651.

Boulanger, J., B.N . McLellan, J. G. Woods, M. F. Proctor, C. Strobeck. 2004a. Sampling design and bias in DNA-based capture-mark-recapture population and density estimates of grizzly bears. Journal of Wildlife Management 68:457–469.

Boulanger, J., G. Stenhouse, and R. Munro. 2004b. Sources of heterogeneity bias when DNA mark-recapture sampling methods are applied to grizzly bear (*Ursus arctos*) populations. Journal of Mammalogy 85:618–624.

Boulanger, J. B., S. Himmer, and C. Swan. 2004c. Monitoring of grizzly bear population trends and demography using DNA mark-recapture methods in the Owikeno Lake area of British Columbia. Canadian Journal Zoology 82:1267–1277.

Boulanger, J., M. Proctor, S. Himmer, G. Stenhouse, D. Paetkau, and J. Cranston. 2006. An empirical test of DNA mark-recapture sampling strategies for grizzly bears. Ursus 17:149–158.

Boulanger, J., G. C. White, M. Proctor, G. Stenhouse, G. Machutchon, and S. Himmer. 2008. Use of occupancy models to estimate the influence of previous live captures on DNA-based detection probabilities of grizzly bears. Journal of Wildlife Management 72:589–595.

Boulanger, J., K. C. Kendall, J. B. Stetz, D. A. Roon, L. P. Waits, and D. Paetkau. 2008. Multiple data sources improve DNA-based mark-recapture population estimates of grizzly bears. Ecological Applications 18:577–589.

Bridges, A. S., M. R. Vaughan, and J. A. Fox. 2011. Reproductive Ecology of American Black Bears in the Alleghany Mountains of Virginia, USA. Journal of Wildlife Management 75:1137–1144.

Bridges, A. S., J. A. Fox, C. Olfenbuttel, and M. R. Vaughan. 2004. American black bear denning behavior: observations and applications using remote photography. Wildlife Society Bulletin 32:188–193.

Brongo, L. L., M. S. Mitchell, and J. B. Grand. 2005. Effects of trapping with bait on bait-station indices to black bear abundance. Wildlife Society Bulletin 33:1357–1361.

Burnham, K. P., and D. R. Anderson. 2002. Model selection and multi-model inference: a practical information-theoretic approach. 2nd edition, Springer Science Media,

New York, New York, USA

Carrel, W.K. 1994. Reproductive history of female black bears from dental cementum. Bears: their biology, and management 9:205–212.

Cattet, M., J. Boulanger, G. Stenhouse, R. A. Powell, and M. J. Reynolds-Hogland. 2008. An evaluation of long-term capture effects of ursids: implications for wildlife welfare and research. Journal of Mammalogy 89:973–990.

Caughlan, L., and K. L. Oakley. 2001. Cost considerations for long-term ecological monitoring. Ecological Indicators 1:123–134.

Clark, J. D., F. T. van Manen, and M. R. Pelton. 2005. Bait stations, hard mast, and black bear population growth in Great Smoky Mountains National Park. Journal of Wildlife Management 69:1633–1640.

Clark, J. D., and R. Eastridge. 2006. Growth and sustainability of black bears at White River National Wildlife Refuge, Arkansas. Journal of Wildlife Management 70:1094–1101.

Clark J. D., R. Eastridge, and M. J. Hooker. 2010. Effects of exploitation on black bear populations at White River National Wildlife Refuge. Journal of Wildlife Management 74:1448–1456.

Clevenger, A. P., and M. A. Sawaya. 2010. Piloting a non-invasive genetic sampling method for evaluating population-level benefits of wildlife crossing structures. Ecology and Society 15(1): http://www.ecologyandsociety.org/vol15/iss1/art7/

Conn, P. B., and D. R. Diefenbach. 2007. Adjusting age and stage distributions for misclassification errors. Ecology 88:1977–1983.

Conn, P. B., D. R. Diefenbach, J. L. Laake, M. A. Ternent, and G. C. White. 2008. Bayesian analysis of wildlife age-at-harvest data. Biometrics 64:1170–1177.

Conroy, M.J., J.P. Runge, R.J. Barker, M.R. Schofield, and C.J. Fonnesbeck. 2008. Efficient estimation of abundance for patchily distributed populations via two-phase, adaptive sampling. Ecology 89: 3362-3370.

Coster, S. S., A. I. Kovach, P. J. Pekins, A. B. Cooper, and A. Timmins. 2011. Genetic mark-recapture population estimation in black bears and issues of scale. Journal of Wildlife Management 75:1128–1136.

Coster, S. S., and A. L. Kovach. 2012. Anthropogenic influences on the spatial genetic structure of black bears. Conservation Genetics 13:1247–1257.

Coy, P. L., and D. L. Garshelis. 1992. Reconstructing reproductive histories of black bears from the incremental layering in dental cementum. Canadian Journal of Zoology 70:2150–2160.

Czetwertynski, S. M., M. S. Boyce, and F. K. Schmiegelow. 2007. Effects of hunting on demographic parameters of American black bears. Ursus 18:1–18.

Davis, M. L., J. Berkson, D. Steffen, and M. K. Tilton. 2007. Evaluation of accuracy and precision of Downing population reconstruction. Journal of Wildlife Management 71: 2297–2303.

Diefenbach, D. R., J. L. Laake, and G. L. Alt. 2004. Spatio-temporal and demographic variation in the harvest of black bears: implications for population estimation. Journal of Wildlife Management 68:947–959.

Downing, R. L. 1980. Vital statistics of animal populations. Pages 247–267 in S.D. Schemnitz, editor. Wildlife Techniques Manual. The Wildlife Society, Washington, D.C., USA.

Dreher, B. P., S. R. Winterstein, K. T. Scribner, P. M. Lukacs, D. R. Etter, et al. 2007. Noninvasive estimation of black bear abundance incorporating genotyping errors and harvested bear. Journal of Wildlife Management 71:2684–2693.

Dreher, B. P., G. J. M. Rosa, P. M. Lukacs, K. T. Scribner, and S. R. Winterstein. 2009. Subsampling hair samples affects accuracy and precision of DNA-based population estimates. Journal of Wildlife Management 73:1184–1188.

Drewry, J. M., F. T. van Manen, and D. M. Ruth. 2013. Density and genetic structure of black bears in coastal South Carolina. Journal of Wildlife Management 77:153–164.

Efford, M. G. 2004. Density estimation in live-trapping studies. Oikos 106:598–610.

Efford, M. G. 2012. secr: spatially explicit capture recapture in R. Dunedin: Department of Zoology, University of Otago, New Zealand.

Eiler, J. H., W. G. Wathen, and M. R. Pelton. 1989. Reproduction in black bears in the southern Appalachian Mountains. Journal of Wildlife Management 53:353–360.

Elowe, K. D., and W. E. Dodge. 1989. Factors affecting black bear reproductive success and cub survival. Journal of Wildlife Management 53:962–968.

Fieberg, J. R., K. W. Shertzer, P. B. Conn, K. V. Noyce, and D. L. Garshelis. 2010. Integrated population modeling of black bears in Minnesota: implications for monitoring and management. PLoS ONE 5(8):e12114. doi:10.1371/journal.pone.0012114

Foran, D. R., S. C. Minta, and K. S. Heinemeyer. 1997. DNA-based analysis of hair to identify species and individuals for population research and monitoring. Wildlife Society Bulletin 25:840–847.

Foster, R. J., and B. J. Harmsen. 2012. A critique of density estimation from camera-trap data. Journal of Wildlife Management 76:224–236.

Freedman, A. H., K. M. Portier, and M. E. Sunquist. 2003. Life history for black bears (*Ursus americanus*) in a changing demographic landscape. Ecological Modeling 167:47–64.

Frumhoff, P. C., J. J. McCarthy, J. M. Melillo, S. C. Moser, and D. J. Wuebbles. 2007. Confronting climate change in the U.S. Northeast: science, impacts, and solutions. Synthesis report of the Northeast Climate Impacts Assessment (NECIA), Union of Concerned Scientists (UCS), Cambridge, Massachusetts, USA.

Gardner, B., J. A. Royle, and M. T. Wegan. 2009. Hierarchical models for estimating density from DNA mark-recapture studies. Ecology 90:1106–1115.

Gardner, B., J. A. Royle, M. T. Wegan, R. E. Rainbolt, and P. D. Curtis. 2010. Estimating black bear density using DNA data from hair snares. Journal of Wildlife Management 74:318–325.

Garshelis, D. L. 1990. Monitoring effects of harvest on black bear populations in North America: a review and evaluation of techniques. Eastern Workshop on Black Bear Research and Management 10:120–144.

Garshelis, D. L., K. V. Noyce, and P. L. Coy. 1998. Calculating average age of first reproduction free of the biases prevalent in bear studies. Ursus 10:437–447.

Garshelis, D. L., and H. Hristienko. 2006. State and provincial estimates of American black bear numbers versus assessments of population trend. Ursus 17:1–7.

Garshelis, D. L., and K. V. Noyce. 2006. Discerning biases in a large scale mark-recapture population estimate for black bears. Journal of Wildlife Management 70:1634–1643.

Gottman, J. 1961. Megalopolis: the urbanized Northeastern Seaboard of the United States. The Twentieth Century Fund, New York, New York, USA.

Gove, N.E., J.R. Skalski, P. Zager, and R.L. Townsend. 2002. Statistical models for population reconstruction using age-at-harvest data. Journal of Wildlife Management 66:310–320.

Hall, E. R. 1981. The mammals of North America 2nd edition. John Wiley and Sons, New York, New York, USA.

Hamilton, R. J., and R. L.Marchinton. 1980. Denning and related activities of black bears in the coastal plain of North Carolina. International Conference on Bear Research and Management 3:121–126.

Hammond, F.M. 2002. The effects of resort and residential development on black bears in Vermont. Final report to Vermont Agency of Natural Resources.

Harley, S. J. R. A. Myers, and A. Dunn. 2001. Is catch-per-unit-effort proportional to abundance? Canadian Journal of Fisheries and Aquatic Sciences 58:1760–1772.

Harlow, H. 2012. Muscle protein and strength retention by bears during winter fasting and starvation. Comparative physiology of fasting, starvation, and food limitation. p. 277–296. DOI: 10.1007/978-3-642-29056-5_17

Harris, R. B., C. C. Schwartz, R. D. Mace, M. A. Haroldson. 2011. Study design and sampling intensity for demographic analyses of bear populations. Ursus 22:24–36.

Hebblewhite, M., M. Percy, and R. Serrouya. 2003. Black bear (*Ursus americanus*) survival and demography in the Bow Valley of Banff National Park, Alberta. Biological Conservation 112:415–425.

Hellgren, E. C., and M. R. Vaughan. 1989. Demographic analysis of a black bear population in the Great Dismal Swamp. Journal of Wildlife Management 53:969–977.

Heppell, S. S., H. Caswell, and L. B. Crowder. 2000. Life histories and elasticity patterns: perturbation analysis for species with minimal demographic data. Ecology 81:654–665.

Herrero, S., A. Higgins, J. E. Cardoza, L. I. Hajduk, and T. S. Smith. 2011. Fatal attacks by American black bear on people: 1900-2009. Journal of Wildlife Management 75:596–603.

Hines, J. E., and J. D. Nichols. 2002. Investigations of potential bias in the estimation of k using Pradel's (1996) model for capture–recapture data. Journal of Applied Statistics 29:573–587.

Hirth, D. H., J. M.Petty, and C. W. Kilpatrick. 2002. Black bear, *Ursus americanus*, hair and apple trees, *Malus pumila*, in Northeast North America. Canadian Field Naturalist 116:305–307.

Hobson, K. A., B. N. McLellan, and J. G. Woods. 1999. Using stable carbon and nitrogen isotopes to infer trophic relationships among black and grizzly bears in the upper Columbia River Basin, British Colombia. Canadian Journal of Zoology 78: 1332–1339.

Howe, E. J., M. E. Obbard, and J. A. Schaefer. 2007. Extirpation risk of an isolated black bear population under different management scenarios. Journal of Wildlife Management 71:603–612.

Howe, E. J., M. E. Obbard, R. Black, and L. L. Wall. 2010. Do public complaints reflect trends in human-bear conflict? Ursus 21:131–142.

Hristienko, H. and J. E. McDonald, Jr. 2007. Going into the 21st century: a perspective on trends and controversies in the management of the American black bear. Ursus 18:72–88.

Huffman, J. E., T. Ombrello, and D. Zellner. 2008. Evaluation of genetic structure and population estimate in New Jersey black bears (*Ursus americanus*). Northeast Wildlife DNA laboratory, East Stroudsburg University, Stroudsburg, Pennsylvania, USA.

Huggins, R. M. 1991. Some practical aspects of a conditional likelihood approach to capture experiments. Biometrics 47:725–732.

