

FOREST WILDLIFE POPULATIONS

Forest Wildlife Populations and Research Group

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CARNIVORE SCENT STATION SURVEY SUMMARY, 2019

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INTRODUCTION

Monitoring the distribution and abundance of carnivores can be important for understanding the effects of harvest, habitat change, and environmental variability on these populations. However, many carnivores are highly secretive, difficult to repeatedly capture, and naturally occur at low to moderate densities, making it difficult to annually estimate abundance over large areas using traditional methods (e.g., mark-recapture, distance sampling, etc.). Hence, indices of relative abundance are often used to monitor such populations over time (Sargeant et al. 1998, 2003, Hochachka et al. 2000, Wilson and Delahay 2001, Conn et al. 2004, Levi and Wilmers 2012).

In the early 1970's, the U.S. Fish and Wildlife Service initiated a carnivore survey designed primarily to monitor trends in coyote populations in the western U.S. (Linhart and Knowlton 1975). In 1975, the Minnesota DNR began to utilize similar survey methodology to monitor population trends for numerous terrestrial carnivores within the state. This year marks the 45th year of the carnivore scent station survey.

METHODS

Scent station survey routes are composed of tracking stations (0.9 m diameter circle) of sifted soil with a fatty-acid scent tablet placed in the middle. Scent stations are spaced at 0.5 km intervals on alternating sides of a road or trail. During the initial years (1975-82), survey routes were 23.7 km long, with 50 stations per route. Stations were checked for presence of tracks on 4 consecutive nights (old tracks removed each night), and the mean number of station visits per night was the basis for subsequent analysis. Starting in 1983, following suggestions by Roughton and Sweeny (1982), design changes were made whereby routes were shortened to 4.3 km, 10 stations/route (still with 0.5 km spacing between stations), and routes were surveyed only once on the day following route placement. The shorter routes and fewer checks allowed for an increase in the number and geographic distribution of survey routes. In either case, the design can be considered two-stage cluster sampling.

Survey routes were selected non-randomly, but with the intent of maintaining a minimum 5 km separation between routes, and encompassing the variety of habitat conditions within the work area of each survey participant. Most survey routes are placed on secondary (unpaved) roads or trails and are completed from September through October. Survey results are currently stratified based on 3 habitat zones within the state (forest (FO), transition (TR), and farmland (FA); Figure 1).

Track presence is recorded at each station and track indices are computed as the percentage of scent stations visited by each species. Confidence intervals (95%) are computed using bootstrap methods (percentile method; Thompson et al. 1998). For each of 1000 replicates, survey routes are randomly re-sampled according to observed zone-specific route sample sizes,

and station visitation rates are computed for each replicate sample of routes. Replicates are ranked according to the magnitude of the calculated index, and the 25th and 975th values constitute the lower and upper bounds of the confidence interval.

RESULTS AND DISCUSSION

A total of 179 routes and 1,638 stations were surveyed this year, the fewest since the survey became fully operational in the early 1980's. Route density varied from 1 route per 901 km² in the Forest Zone to 1 route per 1,660 km² in the Farmland Zone (Figure 1). The recent decline in survey effort is likely a result of staffing shortages and competing workload demands.

Statewide, route visitation rates (% of routes with detection), in order of increasing magnitude, were opossums (4%), domestic dogs (12%), bobcats (13%), wolves (14%), domestic cats (28%), red foxes (30%), raccoons (32%), skunks (35%), and coyotes (36%). Regionally, route visitation rates were as follows: red fox – TR 22%, FA 24%, FO 37%; coyote – FO 18%, TR 39%, FA 61%; skunk – FO 24%, TR 41%, FA 54%; raccoon – FO 13%, TR 35%, FA 71%; domestic cat – FO 9%, TR 43%, FA 54%; domestic dog – FO 3%, TR 17%, FA 27%; opossum - FO 0%, TR 4%, FA 12%; wolf - FA 0%, TR 0%, FO 27%; and bobcat - FA 0%, TR 15%, FO 17%.

Figures 2-5 show station visitation indices (% of stations visited) from the survey's inception through the current year. Although the survey is intended to document long-term trends in populations, confidence intervals (CI) improve interpretation of the significance of any annual changes. However, I refrain from formal significance testing (e.g., determination of whether a CI on the difference between means overlaps 0) and instead use more informal methods (i.e., degree of CI overlap; Cumming and Finch 2005) to highlight changes from last year that likely represent significant differences. Using this approach, the only notable changes this year were increases in bobcat and striped skunk indices in the Transition Zone (Figures 3 and 5).

In the Farmland Zone (Figure 2), red fox indices continue to remain well below their long-term average, as they have for nearly 20 years. Conversely, coyote and raccoon indices continue their increasing trend and are at or near record levels. Low red fox numbers are likely related, in part, to the increased coyote abundance (Levi and Wilmers 2012). No consistent trends are evident for other species in the Farmland Zone over the long term.

Similar to the Farmland, red fox and coyote indices have primarily exhibited inverse patterns in the Transition Zone, with red fox indices remaining low and coyote indices steadily increasing (Figure 3). Although there was a significant increase this year in the striped skunk index in the Transition Zone, long-term data do not show any consistent trend, with current indices near their long-term average. In spite of the large CI for bobcats in the Transition Zone, results suggest a marginally significant increase from last year and a moderate increase over the past decade (Figure 5). Raccoon indices have been comparatively stable and near their long-term averages over the past 2 decades. Wolves had exhibited a mild increase in the Transition Zone over time, but indices have been below the long-term average (and at or near 0) the past 3 years.

No significant changes were observed in the Forest Zone (Figures 4 and 5). Unlike in the Farmland and Transition Zones, the Forest Zone coyote index has not increased over time and has been below average and stable for 2 decades, likely attributable to increased wolf abundance in the Forest Zone (Levi and Wilmers 2012). Red foxes, raccoons, and skunks have not exhibited consistent or notable trends over the past 20 years and all remain near or slightly below their long-term averages. Conversely, wolves and bobcats have exhibited increasing trends over the past 2 decades, though some shorter-term declines have occurred during this period.

ACKNOWLEDGMENTS

I wish to thank all of the cooperators who participated in the 2019 survey: DNR Division of Wildlife staff; Superior National Forest Aurora District; Rydell National Wildlife Refuge; 1854 Treaty Authority, White Earth, Red Lake, and Leech Lake Tribal Natural Resource Departments; Lori Schmidt and Vermillion Community College; Peter Jacobson and Faribault High School; and Steven Hogg and the Three Rivers Park District. This project was funded in part by the Wildlife Restoration Program (Pittman-Robertson).

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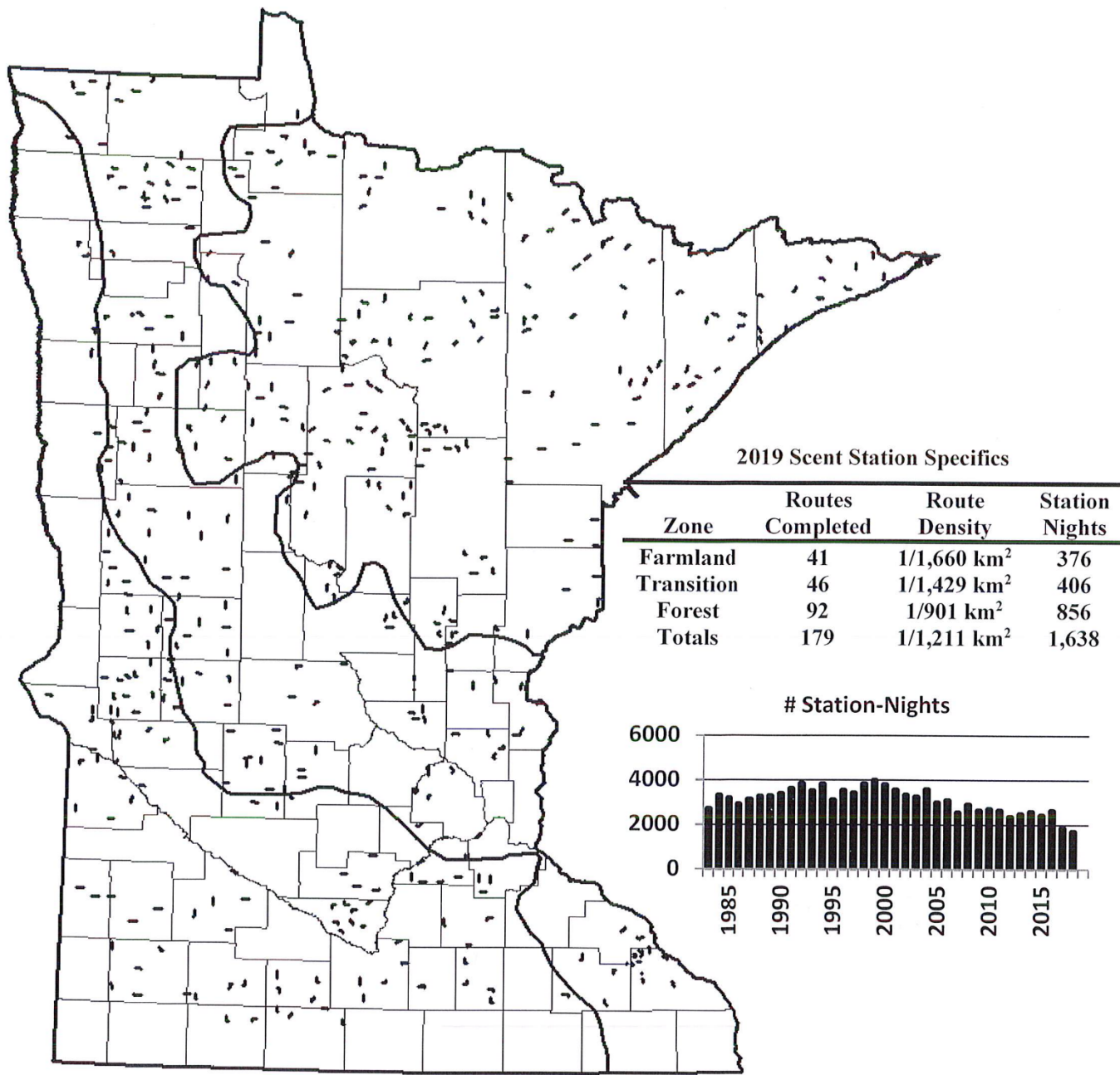


Figure 1. Locations of existing scent station routes (not all completed every year). Insets show 2019 route specifics and the number of station-nights per year since 1983.

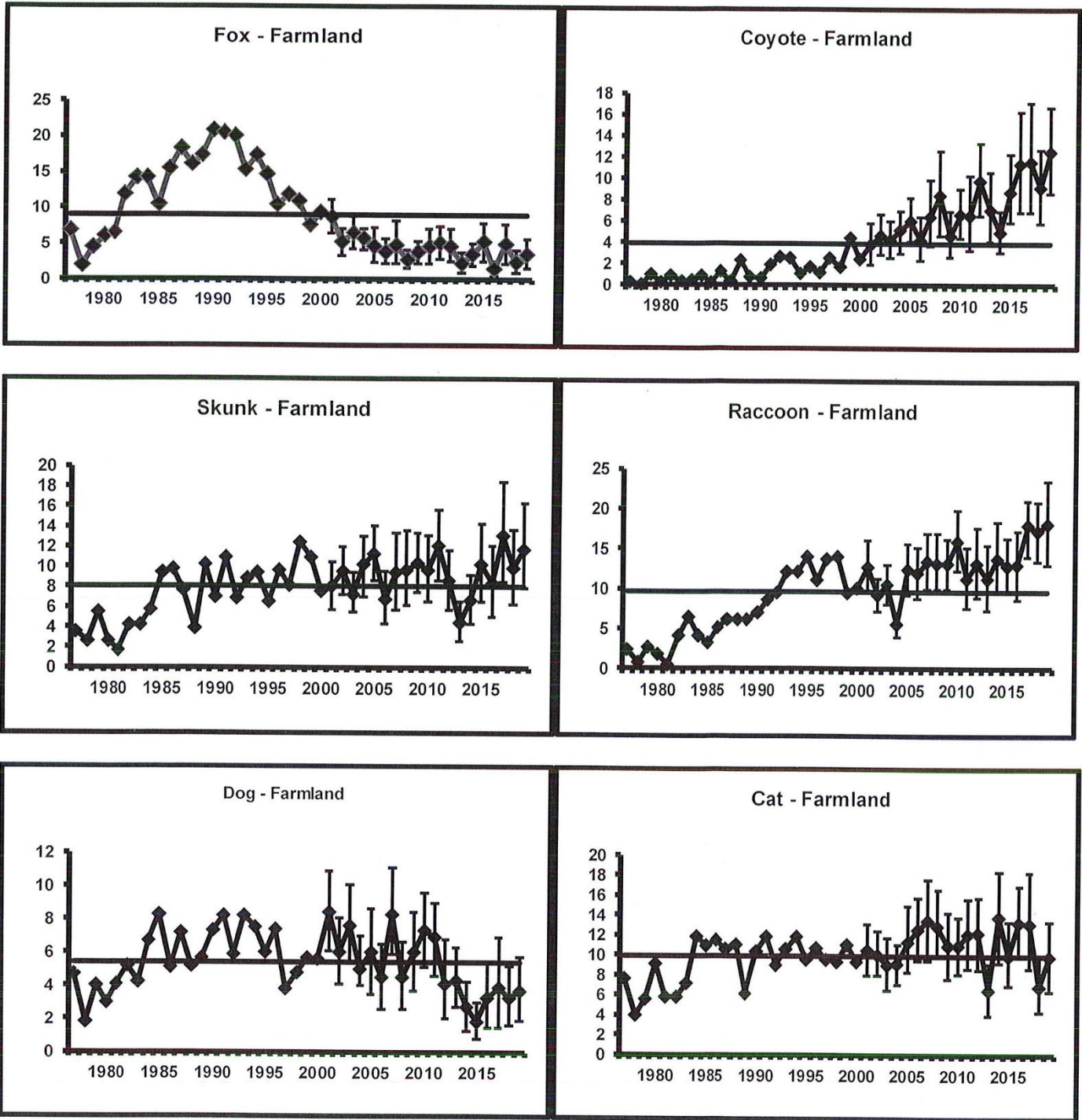


Figure 2. Percentage of scent stations visited by selected species in the Farmland Zone of Minnesota, 1977-2019. Horizontal line represents long-term mean.

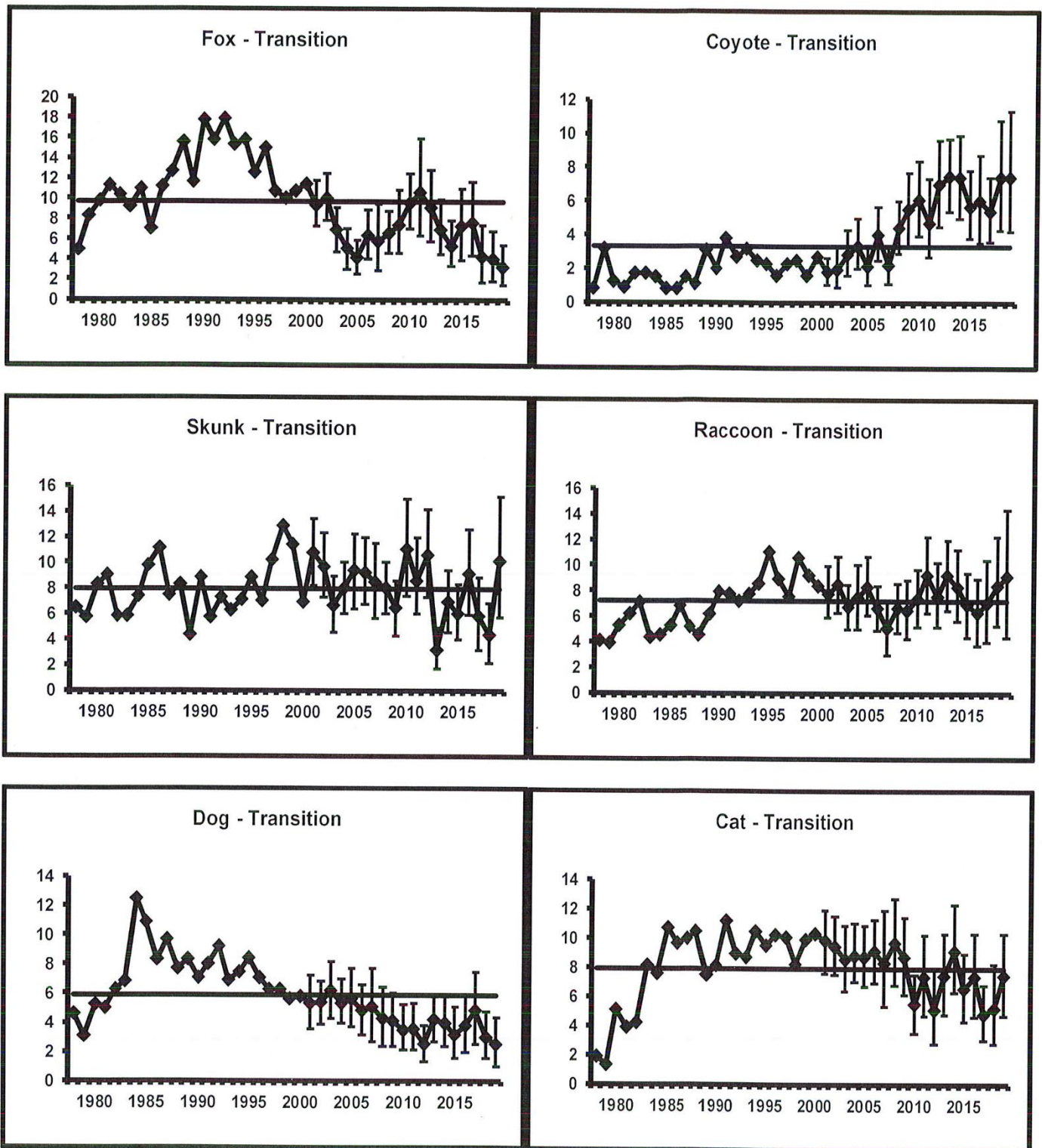


Figure 3. Percentage of scent stations visited by selected species in the Transition Zone of Minnesota, 1978-2019. Horizontal line represents long-term mean.

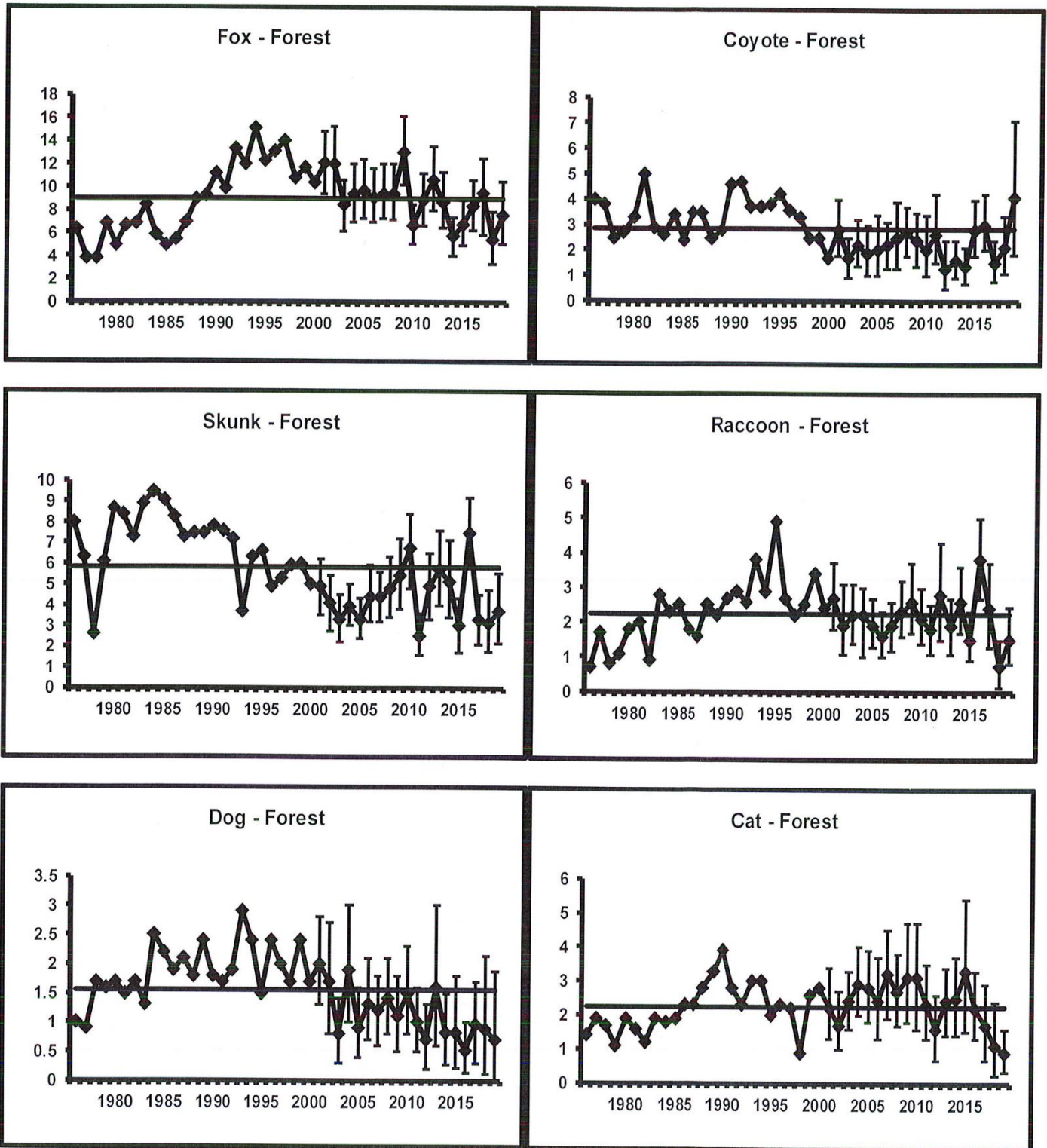


Figure 4. Percentage of scent stations visited by selected species in the Forest Zone of Minnesota, 1976-2019. Horizontal line represents long-term mean.

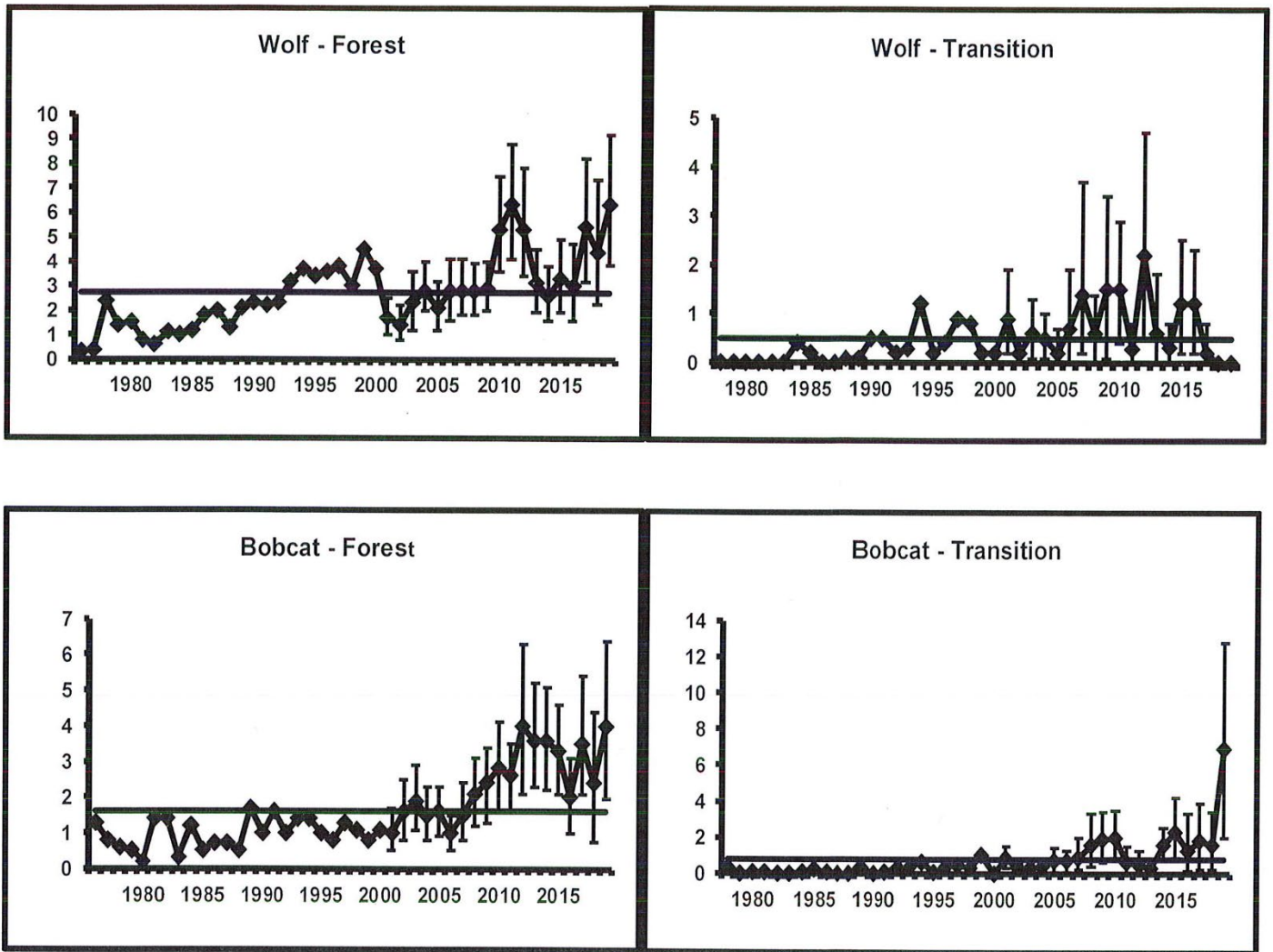


Figure 5. Percentage of scent stations visited by wolves and bobcat in the Forest and Transition Zones of Minnesota, 1976-2019. Horizontal lines represents long-term mean.



FURBEARER WINTER TRACK SURVEY SUMMARY, 2019

John Erb, Minnesota Department of Natural Resources, Forest Wildlife Research Group

INTRODUCTION

Monitoring the distribution and abundance of carnivores can be important for documenting the effects of harvest, habitat change, and environmental variability on their populations. However, many carnivores are highly secretive, difficult to repeatedly capture, and naturally occur at low to moderate densities, making it difficult to estimate abundance over large areas using traditional methods (e.g., mark-recapture, distance sampling, etc.). Hence, indices presumed to reflect relative abundance are often used to monitor populations over time (Hochachka et al. 2000, Wilson and Delahay 2001, Conn et al. 2004).

In winter, tracks of carnivores are readily observable following snowfall. Starting in 1991, Minnesota initiated a carnivore snow-track survey in the northern portion of the State. The survey's primary objective is to use a harvest-independent method to monitor distribution and population trends of fishers (*Pekania pennanti*) and martens (*Martes americana*), two species for which no other survey data is available. Because sign of other carnivores is readily detectable in snow, participants also record tracks for other selected species. After three years of evaluating survey logistics, the survey became operational in 1994. Formal recording of gray fox (*Urocyon cinereoargenteus*) detections did not commence until 2008.

METHODS

Presently, 57 track survey routes are operational across the northern portion of the state (Figure 1); for various reasons, not all are surveyed each year. Each route is 10 miles long and follows secondary roads or trails. Most routes are continuous 10-mile stretches of road or trail, but a few are composed of multiple discontinuous segments. Route locations were subjectively determined based on availability of suitable roads or trails, but were chosen when possible to represent the varying forest habitat conditions in northern Minnesota. For data recording, each 10-mile route is divided into 20 0.5-mile segments.

Each route is surveyed once following a fresh snow, typically from December through mid-February, and track counts are recorded for each 0.5-mile segment. When it is obvious the same animal crossed the road multiple times *within* a 0.5-mile segment, the animal is recorded only once. If it is obvious that an animal ran along the road and entered multiple 0.5 mile segments, which often occurs with canids, its tracks are recorded in all segments but circled to denote it was the same animal. Though these 'duplicate' tracks are not included in calculation of track indices (see below), recording data in this manner allows for future analysis of animal activity in relation to survey 'plot' size and habitat. Snowshoe hares (*Lepus americanus*) are recorded only as present or absent in the first 0.1 miles of each 0.5-mile segment. For standardization, routes are to be surveyed one day after the conclusion of a snowfall (ending by ~ 6:00 pm). However, in most years a few routes are completed two nights following snowfall; track counts on those routes are divided by the number of days post-snowfall.

Because most species of interest occur throughout the area where survey routes are located, calculated indices for all species prior to 2015 utilize data from all surveyed routes. Starting with

the 2015 report, all past marten indices were re-calculated using only those routes that fall within a liberal delineation of marten range (hereafter, the 'marten zone'). However, in general there were minimal differences in temporal patterns observed in this subset versus the full sample of routes.

Currently, I present three summary statistics for each species. First, I compute the percentage of 0.5-mile segments with species presence after removing any duplicates (e.g., if the same red fox (*Vulpes vulpes*) clearly traverses two adjacent 0.5-mile segments of the road, and it was the only 'new' red fox in the second segment, only one of the two segments is considered independently occupied). In addition to this metric, but on the same graph, the average number of tracks per 10-mile route is computed after removing any obvious duplicate tracks across segments. For wolves (*Canis lupus*) traveling through adjacent segments, I use the maximum number of pack members recorded in any one of those segments as the track total for that particular group, though this is likely an underestimate of true pack size. Because individuals from many of the species surveyed tend to be solitary, these two indices (percent of segments occupied and number of tracks per route) will often yield mathematically equivalent results; on average, one tends to differ from the other by a constant factor. In the case of wolf packs, and to a lesser extent red foxes and coyotes (*Canis latrans*) which may still associate with previous offspring or start traveling as breeding pairs in winter, the approximate equivalence of these two indices will still be true if average detected group sizes are similar across years. However, the solitary tendencies in some species are not absolute, potential abundance in relation to survey plot size varies across species, and for wolves, pack size may vary annually. For these reasons, as well as to provide an intuitive count metric, I include both indices on the same graph. Because snowshoe hares are tallied only as present or absent, the two indices are by definition equivalent. Dating back to 1974, hare survey data has also been obtained via counts of hares observed on ruffed grouse drumming count surveys conducted in spring. Post-1993 data for both the spring and winter hare indices are presented for comparison in this report.

In the second graph for each species, I illustrate the percentage of *routes* where each species was detected (hereafter, the 'distribution index'). I compute this measure to help assess whether any notable changes in the above-described track indices are a result of larger-scale changes in distribution (i.e., more or less routes with presence) or finer-scale changes in density along routes.

Using a bootstrapping approach (percentile method; Thompson et al. 1998), I compute confidence intervals (90%) for the percent of segments with species' presence and the percent of routes with species presence. For each of 1000 replicates, survey routes are randomly re-sampled with replacement according to the observed route sample size, replicates are ranked according to the magnitude of the calculated index, and the 50th and 950th values constitute the lower and upper bounds of the confidence interval. Although the survey is intended to document long-term trends in populations, confidence intervals (CI) improve interpretation of the significance of any annual changes. However, I refrain from formal significance testing (e.g., determination of whether a CI on the difference between means overlaps 0) and instead use an informal approach (i.e., degree of CI overlap; Cumming and Finch 2005) to highlight changes from last year that likely represent significant differences.

RESULTS AND DISCUSSION

This winter, 32 of the 57 routes were completed (Figures 1 and 2), the fewest since 2002. Survey routes took an average of 2.1 hours to complete. Snow depths averaged 16" along completed routes, the third deepest since the survey began (Figure 3). Mean overnight low temperature the night preceding the surveys was 2°F, slightly below to the long-term average (Figure 3). Survey routes were completed between November 29th and February 19th, with the mean survey date of January 3rd being close to the long-term average (Figure 3).

Reliable interpretation of changes in track survey results is dependent on the assumption that the probability of detecting animals remains relatively constant across years (Gibbs 2000, MacKenzie et al. 2004). Because this remains an untested assumption, caution is warranted when interpreting changes, particularly annual changes of low to moderate magnitude or short-term trends. Based on

degree of confidence interval overlap, notable changes from last winter included significant increases in the percentage of segments and routes where fishers were detected and the percentage of segments where weasels (*Mustela erminea* and *Mustela frenata*) were detected (Figure 4). Conversely, there were significant decreases in the percentage of routes and route segments where bobcats (*Lynx rufus*) were detected and the percentage of routes where coyotes were detected (Figure 4).

Fishers were detected on 5.8% of the route segments and along 69% of the routes (Figure 4). Over the past decade, fishers have expanded in distribution and abundance along the southern and western edge of their Minnesota range, an area currently with few or no track survey routes. Hence, fisher indices in this report are indicative of population trends in only the northern 'core' of fisher range. Although the observed increase from last year was significant, fisher indices have remained below their long-term average for the past 12 years, and far below the long-term peak around 2002; at their peak, fishers were detected on 14% of route segments and 78% of the survey routes.

Within the 'marten zone', martens were detected on 5.8% of the route segments and 48% of the survey routes (Figure 4); neither represented a significant change from last year. At their peak in 1999, martens were detected on 13% of the 'marten zone' route segments and 83% of the 'marten zone' survey routes. Similar to results for fishers, marten indices have declined over the long-term. Although low and without trend over the last 13 years, marten indices during this period do show indications of 3-5 year cycles, consistent in timing with cyclic fluctuations of some of their rodent prey species in Minnesota (e.g., Berg et al. 2017, Oestricher 2018).

Bobcat indices had increased for approximately 15 years through 2014, and then declined to their (now elevated) long-term average by 2016. Indices from 2016-18 showed a quick rebound from the recent decline. However, the percentage of segments with bobcat detection significantly declined this winter to the lowest levels since 1998; bobcats were detected on 1.5% of the segments and 25% of the routes. Snow depths along routes the past 2 winters were the 2nd and 3rd deepest since the survey began (Figure 3). High snow depths can reduce bobcat populations or movements (i.e., cause lower detection rates), either or both of which could explain the significant decline in bobcat track indices this winter. Bobcats appear to have a negative effect on fishers (Erb et al. 2017); the increase in fisher indices this winter suggests a potential decline in bobcat numbers (not just movement) as a partial explanation for the decline in bobcat indices.

Wolves were detected on 8.8% of the route segments and 88% of the survey routes, neither being significant changes from last year (Figure 4). The average number of wolves detected per route was 3. Coyotes were detected on 2.3% of the route segments and 22% of the routes, the latter constituting a significant decline from last winter. The long-term trend in coyote indices has been stable, but as with martens and weasels (see below), coyote winter indices appear to exhibit 3 - 5 year cycles consistent in timing with fluctuations in some rodent populations in MN. Long-term red fox indices display a 'stair-step' decline over time, being lowest and comparatively stable since 2012. Red foxes were detected on approximately 11% of the segments and 69% of the routes (Figure 4), neither being a significant change from last winter. Although it is premature to characterize longer patterns in gray fox detections, data from the past 12 years suggests that similar to coyotes, martens, and weasels, they may fluctuate in concert with cyclic rodent populations. Gray foxes were detected on 1% of the route segments and on 13% of the routes.

The percentage of route segments on which weasels were detected significantly increased from last year. However, fluctuations continue to be characterized by 4 to 5 year cycles or 'irruptions' superimposed on a long-term declining trend (Figure 4). Weasels were detected on 15% of the route segments (peak of 31% in 1995) and on 63% of the routes (peak of 88% in 1999).

There were no significant changes in the percentage of routes or route segments with hare detection. Both spring and winter hare indices steadily increased from 1994 - 2010, generally declined for five years, and have exhibited no trend for the past four years (Figure 4). Both spring and winter indices are near their post-1994 averages (Figure 4). Historic data (pre-1994; not

presented here) for the spring snowshoe hare index clearly exhibited 10-year cycles. Since then, only subtle signs of a cycle are apparent in both surveys during the first few years of each decade.

ACKNOWLEDGMENTS

I wish to thank all those who participated in this year's survey, including staff with the Minnesota DNR, Superior National Forest (Cook and Ely offices), Fond-du-Lac, and Red Lake Bands of Ojibwe, and the 1854 Treaty Authority. This project was funded in part by the Wildlife Restoration Program (Pittman-Robertson).

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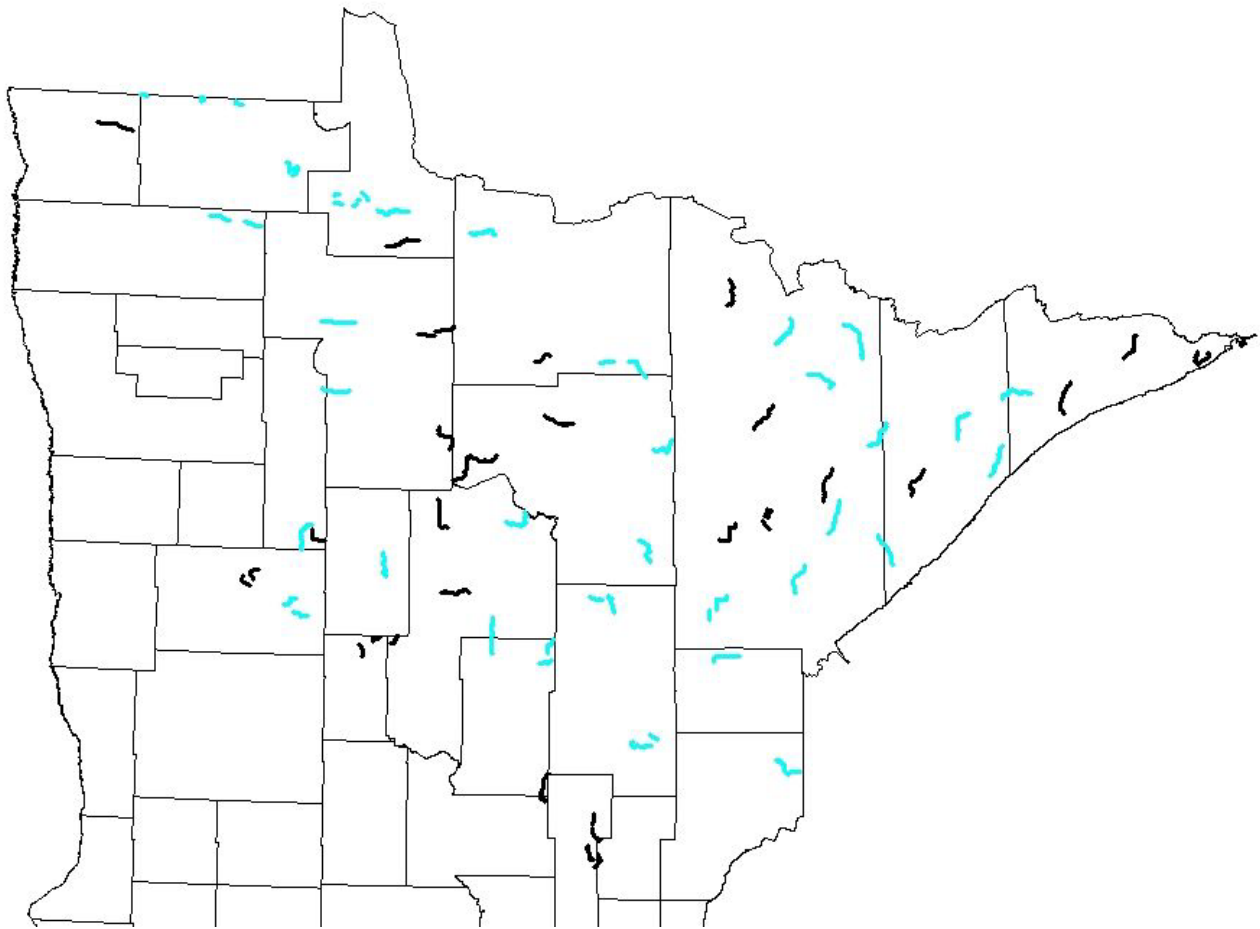


Figure 1. Locations of furbearer winter track survey routes in northern Minnesota. Blue routes are those completed during winter 2019-20.

