FOREST WILDLIFE POPULATIONS

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CARNIVORE SCENT STATION SURVEY SUMMARY, 2018

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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 42nd 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/absence 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/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/absence 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 187 routes and 1,721 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 953 km^2 in the Forest Zone to 1 route per 1,480 km^2 in the Farmland Zone (Figure 1). The decline in survey effort was likely a result of staffing shortages and competing workload demands.

Statewide, route visitation rates (% of routes with detection), in order of increasing magnitude, were bobcats (7%), opossums (8%), wolves (10%), domestic dogs (15%), domestic cats (22%), red foxes (24%), coyotes (29%), skunks (31%), and raccoons (33%). Regionally, route visitation rates were as follows: red fox – FA 17%, TR 24%, FO 28%; coyote – FO 15%, TR 35%, FA 50%; skunk – FO 22%, TR 26%, FA 54%; raccoon – FO 6%, TR 37%, FA 80%; domestic cat – FO 6%, TR 30%, FA 46%; domestic dog – FO 5%, TR 22%, FA 26; opossum - FO 0%, FA 11%, TR 19%; wolf - FA 0%, TR 0%, FO 22%; and bobcat - FA 0%, TR 7%, FO 11%.

Figures 2-5 show station visitation indices (% of stations visited) from the survey's inception through the current year. Although the survey is largely intended to document long- term trends in populations, confidence intervals improve interpretation of the significance of annual changes. Based strictly on the degree of confidence interval overlap, notable changes this year include 1) marginally significant declines in red fox indices in both the Farmland and Forest Zones (Figures 2 and 4), 2) a decline in the domestic cat index in the Farmland Zone (Figure 2), and 3) a decline in the raccoon index in the Forest Zone (Figure 4).

In the Farmland Zone (Figure 2), the red fox index exhibited a marginally significant decline, and indices have remained below the long-term average for nearly 20 years. Although the farmland coyote index has increased over time and remains above the long-term average, indices have been stable over the last 4 years. Raccoon indices also remain above their long- term average, but have been relatively stable over the last 20 years. There has been no consistent trend in Farmland skunk indices for nearly 3 decades, with the current index near the long-term average.

There were no significant changes from last year for any species in the Transition Zone (Figure 3). Coyote and bobcat indices in the Transition have increased over time and are above their long-term averages, whereas red fox indices have been below their long-term averages for most of the last 2 decades. Raccoon and skunk indices have generally been stable and near their long-term averages over the last 2 decades. Wolves had exhibited a mild increase in the Transition Zone over time, but indices have been below the long-term average the past 2 years.

In the Forest Zone (Figures 4 and 5), the raccoon index exhibited a significant decline from last year and was the lowest since the early 1980's. The red fox index exhibited a marginally significant decline, and has been near or slightly below the long-term average in the Forest Zone for the last 2 decades. Unlike in the Farmland and Transition Zones, the Forest Zone coyote index has not increased over time and has been stable and below the long-term average for 2 decades, likely attributable to wolf presence in the Forest Zone. Skunk indices have also remained below their long-term average in the Forest Zone over the past 2 decades. Wolf and bobcat indices have been at peak levels over the past decade and remain above their long-term averages, but both have also exhibited fluctuations during this time.

ACKNOWLEDGMENTS

I wish to thank all of the cooperators who participated in the 2018 survey: DNR Division of Wildlife staff; Superior National Forest Aurora District; Sherburne 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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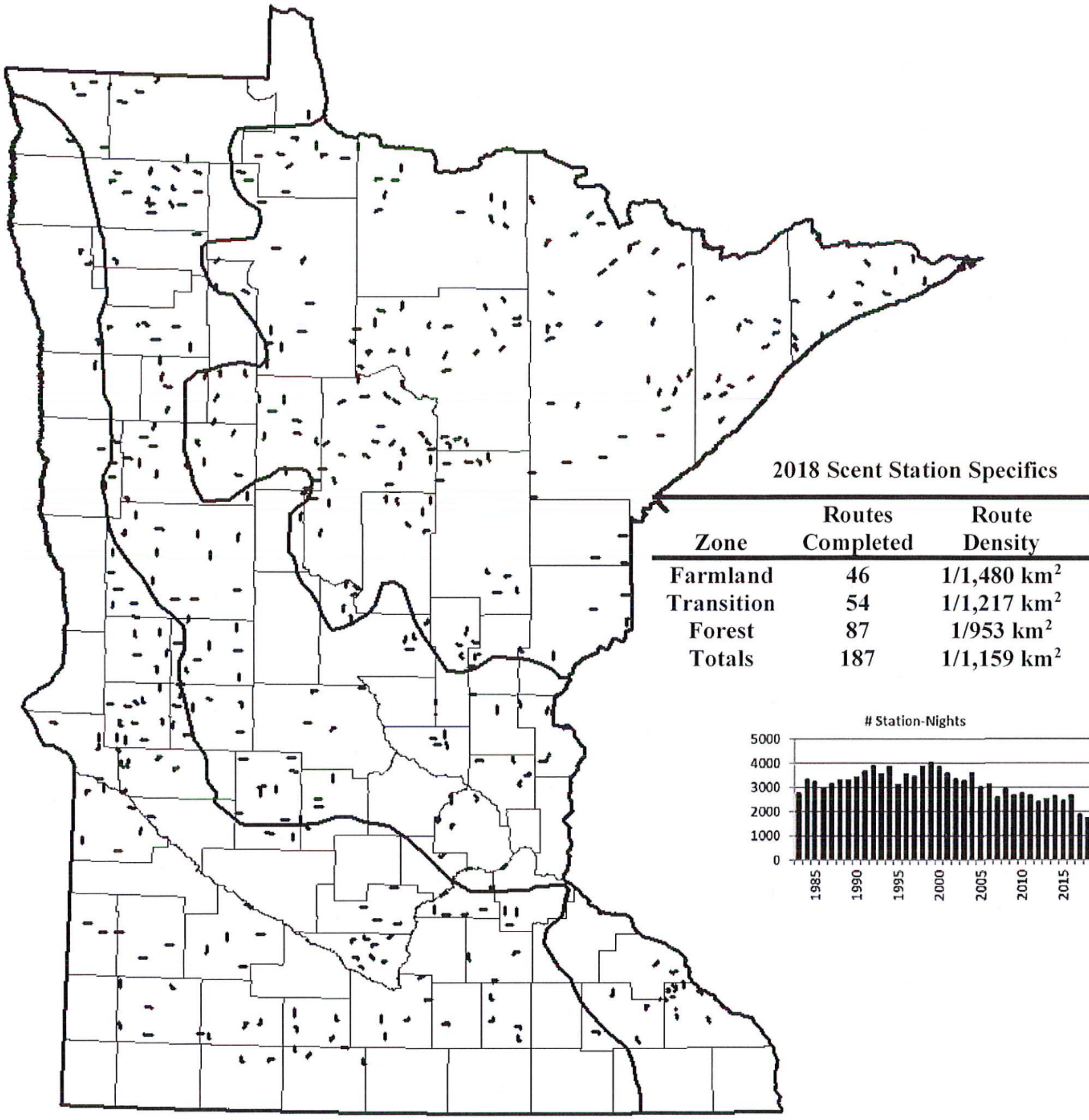

2018 Scent Station Specifics

Zone	Routes Completed	Route Density	Station Nights
Farmland	46	1/1,480 km^2	425
Transition	54	1/1,217 km^2	500
Forest	87	1/953 km^2	796
Totals	187	1/1,159 km^2	1,721

Figure 1. Locations of existing scent station routes (not all completed every year). Insets show 2018 route specifics and the number of station-nights per year since 1983.

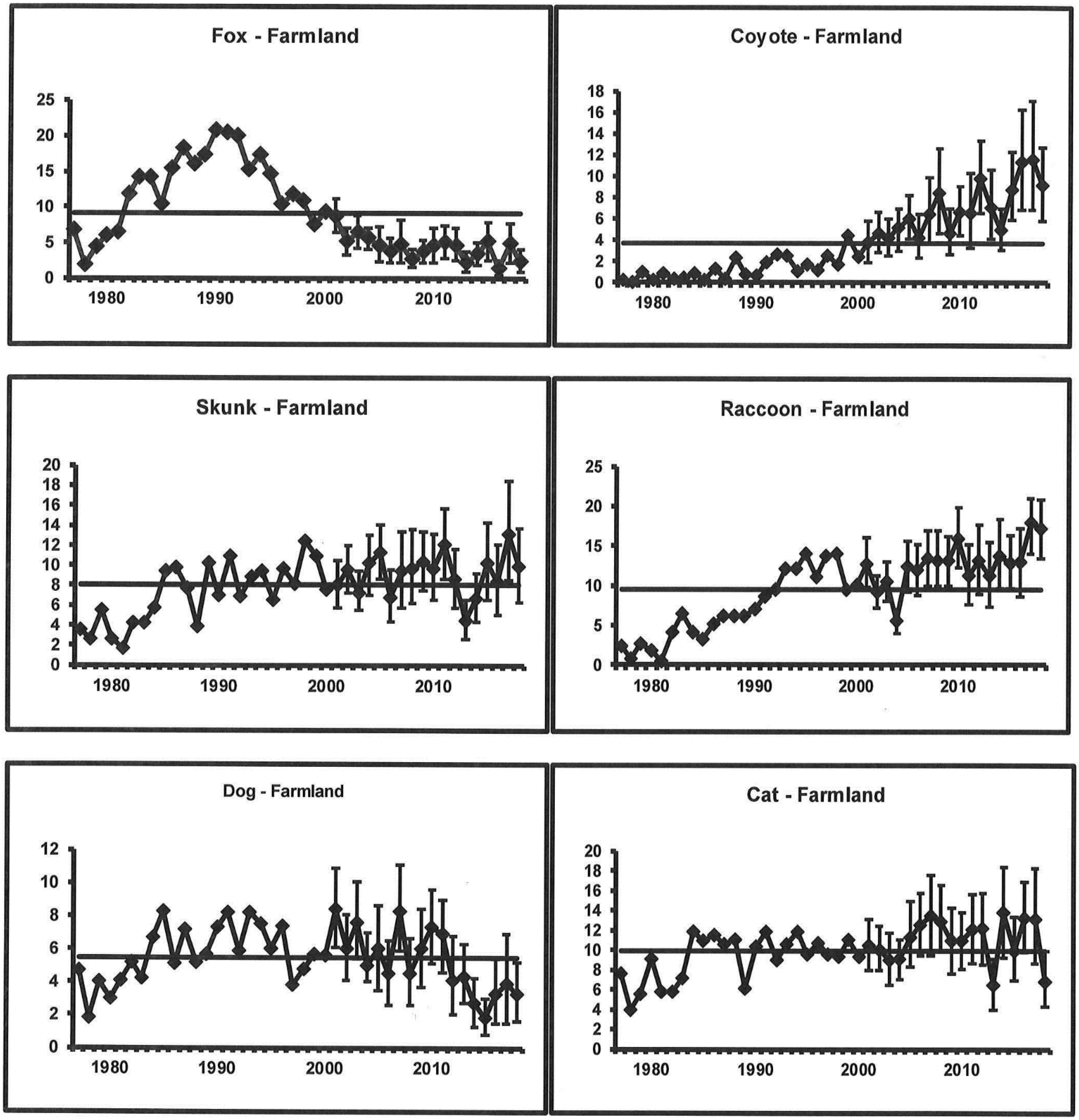

Figure 2. Percentage of scent stations visited by selected species in the Farmland Zone of Minnesota, 1977-2018. Horizontal line represents long-term mean.

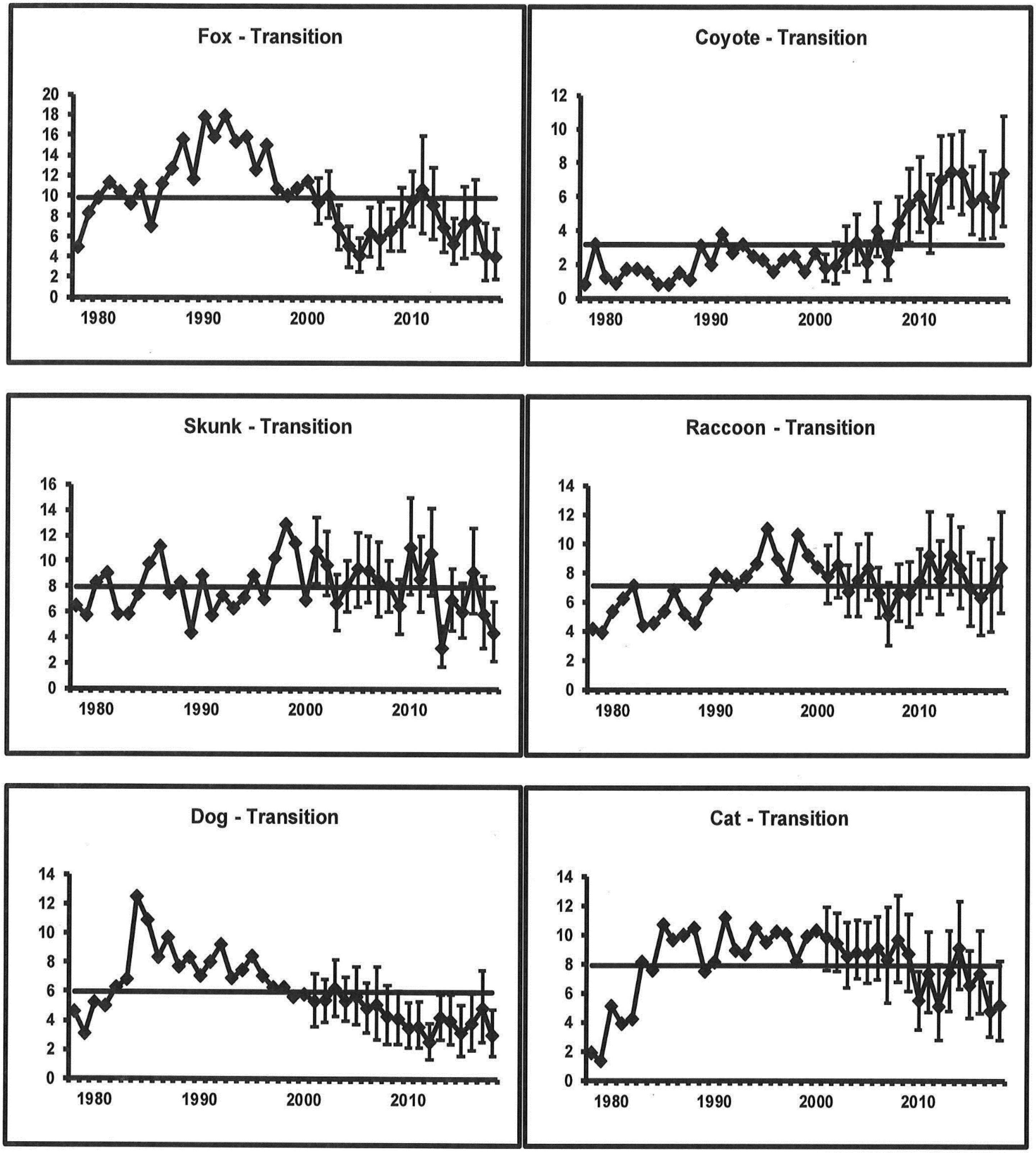

Figure 3. Percentage of scent stations visited by selected species in the Transition Zone of Minnesota, 1978-2018. Horizontal line represents long-term mean.

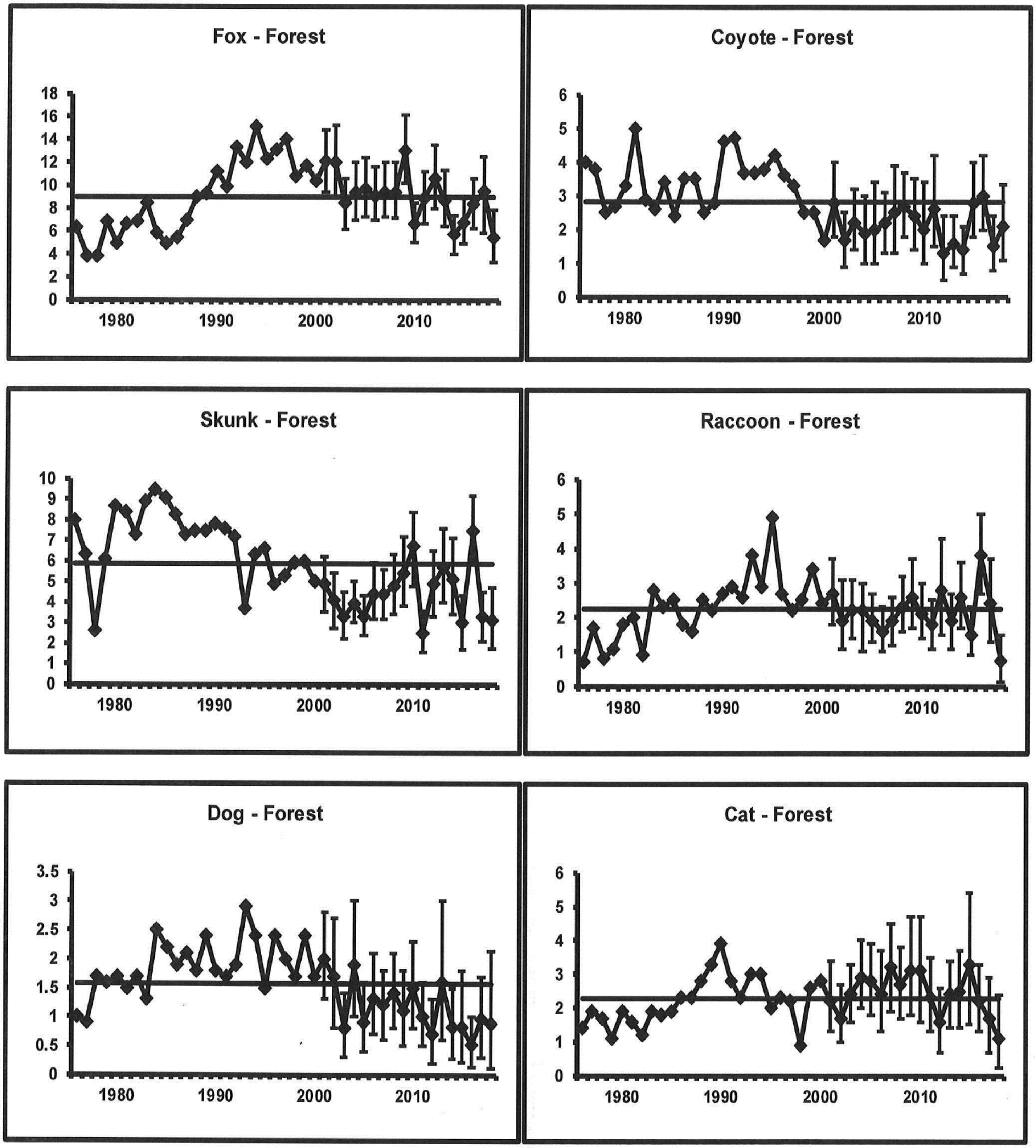

Figure 4. Percentage of scent stations visited by selected species in the Forest Zone of Minnesota, 1976-2018. Horizontal line represents long-term mean.

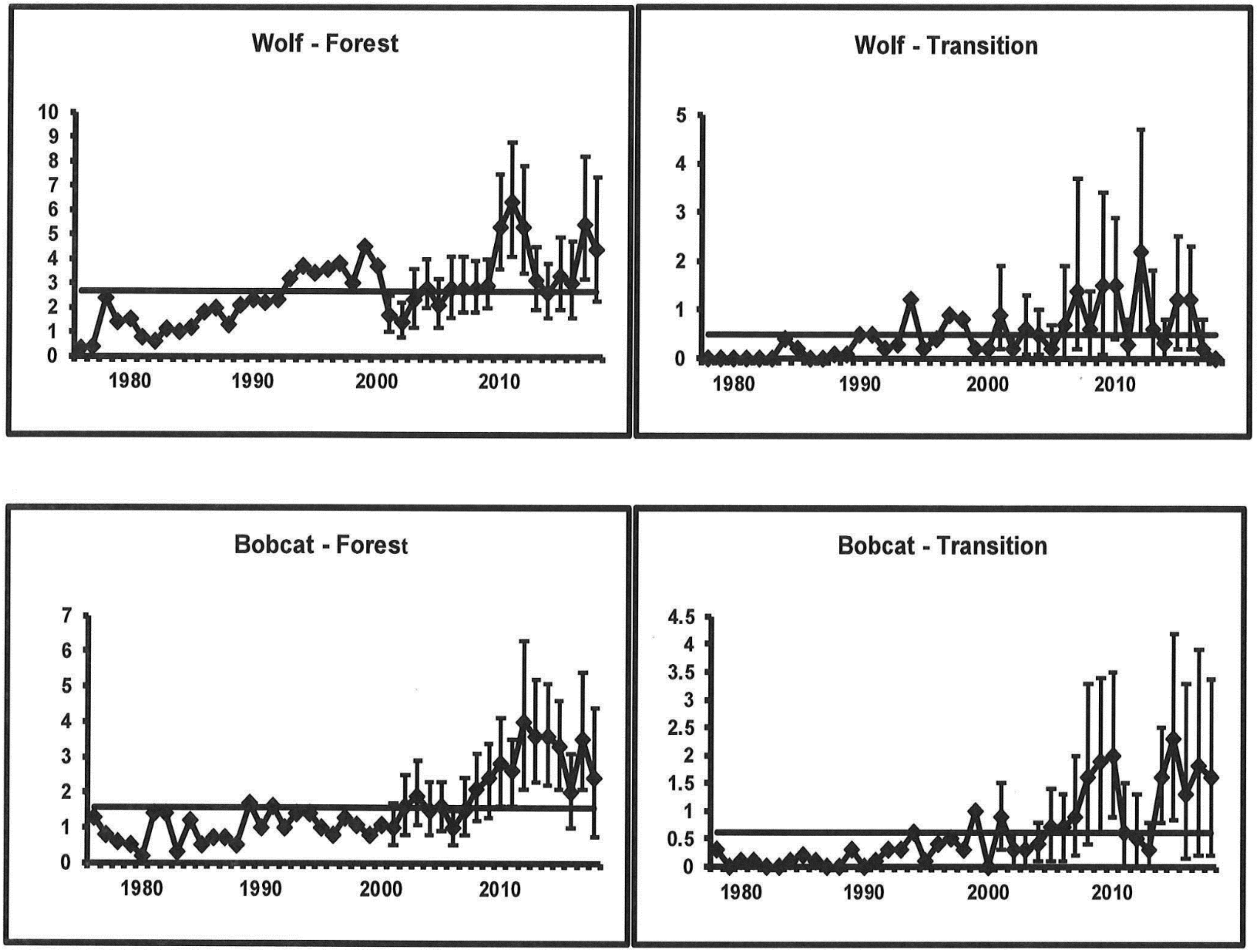

Figure 5. Percentage of scent stations visited by wolves and bobcat in the Forest and Transition Zones of Minnesota, 1976-2018. Horizontal lines represents long-term mean

FURBEARER WINTER TRACK SURVEY SUMMARY, 2018

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 these 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 such 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 fisher (*Pekania pennanti*) and marten (*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 detections did not commence until 2008.

METHODS

Presently, 57 track survey routes are operational across the northern portion of the state (Figure 1). Each route is a total of 10 miles long and follows secondary roads or trails. A majority of routes are continuous 10-mile stretches of road/trail but a few are composed of multiple discontinuous segments. Route locations were subjectively determined based on availability of suitable roads/trails but were chosen where 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 only recorded 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 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 0.1miles of each 0.5-mile segment. Although most routes are surveyed one day after the conclusion of a snowfall (ending by ~ 6:00 pm), thereby allowing one night for tracks to be left, a few routes are usually completed two nights following snowfall. In such cases, track counts on those routes are divided by the number of days post-snowfall.

Because most targeted species 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. However, in general there were minimal differences in temporal patterns observed in this subset versus the full sample of routes.

Currently, three summary statistics are presented 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 fox clearly traverses two adjacent 0.5-mile segments along the road, and it was the only 'new' red fox (*Vulpes vulpes*) 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 presented after removing any obvious duplicate tracks across segments. For wolves (*Canis lupus*) traveling through adjacent segments, the maximum number of pack members recorded in any one of those segments is used 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 (% segments occupied and # 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 fox 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, both indices are currently presented. Because snowshoe hares are tallied only as present/absent, the 2 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'). This measure is computed to help assess whether any notable changes in the above-described track indices are a result of larger-scale changes in distribution (more/less routes with presence) or finer-scale changes in density along routes.

Using bootstrap methods, 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.

RESULTS

This winter, 42 of the 57 routes were completed (Figure 2). Survey routes took an average of 2 hours to complete. Snow depths averaged 18.4" along completed routes, the second-most since the survey began (Figure 3). Mean overnight low temperature the night preceding the surveys was 4°F, similar to the long-term average (Figure 3). Survey routes were completed between November 21st and March 11th, with a mean survey date of January 23rd, the second latest since the survey began (Figure 3).

Based on degree of confidence interval overlap, notable changes from last winter include a significant decrease in red foxes, a marginally significant decline in weasels and wolves, and a marginally significant increase in coyotes (Figure 4). For species monitored on both surveys, these changes mirror results from the fall scent station survey in the Forest Zone.

Fishers were detected on approximately 3% of the route segments and along 40% of the routes (Figure 4). Numerous sources of information indicate that 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 presumed indicative of population trends only in the previous 'core' of fisher range. In the core area, data indicates a longer-term decline, with low but stable numbers since 2012; at their peak (2003/2004), fishers were detected on 14% of route segments and 78% of the survey routes.

Within the 'marten zone', martens were detected on approximately 6% of the route segments and 55% of the survey routes (Figure 4), nearly identical to last year. Similar to results for fishers, marten indices have declined over the long-term, but have been low and without consistent trend over the last 11 years. However, marten fluctuations 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., Oestricher 2018, Berg et al. 2017).

Bobcat indices had increased for approximately 15 years through 2014, and then declined to their long-term average by 2016. Data from the past 2 years show a quick rebound from the recent decline, with the indices approaching peak levels once again. Bobcats were detected on 4.1% of the segments and 45% of the routes.

Wolves were detected on approximately 9% of the route segments and 76% of the survey routes, both down slightly from last winter (Figure 4). The average number of wolves detected per route was 3. Coyotes were detected on 3.6% of the route segments and 45% of the routes. As with martens and weasels (see below), coyote indices appear to exhibit 3 to 5 year cycles consistent in timing with data for some rodent species 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 8% of the segments and 67% of the routes (Figure 4), both significant declines from the previous winter. Gray fox detections have only been formally recorded since 2008. Although it is premature to characterize longer patterns in gray fox detections, data from the past 10 years suggests, similar to coyotes, martens, and weasels, some potential influence of cyclic prey fluctuations. There was a significant decrease in gray fox indices from last winter, with gray foxes being detected on < 1% of the route segments and 2% of the routes.