Immell, D., and R. G. Anthony. 2008. Estimation of black bear abundance using a discrete DNA sampling device. Journal of Wildlife Management 72:324–330.

Johnson, J. B., and K. S. Omland. 2004. Model selection in ecology and evolution. Trends in Ecology and Evolution 19:101–108.

Jolly, G. M. 1965. Explicit estimates from capture-recapture data with both death and immigration stochastic model. Biometrika 52:225–247.

Jonkel, C., and I. M. Cowan. 1971. The black bear in the spruce–fir forest. Wildlife Monographs 27.

Kalinowski, S. T., M. A. Sawaya, and M. L. Taper. 2006. Individual identification and distribution of genotypic differences. Journal of Wildlife Management 70:1148–1150.

Karanth, K. U., and J. D. Nichols. 1998. Estimation of tiger densities in India using photographic captures and recaptures. Ecology 79:2852–2862.

Kelly, M. J., and E. L. Holub. Camera trapping of carnivores: trap success among camera types and across species, and habitat selection by species, on Salt Pong Mountain, Giles County, Virginia. Northeastern Naturalist 15:249–262.

Kendall, W.L., K.H. Pollock, and C. Brownie. 1995. A likelihood-based approach to capture-recapture estimation of demographic parameters under the robust design. Biometrics 51: 293–308

Kendall, W. L., J. D. Nichols, J. E. Hines. 1997. Estimating temporary emigration using capture-recapture data with Pollock's robust design. Ecology 78:563–578.

Kendall, K. C., D. A. Roon, L. P. Waits, J. B. Boulanger, and D. Paetkau. 2008. Grizzly bear density in Glacier National Park. Journal of Wildlife Management 72:1693–1705.

Kendall, K. C., J. B. Stetz, J. Boulanger, A. C. Macleod, D. Paetkau, and G. C. White. 2009. Demography and Genetic Structure of a Recovering Grizzly Bear Population. Journal of Wildlife Management 73:3–17.

Koehler, G. M., and D. J. Pierce. 2005. Survival, cause-specific mortality, sex, and ages of American black bears in Washington state, USA. Ursus 16:157–166.

Kordek, W. S., and J. S. Lindzey. 1980. Preliminary analysis of female reproductive tracts from Pennsylvania black bears. Bear Biology Association Conference Series 3:159–161.

Krause, J., T. Unger, A. Noçon, A. Malaspinas, S. Kolokotronis, et al. 2008. Mitochondrial genomes reveal an explosive radiation of extinct and extant bears near the Miocene-Pliocene boundary. BMC Evolutionary Biology DOI: 10.1186/1471-2148-8-220

Lancia, R. A., J. D. Nichols, and K. H. Pollock. 1996. Estimating the number of animals in wildlife populations. Pages 215–253 *in* T.A. Bookhout, editor. Research and management techniques for wildlife and habitats. Fifth ed., rev. The Wildlife Society. Bethesda, Maryland, USA.

Lariviere, S. 2001. *Ursus americanus*. Mammalian Species 647:1–11.

Laufenberg, J. S. 2010. Effect of subsampling genotyped hair samples on model averaging to estimate black bear population abundance and density. M.Sc. Thesis, University of Tennessee, Knoxville, USA.

Laufenberg, J. S., F. T. van Manen, and J. D. Clark. 2013. Effects of sampling conditions on DNA-based estimates of American black bear abundance. Journal of Wildlife Management (in press).

Leopold, A. 1933. Game management. Charles Scribner's and Sons, New York, New York, USA.

Likert, R. 1932. A Technique for the Measurement of Attitudes. Archives of Psychology 140:1–55.

Lindenmayer, D. B., and G. E. Likens. 2010. The science and application of ecological monitoring. Biological Conservation 143:1317–1328.

Link, W.A. 2003. Nonidentifiability of population size from capture-recapture data with heterogeneous detection probabilities. Biometrics 59:1123–1130.

Long, R. A., T. M. Donovan, P. Mackay, W. J. Zielinski, and J. S. Buzas. 2007. Comparing scat detection dogs, cameras, and hair snares for surveying carnivores. Journal of Wildlife Management 71:2018–2025.

Long, R. A., P. Mackay, W. J. Zielinski, and J. C. Ray. 2008. Noninvasive survey methods for carnivores. Island Press Washington, D.C., USA

Long, R. A., T. E. Donovan, P. Mackay, W. J. Zielinski, and J. S. Buzas. 2011. Predicting carnivore occurrence with noninvasive surveys and occupancy modeling. Landscape Ecology 26:327–340.

Lundberg, D. A., R. A. Nelson, H. W. Wahner, and J. D. Jones. 1976. Protein metabolism in the black bear before and during hibernation. Mayo Clinic Proceedings. Mayo Clinic 51: 716–722.

MacKenzie, D. I., J. D. Nichols, J. A.Royle, K. H. Pollock, L. L. Bailey, and J. E. Hines. 2006. Occupancy estimation and modeling: inferring patterns and dynamics of species occurrence. Elsevier, San Diego, California, USA.

Mace. R. D., S. C. Minta, T. L. Manley, and K. E. Aune. 1994. Estimating grizzly bear

    population size using camera sightings. Wildlife Society Bulletin 22:74–83.

Marescot, L., R. Pradel, C. Duchamp, S. Cubaynes, E. Marboutin, R. Choquet, C. Miquel, and O.

    Gimenez. 2011. Ecological Applications 21:2898–2907.

Martorello, D. A., T. H. Eason, and M. R. Pelton. 2001. A sighting technique using cameras to

    estimate population size of black bears. Wildlife Society Bulletin 29:560–567.

Matthews, S. M., R. T. Golightly, and J. M. Higley. 2008. Mark-resight density estimation for

    American black bears in Hoopa, California. Ursus 19:13–21.

McClintock, B. T., and G. C. White. 2009. A less field-intensive robust design for estimating

    demographic parameters with mark-resight data. Ecology 90:313–320.

McClintock, B. T., and G. C. White. 2012. From NOREMARK to MARK: software for

    estimating demographic parameters using mark–resight methodology. Journal of

    Ornithology 152(2):S641-S650. DOI 10.1007/s10336-010-0524-x

McDonald, J. E., and T. K. Fuller. 2001. Prediction of litter size in American black bears.

    Ursus 12:93–102.

McDonald, J. E., and T. K. Fuller. 2005. Effects of spring acorn availability on black bear diet,

    milk composition, and cub survival. Journal of Mammalogy 86:1022–1028.

McLaughlin, C. R., G. J . Matula, and R. J. O'Conner. 1994. Synchronous reproduction by

    Maine black bears. International Conference on Bear Research and Management 9:471–

    479.

Miller, S. D. 1990. Population management of bears in North America. International

    Conference on Bear Research and Management 8:357–373.

Millner, J. M., E. B. Nilsen, and H. P. Andreassen. 2007. Demographic side effects of selective hunting in ungulates and carnivores. Conservation Biology 21:36–47.

Mills, L.S ., J. J. Citta, K. P. Lair, M. K. Schwartz, and D. A. Tallmon. 2000. Estimating animal abundance using noninvasive genetic sampling: promise and pitfalls. Ecological Applications 10:283–294.

Mills, L. S. 2007. Conservation of wildlife populations: demography, genetics, and management. Blackwell publishing, Malden, Massachusetts, USA.

Mitchell, M.S., L. B. Pacifici, J. B. Grand, and R. A. Powell. 2009. Contributions of vital rates to growth of a protected population of American black bears. Ursus 20:77–84.

Mowat, G., and C. Strobeck. 2000. Estimating population size of grizzly bears using hair capture, DNA profiling, and mark-recapture analysis. Journal of Wildlife Management 64:183–193.

Murphy, M. A., K. C. Kendall, A. Robinson, and L. P. Waits. 2007. The impact of time and field conditions on brown bear (*Ursus arctos*) faecal DNA amplification. Conservation Genetics 8:1219–1224.

Nelson, R. A., G. A. Folk, E. W. Pfeiffer, J. J. Craighead, C. J. Jonkel, and D. L. Steiger. 1983. Behavior, biochemistry, and hibernation in black, grizzly, and polar bears. International Conference on Bear Research and Management 5:284–290.

New York State Department of Environmental Conservation. 2007. Black bears in New York: natural history, range, and interactions with people. Second edition. New York State Department of Environmental Conservation, Albany, New York.

Ngoprasert, D., D. H. Reed, R. Steinmetz, and G. A. Gale. 2012. Density estimation of Asian bears using photographic capture-recapture sampling based on chest marks. Ursus 23:117–133.

Nichols, J. D., and J. E. Hines. 2002. Approaches for the direct estimation of K, and demographic contributions to K, using capture–recapture data. Journal of Applied Statistics 29:539–568.

Nichols, J. D., and B. K. Williams. 2006. Monitoring for conservation. Trends in Ecology and Evolution 21:668–673.

Nicholson, J. M., and F. T. van Manen. 2009. Using occupancy models to determine mammalian responses to landscape changes. Integrative Zoology 4:232–239.

Noss, A. J., B. Gardner, L. Maffei, E. Cuéllar, R. Montaño, A. Romero-Muñoz, R. Sollman, and A. F. O'Connell. 2012. Comparison of density estimation methods for mammal populations with camera traps in the Kaa-Iya del Gran Chaco landscape. Animal Conservation. Online Early. DOI: 10.1111/j.1469-1795.2012.00545.x

Noyce, K. 2012. Summary of States/Provinces Status Reports. Proceedings of the 20th Eastern Black Bear Workshop, edited by C. Ollfenbuttel and G. Warburton. Hendersonville, North Carolina.

Obbard, M. E., and E. J. Howe. 2008. Demography of black bears in hunted and unhunted areas of the boreal forest of Ontario. Journal of Wildlife Management 72:869–880.

Obbard, M. E., B. A. Pond, A. Schenk, R. Black, M. N. Hall, and B. Jackson. 2008. Suspended baits: can they help hunters distinguish male and female American black bears. Ursus 19:33–42.

Obbard, M. E. E. J. Howe, and C. J. Kyle. 2010. Empirical comparison of density estimators for large carnivores. Journal of Applied Ecology 47:76–84.

Otis, D. L., K. P. Burnham, G. C. White, and D. R. Anderson. 1978. Statistical inference from capture data on closed animal populations. Wildlife Monographs 62.

Paetkau, D. 2003. An empirical exploration of data quality in DNA-based population inventories. Molecular Ecology 12:1375–1387.

Pelton, M. R. 2003. Black bear. Pages 504–514 *in* J. A. Chapman and G. A. Feldhammer, editors. Wild mammals of North America: biology, management, and economics. The Johns Hopkins Univ. Press, Baltimore, Maryland, USA.

Pollock, K.H. 1982. A capture-recapture design robust to unequal probability of capture. Journal of Wildlife Management 46: 752–757.

Pollock, K.H., and D.G. Raveling. Assumptions of modern band-recovery models, with emphasis on heterogeneous survival rates. Journal of wildlife Management 46: 88–98.

Pollock, K. H., J. D. Nichols, C. Brownie, and J. E. Hines. 1990. Statistical inference for capture-recapture experiments. Wildlife Monographs 107.

Powell, R. A., J. W. Zimmerman, D. E. Seaman, and J. F. Gilliam. 1996. Demographic analyses of a hunted black bear population with access to a refuge. Conservation Biology 10:224–234.

Pradel, R. 1996. Utilization of capture-mark-recapture for the study of recruitment and population growth rate. Biometrics 52:703–709.

R Development Core Team. 2005. R: A language and environment for statistical computing, reference index version 2.2.1. R Foundation for Statistical Computing, Vienna, Austria. ISBN 3-900051-07-0, URL http://www.R-project.org.

Reynolds-Hogland, M. J., L. B. Pacifici, and M. S. Mitchell. 2007. Linking resources with demography to understand resource limitation for bears. Journal of Applied Ecology 44:1166–1175.

Rice, C. G., J. Rohlman, J. Beecham, and S. Pozzanghera. 2001. Power analysis of bait station surveys in Idaho and Washington. Ursus 12:227–236.

Rogers, L. L., and L. D. Mech. 1981. Interactions of Wolves and Black Bears in Northeastern Minnesota. Journal of Mammalogy 62:434–436.

Roon, D. A., L. P. Waits, and K.C . Kendall. 2003. A quantitative valuation of two methods for preserving hair samples. Molecular Ecology Notes 3:163–166.

Roon, D. A., M. E. Thomas, K. C. Kendall, and L. P. Waits. 2005. Evaluating mixed samples as a source of error in non-invasive genetic studies using microsatellites. Molecular Ecology 14:195–201.

Rossell, C. R., and J. A. Litvaitis. 1994. Application of harvest data to examine responses of black bears to land-use changes. Bears: their biology and management 9:275–281.