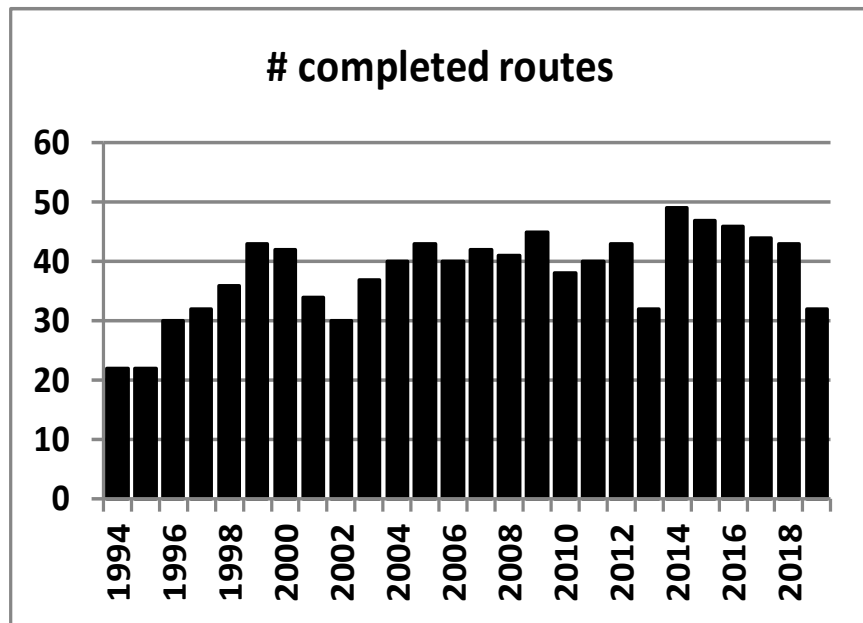


Figure 2. Number of snow track routes surveyed in Minnesota, 1994-2019.

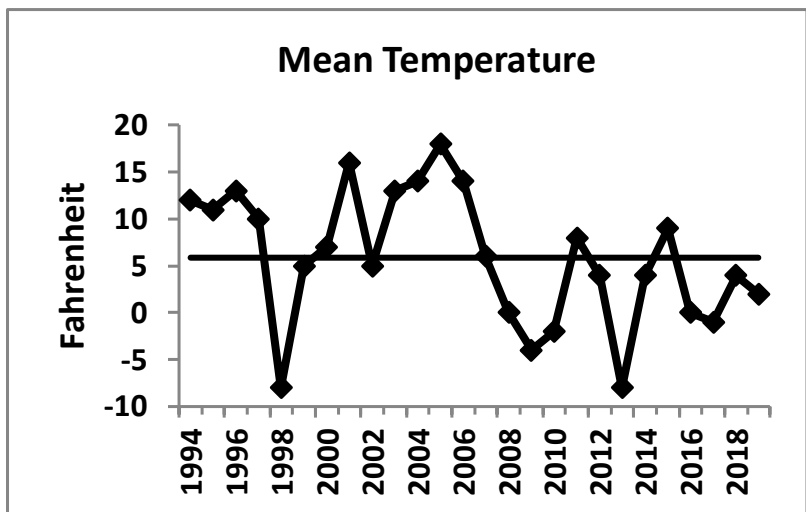
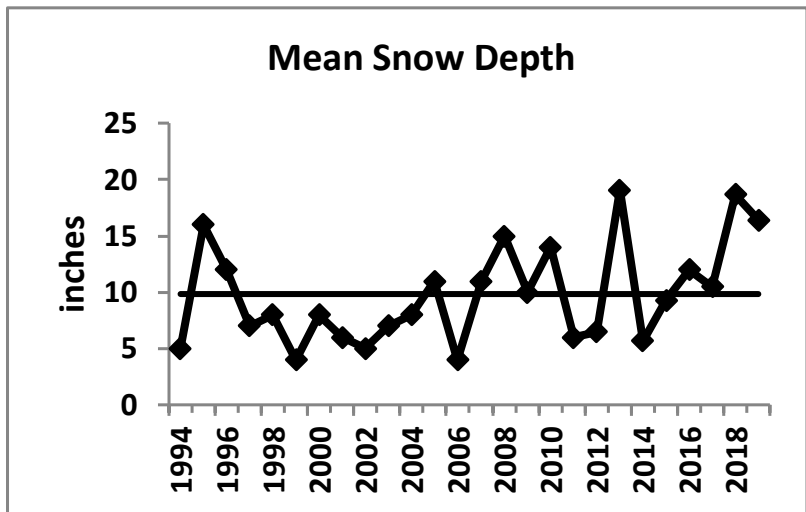
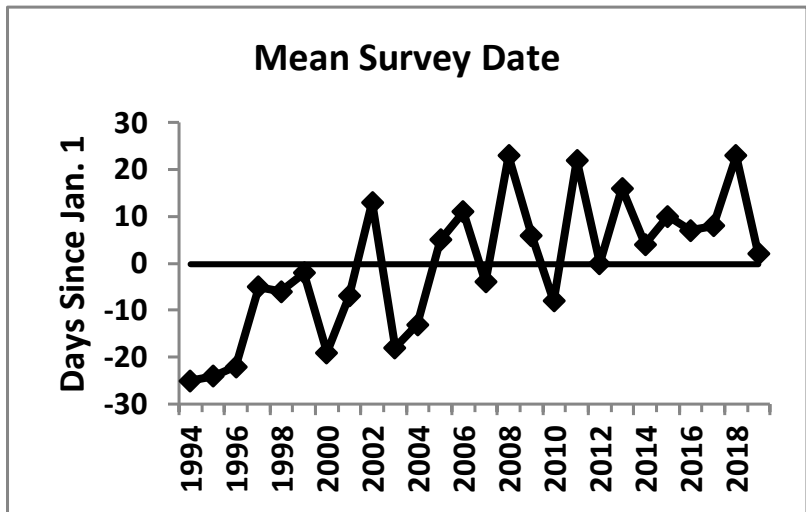


Figure 3. Average survey date, snow depth, and temperature for snow track routes completed in Minnesota, 1994-2019. Horizontal line represents long-term mean.

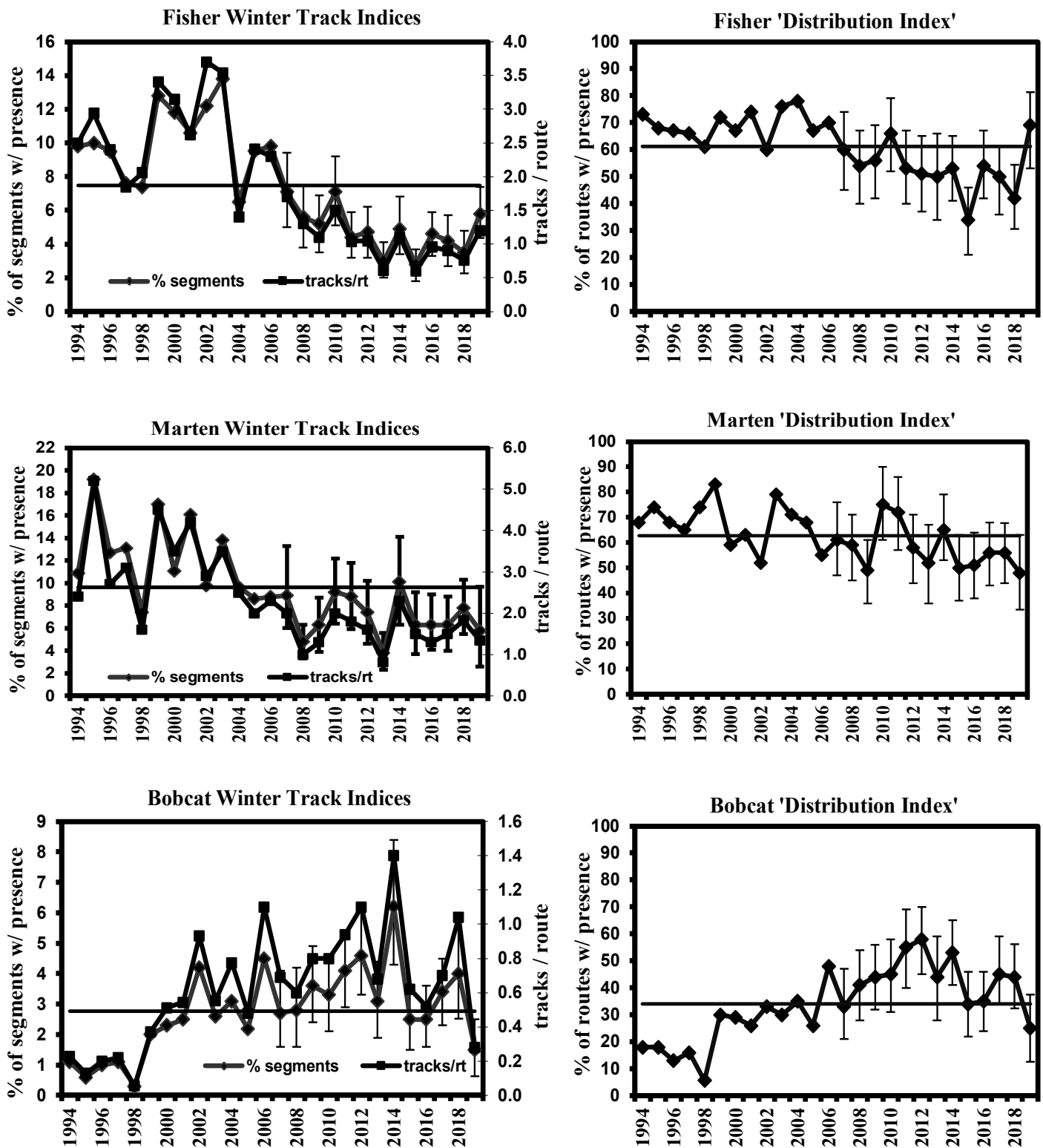


Figure 4. Winter track indices for selected species in Minnesota, 1994-2019. Confidence intervals are presented only for % segments and % routes with track presence; horizontal lines represent their long-term averages.

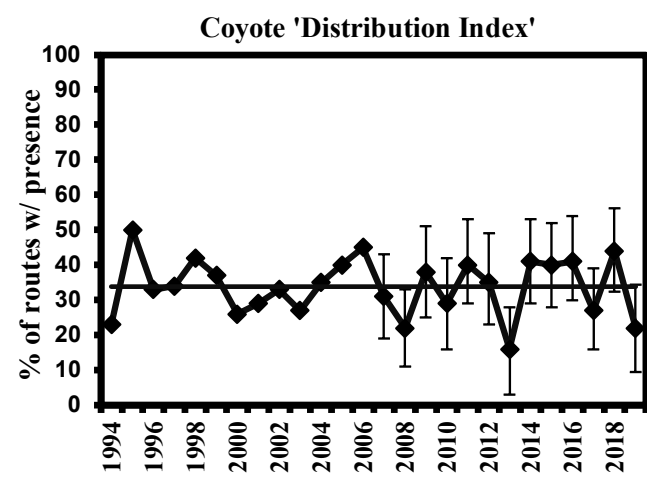
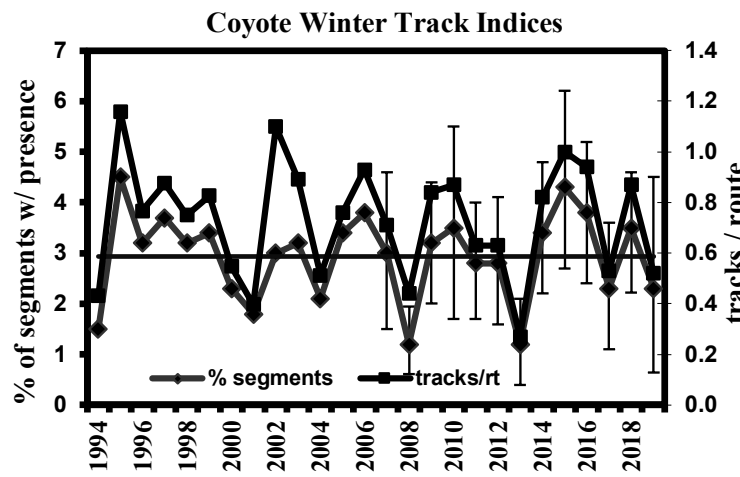
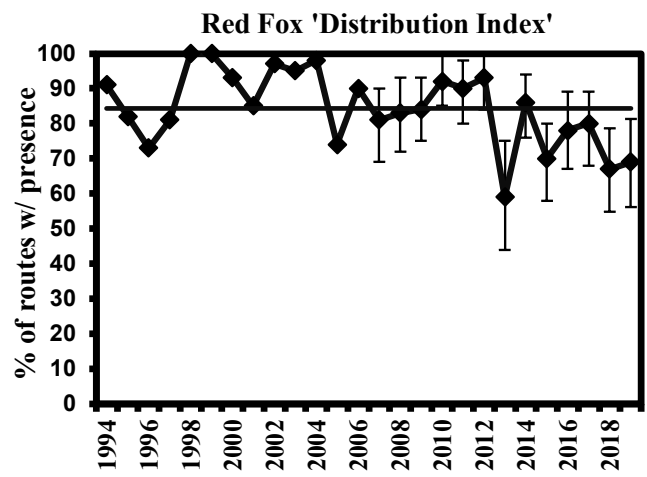
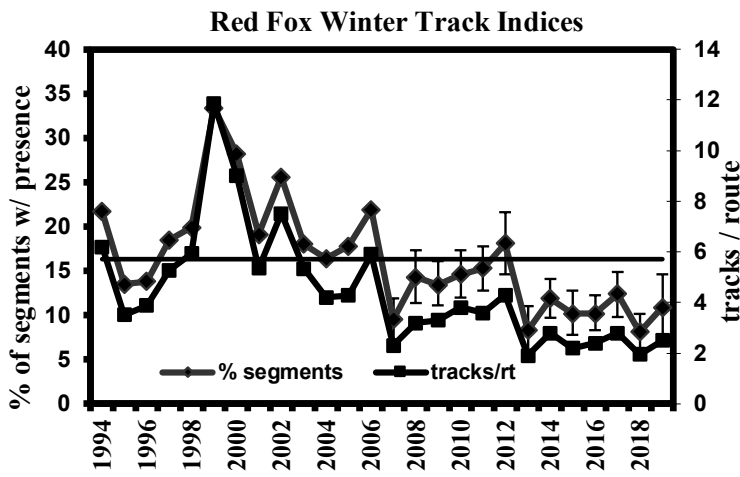
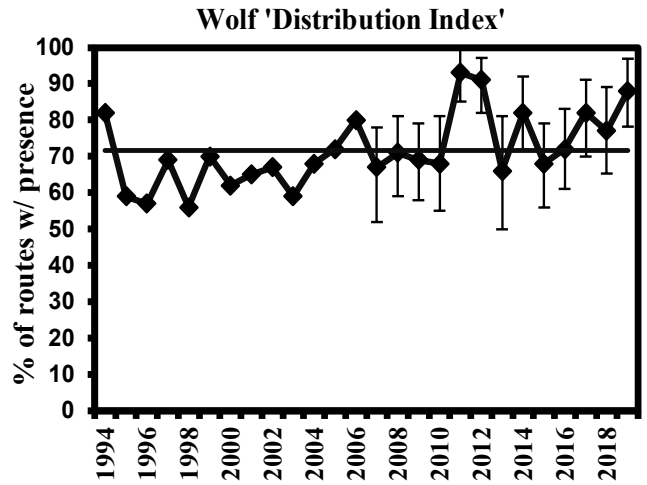
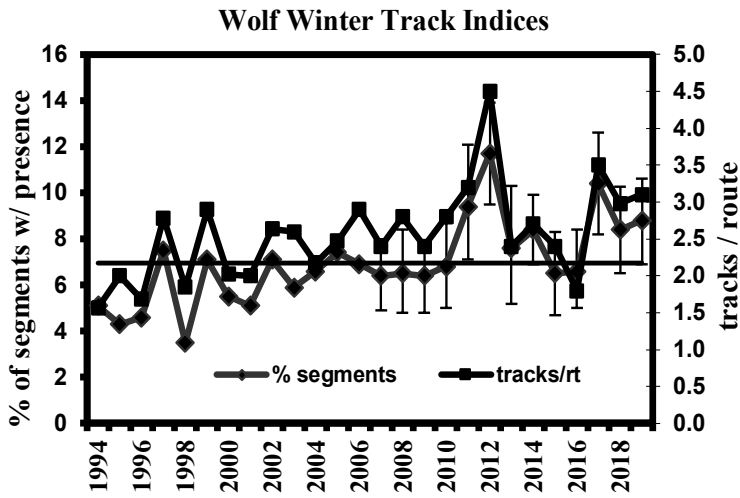


Figure 4 (continued). Winter track indices for selected species in Minnesota, 1994-2019.

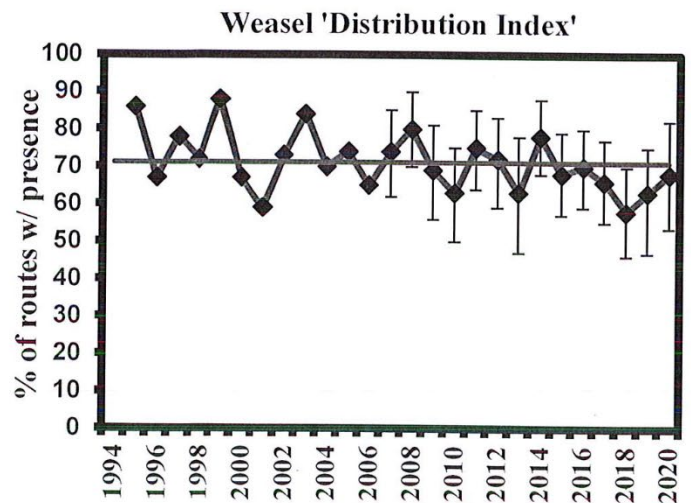
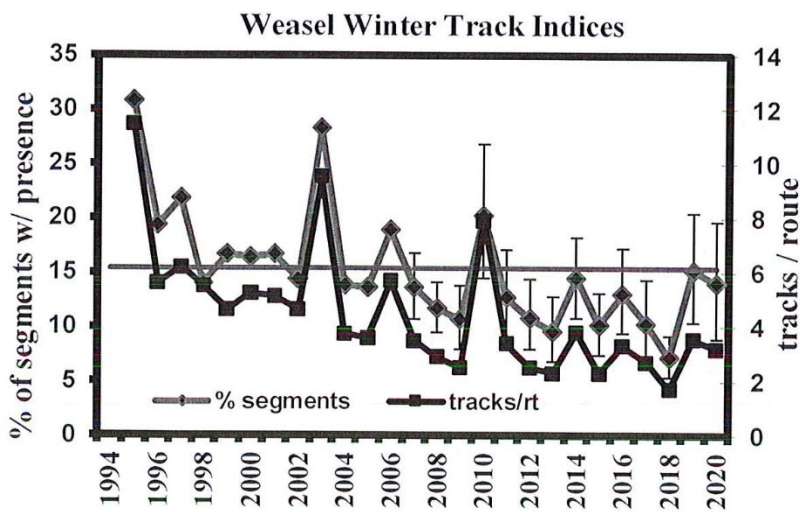
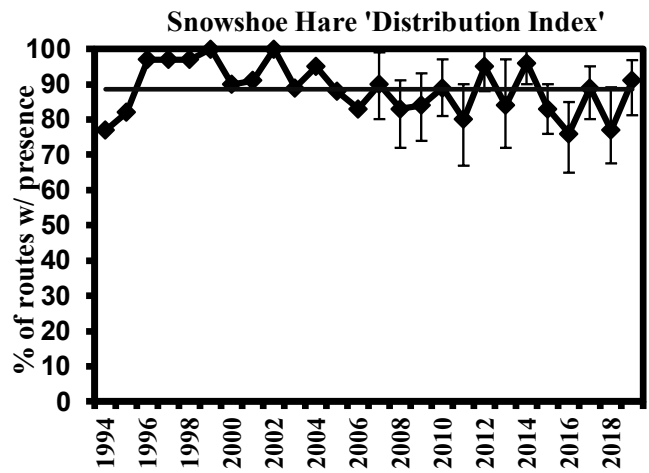
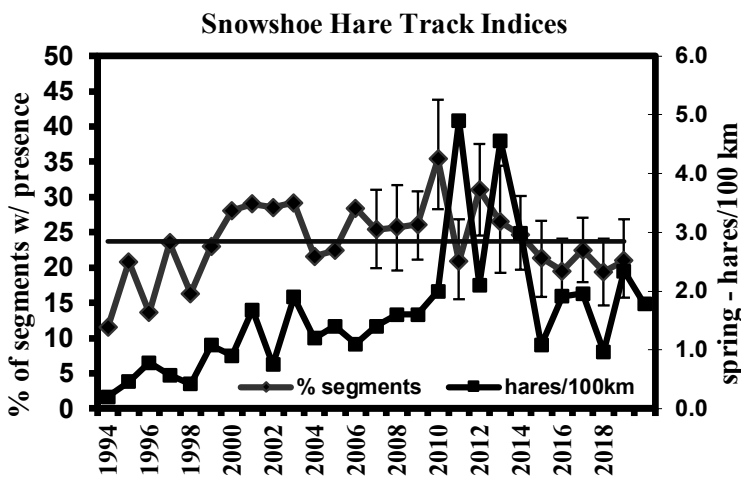
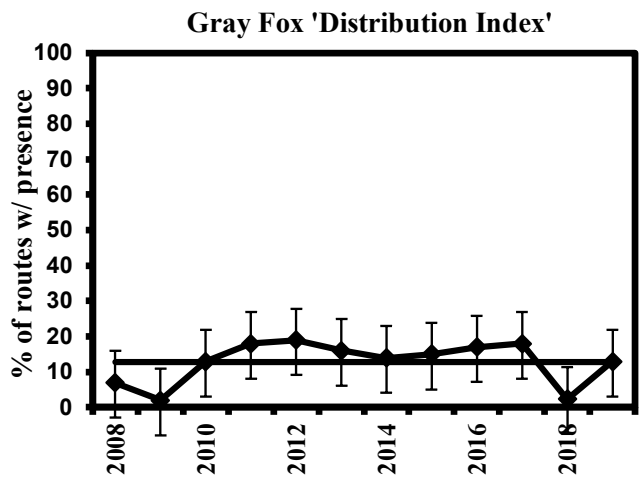
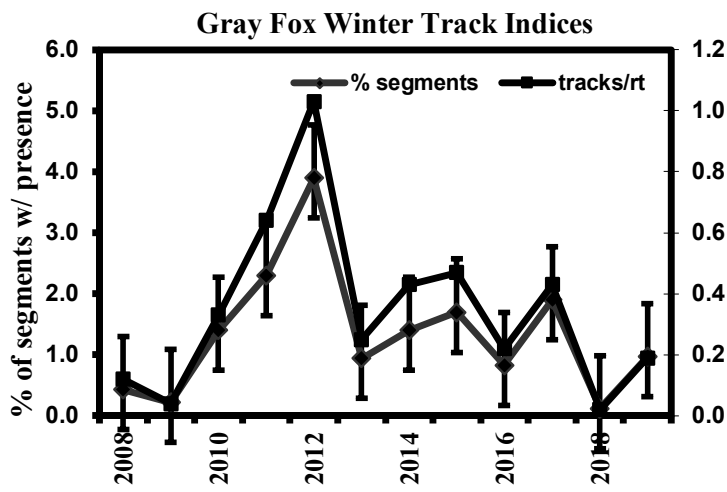


Figure 4 (continued). Winter track indices for selected species in Minnesota, 1994-2019.



REGISTERED FURBEARER POPULATION MODELING UPDATE 2019

John Erb, Forest Wildlife Populations and Research Group

INTRODUCTION

For populations of secretive carnivores, obtaining field-based estimates of population size remains a challenging task (Hochachka et al. 2000; Wilson and Delehay 2001; Conn et al. 2004). This is particularly true when one is interested in annual estimates, multiple species, or large areas. Nevertheless, population estimates are desirable to assist in making management or harvest decisions. Population modeling is a valuable tool for synthesizing our knowledge of population demography, predicting outcomes of management decisions, and approximating population size.

In the late 1970s, Minnesota developed population models for fishers (*Pekania pennanti*), martens (*Martes americana*), bobcats (*Lynx rufus*), and river otters (*Lontra canadensis*) to help estimate population size and monitor population changes. All are deterministic accounting models that do not currently incorporate density-dependence. However, annual adjustments to demographic inputs are often made for bobcats, fishers, and martens in response to the known or assumed influence of factors such as prey fluctuations, winter conditions, or competitor or predator density. Modeling projections are interpreted in conjunction with harvest data and results from any annual field-based track surveys.

METHODS

Primary model inputs include the estimated 1977 'starting' population size, estimates of age-specific survival and reproduction, and sex- and age-specific harvest data. Reproductive inputs were originally based largely on carcass data collected in the early 1980s. However, more recent reproductive data for fishers and martens was collected from 2007 – 2015 as part of a telemetry study (Erb et al. 2017), and for bobcats, additional carcass data was collected in 1992 and from 2003-present. Initial and subsequent survival inputs were based on a review of published estimates in the literature, updated for fishers and martens based on recent Minnesota research, and are periodically adjusted based on presumed relationships as noted above. In some cases, parameter adjustments for previous years are delayed until additional data on prey trends is available. Hence, population estimates reported in previous reports may not always match those reported in current reports.

Harvest data is obtained through mandatory furbearer registration. A detailed summary of 2018-19 harvest information is available in a separate report. Bobcat, marten, and fisher age data is obtained via x-ray examination of pulp cavity width or microscopic counts of cementum annuli from teeth of harvested animals. Although the population models only utilize data for the 3 age-classes (juvenile, yearling, adult), cementum annuli counts have periodically been collected for all non-juveniles either to examine age-specific reproductive output (bobcats) or to obtain periodic information on year-class distribution for selected species. The data was also used for deriving independent estimates of abundance using statistical population reconstruction (e.g., Skalski et al. 2012, Berg et al. 2017). In years where age data was not obtained for a given species, I use average harvest age proportions from the most recent period when data was collected.

For comparison to model projections, field-based track survey indices are presented in this report as running 3-year (t-1, t, t+1) averages of the observed track index, with the most recent year's average computed as $(2/3 \times \text{current index} + 1/3 \times \text{previous index})$. More detailed descriptions of scent station and winter track survey methods and results are available in separate reports.

RESULTS AND DISCUSSION

Bobcat. The 2018-19 state-registered trapping and hunting harvest of bobcats increased 39% to 1,015 (Table 1). Total modeled harvest, which includes reported tribal take, was 1,047. Juveniles accounted for 26% of the harvest, which was also comprised of 1.2 juveniles per adult female. Although both metrics have declined slightly over the past 3 years, they remain within the long-term observed range (Table 1, Figures 1 – 3). Median age for both male and female harvested bobcats was 2.5.

Reproductive data from female bobcats harvested in 2018 was also within previously observed bounds. Although there is a slight increasing trend in average litter sizes over the past 16 years, there has been minimal variation in reproductive output across years. Average litter sizes and pregnancy rates are slightly or significantly lower, respectively, for yearlings compared to older adults (Figures 4 and 5).

Based on projections from the population model, 14% of the fall 2018 population was harvested in 2018. Modeling projects minimal change to the 2019 fall population, projected to be near 8,000 bobcats (Figure 6). Both track indices remain near the upper end of their previously recorded range (Figure 6).

Fisher. The 2018 state-registered trapping harvest of fishers increased ~ 7% to 510 (Table 2). Modeled harvest, which includes reported tribal take, was 564.

After a 15-year lapse, fisher carcass collections were resumed in 2010 to collect current information on harvest age distribution; 488 carcasses were collected in 2018 (Table 2). Juveniles accounted for 54% of the total fisher harvest, similar to the average since aging resumed in 2010 but below the earlier average (64%) from 1977-1994. The juvenile to adult female ratio was 4.5, also similar to the post-2010 average but below the 1977-1994 average (6.6) (Table 2). Median age of harvested male and female fishers was 0.5 and 1.5, respectively (Figures 7 and 8).

Based on model projections, 7% of the fall fisher population was harvested during the 2018 season. Modeling projects a modest population increase over the past 3 years, in contradiction to the stable or slightly declining trend exhibited in the recent snow-track indices (Figure 9). Along the southern and western periphery of fisher range, an area not represented in track surveys, harvest and anecdotal information clearly indicate a population increase over the past 5-10 years. This area of range expansion is a comparatively small portion of overall fisher range, but may explain some of the discordance between track surveys (restricted to northern counties) and the spatially unbounded projections from the model. Acknowledging this caveat, modeling projects a 5% increase to the 2019 fall population, projected to be near 8,900 fishers statewide (Figure 9).

Marten. The 2018 state-registered trapping harvest of martens was 665, a 32% decline from the previous year (Table 3). Modeled harvest, which includes reported tribal take, was 732.

Juveniles accounted for 29% of the total harvest with a juvenile to adult female ratio of 2.3, both the second lowest since data collection began (Table 3, Figure 10). Though data suggests a long-term downward trend in these metrics, the low numbers this year are also likely part of shorter-term cyclic fluctuation in recruitment driven by prey cycles (Berg et al. 2017). Median age for both harvested males and females was 1.5 (Figures 11 and 12).

Based on projections from the marten population model, 7% of the fall 2018 population was harvested (Table 3). Similar to fishers, modeling projects a modest population increase in recent

years, in contradiction to the stable or slightly declining trend exhibited in recent snow-track indices (Figure 13). Contrary to fishers, however, spatial discordance between the track surveys and modeling projections is an unlikely explanation. It remains unclear whether track surveys are becoming biased low, model projections biased high, or both. Acknowledging this uncertainty, modeling projects a 12% increase to the 2019 fall population, projected to be near 11,100 martens (Figure 13).

Otter. From 1977 - 2007, otter harvest was only allowed in the northern part of the state. From 2007-2009, otter harvest was allowed in 2 separate zones with differing individual trapper limits (4 in the north zone, 2 in the southeast zone). Beginning in 2010, otter harvest was allowed statewide with a consistent limit of 4 otters per trapper. The 2018 state-registered trapping harvest of otters increased 4% to 1,351 (Table 4). Modeled statewide otter harvest, which includes tribal take, was 1,398 (Table 4).

An estimated 8% of the fall 2018 otter population was harvested, similar to the previous 2 years. Carcass collections ended in 1986 so no age or reproductive data are available, and no harvest-independent otter survey is currently established. Because demographic parameters in the otter model are usually held constant, fluctuations in population trajectory are largely a function of varying harvest levels. At recent population levels, harvests exceeding ~3,000 for consecutive years typically predict population declines. Since 2002, otter population estimates have varied as a result of notable fluctuations in pelt prices that have altered harvest above and below this threshold. With harvests remaining well below this threshold in recent years, and carrying capacity or density-dependent demographic constraints not currently incorporated in to the model, population projections are likely to be, or to become, unrealistic. Nevertheless, the population clearly remains near its high point estimated over the past 35 years (Figure 14), with the 2019 fall population projected to be ~ 22,000, a 9% increase from 2018.

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Table 1. Bobcat harvest data, 1989 to 2018.

Year	DNR Harvest	Modeled Harvest ¹	% Autumn			% juveniles	% yearlings	% adults	Juv: Ad. Female ratio	% male juveniles	% male yearlings	% male adults	Overall % males	Mean Pelt Price ³
			Pop. Taken ²	Carcasses Examined										
1989	129	129	6	119	39	17	44	2.0	49	53	56	53	\$48	
1990	84	87	4	62	20	34	46	0.8	58	80	44	59	\$43	
1991	106	110	5	93	35	33	32	3.5	59	55	70	61	\$37	
1992	167	167	7	151	28	22	50	1.2	55	45	53	53	\$28	
1993	201	210	8	161	32	20	48	1.4	51	45	52	50	\$43	
1994	238	270	11	187	26	16	58	0.8	64	43	45	50	\$36	
1995	134	152	6	96	31	15	54	2.7	57	71	79	71	\$32	
1996	223	250	10	164	35	20	45	1.8	51	30	49	46	\$33	
1997	364	401	16	270	35	16	49	1.4	60	37	43	48	\$30	
1998	103	107	4	77	29	26	45	1.6	59	60	60	60	\$28	
1999	206	228	8	163	18	24	58	0.8	55	59	62	60	\$24	
2000	231	250	8	183	31	26	43	1.4	54	59	50	53	\$33	
2001	259	278	8	213	30	21	49	1.3	46	45	47	52	\$46	
2002	544	621	15	475	27	25	48	1.1	68	51	48	54	\$72	
2003	483	518	13	425	25	13	62	0.9	62	48	54	55	\$96	
2004	631	709	14	524	28	34	38	1.7	52	40	55	49	\$99	
2005	590	638	13	485	25	13	62	0.8	51	48	47	48	\$96	

Year	DNR Harvest	Modeled Harvest ¹	% Autumn			% juveniles	% yearlings	% adults	Juv: Ad. Female ratio	% male juveniles	% male yearlings	% male adults	Overall % males	Mean Pelt Price ³
			Pop. Taken ²	Carcasses Examined										
2006	890	983	18	813	26	17	57	1.1	60	51	58	57	\$101	
2007	702	758	14	633	34	14	52	1.2	55	60	47	52	\$93	
2008	853	928	15	714	26	25	49	1.1	55	52	50	52	\$75	
2009	884	942	15	844	24	22	54	0.9	57	46	51	51	\$43	
2010	1012	1042	15	955	38	16	46	1.4	62	55	42	52	\$71	
2011	1711	1898	26	1626	23	21	55	0.8	61	73	47	56	\$98	
2012	1875	2026	30	1744	25	19	56	1.0	63	53	54	56	\$144	
2013	1038	1128	20	634	35	18	47	1.4	59	50	48	52	\$89	
2014	1384	1453	27	1296	28	16	56	1.3	60	48	60	58	\$60	
2015	766	803	17	674	24	25	51	1.3	63	63	65	64	\$57	
2016	484	491	9	464	32	21	47	1.9	66	57	64	63	\$36	
2017	731	758	12	682	29	25	46	1.5	65	51	58	58	\$64	
2018	1015	1047	14	984	26	22	52	1.2	59	57	60	59	\$60	

¹Includes DNR and Tribal harvests

²Estimated from population model; includes estimated non-reported harvest of 10%.

³ Average pelt price based on a survey of in-state fur buyers only.

Bobcat Harvest Age-Classes

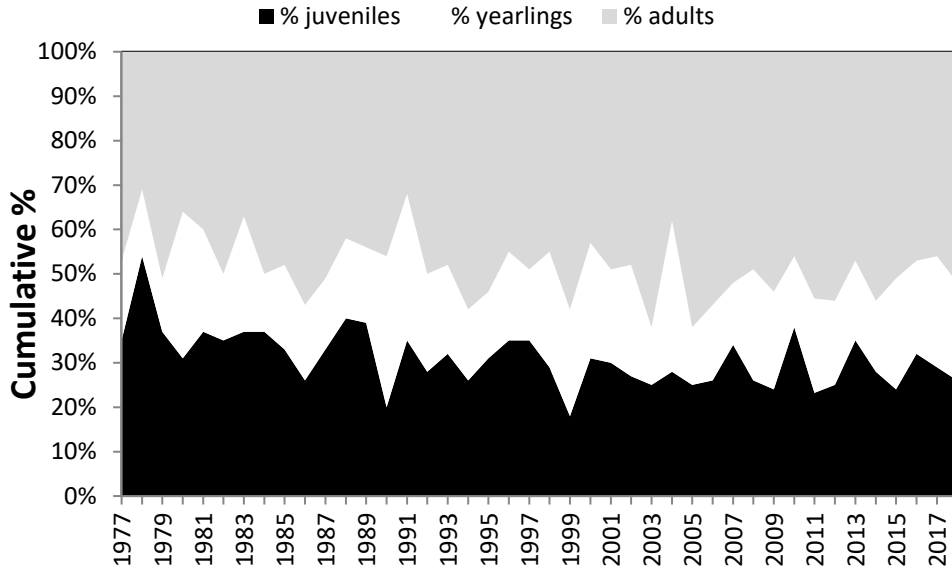


Figure 1. Age-class distribution of bobcats harvested in Minnesota, 1977-2018.

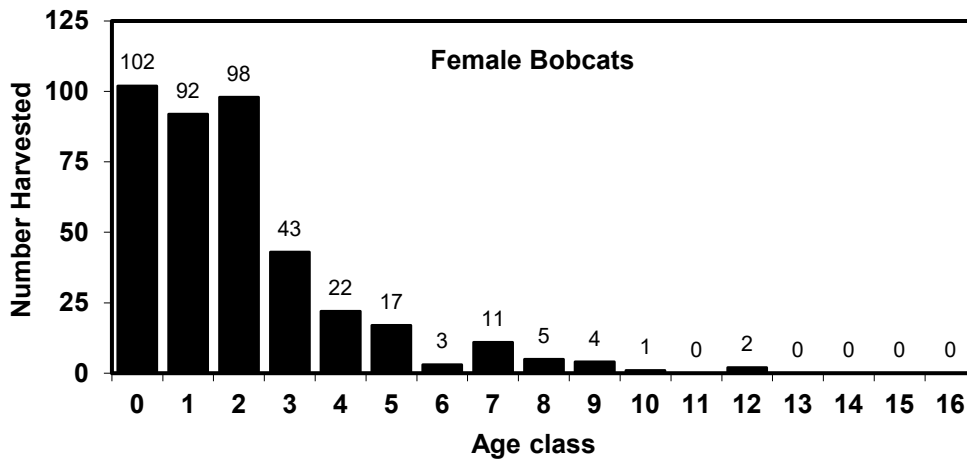


Figure 2. Age structure of female bobcats in the 2018 harvest.

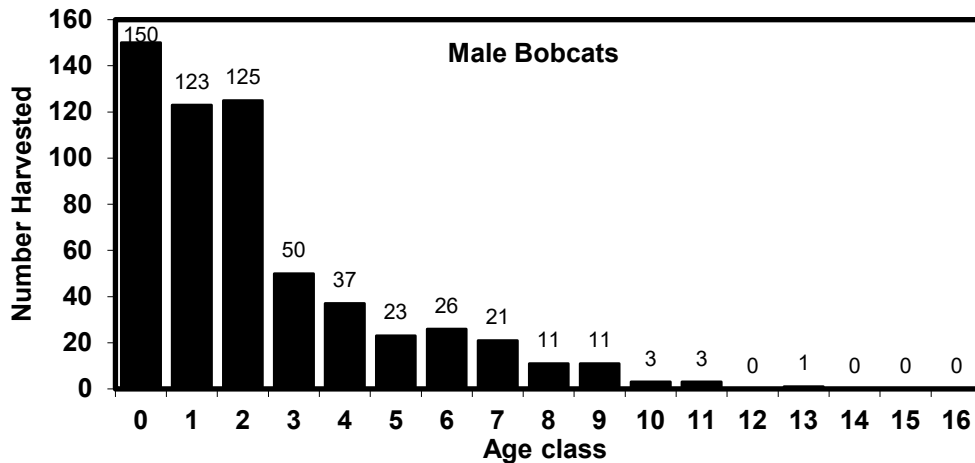


Figure 3. Age structure of male bobcats in the 2018 harvest.