Weasel (*Mustela erminea* and *Mustela frenata*) indices exhibited a marginally significant decline from last winter and their long-term fluctuations continue to be characterized by 4 to 5 year cycles or 'irruptions' superimposed on a declining trend (Figure 4). No significant change was observed in winter snowshoe hare indices from last winter. Since the winter track survey began in 1994, hare indices had steadily increased, leveled off some around 2010, and have slowly declined since (Figure 4). Both the spring and winter indices were slightly below their long-term averages (Figure 4). Historic data (pre-1994; not presented here) for the spring index of snowshoe hares 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.

DISCUSSION

Reliable interpretation of changes in these 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. Notable changes detected this winter were a significant decrease in red foxes, a marginally significant decline in weasels and wolves, and a marginally significant increase in coyotes. With the exception of ambient temperature, the timing and conditions during this winter's survey suggest conditions more 'extreme' than their long-term averages (i.e., second latest average completion date, second highest snow depths). Although this could negatively bias indices for some species as a result of reduced animal activity, it is not currently possible to quantify and adjust for these potential effects and there is no indication that results were consistently biased downward for all species. Nonetheless, it remains a possible factor and inferences from this survey should largely be restricted to examination of long-term trends.

ACKNOWLEDGMENTS

I wish to thank all those who participated in this year's survey, including staff with the Minnesota DNR, Superior National Forest (Cook, Ely, and Grand Marais offices), Leech Lake, 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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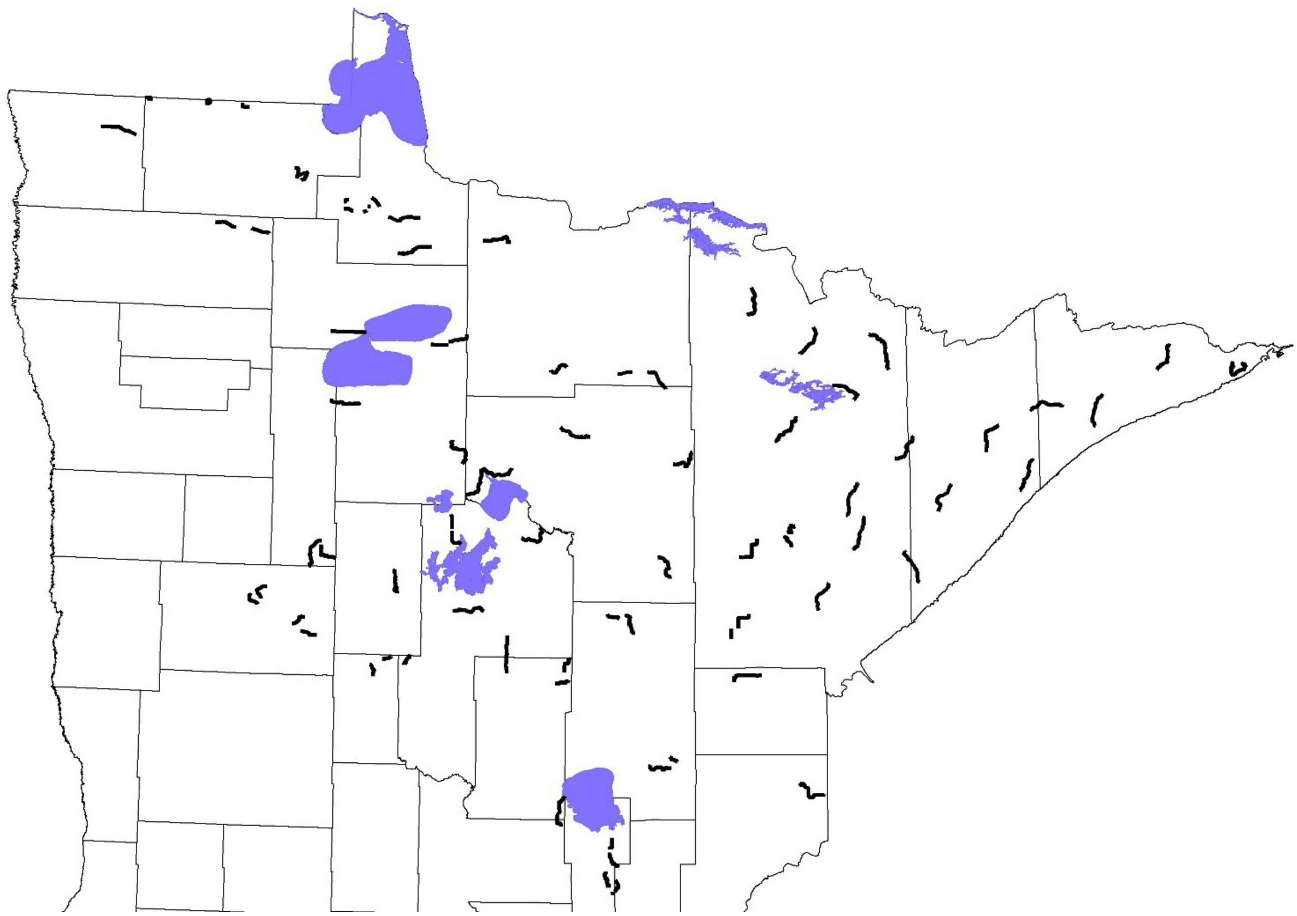

Figure 1. Locations of furbearer winter track survey routes in northern Minnesota.

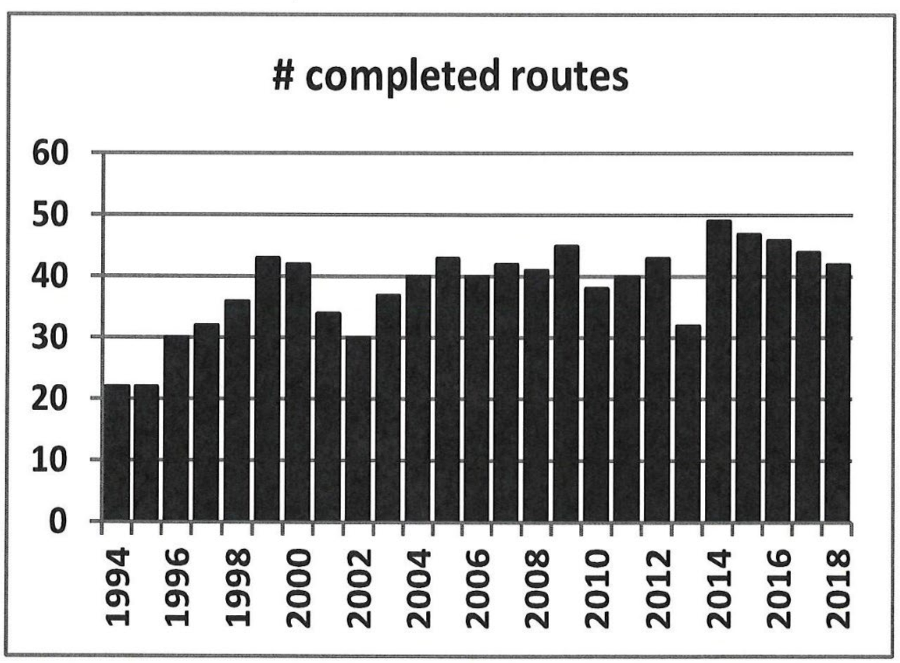

Figure 2. Number of snow track routes surveyed in Minnesota, 1994-2018.

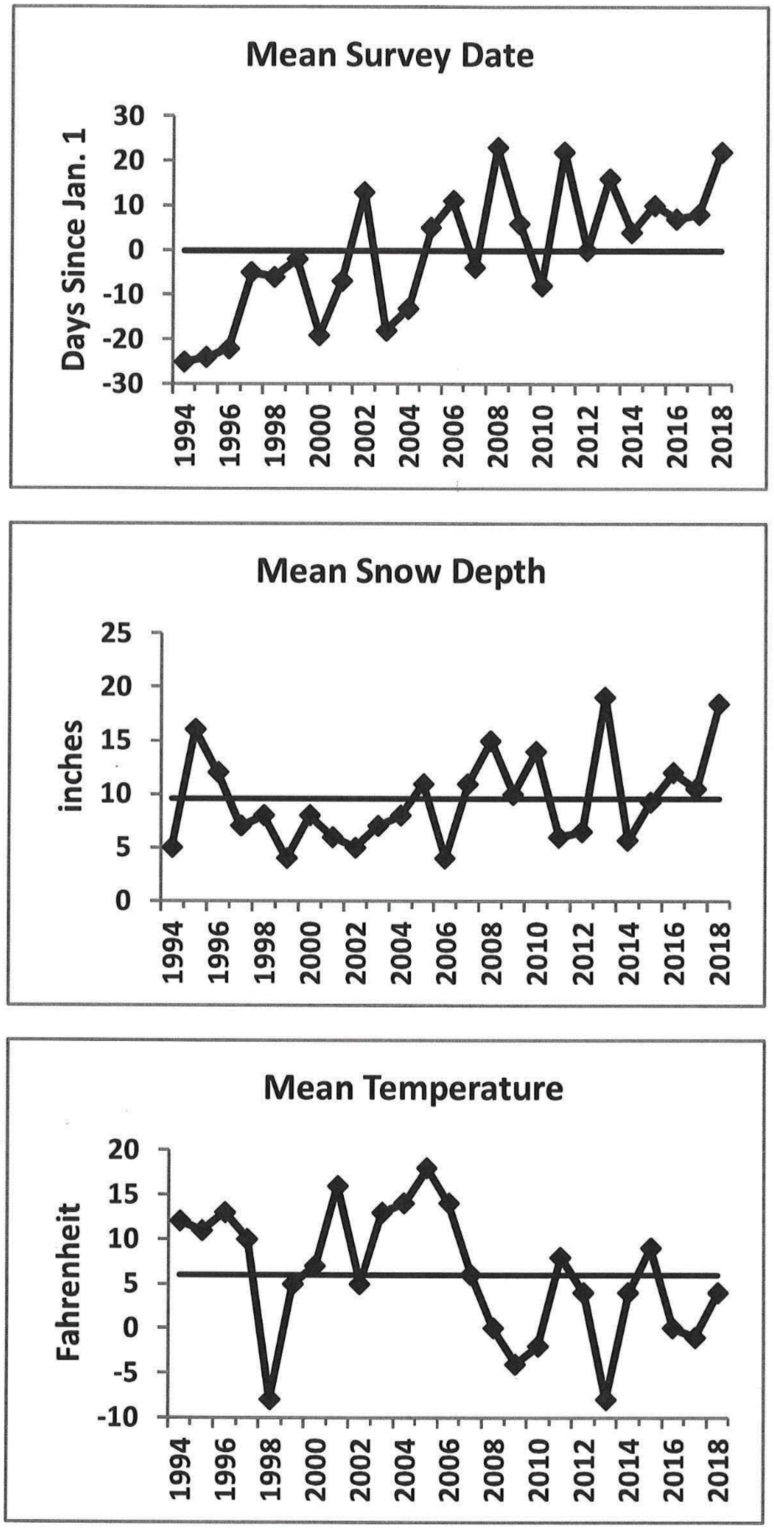

Figure 3. Average survey date, snow depth, and temperature for snow track routes completed in Minnesota, 1994-2018. Horizontal line represents long-term mean.

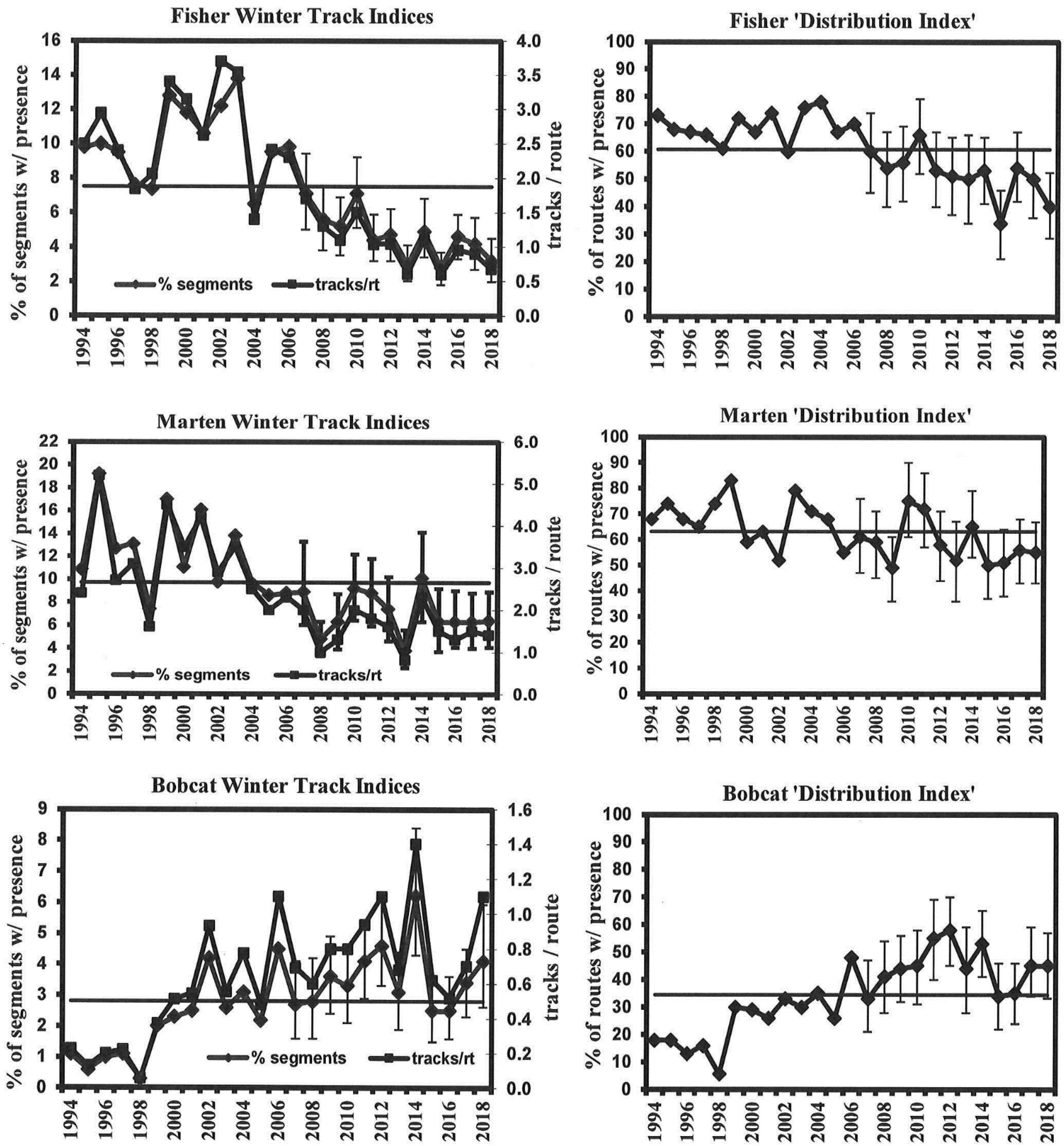

Figure 4. Winter track indices for selected species in Minnesota, 1994-2018. Confidence intervals are presented only for % segments and % routes with track presence; horizontal lines represent their long-term averages.

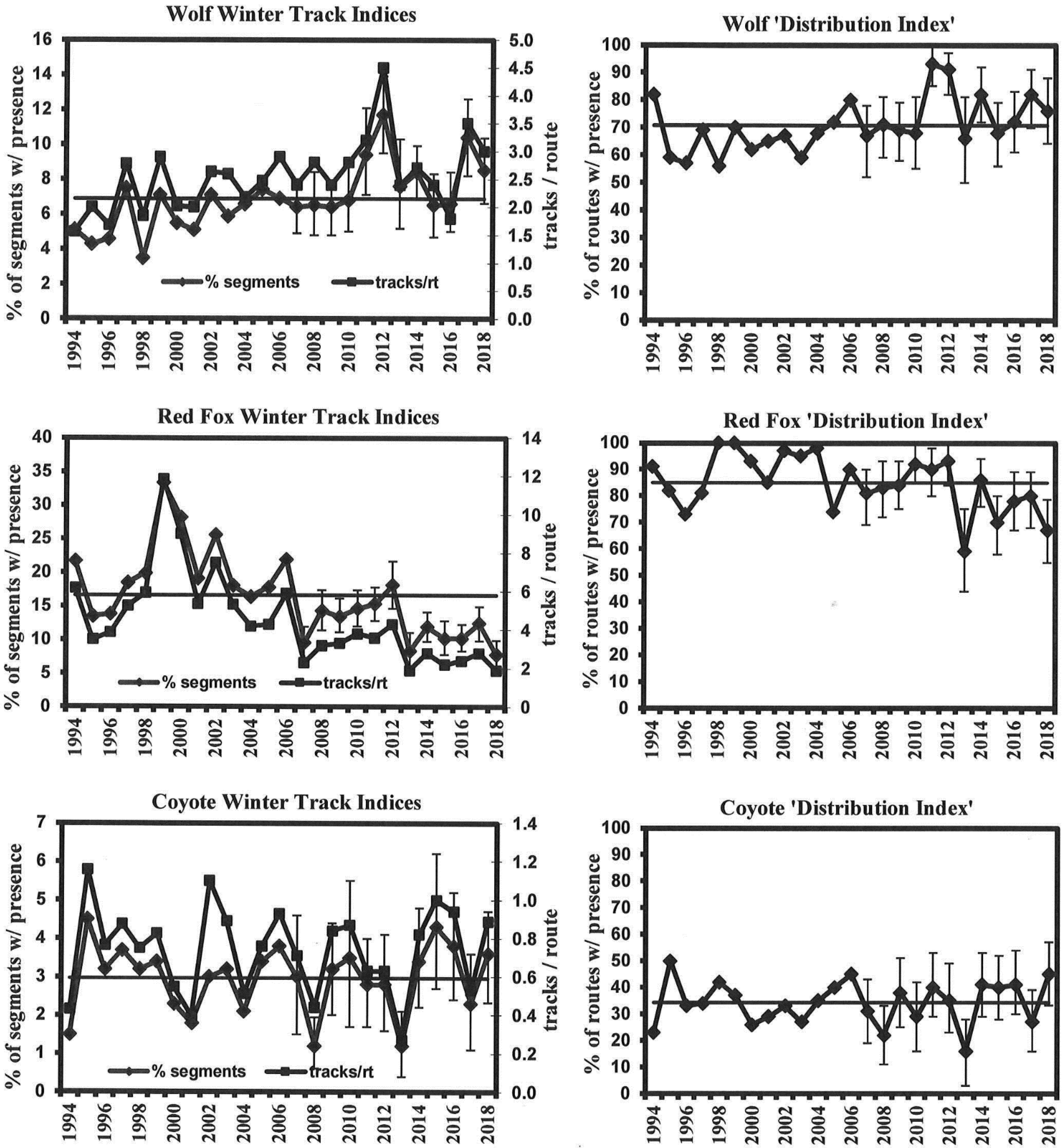

Figure 4 (continued). Winter track indices for selected species in Minnesota, 1994-2018.

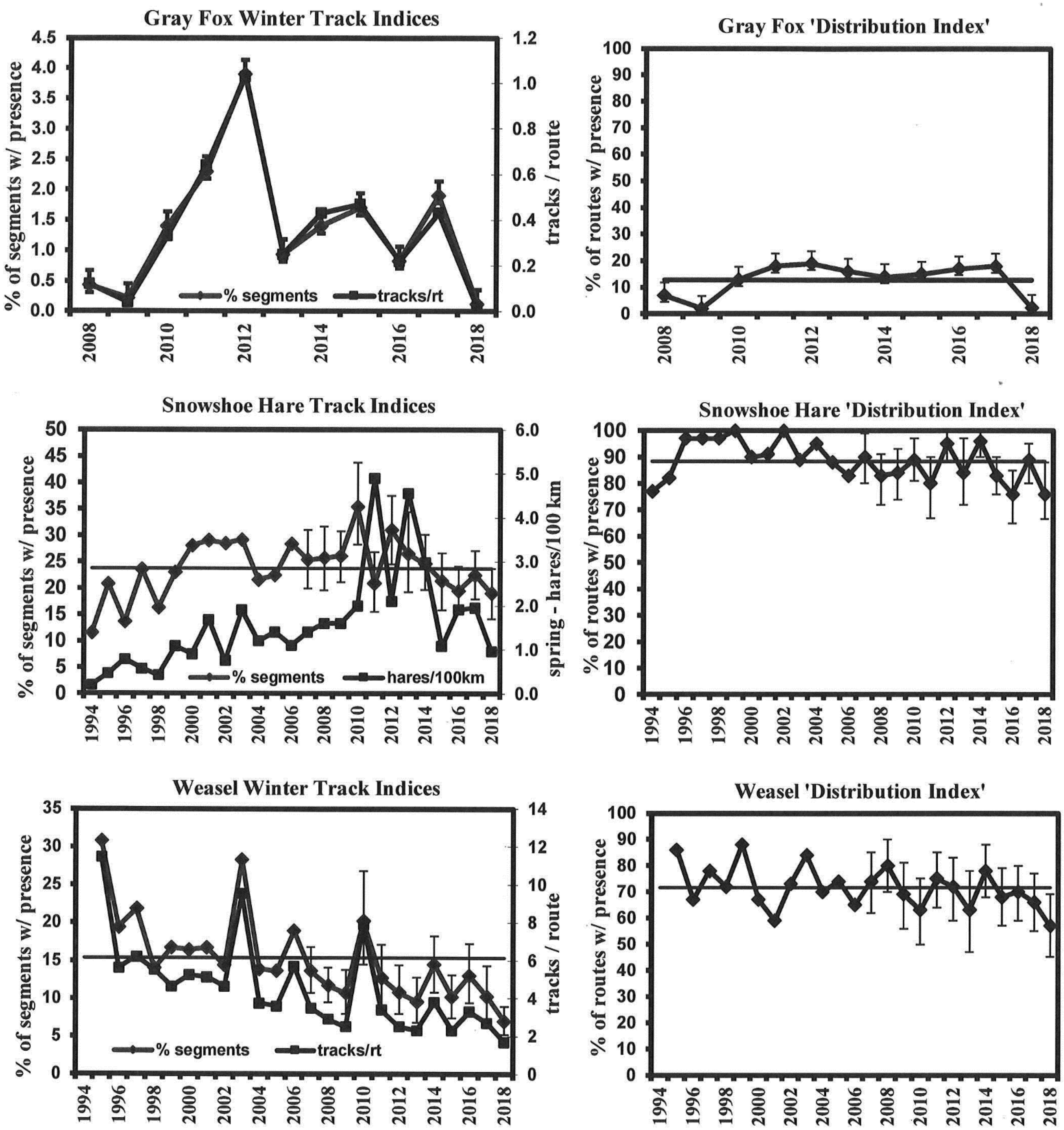

Figure 4 (continued). Winter track indices for selected species in Minnesota, 1994-2018.

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*current index + 1/3*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 Pop. Taken[2]	Carcasses Examined	% juveniles	% yearlings	% adults	Juv: Ad Female ratio	% Male juveniles	% Male yearlings	% Male adults	Overall % males	Mean Pelt Price[3]
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
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

[1]Includes DNR and Tribal harvests

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

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

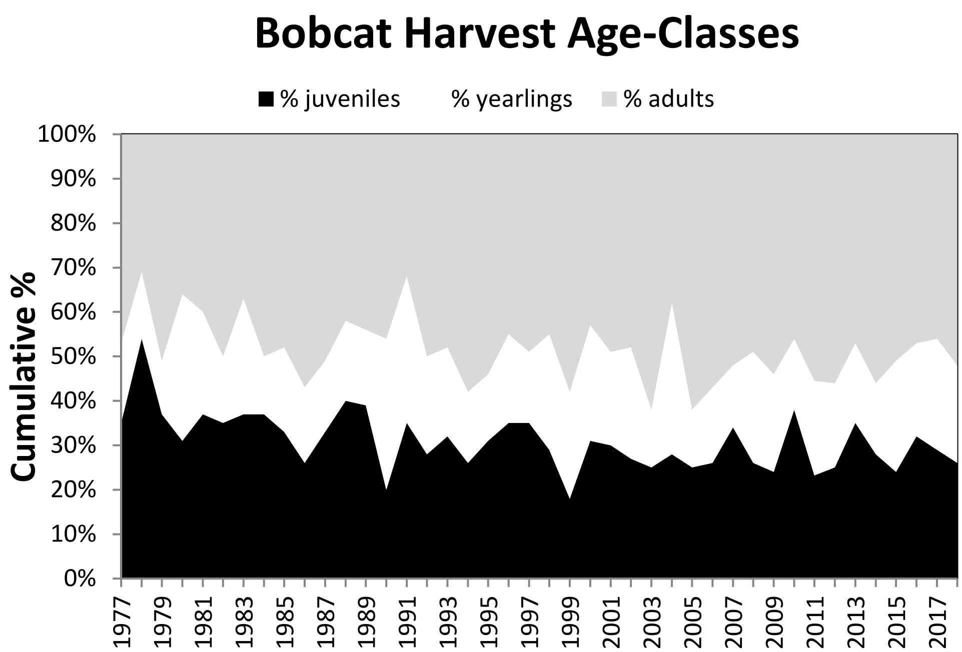


Figure 1. Age-class distribution of bobcats harvested in Minnesota, 1977-2018.

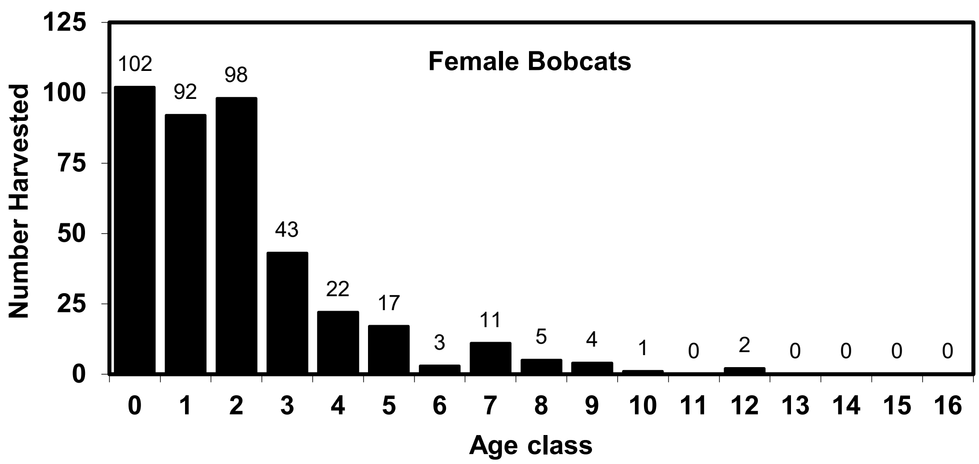


Figure 2. Age structure of female bobcats in the 2018 harvest.