Royle, J. A. and R. M. Dorazio. 2008. Hierarchical modeling and inference in ecology: the analysis of data from populations, metapopulations, and communities. Academic Press, San Diego, California..

Royle, J. A., J. D. Nichols, K. U. Karanth, and A. M. Gopalaswamy. 2009. A hierarchical model for estimating density in camera-trap studies. Journal of Applied Ecology 46:118–127.

Royle, J. A., Nichols, J. D., and M. Ke´ry. 2005. Modelling occurrence and abundance of species when detection is imperfect. Oikos 110: 353–359.

Royle, J. A., and K. V. Young. 2008. A hierarchical model for spatial capture-recapture data. Ecology 89:2281–2289.

Ryan, C. W., J. C. Pack, W. K. Igo, J. C. Rieffenberger, and A. B. Billings. 2004. Relationship of mast production to big-game harvests in West Virginia. Wildlife Society Bulletin 32:786–794.

Ryan, C.W. 2009. Population ecology, residents' attitudes, hunter success, economic impact, modeling management options and retentiton time of Telazol of West Virginia black bears. Dissertation submitted to West Virginia University, Morgantown, WV.

Ryan, C. W., M. R. Vaughan, J. B. Meldrum, R. B. Duncan, and J. W. Edward. 2009. Retention time of Telazol in black bears. Journal of Wildlife Management 73:210–213.

Samson, C., and J. Huot. 1995. Reproductive biology of female black bears in relation to body mass in early winter. Journal of Mammalogy 76:68–77.

Sawaya, M. A., J. B. Stetz, A. P. Clevenger, M. L. Gibeau, and S. T. Kalinowski. 2012. Estimating grizzly and black bear population abundance and trend in Banff National Park using noninvasive genetic sampling. PLoS ONE: 7(5):e34777. doi:10.1371/journal.pone.0034777

Scheick, B., W. McCown, and M. Orlando. 2012. North America Bear Range Mapping Project. Proceedings of the 20th Eastern Black Bear Workshop, edited by C. Ollfenbuttel and G. Warburton. Hendersonville, North Carolina.

Schmidt, J. I., J. M. Ver Hoeff, J. A. K. Maier, and R. T. Bower. 2005. Catch pre unit effort for moose: a new approach using Weibull regression. Journal of Wildlife Management 69:1112–1124.

Schwartz, C. C., and A. W. Franzmann. 1991. Interrelationship of black bears to moose and forest succession in the northern coniferous forest. Wildlife Monographs 113.

Schwartz, C. C., M. A. Haroldson, K. A. Gunther, and D. Moody. 2002. Distribution of grizzly bears in the Greater Yellowstone Ecosystem, 1990–2000. Ursus 13:203–212.

Schwartz, M. K., G. Luikart, and R.S. Waples. 2007. Genetic monitoring as a promising tool for conservation and management. Trends in Ecology and Evolution 22:25–33.

Seber, G. A. F. 1965. A note on the multiple recapture census. Biometrika 52:249-259.

Settlage, K. E., F. T. van Manen, J. D. Clark, and T. L. King. 2008. Challenges of DNA-based mark-recapture studies of American black bears. Journal of Wildlife Management 72: 1035–1042.

Skalski, J. R., M. V. Clawson, and J. J. Millspaugh. 2012a. Model evaluation in statistical population reconstruction. Wildlife Biology 18:225–234.

Skalski, J. R., J. J. Millspaugh and M. V. Clawson. 2012b. Comparison of statistical population reconstruction using full and pooled adult age-class data. PLoS ONE 7(3) doi:10.1371/journal.pone.0033910.

Skalski, J. R., K. E. Ryding, and J. J. Millspaugh. 2005. Wildlife demography: analysis of sex, age, and count data. Academic Press, San Diego, California, USA.

Skalski, J. R., R. L. Townsend, and B. A. Gilbert. 2007. Calibrating population reconstruction models using catch-effort and index data. Journal of Wildlife Management 71:1309–1316.

Sorensen, V. A., 1998. Estimating survival rates of black bears. Canadian Journal of Zoology 76:1335–1343.

Statistics Canada. 2011. Government of Canada's Census of Population. Ottawa, ON, Canada <http://www12.statcan.gc.ca/census-recensement/index-eng.cfm>. Accessed 01 April 2012.

Stetz, J. B., K. C. Kendall, and C. Servheen. 2010. Evaluation of bear rub surveys to monitor grizzly bear population trends. Journal of Wildlife Management 74:860–870.

Stetz, J. B., K. C. Kendall, C. D. Vojta, and GeM Working Group. 2011. Genetic monitoring for managers: a new online resource. Journal of Fish and Wildlife Management 2:216–219.

Stetz, J.B., K.C. Kendall, and A.C. Macleod. 2013. Black bear density in Glacier National Park, Montana. Wildlife Society Bulletin *In Press*.

Stickley, A. R. 1961. A black bear tagging study in Virginia. Proceedings of the Annual Conference of the Southeastern Association of Game and Fish Commissioners 15:43–54.

Stoneberg, R. P., and C. J. Jonkel. 1966. Age determination of black bears by cementum layers. Journal of Wildlife Management 30:411–414.

Taberlet, P., L. P. Waits, and G. Luikart. 1999. Noninvasive genetic sampling: look before you leap. Trends in Ecology and Evolution 14:323–327.

Ternent, M. A, and D. L. Garshelis. 1999. Taste-aversion conditioning to reduce nuisance activity by black bears in a Minnesota military reservation. Wildlife Society Bulletin 27: 720–728.

Ternent, M. A. 2006. Management and biology of black bears in Pennsylvania: ten year plan (2006–2015). Pennsylvania Game Commission, Harrisburg, Pennsylvania.

Ternent, M, and D. Sittler. 2007. Black bear reproduction in Northcentral Pennsylvania. Project annual job report to Pennsylvania Game Commission.

Tredick, C. A., and M. R. Vaughan. 2007. DNA-based population demographics of black bears in coastal North Carolina and Virginia. Journal of Wildlife Management 73:1031–1039.

Tredick, C. A., M. R. Vaughan, D. F. Stauffer, S. L. Simek, and T. Eason. 2007. Sub-sampling genetic data to estimate black bear population size: a case study. Ursus 18:179–188.

van Manen, F. T., M. F. McCollister, J. M. Nicholson, L. M.Thompson, J. L. Kindall, and M. D. Jones. 2012. Short-term impacts of a 4-lane highway on American black bears in eastern North Carolina. Wildlife Monographs 181.

Vashon, J. H., M. R. Vaughan, A. D. Vashon, D. D. Martin, and K. N. Echols. 2003. An expandable radiocollar for black bear cubs. Wildlife Society Bulletin 31:380–386.

Waits, J. L., and P. L. Leberg. 2000. Bias associated with population estimation using molecular tagging. Animal Conservation 3:191–199.

Waits, L. P., and D. Paetkau. 2005. Noninvasive genetic sampling tools for wildlife biologists: a review of applications and recommendations for accurate data collection. Journal of Wildlife Management 69:1419–1433.

Wasser, S. K., B. Davenport, E. R. Ramage, K. E. Hunt, M. Parker, C. Clarke, and G. Stenhouse. 2004. Scat detection dogs in wildlife research and management: application to grizzly and black bears in the Yellowhead Ecosystem, Alberta, Canada. Canadian Journal of Zoology 82:475–492.

Weaver, K. M., and M. R. Pelton. 1994. Denning ecology of black bears In the Tensas River Basin of Louisiana. International Conference on Bear Research and Management 9:427–433.

White, G. C., and K. P. Burnham. 1999. Program MARK: survival estimation from populations of marked animals. Bird Study 46 Supplement, 120–138.

Williams, B. K., J. D. Nichols, and M. J. Conroy. 2002. Analysis and management of animal populations. Academic Press, San Diego, California, USA.

Williams, B. K., R. C. Szaro, and C. D. Shapiro 2007. Adaptive Management: The U.S. Department of the Interior Technical Guide. Adaptive Management Working Group, U.S. Department of the Interior, Washington, D.C., USA.

Wisdom, M. J., L. S. Mills, and D. F. Doak. 2000. Life stage simulation analysis: estimating vital-rate effects on population growth for conservation. Ecology 81:628–641.

Woods, J. G., D. Paetkau, D. Lewis, B. N. McLellan, M. Proctor, and C. Strobeck. 1999. Genetic tagging of free-ranging black and brown bears. Wildlife Society Bulletin 27:616–627.

U.S. Census Bureau. 2010. United States Census 2010. U.S. Census Bureau, Washington, D.C., USA. <http://2010.census.gov/2010census/data/index.php>. Accessed 01 April 2012.

U.S. Department of the Interior, Fish, and Wildife Service, and the U.S. Department of Commerce, U.S. Census Bureau. 2006. National survey of fishing, hunting, and wildlife-associated recreation. U.S. Department of the Interior, Washington, D.C., USA. <http://www.census.gov/prod/www/abs/fishing.html>. Accessed 01 April 2012.

Yoccoz, N. G ., J. D. Nichols, and T. Boulinier. 2001. Monitoring of biological diversity in space and time. Trends in Ecology and Evolution 16:446–453.

Zucchini, W., Borchers, D.L., Erdelmeier, M., Rexstad, E. and Bishop, J. 2007. WiSP 1.2.4. Institut fur Statistik und Okonometrie, Geror-August-Universitat Gottingen, Platz der Gottinger Seiben 5, Gottingen, Germany.

Section VII. Appendices

Appendix A: Glossary

Term	Definition
Age structure	The relative proportions of a population in different age classes.
Birth rate	The average number of offspring produced per individual per unit of time, often expressed as a function of age (x).
Bootstrap analysis	A nonparametric statistical analysis for computing confidence intervals for a phylogeny or a point estimate (e.g., of FST). Re-sampling with replacement to estimate the proportion of times an event (such as the positioning of a node on a phylogenetic tree) appears during multiple re-sampling of a data set.
Carrying capacity	Number of individuals in a population that the resources of a habitat can support; the asymptote, or plateau, of the logistic and other sigmoid equations for population growth.
Census population size	The number of individuals in a population.
Demographic stochasticity	Differences in the dynamics of a population that are the effects of random events on individuals in the population.
Demography	The study of population structure and growth.
Density independent	Having an influence on individuals in a population that does not vary with the density of that population.
Density	Number of individuals in relation to the space, volume, or other resources that are needed.
Density dependent	Having an influence on individuals in a population that varies with the density of that population. Often applied to birth and death rates.
Deterministic	Events that have no random or probabilistic aspects but rather occur in a completely predictable fashion.
Deterministic model	Mathematical model in which all of the relationships are fixed and the concept of probability does not enter; a given input produces an exact prediction as an output.
Distribution	The geographic extent of a local population or ecological unit.
Effective population size (N_e)	The average size of a population expressed in terms of the number of individuals assumed to contribute genes equally to the next generation; generally smaller that the actual size of the population, depending on the variation in reproductive success among individuals.
Environmental stochasticity	Random variation in environmental factors that influence population parameters affecting all individuals in population.
Exponential rate of increase (r)	The natural log of the finite rate of increase. Also called the instantaneous rate of increase.

Fecundity	The potential reproductive capacity of an individual or population. For wildlife populations typically used to represent the number of female offspring per female per unit time).
Fertility	An ecological concept of the actual number of viable offspring produced by an organism, equivalent to realized fecundity.
Finite rate of increase (λ)	The ratio of the population density in one year to that in the previous year (N_t/N_t-1); exponential or intrinsic rate of increase).
Fitness	The ability of an individual, or genotype to survive and produce viable offspring. Quantified as the number of offspring contributed to the next generation, or as proportion of the individual's genes in all the genes contributed to the next generation.
Gene flow	Exchange of genetic information between demes through migration.
Habitat	The place where an animal or plant normally lives, often incorrectly characterized by a dominant vegetation type or plant form.
Hypothesis	Universal propositions that suggenets an explanation for some observed ecological situation
Instantaneous rate of increase (r)	The rate of increase of a population undergoing exponential growth under a given set of ecological conditions; can be positive, negative, or zero, and if birth and death rates are constant for sufficient time will produce a stable age distribution. In the logistic equation, r-r max (1-N/K_0.
Intrinsic rate of increase (r_{max})	The rate of increase of a population undergoing exponential growth under optimum ecological conditions; the maximum instantaneous rate that a species is capable of. It is a characteristic of a species (Cf. exponential and instantaneous rate of increase).
Leslie matrix life history	A matrix of values of age-specific fecundity and survivorship used to project the size and age structure of a population through time; a population matrix.
Life history	The set of adaptations of an organism that influence the life-table values of age-specific survival and fecundity, such as reproductive rate or age at first reproduction.
Likelihood statistics	An approach for parameter estimation and hypothesis testing that involves building a model (i.e., a likelihood function) and the use of the raw data (not a summary statistic), which often provides more precision and accuracy than frequentist statistic approaches (method of moments). The parameter of interest is estimated as the member of the parameter space that maximizes the probability of obtaining your observed data. Likelihood approaches facilitate comparisons between different models (e.g., via likelihood ratio tests) and thus the testing of alternate hypotheses (e.g., stable versus declining population size).