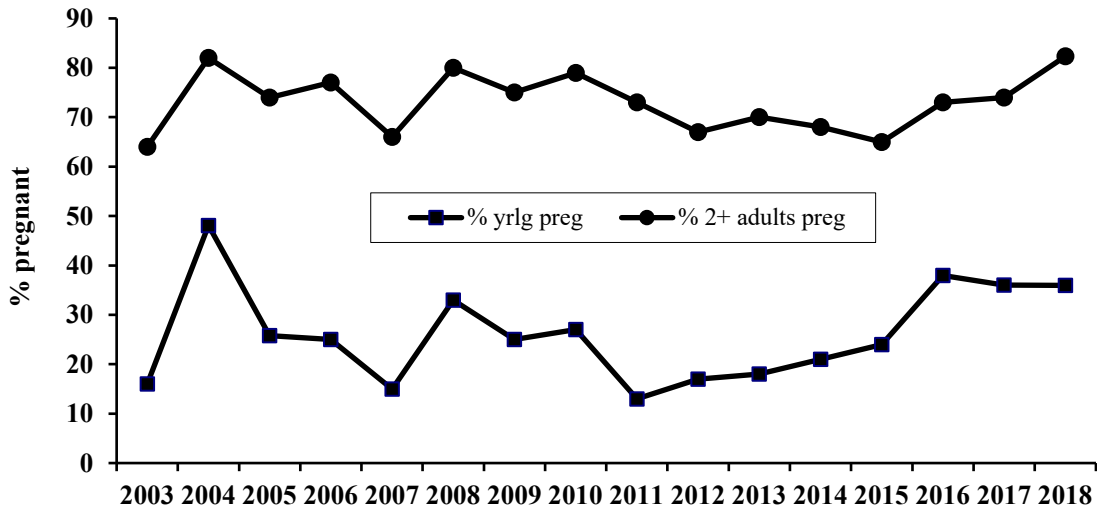


Figure 4. Pregnancy rates for yearling and adult bobcats in Minnesota, 2003-2018.

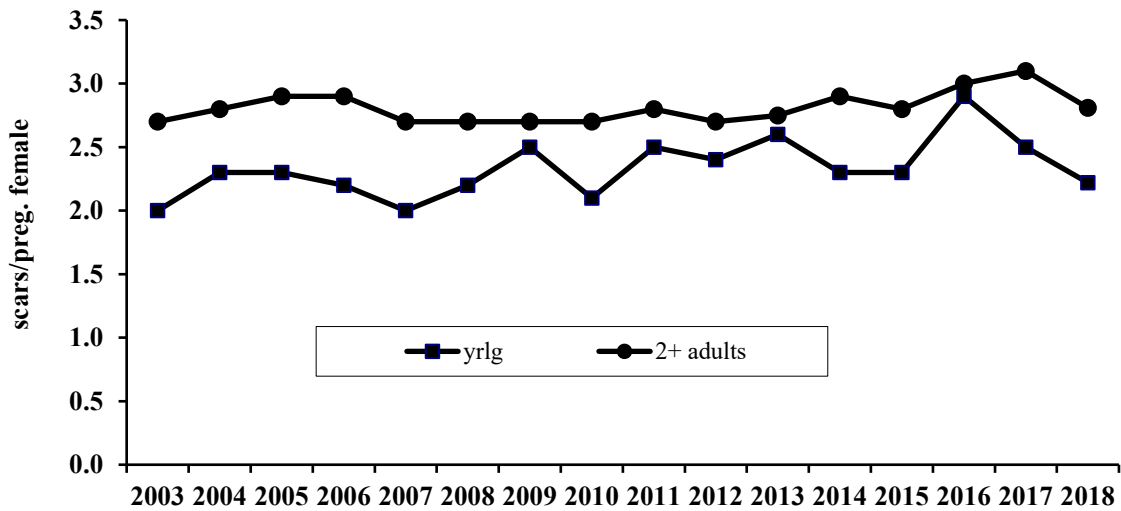


Figure 5. Litter size for parous yearling and adult bobcats in Minnesota, 2003-2018.

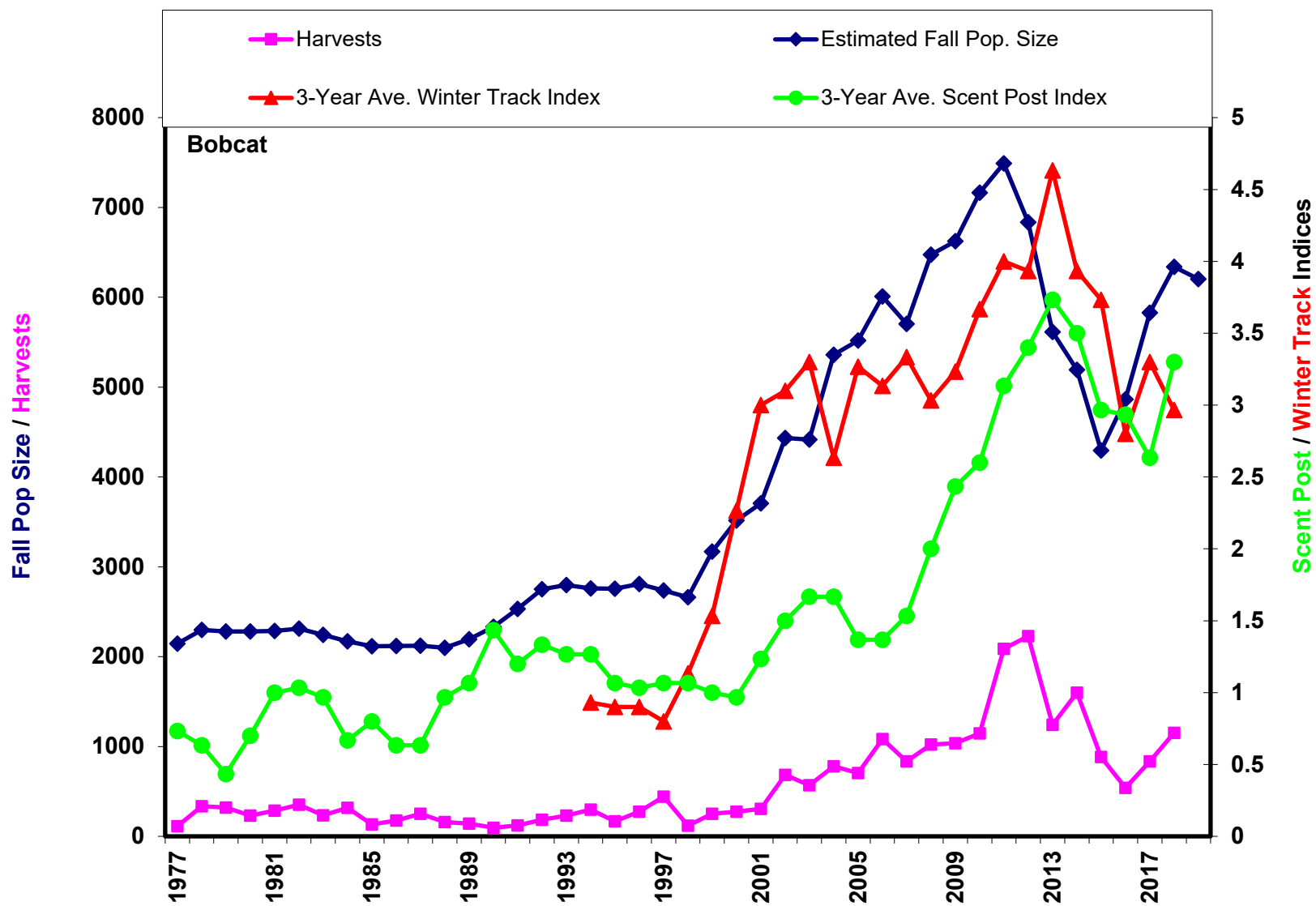


Figure 6. Bobcat population projections, harvests, and survey indices, 1977-2019. Harvests include an estimate of non-reported take.

Table 2. Fisher harvest data, 1989 to 2018.

Year	DNR harvest	Modeled Harvest ¹	% Autumn			% juveniles	% yearlings	% adults	Juv: Ad. Female ratio	% male juveniles	% male yearlings	% male adults	% males overall	Pelt price Males ³	Pelt price Females ³
			Pop. Harvested ²	Carcasses examined											
1989	1243	1243	16	1024	64	19	17	5.8	47	47	36	45	\$26	\$53	
1990	746	756	9	592	65	14	21	4.4	44	55	30	43	\$35	\$46	
1991	528	528	6	410	66	20	14	7.5	50	52	35	48	\$21	\$48	
1992	778	782	8	629	58	21	21	4.8	42	55	45	46	\$16	\$29	
1993	1159	1192	10	937	59	22	19	6.0	47	37	42	44	\$14	\$28	
1994	1771	1932	15	1360	57	18	25	4.0	47	54	44	48	\$19	\$30	
1995	942	1060	8	-	-	-	-	-	-	-	-	45	\$16	\$25	
1996	1773	2000	14	-	-	-	-	-	-	-	-	45	\$25	\$34	
1997	2761	2974	20	-	-	-	-	-	-	-	-	45	\$31	\$34	
1998	2695	2987	20	-	-	-	-	-	-	-	-	45	\$19	\$22	
1999	1725	1880	13	-	-	-	-	-	-	-	-	45	\$19	\$20	
2000	1674	1900	13	-	-	-	-	-	-	-	-	45	\$20	\$19	
2001	2145	2362	15	-	-	-	-	-	-	-	-	54	\$23	\$23	
2002	2660	3028	20	-	-	-	-	-	-	-	-	54	\$27	\$25	
2003	2521	2728	19	-	-	-	-	-	-	-	-	55	\$27	\$26	
2004	2552	2753	20	-	-	-	-	-	-	-	-	52	\$30	\$27	
2005	2388	2454	19	-	-	-	-	-	-	-	-	52	\$36	\$31	
2006	3250	3500	29	-	-	-	-	-	-	-	-	51	\$76	\$68	

Year	DNR harvest	Modeled Harvest ¹	% Autumn				% juveniles	% yearlings	% adults	Juv: Ad. Female ratio	% male juveniles	% male yearlings	% male adults	% males overall	Pelt price Males ³	Pelt price Females ³
			Pop. Harvested ²	Carcasses examined												
2007	1682	1811	18	-	-	-	-	-	-	-	-	-	52	\$63	\$48	
2008	1712	1828	19	-	-	-	-	-	-	-	-	-	52	\$22	\$37	
2009	1259	1323	15	-	-	-	-	-	-	-	-	-	53	\$35	\$34	
2010	903	951	11	759	52	25	23	4.5	55	54	50	54	\$38	\$37		
2011	1473	1651	19	1314	47	28	25	3.2	59	53	42	53	\$48	\$40		
2012	1293	1450	18	1108	51	24	25	3.7	59	53	45	54	\$62	\$63		
2013	1146	1295	17	1040	51	24	25	3.4	55	56	42	52	\$74	\$68		
2014	943	1045	15	881	56	21	23	3.7	57	57	36	52	\$44	\$55		
2015	756	818	12	698	55	19	26	3.8	57	52	44	53	\$35	\$34		
2016	399	434	6	348	56	22	22	4.5	53	56	42	51	\$28	\$37		
2017	477	509	7	440	52	30	18	6.4	65	51	58	58	\$31	\$38		
2018	510	564	7	488	54	24	22	4.5	59	48	46	53	\$43	\$40		

¹ Includes DNR and Tribal harvests

² Estimated from population model, includes estimated non-reported harvest of 20% 1977-1992, and 10% from 1993-present.

³ Average pelt price based on a survey of in-state fur buyers only.

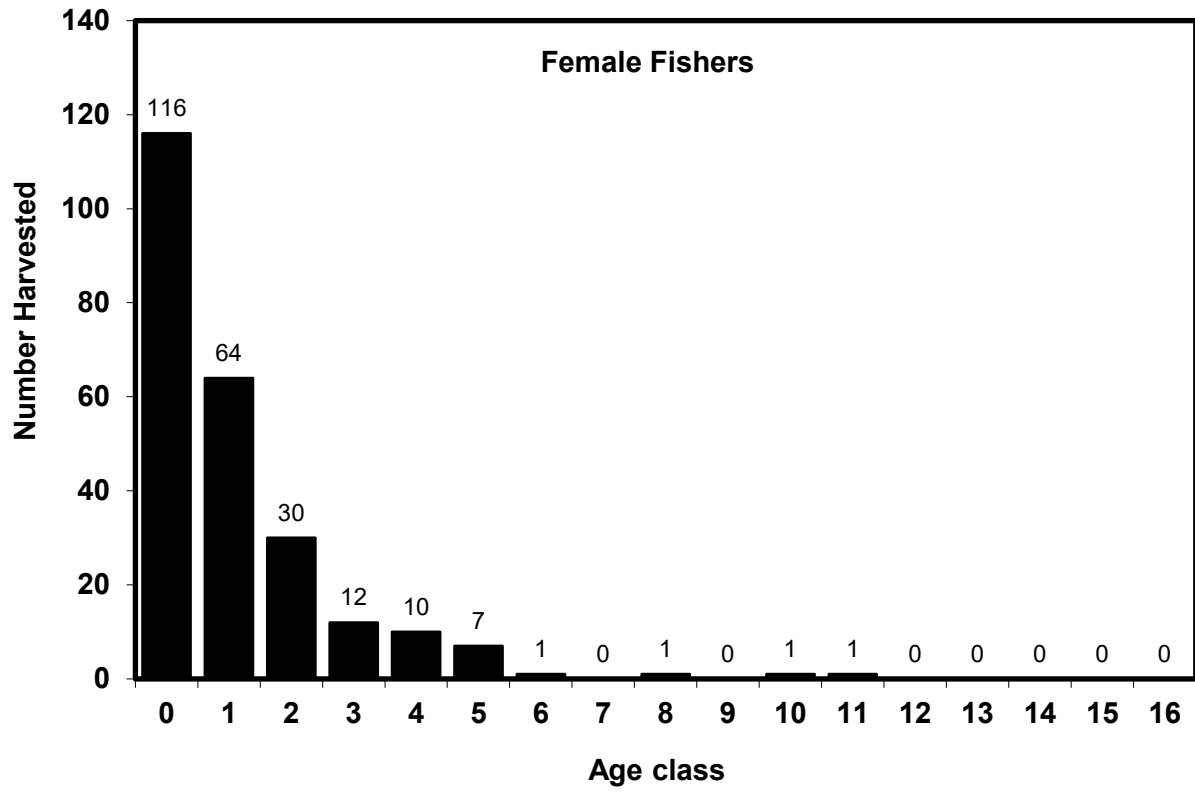


Figure 7. Age structure of female fishers in the 2018 harvest.

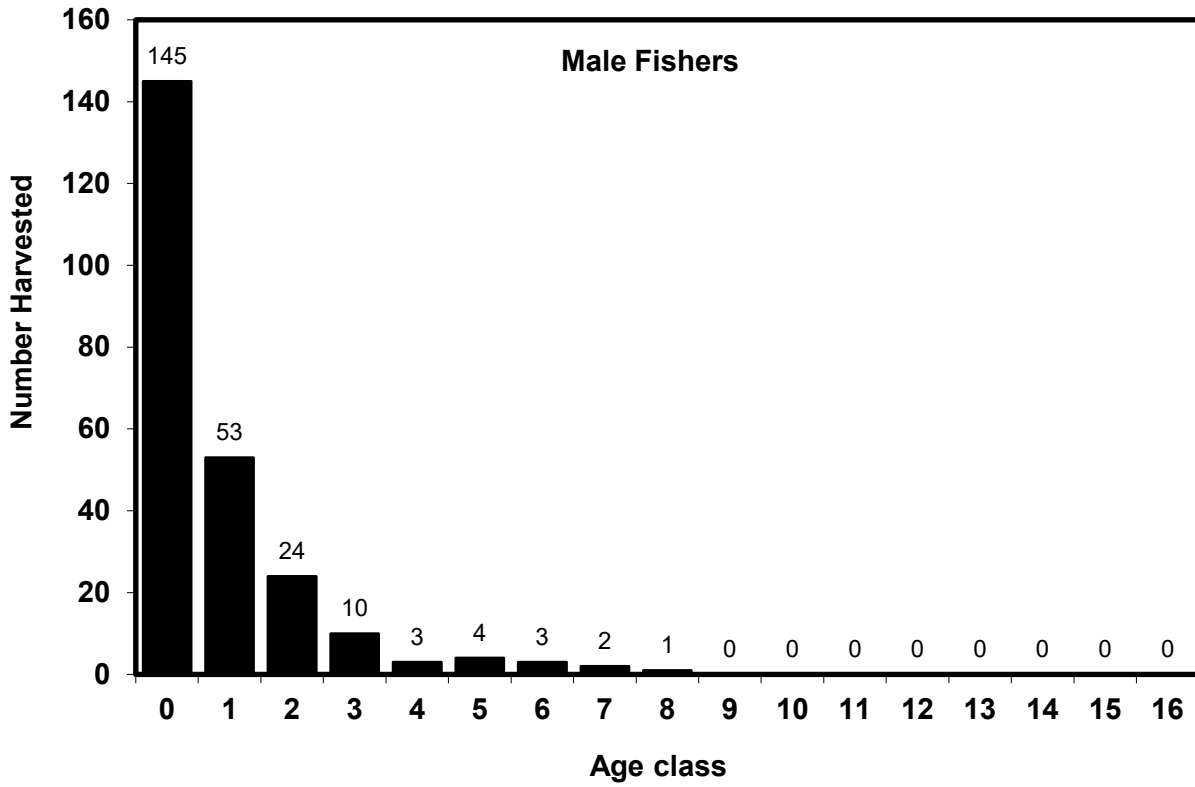


Figure 8. Age structure of male fishers in the 2018 harvest.

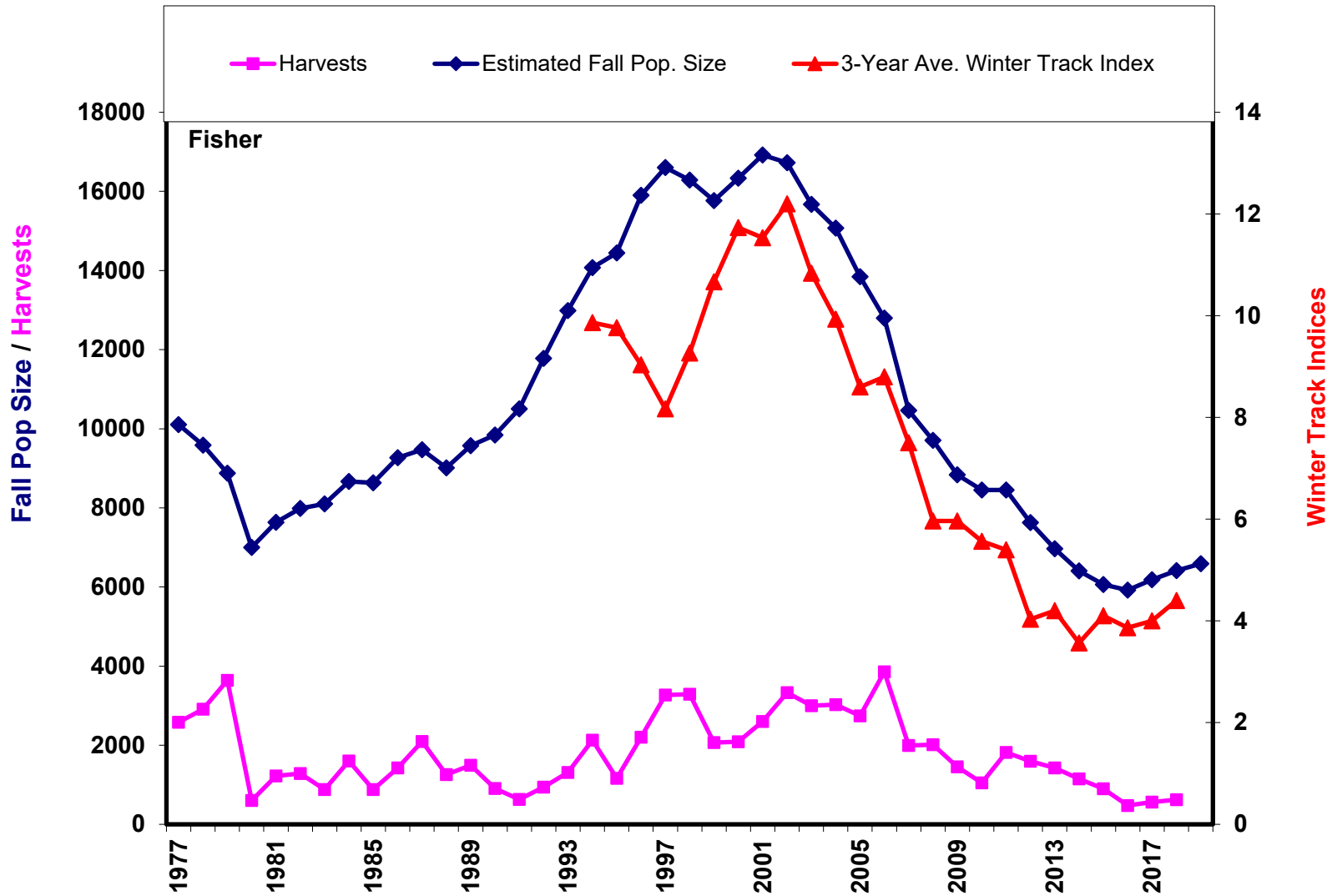


Figure 9. Fisher population projections, harvests, and survey indices, 1977-2019. Harvests include an estimate of non-reported take.

Table 3. Marten harvest data, 1989 to 2018.

Year	DNR harvest	% Autumn			% juveniles	% yearlings	% adults	Juv: Ad. Female ratio	% male juveniles	% male yearlings	% male adults	% males overall	Pelt price Males ⁴	Pelt price Females ⁴
		Modeled Harvest ¹	Pop. Harvested ²	Carcasses Examined ³										
1989	2119	2119	18	1014	68	12	20	9.9	57	63	65	59	\$48	\$47
1990	1349	1447	12	1375	48	18	34	3.6	59	54	61	59	\$44	\$41
1991	686	1000	9	716	74	9	17	13.5	69	71	72	70	\$40	\$27
1992	1602	1802	14	1661	65	18	17	14.8	63	70	75	66	\$28	\$25
1993	1438	1828	13	1396	57	20	23	7.6	61	71	67	64	\$36	\$30
1994	1527	1846	13	1452	58	15	27	6.5	62	76	67	66	\$34	\$28
1995	1500	1774	12	1393	60	18	22	8.2	63	68	66	65	\$28	\$21
1996	1625	2000	14	1372	48	22	30	4.9	62	69	67	65	\$34	\$29
1997	2261	2762	19	2238	61	13	26	6.2	60	60	63	61	\$28	\$22
1998	2299	2795	20	1577	57	18	25	6.5	62	66	65	63	\$20	\$16
1999	2423	3000	20	2013	67	12	21	9.9	65	66	67	66	\$25	\$21
2000	1629	2050	14	1598	56	25	19	8.8	62	69	66	64	\$28	\$21
2001	1940	2250	15	1895	62	15	23	10.7	65	73	74	69	\$24	\$23
2002	2839	3192	19	2451	38	30	32	3.3	59	65	62	62	\$28	\$27
2003	3214	3548	22	2391	49	16	35	4.2	59	66	68	64	\$30	\$27
2004	3241	3592	25	2776	26	28	46	1.4	54	67	59	60	\$31	\$27
2005	2653	2873	22	1992	62	13	25	7.2	66	64	66	66	\$37	\$32
2006	3788	4120	31	1914	64	17	19	9.5	67	68	67	67	\$74	\$66

Year	DNR harvest	% Autumn			Carcasses Examined ³	% juveniles	% yearlings	% adults	Juv: Ad. Female ratio	% male juveniles	% male yearlings	% male adults	% males overall	Pelt price Males ⁴	Pelt price Females ⁴
		Modeled Harvest ¹	Pop. Harvested ²	Pop. Examined ³											
2007	2221	2481	22	1355	30	29	41	1.6	60	68	54	60	\$59	\$50	
2008	1823	1953	20	1095	40	21	39	2.4	62	64	57	60	\$31	\$28	
2009	2073	2250	23	1252	55	16	29	5.1	67	49	63	63	\$27	\$30	
2010	1842	1977	20	1202	47	25	28	4.4	71	56	62	65	\$40	\$37	
2011	2525	2744	28	1615	39	25	36	2.7	64	64	60	62	\$42	\$39	
2012	1472	1610	19	1260	34	30	36	2.6	67	57	64	63	\$57	\$54	
2013	1014	1323	16	942	43	20	37	3.5	59	62	68	63	\$74	\$71	
2014	1059	1124	13	991	58	14	28	5.8	65	67	64	65	\$45	\$34	
2015	877	956	11	812	49	25	26	4.9	64	69	60	64	\$31	\$29	
2016	551	677	7	504	56	23	21	8.1	68	73	68	69	\$30	\$30	
2017	979	1076	11	865	50	25	25	5.0	63	72	60	64	\$39	\$38	
2018	665	732	7	638	29	34	37	2.3	63	69	66	66	\$42	\$33	

¹ Includes DNR and Tribal harvests

² Estimated from population model; includes estimated non-reported harvest of 40% in 1985-1987 and 1991, 20% in 1988-1990 and 1992-1998, and 10% from 1999-present.

³ Starting in 2005, the number of carcasses examined represents a random sample of ~ 70% of the carcasses collected in each year.

⁴ Average pelt price based on a survey of in-state fur buyers only

Marten Harvest Age-Classes

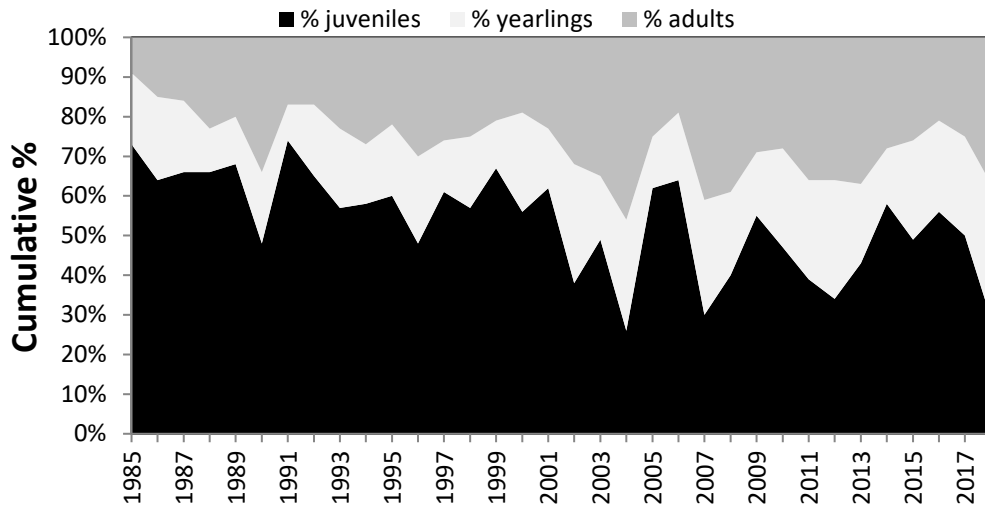


Figure 10. Age-class distribution of martens harvested in Minnesota, 1985 - 2018.

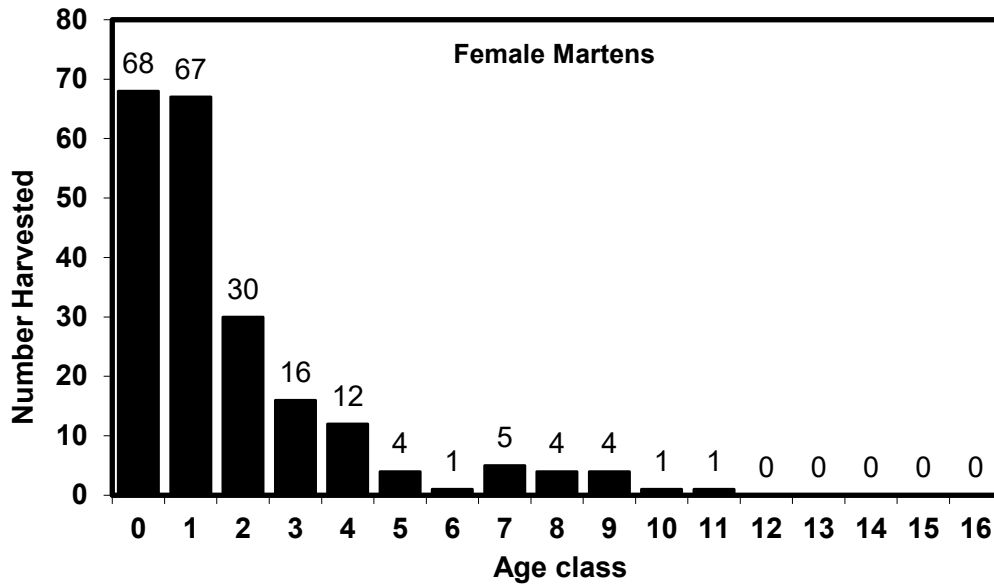


Figure 11. Age structure of female martens in the 2018 harvest.

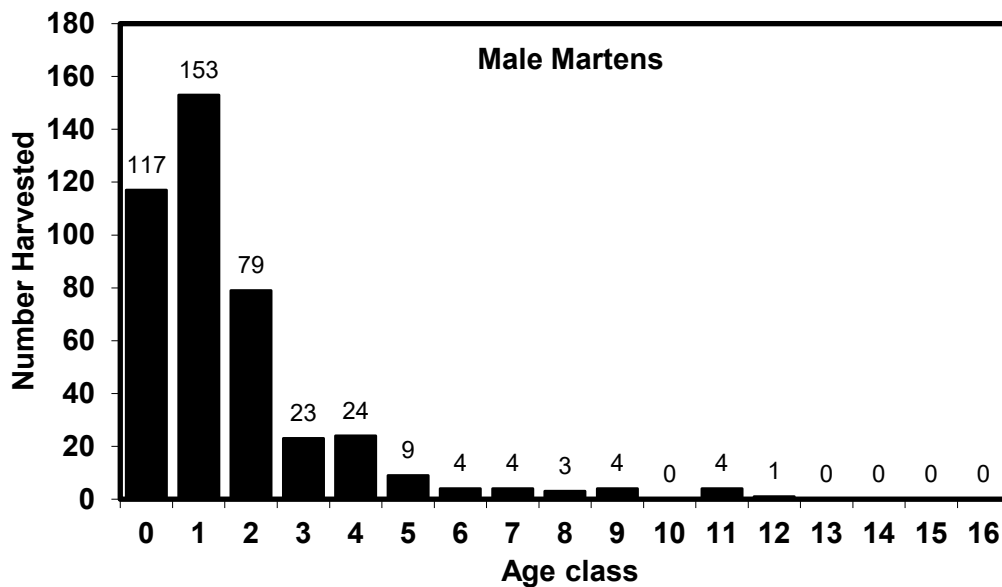


Figure 12. Age structure of male martens in the 2018 harvest.

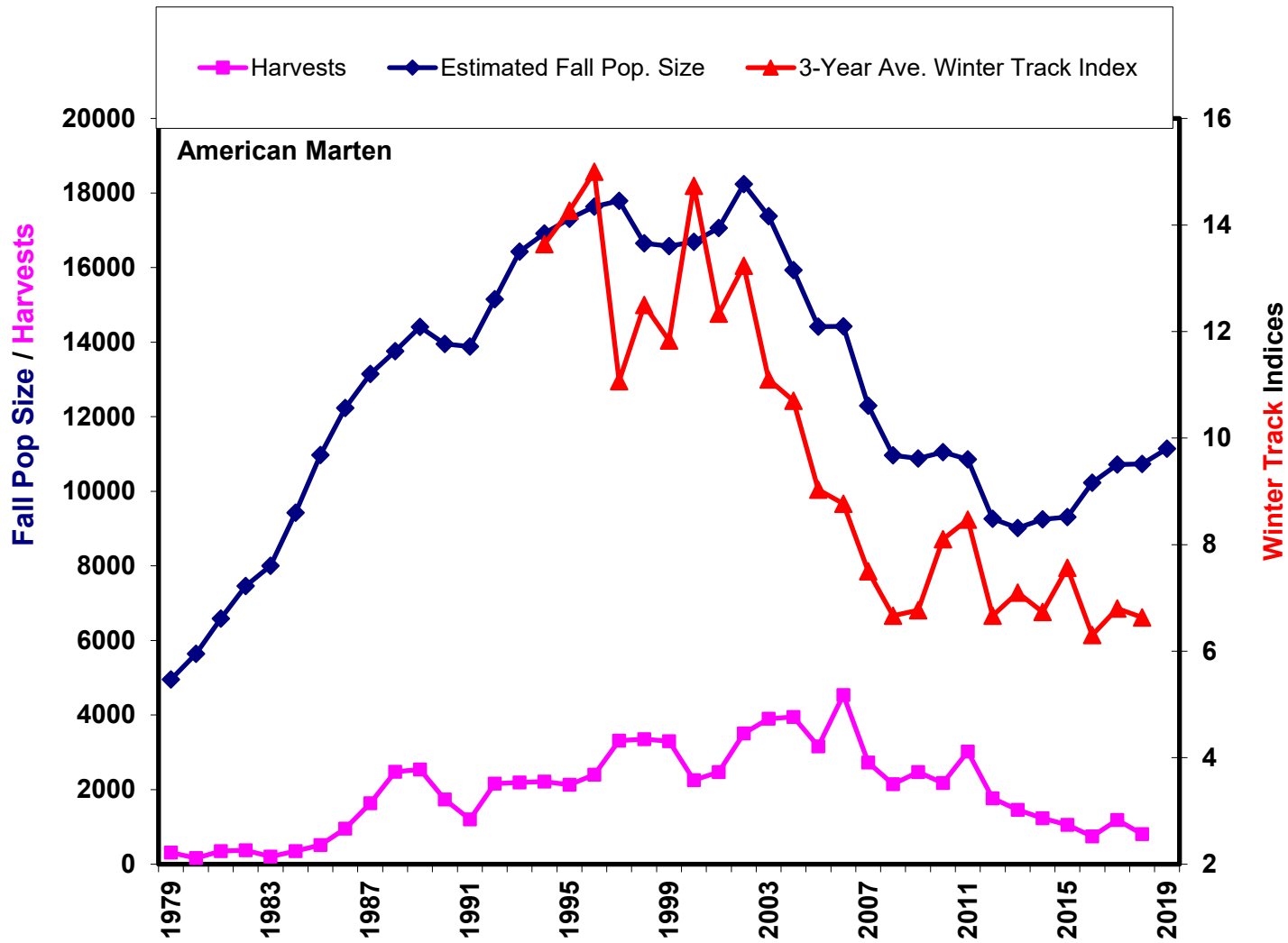


Figure 13. American marten population projections, harvests, and survey indices, 1979-2018. Harvests include an estimate of non-reported take.

Table 4. Otter harvest data¹, 1989 to 2018. Carcasses were only collected from 1980-86.

Year	DNR harvest	% Autumn		Carcasses examined	% juveniles	% yearlings	% adults	Juv:ad. females	% male juveniles	% male yearlings	% male adults	% males overall	Pelt price Otter ³	Pelt price Beaver ³
		Modeled Harvest ¹	Pop. Harvested ²											
1989	1294	1294	12	-	-	-	-	-	-	-	-	52	\$22	\$12
1990	888	903	8	-	-	-	-	-	-	-	-	52	\$24	\$9
1991	855	925	8	-	-	-	-	-	-	-	-	51	\$25	\$9
1992	1368	1365	10	-	-	-	-	-	-	-	-	52	\$30	\$7
1993	1459	1368	10	-	-	-	-	-	-	-	-	52	\$43	\$10
1994	2445	2708	18	-	-	-	-	-	-	-	-	52	\$48	\$14
1995	1435	1646	12	-	-	-	-	-	-	-	-	52	\$39	\$12
1996	2219	2500	17	-	-	-	-	-	-	-	-	52	\$39	\$19
1997	2145	2313	16	-	-	-	-	-	-	-	-	52	\$40	\$17
1998	1946	2139	15	-	-	-	-	-	-	-	-	52	\$34	\$13
1999	1635	1717	12	-	-	-	-	-	-	-	-	52	\$41	\$11
2000	1578	1750	12	-	-	-	-	-	-	-	-	52	\$51	\$14
2001	2301	2531	17	-	-	-	-	-	-	-	-	57	\$46	\$13
2002	2145	2390	15	-	-	-	-	-	-	-	-	59	\$61	\$10
2003	2766	2966	19	-	-	-	-	-	-	-	-	57	\$85	\$12
2004	3450	3700	24	-	-	-	-	-	-	-	-	56	\$87	\$14
2005	2846	3018	22	-	-	-	-	-	-	-	-	58	\$89	\$15

Year	DNR harvest	Modeled Harvest ¹	% Autumn				% juveniles	% yearlings	% adults	Juv:ad. females	%	%	%	%	Pelt price Otter ³	Pelt price Beaver ³
			Pop. Harvested ²	Carcasses examined	male juveniles	male yearlings					male adults	male overall				
2006	2720	2873	21	-	-	-	-	-	-	-	-	-	56	\$43	\$17	
2007	1861	1911	15	-	-	-	-	-	-	-	-	-	55	\$29	\$16	
2008	1938	1983	15	-	-	-	-	-	-	-	-	-	59	\$24	\$12	
2009	1544	1578	12	-	-	-	-	-	-	-	-	-	59	\$36	\$13	
2010	1814	1830	13	-	-	-	-	-	-	-	-	-	57	\$35	\$13	
2011	2294	2490	17	-	-	-	-	-	-	-	-	-	58	\$51	\$17	
2012	3171	3377	22	-	-	-	-	-	-	-	-	-	60	\$72	\$16	
2013	2824	2993	21	-	-	-	-	-	-	-	-	-	48	\$61	\$17	
2014	2154	2235	16	-	-	-	-	-	-	-	-	-	59	\$35	\$12	
2015	1955	2030	14	-	-	-	-	-	-	-	-	-	62	\$30	\$8	
2016	1195	1227	8	-	-	-	-	-	-	-	-	-	62	\$21	\$8	
2017	1295	1336	8	-	-	-	-	-	-	-	-	-	60	\$22	\$10	
2018	1351	1398	8	-	-	-	-	-	-	-	-	-	57	\$25	\$9	

¹ Includes DNR and Tribal harvests

² Estimated from population model. Incl. estimated non-reported harvest of 30% to 1991, 22% from 1992-2001, and 15% from 2002-present.