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

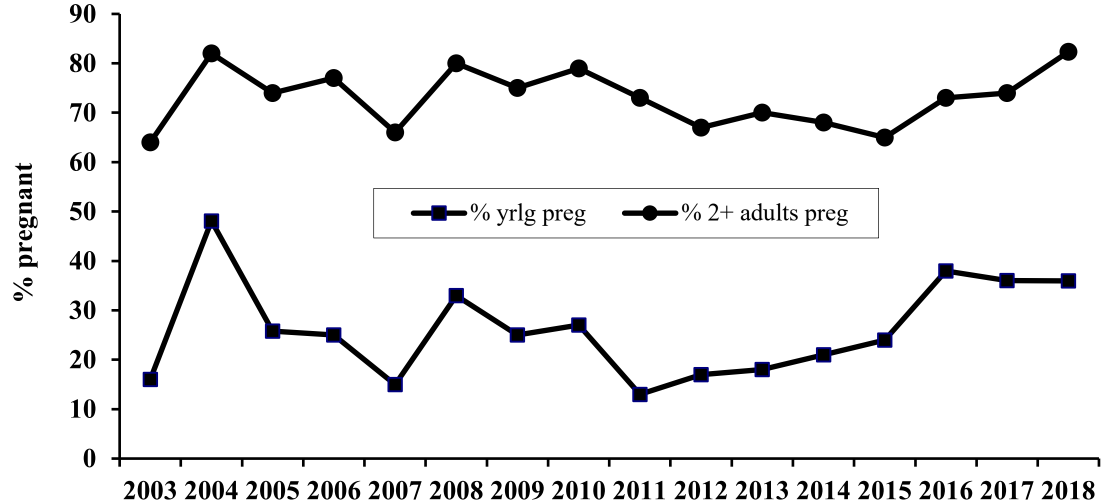


Figure 4. Pregnancy rates for yearling and adult bobcats in Minnesota, 2003-2018.

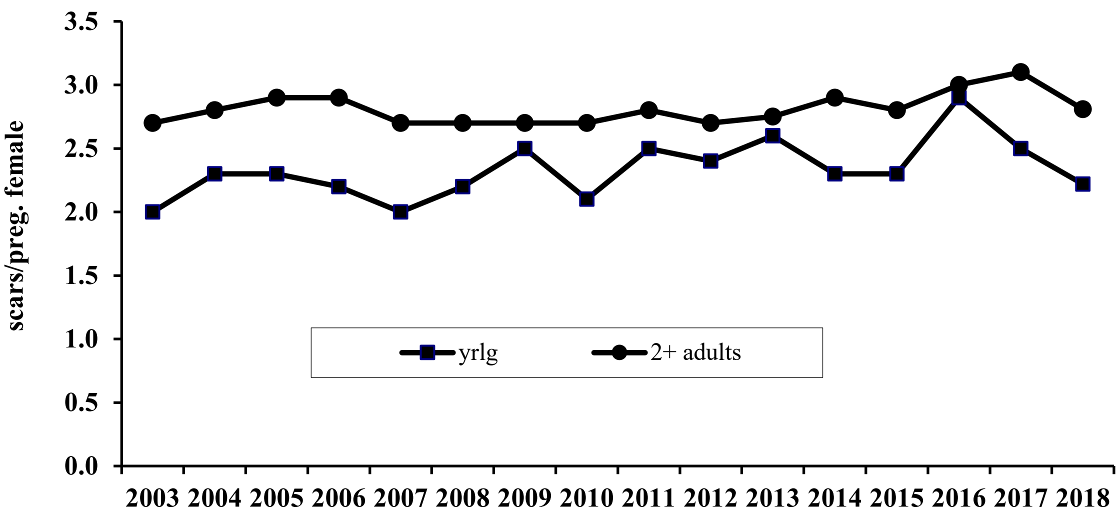


Figure 5. Litter size for parous yearling and adult bobcats in Minnesota, 2003-2018.

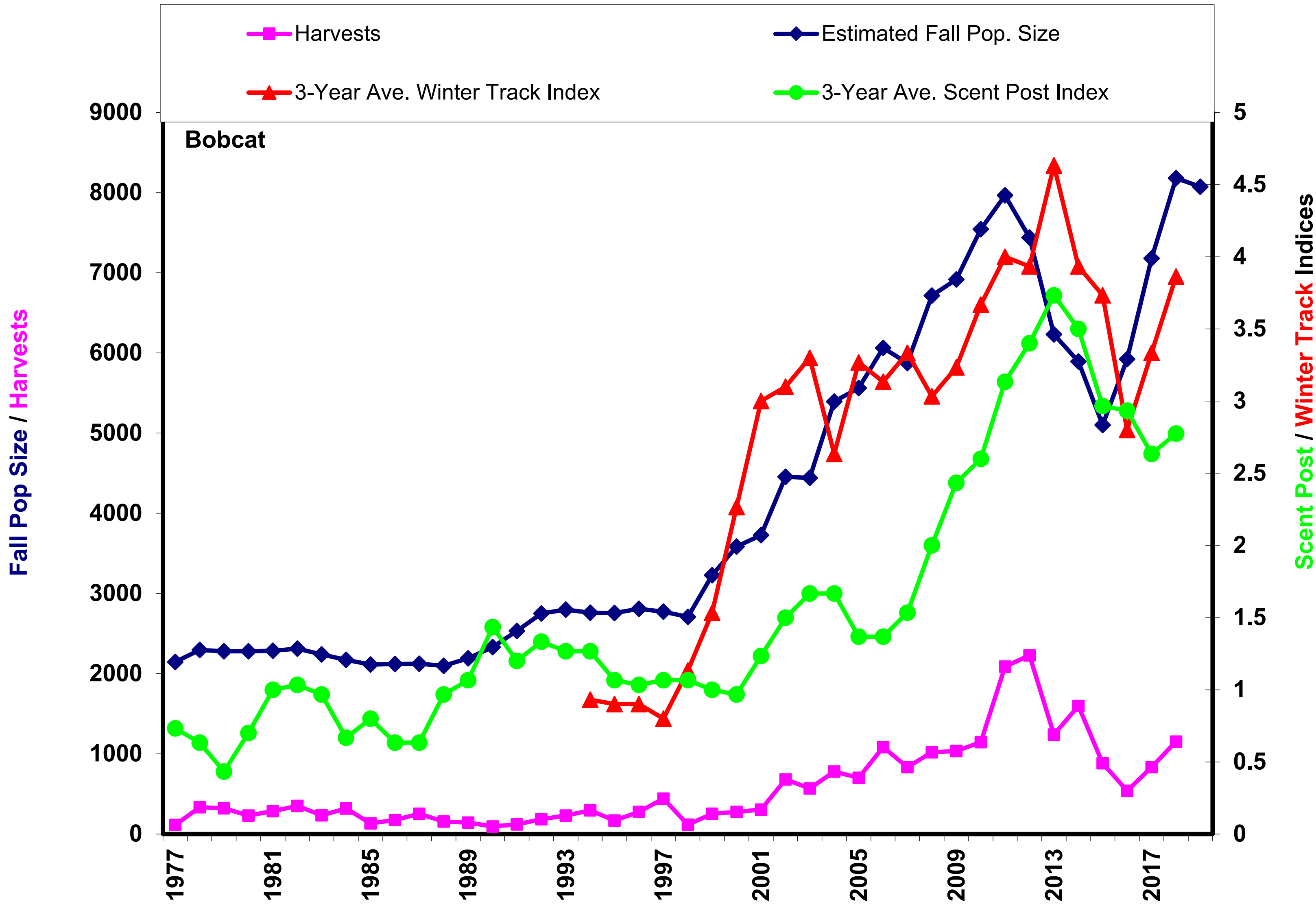


Figure 6. Bobcat population projections, harvests, and survey indices, 1977-2019. Harvests include an estimate of non-reported take.

Table 2. Fisher harvest data, 1989 to 2018.

Year	DNR harvest	Modeled Harvest[1]	% Autumn Pop. Harvested[2]	Carcasses examined	% juveniles	% yearlings	% adults	Juv: Ad. Female ratio	% male juveniles	% male yearlings	% male adults	% males overall	Pelt price Males[3]	Pelt price Females[3]
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
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.

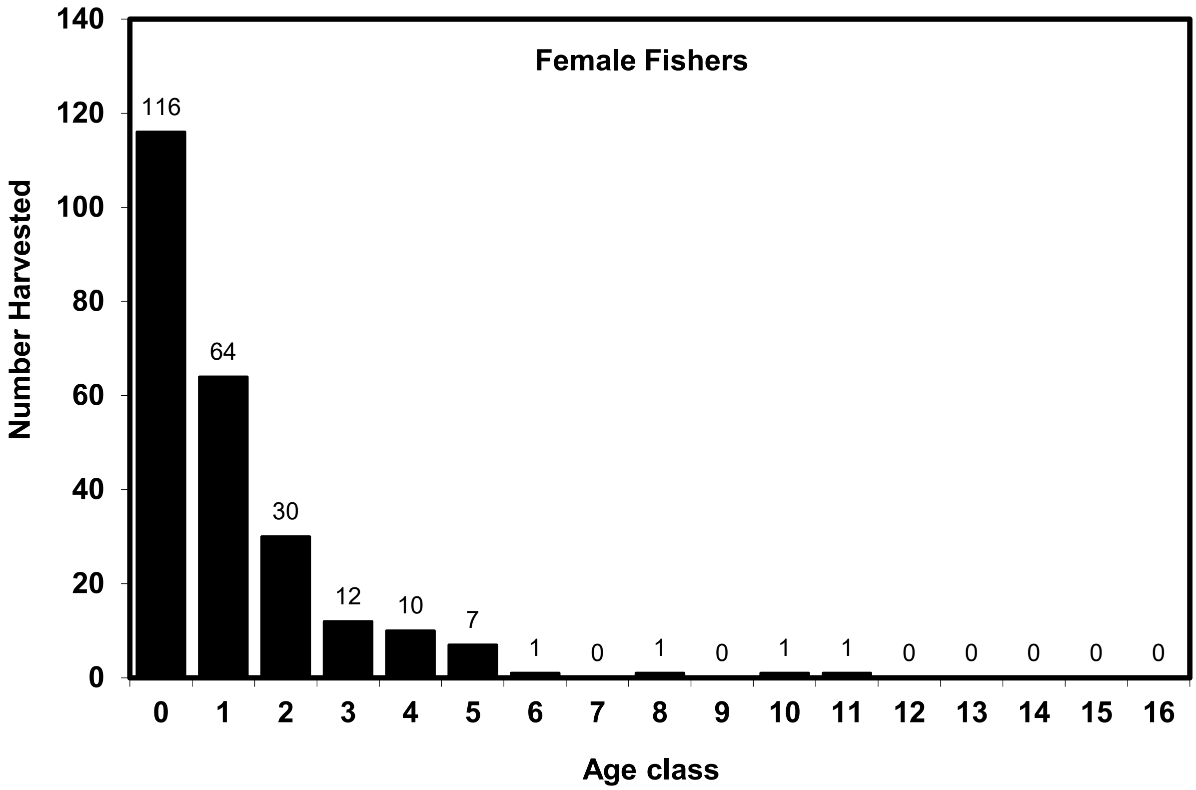


Figure 7. Age structure of female fishers in the 2018 harvest.

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

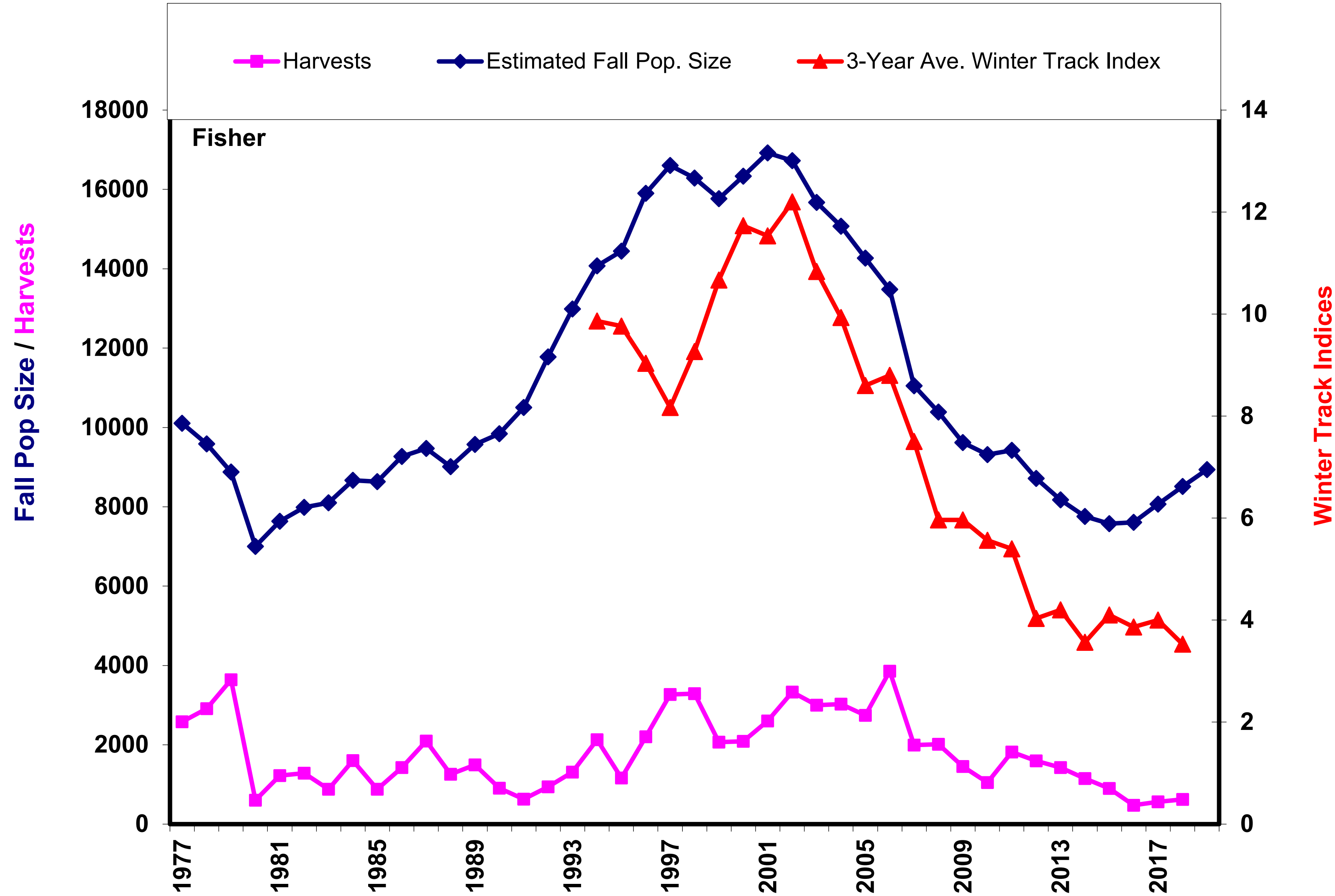


Figure 9. Fisher population projections, harvests, and survey indices, 1977-2019. Harvests include an estimate of non-reported take.

Table 3. Marten harvest data, 1989 to 2018.

Year	DNR harvest	Modeled Harvest[1]	% Autumn Pop. Harvested[2]	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]
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
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

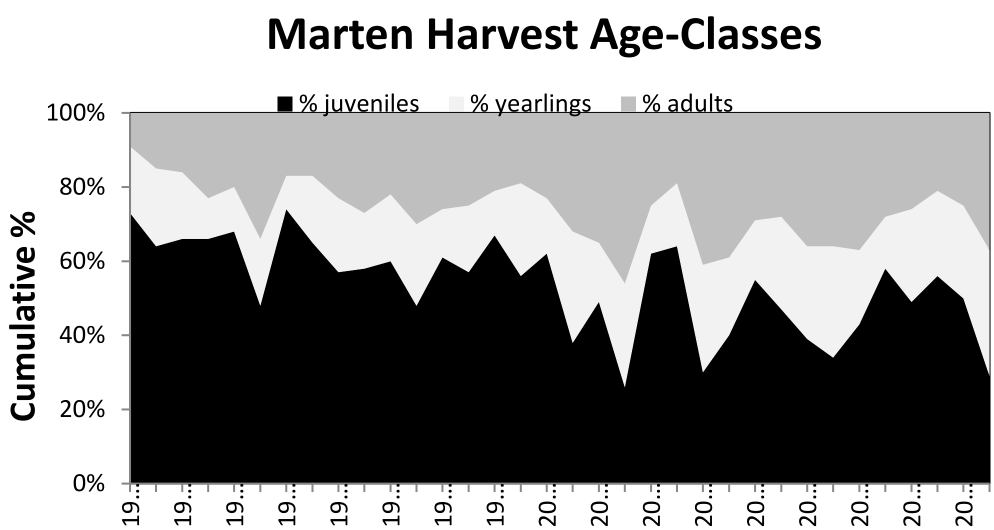


Figure 10. Age-class distribution of martens harvested in Minnesota, 1985 - 2018.

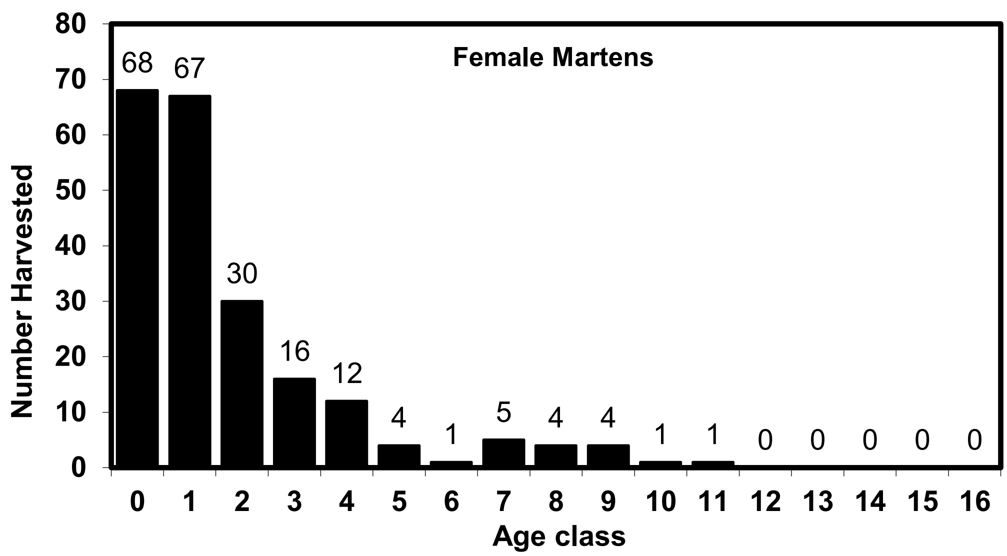


Figure 11. Age structure of female martens in the 2018 harvest.

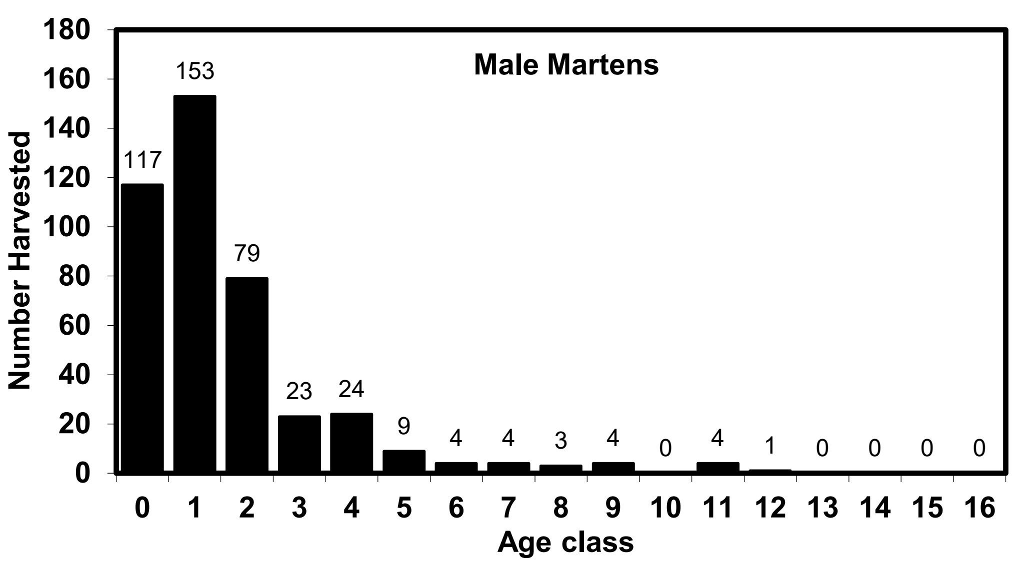


Figure 12. Age structure of male martens in the 2018 harvest.

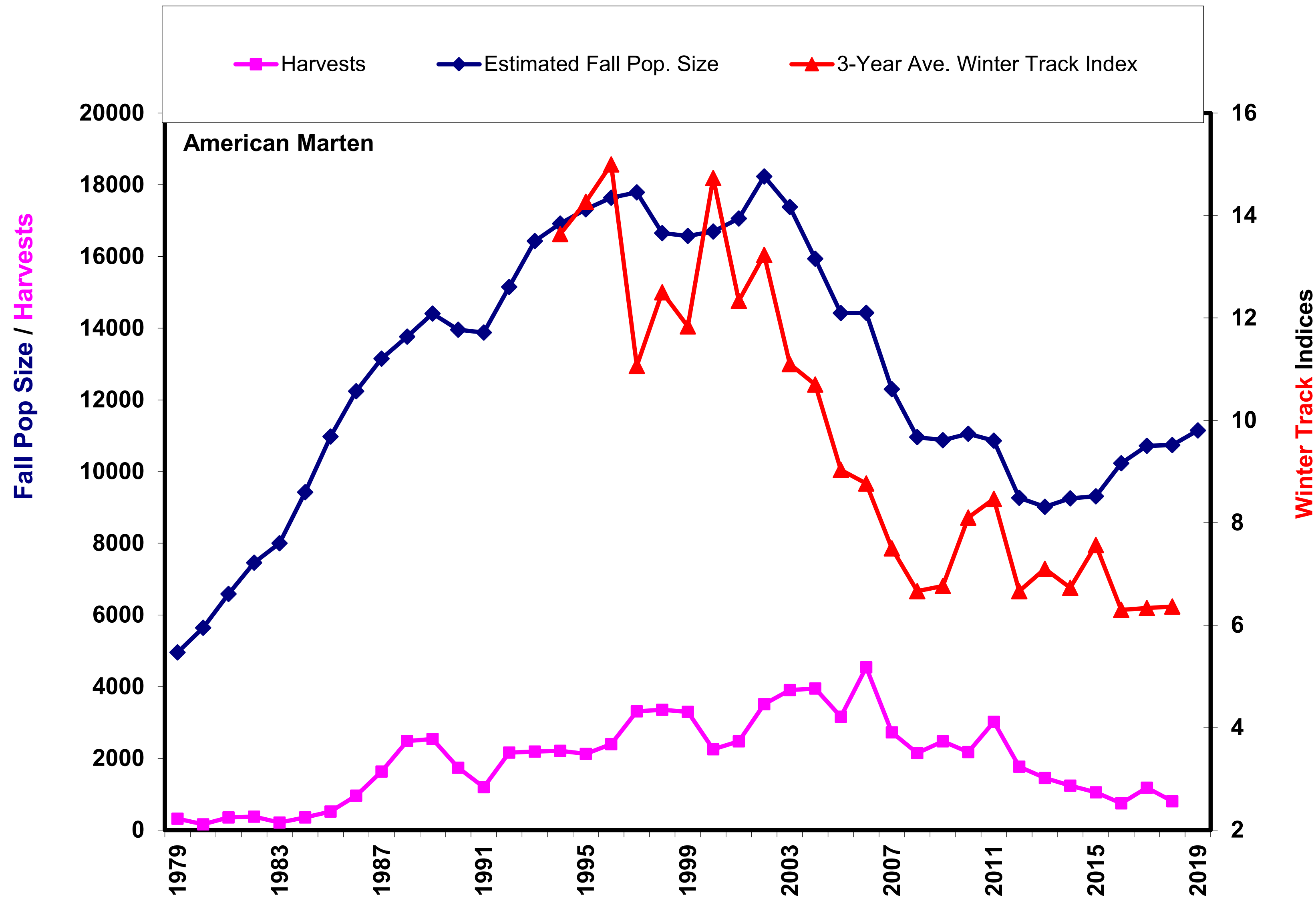


Figure 13. American marten population projections, harvests, and survey indices, 1979-2018. Harvests include an estimate of non-reported take.

Table 4. Otter harvest data[1], 1989 to 2018. Carcasses were only collected from 1980-86.

Year	DNR harvest	Modeled Harvest[1]	% Autumn Pop. Harvested[2]	Carcasses examined	% juveniles	% yearlings	% adults	Juv:ad. females	% Male juveniles	% Male yearlings	% Male adults	% Males overall	Pelt price Otter[3]	Pelt price Beaver[3]
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
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.