Maximum likelihood estimate (MLE)	A method of parameter estimation that obtains the parameter value that maximizes the likelihood of the observed data.
MCMC	Markov chain Monte Carlo. A tool or algorithm for sampling from probability distributions based on constructing a Markov chain. The state of the chain after many steps is then used as a sample from the desired distribution. Sometimes called a random walk Monte Carlo method.
Microsatellite	Tandemly repeated DNA consisting of short sequences of one to six nucleotides repeated between approximately five and 100 times. Also known as VNTRs, SSRs, or STRs.
Mortality (m_x)	Ratio of the number of deaths to individuals at risk, often described as a function of age (x).
Net reproductive rate (R_e)	The expected number of offspring produced by a female during her lifetime.
Niche	The set of resources and environmental conditions that allow a single species to persist in a particular region, often conceived of as multidimensional space. Also called fundamental niche.
Omnivore	An organism whose diet is broad, including both plant and animal foods. Specifically, an organism that feeds at more than one trophic level.
Parameter	In statistics, an unknown true characteristic of a ststicical population. It is usually impossible to know the value of a parameter. A statistic estimates a parameter.
Phenology	Study of the periodic (seasonal) phenomena of animal and plant life and their relations to the weather and climate (e.g. the time of flowering in plants).
Poisson distribution	A probability distribution, with identical mean and variance, that characterizes discrete events occurring independently of one another in time, when the mean probability of that event on any one trial is very small. Earthquake hazards, radioactive decay, and mutation events follow a Poisson distribution. The Poisson is a good approximation to the binomial distribution when the probability is small and the number of trials is large.
Polymerase chain reaction (PCR)	A technique to replicate a desired segment of DNA. PCR starts with primers that flank the desired target fragment of DNA. The DNA strands are first separated with heat, and then cooled allowing the primers bind to their target sites. Polymerase then makes each single strand into a double strand, starting from the primer. This cycle is repeated multiple times creating a 106 increase in the gene product after 20 cycles and a 109 increase over 30 cycles.
Population growth rate (dN/dt)	The rate of growth of a population over a short period of time, defined by the product of population size, N, and the instantaneous rate of increase, r. (see logistic equation).

Recruitment	Increment to a natural population, usually from young animals entering the adult population.
Resource	A substance or object required by an organism for normal maintenance, growth, and reproduction.
Sex ratio	Ratio of the number of individuals of one sex to that of another sex in a population.
Shadow effect	A case usually caused by low marker polymorphism in mark–recapture studies in which a novel capture is labeled as a recapture due to identical genotypes at the loci studied.
Single nucleotide polymorphism (SNP)	A nucleotide site (base pair) in a DNA sequence that is polymorphic in a population either due to transitions or transversions and can be used as a marker to assess genetic variation within and among populations. Usually only two alleles exist for a SNP in a population.
Stable age distribution	The proportions of individuals in various age classes in a population that has a constant instantaneous rate of growth, r.
Stage-classified population	A population containing individuals of different developmental stages (e.g., adults and larvae) in the same or different habitats.
Stochastic	The presence of a random variable in determining the outcome of an event.
Stochastic model	Mathematical model based on probabilities; the prediction of the model is not a single fixed number but a range of possible numbers (Cf. deterministic model).
Survival (l_x)	Proportion of newborn individuals alive at age x; also called survivorship.
Survivorship curve	Curve showing the number of individuals surviving to age x (log scale) plotted against age.
Sympatric	Occurring in the same place; usually refers to areas of overlap in species distribution (Cf. allopatric).
Type I error	The rejection of a true null hypothesis
Type II error	The failure to reject a false null hypothesis

Appendix B: Survey Questions and Results

In January and February of 2012 we requested that the primary representatives from each NEBBTC member jurisdiction complete an online survey to help us better understand elements of bear research and management that could not be captured from the available literature. Only one representative per jurisdiction was invited to complete the survey. To minimize nonresponse issues, we reminded members about the survey on at least three separate occasions.

We attempted to keep the survey as simple as possible to meet our objectives and maximize response rate. Questions were designed with the assistance of social scientists with experience creating surveys of similar nature. Further, we enlisted wildlife biologists both inside and outside of the NEBBTC who are familiar with bear management issues to review the questions. To maximize efficiency and ease for the respondents, the survey was conducted online using SurveyMonkey, with most questions requiring simply clicking a radial button.

Below are the exact questions paired with a summary of responses, either in narrative or graphical format.

_____

Introduction
The following set of questions is intended to give us a better understanding of how black bear managers and biologists in the northeast conduct their work. As such, we request that you answer the following questions as they pertain to the populations in your jurisdiction. Most questions use the Likert scale, where answers are ranked from strongly disagree to strongly agree. This is a very powerful, and rapid, way of clustering beliefs to identify patterns across a sample of people. We will use these results in the Technical Report that we are preparing for the NEBBTC to identify relevant black bear monitoring and management options across jurisdictions.

We are requesting your name, position, and jurisdiction simply to help us track how representative our sample is. Individual responses will not be published

Definitions
1) Monitoring: for our purposes, this refers to long-term, focused monitoring efforts (e.g., multi-year population growth rate estimates) as opposed to status assessments (e.g., one-time abundance estimate).
2) Precision: any measure of uncertainty related to monitoring. These can include standard errors (SE), coefficients of variation (CV), confidence intervals (CI), or others. A measure of uncertainty is what separates parameter estimates (e.g., abundance) from an index.
3) Adaptive Management: a structured, iterative process of optimal decision making in the face of uncertainty, with an aim to reducing uncertainty over time via system monitoring. In this way, decision making simultaneously maximizes one or more resource objectives and, either passively or actively, accrues information needed to improve future management. For our purposes,

"adaptive management" includes variations such as adaptive impact management and adaptive resource management.

4) Jurisdiction: State or Province.

Questions

1. What is your name?

2. What is your current job title?

3. What jurisdiction do you represent?

At what level within your organization are management decisions typically made?	Fourteen of 15 respondents answered that decisions are made at the level of the state/province natural resources agency, with the remainder being at the local level. Three respondents noted, however, that legislators or game commissions or other levels of government can intervene in part depending on the political climate or specific issue.
5. Management decisions are made at the most effective level within my jurisdiction.	
6. There is strong administrator support for new programs or research and monitoring efforts in my jurisdiction.	
7. If you answered Disagree or Strongly Disagree to the previous question, what do you see as the primary constraints affecting current or potential future monitoring efforts?	Seven respondents provided narrative answers to question 7. The general consensus was that support is a function of funding and the attitude of the current administration. Either or both can be factors.

8. Establishing Northeastern region-wide standards for monitoring is important to facilitate communication and improve management across jurisdictions.	Strongly Agree	Agree	No Opinion	Disagree	Strongly Disagree
	0	11	0	4	0

9. There is good communication about black bear population status across jurisdictions.	Strongly Agree	Agree	No Opinion	Disagree	Strongly Disagree
	4	7	0	4	0

10. There is good communication about black bear monitoring across jurisdictions.	Strongly Agree	Agree	No Opinion	Disagree	Strongly Disagree
	2	10	0	3	0

11. There is good communication about black bear management objectives across jurisdictions.	Strongly Agree	Agree	No Opinion	Disagree	Strongly Disagree
	0	11	0	4	0

12. There is strong concordance of black bear management objectives across jurisdictions.	Strongly Agree	Agree	No Opinion	Disagree	Strongly Disagree
	0	3	3	9	0

13. I am familiar with the concept of adaptive management.	Bar chart: Strongly Agree: 4, Agree: 10, No Opinion: 1, Disagree: 0, Strongly Disagree: 0 (y-axis 0–15)
14. It is more important to manage bear populations based on numerical population goals or based on bear impacts to people and property.	Bar chart: Strongly Agree: 2, Agree: 5, No Opinion: 3, Disagree: 4, Strongly Disagree: 1 (y-axis 0–6)
15. What proportion of populations in your jurisdiction are at or beyond social carrying capacity?	Bar chart: <20%: 5, 20-40%: 4, 40-60%: 4, 60-80%: 1, >80%: 0 (y-axis 0–6)
16. There is strong public support for black bear conservation in my jurisdiction.	Bar chart: Strongly Agree: 4, Agree: 6, No Opinion: 2, Disagree: 3, Strongly Disagree: 0 (y-axis 0–8)
17. There is strong public support for black bear harvest in my jurisdiction.	Bar chart: Strongly Agree: 3, Agree: 8, No Opinion: 2, Disagree: 2, Strongly Disagree: 0 (y-axis 0–10)
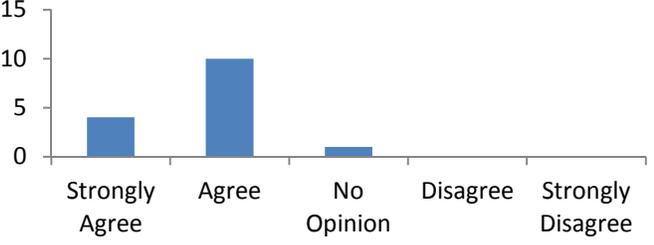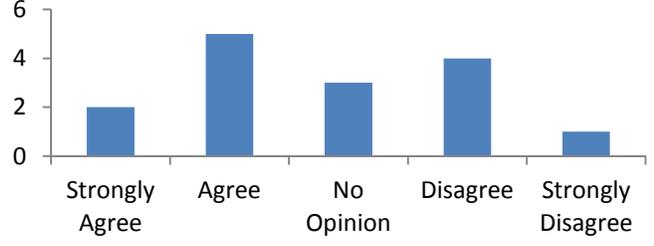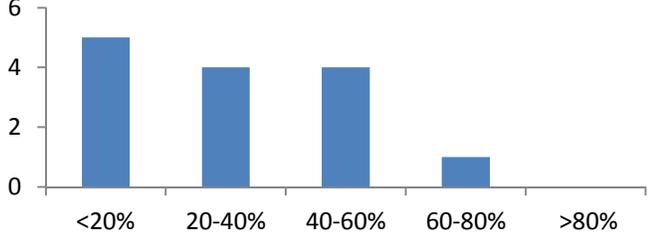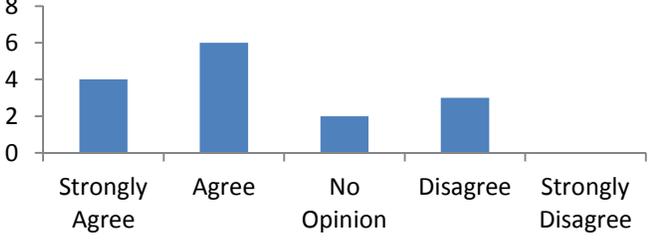
18. Bear hunting advocacy groups play an important role in helping to manage bear populations in my jurisdiction.	
19. Adaptive Management plays an important role in managing bear populations in my jurisdiction.	
20. Current management efforts and methods are adequate for reaching/maintaining objectives.	
21. Adjusting harvest levels (through season length, bag limits, permit quotas, open areas, etc) is the most important management tool I use.	
22. Educational efforts are the most important investment for reaching management objectives in my population.	