³ Weighted average of spring (beaver only) and fall prices based on a survey of in-state fur buyers.

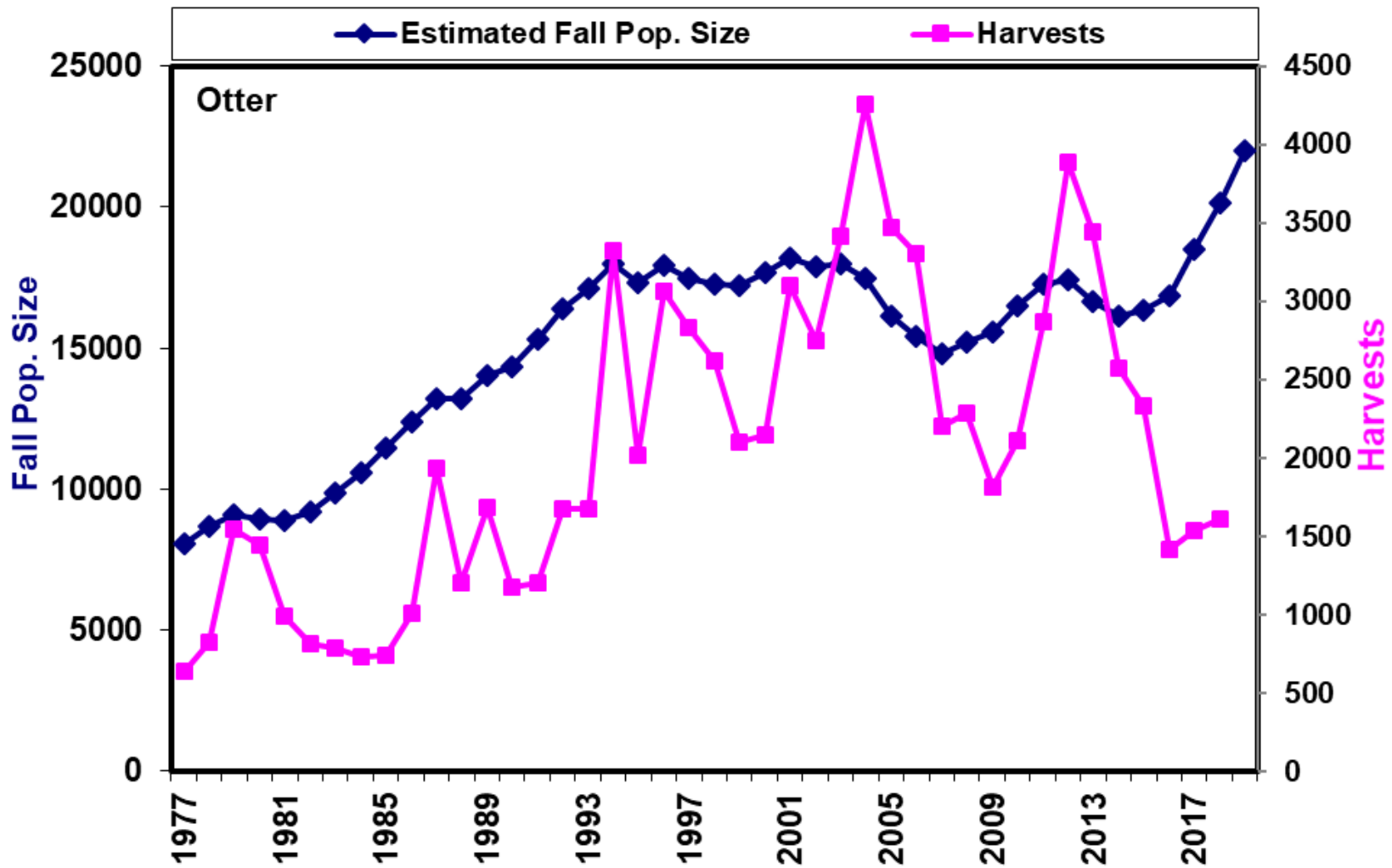


Figure 14. Otter population projections and harvests, 1977-2018. Harvests include an estimate of non-reported take.



STATUS OF MINNESOTA BLACK BEARS, 2019

Dave Garshelis and Andy Tri, Forest Wildlife Research Group

INTRODUCTION

The size of the Minnesota bear population has been estimated in the past using a biomarker (tetracycline) and mark–recapture based on hunter-submitted samples (Garshelis and Visser 1997, Garshelis and Noyce (2006). The last estimate was produced in 2008, and the use of that biomarker may no longer be permitted. Since then, trends in the population have been assessed using various modelling approaches, based on composition (sex-age) of harvest data. Additionally, population information may be inferred by examination of nuisance bear complaints and the seasonal abundance of natural bear foods.

METHODS

Successful hunters must register their bears and submit a tooth sample, which is used to estimate age, and thus harvest age structure. Hunters also report the sex of their harvested bear; we adjust this for a known bias in hunter-reported sex (11% of female bears reported as males). Ages and sexes of harvested bears accumulated since 1980 were used to reconstruct minimum statewide population sizes through time (i.e., the size of the population that eventually died due to hunting) using a technique formulated by Downing (1980): each sex was estimated separately, and then summed. Age groups were collapsed to 1, 2, and 3+ years in order to estimate population size 3 years in the past (no more recent estimates can be obtained using this technique). This technique only estimates the size of the population that eventually dies due to hunting; to account for bears that die of other causes, the trend lines are scaled upward to attempt to match tetracycline-based estimates.

A second, independent assessment of population trend is obtained by investigating harvest rates (% of living bears harvested each year). A relatively low harvest rate would signify a population with more potential growth. Harvest rate is estimated from the inverse of the age at which the number of males and females in the harvest is equal, based on methodology of Fraser (1984).

RESULTS

Population trend statewide

Ages of harvested bears accumulated since 1980 were used to reconstruct minimum statewide population sizes through time (i.e., the size of the population that eventually died due to hunting) using a technique formulated by Downing. This was scaled upwards (to include bears that died of other causes), using 4 statewide tetracycline mark–recapture estimates as a guide. One

trajectory, which assumed non-harvest mortality, was 23% of total mortality (curves elevated x1.3) matched the 1991 tetracycline estimate, but fell below the other tet-estimates. Another trajectory, which assumed non-harvest mortality was 44% of all mortality (curves elevated x1.8) matched the 1997, 2002, and 2008 tet-estimates. The curves show an increasing population from the early 1980s to mid-1990s, leveling off in the late 1990s, followed by a steep decline through the 2000s. Since 2013, quotas were maintained at a low and consistent level (Table 1) in an attempt to reverse the population decline (and also to allow the models to perform better, without the confounding issue of changing hunter effort). The reduced harvest pressure stabilized the population (Figure 1).

Population trend: quota vs no-quota zones

Downing population reconstruction indicated that the quota zone population declined by ~50% from 2000 to 2014 (Figure 2). With reduced quotas and lower harvests since then, the quota zone population increased over 10% from 2014 to 2016, according to this model, but then dipped following the higher than expected harvest in 2016. The Downing model does not produce population estimates for the most recent 2 years, so the effects of lower harvests in 2017–2019 is not yet reflected.

A new Bayesian model developed by Allen et al. (2018) for bear monitoring in Wisconsin includes not only the sex-age composition of harvested bears (like the Downing), but also reproductive and survival parameters (obtained from data collected from long-term monitoring of radio-collared bears in different study sites across Minnesota). This model does not have a lag time (so projections are available to pre-hunt 2020). The trajectory of this model was remarkably similar to that of the Downing model for the quota zone, and indicated that the population there is slowly recovering.

Meanwhile, despite a surge in “overflow” hunters in the no-quota zone (Figure 4) prompted by the more restrictive quota zone permit allocations, harvests in the no-quota zone have not increased, and both the Downing and Allen models show a recent population increase.

Trends in harvest rates

The sex ratio of harvested bears varies by age (Figure 3). Male bears are more vulnerable to harvest than females, so males always predominate among harvested 1-year-olds (67–75%). Males also predominate, but less strongly among 2 and 3-year-old harvested bears. However, older-aged harvested bears (≥ 8 years) are nearly always dominated by females, because, although old females continue to be less vulnerable as individuals, there are far more of them than old males in the living population. The age at which the line fitted to these proportions crosses the 50:50 sex ratio is approximately the inverse of the harvest rate. Segregating the data into time blocks showed harvest rates increasing from 1980–1999, then declining with reductions in hunter numbers (Figure 5). Based on this method, harvest rates since 2015 have been significantly less than what they were in the early 1980s, when the bear population was increasing (Figure 1).

One problem in using this very simple method is that it assumes that the relative difference for males versus females in their vulnerability to harvest does not change systematically through time. This may not be true, given the steadily increasing male-skewed harvests since the late 1990s, and especially in recent years (Figures 6 & 7).

Nuisance complaints and kills

The total number of recorded bear complaints slowly increased over the past decade, reaching a peak in 2015 and 2016 (Table 2, Figure 8). Number of complaints declined in 2017, despite a higher number of DNR personnel recording complaints, and declined again in 2018, with

abundant natural foods all summer (Tables 3 & 4). Average or below-average foods during the summer of 2019 led to higher numbers of complaints. A new recording system was instituted in 2017 whereby Wildlife Managers recorded all bear complaints online as they were received, instead of submitting reports at the end of each month (thus, unlike previous years, Managers who had no complaints were not counted in the number of personnel participating). Conservation Officers implemented a similar system beginning July 2019. This dramatically increased the number of officers reporting bear complaints. Also, a relatively high number of the reports from officers involved a bear being killed by a private party. In 2018 and 2019, a list was distributed of “area 88” hunters, who expressed interest in taking a nuisance bear in the quota area on a no-quota license. Only 13 hunters purchased an “88” license and 3 hunters were successful (23% success rate).

Food abundance

The composite range-wide, all-season abundance of natural bear foods (fruits and nuts) in 2019 was comparable to 2017 and significantly lower than 2018 (Table 3). Abundance of many summer foods was below the long-term (35-year) average in all but the northwest region. In general, summer food conditions were average and patchy across the state (Table 4); in many areas, especially in the northeast and north-central, summer foods were delayed 2–3 weeks. On the other hand, fall foods tended to be average across the state (Table 5). The statewide fall food index (productivity of dogwood+oak+hazel), which helps predict annual harvest after accounting for hunter effort (Figures 9 & 10), was average in all regions. Hazelnut production was above average in much of the state (with patches near Bemidji, Grand Rapids, and Cloquet with exceptional production). Dogwood production was generally average to above-average across the range. Oak production was average across the state with patches of above-average production near Detroit Lakes and Cloquet. Stations in the North-Central and East-Central regions reported acorn drop nearly 2 weeks later than last year, which may have contributed to the high hunting success (and higher than expected harvest).

Predictions of harvest from food abundance

The 2019 statewide harvest was nearly 20% higher than expected (2340 actual vs. 1959 predicted), based on regression of harvest as a function of hunter numbers and the fall food productivity index (Figure 10). This regression is nearly as strong (and has accurately predicted previous harvests) when only the past 15 years are considered. For the quota zone, the actual harvest in 2019 was also nearly 20% higher (1659 actual vs. 1391 predicted) than predicted by this regression.

All data contained herein are subject to revision, due to updated information, improved analysis techniques, and/or regrouping of data for analysis.

Table 1. Number of bear hunting quota area permits available, 2013–2019. Highlighted values show a change from the previous year. BMUs 26 and 44 were divided into 27/28 and 46/47, respectively, in 2016.

BMU	2014	2015	2016		2017	2018	2019
			Before BMU	After BMU			
			split ^a	split			
12	200	150	150	150	125	125	125
13	250	250	250	250	225	225	225
22	50	50	50	50	50	50	50
24	200	200	200	200	175	175	175
25	500	500	500	500	400	400	400
26	350	350	325				
27				250	225	225	225
28				75	60	60	60
31	550	550	550	550	500	500	500
41	150	150	125	125	125	125	150
44	450	450	450				
46				400	350	350	350
47				50	40	40	40
45	150	150	250	250	175	175	200
51	900	900	1000	1000	900	900	900
Total	3750	3700	3850	3850	3350	3350	3400

^a In 2016, the Leech Lake Reservation was split from BMUs 26 and 44 to form BMUs 28 (north) and 47 (south), with the remaining area of BMU 26 renamed BMU 28 and remaining area of BMU 44 renamed BMU 46. The column shows permit allocation before the split in order to compare with previous years.

Table 2. Number of nuisance bear complaints registered by Conservation Officers and Wildlife Managers during 1999–2019, including number of nuisance bears killed and translocated, and bears killed in vehicular collisions.

	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017 ^j	2018 ^j	2019 ^j
Number of personnel participating in survey ^a	52	60	54	50	39	34	42	46	46	37	51	40	34	56	63	64	61	55	86 (51,35)	78 (56,23)	126 (60,66)
Complaints examined on site	189	105	122	75	81	75	61	57	63	59	65	70	37	113	69	79	97	118	71 (22,49)	40 (21,19)	82 (37,45)
Complaints handled by phone ^b	987	618	660	550	424	507	451	426	380	452	535	514	396	722	623	570	840	780	644 (450,194)	438 (369,69)	736 (599,137)
Total complaints received	1176	723	782	625	505	582	512	483	443	511	600	584	433	835	692	649	937	898	715	478	818
• % Handled by phone	84%	85%	84%	88%	84%	87%	88%	88%	86%	88%	89%	88%	91%	86%	90%	88%	90%	87%	90%	92%	90%
Bears killed by:																			22 (4,18)	9 ^k (4,5)	45 (5,40)
• Private party or DNR	25	25	22	12	13	25	28	11	21	22	23	22	9 ^k	16	24	26	45	53			
• Hunter before season ^c																					
– from nuisance survey	5	7	4	0	3	3	6	2	18	3	4	3	3	11	0	0	1	13	1	2	0
– from registration file	24	43	20	11	8	4	13	6	25	5	15	10	5	12	0	1	4	6	3	11	5
• Hunter during/after season ^d	0	1	1	0	0	0	1	0	0	0	0	0	0	0	1	0	1	1	1	0	0
• Hunter by Area 88 license ^e																			1		3 ^m
• Permittee ^f	7	2	6	4	6	1	5	4	5	1	3	5	0	0	1	0	3	0	0	1	2
Bears translocated	29	1	6	3	1	3	3	3	1	3	2	2	2	0	3	2	0	0	0	0	0
• % bears translocated ^g	15	1	5	4	1	4	5	5	2	5	3	3	5	0	4	3	0	0	0	0	0
Bears killed by cars ^h	60	39	43	26	25	16	22	18	20	27	18	28	15	33	32	28	47 ^h	27	9 (0,9) ^h	25 (15,10) ^h	16 (11,5) ^h

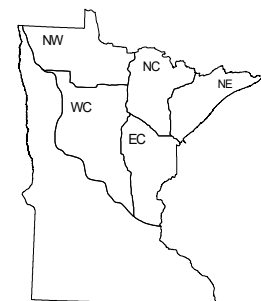
^a Maximum number of people turning in a nuisance bear report each month. Monthly reports were required beginning in 1984, and included cases of zero complaints. In 2017, the recording system was changed, where Wildlife Managers only recorded actual complaints (not zero complaints), generally at the time the complaint was received. Since then, the number reflects the total number of people receiving and recording at least 1 complaint during that year. For consistency, the records from Conservation Officers were handled the same way. Beginning July 2019, COs recorded complaints electronically and individually (as they occurred), similar to Wildlife Managers (but using a different recording system).

- ^b If a complaint was handled by phone, it means a site visit was not made.
- ^c The discrepancy between the number recorded on the nuisance survey and the number registered before the opening of the season indicates incomplete data. Similarity between the two values does not necessarily mean the same bears were reported.
- ^d Data only from nuisance survey because registration data do not indicate whether bear was a nuisance.
- ^e In 2017, hunters could choose Area 88 in the quota lottery, and if drawn, could hunt for a nuisance bear, if authorized (11 were authorized, 1 killed a bear). In 2018, Area 88 was only a designation for hunters willing to take a nuisance bear in the quota area on a no-quota license, if so authorized; 116 hunters were on this list. However, none of the 4 hunters with NQ licenses who killed a bear in the quota area (Table 5) were authorized to do so. It is not known from these records if others were authorized but did not kill a bear.
- ^f A permit for non-landowners to take a nuisance bear before the bear season was officially implemented in 1992, but some COs individually implemented this program in 1991. Data are based on records from the nuisance survey, not directly from permit receipts. Only 7 bears have been killed by permittees since 2011. In 2019, 7 permits were issued but only 2 bears killed.
- ^g Percent of on-site investigations resulting in a bear being captured and translocated. According to DNR nuisance policy, trapped nuisance bears should not be translocated.
- ^h Car kill data were reported on the monthly nuisance form beginning in 2005. In all previous years, car kill data were from Enforcement's confiscation records. In 2015, confiscation records had more car-kills than the nuisance survey (47 vs 33), so the higher number is shown here. In 2017, only 1 car-kill was in the confiscation records, and in 2018 there were just 2. In 2017, the electronic system used by managers did not allow for recording of car kills. In 2018, an effort was made to increase car-kill reporting by managers, which was further increased in 2019 by adding a distinct coding for non-confiscated car kills that were either observed or reported by the public.
- ^j Beginning in 2017, Wildlife Managers recorded nuisance bear complaints on an all-species wildlife damage app, whereas Conservation Officers continued to submit monthly nuisance bear survey forms (April–Oct). Beginning in 2019, COs also used an electronic app to record bear complaints (but a different app than wildlife). Because the 2 survey tools are not exactly the same, data are presented separately for each in parenthesis (Wildlife Managers, COs). For consistency, only April–October data are included (in 2017 managers recorded 10 calls in other months, in 2018 14 calls were in other months, in 2019 16 calls were in other months).
- ^k Lowest number of nuisance bears were killed in 2011 and 2018, since recording began in 1982.
- ^m 13 NQ hunters were authorized to take nuisance bears in the quota area in 2019, of which 3 were successful.

Table 3. Regional bear food indices^a in Minnesota's bear range, 1984–2019.

Shaded blocks indicate particularly low (<45; pink) or high (≥70; green) values.

Year	Survey Area					Rangewide
	NW	NC	NE	WC	EC	
1984	32.3	66.8	48.9	51.4	45.4	51.8
1985	43.0	37.5	35.3	43.5	55.5	42.7
1986	83.9	66.0	54.7	74.7	61.1	67.7
1987	62.7	57.3	46.8	67.4	69.0	61.8
1988	51.2	61.1	62.7	54.4	47.3	56.0
1989	55.4	58.8	48.1	47.8	52.9	51.6
1990	29.1	39.4	55.4	44.0	47.9	44.1
1991	59.7	71.2	64.8	72.1	78.9	68.4
1992	52.3	59.9	48.6	48.1	63.3	58.2
1993	59.8	87.8	75.0	73.9	76.8	74.3
1994	68.6	82.3	61.3	81.5	68.2	72.3
1995	33.8	46.5	43.9	42.0	50.9	44.4
1996	89.5	93.2	88.4	92.2	82.1	87.6
1997	58.2	55.5	58.8	62.0	70.1	63.9
1998	56.9	72.8	66.4	72.3	84.5	71.1
1999	63.7	59.9	61.1	63.2	60.6	62.0
2000	57.7	68.0	54.7	69.2	67.4	62.3
2001	40.6	48.7	55.6	62.2	66.0	55.8
2002	53.1	63.4	60.4	68.6	68.3	66.8
2003	59.1	57.5	55.2	58.6	49.7	58.8
2004	57.0	60.5	61.1	70.3	67.9	64.4
2005	53.4	65.9	61.4	59.9	72.6	62.3
2006	51.0	64.9	53.4	51.0	52.1	56.9
2007	68.4	79.0	67.3	67.6	70.0	69.4
2008	58.6	74.1	64.7	66.6	71.4	65.4
2009	59.9	67.8	63.2	69.2	69.5	66.5
2010	70.0	71.3	79.0	60.8	57.3	68.0
2011	61.4	59.6	57.9	66.7	63.5	62.5
2012	49.1	50.3	59.4	50.5	41.5	50.7
2013	71.9	77.1	76.0	59.1	63.2	71.8
2014	71.4	70.7	71.4	61.0	66.5	70.2
2015	47.2	56.3	44.8	57.2	46.5	50.7
2016	79.5	64.3	75.8	64.4	60.6	70.3
2017	67.1	57.5	56.2	70.6	73.9	61.3
2018	72.6	82.4	101.8 ^b	71.5	88.3 ^b	83.9 ^b
2019	68.8	60.9	64.4	59.8	65.1	63.9



^a Each bear food index value represents the sum of the mean index values for 14 species, based on surveys conducted in that area. Range-wide mean is derived directly from all surveys conducted in the state (i.e., not by averaging survey area means).

^b Record high food rating in NE and EC regions, and second-highest statewide.

Table 4. Regional mean index values^a for bear food species in 2019 compared to the previous 35-year mean (1984-2018) in Minnesota's bear range. Shading indicates particularly high (green) or low (pink) fruit abundance relative to average (≥ 1 point difference for individual foods; ≥ 5 points difference for totals).

FRUIT	NW		NC		NE		WC		EC		Rangewide	
	35yr mean	2019 (n = 10 ^b)	35yr mean	2019 (n = 8)	35yr mean	2019 (n = 6)	35yr mean	2019 (n = 10)	35yr mean	2019 (n = 8)	35yr mean	2019 (n = 33)
SUMMER												
Sarsaparilla	4.7	5.3	5.9	6.9	5.3	6.5	4.5	3.5	5.3	3.8	5.0	5.2
Pincherry	3.4	4.3	4.5	3.8	4.3	4.0	3.8	3.1	3.7	3.0	3.9	3.4
Chokecherry	5.8	6.2	5.5	4.0	4.7	4.3	5.4	4.4	4.7	5.0	5.3	4.6
Juneberry	5.2	5.1	4.9	2.6	5.1	3.8	3.7	2.6	4.0	3.8	4.5	3.9
Elderberry	1.6	2.3	3.0	2.6	3.6	5.5	3.1	3.5	3.2	4.3	3.0	3.1
Blueberry	5.1	4.6	5.5	3.7	5.1	6.2	3.7	5.1	3.9	5.0	4.5	4.9
Raspberry	6.5	7.1	7.9	5.9	8.0	7.7	7.1	8.1	7.1	7.1	7.2	6.9
Blackberry	1.3	2.3	2.4	2.6	1.2	0.0	3.6	3.5	4.4	3.2	2.9	2.9
FALL												
Wild Plum	2.3	3.4	1.9	1.6	1.3	1.5	2.7	1.9	2.4	1.1	2.3	2.3
HB Cranberry	5.3	6.9	4.4	4.4	4.0	4.4	3.8	3.5	3.8	6.1	4.2	4.7
Dogwood	6.2	6.9	5.7	4.6	4.9	5.2	5.9	3.9	5.9	4.3	5.8	5.1
Oak	3.5	3.8	3.2	3.9	2.0	2.0	5.9	5.9	5.7	6.9	4.5	5.1
Mountain Ash	1.6	2.5	2.6	3.0	4.7	6.0	1.8	2.5	2.4	4.2	2.7	4.0
Hazel	6.4	8.1	7.4	11.3	7.0	7.3	7.9	8.5	7.7	7.3	7.2	7.9
TOTAL^d	59.0	68.8	64.6	60.9	61.2	64.4	62.8	59.8	64.2	65.1	63.0	63.9

^a Food abundance indices were calculated by multiplying species abundance ratings x fruit production ratings.

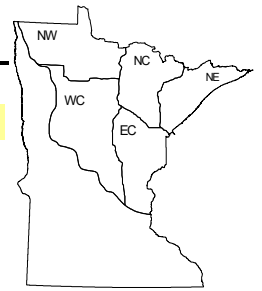
^b n = Number of surveys used to calculate area-specific means

^c Sample size for the entire range does not equal the sum of the sample sizes of 5 survey areas because some surveys were conducted on the border of 2 or more areas and were included in calculations for both.

^d Because of rounding error, these totals may be slightly different than the sum of adding down the columns.

Table 5. Regional productivity index^a for important fall bear foods (oak + hazel + dogwood), 1984–2019. Particularly low (≤ 5.0 ; yellow) or high (≥ 8.0 ; tan) values are shaded.

Year	Survey Area					Entire Range
	NW	NC	NE	WC	EC	
1984	4.2	7.6	7.0	6.2	7.0	6.5
1985	4.9	2.8 ^b	4.2	4.7	5.3	4.4 ^b
1986	7.2	5.0	4.0	7.0	6.2	6.2
1987	8.0	7.8	7.3	7.6	8.0	7.7
1988	5.5	7.2	7.3	6.8	6.1	6.7
1989	6.0	5.3	4.1	5.7	6.4	5.8
1990	3.3 ^b	4.2	6.4	5.7	6.4	5.2
1991	6.2	6.2	5.4	7.2	7.7	6.7
1992	4.7	5.0	4.4	4.4 ^b	6.8	5.1
1993	5.3	7.1	6.7	6.2	7.7	6.5
1994	7.1	7.8	5.8	7.8	7.1	7.2
1995	4.8	4.8	5.1	4.6	5.3	4.9
1996	8.7	8.6	8.1	9.2	8.5	8.6
1997	5.8	5.4	5.1	6.8	6.5	6.2
1998	5.8	6.0	6.3	7.1	7.8	6.7
1999	6.4	5.1	5.9	6.6	6.0	6.2
2000	5.8	7.7	7.2	7.5	8.5	7.0
2001	3.4	4.1	5.7	6.0	6.5	5.2
2002	8.7	7.1	6.6	8.8	8.2	8.1
2003	6.3	6.0	5.5	6.2	6.0	6.1
2004	6.1	5.4	5.4	6.4	6.1	5.9
2005	5.8	5.8	6.1	6.4	7.0	6.2
2006	6.7	6.1	6.0	6.7	5.8	6.3
2007	6.0	5.8	5.7	6.6	6.4	6.2
2008	6.6	7.3	6.2	7.0	8.9	7.1
2009	5.1	6.2	5.3	6.3	6.5	6.0
2010	7.7	6.4	6.5	6.2	5.4	6.6
2011	5.8	6.5	6.2	7.0	7.4	6.5
2012	6.2	6.3	6.3	6.5	4.8	6.1
2013	6.8	6.0	5.7	6.7	6.9	6.3
2014	7.0	5.6	5.4	7.7	6.1	6.7
2015	5.8	5.9	3.5 ^b	8.2	3.7 ^b	5.6
2016	5.7	5.2	6.0	5.4	5.2	5.3
2017	6.8	5.6	5.1	7.4	7.1	6.5
2018	5.8	6.1	7.7	8.3	8.4	7.2
2019	6.2	7.1	6.6	6.5	7.1	6.7



^a Values represent the sum of mean production scores for hazel, oak, and dogwood, derived from surveys conducted in each survey area. Range-wide mean is for all surveys conducted in the state (i.e. not an average of survey area means).

^b Record low fall food score in survey area.

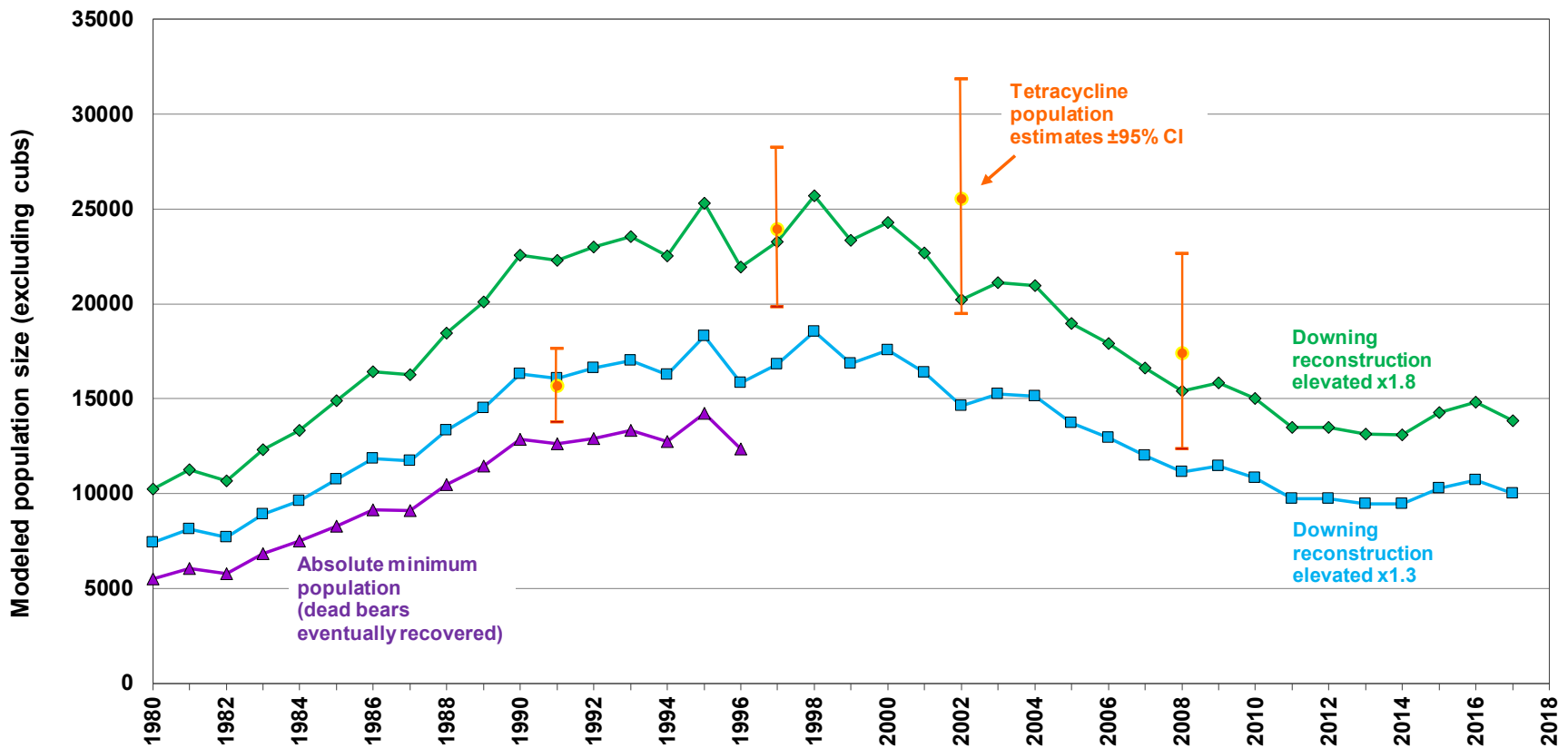


Figure 1. Statewide bear population trend (pre-hunt) derived from Downing reconstruction, scaled (elevated to account for non-harvest mortality) to various degrees to attempt to match the tetracycline-based mark-recapture estimates (2 such curves shown here; estimates beyond 2017 are unreliable).

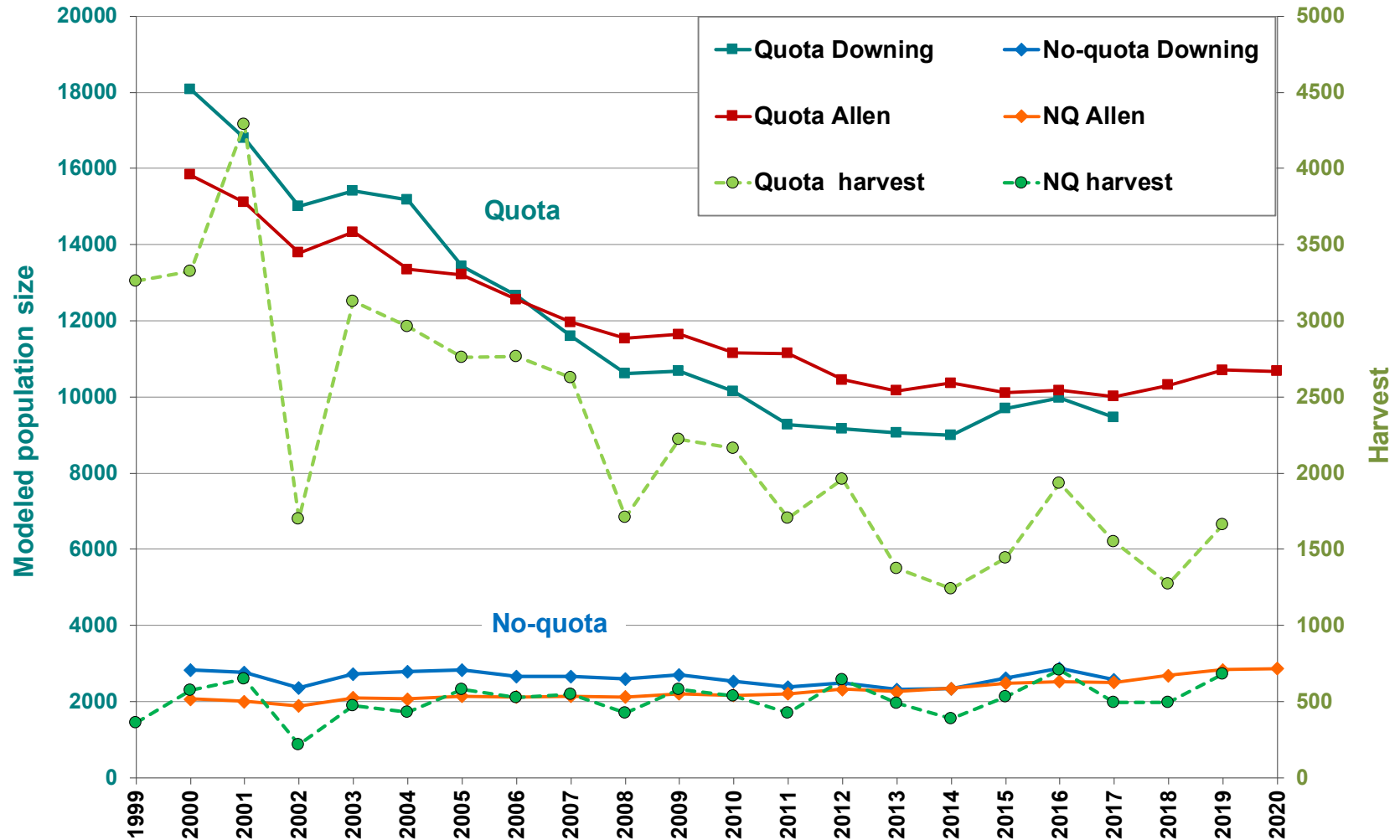


Figure 2. Population trends during 2000s derived from two independent population models (Downing and Allen) for quota and no-quota zones, compared to respective harvests. Downing reconstruction-based estimates <2 years from the most recent harvest age data are unreliable (hence these curves terminate 2017). Downing curves were scaled (elevated to account for non-harvest mortality) to fall between the two curves in Fig. 18 (i.e., the actual scale of the population estimates is not empirically-based, but happens to approximately match the magnitude of the Allen estimates).

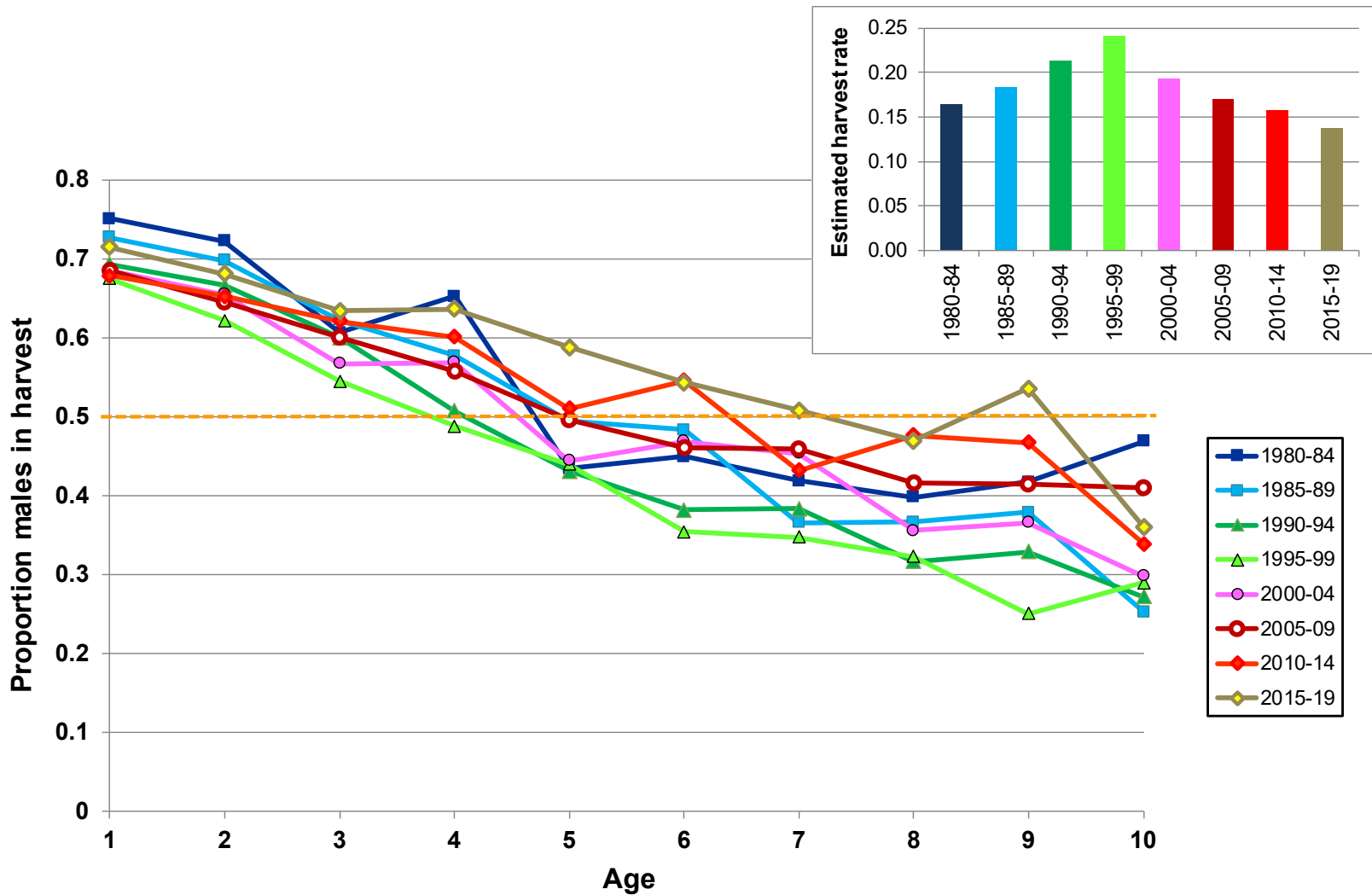


Figure 3. Trends in proportion of male bears in statewide harvest at each age, 1–10 years, grouped in 5-year time blocks, 1980–2019. Higher harvest rates result in steeper curves because males in the living population are reduced faster than females. Fitting a line to the data for each time block and predicting the age at which 50% of the harvest is male (dashed tan horizontal line) yields approximately the inverse of the harvest rate (derived rates are shown in inset). Flatter curves in recent years indicate lower harvest rates (2015–19 lower than 1980–84).