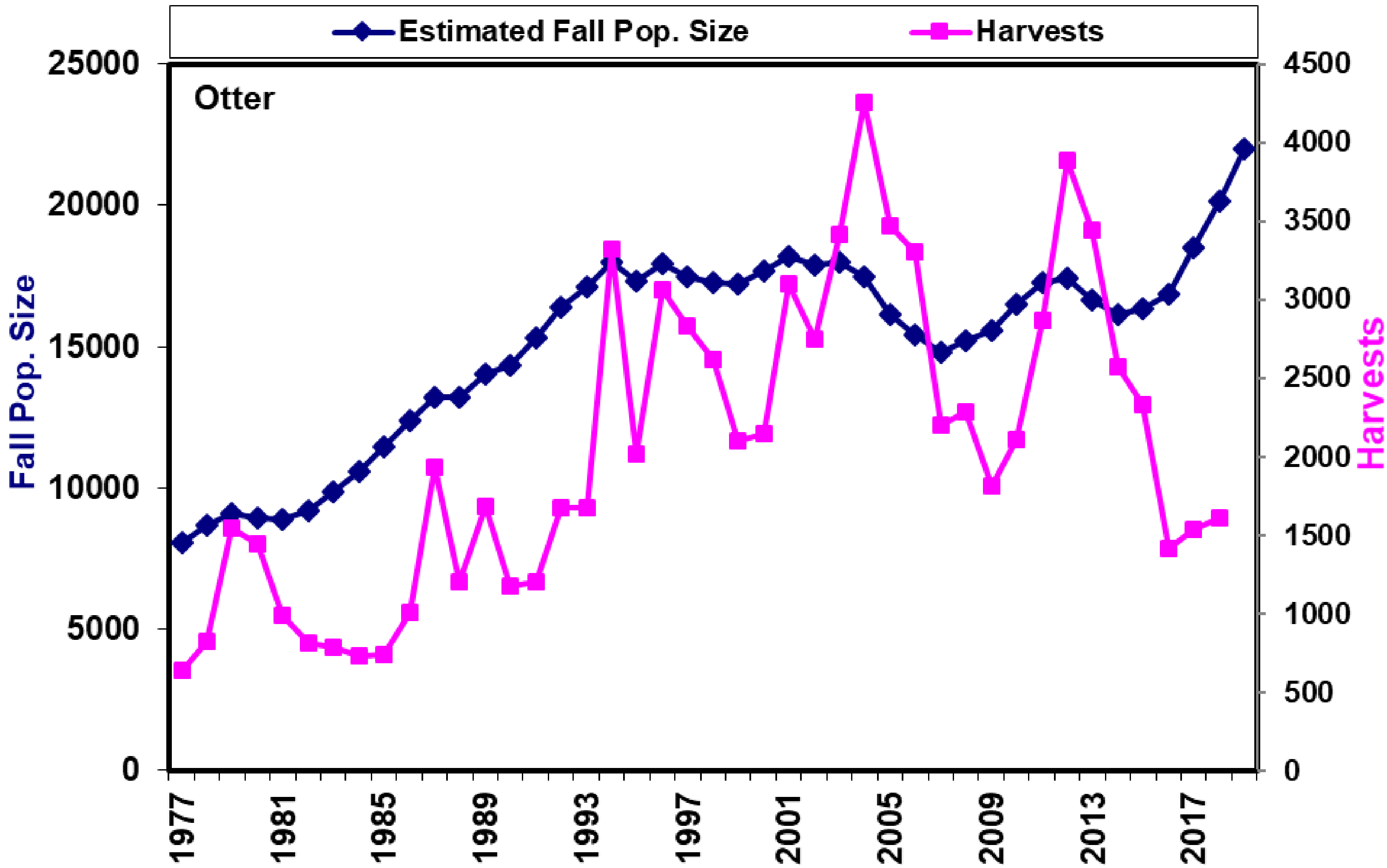


Figure 14. Otter population projections and harvests, 1977-2018. Harvests include an estimate of non-reported take.

STATUS OF MINNESOTA BLACK BEARS, 2018

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 (Figures 1 & 2).

This year another population trajectory was added, derived from a Bayesian model recently developed by Allen et al. (2018) for bear monitoring in Wisconsin. Besides the sex-ages of harvested bears, this model also includes reproductive and survival parameters.

From 1980 to 2000, the Allen matched the Downing model that included 23% non-harvest mortality. But in the last 10 years, the Allen model better matched the Downing model with 44% non-harvest mortality. However, whereas both models show a decline since the late 1990s, that decline is much less steep in the Allen model.

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 Downing model indicates the reduced hunting pressure has worked, enabling a population increase from 2014 to 2016 (although estimates for 2017 and 2018 are not obtainable with this model). The Allen model, in contrast, shows a continued decline until pre-hunt 2015, and then a leveling off (at 11–12,000 bears, excluding cubs) through 2018.

Of note, Downing population reconstruction assumes equal harvest pressure through time. As harvest pressure is diminished, and fewer bears are killed (as has been the trend since 2003), non-harvest mortality should comprise a greater proportion of total mortality. Therefore, it is possible that the Downing curve should be higher in recent years (which have lower harvest rates; see Fig. 3). That would make the disparity between the Allen and Downing trajectories greater during the most recent years.

Population trend: quota vs no-quota zones

Downing reconstruction indicated vastly different population trajectories for the quota and no-quota zones (Figure 2). Whereas the quota zone has shown a decline of about 50% of the population from 2000 to 2014, the no-quota zone remained relatively stable. With reduced quotas and lower harvests since then, the quota zone population increased almost 10% in 2 years (2014–2016), according to this model. Meanwhile, despite a surge in "overflow" hunters in the no-quota zone (Figure 4) prompted by the lower number of quota zone permits available, harvests in the no-quota zone have not increased, and the Downing model shows a recent population increase.

The Downing model does not produce population estimates for the most recent 2 years, so the effects of the high harvest in 2016 (in both quota and no-quota zones) is not yet reflected in the trajectories of this model.

Trends in harvest rates

The sex ratio of harvested bears varies by age in accordance with the relative vulnerability of the sexes (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, 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). 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 continued to use the monthly reporting system (and recorded zero when they had no complaints). In 2018, although the total number of complaints was the lowest since 2011, hotspots of nuisance activity were apparent: Little Falls, Park Rapids, Brainerd, Bemidji (all with 30–50 recorded complaints) and Cloquet (85 complaints). The number of nuisance bears killed equaled that of 2011, the lowest since recording began in 1982. In 2018 a list was distributed of 116 “area 88” hunters, who expressed interest in taking a nuisance bear in the quota area on a no-quota license. We have no records of any hunters doing so (it is unclear how many were authorized to do so).

Food abundance

The composite range-wide, all-season abundance of natural bear foods (fruits and nuts) in 2018 was the second highest on record and considerably higher than 2015–2017 (Table 3). Abundance of nearly all the summer foods was well above the long-term (34-year) average (Table 4), in all but the west-central region. On the other hand, fall foods were high in the west-central and east-central regions (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 the highest since 2002, because fall foods were so high in the west-central and east-central areas (but near normal in the northwest). Hazelnut production was average in the northwest, and above-average in most other areas (with patches of exceptional production). Dogwood production was generally above-average across the range. Oak production occurred in 3 bands, increasing from average to exceptional along a northwest to southeast gradient.

Predictions of harvest from food abundance

The 2018 statewide harvest was close, but slightly higher than expected (1766 actual vs. 1715 predicted), based on regression of harvest as a function of hunter numbers and the fall food productivity index (Figure 10). This regression is even stronger (and has accurately predicted previous harvests) when only the past 15 years are considered. For the quota zone, the actual harvest in 2018 was also close but higher (1272 actual vs. 1201 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, 2013–2018. Highlighted values show a change from the previous year. BMUs 26 and 44 were divided into 27/28 and 46/47, respectively, in 2016.

				2016			
BMU	**2013**	**2014**	**2015**	**Before BMU split [a]**	**After BMU split**	**2017**	**2018**
12	200	200	150	150	150	125	125
13	250	250	250	250	250	225	225
22	50	50	50	50	50	50	50
24	200	200	200	200	200	175	175
25	500	500	500	500	500	400	400
26	350	350	350	325			
27					250	225	225
28					75	60	60
31	550	550	550	550	550	500	500
41	150	150	150	125	125	125	125
44	450	450	450	450			
46					400	350	350
47					50	40	40
45	150	150	150	250	250	175	175
51	900	900	900	1000	1000	900	900
Total	**3750**	**3750**	**3700**	3850	**3850**	**3350**	**3350**

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 1998–2018, including number of nuisance bears killed and translocated, and bears killed in vehicular collisions.

	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017 [j]	2018 [j]
Number of personnel participating in survey [a]	71	52	60	54	50	39	34	42	46	46	37	51	40	34	56	63	64	61	55	86 (51,35)	78 (56,23)
Complaints examined on site	226	189	105	122	75	81	75	61	57	63	59	65	70	37	113	69	79	97	118	71 (22,49)	40 (21,19)
Complaints handled by phone [b]	743	987	618	660	550	424	507	451	426	380	452	535	514	396	722	623	570	840	780	644 (450,194)	438 (369,69)
Total complaints received	969	1176	723	782	625	505	582	512	483	443	511	600	584	433	835	692	649	937	898	715	478
• % Handled by phone	77%	84%	85%	84%	88%	84%	87%	88%	88%	86%	88%	89%	88%	91%	86%	90%	88%	90%	87%	90%	92%
Bears killed by:																					
• Private party or DNR	31	25	25	22	12	13	25	28	11	21	22	23	22	9	16	24	26	45	53	22 (4, 18)	9 [k] (4,5)
• Hunter before season [c]																					
– from nuisance survey	23	5	7	4	0	3	3	6	2	18	3	4	3	3	11	0	0	1	13	1	2
– from registration file	31	24	43	20	11	8	4	13	6	25	5	15	10	5	12	0	1	4	6	3	11 [m]
• Hunter during/after season [d]	3	0	1	1	0	0	0	1	0	0	0	0	0	0	0	1	0	1	1	1	0
• Hunter by Area 88 license [e]																				1	[m]
• Permittee [f]	11	7	2	6	4	6	1	5	4	5	1	3	5	0	0	1	0	3	0	0	1
Bears translocated	24	29	1	6	3	1	3	3	3	1	3	2	2	2	0	3	2	0	0	0	0
• % bears translocated [g]	11	15	1	5	4	1	4	5	5	2	5	3	3	5	0	4	3	0	0	0	0
Bears killed by cars [h]	61	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]

Table 8. (continued)

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 it was no longer possible to differentiate Wildlife Managers who participated month by month. Instead, 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.

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 Beginning in 2017, hunters could choose Area 88 in the quota lottery, and if drawn, could hunt for a nuisance bear, if authorized. In 2017, 11 hunters were authorized, but only 1 killed 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 4 bears have been killed by permittees since 2011.

g Percent of on-site investigations resulting in a bear being captured and translocated.

h Car kill data were reported on the monthly nuisance form for the first time 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. The number of reported car-kills in 2017 was the lowest since record-keeping began in 1981.

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). The 2 survey tools are not exactly the same, so data are presented separately for each in parenthesis (Wildlife Managers, COs). For consistency, only April–October data are included (in 2017 10 calls were received in other months).

[k] Lowest number of nuisance bears were killed in 2011 and 2018, since recording began in 1982.

[m] 9 of the 11 pre-season hunters in 2018 were in BMU 11. None were NQ hunters authorized to hunt in the quota zone (Area 88).

Table 3. Regional bear food indices[a] in Minnesota's bear range, 1984–2018. Shaded blocks indicate particularly low (<45; pink) or high (≥70; green) values.

Year	Survey Area					
	NW	NC	NE	WC	EC	Range wide
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	52.7	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	57.3	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]

[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 2018 compared to the previous 34-year mean (1984-2017) 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).

	NW		NC		NE		WC		EC		Rangewide	
FRUIT	34yr mean	2018 (*n* = 11[b])	34yr mean	2018 (*n* = 10)	34yr mean	2018 (*n* = 5)	34yr mean	2018 (*n* =7)	34yr mean	2018 (*n* = 11)	34yr mean	2018 (*n* = 36)
SUMMER												
Sarsaparilla	4.6	6.5	5.8	7.2	5.3	8.4	4.5	4.0	5.3	6.0	5.0	6.3
Pincherry	3.3	5.1	4.4	6.1	4.2	9.4	3.8	3.8	3.7	5.4	3.9	5.8
Chokecherry	5.7	9.4	5.4	8.8	4.5	9.8	5.4	8.3	4.6	6.8	5.2	8.9
Juneberry	5.2	6.6	4.9	6.7	5.0	8.8	3.7	4.3	3.9	8.4	4.5	6.8
Elderberry	1.6	0.5	3.0	3.2	3.6	4.5	3.1	2.5	3.3	3.6	2.9	2.7
Blueberry	5.1	7.5	5.4	9.9	4.9	8.7	3.6	5.0	3.8	5.2	4.4	7.4
Raspberry	6.4	8.1	7.9	9.0	7.9	12.4	7.1	6.1	7.0	9.2	7.1	8.7
Blackberry	1.3	1.5	2.4	1.0	1.2	1.0	3.6	4.0	4.4	6.9	2.9	3.7
FALL												
Wild Plum	2.2	4.2	1.8	6.1	1.1	6.3	2.7	5.6	2.4	3.0	2.2	4.7
HB Cranberry	5.3	5.3	4.5	4.0	3.9	6.5	3.8	2.6	3.8	4.6	4.2	4.3
Dogwood	6.2	7.0	5.7	5.1	4.9	6.3	5.9	7.7	5.9	6.6	5.7	6.8
Oak	3.5	3.1	3.1	3.3	1.9	4.3	5.8	9.0	5.6	8.7	4.4	6.4
Mountain Ash	1.6	1.5	2.5	4.4	2.5	7.3	1.7	1.3	2.3	4.1	2.6	3.7
Hazel	6.3	6.2	7.3	7.4	7.3	8.2	7.9	7.3	7.6	9.8	7.2	7.7
TOTAL	58.3	72.6	64.1	82.4	58.2	101.8	62.6	71.5	63.4	88.3	62.3	83.9

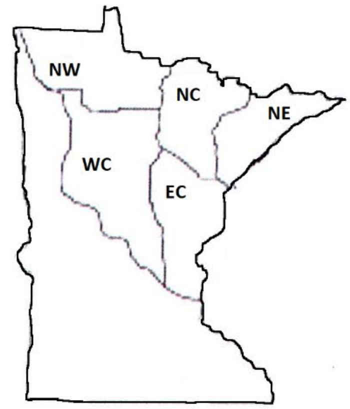
[a] Food abundance indices were calculated by multiplying species abundance ratings x fruit production ratings.

[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.

Table 5. Regional productivity index[a] for important fall foods (oak + hazel + dogwood) in Minnesota's bear range, 1984–2018. Shading indicates particularly low (□ 5.0; yellow) or high (≥8.0; tan) values.

Year	Survey Area					
	NW	NC	NE	WC	EC	Entire Range
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
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



[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.

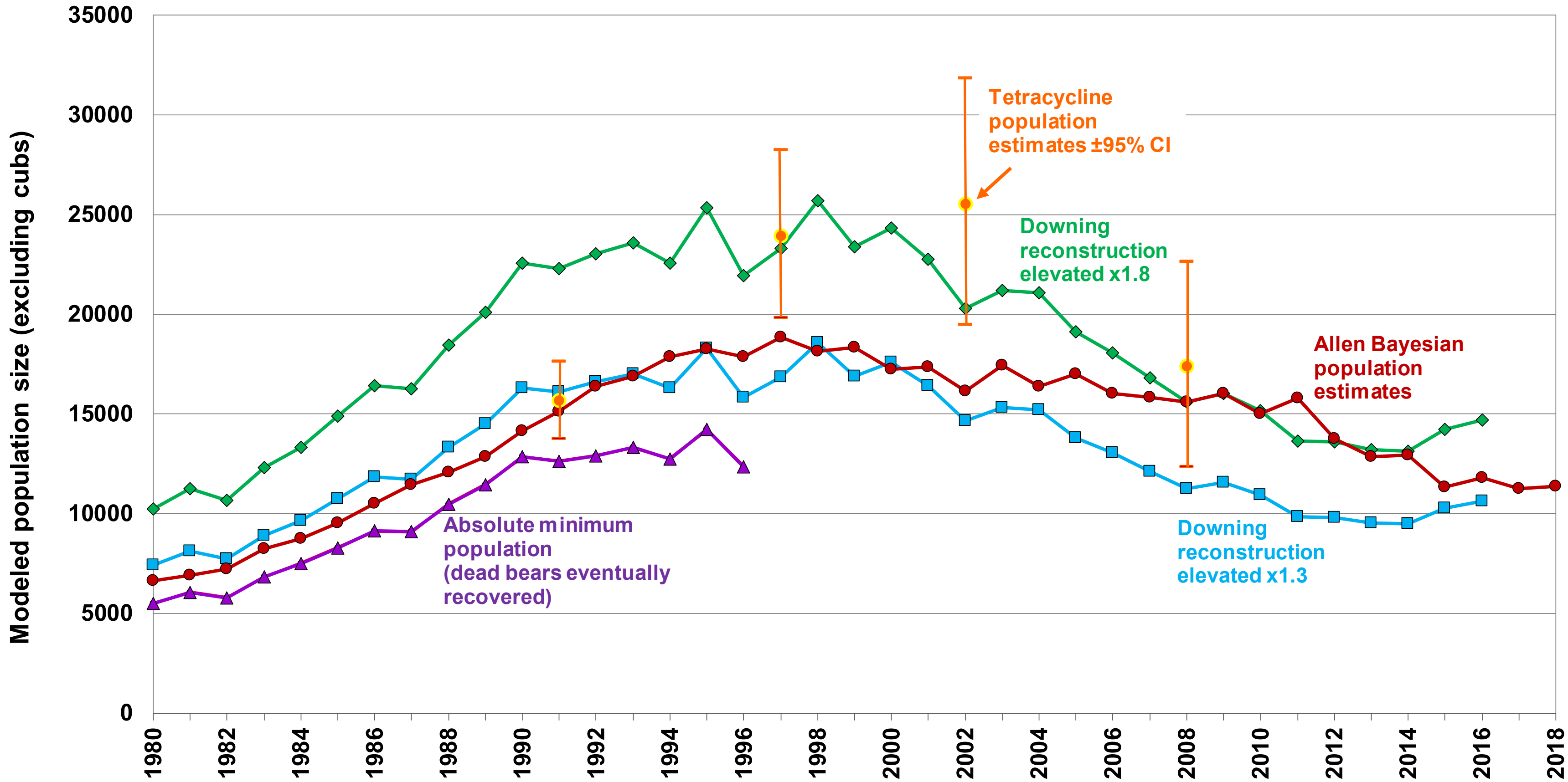


Figure 1. Statewide bear population trend (pre-hunt) derived from 2 population models: (1) Downing reconstruction, based solely on sex-specific harvest age structures, scaled (elevated to account for non-harvest mortality) to various degrees to attempt to match the tetracycline-based mark–recapture estimates (2 such curves shown here; estimates beyond 2016 are unreliable); and (2) a new Bayesian population model by Allen et al. (2018), which, besides harvest data includes estimates of reproduction and survival as well as an initial population size, and allows for estimates of the current year.

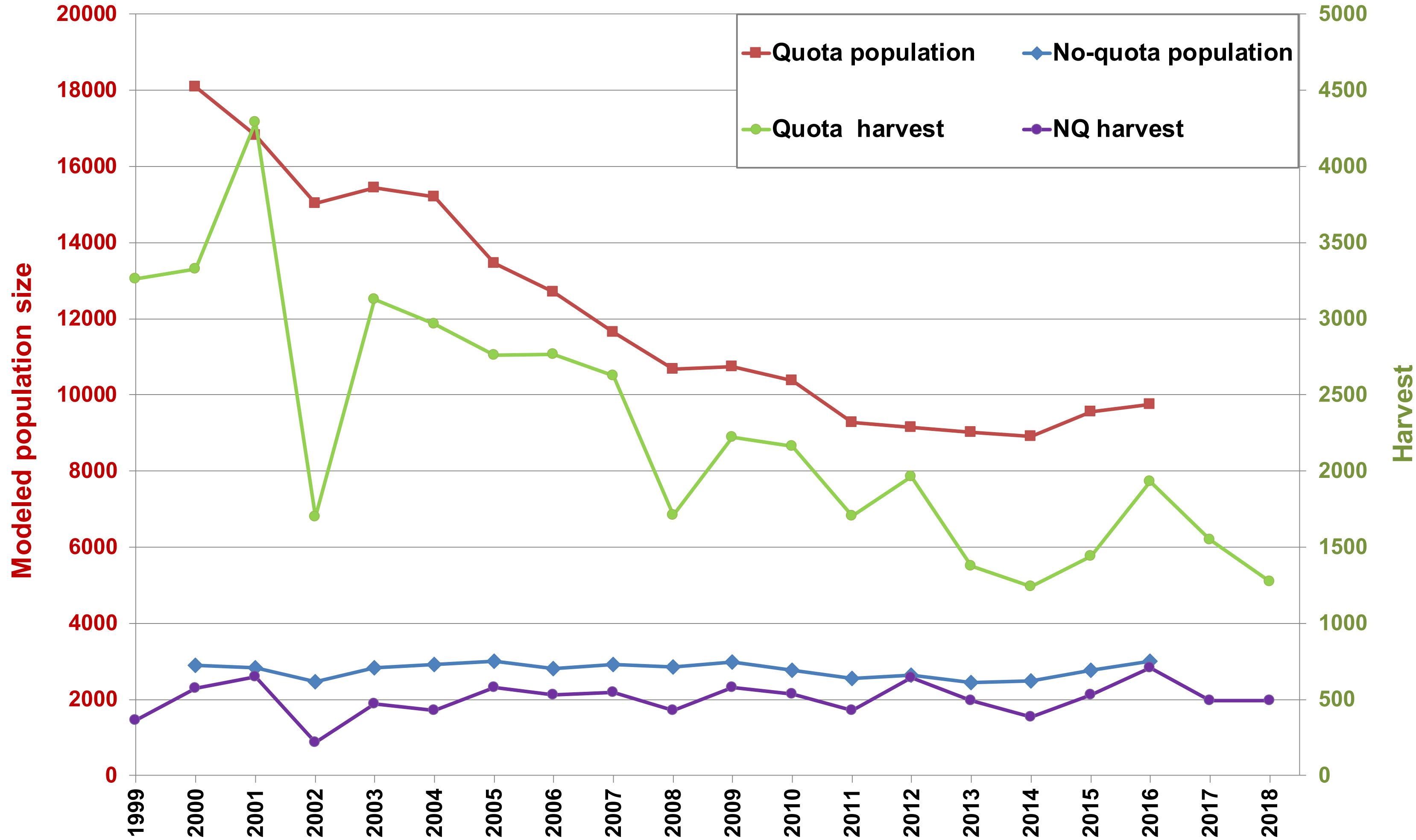


Figure 2. Population trends during 2000s derived from Downing reconstruction for quota and no-quota zones compared to respective harvests. Reconstruction-based estimates <2 years from the most recent harvest age data are unreliable (hence curves terminate in pre-hunt 2016). Population curves were scaled (elevated to account for non-harvest mortality) to fall between the 2 Downing curves in Figure 2 (i.e., the actual scale of the population estimates is not empirically-based).

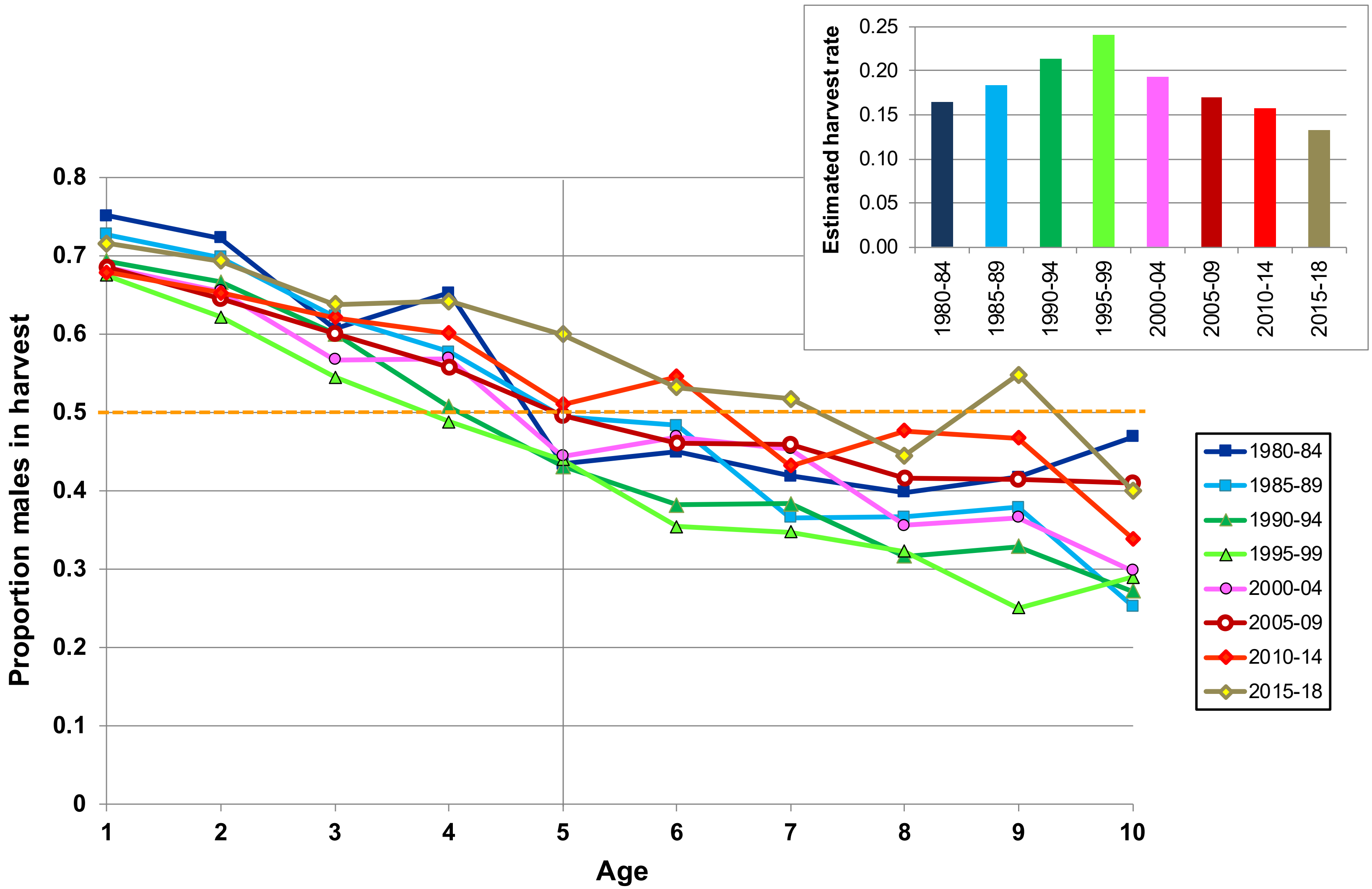


Figure 3. Trends in proportion of male bears in statewide harvest at each age, 1–10 years, grouped in 5-year time blocks, 1980–2018 (last interval = 4 years). Higher harvest rates result in steeper curves because males in the living population are reduced faster than females. Fitting a line to the data for each time block and predicting the age at which 50% of the harvest is male (dashed tan line) yields approximately the inverse of the harvest rate (derived rates are shown in inset). Flatter curves in recent years indicate lower harvest rates.