23. Game laws enforcement is the most important investment for reaching management objectives in my population.	Bar chart (y-axis 0 to 10): Strongly Agree = 0, Agree = 1, No Opinion = 3, Disagree = 8, Strongly Disagree = 3
24. Feeding/sanitation enforcement is the most important investment for reaching management objectives in my population.	Bar chart (y-axis 0 to 10): Strongly Agree = 2, Agree = 2, No Opinion = 3, Disagree = 8, Strongly Disagree = 0
25. Minimizing human-bear conflicts (e.g., via improved sanitation) is the most important investment for reaching management objectives in my population.	Bar chart (y-axis 0 to 8): Strongly Agree = 3, Agree = 6, No Opinion = 2, Disagree = 4, Strongly Disagree = 0
26. Q425 follow-up: For your top choice among the last three questions, please explain why.	Several respondents noted that there is no single most important component for reaching management objectives, and that many are interrelated. In particular, education and conflict reduction (largely via sanitation) are linked. Education leads to fewer conflicts leads to increasing social carrying capacity. This, then, has direct implications to management as populations grow and new problems and opportunities develop. Clearly, reducing human-bear conflicts is among the top priorities for all jurisdictions.
27. Abundance is the most important parameter for management decision making.	Bar chart (y-axis 0 to 8): Strongly Agree = 2, Agree = 3, No Opinion = 3, Disagree = 7, Strongly Disagree = 0
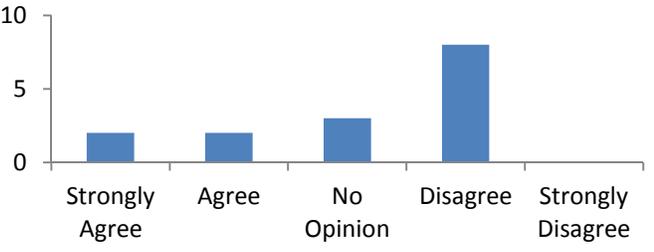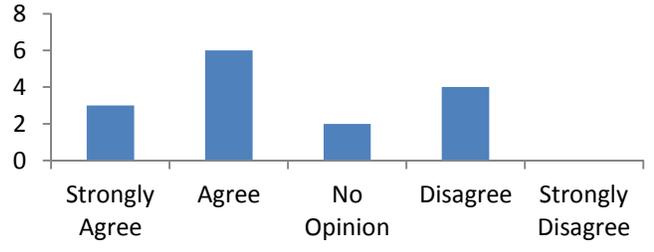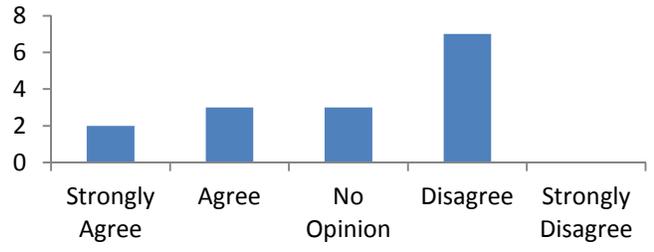
28. Population growth rate is the most important parameter for management decision making.	Bar chart (y-axis 0–8): Strongly Agree = 1, Agree = 7, No Opinion = 2, Disagree = 5, Strongly Disagree = 0
29. Survival is the most important parameter for management decision making.	Bar chart (y-axis 0–8): Strongly Agree = 0, Agree = 7, No Opinion = 2, Disagree = 6, Strongly Disagree = 0
30. Fecundity is the most important parameter for management decision making.	Bar chart (y-axis 0–8): Strongly Agree = 0, Agree = 6, No Opinion = 3, Disagree = 6, Strongly Disagree = 0
31. Population persistence from Population Viability Analysis (PVA) is the most important parameter for management decision making.	Bar chart (y-axis 0–8): Strongly Agree = 0, Agree = 1, No Opinion = 4, Disagree = 7, Strongly Disagree = 3
32. Monitoring habitat is more important than monitoring demographic questions.	Bar chart (y-axis 0–15): Strongly Agree = 0, Agree = 1, No Opinion = 2, Disagree = 12, Strongly Disagree = 0
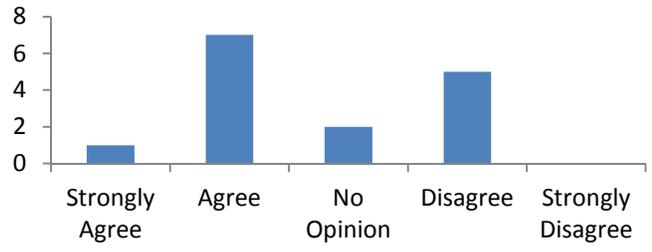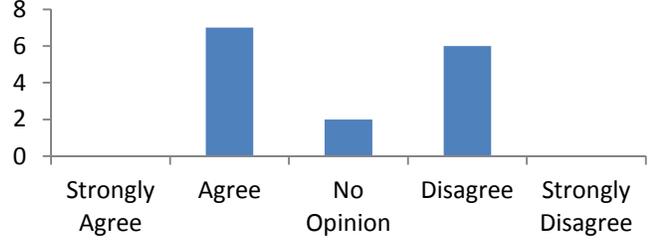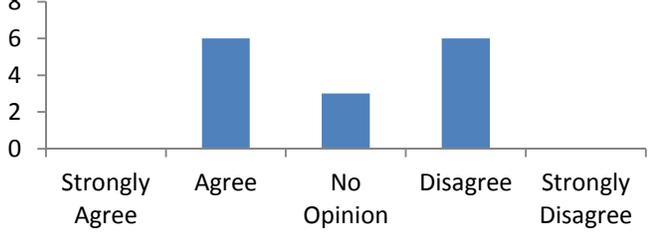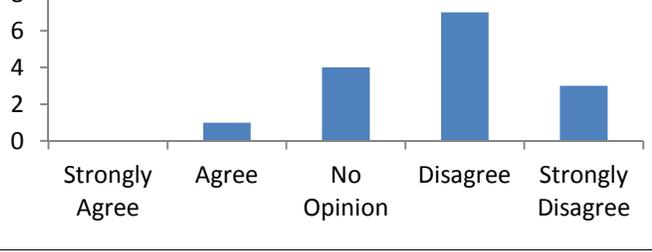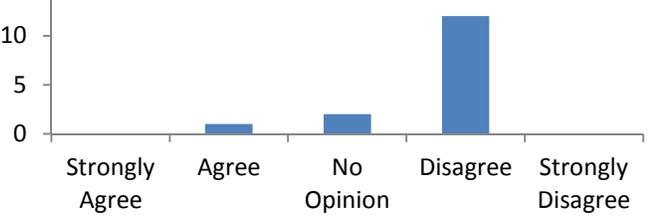
33. Precision of estimates plays an important role in how I perceive and use information in decision making.	Bar chart with y-axis from 0 to 10. Strongly Agree: 3, Agree: 8, No Opinion: 2, Disagree: 2, Strongly Disagree: 0.
34. There are thresholds of precision after which I discount or ignore scientific results.	Bar chart with y-axis from 0 to 8. Strongly Agree: 0, Agree: 6, No Opinion: 7, Disagree: 2, Strongly Disagree: 0.
35. If you answered "agree" or "strongly agree" to the previous question, what threshold CV is acceptable for decision making? For example, a point estimate of 100, with a SE=20 would yield a CV of 20%. This equates to a simple 95% confidence interval of 60.8-139.2.	Bar chart with y-axis from 0 to 15. 0-5%: 1, 5-10%: 1, 10-20%: 12, 20-30%: 1, >30%: 0.
36. For decision making in my jurisdiction, the most appropriate time frame to monitor a population after management objectives (e.g., abundance) have been met is:	Bar chart with y-axis from 0 to 8. 0 Years: 1, 1-5 Years: 6, 5-10 Years: 4, <10 years: 0, Indefinitely: 3.
37. Understanding the causes of population changes is more important than only knowing that populations are changing.	Bar chart with y-axis from 0 to 8. Strongly Agree: 5, Agree: 7, No Opinion: 1, Disagree: 2, Strongly Disagree: 0.

38. Establishing measurable management objectives is important for adaptive management of wildlife populations.	**Chart:** Bar chart (y-axis 0 to 15). Strongly Agree: 5, Agree: 10, No Opinion: 0, Disagree: 0, Strongly Disagree: 0
39. My jurisdiction has explicit bear management objectives or a formal bear management plan.	**Chart:** Bar chart (y-axis 0 to 8). Strongly Agree: 5, Agree: 6, No Opinion: 1, Disagree: 2, Strongly Disagree: 1
40. We have reached management objectives in the following percentage of my jurisdiction:	**Chart:** Bar chart (y-axis 0 to 6). <20%: 5, 20-40%: 3, 40-60%: 5, 60-80%: 1, >80%: 1
41. Precise parameter estimates are more important for populations that are declining or threatened.	**Chart:** Bar chart (y-axis 0 to 8). Strongly Agree: 1, Agree: 6, No Opinion: 2, Disagree: 5, Strongly Disagree: 1
42. Monitoring demography is an important component of successful conservation and management.	**Chart:** Bar chart (y-axis 0 to 15). Strongly Agree: 2, Agree: 12, No Opinion: 1, Disagree: 0, Strongly Disagree: 0

43. Q42 follow-up: Why?	There was some confusion about question 42, which invalidates the response for it and this question. From this follow-up question, it became apparent that several respondents interpreted question 42 as being related to human demography.
44. Successful management does not require monitoring population demography.	Strongly Agree: 0, Agree: 1, No Opinion: 2, Disagree: 9, Strongly Disagree: 3
45. Black bear populations are robust and therefore do not require focused monitoring efforts.	Strongly Agree: 0, Agree: 0, No Opinion: 2, Disagree: 11, Strongly Disagree: 2
46. More detailed information on population trends would improve management in my population.	Strongly Agree: 3, Agree: 9, No Opinion: 2, Disagree: 1, Strongly Disagree: 0
47. Genetic monitoring methods are inferior to other monitoring methods.	Strongly Agree: 0, Agree: 0, No Opinion: 6, Disagree: 7, Strongly Disagree: 2
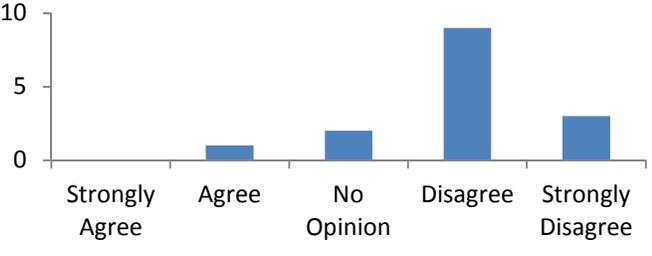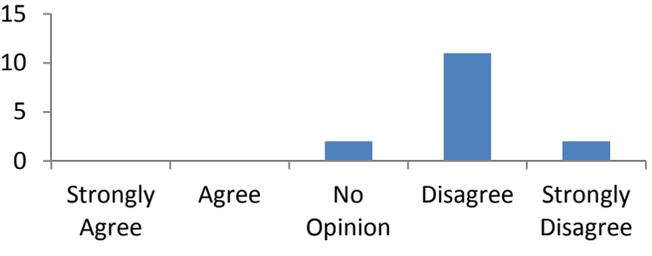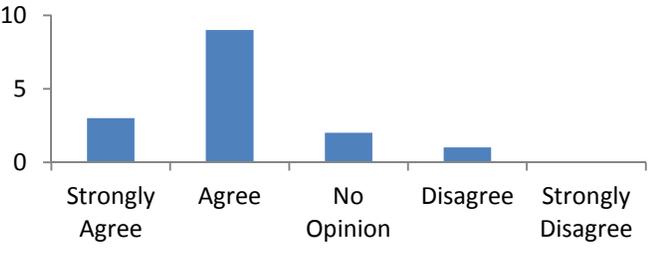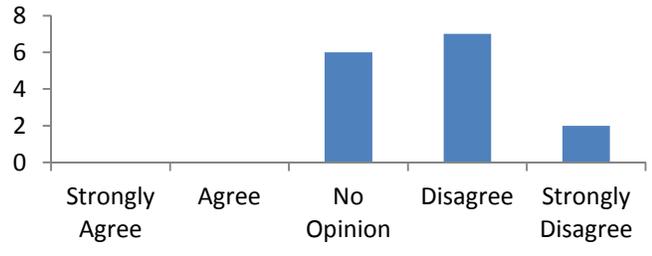
48. Q47 follow-up: Why?	With near consensus, genetic methods are considered superior to, or an excellent complement to, traditional methods, depending on the research question. The general perception, however, is that genetic methods are prohibitively expensive due to sampling demands.
49. The status of a population should determine how much is invested in monitoring, e.g., declining populations should receive greater funding to determine causes.	Bar chart (y-axis 0–10): Strongly Agree = 0, Agree = 7, No Opinion = 0, Disagree = 8, Strongly Disagree = 0
50. I attempt to stay current with the scientific literature, eg the Journal of Wildlife Management.	Bar chart (y-axis 0–15): Strongly Agree = 2, Agree = 11, No Opinion = 2, Disagree = 0, Strongly Disagree = 0
51. Sources other than the peer-reviewed literature are more important for learning about new monitoring methods.	Bar chart (y-axis 0–8): Strongly Agree = 0, Agree = 2, No Opinion = 7, Disagree = 6, Strongly Disagree = 0
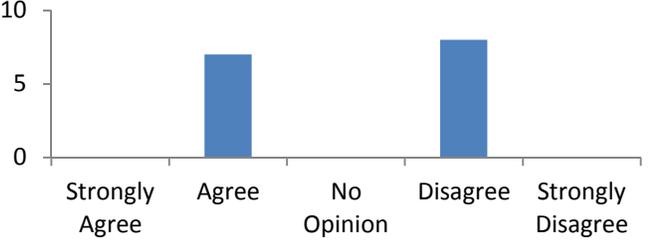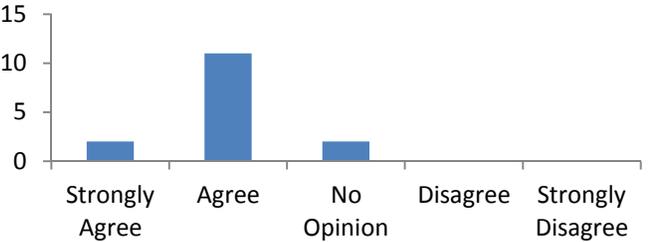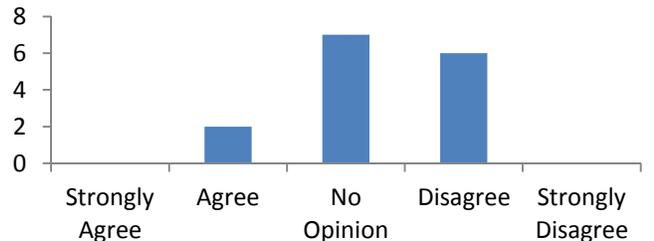
Appendix C: Noninvasive Genetic Sampling Literature Summary

Table C.1. Summary of publications related to the use of noninvasive genetic sampling methods to estimate black bear population parameters and model capture probabilities (Chapter 8).