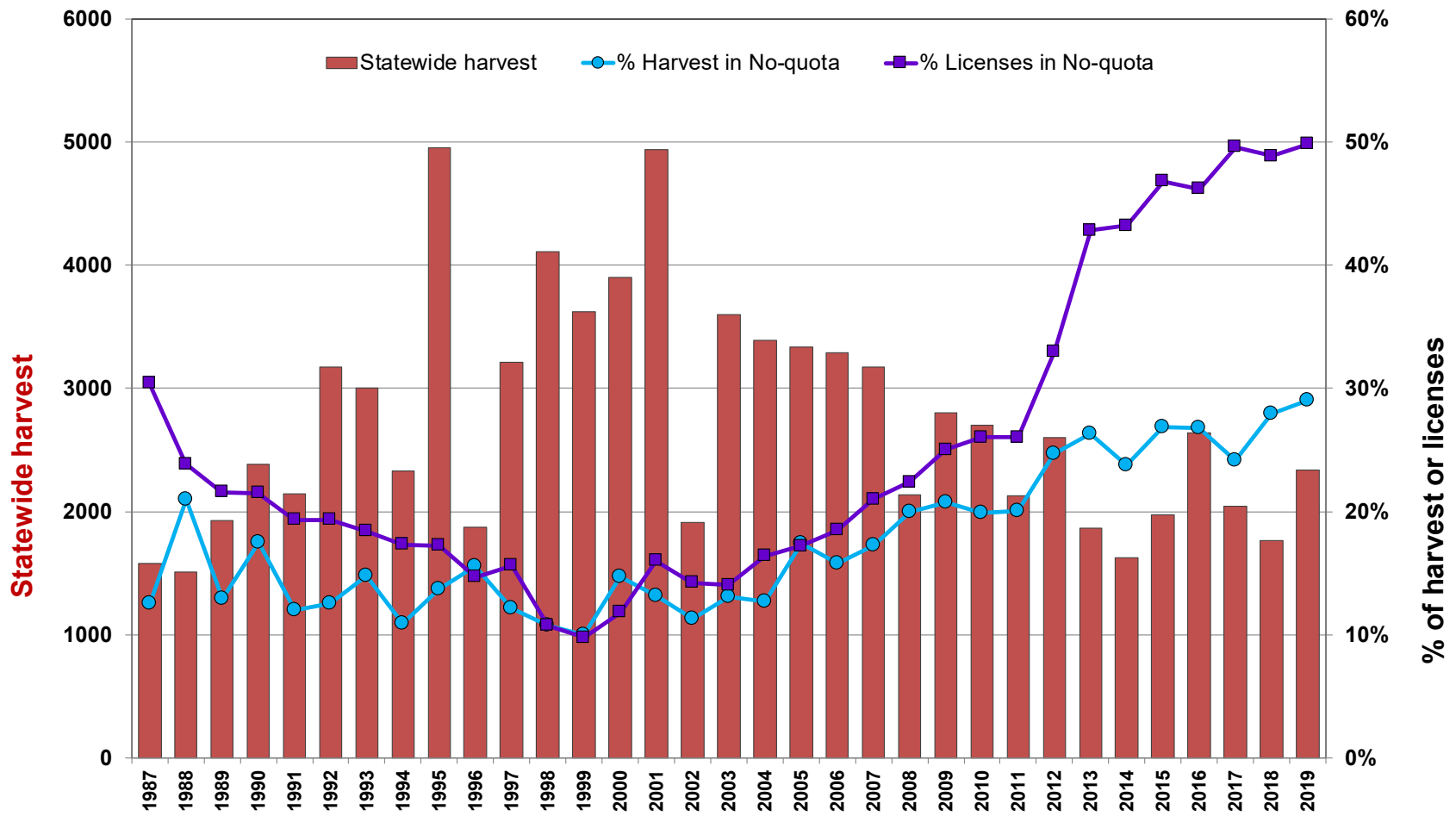


Figure 4. Trends in statewide bear harvest and proportions of harvest and licenses in the no-quota zones, 1987–2019.

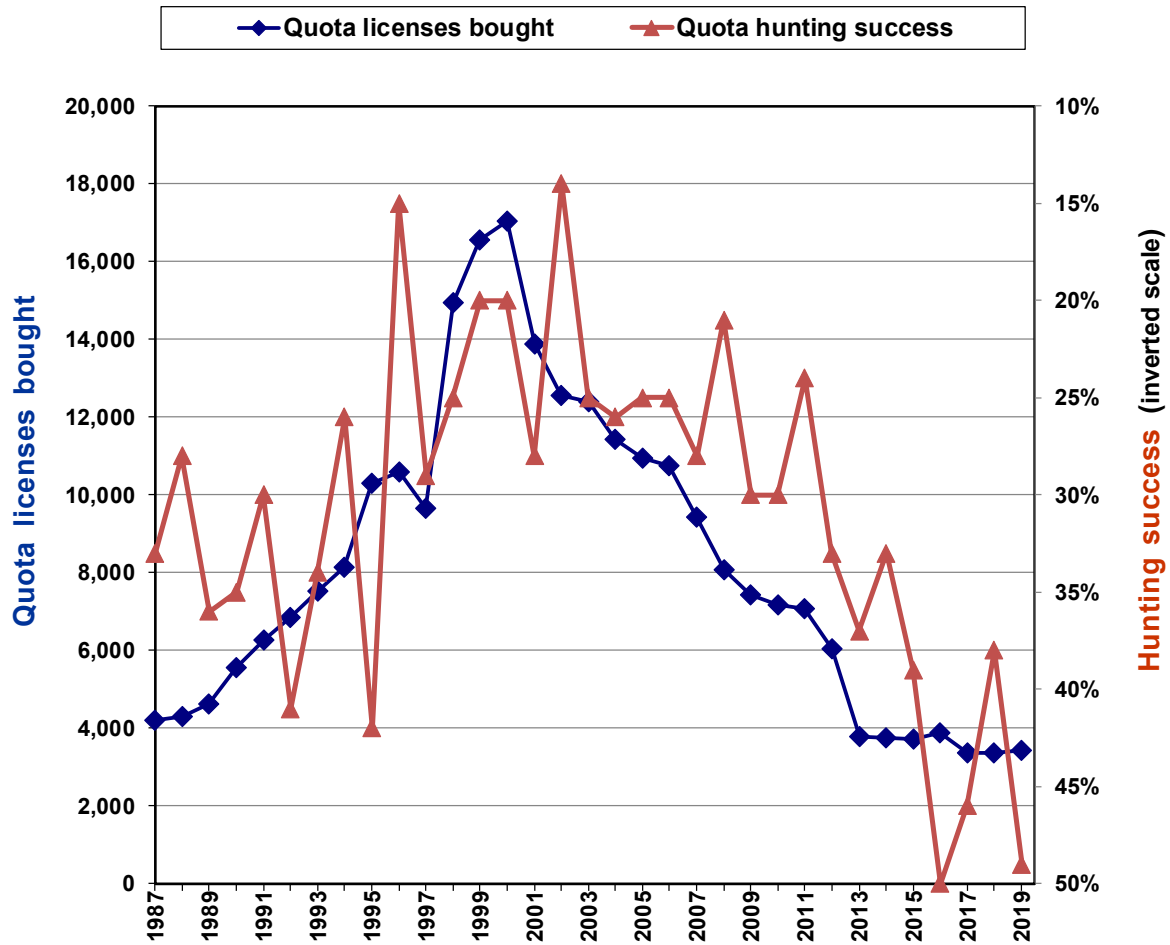


Figure 5. Relationship between licenses sold and hunting success (note inverted scale) in quota zone, 1987–2019 (quota and no-quota zones first partitioned in 1987). Number of licenses explains 53% of variation in hunting success during this period. Large variation in hunting success is also attributable to food conditions (e.g., during 2013–2019, when licenses were held relatively constant).

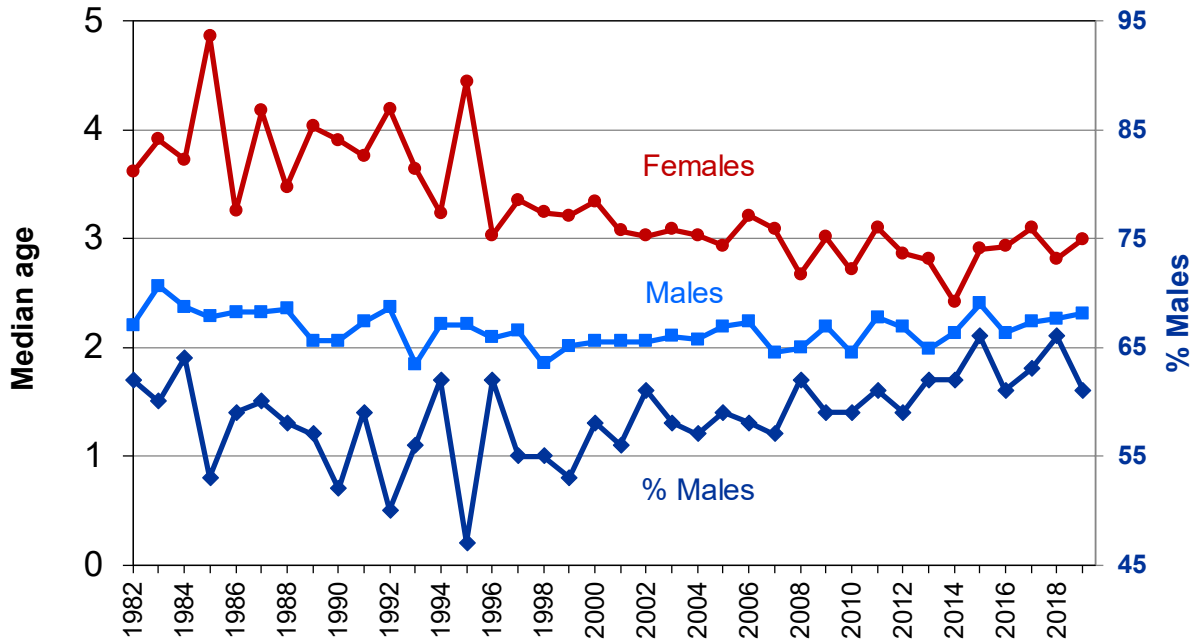


Figure 6. Statewide median ages (years) and sex ratio of harvested bears, 1982–2019.

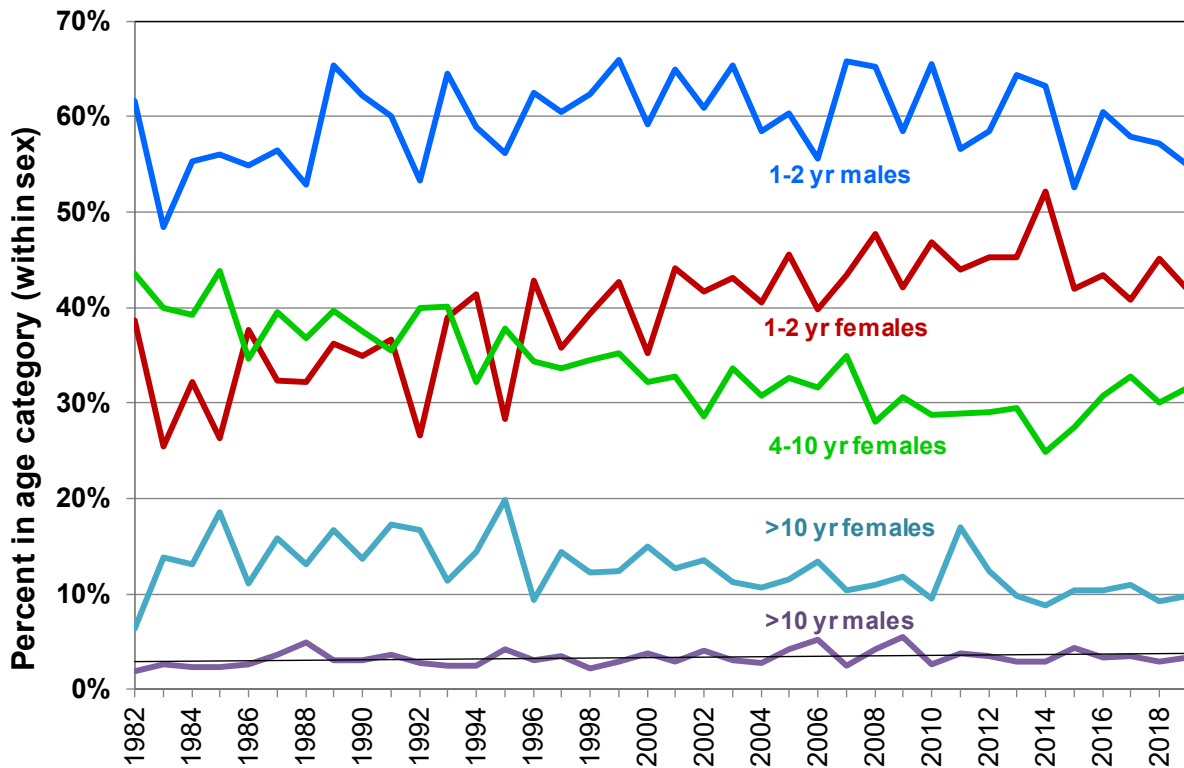


Figure 7. Statewide harvest structure: proportion of each sex in age category, 1982–2019.

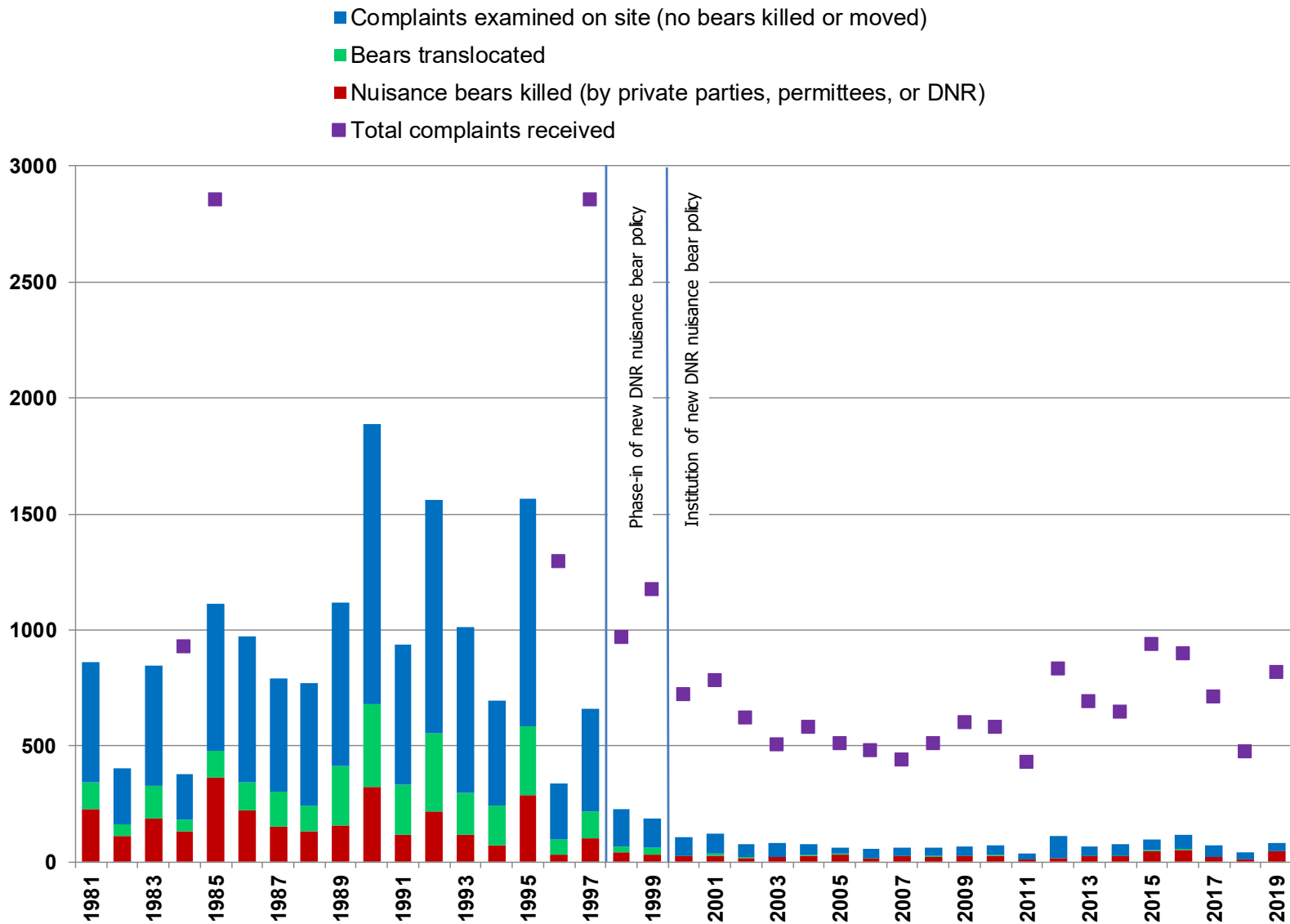


Figure. 8. Trends in nuisance bear complaints, and nuisance bears killed and moved, 1981–2019, showing dramatic effect of change in nuisance bear policy, and slight increasing trend over past decade.

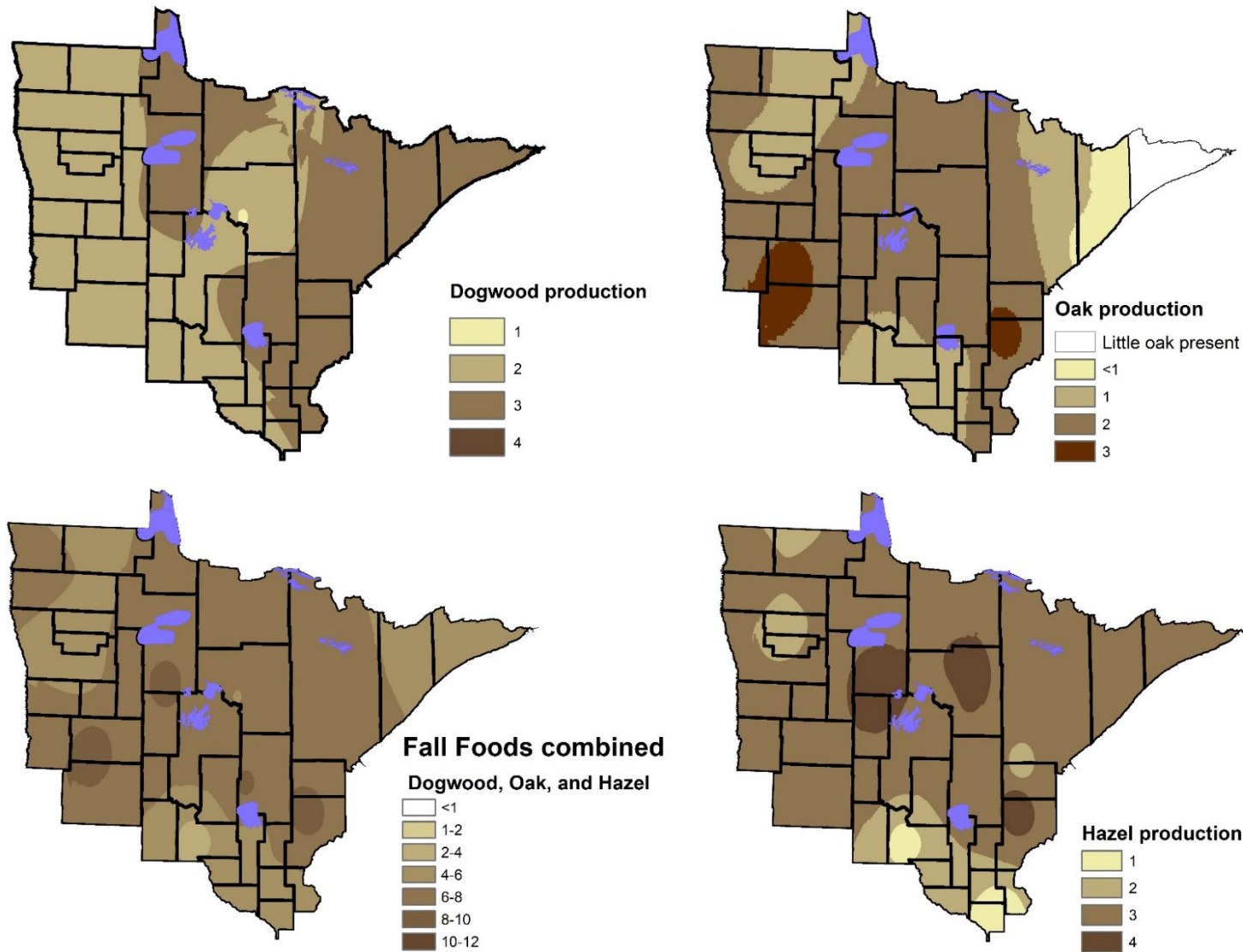


Figure 9. Production of fall bear foods (dogwood, oak, hazel) across Minnesota, 2019.

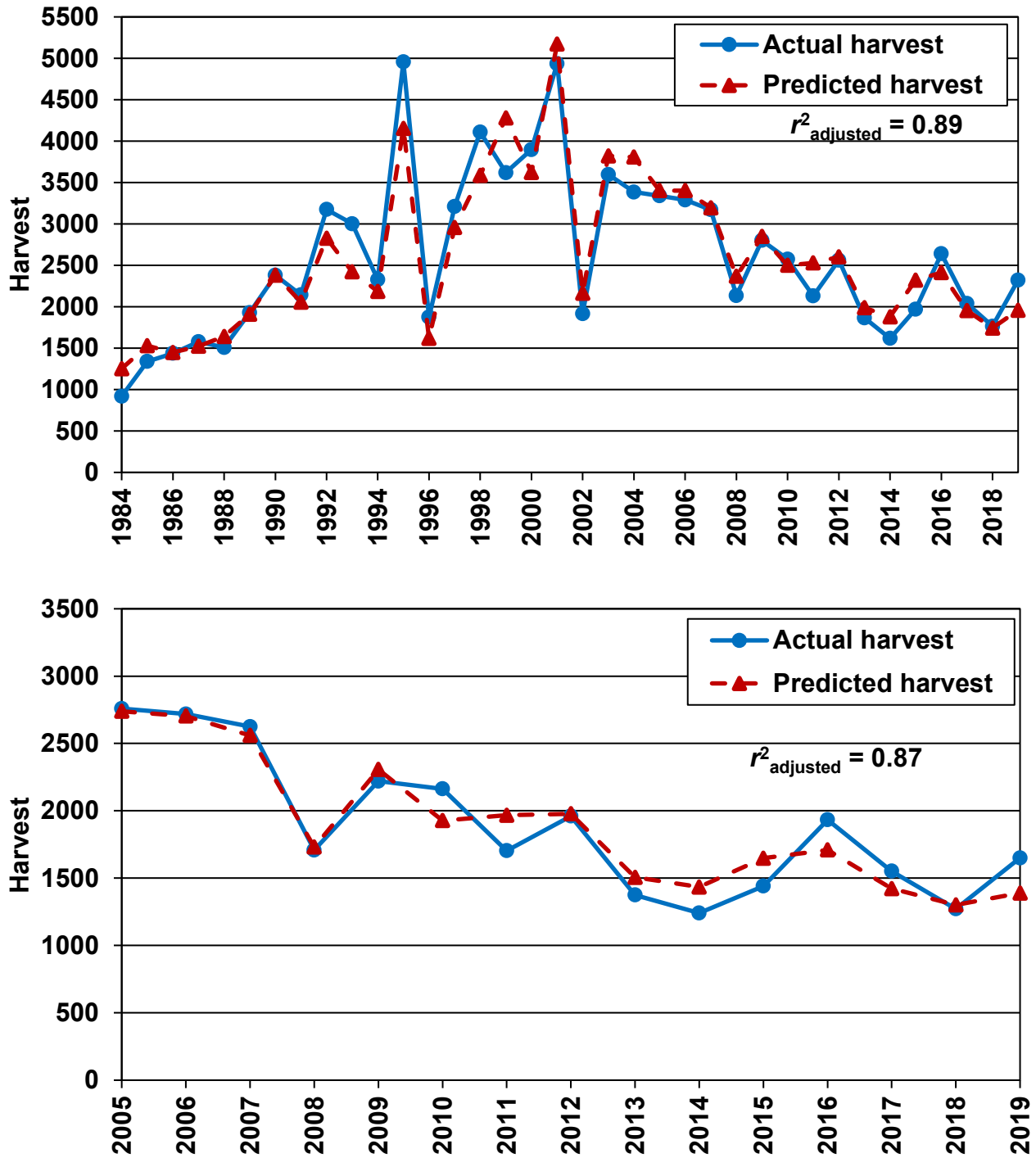


Figure 10. Number of bears harvested vs. number predicted to be harvested based on number of hunters and fall food production — top panel: statewide 1984–2019; bottom panel: quota zone only, most recent 15 years. Regression for both datasets included an interaction term between food and hunters to better predict the drastic changes in harvest when fall foods were extremely high or low.



2020 MINNESOTA SPRING GROUSE SURVEYS

Charlotte Roy, Forest Wildlife Populations and Research Group

SUMMARY OF FINDINGS

The Minnesota DNR coordinates ruffed grouse (*Bonasa umbellus*) and sharp-tailed grouse (*Tympanuchus phasianellus*) surveys each spring with the help of wildlife staff and cooperating federal, tribal, and county biologists. In 2020, Governor Walz issued Executive Orders 20-20 and 20-33, directing state employees to stay home except to perform essential work during 27 March – 4 May because of the COVID-19 pandemic. Federal and tribal biologists were exempted from these Executive Orders and were able to conduct fieldwork while social distancing to prevent the spread of COVID-19. Observers conducted ruffed grouse surveys in the northern portion of the state between 21 April – 13 May, but surveys in southern regions did not occur during the survey window. Mean ruffed grouse drums per stop (dps) were 1.6 statewide (95% confidence interval = 1.2–1.9) which is similar to last year, but the lack of surveys in the southern part of the state, a region where survey counts are usually lower, is likely causing the statewide index to be higher than it would be if southern regions were included. In the northern survey regions, counts were similar to or down from last year. High points in the population cycle occur on average every 10 years, and surveys indicate that the last peak was in 2017. Ruffed grouse in Minnesota are currently in the declining phase of the 10-year cycle.

DNR Wildlife Staff did not conduct sharp-tailed grouse surveys during 2020 due to the Governor's Stay at Home Orders during the COVID-19 pandemic. Thus, sharp-tailed grouse survey data are not available this year.

INTRODUCTION

The ruffed grouse (*Bonasa umbellus*) is the most popular game bird in Minnesota, with an annual harvest of 200,000–500,000 birds. Ruffed grouse hunter numbers have been as high as 92,000 during the last decade, although hunter numbers did not peak with recent peaks in grouse numbers, as they have traditionally. Sharp-tailed grouse (*Tympanuchus phasianellus*) are also popular among hunters, with an annual harvest of 5,000–22,000 birds since the early-1990s and 4,000–10,000 hunters in Minnesota.

The Minnesota DNR coordinates grouse surveys each year to monitor changes in grouse populations through time. These surveys provide a reasonable index to population trends, when the primary source of variation in counts among years is change in densities. However, weather, habitat conditions, observer ability, and grouse behavior, also vary over time and can influence survey counts. Thus, making inferences from survey data over short time periods (e.g., a few years) can be tenuous. Nevertheless, over longer time periods and when large changes in index values occur, these surveys can provide a reasonable index to long-term grouse population trends. Spring surveys provide evidence that the ruffed grouse population cycles at approximately 10-year intervals. The spring survey data also correlated strongly with the fall harvest before the early 2000s, but in recent decades, this relationship has weakened.

The first surveys of ruffed grouse in Minnesota occurred in the mid-1930s, and the first spring survey routes were established along roadsides in 1949. By the mid-1950s, ~50 routes were established with ~70 more routes added during the late-1970s and early-1980s. Since then, staff and cooperators have conducted spring drumming counts annually to survey ruffed grouse in the forested regions of the state where ruffed grouse habitat occurs. Drumming is a low sound produced by males as they beat their wings rapidly and in increasing frequency to signal the location of their territory. These drumming displays also attract females that are ready to begin nesting, so the frequency of drumming increases in the spring during the breeding season. The sound produced when male grouse drum is easy to hear and thus drumming counts are a convenient way to survey ruffed grouse populations in the spring.

The first surveys of sharp-tailed grouse in Minnesota occurred between the early-1940s and 1960. The current survey is based on counts at dancing grounds during the spring and was first conducted in 1976. Male sharp-tailed grouse display, or dance, together in open areas to attract females in the spring. This display consists of the males stomping their feet with out-stretched wings. Females visit the dancing grounds to select males for breeding. These dancing grounds, or leks, are reasonably stable in location from year to year, allowing surveyors to visit and count individuals each spring. Staff and cooperators conduct surveys in openland portions of the state where sharp-tailed grouse persist, although sharp-tailed grouse were formerly much more widely distributed in Minnesota at the early part of the 20th century.

METHODS

Ruffed Grouse

Observers conducted ruffed grouse surveys along established routes throughout the state. Each route consisted of 10 listening stops at approximately 1.6-km (1-mile) intervals. The placement of routes on the landscape was determined from historical survey routes, which were originally placed near ruffed grouse habitat in low traffic areas. Annual sampling of these historical routes provides information about temporal changes along the routes but may not be representative of the counties or regions where the routes occurred.

I engaged survey observers from among state, federal, tribal, private, and student biologists that had a professional background in wildlife science. Most observers had previously participated in the survey. I provided each observer a set of instructions and route location information but did not provide formal survey training. I asked participants to conduct surveys at sunrise during peak drumming activity (in April or May) on days that had little wind and no precipitation. I provided guidance about the timing of the usual peak in drumming but allowed flexibility in timing to match the peak if it occurred outside the usual survey windows. Each observer drove the survey route once and listened for drumming at each stop for 4 minutes. Observers recorded the number of drums heard at each stop (not necessarily the number of individual grouse), along with information about phenology and weather at the time of the survey.

I used the number of drums heard per stop (dps) as the survey index value. I determined the mean dps for each route, for each survey region (Figure 1), and for the entire state. For each survey region, I calculated the mean of route-level means for all routes partially or entirely within each Ecological Classification System (ECS) section. Routes that traversed section boundaries were included in the means for both sections. Because the number of routes within sections was not related to any proportional characteristic, I used the weighted mean of index values for the 4 ECS sections in the Northeast region and the 7 ECS sections in the state. I used the geographic area of the section as the weight for each section mean (i.e., Lake Agassiz, Aspen Parklands = 11,761 km², Northern Minnesota and Ontario Peatlands = 21,468 km², Northern Superior Uplands = 24,160 km², Northern Minnesota Drift and Lake Plains = 33,955 km², Western Superior Uplands = 14,158 km², Minnesota and Northeast Iowa Morainal (MIM) =

20,886 km², and Paleozoic Plateau (PP) = 5,212 km²). I reduced the area used to weight drum index means for the MIM and PP sections to reflect the portion of these areas within ruffed grouse range (~50%) using subsection boundaries. I calculated a 95% confidence interval (CI) to convey the uncertainty of each mean index value using 10,000 bootstrap samples of route-level means for survey regions and the whole state. I defined confidence interval boundaries as the 2.5th and 97.5th percentiles of bootstrap frequency distributions.

The COVID-19 pandemic affected the operation of ruffed grouse surveys in 2020. Governor Walz restricted non-essential fieldwork under Executive Orders 20-20 and 20-33 during 27 March – 4 May to prevent the spread of COVID-19. Usually ruffed grouse drumming surveys are completed during 8 – 30 April in the southern part of the state, 15 April – 5 May in the central region, and 20 April – 10 May in the northern part of the state, with some flexibility to match the peak of drumming when it occurs outside these survey windows. Thus, DNR Wildlife Staff missed the entire survey window in the southern region, began at the tail end of the window in the central region, and began in the last quarter of the survey window in the north. Nevertheless, some cooperators were exempt from the Executive Orders (e.g., federal biologists with the Forest Service and tribal biologists), and these cooperators conducted surveys when DNR Wildlife Staff could not. Many external cooperators also ran additional routes to maintain the integrity of the long-term data set, which is of value to their respective natural resource agencies, and because they could perform the survey while adhering to social distancing guidelines. The Governor issued Executive Order 20-48, allowing for “field research, monitoring, and surveying” to resume on 4 May, and the DNR Commissioner approved some DNR Wildlife Staff to conduct ruffed grouse surveys on 4 May. Observers conducted a few ruffed grouse surveys in the southern part of the state after the usual survey window and the peak in drumming, but I did not include these surveys in the analysis to ensure consistency in protocol and better comparability among years. General adherence to these protocols facilitates interpretation of population patterns in the context of the 10-year cycle.

SHARP-TAILED GROUSE

Wildlife staff and volunteers usually survey known sharp-tailed grouse lek locations in the Northwest (NW) and East Central (EC) portions of the state (Figure 2) during the peak in lek attendance, which usually occurs in the latter half of April and the first week of May. Although Governor Walz exempted “field research, monitoring and surveying” on 4 May, staff participation in sharp-tailed grouse surveys was not approved in time to complete the work. Therefore, DNR Wildlife Staff did not conduct any sharp-tailed grouse surveys during the peak in lek attendance in 2020. Unlike ruffed grouse surveys, few external cooperators participate in sharp-tailed grouse surveys.

RESULTS & DISCUSSION

Ruffed Grouse

Observers from 11 cooperating organizations surveyed 102 routes (80% of all routes) between 21 April and 13 May 2020, with 84% of northern routes completed and 42% of southern routes completed. Most routes (89%) were surveyed between 21 April and 10 May, with a median survey date of May 6, which is similar to last year (May 4) and the median survey date for the most recent 10 years (May 3). Observers reported Excellent (61%), Good (37%), and Fair (2%) survey conditions for 95 routes reporting conditions.

Statewide counts of ruffed grouse drums averaged 1.6 dps (95% confidence interval = 1.2–1.9 dps) during 2020 (Figure 3). Drum counts were 1.7 (1.3–2.0) dps in the Northeast survey region ($n = 92$ routes), 1.2 (1.0–1.3) dps in the Northwest survey region ($n = 5$), 1.2 (0.4–2.2) dps in the

Central Hardwoods survey region ($n = 10$), and no routes were completed during the appropriate survey window in the Southeast survey region (Figure 4a-d).

Statewide drum counts were similar to last year. I received 5 surveys from 2019 after the report was written last year, and updated results are included here. The southern survey regions tend to have lower average counts than the northern regions each year, and because southern regions were not surveyed in 2020, the statewide index is likely higher than it would be if southern routes were included. In the Northeast and Northwest, counts were similar to or down from last year, respectively. In the Central Hardwoods, observers surveyed only the northern portion of the region where counts tend to be higher, which likely explains the slightly higher, although not statistically different, dps in this region in 2020 compared to 2019. The most recent peak in the 10-year cycle occurred in 2017. Although peaks in the cycle occur on average approximately every 10 years, they vary from 8 to 11 years apart (Figure 3). Recent survey data indicate that ruffed grouse are in the declining phase of the 10-year cycle in Minnesota.

Sharp-tailed Grouse

Cooperators and staff did not collect sharp-tailed grouse survey data during the survey window in 2020. Survey data from recent years indicate a declining trend in both the NW and EC survey regions. The EC population has declined the most, with fewer than 300 birds (163 – 286) counted annually since 2013, and a drop in leks counted from 70 in 2010 to 30 in 2019. Continued monitoring of these populations will provide information that wildlife managers can use to make management decisions.

ACKNOWLEDGMENTS

I would like to extend a special thanks to federal biologists from the Superior National Forest (USDA Forest Service), and tribal biologists with 1854 Treaty Authority and White Earth Reservation for surveying additional ruffed grouse routes this spring while exempted from the Governor's Stay at Home Order. The extra efforts of H. Becker, T. Brannock, D. Garrison, D. Grosshuesch, S. Malick-Wahls, D. McArthur, D. Ryan, S. Swanson, M. Swingen, and others ensured that surveys were conducted during the appropriate temporal window, and that survey data collected annually since 1949 and used by numerous natural resource agencies and cooperators to make decisions, could continue during the pandemic. The ruffed grouse survey was also accomplished this year through the combined efforts of staff and volunteers at Chippewa National Forests; Fond du Lac and Red Lake Reservations; Blandin Paper; Beltrami County and Cass County Land Departments; and DNR staff at Baudette, Bemidji, Cloquet, Crookston, Detroit Lakes, Fergus Falls, Grand Rapids, International Falls, Park Rapids, Red Lake WMA, Roseau River WMA, Sauk Rapids, Thief Lake WMA, Thief River Falls, Tower, and Two Harbors work areas. Several other staff and volunteers usually participate in the ruffed grouse drumming survey but were not able to during the appropriate survey window due to the pandemic. These work areas included Aitkin, Brainerd, Carlos Avery Wildlife Management Area, Karlstad, Little Falls, Mille Lacs WMA, Rochester, Whitewater WMA, and Winona work areas, as well as Vermilion Community College, Leech Lake Reservation, and Agassiz National Wildlife Refuge. Gary Drotts, John Erb, and Rick Horton organized an effort to enter the ruffed grouse survey data for 1982–2004, and Doug Mailhot and another volunteer helped enter the data. In 2020, Jackson Bates and Nicole Dotson entered ruffed grouse survey data for 1979-1981. I would also like to thank Lindsey Shartell and Leslie McInenly for making helpful comments on this report. This work was funded in part through the Federal Aid in Wildlife Restoration Act.

Table 1. Sharp-tailed grouse / lek (≥ 2 males) at all leks observed during spring surveys each year in Minnesota.

Year	Statewide			Northwest ^a			East Central ^a		
	Mean	95% CI ^b	<i>n</i> ^c	Mean	95% CI ^b	<i>n</i> ^c	Mean	95% CI ^b	<i>n</i> ^c
2004	11.2	10.1 – 12.3	183	12.7	11.3 – 14.2	116	8.5	7.2 – 9.9	67
2005	11.3	10.2 – 12.5	161	13.1	11.5 – 14.7	95	8.8	7.3 – 10.2	66
2006	9.2	8.3 – 10.1	161	9.8	8.7 – 11.1	97	8.2	6.9 – 9.7	64
2007	11.6	10.5 – 12.8	188	12.7	11.3 – 14.1	128	9.4	8.0 – 11.0	60
2008	12.4	11.2 – 13.7	192	13.6	12.0 – 15.3	122	10.4	8.7 – 12.3	70
2009	13.6	12.2 – 15.1	199	15.2	13.4 – 17.0	137	10.0	8.5 – 11.7	62
2010	10.7	9.8 – 11.7	202	11.7	10.5 – 12.9	132	8.9	7.5 – 10.5	70
2011	10.2	9.5 – 11.1	216	11.2	10.2 – 12.2	156	7.8	6.7 – 8.9	60
2012	9.2	8.2 – 10.3	153	10.7	9.3 – 12.3	100	6.3	5.4 – 7.3	53
2013	9.2	8.2 – 10.2	139	10.5	9.3 – 11.7	107	4.8	3.8 – 5.9	32
2014	9.8	8.8 – 10.9	181	10.9	9.8 – 12.1	144	5.4	4.5 – 6.4	37
2015	9.8	8.9 – 10.7	206	10.8	9.9 – 11.9	167	5.3	4.4 – 6.4	39
2016	9.5	8.6 – 10.5	182	10.2	9.2 – 11.4	152	6.0	4.9 – 7.3	30
2017	9.7	8.7 – 10.8	181	10.4	9.2 – 11.8	141	7.2	5.8 – 8.6	40
2018	9.3	8.4 – 10.3	161 ^d	9.8	8.8 – 10.9	130	7.3	5.4 – 9.6	30
2019	10.2	9.1 – 11.4	152	11.0	9.7 – 12.3	122	7.2	5.4 – 9.5	30
2020	NA ^e	NA	NA	NA	NA	NA	NA	NA	NA

^a Survey regions; see Figure 1.