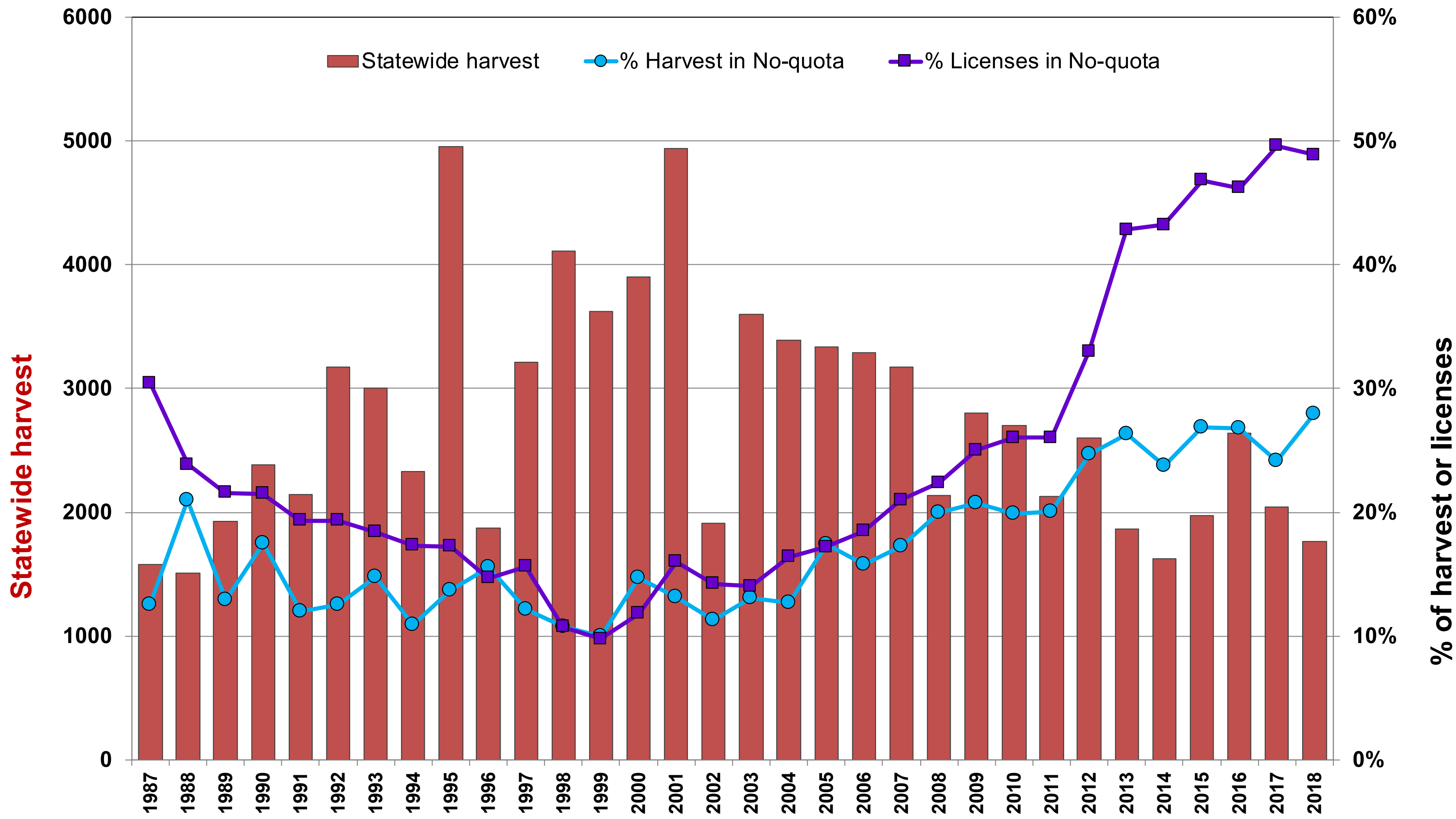


Figure 4. Trends in statewide bear harvest and proportions of harvest and licenses in the no-quota zones, 1987–2018.

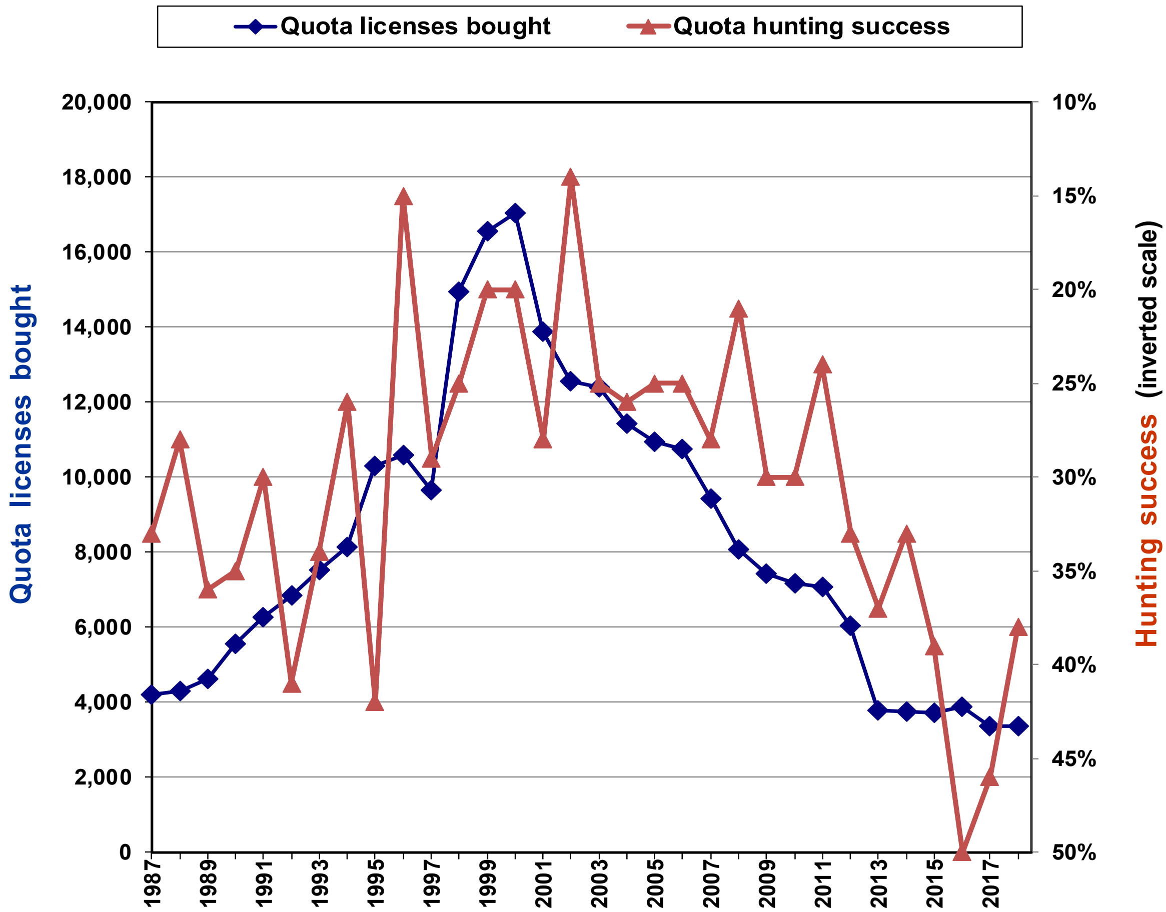


Figure 5. Relationship between licenses sold and hunting success (*note inverted scale*) in quota zone, 1987–2018 (quota and no-quota zones first partitioned in 1987). Number of licenses explains 47% of variation in hunting success during this period. Large variation in hunting success is also attributable to food conditions (e.g., during 2013–2018, when licenses were held relatively constant).

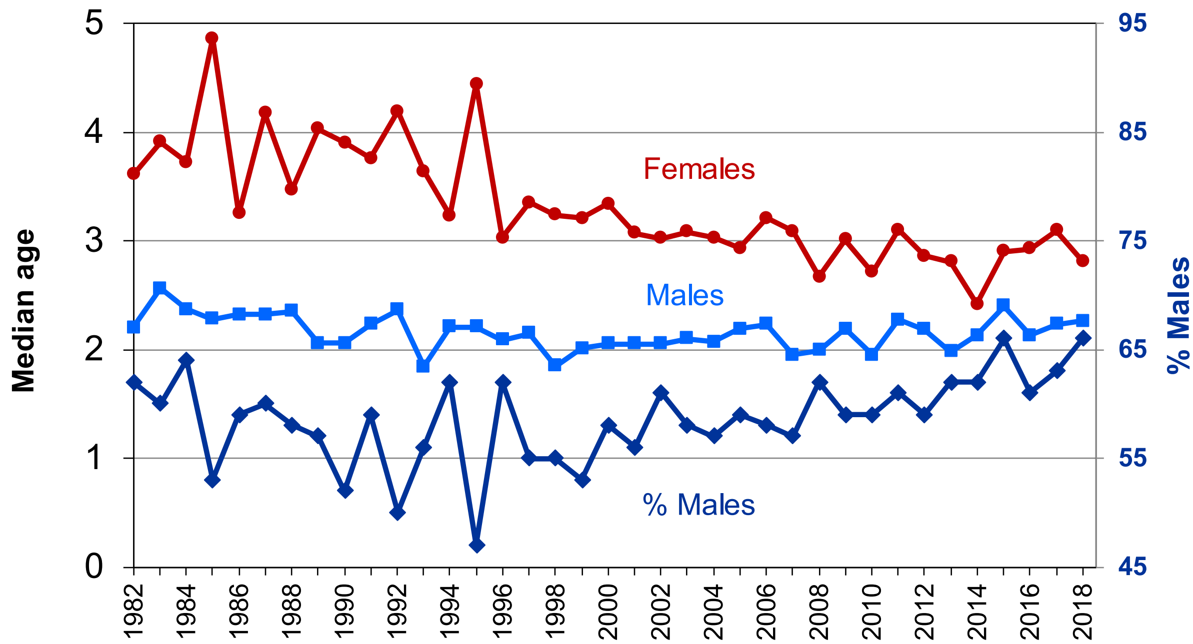


Figure 6. Statewide median ages (years) and sex ratio of harvested bears, 1982–2018.

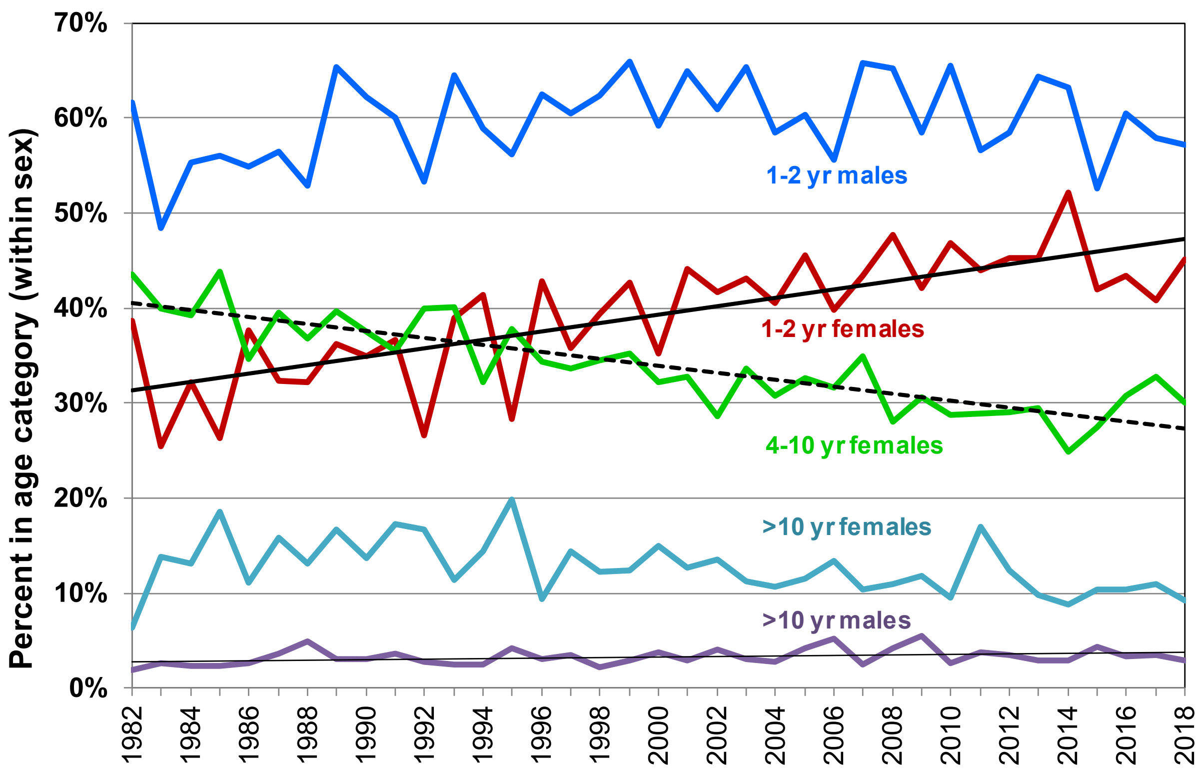


Figure 7. Statewide harvest structure: proportion of each sex in age category, 1982–2018. Trend lines shown are significant, but since 2008 the trend is level.

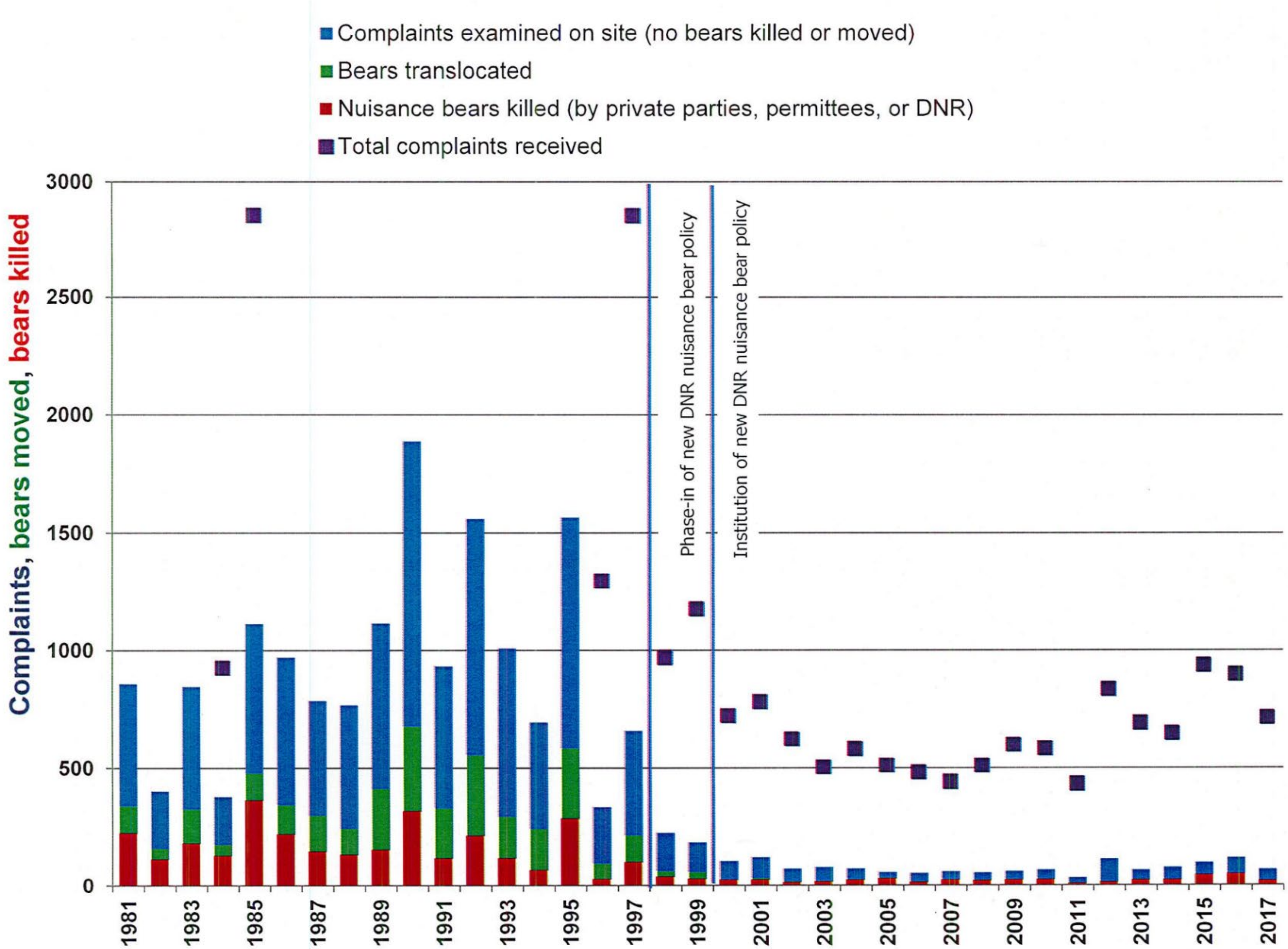


Figure. 8. Trends in nuisance bear complaints, and nuisance bears killed and moved, 1981–2018, showing dramatic effect of change in nuisance bear policy, and slight increasing trend over past decade, until 2018.

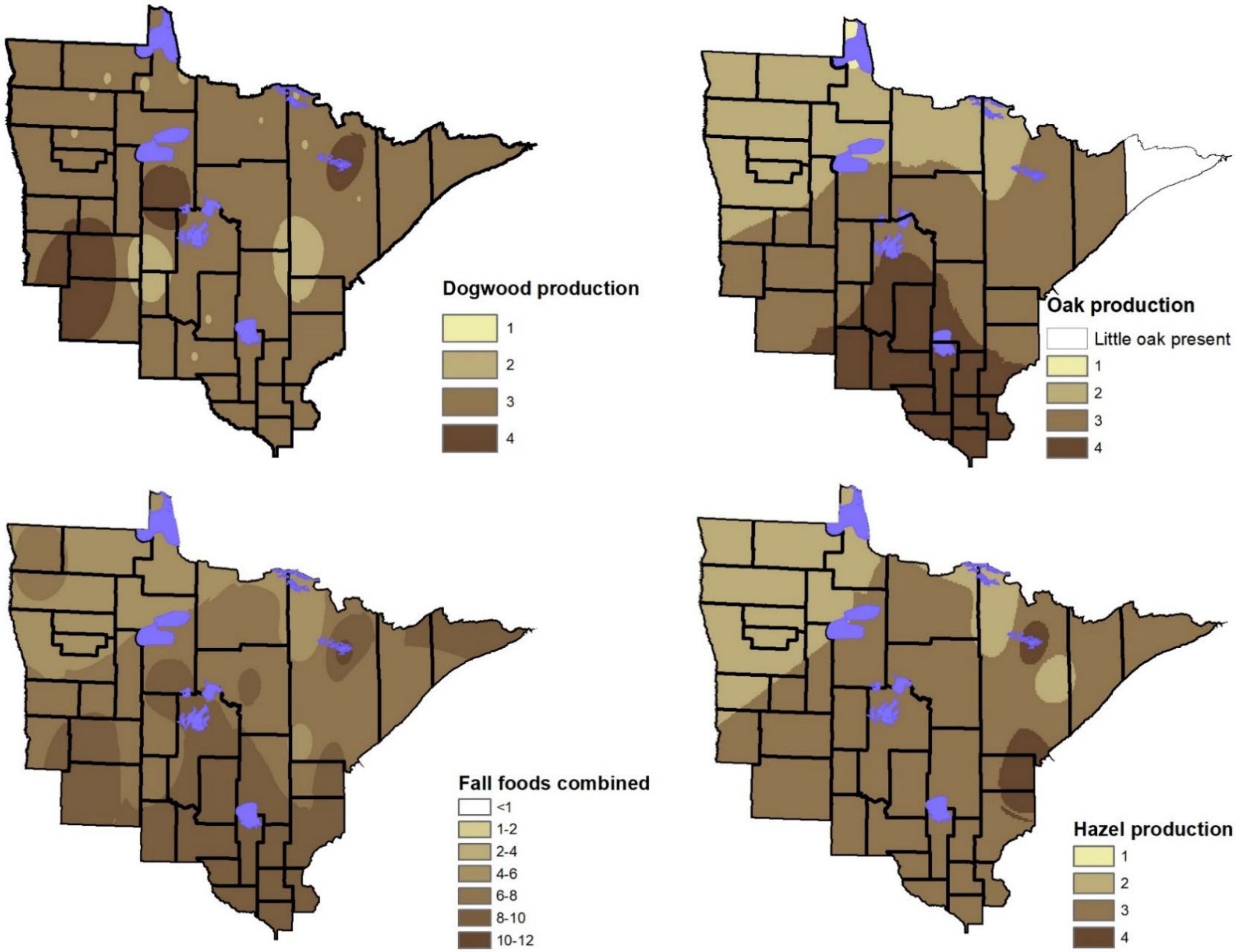


Figure 9. Production of fall bear foods (dogwood, oak, hazel) across Minnesota, 2018.

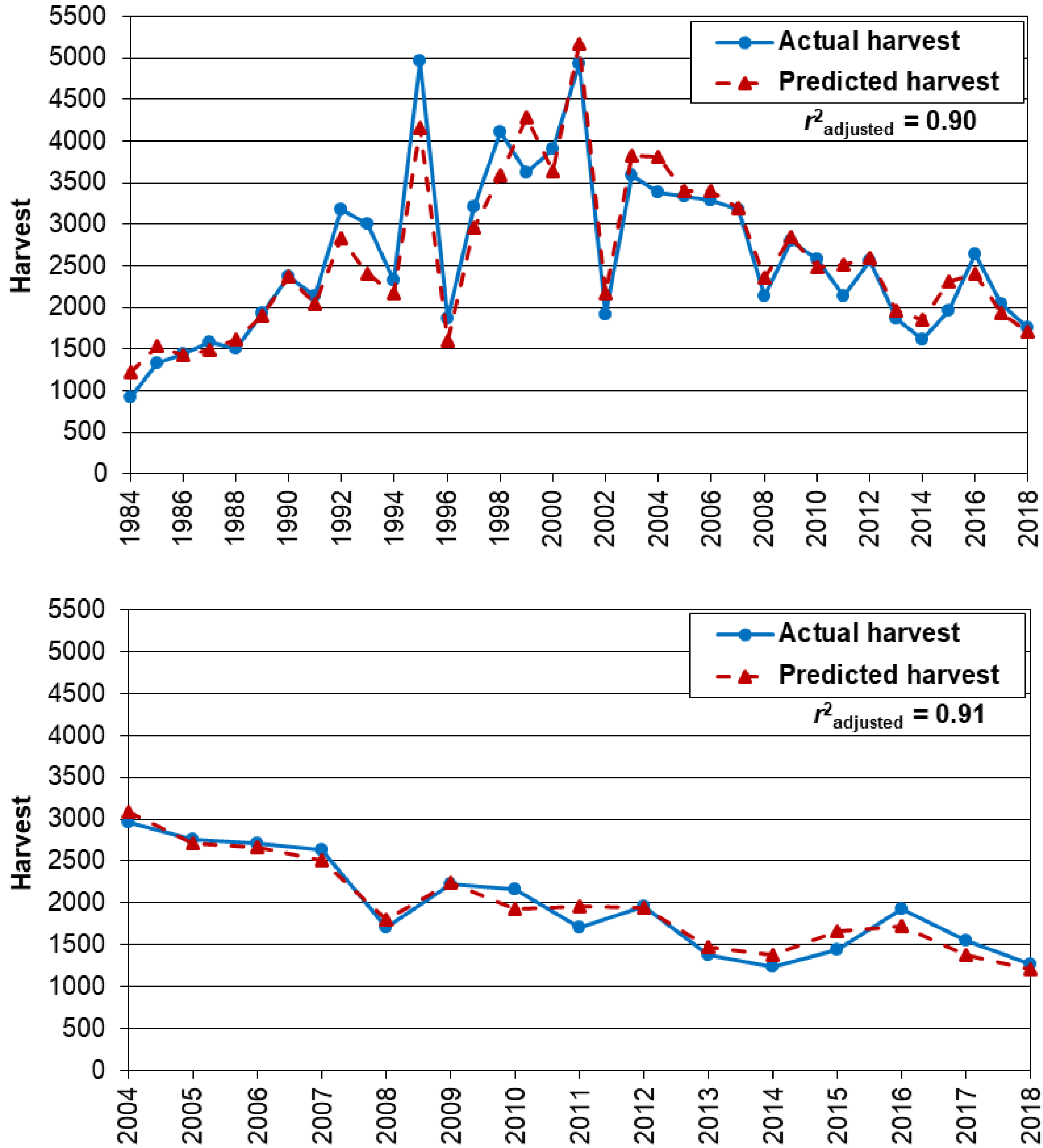


Figure 10. Number of bears harvested vs. number predicted to be harvested based on number of hunters and fall food production — top panel: statewide 1984–2018; bottom panel: quota zone only, most recent 15 years. Regression for the full dataset included an interaction term between food and hunters to better predict the drastic changes in harvest when fall foods were extremely high or low.

2019 MINNESOTA SPRING GROUSE SURVEYS

Charlotte Roy, Forest Wildlife Populations and Research Group Minnesota

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 agencies. In 2019, ruffed grouse surveys were conducted between 15 April and 17 May. Mean ruffed grouse drums per stop (dps) were 1.5 statewide (95% confidence interval = 1.3–1.7) which is similar to last year. High points in the population cycle occur on average every 10 years, and surveys indicate that the last peak occurred in 2017, with counts similar to the previous peak in 2009.

Sharp-tailed grouse surveys were conducted between 18 March and 5 May 2019, with 1,555 birds (males and birds of unknown sex) observed at 152 leks. The mean numbers of sharp-tailed grouse/lek were 7.2 (5.4–9.5) in the East Central (EC) survey region, 11.0 (9.7– 12.3) in the Northwest (NW) region, and 10.2 (9.1–11.4) statewide. Comparisons between leks observed in consecutive years (2018 and 2019) indicated similar numbers of birds/lek statewide ($t = 0.5$, $P = 0.65$) and in the NW region ($t = 0.05$, $P = 0.96$, $n = 101$). In the EC region, a 23% decrease in birds/lek observed in consecutive years occurred but was not statistically significant ($t = 1.7$, $P = 0.10$, $n = 31$), likely due to the smaller number of leks surveyed in the EC region and the impact that sample size has on the statistical power to detect differences between years.

INTRODUCTION

The ruffed grouse (*Bonasa umbellus*) is the most popular game bird in Minnesota, with an annual harvest averaging >500,000 birds (~150,000 to 1.4 million birds). Ruffed grouse hunter numbers have been as high as 92,000 during the last decade, although hunter numbers did not peak with the recent peak in grouse numbers, as they have traditionally. Sharp-tailed grouse (*Tympanuchus phasianellus*) are also popular among hunters, with an annual harvest of 6,000-22,000 birds since the early-1990s and 5,000–10,000 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 also used to correlate strongly with the fall harvest, but since the early 2000's, 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 that time, spring drumming counts have been conducted 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.