Reference	Site name	Year	State/Province	Area (km^2)
Coster et al. (2011)	Pittsburg	2006	NH	196
Coster et al. (2011)	Milan	2006	NH	223
Coster et al. (2011)	Pittsburg	2007	NH	196
Coster et al. (2011)	Milan	2007	NH	223
Tredick and Vaugahn (2009)	Great Dismal Swamp NWR	2001	VA	175
Tredick and Vaugahn (2009)	Pocosin Lakes NWR	2002	NC	115
Tredick and Vaugahn (2009)	Alligator River NWR	2003	NC	150
Tredick and Vaugahn (2009)	Great Dismal Swamp NWR	2002	VA	175
Tredick and Vaugahn (2009)	Pocosin Lakes NWR	2003	NC	115
Tredick and Vaugahn (2009)	Alligator River NWR	2004	NC	150
Dreher et al. (2007)	Northern Lower Peninsula	2003	MI	36,848
Mowat et al. (2005)	Sout-central Selkirks	1996	BC	5,226
Mowat et al. (2005)	North-central Selkirks	1996	BC	4,640
Poole et al. (2002)	Prophet Plateau	1998	BC	5,413
Poole et al. (2001)	Prophet Mountains	1998	BC	3,114
Mowat et al. (2005)	Yellowhead	1999	BC	5,352
Mowat et al. (2005)	Parsnip Plateau	2000	BC	3,016
Mowat et al. (2005)	Parsnip Mountains	2000	BC	3,636
Mowat et al. (2005)	Bowron	2001	BC	2,494
Tredick et al. (2007)	Pungo Unit of Pocosin Lakes, NWR	2002	NC	50
Tredick et al. (2007)	St. Johns	2001	FL	967

Gardner et al. (2010)	Fort Drum	2006	NY	157
Belant et al. (2005)	Stockton Island	2002	WI	41
Belant et al. (2005)	Sand Island	2002	WI	12
Bittner et al. (2013)	Alleghany/Garrett County	2000	MD	2152
Settlage et al. (2008), Laufenberg et al. (2013,	Great Smoky Mountains NP	2003	TN	200
Boersen et al. (2003)	Tensas River Tract	1999	LA	329
Immell and Anthony (2008)	Steamboat	2003	OR	112
Immell and Anthony (2008)	Toketee	2003	OR	155
Immell and Anthony (2008)	Steamboat	2004	OR	138
Immell and Anthony (2008)	Toketee	2004	OR	145
Settlage et al. (2008)	Great Smoky Mountains NP	2003	TN	160
Settlage et al. (2008)	3 National Forests	2003	NC, SC, GA	329
Triant et al. (2004)	Inland	1999	LA	208
Triant et al. 2004)	Coastal	1999	LA	142
Stetz et al., unpublished data	Glacier NP	2004	MT	6,600
Stetz et al., unpublished data	Glacier NP	2005	MT	6,600
Obbard et al. (2010)	11 WMUs	2004, 2005	ON	
Sawaya et al. (2012)	Banff NP	2006	AB	2,246
Sawaya et al. (2012)	Banff NP	2008	AB	2,247

Appendix D: Mark-recapture Models

Black bear population parameters (e.g., abundance, density) can be estimated from a variety of mark-recapture models, each with their own ability to accommodate assumptions (see Chapters 8, 9, and 10; Table D.1). Some of these assumptions apply to all models (e.g., demographic closure, marks are not lost), whereas other assumptions can only be relaxed with specific model types. Managers should examine the model assumptions and decide which ones would apply to their populations before finalizing their selections. Independent of which model type is chosen, managers should gain a thorough understanding of the assumptions that apply to their model before making management decisions based on its results. This is particularly important for interpreting density estimates based on models that relax the assumption of geographic closure.

Although there are many more mark-recapture models than we can possibly cover here, the models we presented (Table D.1) have the greatest potential to provide reliable estimates of population parameters for black bears in the Northeast. Of these, the Huggins closed-capture model (Huggins 1991) has been used most frequently to estimate black bear abundance because of its ability to incorporate individual covariates to model detection probabilities (i.e., increase estimate precision). Closed-population mixture models have also been used when individual capture heterogeneity is known to bias estimates because of differences in individual detection probabilities. Recently developed SECR models allow for precise estimates while addressing the geographic closure assumption, but these models have their own assumptions that may not be biologically feasible (e.g., circular home ranges that are stationary during sampling) or may be violated in real-world sampling scenarios (e.g., capture probability highest at home-range

center). When demographic or geographic closure is known to be violated, we suggest that managers consider an open population model such as Pollock's robust design (Pollock et al. 1990), which requires multiple primary sampling periods (usually years).

Table D.1. Assumptions of mark-recapture models for estimating American black bear population parameters in jurisdictions of the Northeastern Black Bear Technical Committee.

Model suitability[a]	Closed population	Closed pop w/ mixture	Huggins-closed	Spatially-explicit	Integrated	Mark-resight	Open population
Parameter of interest							
Abundance	+	+	++	+	+	+	+
Density	+	+	+	++	+	+	+
Survival					+		++
Reproduction					+		++
Population growth					+		++
Model assumptions							
Every animal has chance of being readably marked	+	+	+	+	+	+	+
Marks are read correctly and not lost	+	+	+	+	+	+	+
Every animal with equal capture probability	+					+	
Every animal has circular home range				+			
Detection probability highest at home range center				+			
Detection probability can be related to covariates			+	+	+	+	
Population has been representatively sampled	+	+	+	+	+	+	+
Population is demographically closed	+	+	+	+	+	+	
Population is geographically closed	+	+	+		+	+	
Study area boundaries do not change							+

[a] "++" = most suitable or applicable, "+ = suitable or applicable, null = not suitable or applicable; based on synthesis of report findings.

Appendix E: Statistical Analysis Software

Most of the mark-recapture model types presented in Section IV (see Chapters 8, 9, and 10; Table D.1) can be implemented with various computer software packages, many of which are freely available. These software programs can estimate a range of population parameters and offer a number of appealing features (Table E.1). Some of the software programs have excellent graphic user interfaces (GUI) which makes obtaining parameter estimates easy for wildlife managers. However, caution should always be applied when using these programs because different settings can have profound effects on the reliability of population parameter estimates (e.g., different link functions in Program MARK).

Among the many software programs available to managers, Program MARK (White and Burnham 1999) stands out with the most proven track record of reliable use for estimating black bear abundance and population growth. Program MARK can accommodate a variety of data types (e.g., hair snares, bear rubs, telemetry, remote photographs, observations) and can implement most mark-recapture models that are commonly used by researchers and managers to estimate black bear population parameters (Table E.1).

Program R packages have particular appeal over stand-alone programs such as MARK because they generally offer greater versatility to estimate parameters of interest (Table E.1). Nonetheless, Program R requires an extensive time investment to adequately learn the programming language. Therefore, managers may encounter the frustration of a steep learning curve to become proficient with producing reliable results. However, advantages to investing the time to learn R code are that the program is free and open source, the programming language is

consistent across many different packages, extensive online resources are are available, and new

packages are constantly being developed to accommodate new sampling methods and mark-

recapture models.

Table E.1. Software packages for analyzing mark-recapture models for American black bear populations in jurisdictions of the Northeastern Black Bear Technical Committee.

| | | | | | Program R packages | |
Software suitability[a]	DENSITY	MARK	RISKMAN	SECR	RMARK	WiSP
Parameter of interest						
Abundance	+	++		+	+	+
Density	++	+		++	+	+
Survival		++			++	
Recruitment		++			++	
Population growth	[b]	++	++	[b]	++	
Features						
Open access software	+	+	+	+	+	+
Open-source code				+	+	+
Documentation	++	++	++	++	+	+
Graphic user interface	++	+	++			+
Individual covariates	+	+		+	+	+
Sex-specific parameter estimates	+	+	+	+	+	+
Simulations	++	++	++	+		++

[a] "++" = most suitable or applicable, "+ = suitable or applicable, null = not suitable or applicable; based on synthesis of report findings.
[b] = new models under development

Appendix F: R Code for Closed Population Abundance Simulations

NEBBTC Simulation of closed population abundance estimation using routines in ## the R package 'WiSP'. Modified by Jeff Stetz and Mike Sawaya - summer 2012

WiSP is not on CRAN - must be downloaded from developer website: ## http://www.ruwpa.st-and.ac.uk/estimating.abundance/WiSP/index.html

I had to extract files to a folder not in C:\Program Files, then copy/paste to C:\Program Files\R \R-2.15.1\library


```
## ---- set working directory ------------------------
setwd('C:\...\'); getwd()
require(wisp); require(rgl); require(xlsx)
x.len <- c(100,200) study region
y.len <- c(100,200) study region
ngroups <- seq(400,900,by=100) this is the number of individuals
occ <- c(5,7,10) occasions
for(x in 1:length(x.len)) { for(y in 1:length(y.len)) {
for(n in 1:length(ngroups)) { for(o in 1:length(occ)) {
for(repl in 1:50){
my.region <- generate.region(x.length=x.len[x], y.width=y.len[y])
dimensions (aka survey region)
my.density <- generate.density(my.region, southwest=1, (simple plane in this case)
southeast=1, northwest=1) #plot.density.population(my.density)
resolution plots can slow things down
my.pop.pars <- setpars.population(my.density, number.groups=ngroups[n],
population (here, #groups=#individuals) size.method="user",
#
# x-dimension values for # y-dimension values for Number of groups; for us # Number of sampling
# Creates population
# Defines density surface # 3D wire plot; high
# Number of animal groups in
# Method of how animal group
sizes are determined;
size.min=1, size.max=1, size.mean=1,
size values - if 'size.method' has been set to user.
only active if 'size.method' set to 'poisson'
reflecting individuals are independently detected exposure.method="beta",
# 'size.method = "user"' allows the user to enter possible group size values and their probabilities.
# Method of how group exposure is determined. 'method = beta' for Poisson-distributed group exposure values;
user provides possible group exposure values and their probabilities.
```

```
exposure.min=0, exposure.max=1,
exposure values (only used when 'exposure.method = user'.
exposure.mean=0.5, (only if 'exposure.method = beta').
exposure.shape=0.1,
Beta distribution (only if 'exposure.method = beta').
type.values = c("Male","Female"), properties for animal groups.
# If 'method = user' the
# Lower and upper bounds of # Mean group exposure value # Shape parameter of the
# Vector of possible type
# Vector of possible group # min, max, and mean size
# I've set group size to 1,type.prob = c(0.45,0.55)) # Vector of respective type my.pop <-
 generate.population(my.pop.pars)
# summary(my.pop)
#plot.population(my.pop, type='details', show.sizes=T, show.exp=T, dsf=0.75, title='my.pop')
my.cr.design.pars <- generate.design.cr(my.region, # Capture-recapture design parameters;
n.occ=occ[o], effort=rep(1,occ[o])) # number of occasions; relative effort across occasions
"effort=c(1,1,1,1,1)"
my.sample.cr.pars <- min/max values
my.point.est.crMh <- point.est.crMh(my.cr.sample,num.mix =2,init.N =-1) # Currently set to model Mh
 with 2 mixtures
#summary(my.point.est.crMh) # Currently have summaries turned off to limit clutter
my.interval.est.crMh <- int.est.crMh(my.cr.sample,num.mix =2,init.N =-1, # Nonparametric bootstrap CIs
 with 99 runs
ci.type='boot.nonpar', nboot=99, plot=F) #summary(my.interval.est.crMh)
##---- A new row for a dataframe with each element, for instance ---------------------------------
if(my.pop.pars$size.method=="user"){ groupsize=mean(my.pop.pars$size.values)
} if(my.pop.pars$size.method!="user"){
groupsize=my.pop.pars$size.mean }
#A single row of the table
new.row <- data.frame(Nhat.ind=my.interval.est.crMh$boot.mean$Nhat.ind,
 effort=my.cr.design.pars$effort[1],
occasions=my.cr.design.pars$number.occasions,
SE=my.interval.est.crMh$se$Nhat.ind, replicate=repl,x=x.len[x],y=y.len[y],ngroup=ngroups[n],
 nindivid=ngroups[n]*groupsize,occ=occ[o],
min.cp.mark=my.sample.cr.pars$theta0.marked, max.cp.mark=my.sample.cr.pars$theta0.marked,
 min.cp.unmark=my.sample.cr.pars$theta1.unmarked, max.cp.unmark=my.sample.cr.pars
$theta1.unmarked )
##---- Within a loop, you do the following to add the new row to your table (or create a table)
if(exists("out.table")){
out.table <- rbind(out.table,new.row)
} if(!exists("out.table")){
out.table <- new.row
probabilities.
# was pmin.unmarked=0.01, pmax.unmarked=0.25, # Re/capture probability
setpars.survey.cr(my.pop, my.cr.design.pars, pmin.marked=0.01, pmax.marked=0.25,
improvement=0) # Improvement in detection my.cr.sample <- generate.sample.cr(my.sample.cr.pars)
across sessions #summary(my.cr.sample)
}
}#end repl
##---- Save output to .xlsx file; static destination file name --------------------------------------
```

```
##---- Destination .xlsx has to be created first (one time); worksheets added after that -------------
filespot <- ("C:/.../NEBB.wisp.sims.output.xlsx")
##---- Dynamic worksheet name; worksheet added to common destination file ----------------------------
##---- R will return an error if worksheet with same name exists or if file is open -----------------
SaveExcel <- write.xlsx(out.table, filespot,
 sheetName=paste("XY",x.len[x],y.len[y],"N",ngroups[n],"Occ",occ[o],
"Mh.005.5",sep="."), col.names=T, row.names=F, append=T)
rm(out.table)
}#end o
}#end n }#end y
}#end x
### ---- Simulations based on conditions defined above ------------------------- #
#my.Mh.cr.sim <- point.sim.crMt(pop.spec=my.pop.pars,
'mypop' and 'mydens' allows randomization
# design.spec=my.cr.design.pars,
# survey.spec=my.sample.cr.pars, B=99, seed=123456)
repllicates; setting seed makes it replroducible #
# using user defined # B=num
#save(out.table,file=paste("wisp.X",x,"Y",y,"N",n,"Occ",o,"Mh.005.5.RData", sep=".")) #
#summary(my.Mh.cr.sim)
##plot(my.Mh.cr.sim)
#
#
## Suggested citation:
## Zucchini, W., Borchers, D.L., Erdelmeier, M., Rexstad, E. and Bishop, J. 2007.
## WiSP 1.2.4. Institut fur Statistik und Okonometrie, Geror-August-Universitat Gottingen, ## Platz der
 Gottinger Seiben 5, Gottingen, Germany.
```