^b 95% CI = 95% confidence interval

^c *n* = number of leks in the sample.

^dOne lek was located just south of the NW region in Clearwater County.

^eNo data were collected in 2020 due to the Governor's Stay at Home Order during the COVID-19 pandemic.

Table 2. Difference in the number of sharp-tailed grouse / lek observed during spring surveys of the same lek in consecutive years in Minnesota.

Comparison ^b	Statewide			Northwest ^a			East Central ^a		
	Mean	95% CI ^c	<i>n</i> ^d	Mean	95% CI ^c	<i>n</i> ^d	Mean	95% CI ^c	<i>n</i> ^d
2004 – 2005	-1.3	-2.2 – -0.3	186	-2.1	-3.5 – -0.8	112	0.0	-1.0 – 1.1	74
2005 – 2006	-2.5	-3.7 – -1.3	126	-3.6	-5.3 – -1.9	70	-1.1	-2.6 – -0.6	56
2006 – 2007	2.6	1.5 – 3.8	152	3.3	1.7 – 5.1	99	1.2	0.1 – 2.3	53
2007 – 2008	0.4	-0.8 – 1.5	166	0.0	-1.6 – 1.6	115	1.2	0.1 – 2.5	51
2008 – 2009	0.9	-0.4 – 2.3	181	1.8	-0.1 – 3.8	120	-0.8	-2.1 – -0.6	61
2009 – 2010	-0.6	-1.8 – 0.6	179	-0.8	-2.6 – 1.0	118	-0.1	-1.2 – 1.0	61
2010 – 2011	-1.7	-2.7 – -0.8	183	-1.8	-3.1 – -0.5	124	-1.5	-2.8 – -0.3	59
2011 – 2012	-2.0	-2.9 – -1.1	170	-1.7	-2.9 – -0.4	112	-2.4	-3.3 – -1.6	58
2012 – 2013	-0.8	-2.0 – 0.4	140	0.4	-1.3 – 2.3	88	-2.9	-4.2 – -1.8	52
2013 – 2014	1.4	0.1 – 2.7	121	1.6	-0.3 – 3.5	79	1.1	-0.1 – 2.3	42
2014 – 2015	-0.2	-1.4 – 0.9	141	-0.3	-1.9 – 1.3	102	-0.1	-1.1 – 1.1	39
2015 – 2016	-1.3	-2.3 – -0.2	167	-1.6	-2.9 – -0.2	129	-0.2	-1.3 – 0.9	38
2016 – 2017	-0.3	-1.5 – 0.9	166	-0.3	-1.8 – 1.2	128	-0.2	-1.2 – 0.8	38
2017 – 2018	-2.2	-3.3 – -1.1	159 ^e	-2.4	-3.9 – -0.4	123	-1.4	-2.8 – 0.2	36
2018 – 2019	-0.3	-1.5 – 1.0	132	0.0	-1.5 – 1.6	101	-1.4	-3.0 – 0.1	31
2019 – 2020 ^f	NA	NA	NA	NA	NA	NA	NA	NA	NA

^a Survey regions; see Figure 1.

^b Consecutive years for which comparable leks were compared.

^c 95% CI = 95% confidence interval

^d *n* = number of leks in the sample. Here, a lek can have a 0 count in 1 of the 2 years and still be considered.

^eOne lek was located just south of the NW region in Clearwater County.

^fNo data were collected in 2020 due to the Governor's Stay at Home Order during the COVID-19 pandemic.

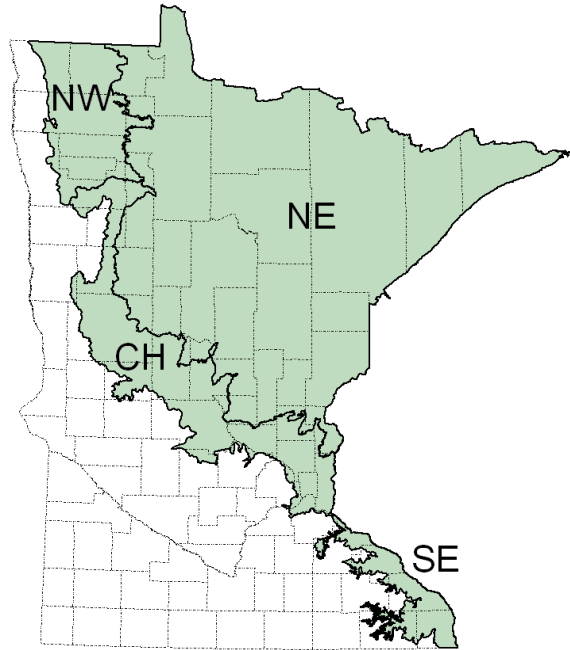


Figure 1. Survey regions for ruffed grouse in Minnesota. Northwest (NW), Northeast (NE), Central Hardwoods (CH), and Southeast (SE) survey regions are depicted relative to county boundaries (dashed lines) and influenced by the Ecological Classification System.

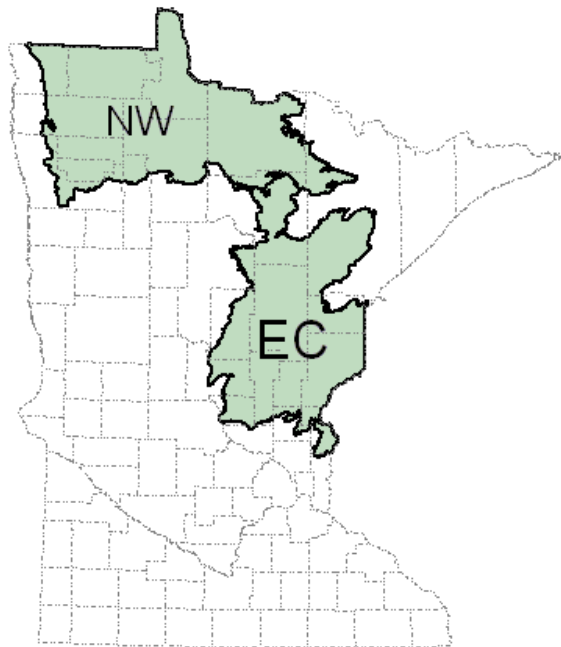


Figure 2. Survey regions for sharp-tailed grouse in Minnesota. Northwest (NW) and East Central (EC) survey regions are depicted relative to county boundaries (dashed lines) and influenced by Ecological Classification System Subsection boundaries.

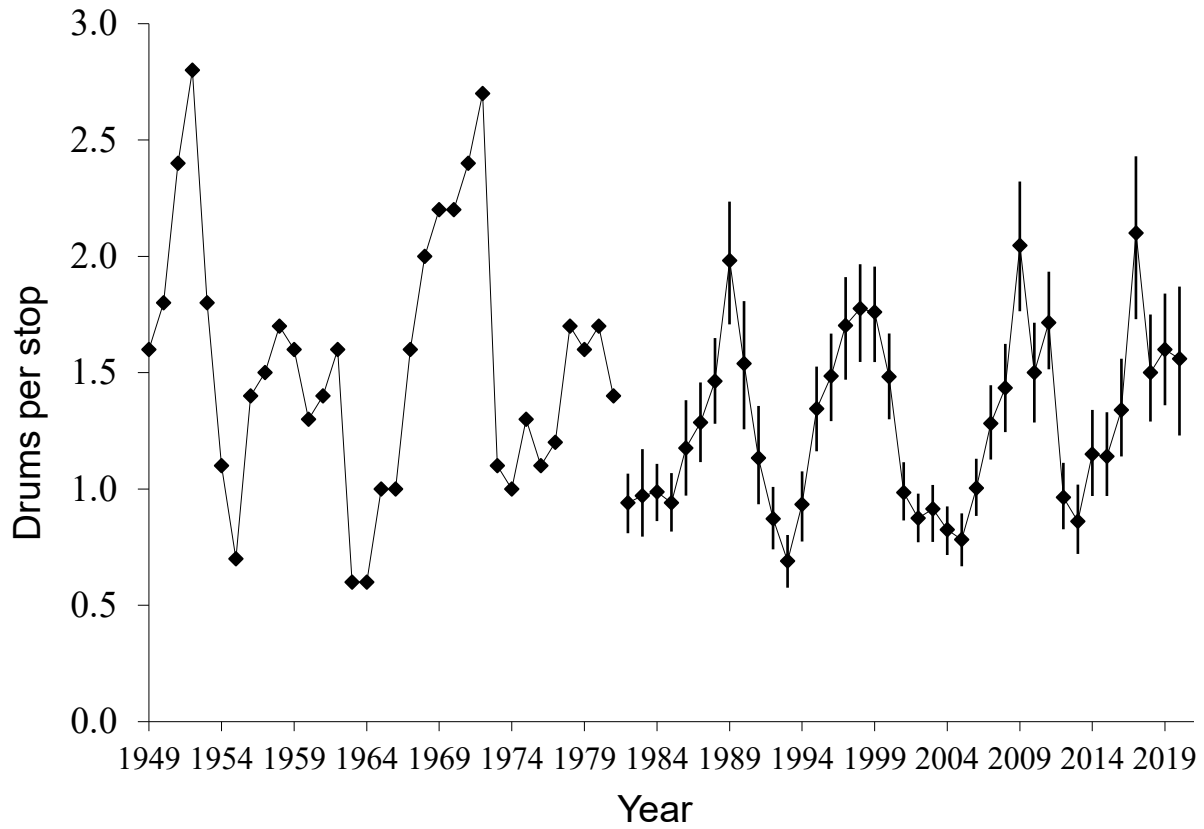
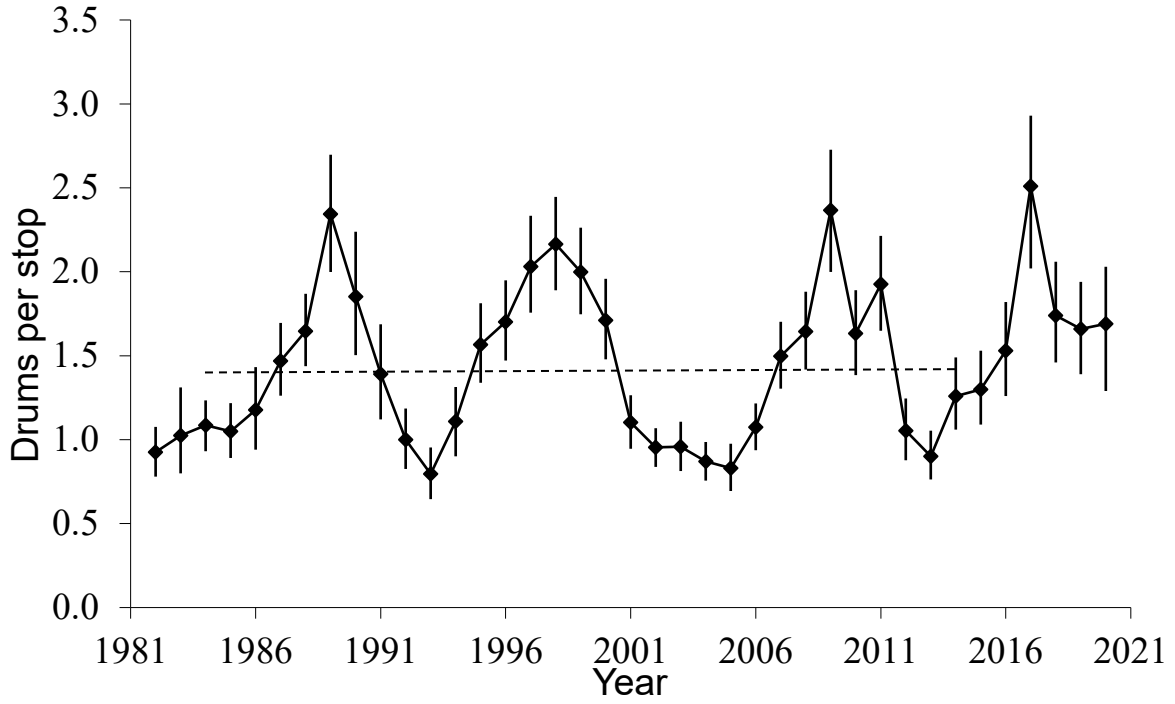
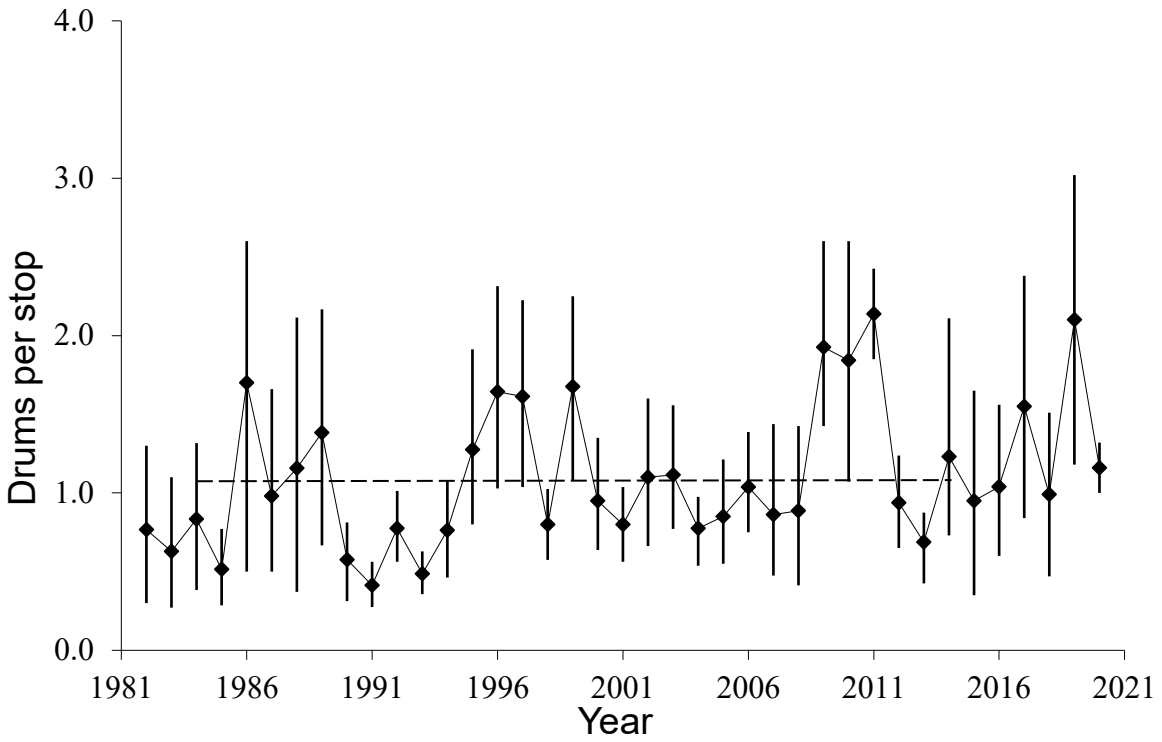


Figure 3. Statewide **ruffed grouse** population index values in Minnesota. Bootstrap (95%) confidence intervals (CI) are provided after 1981, but different analytical methods were used prior to this and thus CI are not available for earlier years. The difference between 1981 and 1982 is biological and not an artifact of the change in analysis methods.



a.



b.

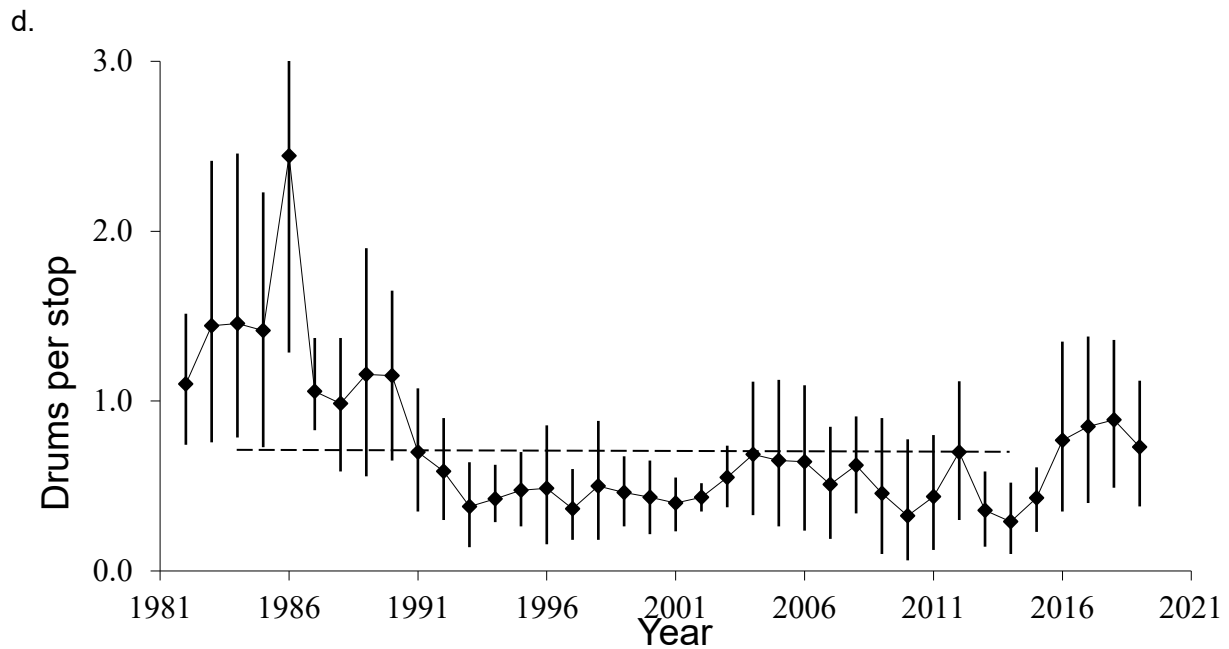
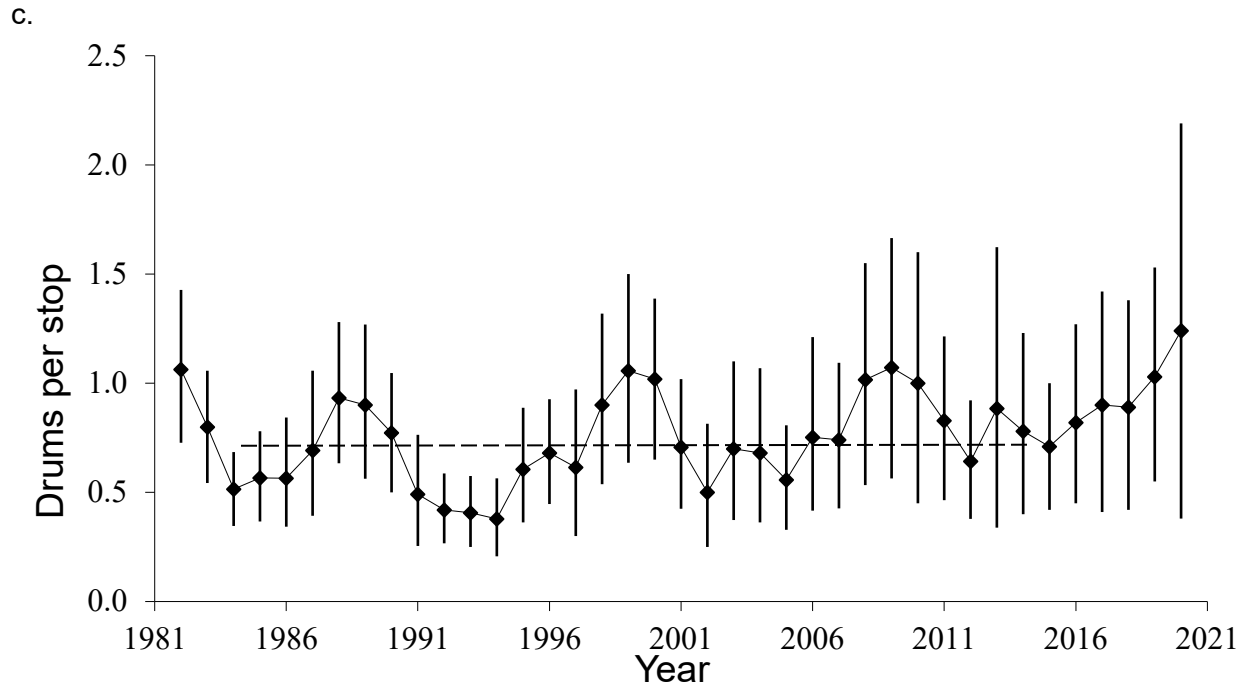


Figure 4a,b,c,d. **Ruffed grouse** population index values in the **Northeast** (a), **Northwest** (b), **Central Hardwoods** (c), and **Southeast** (d) survey regions of Minnesota. The mean for 1984-2014 is indicated by the dashed line. Bootstrap (95%) confidence intervals are provided for each mean. In the bottom panel, the CI for 1986 extends beyond area depicted in the figure. Data were not collected during the survey window in the Southeast during the COVID-19 pandemic in 2020, so the last point is from 2019.

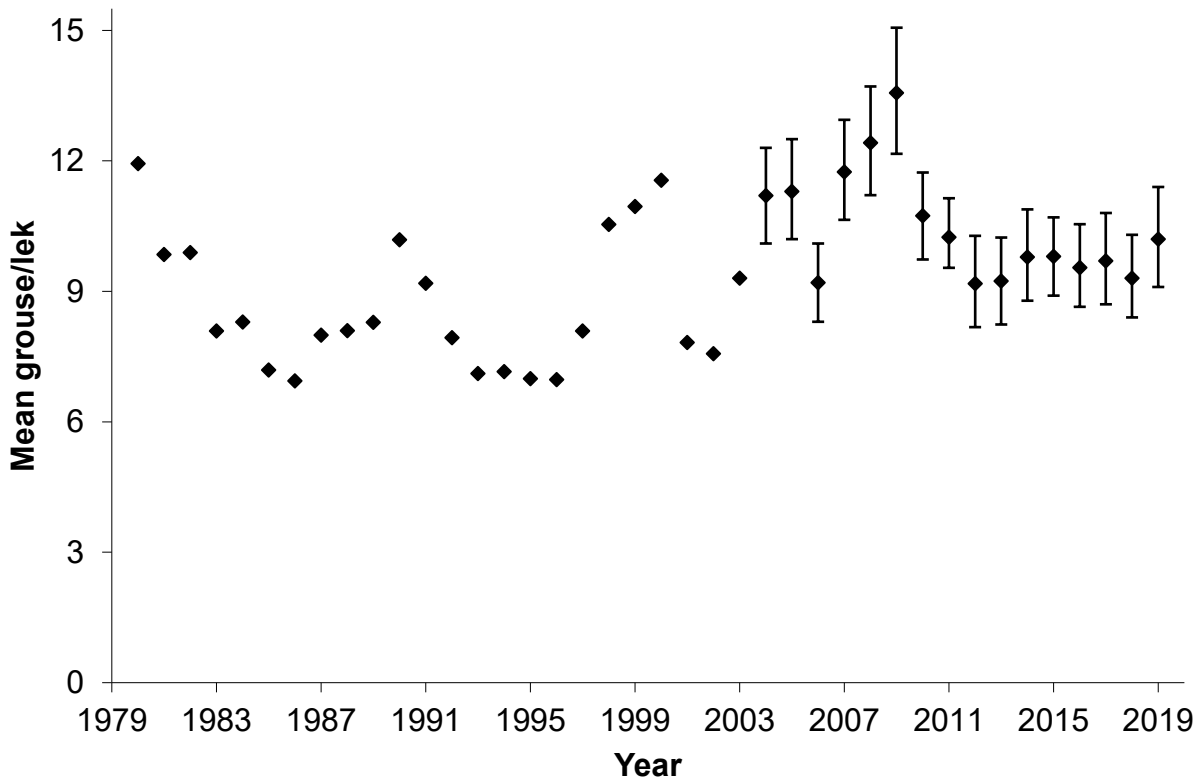


Figure 5. **Sharp-tailed grouse** counted in spring lek surveys statewide in Minnesota during 1980–2019. Bootstrap (95%) confidence intervals are provided for recent years. Annual means are not connected by lines because the same leks were not surveyed every year. No data were collected in 2020 due to the Governor’s Stay at Home Order during the COVID-19 pandemic, so data are presented through 2019.

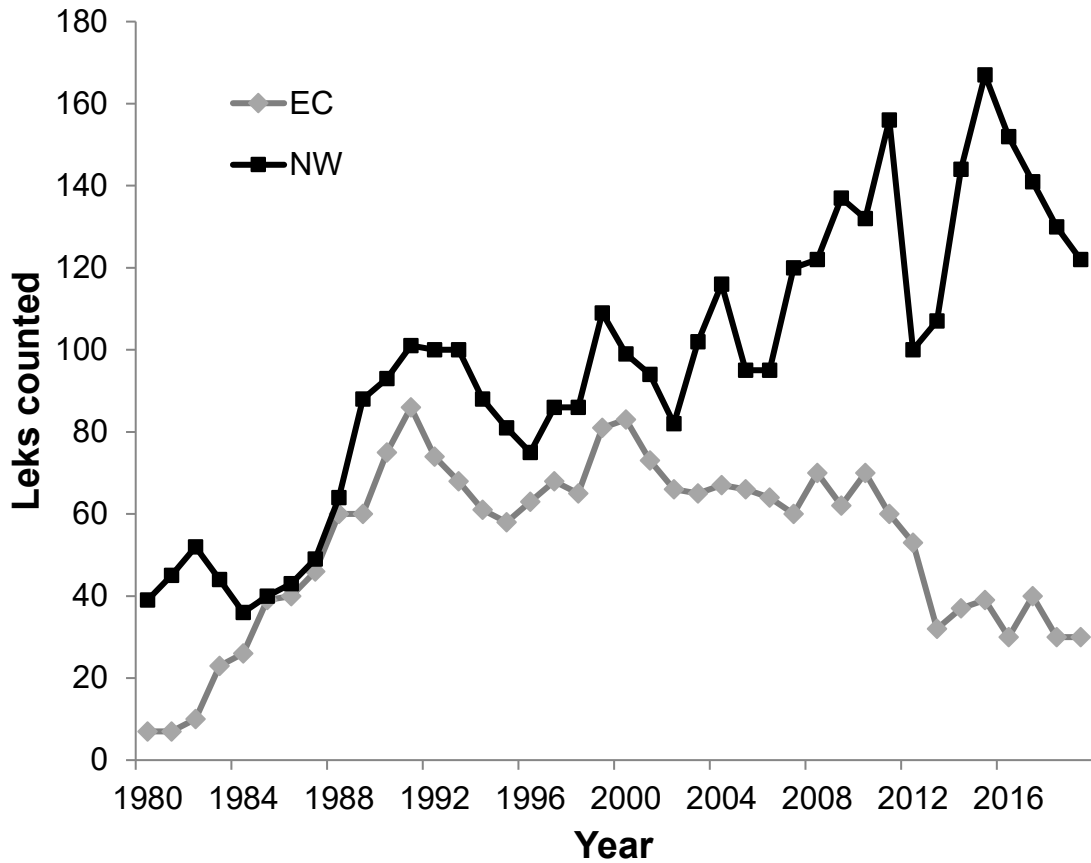


Figure 6. The number of **sharp-tailed grouse** leks with 2 or more birds counted in spring lek surveys in the Northwest (NW) and East Central (EC) survey regions of Minnesota during 1980-2019. Survey data were not collected in 2020 due to the Governor’s Stay at Home Order during the COVID-19 pandemic, so data are presented through 2019.



2020 MINNESOTA PRAIRIE-CHICKEN POPULATION SURVEY

Charlotte Roy, Forest Wildlife Populations and Research Group

SUMMARY OF FINDINGS

Greater prairie-chickens (*Tympanuchus cupido pinnatus*) were surveyed in 13 of 17 survey blocks during the spring of 2020. Observers located 39 booming grounds and counted 452 males and birds of unknown sex in the survey blocks, which is fewer than last year when all 17 blocks were surveyed. Including areas outside the survey blocks, observers located 98 booming grounds, 920 male prairie-chickens, and 84 birds of unknown sex throughout the prairie-chicken range. Estimated densities of 0.07 (0.04–0.10) booming grounds/km² and 11.6 (9.3–13.9) males/booming ground within the survey blocks were similar to densities during recent years and during the 10 years preceding modern hunting seasons (i.e., 1993–2002). However, in 2020, 4 of 7 peripheral survey blocks were not surveyed due to restrictions on nonessential fieldwork during the COVID-19 pandemic, and counts in these peripheral blocks are typically lower, which may have caused estimates of prairie-chicken densities this year to be biased high. Both population indices began to decline in 2008, with greater declines in booming grounds/km² and more stable estimates of males/booming ground.

INTRODUCTION

Historically, greater prairie-chicken (*Tympanuchus cupido pinnatus*) range in Minnesota was restricted to the southeastern portion of the state. However, dramatic changes in their range occurred in the 19th century as settlers expanded and modified the landscape with farming and forest removal, providing abundant food sources and access to new areas. As grass was lost from the landscape, prairie-chicken populations began to decline, their range contracted, and hunting seasons closed after 1942. In an attempt to bolster populations and expand prairie-chicken range, the Minnesota Department of Natural Resources (MNDNR) conducted a series of translocations in the Upper Minnesota River Valley during 1998–2006. Today, the beach ridges of glacial Lake Agassiz hold most of Minnesota's prairie-chickens, but their populations do extend southward (Figure 1). Hunting was re-opened using a limited-entry season in 2003, and <100 prairie-chickens are now harvested annually.

With the opening of the new hunting season, the DNR had a greater interest in the monitoring of prairie-chicken populations, which the Minnesota Prairie-Chicken Society (MPCS) had been coordinating since 1974. The DNR, in collaboration with MPCS members, began coordinating prairie-chicken surveys and adopted a standardized survey design in 2004. These surveys are conducted at small open areas called leks, or booming grounds, where male prairie-chickens display for females in the spring and make a low-frequency booming vocalization that can be heard for miles.

Prairie-chickens continue to be surveyed to monitor changes in population densities over time. However, density estimates can be costly and difficult to obtain, so instead we count individuals and make the assumption that changes in density are the primary source of variation in counts

among years. If true, counts should provide a reasonable index to long-term trends in prairie-chicken populations. However, counts are also influenced by weather, habitat conditions, observer ability, and bird behavior among other factors, which make it difficult to make inferences over short periods of time (e.g., a few annual surveys) or from small changes in index values. Nevertheless, over long time periods and when changes in index values are large, inferences from prairie-chicken surveys are more likely to be valid.

METHODS

Cooperating biologists and volunteers surveyed booming grounds in 13 of 17 designated survey blocks in western Minnesota (Figure 2) during April and May. Each survey block was nonrandomly selected so that surveys would be conducted in areas where habitat was expected to be good (i.e., grassland was relatively abundant) and leks were known to occur. Each observer attempted to find and survey each booming ground repeatedly in his/her assigned block, which comprised 4 sections of the Public Land Survey (approximately 4,144 ha). Observers obtained multiple counts at each booming ground in the morning because male attendance at leks varies throughout the season and throughout the day.

During each survey, observers obtained visual counts of males, females, and birds of unknown sex from a distance with binoculars. Sex was determined through behavior; males display conspicuously, and females do not. If no birds were displaying during the survey period, then sex was recorded as unknown. When a reliable count could not be obtained visually because vegetation or topography prevented it, birds were flushed for counts and sex was recorded as unknown. Most birds for which sex was unknown were likely male because female attendance at leks is sporadic, and they are less conspicuous during lek attendance than displaying males.

In the analysis, I used counts of males and unknowns at each booming ground but not females.

Leks were defined as having ≥ 2 males, so observations of single males were not counted as leks. Data were summarized by hunting permit area and spring survey block. The survey blocks were separated into a core group and a periphery group for analysis. The core group had a threshold density of approximately 1.0 male/km² during 2010, and was located proximally to other such blocks (Figure 2). I compared densities of leks and prairie-chickens to estimated densities from previous years.

I also encouraged observers to submit surveys of booming grounds outside the survey blocks because these observations may provide additional information that is helpful to prairie-chicken management. These data were included in estimates of minimum abundance of prairie-chickens. However, these data were not used in the analysis of lek and prairie-chicken densities because effort and methods may have differed from those used in the survey blocks.

In 2020, the COVID-19 pandemic affected the implementation of the prairie-chicken survey. Four survey blocks in the periphery of the survey region, survey blocks 1 and 2 in Otter Tail County and survey blocks 2 and 3 in Wilkin County, could not be surveyed due to operator restrictions on nonessential field work during the pandemic. Furthermore, the 3 Norman County survey blocks and 1 block in each of Becker and Wilkin Counties were surveyed later than usual because prairie-chicken surveys were not exempted from restrictions on field work for MNDNR Wildlife Staff until May 7. For the analysis, I only used the surveyed blocks to calculate densities for the periphery of the survey region.

RESULTS & DISCUSSION

Observers from MNDNR Section of Wildlife, the U.S. Fish & Wildlife Service, and The Nature Conservancy, as well as many unaffiliated volunteers counted prairie-chickens between 10 March and 15 May 2020. Observers located 98 booming grounds and observed 920 male

prairie-chickens and 84 birds of unknown sex within and outside the survey blocks (Table 1). These counts represent a minimum number of prairie-chickens in Minnesota during 2020, but because survey effort outside of survey blocks is not standardized among years, these counts should not be compared among years or permit areas.

Within the standardized survey blocks, 452 males and birds of unknown sex were counted on 39 booming grounds during 2020 (Table 2). These counts are the lowest since the standardized survey began in 2004 when 1,566 males and 95 booming grounds were counted. This contrasts with the high count of 1,618 males and 114 booming grounds in 2007. Each lek was observed an average of 3.6 times (median = 2), with 28% of booming grounds observed just once. These counts should not be regarded as estimates of abundance because detection probabilities of leks and birds were not estimated. However, if detection probabilities and effort are similar among years in the survey blocks, then population indices based on survey block data can be used to monitor changes in abundance among years.

Densities of prairie-chickens in the 10 core survey blocks were 0.08 (0.05–0.12) booming grounds/km² and 11.4 (9.0–13.9) males/booming ground which is similar to recent years (Table 2, Figure 2). In the 3 of the 7 peripheral survey blocks that were surveyed in 2020, densities were 0.04 (0.02–0.06) booming grounds/km² and 12.6 (5.7–19.5) males/booming ground. These estimates were likely biased high by the lack of surveys in 4 peripheral survey blocks during the COVID-19 pandemic; these blocks usually have low densities. For all survey blocks, the density of 0.07 (0.04–0.10) booming grounds/km² during 2020 was similar to densities during recent years (Table 2, Figure 3) and the average of 0.08 (0.06–0.09) booming grounds/km² during the 10 years preceding recent hunting seasons (i.e., 1993–2002). Similarly, the density of 11.6 (9.3–13.9) males/booming ground in all surveyed blocks during 2020 was comparable to densities during recent years and similar to the average of 11.5 (10.1–12.9) males/booming ground observed during 1993–2002 (Table 2, Figure 3). However, due to a lack of surveys in 4 peripheral survey blocks during the pandemic, estimates for all survey blocks combined are likely biased high this year.

The observed densities are lower than the years preceding 2008 when CRP enrollments in the counties containing the survey blocks were highest. These changes in the population indices coincide with gains and losses in enrollments in the Conservation Reserve Program. More explicit examination of these patterns can be found in the recent publication, *Adkins, K., C. L. Roy, D. E. Anderson, R. Wright. 2019. Landscape-scale Greater Prairie-chicken Habitat Relations and the Conservation Reserve Program. The Journal of Wildlife Management DOI: 10.002/jwmg.21724*

ACKNOWLEDGMENTS

I would like to thank cooperators who conducted and helped coordinate the prairie-chicken survey, with a special thanks to those that stepped up to complete extra surveys this year when many were restricted from fieldwork during the pandemic. Cooperators with The Nature Conservancy included Brian Winter, Travis Issendorf, and volunteers Pat Beauzay, Matt Mecklenburg, Casey Reep, Derek Savage, and Carl Altenbernd; cooperators within MNDNR included Emily Hutchins, Rob Baden, Greg Henderson, Mark Palm, Michael Oehler, and Matt Morin; cooperators with the US Fish and Wildlife Service usually include Shawn Papon, Chad Raitz, Ben Walker, Erin Lentz, Traver Fields, and Stacy Salvevold; and numerous additional volunteers participate, including Dan Svedarsky, Doug Wells, Jon Voz, Ross Hier, Phil Doll, and Doug Hedtke. This survey was funded in part by the Wildlife Restoration (Pittman-Robertson) Program. Lindsey Shartell provided assistance and comments which improved this report.

Table 1. Minimum abundance of prairie-chickens within and outside hunting permit areas in Minnesota during spring 2020. Lek and bird counts are not comparable among permit areas or years.

Permit Area	Area (km ²)	Leks	Males	Unknown ^a
803A	1,411	11	72	0
804A	435	0	0	0
805A	267	12	90	0
806A	747	8	29	45
807A	440	17	166	0
808A	417	21	269	0
809A	744	11	130	0
810A	505	3	36	5
811A	706	1	7	0
812A	914	3	23	0
813A	925	3	44	3
PA subtotal	7,511	90	866	53
Outside PAs ^b	NA ^c	8	54	31
Grand total	NA ^c	98	920	84

^a Unknown = prairie-chickens for which sex was unknown, but which were probably males.

^b Counts done outside permit areas (PA).

^c NA = not applicable because the area outside permit areas was not defined.

Table 2. Prairie-chicken counts within survey blocks in Minnesota during spring 2020, and change in counts compared to 2019.

Range ^b	Survey Block	Area (km ²)	2020		Change from 2019 ^a	
			Booming grounds	Males ^c	Booming grounds	Males ^c
Core	Polk 1	41.2	4	21	-1	-5
	Polk 2	42.0	3	35	0	3
	Norman 1	42.0	2	15	1	12
	Norman 2	42.2	1	17	-2	-3
	Norman 3	41.0	5	46	2	21
	Clay 1	46.0	8	125	1	-1
	Clay 2	41.0	3	39	1	-16
	Clay 3	42.0	6	56	2	-5
	Clay 4	39.0	0	0	-2	-7
	Wilkin 1	40.0	2	35	0	-3
	Core subtotal	415.0	34	389	2	-4
Periphery	Mahnomen	41.7	2	42	0	0
	Becker 1	41.4	2	15	-2	-2
	Becker 2	41.7	1	6	0	0
	Wilkin 2	NA	NA ^d	NA	NA	NA
	Wilkin 3	NA	NA	NA	NA	NA
	Otter Tail 1	NA	NA	NA	NA	NA
	Otter Tail 2	NA	NA	NA	NA	NA
Periphery subtotal	124.8	5	63	-2	-2	
Grand total	539.8	39	452	0	-6	

^a The 2019 count was subtracted from the 2020 count, so positive values indicate increases.

^b Survey blocks were categorized as within the core or periphery of the Minnesota prairie-chicken range based upon bird densities and geographic location.