Sharp-tailed grouse were first surveyed in Minnesota 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. Surveys are conducted in openland portions of the state where sharp-tailed grouse persist, although they were formerly much more widely distributed in Minnesota at the early part of the 20th century.

METHODS

Ruffed Grouse

Surveys for ruffed grouse were conducted 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.

Survey observers were solicited from among state, federal, tribal, private, and student biologists. Each observer was provided a set of instructions and route location information. No formal survey training was conducted but all observers had a professional background in wildlife science, and most had previously participated in the survey. Participants were asked to conduct surveys at sunrise during peak drumming activity (in April or May) on days that had little wind and no precipitation. 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.

The number of drums heard per stop (dps) was used as the survey index value. I determined the mean dps for each route, for each of 4 survey regions (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 the region. Routes that traversed regional boundaries were included in the means for both regions. Because the number of routes within regions was not related to any proportional characteristic, I used the weighted mean of index values for the 4 Ecological Classification Sections (ECS) in the Northeast region and the 7 ECS sections in the state. The geographic area of the section was used as the weight for each section mean (i.e., Lake Agassiz, Aspen Parklands = 11,761 km^2, Northern Minnesota and Ontario Peatlands = 21,468 km^2, Northern Superior Uplands = 24,160 km^2, Northern Minnesota Drift and Lake Plains = 33,955 km^2, Western Superior Uplands = 14,158 km^2, Minnesota and Northeast Iowa Morainal (MIM) = 20,886 km^2, and Paleozoic Plateau (PP) = 5,212 km^2). The area used to weight drum index means for the MIM and PP sections was reduced to reflect the portion of these areas within ruffed grouse range (~50%) using subsection boundaries. A 95% confidence interval (CI) was calculated 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. Confidence interval boundaries were defined as the 2.5th and 97.5th percentiles of bootstrap frequency distributions.

Sharp-tailed Grouse

Wildlife staff and volunteers surveyed known sharp-tailed grouse lek locations in their work areas in the Northwest (NW) and East Central (EC) portions of the state (Figure 2). The NW region consisted of Lake Agassiz & Aspen Parklands, Northern Minnesota & Ontario Peatlands, and Red River Valley ECS sections. The EC region consisted of selected subsections of the Northern Minnesota Drift & Lake Plains, Western Superior Uplands, and Southern Superior Uplands sections. In the EC region, and in eastern portions of the NW region where sharp-tailed grouse occur at low densities, most known leks are surveyed each year.

Some leks may have been missed, but most managers in these regions believed that they included most of the leks in their work area, with the exception of Aitkin and Tower work areas where workloads do not permit exhaustive surveys. In the western part of the NW region, sharp-tailed grouse occur at higher densities, and thus surveying all leks is not feasible. Therefore, in the western portion of the NW region (e.g., Roseau, Thief River Falls), managers conduct surveys along 20-25 mile (32-40 km) routes. Given the uncertainty in the proportion of leks missed, especially those occurring outside traditional areas, the survey may not necessarily reflect sharp-tailed grouse numbers in larger areas such as counties or regions.

Each cooperator was provided with instructions and asked to conduct surveys on $\geq$1 day in an attempt to obtain a maximum count of male sharp-tailed grouse attendance at each lek. Observers were asked to conduct surveys within 2.5 hours of sunrise under clear skies and during low winds (<16 km/hr, or 10 mph) when lek attendance and ability to detect leks were expected to be greatest. Data recorded during each lek visit included the number of males, females, and birds of unknown sex. Observed lek size can vary as a function of population changes, lek numbers, and the timing, effort, and conditions of surveys, so it is important to consider all these factors when collecting data.

The number of sharp-tailed grouse per dancing ground was used as the index value and was averaged for the NW region, the EC region, and statewide, using known males and birds of unknown sex. Observations of just 1 grouse were not included in the index. Data from former survey years were available for comparison, however, survey effort and success varied among years rendering comparisons of the full survey among years invalid. Therefore, to make valid comparisons between 2 consecutive years, only counts of birds from dancing grounds that were surveyed during both years were considered. Paired t-tests were used to test the significance of comparisons among years. Confidence intervals (95%) were calculated using 10,000 bootstrap samples of lek counts for each region and statewide.

RESULTS & DISCUSSION

Ruffed Grouse

Observers from 14 cooperating organizations surveyed 131 routes between 15 April and 17 May 2019. Most routes (97%) were surveyed between 15 April and 15 May, with a median survey date of May 3, which is similar to the last 2 years (May 3) and the median survey date for the most recent 10 years. Excellent (68%), Good (29%), and Fair (3%) survey conditions were reported for 121 routes reporting conditions.

Statewide counts of ruffed grouse drums averaged 1.5 dps (95% confidence interval = 1.3–1.7 dps) during 2019 (Figure 3). Drum counts were 1.6 (1.3–1.9) dps in the Northeast (n = 103 routes), 2.1 (1.2–3.0) dps in the Northwest ($n = 5$), 0.8 (0.5–1.4) dps in the Central Hardwoods ($n = 15$), and 0.7 (0.4–1.1) dps in the Southeast ($n = 8$) regions (Figure 4a-d).

Statewide drum counts were similar to last year. Surveys indicate the most recent peak 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).

Sharp-tailed Grouse

A total of 1,555 male sharp-tailed grouse and grouse of unknown sex were counted at 152 leks (Table 1) during 18 March to 5 May 2019. The statewide index value of 10.2 (9.1–11.4) grouse/lek was centrally located among values observed since 1980 (Figure 5). In the EC survey region, 216 grouse were counted on 30 leks, and 1,339 grouse were counted on 122 leks in the NW survey region. The grouse/lek index was similar statewide and in both survey regions compared to 2018 (Table 1). Leks with ≥2 grouse were observed an average of 1.7 times. Counts at leks observed during both 2018 and 2019 were similar statewide ($t = 0.5$, $P = 0.65$) and in the NW region ($t = 0.5$, $P = 0.96$). However, a 23% decline in the EC region was not significant ($t = 1.7$, $P = 0.10$; Table 2), likely because fewer leks were surveyed in that region, which limits statistical power to detect differences statistically (Figure 6). Furthermore, a loss of small leks would tend to maintain or increase the average lek size, whereas it would cause comparisons of leks surveyed in successive years to decline.

Sharp-tailed grouse population index values peaked with those for ruffed grouse in 2009 and appear to have troughed with them in 2013, but sharp-tailed grouse peaks can follow those of ruffed grouse by as much as 2 years. This year, ruffed grouse and sharp-tailed grouse populations both remained similar to last year.

ACKNOWLEDGMENTS

The ruffed grouse survey was accomplished this year through the combined efforts of staff and volunteers at Chippewa and Superior National Forests (USDA Forest Service); Fond du Lac, Leech Lake, Red Lake, and White Earth Reservations; 1854 Treaty Authority; Blandin Paper; Vermilion Community College; Beltrami County and Cass County Land Departments; and DNR staff at Aitkin, Baudette, Bemidji, Brainerd, Carlos Avery Wildlife Management Area (WMA), Cloquet, Crookston, Detroit Lakes, Fergus Falls, Grand Rapids, International Falls, Karlstad, Little Falls, Mille Lacs WMA, Park Rapids, Red Lake WMA, Rochester, Roseau River WMA, Sauk Rapids, Thief Lake WMA, Thief River Falls, Tower, Two Harbors, Whitewater WMA, and Winona work areas. I would like to thank DNR staff and volunteers at Aitkin, Baudette, Bemidji, Cloquet, Crookston, Karlstad, International Falls, Tower, Thief River Falls, and Thief Lake work areas, and staff and volunteers at Red Lake and Roseau River WMAs for participating in sharp-tailed grouse surveys. Pam Coy, Alex Elliott, Joe Rohm, and Ben Bullard also helped with lek surveys this year. Laura Gilbert helped enter ruffed grouse data. 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. I would also like to thank Mike Larson 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.

	Statewide			Northwest[a]			East Central[a]		
Year	Mean	95% CI[b]	*n*[c]	Mean	95% CI[b]	*n*[c]	Mean	95%CI[b]	*n*[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

[a] Survey regions; see Figure 1.

[b] 95% CI = 95% confidence interval

[c] *n* = number of leks in the sample.

[d]One lek was located just south of the NW region in Clearwater County.

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

	Statewide			Northwest [a]			East Central [a]		
Comparison[b]	Mean	95% CI[c]	*n*[d]	Mean	95% CI[c]	*n*[d]	Mean	95%CI[c]	*n*[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	99.0	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

[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.

[e]One lek was located just south of the NW region in Clearwater County.

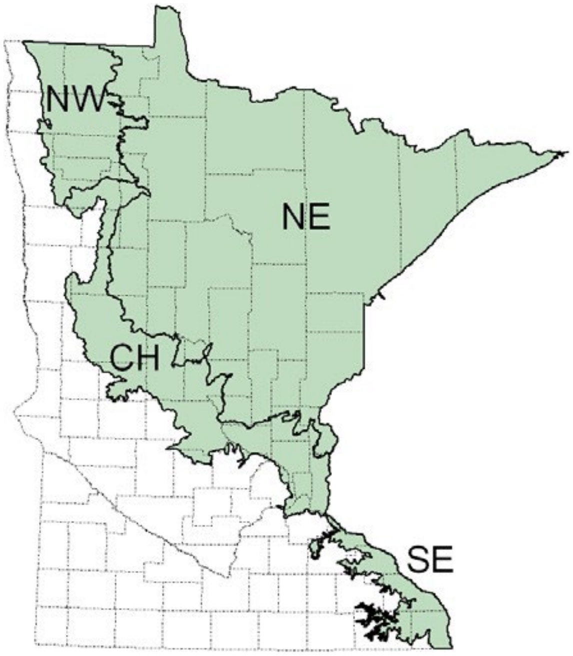


Figure 1. Survey regions for **ruffed grouse** in Minnesota. Northwest (NW), Northeast (NE), Central Hardwoods (CH), and Southeast (SE) survey regions are depicted relative to county boundaries (dashed lines) and influenced by the Ecological Classification System.

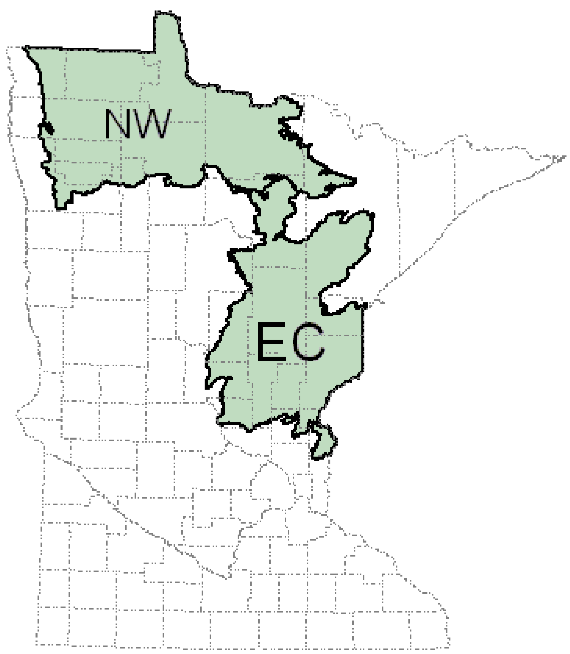


Figure 2. Survey regions for **sharp-tailed grouse** in Minnesota. Northwest (NW) and East Central (EC) survey regions are depicted relative to county boundaries (dashed lines) and influenced by Ecological Classification System Subsections boundaries.

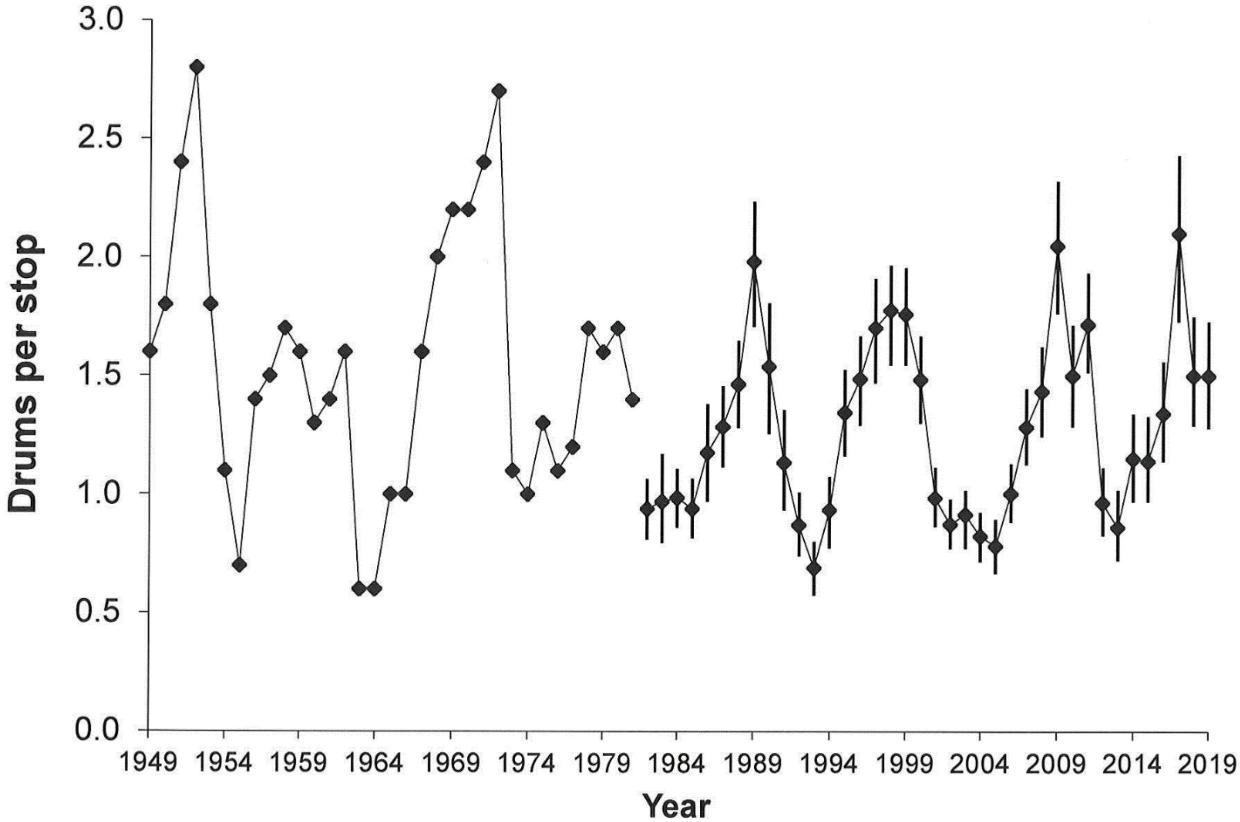
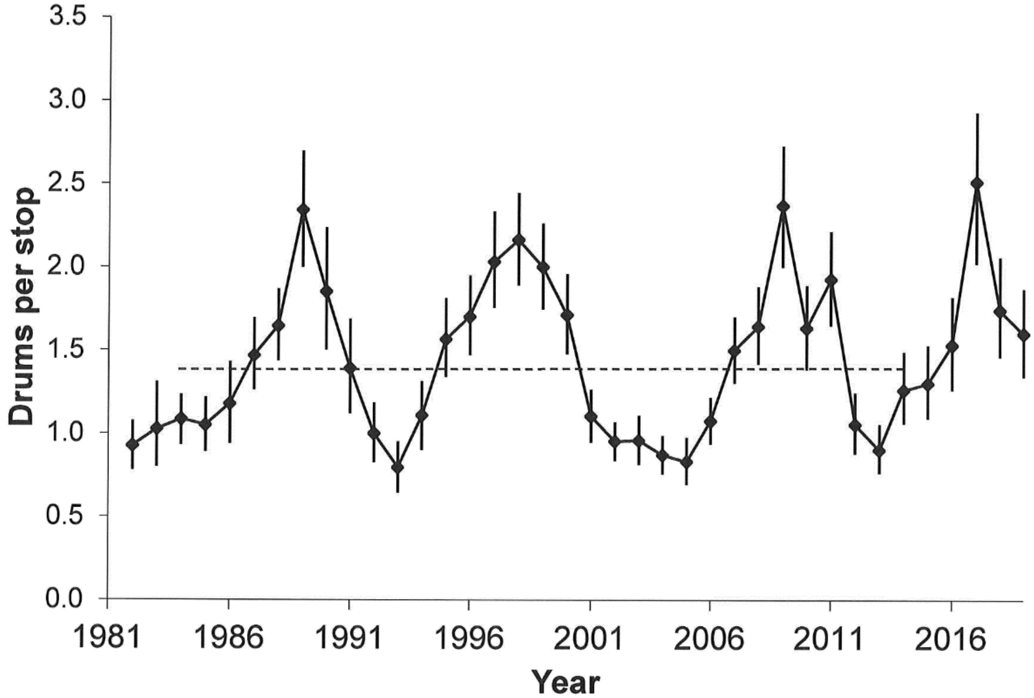
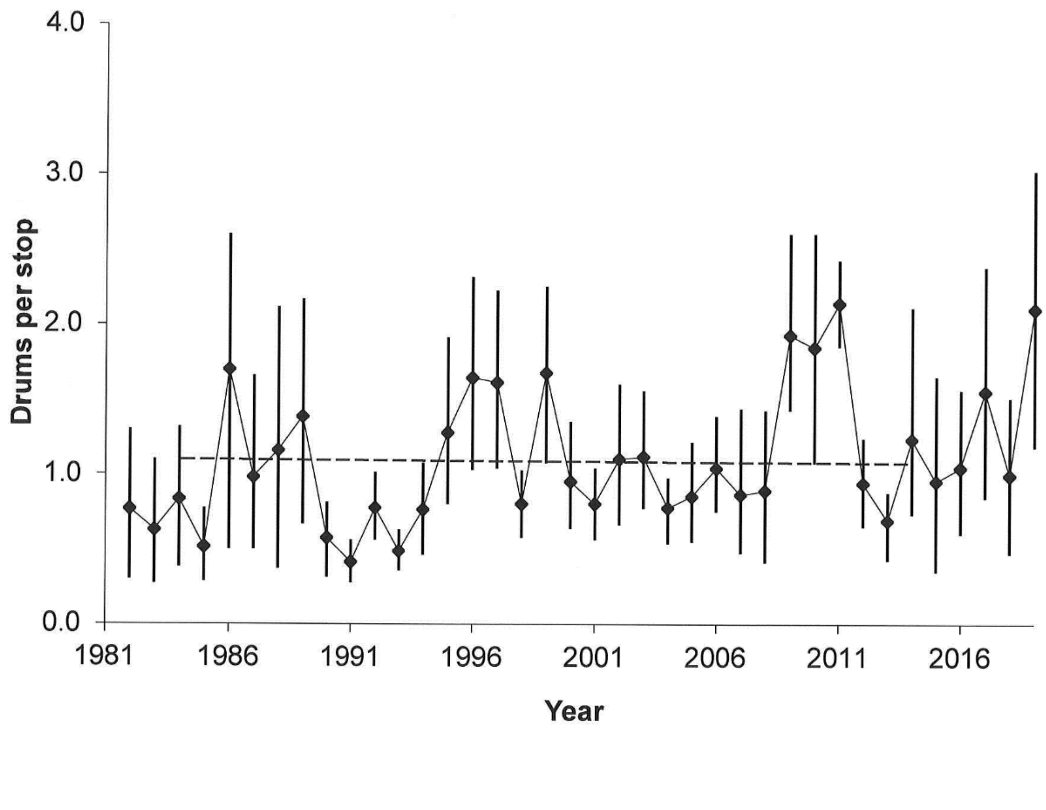


Figure 3. Statewide ruffed grouse population index values in Minnesota. Bootstrap (95%) confidence intervals (CI) are provided after 1981, but different analytical methods were used prior to this and thus CI are not available for earlier years. The difference between 1981 and 1982 is biological and not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

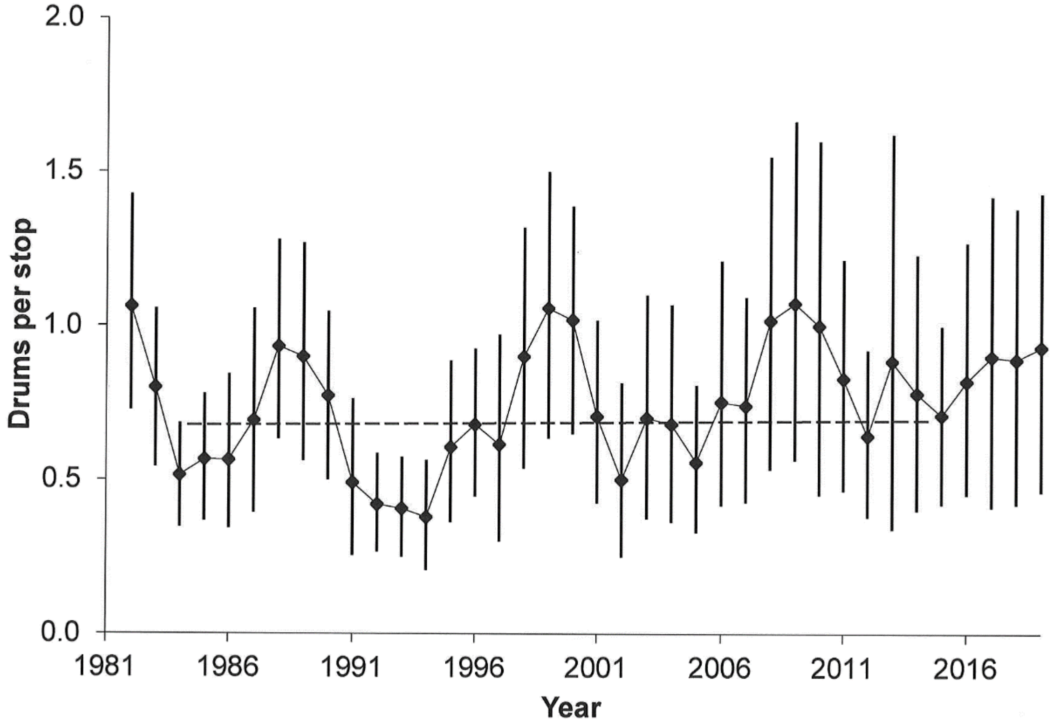
a.


3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.0
Drums per stop
1981
1986
1991
1996
2001
2006
2011
2016
Year

b.


4.0
3.0
2.0
1.0
0.0
Drums per stop
1981
1986
1991
1996
2001
2006
2011
2016
Year

c.



d.

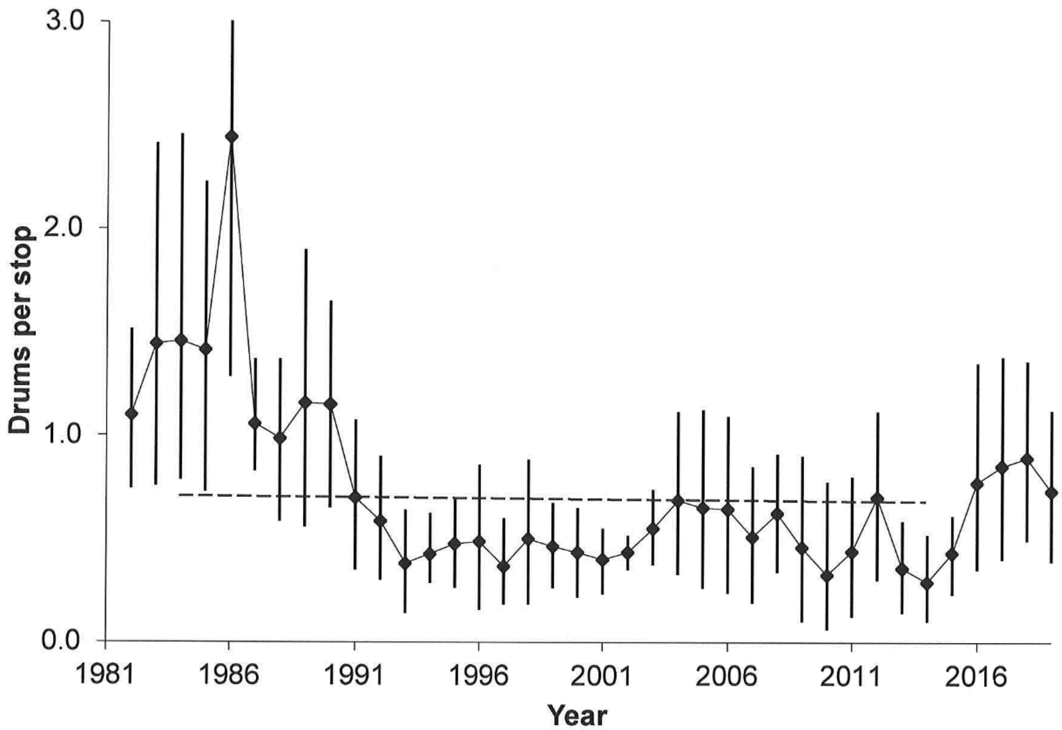


Figure 4a,b,c,d. Ruffed grouse population index values in the **Northeast** (a), **Northwest** (b), **Central Hardwoods** (c), and **Southeast** (d) survey regions of Minnesota. The mean for 1984-2014 is indicated by the dashed line. Bootstrap (95%) confidence intervals are provided for each mean. In the bottom panel, the CI for 1986 extends beyond area depicted in the figure.