Appendix G: R Code for Spatially-explicit Capture Recapture Abundance Simulations

```
## An R function to run a suite of SECR simulations
## Modified from Murray Efford's 2012-05-31 code by J.Stetz and A.Mynsberge

## Set working directory!!
setwd('C:/Sinopah/NEBBTC/Simulations/secr/Results');
getwd()

require(secr)

runsim <- function(nrepl = 25, outputfile = 'sim.output.RData') {


## ---- Parameter values ---------------------------------------
   D <- c(0.001, 0.005, 0.01, 0.015)
   g0 <- c(0.05, 0.01, 0.15)
   sigma <- c(400,1000, 2000, 3000)

## ---- Design variables ---------------------------------------
   spacing <- c(1000, 2000, 3000)
   occasions <- c(5,7,10)
   nspacing <- length(spacing)
   noccasions <- length(occasions)

## ---- Grid dimensions ----------------------------------------
   nx <- 25; ny <- 25

## ---- Simulation variables  ----------------------------------
   buff <- 20000                                    # common to use 4*sigma; went larger to be safe

## ---- array to hold results ----------------------------------
   output <- array(dim = c(nspacing, noccasions, 3, nrepl))
#   output.SE <- array(dim = c(nspacing, noccasions, nrepl))
#   output.CV <- array(dim = c(nspacing, noccasions, nrepl))
   dimnames(output) <- list(spacing, occasions, c("est","se","cv"),NULL)
#   dimnames(output.SE) <- list(spacing, occasions, NULL)
#   dimnames(output.CV) <- list(spacing, occasions, NULL)

   cat('Starting simulations', date(), '\n')
   flush.console()
## ---- loop over replicates, spacing, and noccasions ------------------
   for (r in 1:nrepl) {
      for (sp in 1:nspacing) {
         grid <- make.grid (nx = nx, ny = ny, spacing = spacing[sp])
         for (nocc in 1:noccasions) {
```

```
          temppop <- sim.popn (grid, D = D, buffer = buff)
          tempCH <- sim.capthist (grid, popn = temppop,
             detectfn = 0, noccasions = occasions[nocc],
             detectpar = list(g0 = g0, sigma = sigma))

## bracketing with try() allows us to continue if there is an error in secr.fit
          tempfit <- try (secr.fit (tempCH, detectfn = 0, buffer = buff,
             trace = FALSE, verify = FALSE,
             start = log(c(D,g0,sigma))), silent = TRUE)
          if (!inherits(tempfit, 'try-error')) {
             temppred <- unlist(predict(tempfit)['D',])
  ## here we save only the relative SE of D-hat...replace as desired
             output[sp,nocc,"est",r] <- temppred['estimate']
             output[sp,nocc,"se",r]<-temppred['SE.estimate']
             output[sp,nocc,"cv",r]<-temppred['SE.estimate'] / temppred['estimate']


          }
          else{
          cat("!\n")
          }
        }
      }
      cat('Completed replicate', r, date(), '\n')
      flush.console()
      save(output, file = outputfile)
      #save(output.SE, file = gsub("output","output.SE.",outputfile))
      #save(output.CV, file = gsub("output","output.CV.",outputfile))
    }
    output
 }



## ---- Output filename is NOT dynamic -------------------------------
runsim (nrepl = 25, outputfile = 'sim.output.D.005.g0.15.sigma.3000.25b.RData')



##---- Convert sigma to home range in km2 -----------------------------
for (hr in 1:length(sigs)) {
 homerange = ((sigs*(qchisq(0.95,2)^0.5))^2)*3.1415}
homerange/1000000
##---- NEBBTC secr simulation output extraction, formatting, and analysis -----------------------
##---- Manual file selection; mix of static and dynamic outputs ---------------------------------
##---- J. Stetz and M. Sawaya - last modified 05 August 2012 -------------------------------------

require(xlsx)

setwd('C:/Sinopah/NEBBTC/Simulations/secr/Results');
getwd()

##---- Provide filename within parantheses; influde suffix ".RData" ------------------------------
```

```
filename="sim.output.D.001.g0.05.sigma.1000.10.RData"
load(filename)
dim(output)
dimnames(output)
id<-dimnames(output)
spacing=id[[1]]
occ=id[[2]]
rep=dim(output)[3]
for(i in 1:length(spacing)){
        for(j in 1:length(occ)){

                        new.rows=data.frame(spacing=spacing[i],occasions=occ[j],
                        rep=1:25,estimate=output[spacing[i],occ[j],"est",],
                        SE=output[spacing[i],occ[j], "se",],
                        CV=output[spacing[i],occ[j], "cv",])
                if(exists("out.table")){
                        out.table=rbind(out.table,new.rows)
                }
                else{
                        out.table=new.rows

                }
        }
}

##---- Name components coming from file name --------------------------------------------------------
##---- Requries that the naming convention stays the same -------------------------------------------
##---- The "\\" are to take the special meaning out of the period -----------------------------------
##---- Note that there are some factors (vs. numeric); changed later --------------------------------

out.table$D=unlist(strsplit(filename,"\\."))[[4]]
out.table$g0=unlist(strsplit(filename,"\\."))[[6]]
out.table$sigma=unlist(strsplit(filename,"\\."))[[8]]
out.table$gridsize=unlist(strsplit(filename, "\\."))[[9]]
out.table$filename=(filename)
d = unlist(strsplit(filename,"\\."))[[4]]
g = unlist(strsplit(filename,"\\."))[[6]]
sig = unlist(strsplit(filename,"\\."))[[8]]
gr = unlist(strsplit(filename, "\\."))[[9]]
strsplit(filename,"\\."); sapply(out.table,"class")

##---- Converting whole columns, hence use of sapply function --------------------------------------

#formatC(out.table$est, digits=4, format="f", flag=0, ignoreNA=T, zero.print=T)
#initial attempt to reduce number of decimals; probably a problem with NAs?

out.table[,sapply(out.table,"class")=="factor"] <-
sapply(out.table[,sapply(out.table,"class")=="factor"],"as.character")
out.table[, c(7:10)] <- sapply(out.table[, c(7:10)], as.numeric)
```

out.table$D=(out.table$D/1000); out.table$g0=(out.table$g0/100)

##---- Convert 'Inf' SE's to 'NA' for calculating CIs and/or confidence interval coverage ('CIC') --
is.na(out.table$SE)=!is.finite(out.table$SE)
out.table$PRB=((out.table$estimate-out.table$D)/out.table$D)
out.table$lowCI=(out.table$estimate-
(1.96*out.table$SE));out.table$upperCI=(out.table$estimate+(1.96*out.table$SE))

#out.table$CIC=ifelse((out.table$estimate<out.table$upperCI)&(out.table$estimate>out.table$lowCI), 1,
0)

out.table$CIC2=ifelse(((out.table$estimate-
(1.96*out.table$SE))&(out.table$estimate+(1.96*out.table$SE))),1,0)

#out.table

##---- First save output as RData file, then .xlsx ---

save(out.table,   file=paste("D",d,"g0",g,"sigma",sig,"grid",gr,"frmtd.RData",sep="."))

##---- Save output to .xlsx file; static destination file name --------------------------------------
##---- Destination .xlsx has to be created first (one time); worksheets added after that -------------

filespot <- ("C:/Sinopah/NEBBTC/Simulations/secr/Results/NEBB.secr.sims.output.summary.10b.xlsx")

##---- Dynamic worksheet name; worksheet added to common destination file ----------------------
##---- R will return an error if worksheet with same name exists or if file is open -----------------
SaveExcel <- write.xlsx(out.table, filespot, sheetName=paste("D",d,"g0",g,"sigma",sig,"grid",gr,sep="."),

        col.names=T, row.names=F, append=T)

Appendix H: MATLAB Code for Demographic Analysis Simulations

% Lamvaresti.m: a program to estimate the sampling variance in log stochastic lambda
% using approximation formulae from Doak et al. 2005 (equation numbers refer to this
% paper and its appendix 3).
% Further modified from code provided by R. Harris for NEBBTC Technical Report
% J.Stetz and M.Sawaya - last modified 24 FEb 2013

% You must have the symbolic math toolbox of Matlab to use this program.
% This program uses two functions (secder.m and eigenall.m) from the website of
% programs that accompany Morris and Doak (2002): www.sinauer.com/PVA/
% The general form of data entry used here is quite similar to other, simpler
% programs also on this website, including Vitalsens.m and Stochsens.m; reading
% through these programs may help you understand the structures used here if you
% having trouble.

% One warning: the symbolic logic routines and the simulations to estimate correlations
% in beta variable means and variances are time-consuming, with one to several minutes
% between different steps. Be patient.

% 25 age classes; 6 vital rates (4 survival, 1 fecundity) - zero correlation throughout


```
clear all;
global yrsam kknums mmnums     % global variables used by called functions
randn('state',sum(100*clock)); % seeding random numbers
rand('state',sum(100*clock));  % seeding random numbers
warning off               % MATLAB:divideByZero
```

%**************** Parameters that must be input by user **********************
% First, give symbolic names for each vital rate to be used in this program. For the desert tortoise,
% these are: first, six survival rates (for stages 2-7); next, 5 growth rates (stages 2-6);
% and, finally, three fecundities (stages 5-7).
% These symbolic definitions are given below, and then the vector of these names (Svr) is defined.

```
syms  v1 v2 v3 v4 v5  % vital rates as symbolic variables
Svr = [v1 v2 v3 v4 v5 ]; % vector of symbolic vital rates
```

% Next, give the mean Vital rate values:

% CUB surv(v1)=0.87; YEARLING surv(v2)=0.9; SUBADULT (2-3yr) surv(v3)=0.82;
% ADULT (4+) surv(v4)= 0.92; fecALL(v5)= 0.312 [; fec6+(v5) = 0.350] (note these are

% fx, not mx values)

% NOTE! INCLUDES ONLY ONE FECUNDITY!!