^c Includes birds recorded as being of unknown sex but excludes lone males.

^d NA = not applicable because 4 survey blocks were not completed in 2020 due to cooperators restrictions on nonessential field work during the COVID-19 pandemic.

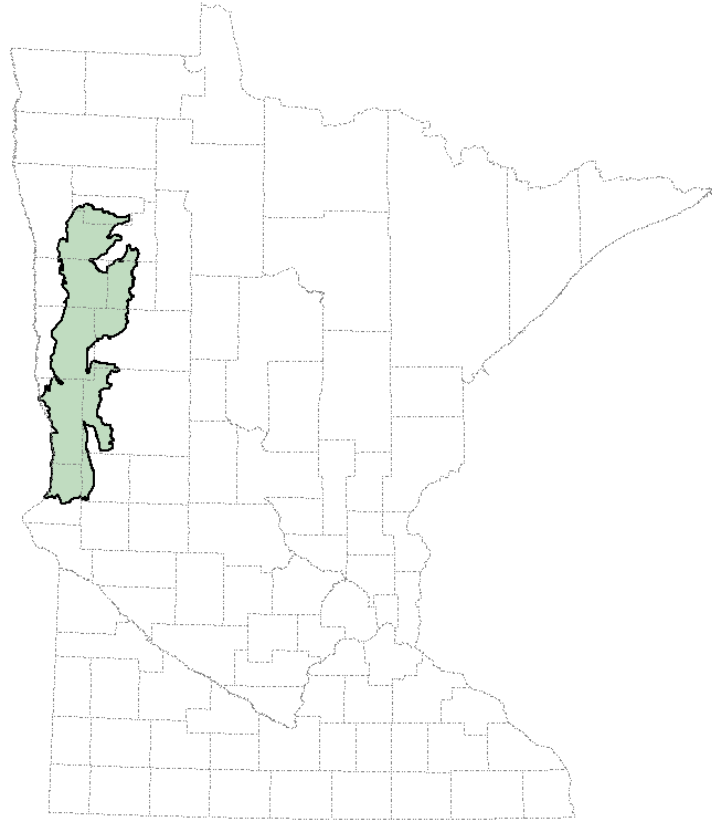


Figure 1. Primary greater prairie-chicken range in Minnesota (shaded area) relative to county boundaries. The range boundary was based on Ecological Classification System Land Type Associations and excludes some areas known to be occupied by prairie-chickens.

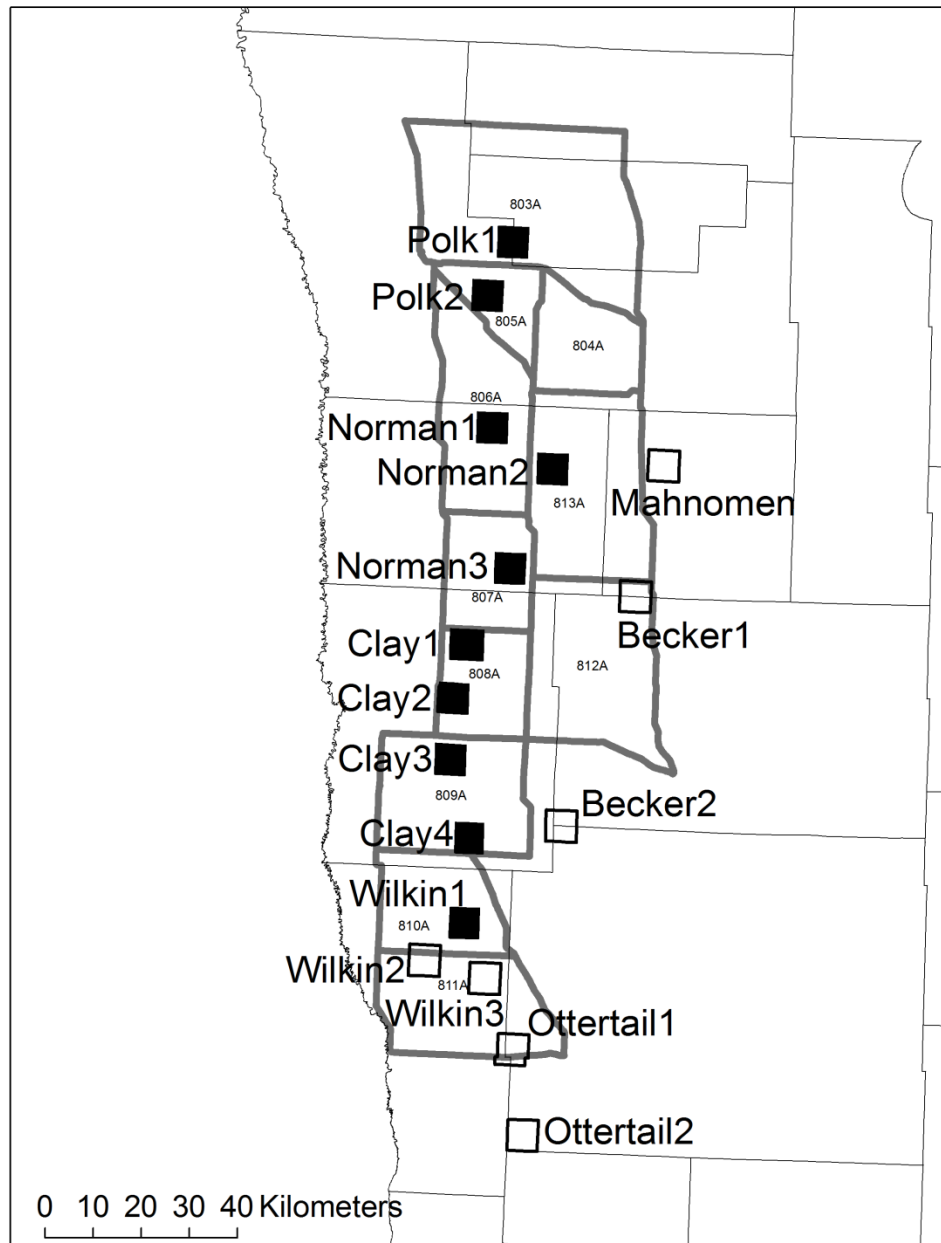


Figure 2. Prairie-chicken lek survey blocks (41 km², labeled squares) and hunting permit areas (thick grey lines) in western Minnesota. Survey blocks were either in the core (black) or periphery (white) of the range with a threshold of 1.0 male/km² in 2010, and were named after their respective counties (thin black lines). Permit areas were revised in 2013 to eliminate 801A and 802A, modify 803A, and add 812A and 813A. See previous reports for former permit area boundaries.

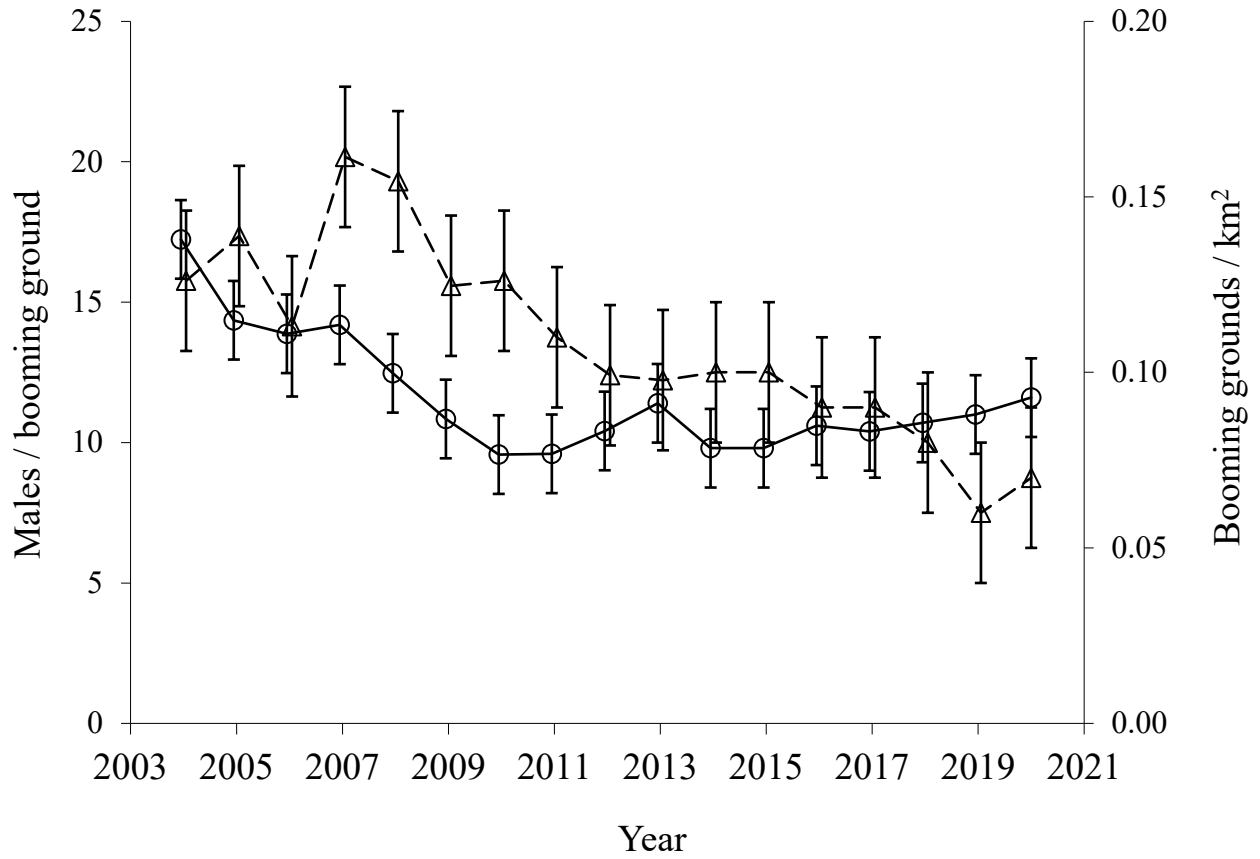


Figure 3. Mean prairie-chicken males/booming ground (circles connected by solid line) and booming grounds/km² (triangles connected by dashed line) in survey blocks in Minnesota with 95% confidence intervals.

2020 NW MN ELK SURVEYS

Doug Franke, Area Wildlife Manager, Thief River Falls

INTRODUCTION

Minnesota DNR Fish and Wildlife and Enforcement staff used a single fixed-wing aircraft (Cessna 185 Skywagon) to conduct aerial elk surveys for the Grygla and Lancaster elk herds between February 11th and February 19th, 2020. As in the past, survey transects were spaced 1/5 mile apart and flown at an altitude of 300 to 400 feet and speeds of 80-90 mph. A pilot and two observers recorded elk locations and documented antlerless and antlered elk. Cow and calf elk were combined and recorded as antlerless since differentiating the two is difficult due to moving animals and the altitude and speed of the fixed-wing aircraft. Antlered elk were recorded as either branch antlered or spike bulls.

The Thief Lake Area Wildlife Manager requested to exclude the lower 22 survey plots of the Grygla survey block this year and in future years due to the large number of cattle farms and lack of elk observations in that area for the past decade. The same predetermined transects used in 2019 were flown for the Lancaster survey block. The Caribou-Vita elk survey block has not been completed at the time of this report pending improved snow conditions in Manitoba.

Observability conditions were considered very good this year. Snow depths and snow condition were fairly consistent and considered very good for both elk survey blocks. Snow depths ranged from 12 inches in the Grygla area and 16 to 20 in the Lancaster area. Weather conditions were average for this time of the year with temperatures ranging from a low of -20°F to a high of 28°F with sunny to cloudy skies. There was a one-day weather delay between the first and second days of the Grygla survey; and there was a two-day weather delay between the first and second days of the Lancaster survey and due to snow, low ceilings, and high winds.

Grygla Survey Block

This survey started on February 11th and after a one-day weather delay was completed on February 13, 2020. The area surveyed was 122 mi² (22 square miles were not flown this year as previously mentioned). Total aircraft engine time to complete this survey (takeoff to landing) was 9.8 hours. The fixed-wing crew recorded elk at 5 separate locations within the survey boundary--all elk were observed on the second survey day. Total elk observed was 24 and included: 14 antlerless and 10 bulls (9 branch antlered and 1 spike). Of special note is that all but one of the elk were located on State Wildlife Management Area land at the time of the survey.

Lancaster Survey Block—Water Tower and Percy WMA herds

This survey started on February 16th and was completed on February 19, 2020. The area surveyed was the same 167 mi² area that has been flown the past several years. Total aircraft time to complete the survey was 14.7 hours (takeoff to landing). The fixed-wing crew recorded elk at 10 separate locations within the survey boundary. Total elk recorded within the Lancaster block was 102 and included: 69 antlerless and 33 bulls (27 branch antlered and 6 spikes).

The Water Tower herd had 41 antlerless elk and was again located in the same exact woodlot that the antlerless group was recorded in during the 2019 and 2018 surveys. This group was located one mile east of this woodlot in 2017. In addition, there were 12 branch antlered and 3 spike bulls located within 1 to 7 miles of the antlerless group.

The Percy WMA herd had 29 antlerless that were located approximately 0.5 miles east of the Percy WMA (about 5 miles southeast of where they had spent the winters of 2018 and 2019). For the first time since I have been surveying these elk (2013), the antlerless herd was actually split into two groups of 19 and 9 animals separated by 1.25 miles. The antlerless groups for all the elk herds in the recent past have always been observed together during winter months. There were 13 branch antlered and 3 spike bulls observed 3.5 miles north of the antlerless group(s). Two branch antlered bulls were located 3.5 miles west of Lancaster.

Caribou-Vita Survey Block (a.k.a. border herd)

This survey block was not completed this year. Poor snow conditions in Manitoba and lack of funding precluded the Manitoba Wildlife staff from initiating the survey. Table 2 was included again this year as a reference—it details the age/sex breakdown for these two populations in Canada for 2017 and 2018.

Table 1 on page three summarizes MN DNR elk observations during the past five years of NW MN aerial elk surveys. The maps on pages 4 and 5 show the 2020 locations of elk within each survey block. The map on page 6 shows elk observations during surveys conducted from 2004 to 2018.

ACKNOWLEDGMENTS

I would like again to thank all those that helped with the survey this year, especially the fixed-wing pilot Bob Geving who provided safe flying and A+++ landings for all of us (we won't hold the flat tire on the last landing against you!). Observers this year included: Kyle Arola (Thief Lake Area Wildlife Manager), Jason Wollin (Karlstad Assistant Area Wildlife Manager), and myself. Special thanks again to Brian Haroldson who put together all of the survey materials and computer used during the survey—much appreciated!

Table 1. Comparison of aerial survey elk observations between 2016 and 2020 for the Lancaster, Caribou-Vita, and Grygla herds.

	Lancaster					Caribou-Vita (US side of border)					Grygla				
	2016	2017	2018	2019	2020	2016	2017	2018	2019*	2020*	2016	2017	2018	2019	2020
Spike bull	6	2	5	11	6	0	0	1	-	-	2	4	2	1	1
Branch antlered bull	12	14	13	22	27	6	1	6	-	-	9	6	6	10	9
Total bulls	18	16	18	33	33	6	1	7	-	-	11	10	8	11	10
Antlerless	34	45	57	61	69	4	0	0	-	-	10	7	7	8	14
Total elk	52	61	75	94	102	10	1	7	-	-	21	17	15	19	24

* Survey was not completed in 2019 or 2020

Table 2. Aerial survey elk observations recorded by Manitoba Wildlife—2017 and 2018

	Border (Caribou)		Vita		Combined Total	
	2017	2018	2017	2018	2017	2018
Spike bull	2	3	4	2	6	5
Branch antlered bull	17	12	7	5	24	17
Total bulls	19	15	11	7	30	22
Cow	68	*	32	*	100	*
Calf	21	*	12	*	33	*
Total antlerless	89	65	44	39	133	104
Total elk	108	80	55	46	163	126

* Manitoba Wildlife did not differentiate antlerless elk between cows and calves in 2018

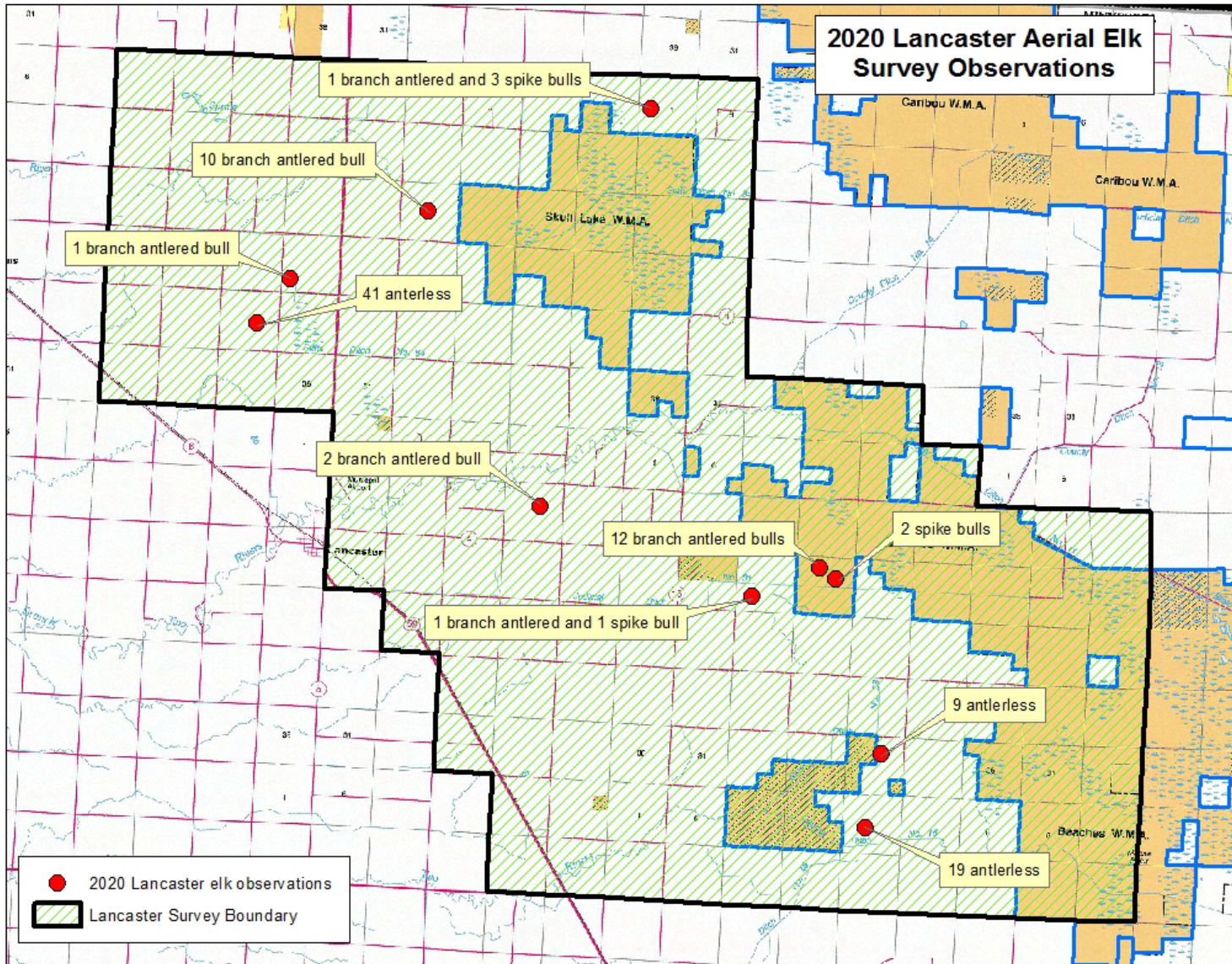


Figure 1. Locations of elk observed within the Lancaster area survey blocks, 2020

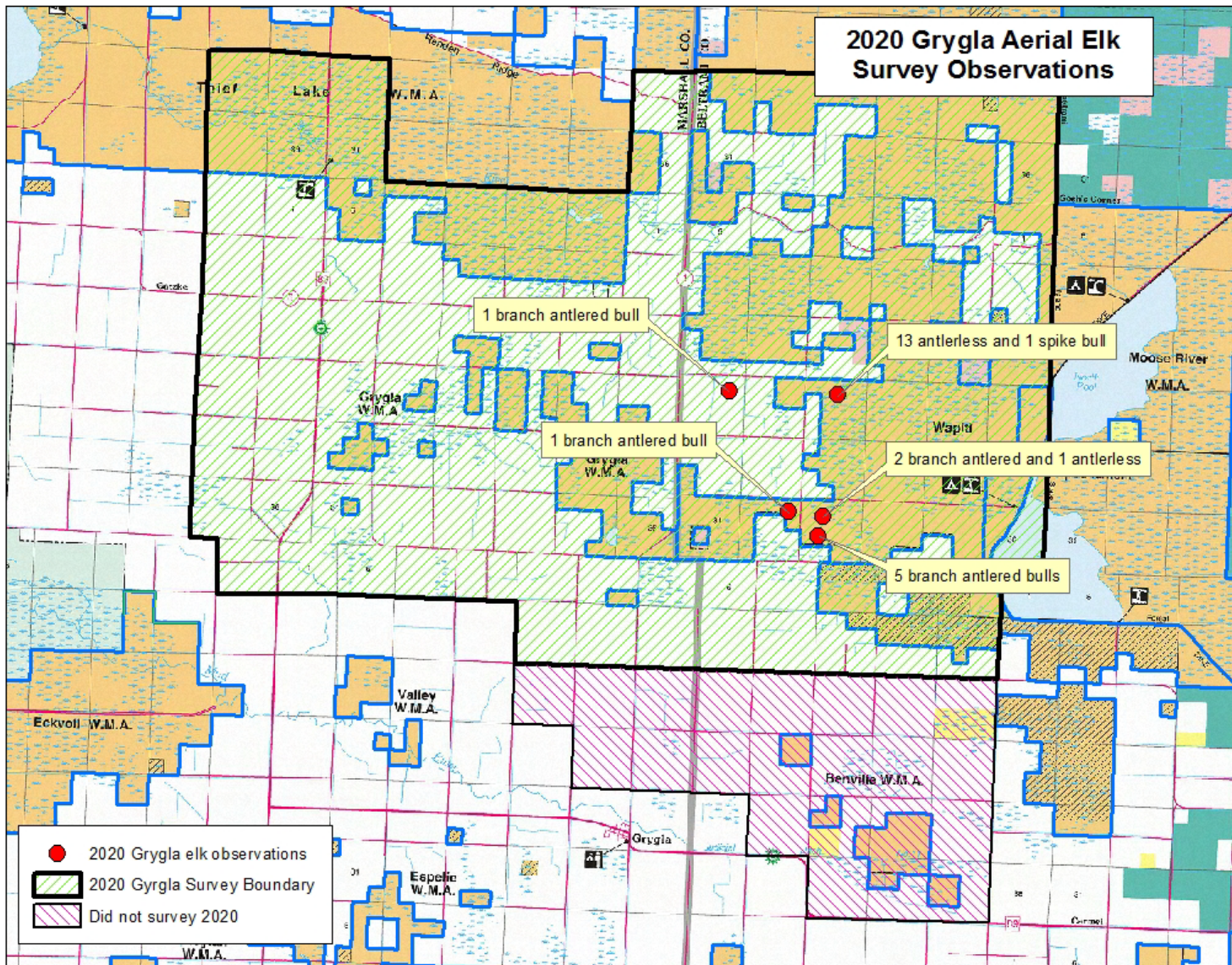


Figure 2. Locations of elk observed within the Grygla area survey blocks, 2020

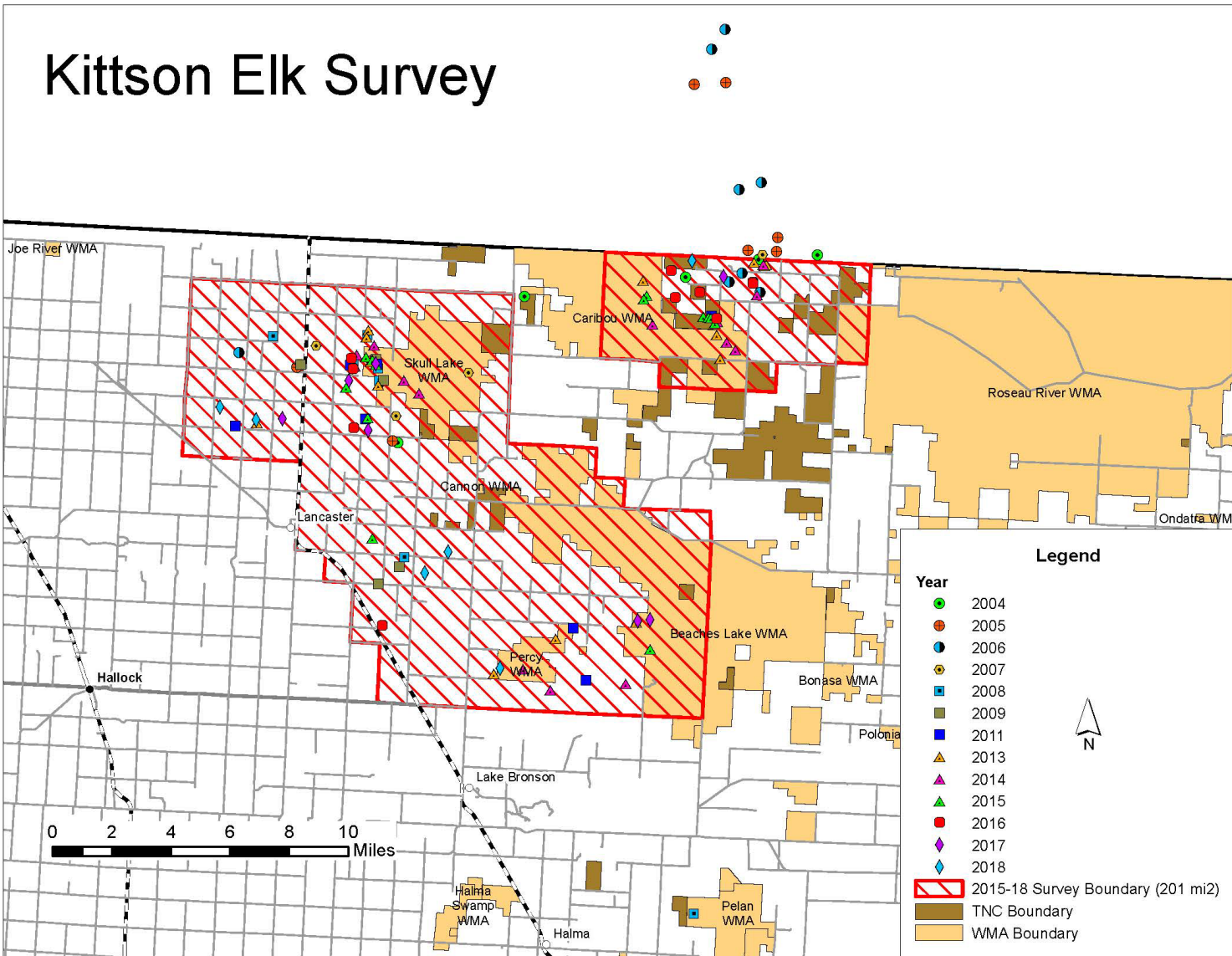


Figure 3. Locations of elk observed within the Kittson area survey blocks, 2020



2020 AERIAL MOOSE SURVEY

Glenn D. DelGiudice, Forest Wildlife Populations and Research Group

INTRODUCTION

Each year we conduct an aerial survey in northeastern Minnesota to estimate the moose (*Alces alces*) population and to monitor and assess changes in the overall status of the state's largest deer species. Specifically, the primary objectives of this annual survey are to estimate moose abundance, percent calves, and calf:cow and bull:cow ratios. These demographic data help us to 1) best determine and understand the population's long-term trend (decreasing, stable, or increasing), composition, and spatial distribution; 2) set the harvest quota for the subsequent State hunting season (when applicable); 3) with research findings, improve our understanding of moose ecology; and 4) otherwise contribute to sound future management strategies.

METHODS

The survey area is approximately 5,985 mi² (almost 4 million acres, Lenarz 1998, Giudice et al. 2012). We estimate moose numbers and age and sex ratios by flying transects within a stratified random sample of 436 total survey plots that cover the full extent of moose range in northeastern Minnesota (Figure 1). To keep the stratification current, all survey plots are reviewed and re-stratified as low, medium, or high moose density about every 5 years, based on past survey observations of moose, locations of recently harvested moose, and extensive field experience of moose managers and researchers. Low, medium, and high density classes are based on whether up to 2, 3–7, or 8 or more moose, respectively, would be expected to be observed in a specific plot. The most recent re-stratification review was conducted in October 2018. Additionally, individual plots may be re-stratified after each annual survey as warranted by aerial observations. Stratification is most important to optimizing precision of our survey estimates. In 2012, we added a 4th stratum to the survey approach, represented by a series of 9 plots (referred to as “habitat plots”) which have already undergone, or will undergo significant disturbance by wildfire, prescribed burning, or timber harvest. These same 9 plots are surveyed each year in an effort to better understand moose use of disturbed areas and evaluate the effect of forest disturbance on moose density over time. In total, we surveyed 52 (43 randomly sampled and the 9 habitat plots) of the 436 plots this year.

All 436 survey plots in the grid (designed in 2005) are 13.9-mi² rectangles (5 x 2.77 mi), oriented east to west, with 8 flight-transects similarly oriented and evenly spaced 0.3 mi apart. Minnesota Department of Natural Resources (MNDNR) Enforcement pilots flew the 2 helicopters used to conduct the survey—1 Bell Jet Ranger (OH-58) and 1 MD500E. We determined the sex of moose using the presence of antlers or the presence of a vulva patch (Mitchell 1970), nose coloration, and bell size and shape. We identified calves by size and behavior. We used the program DNRSurvey on tablet-style computers (Toughbook®) to record survey data (Wright et al. 2015). DNRSurvey allowed us to display transect lines superimposed on aerial photography, topographical maps, or other optional backgrounds to observe each aircraft's flight path over the selected background in *real time*, and to efficiently record data using a tablet pen with a menu-driven data-entry form. Two primary strengths of this aerial

moose survey are the consistency and standardization of the methods since 2005 and the long-term consistency of the survey team’s personnel, survey biometrician, and geographic information system (GIS) specialists.

We accounted for visibility bias using a sightability model (Giudice et al. 2012). This model was developed between 2004 and 2007 using adult moose that were radiocollared as part of a study of survival and its impact on dynamics of the population (Lenarz et al. 2009, 2010). Logistic regression indicated that “visual obstruction” (VO) was the most important covariate in determining whether radiocollared moose were observed. We estimated VO within a 30-ft radius (roughly 4 moose lengths) of the observed moose. Estimated VO was the proportion of a circle where vegetation would prevent you from seeing a moose from an oblique angle when circling that spot in a helicopter. If we observed more than 1 moose (a group) at a location, VO was based on the first moose sighted. We used uncorrected estimates (no sightability correction) of bulls, cows, and calves, adjusted for sampling, to calculate the bull:cow and calf:cow ratios at the population level (i.e., using the combined ratio estimator; Cochran 1977:165).

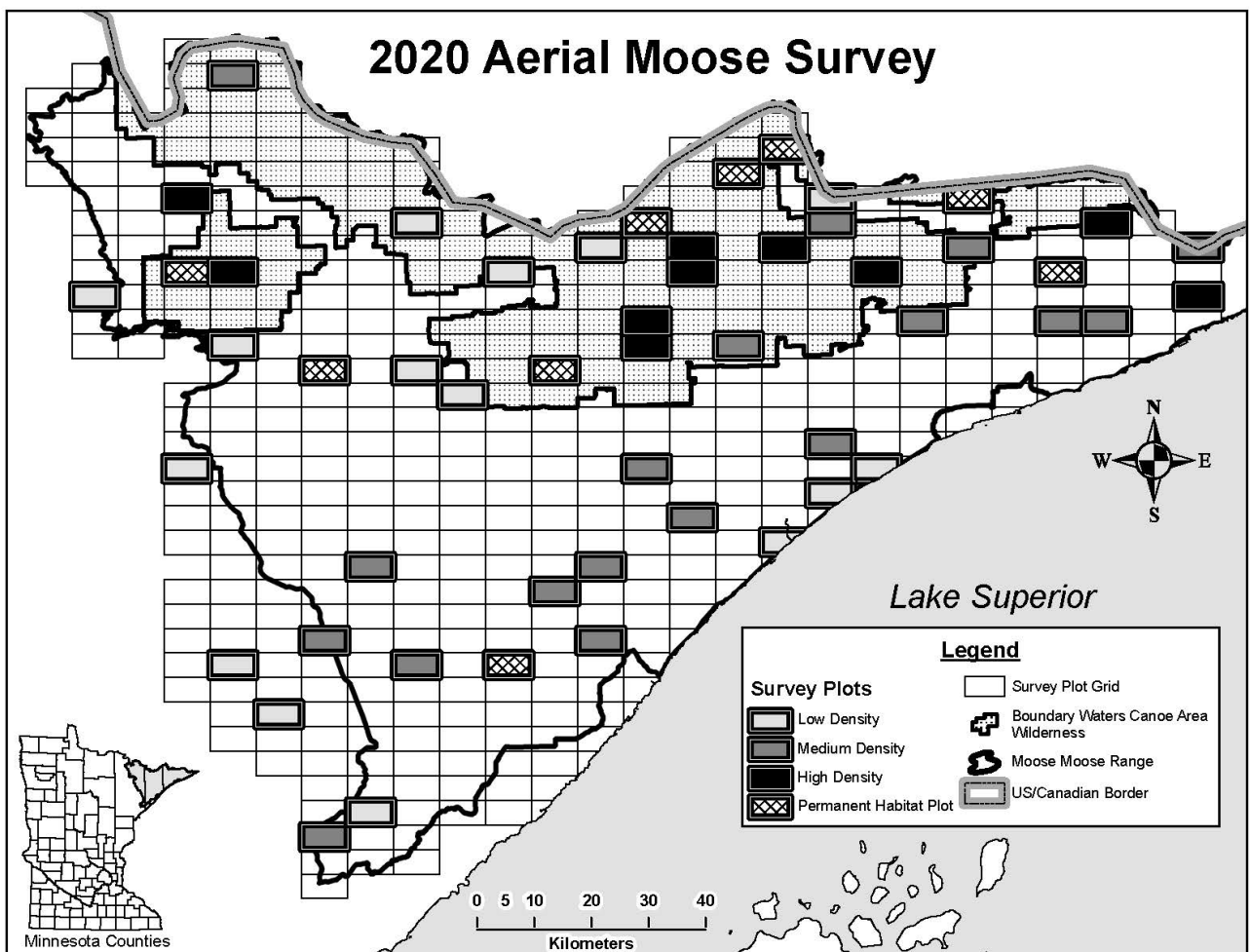


Figure 1. Moose survey area and 52 sample plots flown in the 2020 aerial moose survey.

RESULTS AND DISCUSSION

The survey was conducted from 6 to 17 January 2020. It consisted of 9 actual survey days, and as from 2014 to 2019, it included a sample of 52 survey plots. This year, based on optimal allocation analyses, we surveyed 15 low-, 18 medium-, and 10 high-density plots, and the 9 permanent or habitat plots (Giudice 2020). Generally, 8" of snow cover is our minimum threshold depth for conducting the survey. Snow depths were greater than 16" on 100% of the sample plots. Overall, survey conditions were rated as good for 88%, fair for 12%, and poor for 0% of the plots when surveyed. Average survey intensity was 48 minutes/plot (13.9 mi²) and ranged from 25 to 60 minutes/plot (Giudice 2020).

This year 308 moose were observed on 39 (75%) of the 52 plots surveyed (a total 723 mi²), less than the 429 moose observed on 43 of 52 plots during the 2019 survey. An average of 7.9 moose (range = 1–28) were observed per "occupied" plot. Plot occupancy during the past 16 years averaged 81% (range = 65–95%) with a mean 11.6 moose observed per occupied plot. The average group size was 2.1 moose, similar to the previous 16 years (2 moose) and ranged from 1 to 8 moose per group. This year's 308 observed moose included 131 bulls, 138 cows, 37 calves, and 2 unclassified adults. Overall, estimated VO averaged 44% (range = 0–85%) and average estimated detection probability was 0.55 (range = 0.23–0.85). Both VO and detection probability have remained relatively constant since 2005.

After adjusting for sampling and sightability, we estimated the population in northeastern Minnesota at 3,150 (2,400–4,320, 90% confidence interval [CI]) moose (Table 1, Figure 2). As can be noted from the 90% confidence intervals associated with the population point estimates, statistical uncertainty inherent in aerial wildlife surveys can be quite large, even when surveying large, dark, relatively conspicuous animals such as moose against a white background during winter. This is attributable to the varied (1) occurrence of dense vegetation, (2) habitat use by moose, (3) behavioral responses to aircraft, (4) effects of annual environmental conditions (e.g., snow depth, ambient temperature) on their movements, and (5) interaction of these and other factors. Consequently, year-to-year statistical comparisons of population estimates are *not* supported by these surveys. These data are best suited to establishing long-term trends; even short-term trends must be viewed cautiously.

Past aerial survey and research results have indicated that the long-term trend of the population in northeastern Minnesota has been declining since 2006 (Lenarz et al. 2010, DelGiudice 2019). The current population estimate is 64% less than the estimate in 2006 and the declining linear trend during the past decade remains statistically significant ($r^2 = 0.762$, $P < 0.001$, Figure 2). However, the leveling since 2012 persists, and a piecewise polynomial curve indicates that the trend from 2012 to 2020 is not declining (Figure 3). While this recent short-term trend (9 years) is noteworthy, it applies only to the existing survey estimates, and does not forecast the future trajectory of the population (Giudice 2020).

The January 2020 calf:cow ratio of 0.36 is similar to the 15-year average since 2005 (0.35, Table 1, Figure 4). Calves were 12% of the total 308 moose actually observed and represented 18% of the estimated population (Table 1, Figure 4). Twin calves were observed with 3 of the 138 (2%) cow moose (Table 1). Although we know from recent field studies that fertility (pregnancy rates) of the population's adult females has been robust, overall, survey results indicate calf survival to January 2020 remains low, typical compared to most years since the population decline began following the 2006 survey (Table 1). Calf survival during the January–April interval can decline markedly (Schrage et al., unpublished data), and annual spring recruitment of calves (survival to 1 year old) can have a significant influence on the population's performance and dynamics. Findings of a recent field study documented similar low calf survival (0.442–0.485) to early winter in 2015–16 and 2016–17 (Obermoller 2017,

Severud 2017). Calf survival by spring 2017 (recruitment) had declined to just 0.33. But it is also important to note that adult moose survival has the greatest long-term impact on annual changes in the moose population (Lenarz et al. 2010). Consistent with the recent relative stability of the population trend, the annual survival rate of adult GPS-collared moose has changed little (85–88%) during 2014–2017 (Carstensen et al. 2017, unpublished data), but is slightly higher than the previous long-term (2002–2008) average of 81% (Lenarz et al. 2009).

The January 2020 estimated bull:cow ratio (0.90, Table 1; Figure 5) is similar to the long-term average of 1.00 during 2005–2019 and compared to the mean ratio of 2009–2012 (0.87), when the population decline was steepest. However, this ratio has been as low as 0.64 (2011) during the steep decline. During the recent 9-year trend of stability, the average bull:cow ratio has been 1.12. However, due to the notable annual variability associated with the bull:cow ratios, the apparent upward trend line is not statistically meaningful (Figure 5).