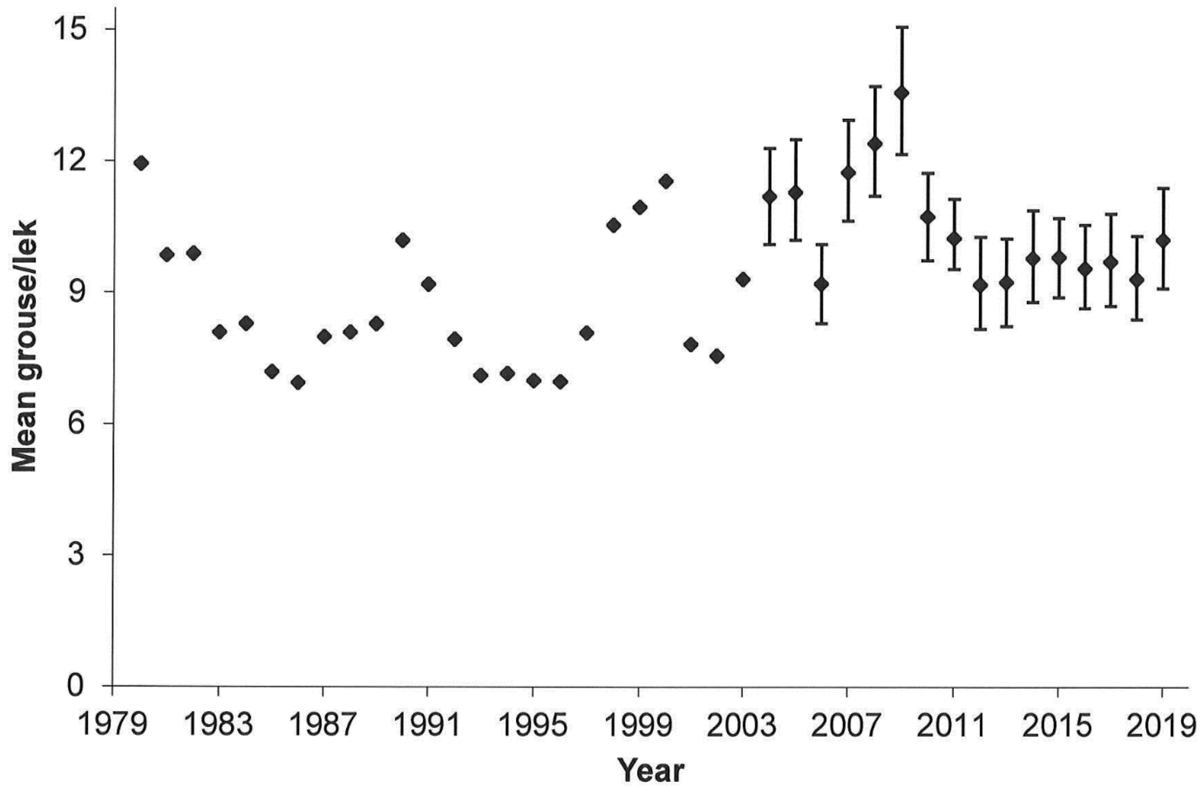


Figure 5. **Sharp-tailed grouse** counted in spring lek surveys statewide in Minnesota during 1980–2019. Bootstrap (95%) confidence intervals are provided for recent years. Annual means are not connected by lines because the same leks were not surveyed every year.

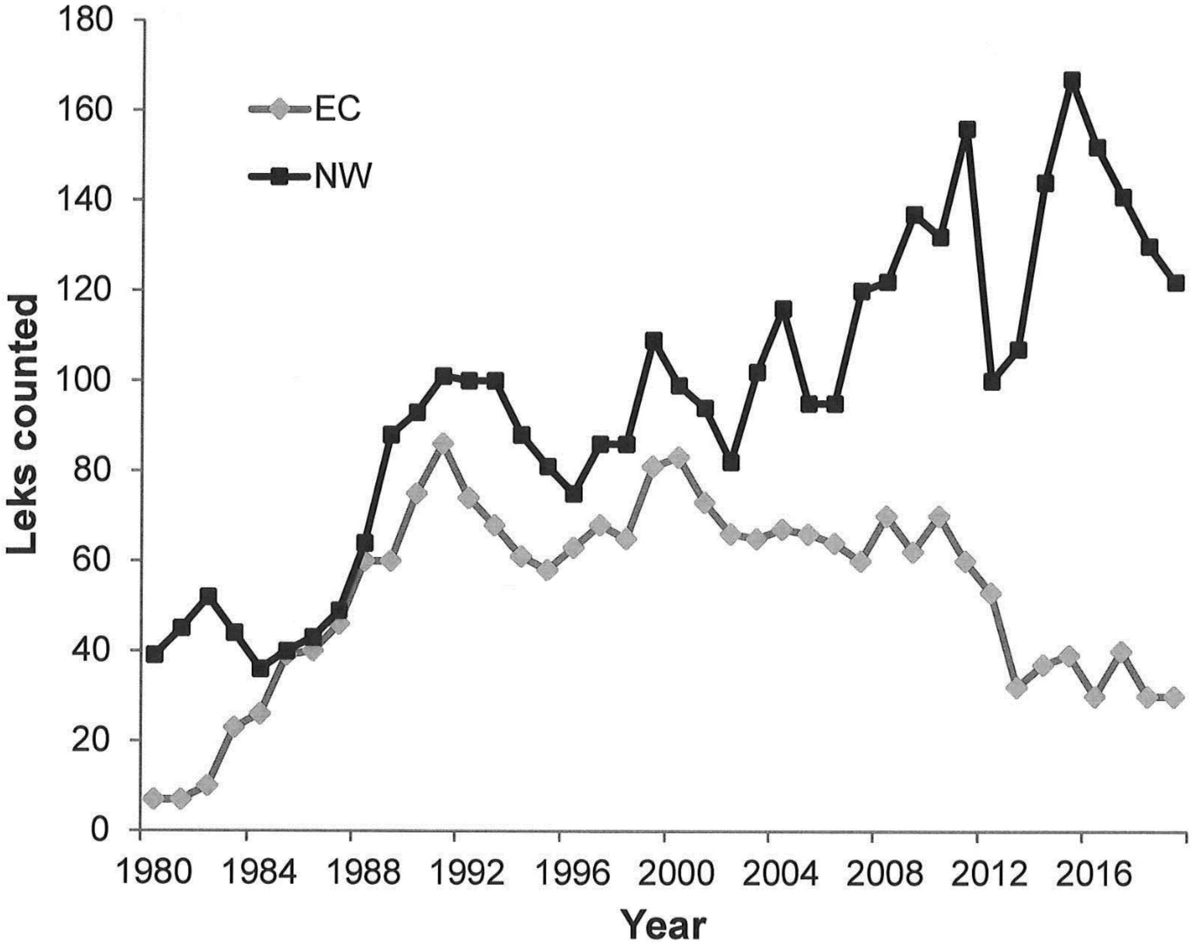


Figure 6. The number of **sharp-tailed grouse** leks with 2 or more birds counted in spring lek surveys in the Northwest (NW) and East Central (EC) survey regions of Minnesota during 1980- 2019.

2019 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 all 17 survey blocks during the spring of 2019. Observers located 45 booming grounds and counted 497 males and birds of unknown sex in the survey blocks, which is a decline of more than 20% in the number of leks and birds counted compared to last year. Including areas outside the survey blocks, observers located 113 booming grounds, 1,039 male prairie-chickens, and 115 birds of unknown sex throughout the prairie-chicken range. Estimated densities of 0.06 (0.05–0.08) booming grounds/km^2 and 11.0 (8.5–13.6) 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). All population indices began to decline in 2008, but seem to have stabilized in recent years at a lower level.

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. However, as grass was lost from the landscape, prairie-chicken populations began to decline, 2 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 (DNR) conducted a series of translocations in the Upper Minnesota River Valley during 19982006. 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 approximately 120 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 all 17 designated survey blocks in western Minnesota (Figure 2) during April and May. Each survey block was 3 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/km^2 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. 4

RESULTS & DISCUSSION

Observers from DNR Division of Fish and Wildlife, the U.S. Fish & Wildlife Service, and The Nature Conservancy, as well as many unaffiliated volunteers counted prairie-chickens between 6 April and 14 May 2019. Observers located 113 booming grounds and observed 1,039 male prairie-chickens and 115 birds of unknown sex within and outside the survey blocks (Table 1). These counts represent a minimum number of prairie-chickens in Minnesota during 2019, but because survey effort outside of survey blocks is not standardized among years, these counts should not be compared among years or permit areas.

Table 1. Minimum abundance of prairie-chickens within and outside hunting permit areas in Minnesota during spring 2019. Lek and bird counts are not comparable among permit areas or years.

Permit Area	Area (km2)	Leks	Males	Unk[a]
803A	1,411	11	68	0
804A	435	1	8	0
805A	267	12	89	4
806A	747	13	58	19
807A	440	14	164	25
808A	417	20	309	0
809A	744	13	161	0
810A	505	3	39	11
811A	706	7	31	15
812A	914	6	23	0
813A	925	4	29	2
PA subtotal	7,511	104	979	76
Outside PAs[b]	NAc	9	60	39
Grand total	NAc	113	1,039	115

[a] Unk = 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.

Within the standardized survey blocks, 497 males and birds of unknown sex were counted on 45 booming grounds during 2019 (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 2.5 times (median = 2), with 35% of booming grounds observed 5 just once. These counts should not be regarded as estimates of abundance because detection probabilities of leks and birds were not estimated. However, if we assume that 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 (0.05–0.10) booming grounds/km^2 and 12.3 (9.2–15.4) males/booming ground (Table 2, Figure 2). In the 7 peripheral survey blocks, densities were 0.04 (0.02–0.07) booming grounds/km^2 and 8.0 (4.1– 11.9) males/booming ground. The density of 0.06 (0.05–0.08) booming grounds/km^2 in all survey blocks during 2019 was similar to densities during recent years (Table 2, Figure 3) and the average of 0.08 (0.06–0.09) booming grounds/km^2 during the 10 years preceding recent hunting seasons (i.e., 1993–2002). Similarly, the density of 11.0 (8.5–13.6) males/booming ground in all survey blocks during 2019 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, these densities are lower than the years preceding 2008 when CRP enrollments in the counties containing the survey blocks were highest.

Densities appear to have stabilized over the last several years at a new lower level. 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*

Table 2. Prairie-chicken counts within survey blocks in Minnesota.

			2019		Change from 2018[a]	
Range[b]	Survey Block	Area (km^2)	Booming grounds	Males[c]	Booming grounds	Males[c]
Core	Polk 1	41.2	5	26	1	-9
	Polk 2	42.0	3	32	-2	-33
	Norman 1	42.0	1	3	0	-5
	Norman 2	42.2	3	21	-2	-10
	Norman 3	41.0	3	25	-4	-25
	Clay 1	46.0	7	126	1	22
	Clay 2	41.0	2	55	0	0
	Clay 3	42.0	4	61	-1	-25
	Clay 4	39.0	2	7	1	4
	Wilkin 1	40.0	2	38	-2	-3
	Core subtotal	415.0	32	393	-8	-84
Periphery	Mahnomen	41.7	2	42	-1	-20
	Becker 1	41.4	4	17	-3	-31
	Becker 2	41.7	1	6	-1	1
	Wilkin 2	41.7	1	10	0	6
	Wilkin 3	42.0	3	13	0	0
	Otter Tail 1	41.0	1	8	-1	-3
	Otter Tail 2	40.7	1	9	-2	-12
	Periphery subtotal	290.6	13	104	-8	-59
Grand Total		705.5	45	497	-16	-143

[a] The 2018 count was subtracted from the 2019 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.

ACKNOWLEDGMENTS

I would like to thank cooperators who conducted and helped coordinate the prairie-chicken survey. Cooperators within the DNR included Emily Hutchins, Brian Torgusson, Rob Baden, Michael Oehler, Matt Morin, and Becky Ekstein; cooperators with The Nature Conservancy included Brian Winter, Travis Issendorf, and volunteers Pat Beauzay, Matt Mecklenburg, Tyler Larson, Derek Savage, Tony Nelson, and Carl Altenbernd; cooperators with the US Fish and Wildlife Service included Shawn Papon, Chad Raitz, Ben Walker, Erin Lentz, Traver Fields, and Stacy Salvevold; and numerous additional volunteers participated, including Dan Svedarsky, Doug Wells, Jon Voz, Ross Hier, Phil Doll, and Doug Hedtke. University of North Dakota students also assisted with surveys this year. This survey was funded in part by the Wildlife Restoration (Pittman-Robertson) Program W-69-S-16 Project #12. Mike Larson provided assistance and comments which improved this report.

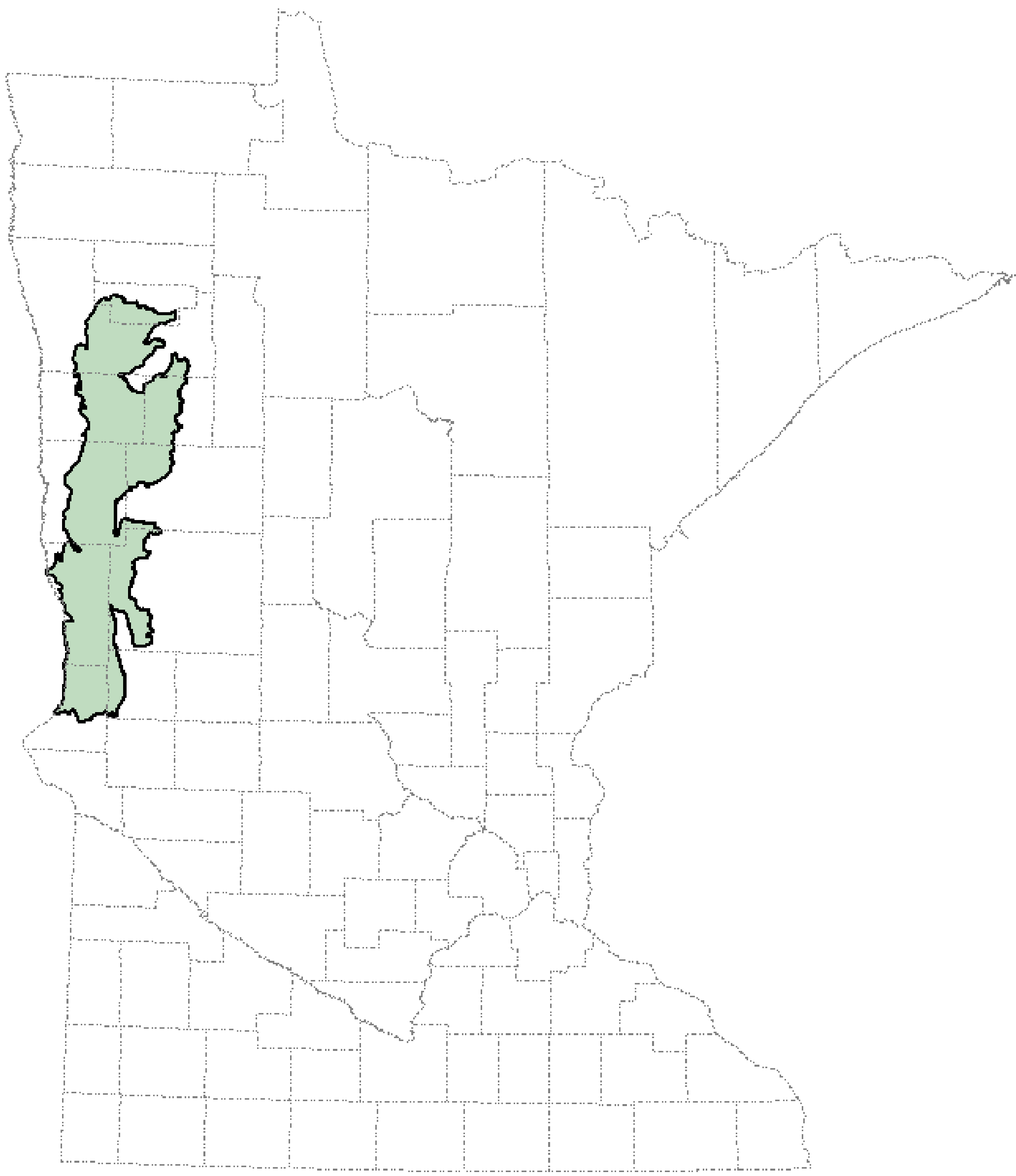


Figure 1. Primary greater prairie-chicken range in Minnesota (shaded area) relative to county boundaries. The range boundary was based on Ecological Classification System Land Type Associations and excludes some areas known to be occupied by prairie-chickens.

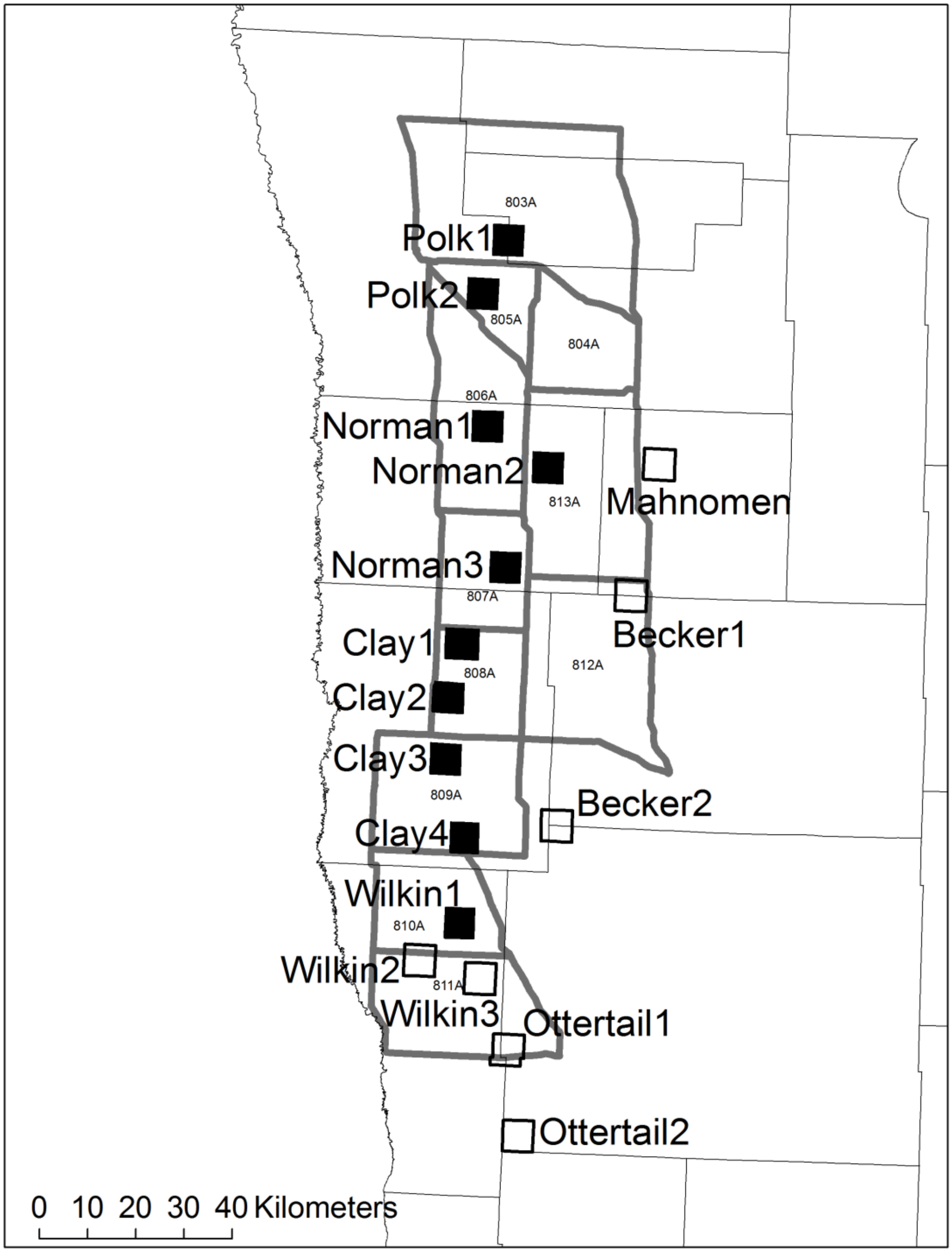


Figure 2. Prairie-chicken lek survey blocks (41 km^2, labeled squares) and hunting permit areas (thick grey lines) in western Minnesota. Survey blocks were either in the core (black) or periphery (white) of the range with a threshold of 1.0 male/km^2 in 2010, and were named after their respective counties (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.

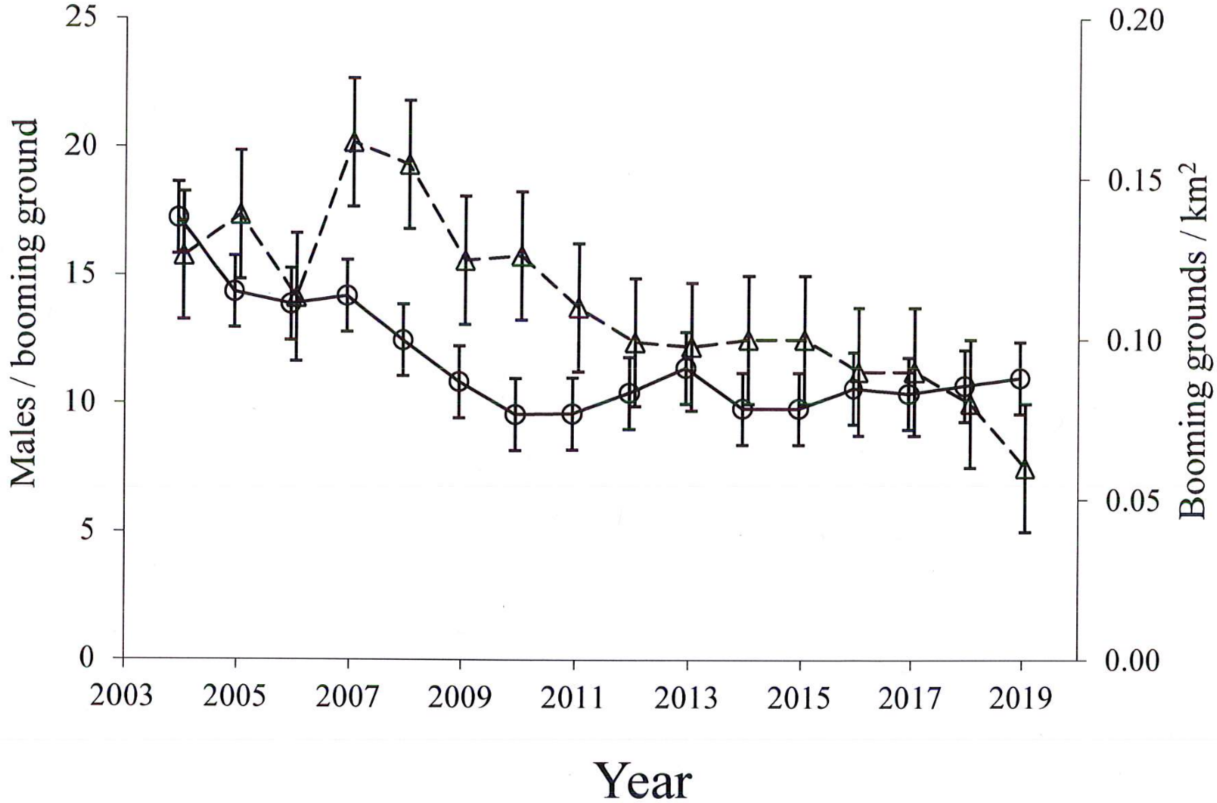


Figure 3. Mean prairie-chicken males/booming ground (circles connected by solid line) and booming grounds/km^2 (triangles connected by dashed line) in survey blocks in Minnesota with 95% confidence intervals.

DEPARTMENT OF NATURAL RESOURCES

2019 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 10th and February 16th, 2019. 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 80-85 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 the animals moving and the altitude and speed of the fixed-wing aircraft. Antlered elk were recorded as either branch antlered or spike bulls.

The survey block for Grygla was expanded this year by fourteen square miles in the northeast corner after local landowners reported two mature bulls frequenting an area outside of the previous boundary. The same predetermined transects used in 2018 were flown for the Lancaster survey block. The Caribou-Vita elk survey block was not flown this year since Manitoba Wildlife was not able to fund an aerial elk survey on the Canadian side.

Observability conditions were excellent this year. Snow depths and conditions were very consistent and considered very good for both elk survey blocks. Snow depths ranged from 20 to 25 inches across both the Grygla and Lancaster areas. Weather conditions were also very good for this time of the year with temperatures ranging from a low of -10°F to a high of 13°F with mostly cloudy skies. There was a two-day weather delay between the first and second days of the Grygla survey due to snow and high winds.

Grygla Survey Block

This survey started on February 10th and after a two-day weather delay was completed on February 13, 2019. The area surveyed was the same 133 mi^2 block that has been used the past two years with an additional 14 mi^2 added in the northeast corner—147 mi^2 total (Figure 2). After the 2018 survey, Thief Lake WMA staff received information that a landowner had been feeding two bull elk just north of the survey block. This prompted the decision to expand the survey boundary. Total aircraft engine time to complete this survey (takeoff to landing) was 10.9 hours. The fixed-wing crew recorded elk at 4 separate locations within the survey boundary--all elk were observed on the first day. Total elk observed was 19 and included: 8 antlerless and 11 bulls (10 branch antlered and 1 spike). Of special note is that many 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 15th and was completed on February 16, 2019. The area surveyed was the same 167 mi^2 area that has been flown the past several years (Figure 1). Total aircraft time to complete the survey was 14.5 hours (takeoff to landing). The fixed-wing crew recorded elk at 7 separate locations within the survey boundary. Total elk recorded within the Lancaster block was 94 and included: 61 antlerless and 33 bulls (22 branch antlered and 11 spikes). As with the Grygla elk herd, there were several elk either directly located on or in close proximity to State Wildlife Management Area land at the time of the survey.

- The Water Tower herd had 37 antlerless and 2 spike bull elk and were located in the same exact woodlot the antlerless group was recorded in 2018. In addition, there were 7 branch antlered and 5 spike bulls located within one to five miles of the antlerless group.
- The Percy WMA herd had 24 antlerless and 1 spike bull elk and were located approximately four miles northwest of the Percy WMA (within one mile of the 2018 location). There were 14 branch antlered and 4 spike bulls observed within 2 to 3 miles east of the antlerless group. One lone branch antlered bull was located near the western edge of the Percy WMA (similar location where a single spike bull was observed in 2018).

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

This survey block was not completed in 2019. 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 summarizes MN DNR elk observations during the past five years of NW MN aerial elk surveys. The last two pages are maps showing the 2019 locations of elk within each survey block.

ACKNOWLEDGMENTS

I would like 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! 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 2015 and 2019 for the Lancaster, Caribou-Vita, and Grygla herds.