%Fx = total # offspring produced during x.
%mx = fecundity: mean # offspring produced

realvrmeans = [0.73   0.87   0.933   0.93   0.6875];

% Then estimated true temporal variances (not standard deviations) of the Vital Rates:

realvrvars= [0.00493   0.00435 0.0036   0.00271   0.020];

% Next, you must say what the distribution is for each vital rate: this program only distinguishes
% between beta-distributed variables (coded as 1) and all others, assumed to be fecundities
% or similarly distributed parameters (coded 2).
% I interpret this as survival rates (n=4) are betas, whereas fx (n=2) are
% fecundities
vrtypes= [ones(1,4),2];
% Then, you must give a the full estimated matrix of temporal correlations between the vital rates.
% We do this here by putting the matrix for the desert tortoise directly in the code (see also
% Table 8.2 in Morris and Doak 2002). You could also load a matlab binary data file that has your
% correlation matrix.
% Harris comment: Seems to me for NCDE griz this should be a 6 x 6 matrix of zeros
% Kept as all zeros; justification discuss in Harris et al 2011 a little (ie, lack of evidence
% in the literature for covarying vital rates, despite logical likelihood
realcorrmx = [0 0 0 0 0; 0 0 0 0 0; 0 0 0 0 0; 0 0 0 0 0; 0 0 0 0 0];

 % Define how the different vital rates combine to make each matrix element, doing this
% by defining the entire symbolic matrix:
% 25 x 25 Leslie matrix for NEBBTC black bears (25 years based on Beston 2011, from Hebblewhite et al. 2003)

symmx = [0 0 0 v5 v5 v5 v5 v5 v5 v5 v5 v5 v5 v5 v5 v5 v5 v5 v5 v5 v5 v5 v5
    v1 0 00 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0
    0 v2 00 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0
    0 0 v3 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0
    0 0 0 v3 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0
    0 0 00 v4 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0
    0 0 00 0 v4 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0
    0 0 00 0 0 v4 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0
    0 0 00 0 0 0 v4 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0
    0 0 00 0 0 0 0 v4 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0
    0 0 00 0 0 0 0 0 v4 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0
    0 0 00 0 0 0 0 0 0 v4 0 0 0 0 0 0 0 0 0 0 0 0 0 0
    0 0 00 0 0 0 0 0 0 0 v4 0 0 0 0 0 0 0 0 0 0 0 0 0
    0 0 00 0 0 0 0 0 0 0 0 v4 0 0 0 0 0 0 0 0 0 0 0 0
    0 0 00 0 0 0 0 0 0 0 0 0 v4 0 0 0 0 0 0 0 0 0 0 0
    0 0 00 0 0 0 0 0 0 0 0 0 0 v4 0 0 0 0 0 0 0 0 0 0
    0 0 00 0 0 0 0 0 0 0 0 0 0 0 v4 0 0 0 0 0 0 0 0 0

```
0 0 00 0 0 0 0 0 0 0 0 0 0 0 0 v4 0 0 0 0 0 0 0 0
0 0 00 0 0 0 0 0 0 0 0 0 0 0 0 0 v4 0 0 0 0 0 0 0
0 0 00 0 0 0 0 0 0 0 0 0 0 0 0 0 0 v4 0 0 0 0 0 0
0 0 00 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 v4 0 0 0 0 0
0 0 00 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 v4 0 0 0 0
0 0 00 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 v4 0 0 0
0 0 00 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 v4 0 0
0 0 00 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 v4 0 ];
```

% Now, what are the sampling intensities for each vital rate and the durations of sampling that
% you want to have run calculations for? insams is a matrix with columns of sampled number of
% individuals used to estimate each vital rate (in the same order as for the means and variances
% above) and rows for different sets of these samples to run. For example, the insams defined
% below has one set of sampling of 30 individuals for each vital rate, and one set of sampling 100
% individuals for each rate; remember that these sampling patterns can be those used or ones you
% might want to consider.

insams = [10 10 10 10 10; 30 10 10 10 10; 10 30 10 10 10; 10 10 30 10 10; 10 10 10 30 10; 10 10 10 10
30; 30 30 10 10 30; 30 30 30 30 30];

% Then input each sampling duration that you want to consider: each number here is one duration to try:

yrsams = [3 5 10 20];

% Rename output each time

outputfilename = 'NEBBTC_Var8.txt'; % The name of the file to save output data to

%*************** End of Parameter inputs: Proceeding to calculations
****

%First Step: Basic calculations and estimation of the deterministic vital rate sensitivities
estiouts=[];   % The variable to store output data
realmx = subs(symmx,Svr,realvrmeans);   % Making a matrix of the mean numerical values
nmx = length(realmx);   % Size of pop mx.
nvr = length(realvrvars);   % Number of vital rates

[lambdas,lambda1,W,w,V,v]= eigenall(realmx);   % Use eigenall.m to get eigenvalues
sensmx = v*w'/(v'*w);   % Get sensitivities of matrix elements
vrsens = zeros(1,nvr);   % Initialize vital rate sens.
for xx=1:nvr % A loop to calculate sensitivity for each vital rate
        % First get derivatives of elements with respect to vital rates:
        diffofvr = double(subs(diff(symmx,Svr(xx)),Svr,realvrmeans));
   vrsensbyelements(:,:,xx) = diffofvr;
   % Then, sum up to get row of total vital rate sensitivities:
        vrsens(xx) = double(sum(sum(sensmx.*diffofvr)));
end; % xx

% Second Step: Calculate stochastic lambda and its sensitivities to the matrix element means
mx = realmx; % Set mx equal to the name of stored pop'n matrix

```
vrcovmx = realcorrmx.*(sqrt(realvrvars')*sqrt(realvrvars)); % Make a covariance matrix
tau=(vrsens)*vrcovmx*(vrsens'); % tau as in Tuljapurkar (1991), but estimated by vital rates
% Estimate  log(lambda_S), the log of stochastic lambda:
loglamS = log(lambda1) - 0.5*(1/(lambda1^2))*tau;

squloglamderivs=[]; % Here, we are define the three storage variables for the final calcs:
squVarsums =[];
squCorrsums = [];

for ii=1:nvr % Loop to get the values needed to estimate the derivatives: d(log(lambda_S))/d(vi)
   kkllsum=0;
   for kk=1:nvr
      for ll = 1:nvr
         dSldi =0;
         dSkdi =0;
dSldi = sum(sum( sensmx.*double(subs(diff(diff(symmx,Svr(ll)),Svr(ii)),Svr,realvrmeans)) ));
dSkdi = sum(sum( sensmx.*double(subs(diff(diff(symmx,Svr(kk)),Svr(ii)),Svr,realvrmeans)) ));
         for xx = 1:nmx
            for yy =1:nmx
dSldi = dSldi + vrsensbyelements(xx,yy,ii)*sum(sum(secder(mx,xx,yy).*vrsensbyelements(:,:,ll) ));
dSkdi = dSkdi + vrsensbyelements(xx,yy,ii)*sum(sum(secder(mx,xx,yy).*vrsensbyelements(:,:,kk) ));
            end
         end
         kkllsum= kkllsum + vrcovmx(kk,ll)*(dSldi*vrsens(kk) +dSkdi*vrsens(ll));
      end
   end
   % The derivatives of log(lambda_S) with respect to each vital rate:
   loglamderivs(ii) =     vrsens(ii)/lambda1 + vrsens(ii)*tau/(lambda1^3) - kkllsum/(2*lambda1^2);
   % The square of each derivative, which multiples with the variance in each rate in equation 2.
   squloglamderivs(ii) = (loglamderivs(ii))^2;
   % The sums that multiple with the variances of the variances terms in equation 2:
   squVarsums(ii) = (1/lambda1^4)*(  sum( vrsens(ii)*vrsens.*sqrt(realvrvars).*realcorrmx(ii,:)) )^2;

    disp('The vital rate number and sensitivity of log(lambda_S) to this vital rate');
    disp([ii,loglamderivs(ii)]);
end; %ii
% Finally, the matrix of values that multiple with the variances of correlations in equation 2:
squCorrsums = (1/lambda1^4)*((sqrt(realvrvars')*sqrt(realvrvars)).*(vrsens'*vrsens)).^2;
clear v1 v2 v3 v4 v5 Svr symmx; %making space in memory

% Third Step: estimate sampling variance in log(lambda_S)for different sampling patterns
for ii = 1:length(insams(:,1))% Loop through each set of sampling intensities
   SamNs = insams(ii,:); % The vector of within year sample sizes to use
   for jj=1:nvr        % A loop to use simulation to estimate the correlation of means and standard
      if vrtypes(jj) ==1  %  deviations in sampled values for beta-distributed variables:
         mn = realvrmeans(jj);
         va = realvrvars(jj);
         vv = mn*((mn.*(1-mn)/(va))-1); % calculate the beta parameters
         ww = (1-mn).*((mn.*(1-mn)/(va))-1);
          aa = betarnd(vv,ww,SamNs(jj),10000); % Draw 10,000 sets of values
```

```
        aavars = var(aa);
        aaSD= sqrt(aavars);
        aameans=mean(aa);

        aacov=cov([aaSD',aameans']);
        vrvrvarcovs(jj) = aacov(1,2);
    else vrvrvarcovs(jj)=0;
    end;
    betacorrcontribut(jj) = 2*vrvrvarcovs(jj).*loglamderivs(jj).*(1/lambda1^2).*(sum( ...
        vrsens(jj)*vrsens.*sqrt(realvrvars(jj)).*realcorrmx(jj,:)) );
    disp('The vital rate number and, next line, beta-value correlation contribution to variance');
    disp(jj); disp(betacorrcontribut(jj));

  end;
  clear aa aavars aaSD aameans aacov; % making space in memory
  for yy=1:length(yrsams); % Loop through the sampling durations
    yrs = yrsams(yy); % number of years of data

    for xx=1:nvr %loop to estimate  within-year sampling variances of each vital rate:
        if vrtypes(xx) == 1;  inyrvar(xx) = realvrmeans(xx)*(1-realvrmeans(xx));  end; % binomials
        if vrtypes(xx) == 2;  inyrvar(xx) = realvrmeans(xx);   end; % using Poisson variance for
fecundities
    end
    % Next, estimate the total sampling variance for mean values (equation A6):
    meanvars = (1/yrs).*(realvrvars + inyrvar./SamNs);
    % Then, the variances for the corrected variance estimates (equation A9):
    correctedvarvars = (2*yrs/(yrs-1)^2)*realvrvars.*(realvrvars + 2*(inyrvar./SamNs));
    SDvars = (correctedvarvars./(4.*realvrvars)); % Transform correctedvarvars to get variances of SDs
    SDvars(isnan(SDvars)) = 0;
    correlvars = (yrs/(yrs-1)^2)*(realcorrmx.^2 -1).^2; % The variances of the correlations

    % At Last, get the outputs:
    % 1. The sampling variance in the estimate of deterministic log(lambda): this is also the
    %     sampling variance in log(lambda_S) generated by sampling variance of the mean vital rates:
    DeterLogLamVar = sum(squloglamderivs.*meanvars);
    % 2. Sampling variance of log(lambda_S) from just variance in means and variances of vital rates:
    VarLogLamVar   = sum(squloglamderivs.*meanvars + squVarsums.*SDvars );
    % 3. Sampling variance of log(lambda_S) from variances of means, variances, and correlations,
    %     but without the effects of beta variable correlations
    FullLogLamVar = sum(squloglamderivs.*meanvars + squVarsums.*SDvars +
0.5*sum(squCorrsums.*correlvars) );
    % 4. The best of sampling variance of log(lambda_S) with the effects of beta variable correlations
    FullLogLamVarADDED = FullLogLamVar+sum(betacorrcontribut);

    % Save the data: as now written, the outputs are one row for each combination of sampling duration
and
    % intensity. The columns of data are: sampling intensity for the first vital rate; sampling
    % duration; sampling variance (SV) for deterministic log(lambda); SV for log(lambda_S) from SV
in
```

% vital rate means and variances; SV for log(lambda_S) from SV in means, variances, and correlations;
    % SV for log(lambda_S) from all sources; estimated log(lambda_S) for the input parameters; and,
    % estimated log(deterministic lambda).
    estiouts = [estiouts;[SamNs(1) yrs DeterLogLamVar VarLogLamVar FullLogLamVar FullLogLamVarADDED ...
            loglamS log(lambda1) meanvars SDvars]];
    disp('The sampling intensity set, sampling duration set, and sampling variance in log(lambda_S)');
    disp([ii,yy, FullLogLamVarADDED]);
   end; %yy
end; %ii
save(outputfilename, 'estiouts','-ASCII'); % This saves a file with the data in estiouts
disp('DONE!');