Table 1. Estimated moose abundance, 90% confidence intervals, calf:cow ratios, percent calves in the population, percent cows with twins, and bull:cow ratios estimated from aerial surveys in northeastern Minnesota, 2005–2020.

Survey	Estimate	90% Confidence Interval	Calf: Cow	% Calves	% Cows w/ twins	Bull: Cow
2005	8,160	6,090 – 11,410	0.52	19	9	1.04
2006	8,840	6,790 – 11,910	0.34	13	5	1.09
2007	6,860	5,320 – 9,150	0.29	13	3	0.89
2008	7,890	6,080 – 10,600	0.36	16	2	0.77
2009	7,840	6,270 – 10,040	0.32	14	2	0.94
2010	5,700	4,540 – 7,350	0.28	13	3	0.83
2011	4,900	3,870 – 6,380	0.24	13	1	0.64
2012	4,230	3,250 – 5,710	0.36	15	6	1.08
2013	2,760	2,160 – 3,650	0.33	12	3	1.23
2014	4,350	3,220 – 6,210	0.44	17	3	1.24
2015	3,450	2,610 – 4,770	0.29	13	3	0.99
2016	4,020	3,230 – 5,180	0.42	17	5	1.03
2017	3,710	3,010 – 4,710	0.36	15	4	0.91
2018	3,030	2,320 – 4,140	0.37	15	4	1.25
2019	4,180	3,250 – 5,580	0.32	13	3	1.24
2020	3,150	2,400 – 4,320	0.36	18	2	0.90

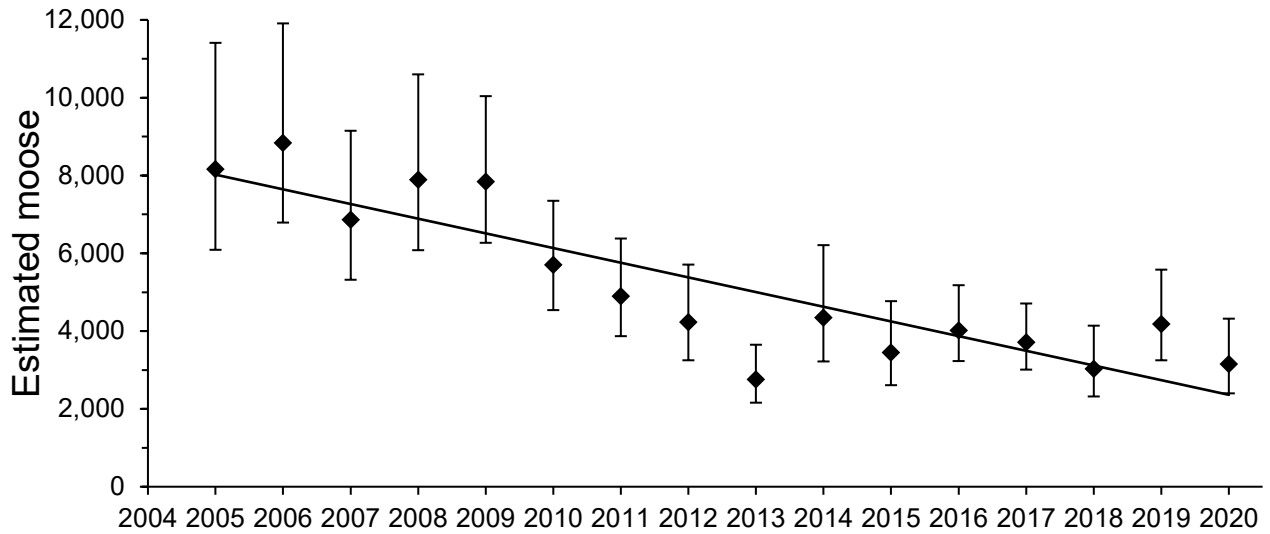


Figure 2. Point estimates, 90% confidence intervals, and a linear trend line of estimated moose abundance in northeastern Minnesota, 2005–2020 ($y = -377x + 764585$, $r^2 = 0.762$, $P < 0.001$). Note: The 2005 survey was the first to be flown with helicopters, and to include a sightability model and a uniform grid of east-west oriented, rectangular 13.9-mi² plots.

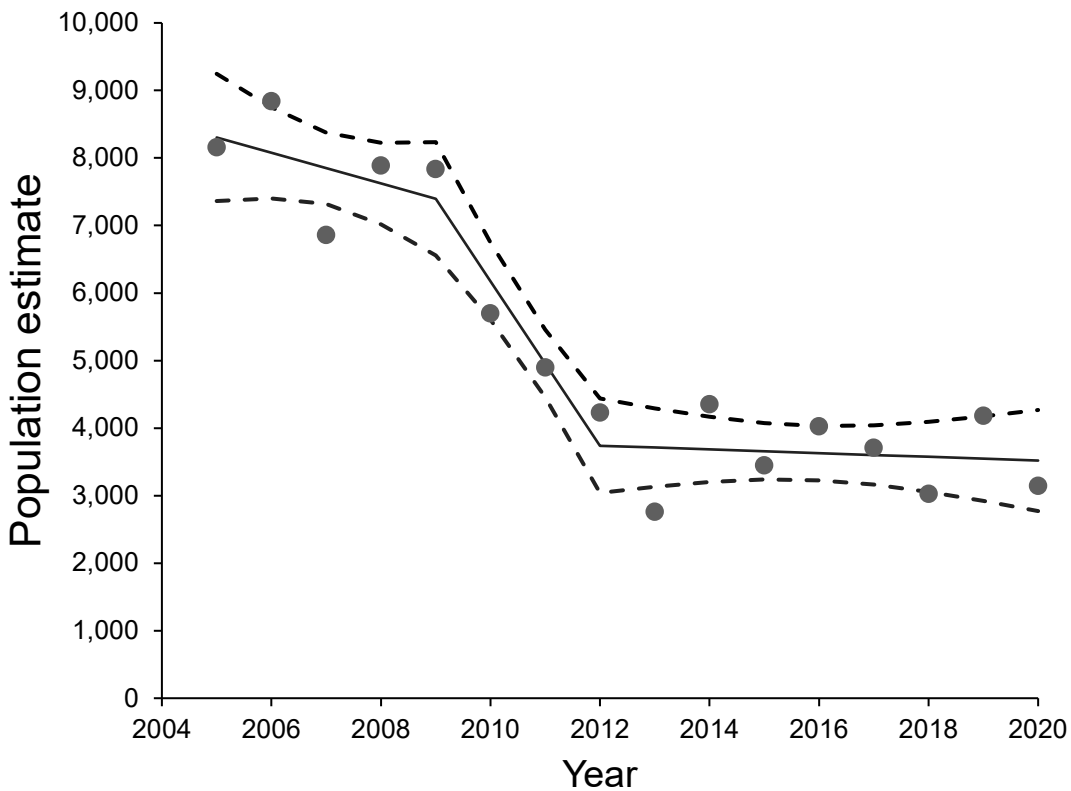


Figure 3. Point estimates, 95% confidence intervals (dashed lines), and a piecewise polynomial curve of moose abundance in northeastern Minnesota, 2005–2020 (Giudice 2020). This curve shows a change in the short-term slope of the trend from 2012 to 2020 compared to 2009 to 2012.

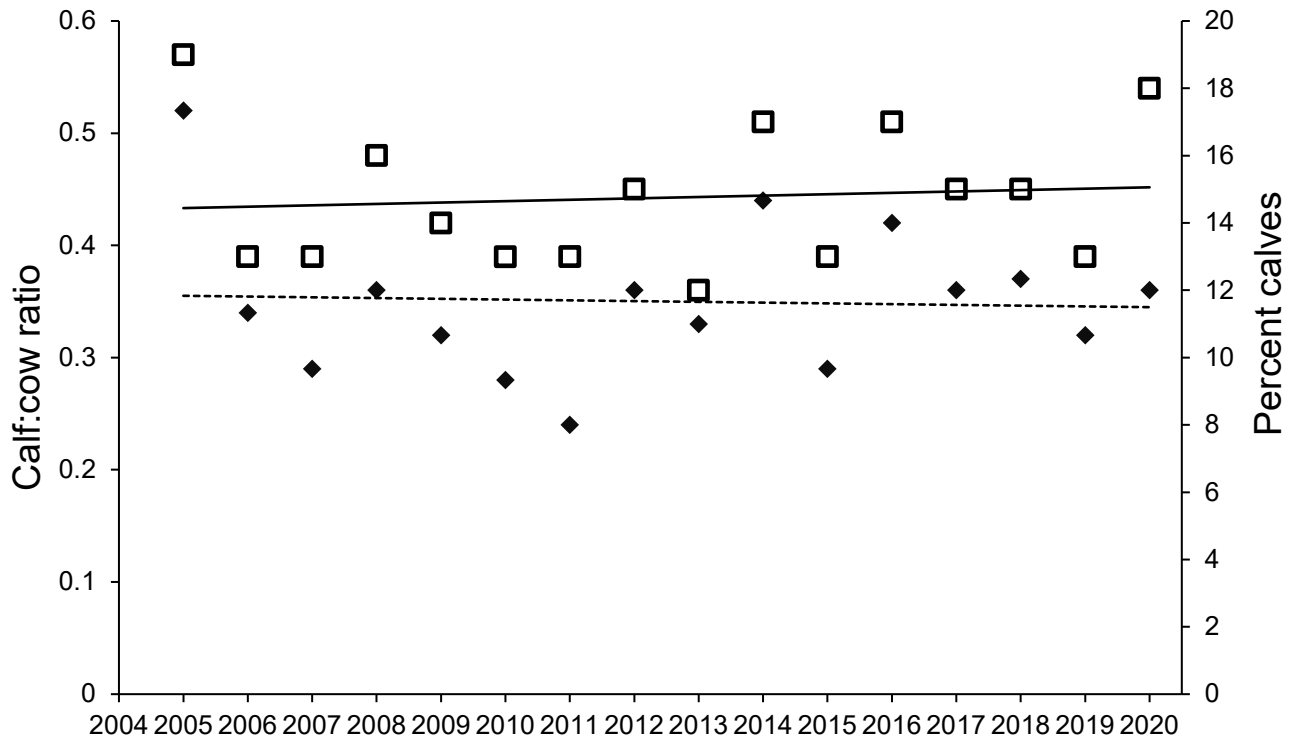


Figure 4. Estimated calf:cow ratios (solid diamonds, dashed trend line) and percent calves (open squares, solid trend line) of the population from aerial moose surveys in northeastern Minnesota, 2005–2020.

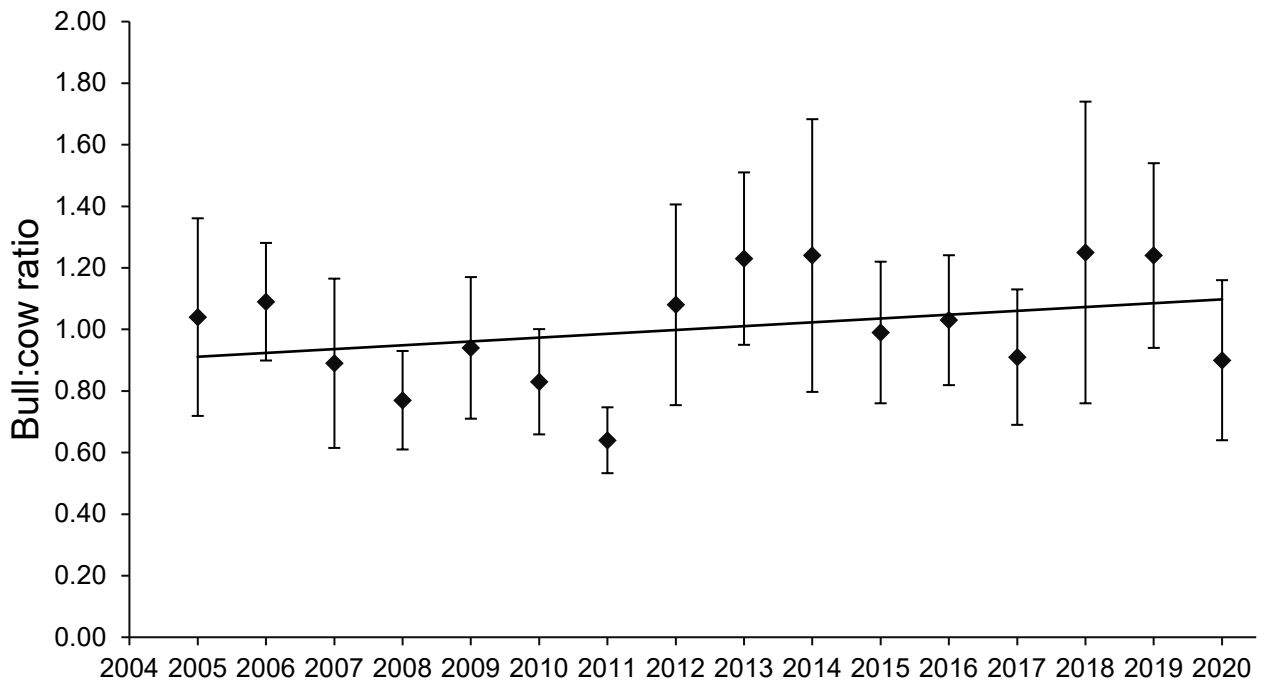


Figure 5. Estimated bull:cow ratios, 90% confidence intervals, and trend line from aerial moose surveys in northeastern Minnesota, 2005–2020.

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MINNESOTA WOLF POPULATION UPDATE 2020

John Erb and Carolin Humpal, Forest Wildlife Populations and Research Group

INTRODUCTION

Since the late 1970's, Minnesota has monitored its statewide wolf population using an approach that combines attributes of territory mapping with an *ad hoc* approach to determine the total area of the state occupied by wolf packs. The methods employed have changed only slightly during this time. Initially, surveys were conducted at approximately 10-year intervals (1978, 1988, 1997), thereafter at approximately 5-year intervals (2003, 2007, 2012). Results indicated a geographically and numerically expanding population through the 1997-98 survey, with little geographic expansion from 1998 to 2007 (Erb and DonCarlos 2009). These results were generally consistent with separate wolf population trend indicators (annual scent station survey, winter track survey, and number of verified depredations) in Minnesota.

In 2012, wolves in the Western Great Lakes Distinct Population Segment were removed as a listed species under the federal Endangered Species Act. The de-listing coincided with the normally scheduled (every 5th year) wolf survey as well as survey timeline specifications in the Minnesota Wolf Management Plan (i.e., first and fifth year after delisting; Minnesota Department of Natural Resources 2001). The 2012-13 survey (Erb and Sampson 2013) concluded that overall wolf range had expanded along its south and west edge, but with only minor change in the total amount of land occupied by wolf packs; similar patterns were found 5 years later as part of the winter 2017-18 survey (Erb et al. 2018).

After federal de-listing in 2012, wolf harvest seasons were established and population surveys have been conducted annually to better inform annual management decisions. In the first three winters after de-listing, wolf population point estimates varied from approximately 2,200 to 2,400 (Erb et al. 2014). In December 2014, following the third consecutive wolf harvest season, wolves in Minnesota were returned to the list of federally threatened species as a result of a court ruling. Since that time, wolf surveys have continued on an annual basis. Herein we provide an update of population status from the 2019-20 winter survey.

METHODS

The methodology used to estimate wolf population size in Minnesota utilizes three primary pieces of information: 1) an estimate of the total area of land occupied by wolf packs; 2) an estimate of average wolf pack territory size; and 3) an estimate of average mid-winter pack size. It is likely that occupied range changes on a comparatively slow timescale compared to fluctuations in average territory and pack size. As such, occupied range is estimated only once every 5 years, with the last being during winter 2017-18; we assume that occupied range has remained unchanged (i.e., 73,972 km²; Erb et al. 2018) and use that in our population calculations for winter 2019-20.

To radio-collar wolves, we and various collaborators captured wolves using foothold traps (LPC # 4, LPC #4 EZ Grip, or LPC #7 EZ Grip) approved as part of research conducted under the Association of Fish and Wildlife Agencies Best Management Practices for trapping program.

Some wolves are also captured with the use of live-restraining neck snares (Gese et al. 2019), and a few by helicopter dart-gun. Wolves were typically immobilized using a mixture of either Ketamine:Xylazine or Telazol:Xylazine. After various project-specific wolf samples and measurements were obtained, the antagonist Yohimbine and an antibiotic were typically administered to all animals prior to release.

Various models of radio-collars were deployed depending on study area and collar availability. Most GPS radio-collars were programmed to take 3-6 locations per day, and wolves fitted with VHF-only radio-collars were relocated at approximately 7- to 10-day intervals throughout the year, or in some cases, primarily from early winter through spring.

To estimate average territory size, we delineated territories of radio-collared packs using minimum convex polygons (MCP) for consistency with previous surveys. Prior to delineating wolf pack territories, we removed 'outlier' radiolocations using the following guidelines, though subjective deviations were made in some cases as deemed biologically appropriate: 1) for wolves with approximately weekly VHF radiolocations only, locations > 5 km from other locations were excluded as extraterritorial forays (Fuller 1989); 2) for GPS collared wolves with temporally fine-scale movement information, we removed obvious movement paths if the animal did not travel to that area on multiple occasions and if use of the path would have resulted in inclusion of obviously unused areas in the MCP; and 3) for consistency with the way in which the data is used (i.e., to estimate number of packs), points that result in notable overlap with adjacent territories are removed.

In past surveys where all or the majority of territories were delineated using VHF radiolocations, raw territory sizes were increased 37% to account for the average amount of interstitial space between delineated wolf pack territories, as estimated from several Minnesota studies (Fuller et al. 1992:50) where the number of radiolocations per pack typically averaged 30-60. Interstitial spaces are a combination of small voids created by landscape geometry and wolf behavior, but can also be an artifact of territory underestimation when there are comparatively sparse radiolocations. Hence, for packs with < 100 radiolocations ($n = 9$; mean number of radiolocations = 21), we multiplied each estimated territory size by 1.37 as in the past. For packs with > 100 radiolocations ($n = 35$; mean number of radiolocations = 2,904), territories were assumed to be fully delineated and were not re-scaled.

To estimate average mid-winter pack size, radio-marked wolves were repeatedly located via aircraft during winter to obtain visual counts of pack size. In cases where visual observations were insufficient, we also rely on any estimates of pack size based on tracks observed in the snow and trail camera images from within the pack's territory. If any reported count produced uncertain estimates (e.g., 4 to 5 wolves), we used the lower estimate. Overall, counts are assumed to represent minimum known mid-winter pack size.

The estimated number of packs within occupied wolf range is computed by dividing the area of occupied range by average scaled territory size. The estimated number of packs is then multiplied by average mid-winter pack size to produce an estimate of pack-associated wolves, which is then divided by 0.85 to account for an estimated 15% lone wolves in the population (Fuller et al. 1992:46, Fuller et al. 2003:170). Specifically,

$$N = ((\text{km}^2 \text{ of occupied range} / \text{mean scaled territory size}) * \text{mean pack size}) / 0.85.$$

Using the accelerated bias-corrected method (Manly 1997), the population size confidence interval (90%) was generated from 9,999 bootstrapped re-samples of the pack and territory size data and does not incorporate uncertainty in estimates of occupied range or percent lone wolves. For purposes of discussion, we base our informal assessments of significant differences

in results across years on visual comparison of the degree of confidence interval overlap (Cumming and Finch 2005).

RESULTS AND DISCUSSION

Pack and Territory Size

We obtained data on 48 packs that were monitored during all or part of the survey period (April 2019 to April 2020). We obtained territory and winter pack size data from 31 radio-marked wolf packs (Figure 1). Thirteen additional wolf packs had adequate radiolocation data to delineate territories, but we were unable to obtain mid-winter pack counts, and we obtained pack counts on 4 packs for which there was insufficient data to delineate a territory.

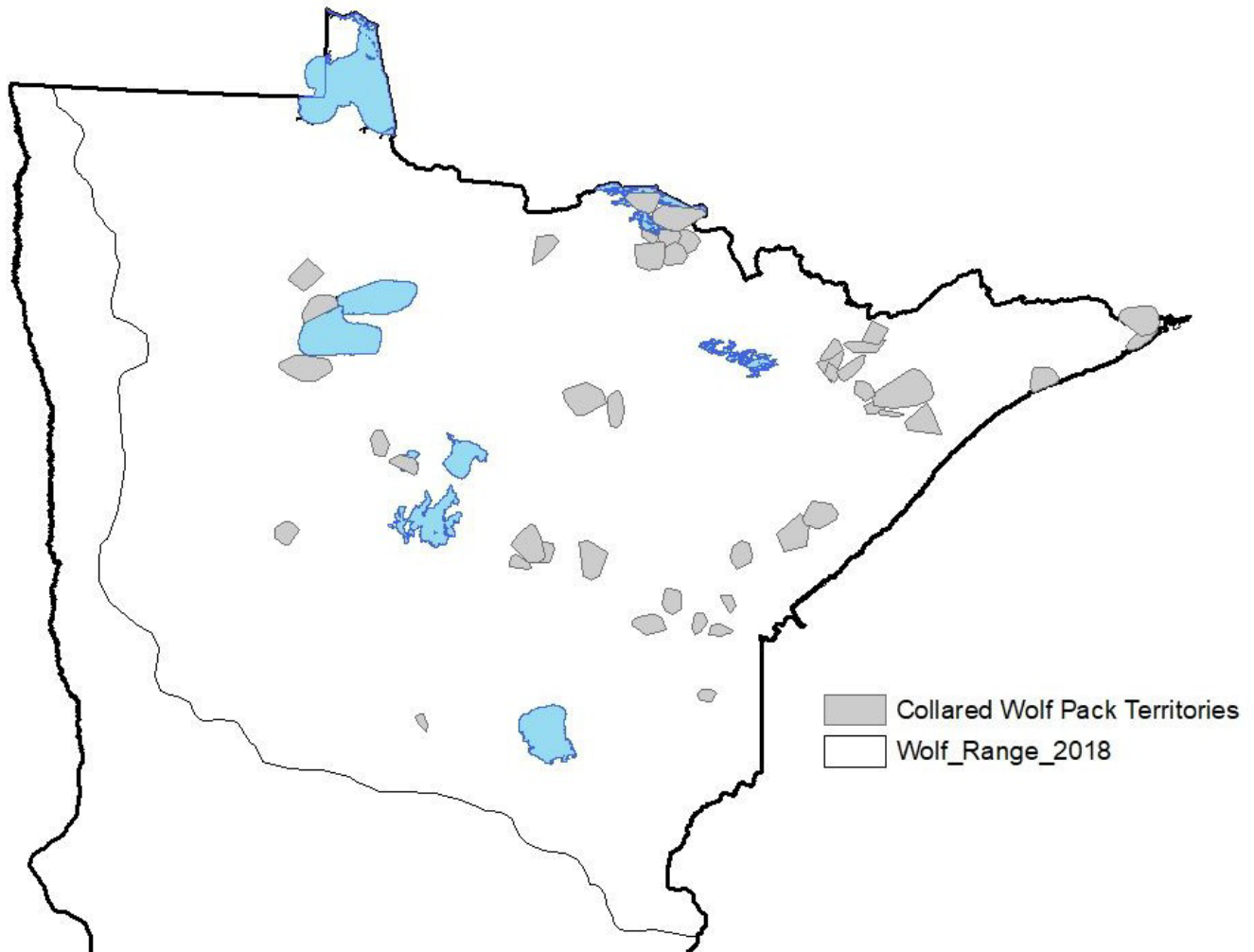


Figure 1. Location of radio-marked wolf packs during the 2019-20 survey.

A land cover comparison using the 2011 National Land Cover Database suggests that the location of collared packs this winter led to some under-representation of cover types classified as woody wetlands (Table 1), likely a result of under-sampling of packs in portions of northwest MN around Red Lake where territories tend to be larger. There was also under-representation of cover types classified as pasture-hay-grassland (Table 1), likely a result of fewer collared packs in our south and southwest study areas where territories tend to be closer to, or smaller than, the statewide average. Using spring 2019 deer density data (MNDNR, unpublished data) for deer hunting permit areas, weighted by number of radio-collared wolf packs in a permit area,

we estimate an average of approximately 9.2 deer/mi² (pre-fawn) in territories of radio-marked packs during spring 2019. In comparison, 2019 spring deer density for the entirety of occupied wolf range (weighted by permit area) in Minnesota was approximately 13 deer/mi².

Table 1. Comparison of land cover^a in territories of radio-collared wolf packs with land cover in all of occupied wolf range in Minnesota.

Land Cover Category	Overall Occupied Wolf range	Radio-collared Wolf Territories
	% Area	% Area
Woody Wetlands	32.6	31.5
Deciduous Forest	23.6	21.4
Emergent Herbaceous Wetlands	9.9	5.3
Mixed Forest	7.2	11.2
Evergreen Forest	7.0	12.7
Open Water	5.4	6.4
Shrub/Scrub	4.5	6.8
Pasture/Hay/Grassland/Crops	7.7	2.9
Developed, All	2.2	1.8

^a Land cover data derived from the 2011 National Land Cover Database

The point estimate for average territory size in winter 2019-20 declined 21% from the previous winter. This represents a marginally significant decline and the lowest point estimate for average territory size since surveys began (Figure 2). After applying the territory scaling factors, average estimated territory size for radio-marked packs during the 2019-20 survey was 117 km² (range = 33 – 378 km²).

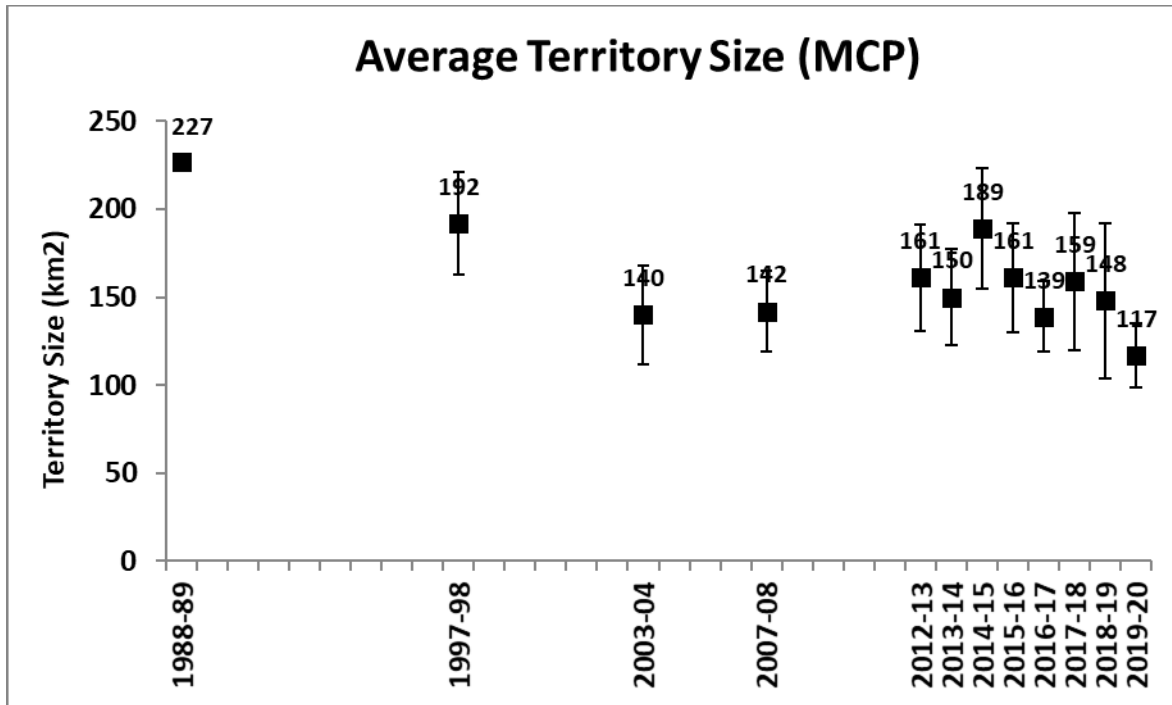


Figure 2. Average scaled territory size for radio-marked wolf packs in Minnesota from winter 1988-89 to 2019-20.

The point estimate for average winter pack size also declined by 21%, a significant decline and the lowest average pack size since surveys began. Average winter pack size in 2019-20 was estimated to be 3.6 (range = 2 – 7, Figure 3).

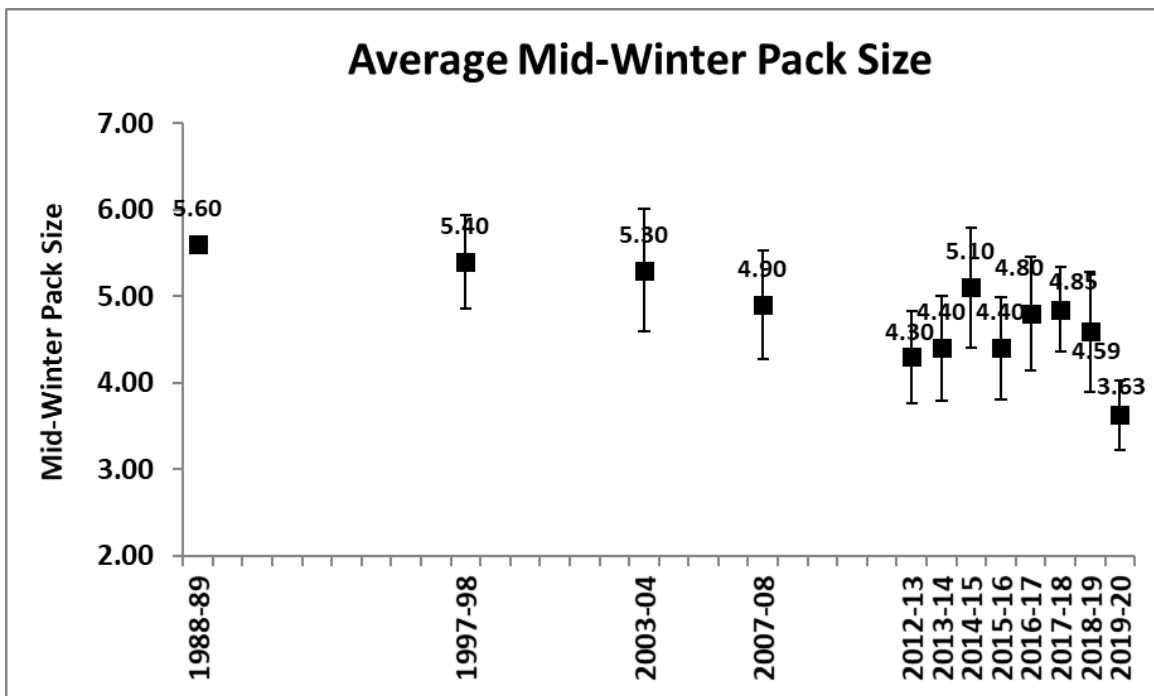


Figure 3. Average mid-winter pack size for radio-marked wolf packs in Minnesota from winter 1988- 89 to 2019-20.

Wolf Numbers

Given an average territory size of 117 km² and assuming occupied range has not changed since the 2017-18 survey (73,972 km²; Erb et al. 2018), we estimated a total of 631 wolf packs in Minnesota during winter 2019-20. Although also influenced by the estimated amount of occupied range, trends in the estimated number of packs are inversely correlated with trends in estimated territory size (i.e., for a given amount of occupied range, increases in average territory size yield lower estimates of the number of packs within the state).

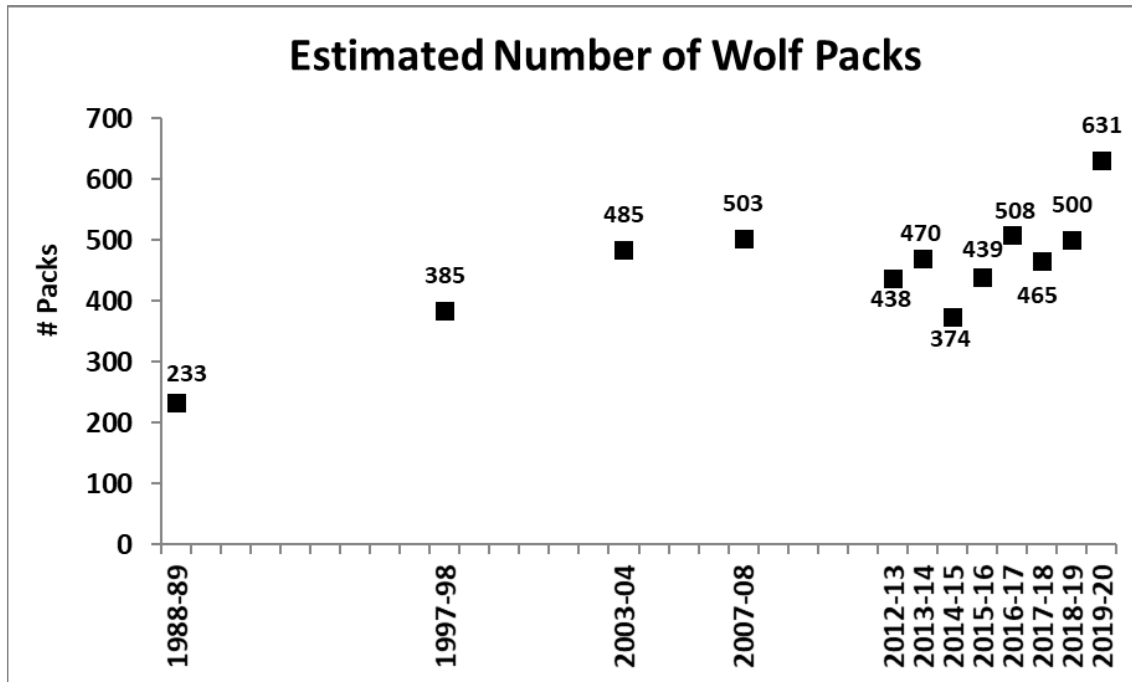


Figure 4. Estimated number of wolf packs in Minnesota at periodic intervals from winter 1988-89 to 2019-20.

After accounting for the assumed 15% lone wolves in the population, we estimated the 2019-20 mid-winter wolf population at 2,696 wolves, or 3.64 wolves per 100 km² of occupied range. The 90% confidence interval was approximately +/- 500 wolves, specifically 2,244 to 3,252. Given the nearly complete overlap with the 2019-20 confidence interval, we conclude that the 2019-20 statewide wolf population size was unchanged from the previous winter, but with results suggesting more but smaller packs.

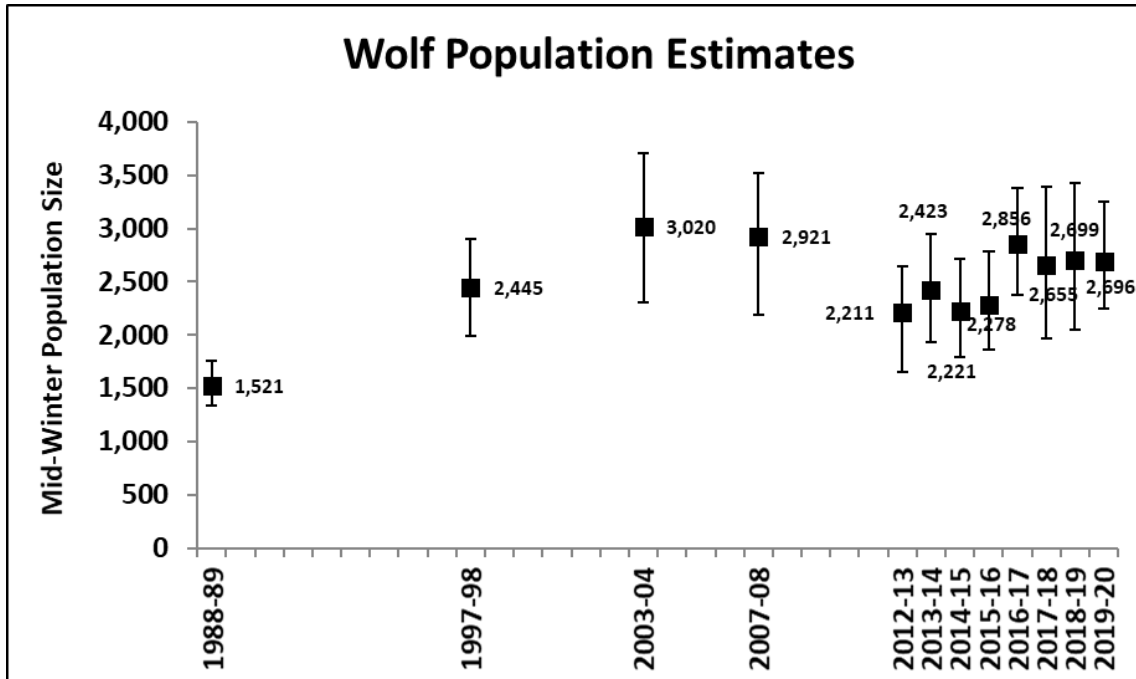


Figure 5. Wolf population estimates from periodic standardized surveys in Minnesota from winter 1988-89 to 2019-20.

From spring 2018 to spring 2019, overall average deer density within wolf range remained stable. Over the past 8 years, the trend in winter wolf population size has been positively correlated with average deer density within wolf range the preceding spring (Figure 6).

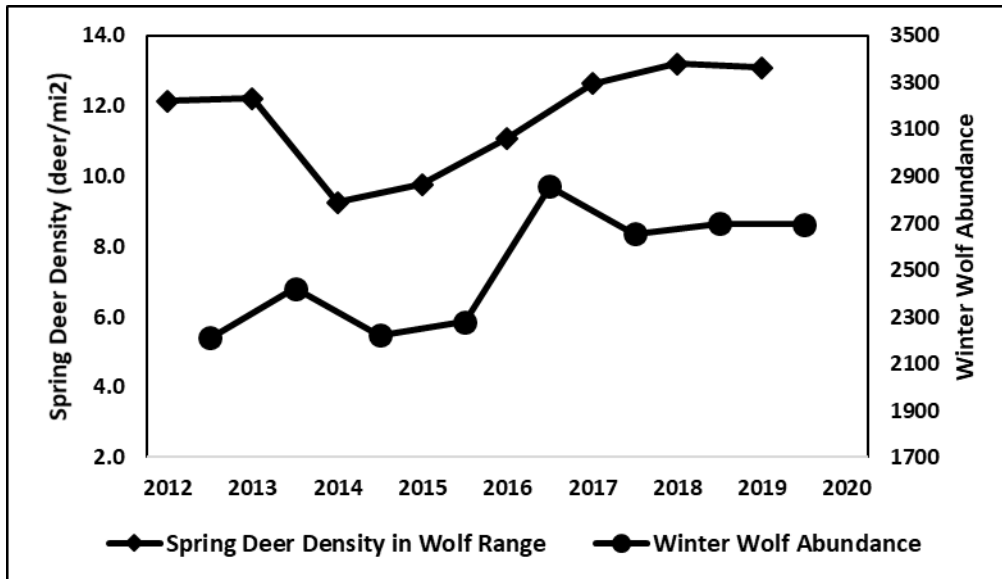


Figure 6. Comparison of estimated spring (pre-fawn) deer density and winter wolf abundance in Minnesota, 2012-2019.

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