	Lancaster					Caribou-Vita (US side of border)					Grygla				
	2015	2016	2017	2018	2019	2015	2016	2017	2018	2019*	2015	2016	2017	2018	2019
Spike bull	2	6	2	5	11	5	0	0	1	-	3	2	4	2	1
Branch antlered bull	16	12	14	13	22	17	6	1	6	-	6	9	6	6	10
Total bulls	**18**	**18**	**16**	**18**	**33**	**22**	**6**	**1**	**7**	**-**	**9**	**11**	**10**	**8**	**11**
Antlerless	**16**	**34**	**45**	**57**	**61**	**57**	**4**	**0**	**0**	**-**	**9**	**10**	**7**	**7**	**8**
Total elk	**34**	**52**	**61**	**75**	**94**	**79**	**10**	**1**	**7**	**-**	**18**	**21**	**17**	**15**	**19**

* Survey was not completed in 2019

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

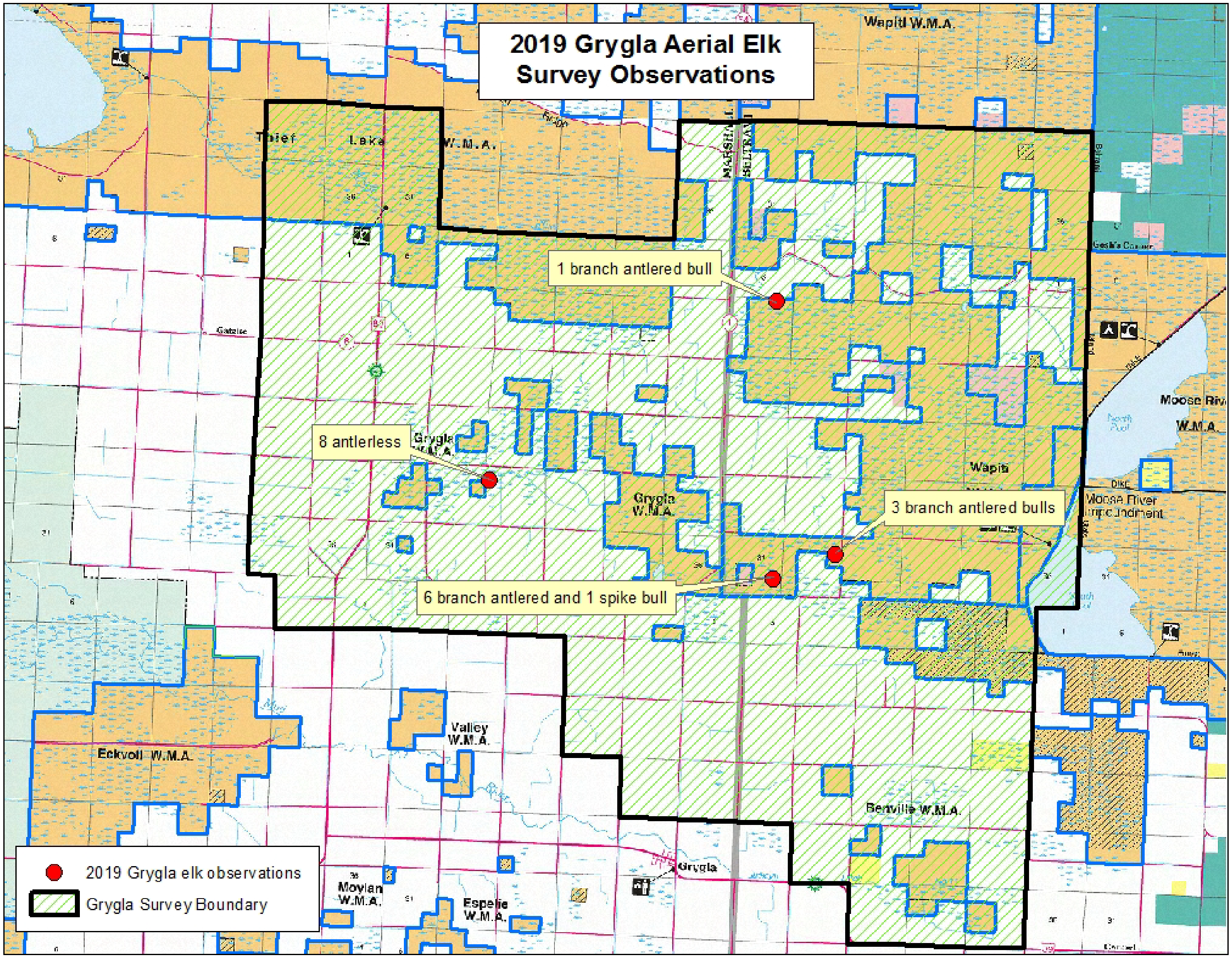


Figure 1. Locations of elk observed within the Grygla area survey blocks, 2019

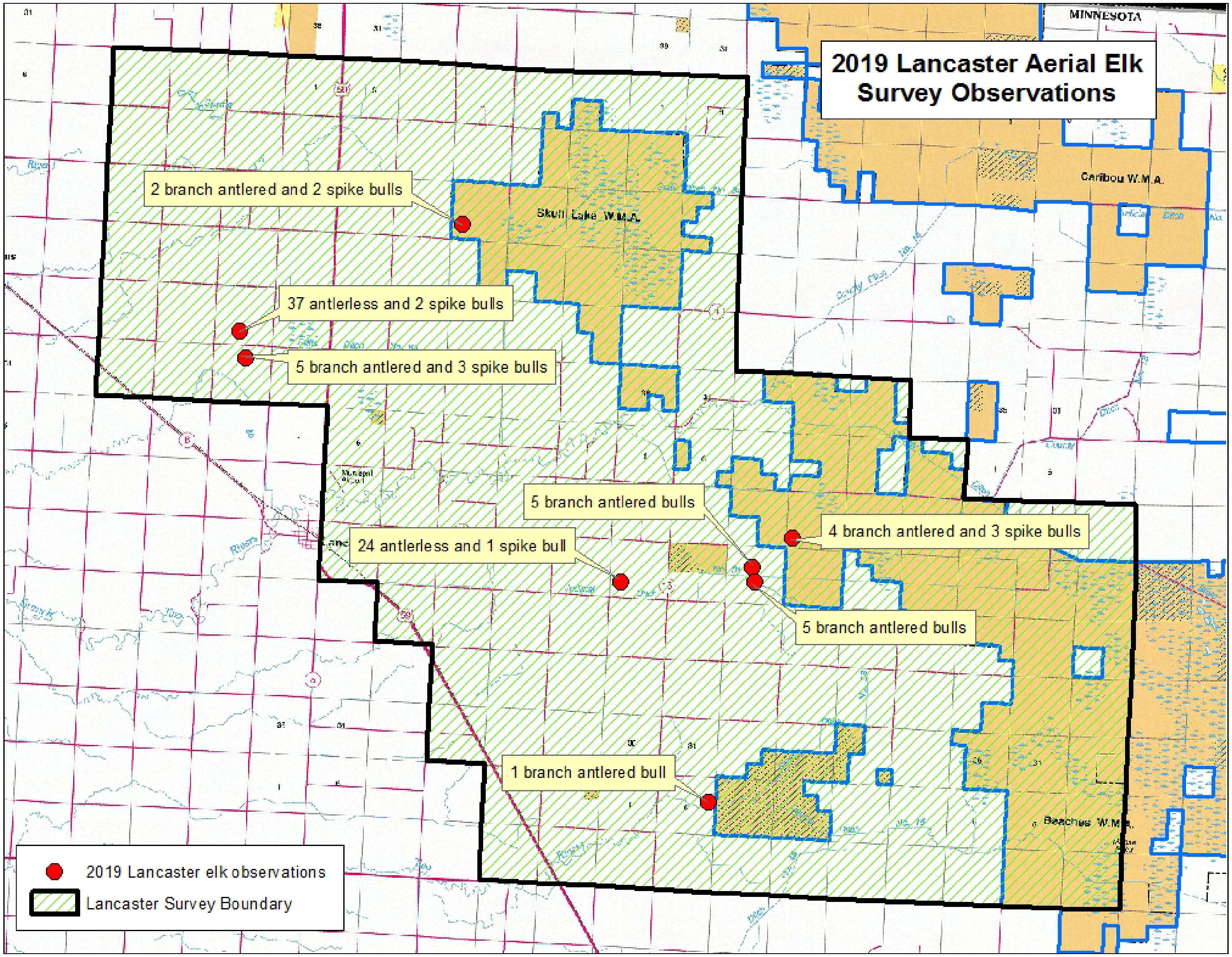


Figure 2. Locations of elk observed within the Lancaster area survey blocks, 2019.

2019 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 mi^2 (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 the 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, or high density classes are based on whether ≤ 2, 3–7, or $\geq$8 moose, respectively, would be expected to be observed in a specific plot. The most recent re-stratification was conducted in October 2018 for the 2019 survey. 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 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-mi^2 rectangles (5 x 2.77 mi), oriented east to west, with 8 flight-transects 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).

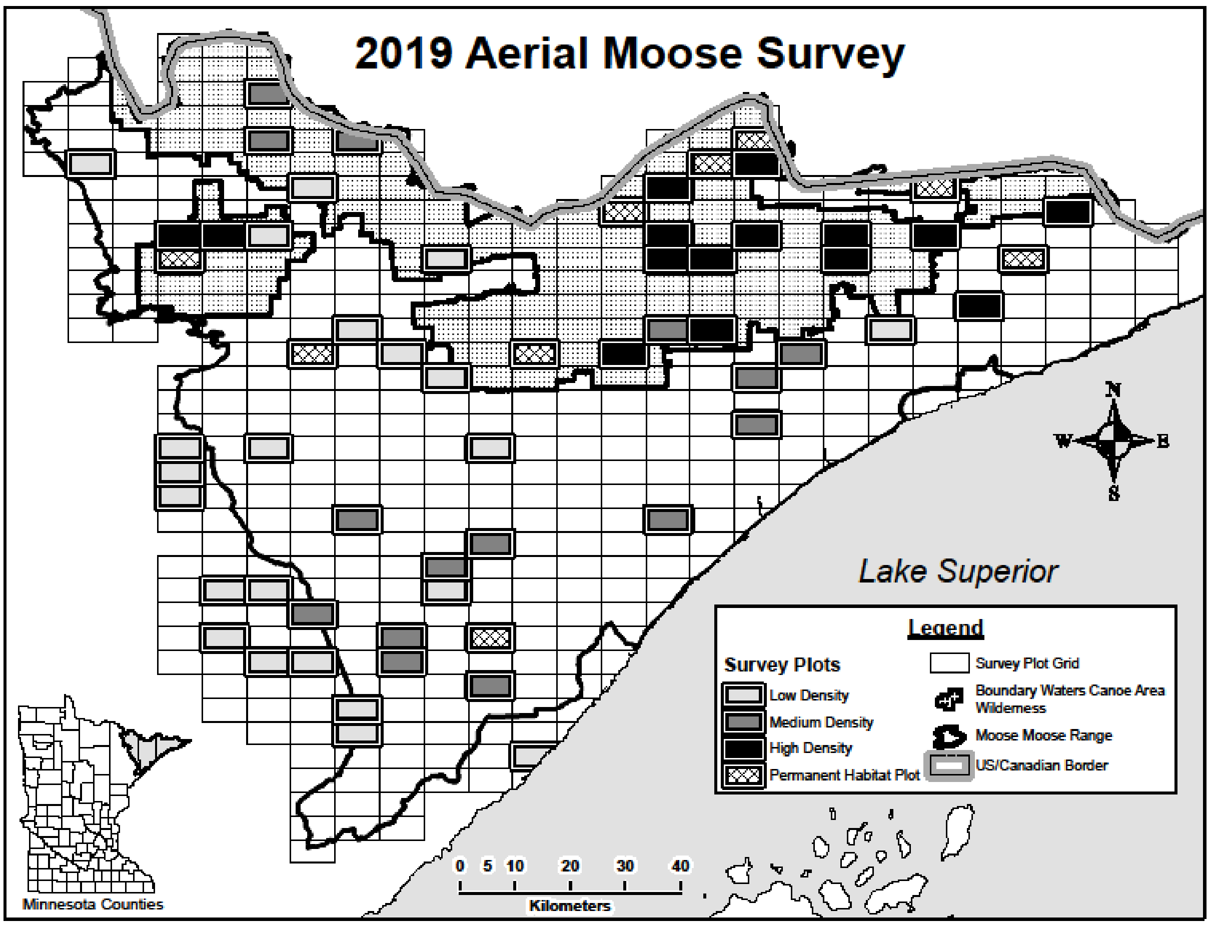


Figure 1. Moose survey area and 52 sample plots flown in the 2019 aerial moose survey.

RESULTS AND DISCUSSION

The survey was conducted from 3 to 17 January 2019. It consisted of 10 actual survey days, and as from 2014 to 2018, it included a sample of 52 survey plots. This year, based on optimal allocation analyses, we surveyed 19 low-, 12 medium-, and 12 high-density plots, and the 9 permanent or habitat plots (Giudice 2019). Generally, 8" of snow cover is our minimum threshold depth for conducting the survey. Snow depths were 8–16" and >16" on 8% and 92% of the sample plots, respectively. Overall, survey conditions were rated as good for 86%, fair for 12%, and poor for 2% of the plots when surveyed. Average survey intensity was 47 minutes/plot (13.9 mi^2) and ranged from 30 to 55 minutes/plot (Giudice 2019).

This year 429 moose were observed on 43 (83%) of the 52 plots surveyed (a total 723 mi^2), more than the 415 moose observed on 37 of 52 plots during the 2018 survey. An average of 10.0 moose (range = 1–38) were observed per "occupied" plot. Plot occupancy during the past 15 years averaged 81% (range = 65–95%) with a mean 11.7 moose observed per occupied plot. This year's 429 observed moose included 179 bulls, 182 cows, 61 calves, and 7 unclassified adults. Overall, estimated VO averaged 40% (range = 5−90%) and average estimated detection probability was 0.59 (range = 0.20−0.83). 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 4,180 (3,250–5,580, 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 2018). The current population estimate is 53% less than the estimate in 2006 and the declining linear trend during the past decade remains statistically significant (r^2 = 0.76, P< 0.001, Figure 2). However, the leveling since 2012 persists, and a piecewise polynomial curve indicates that the trend from 2012 to 2019 is not declining (Figure 3). While this recent short-term trend (8-year) is noteworthy, it applies only to the existing survey estimates, and does not forecast the future trajectory of the population (Giudice 2019).

The January 2019 calf:cow ratio of 0.33 is lower than the 14-year average since 2005 (0.40, Table 1, Figure 4). Calves were 14% of the total 429 moose actually observed and represented 13% of the estimated population (Table 1, Figure 4). Twin calves were observed with 5 of the 182 (3%) 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 2019 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 2019 estimated bull:cow ratio (1.23, Table 1; Figure 5) appears to be elevated compared to the long-term average of 1.00 during 2005–2018, and compared to the mean ratio (0.87) of 2009−2012, when the population decline was steepest. Estimated bull:cow ratios have been this high previously (2013 and 2014) during the recent interval of apparent stability; however, due to the notable annual variability associated with the bull:cow ratios, there is no apparent upward or downward long-term trend (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, 2005–2019.

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 – 9,100	0.29	13	3	0.89
2008	7,890	6,080 – 10,600	0.36	17	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	13	3	1.23
2014	4,350	3,220 – 6,210	0.44	15	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	0.33	13	3	1.23

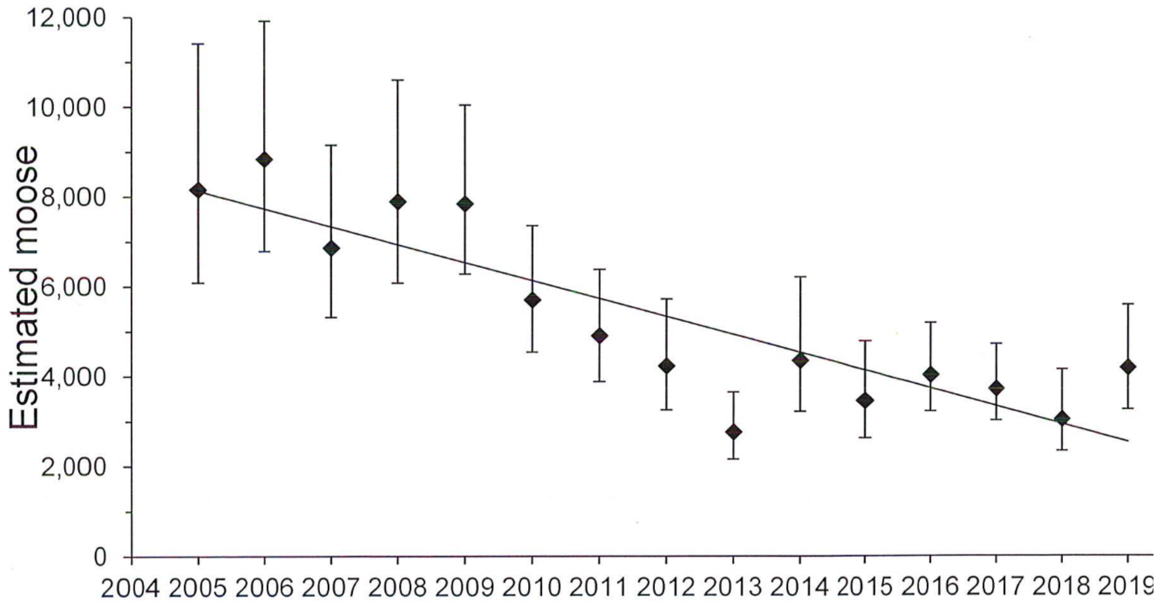


Figure 2. Point estimates, 90% confidence intervals, and a linear trend line of estimated moose abundance in northeastern Minnesota, 2005–2019 (y = -400*x* + 809841, $r^2 = 0.76$, $P < 0.001$). Note: The 2005 survey was the first to be flown with helicopters, and to include a sightability model and a uniform grid of east-west oriented rectangular 13.9-mi² plots.

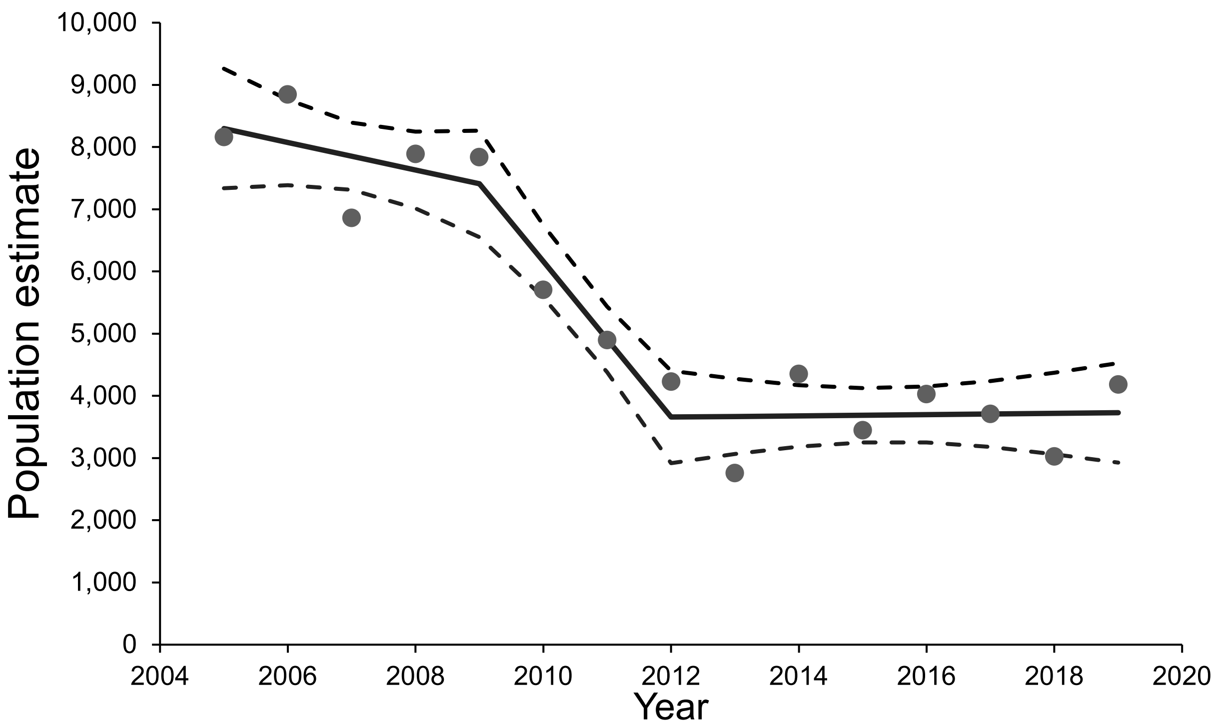


Figure 3. Point estimates, 95% confidence intervals, and a piecewise polynomial curve of moose abundance in northeastern Minnesota, 2005–2019 (Giudice 2019). This curve shows a change in the short-term slope of the trend from 2012 to 2019 compared to 2009 to 2012.

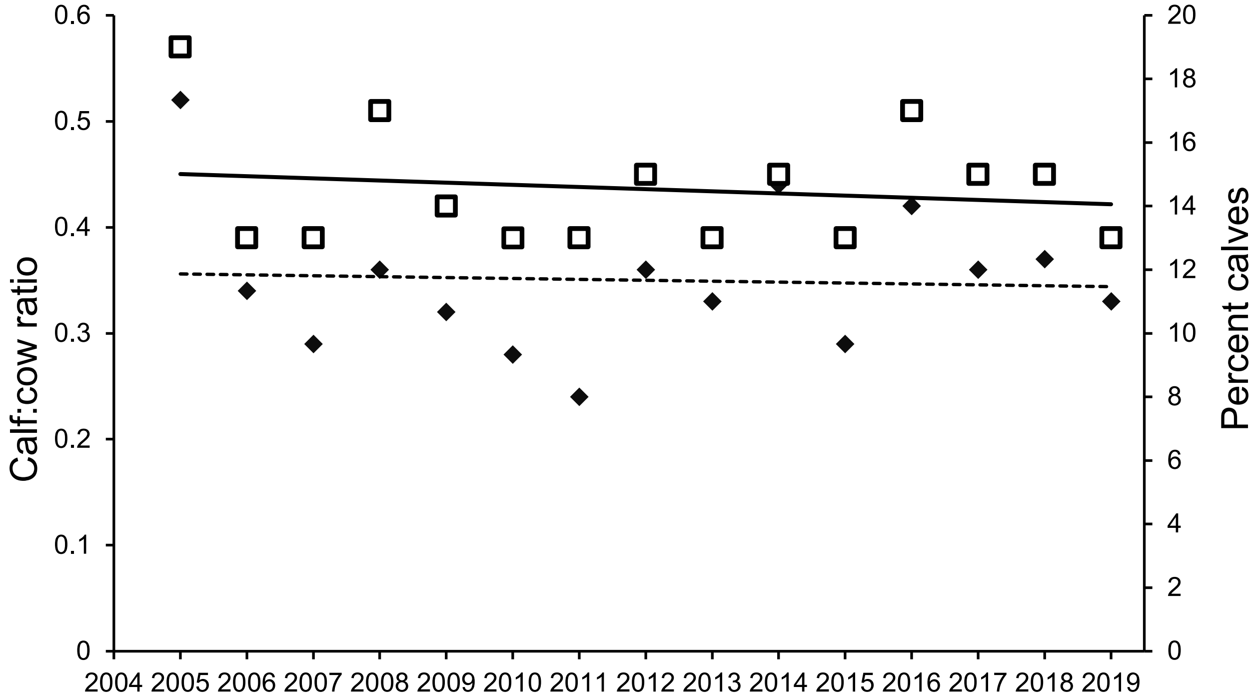


Figure 4. Estimated calf:cow ratios (solid diamonds, dashed trend line) and percent calves (open squares, solid trend line) of the population from aerial moose surveys in northeastern Minnesota, 2005–2019.

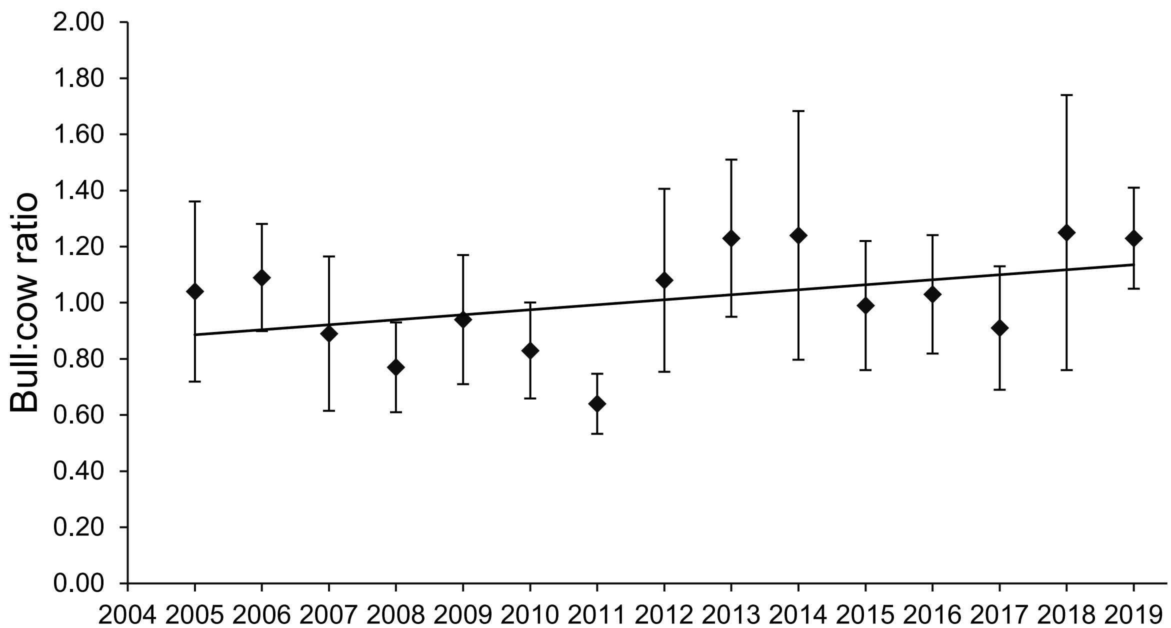


Figure 5. Estimated bull:cow ratios, 90% confidence intervals, and trend line from aerial moose surveys in northeastern Minnesota, 2005–2019.

ACKNOWLEDGMENTS

This survey is an excellent partnership between the Divisions of Enforcement and of Fish and Wildlife, the Fond du Lac Band of Lake Superior Chippewa, and the 1854 Treaty Authority. Specifically, thank you to Thomas Buker, Chief Pilot, for coordinating all of the aircraft and pilots; Tom Rusch for coordinating flights, survey crews, and other important parts of this effort; and Mike Schrage (Fond du Lac Band of Lake Superior Chippewa) and Darren Vogt (1854 Treaty Authority) for securing supplemental survey funding from their respective groups. Enforcement pilots, Brad Maas and John Heineman, and Luke Ettl (MIFC) skillfully piloted the aircraft during the surveys; Tom Rusch, Mike Schrage, and Nancy Hansen flew as our seasoned observers, and welcome to Morgan Swingen, our newest observer and member of the team. Thank you to John Giudice who continues to provide critical statistical consultation and analyses, and to Barry Sampson (now retired) for creating the process to generate the GIS survey maps and GPS coordinates for the transect lines, and for his work on re-stratification of the survey plots. We gratefully acknowledge Bob Wright, Brian Haroldson, and Chris Pouliot for creating the program, DNRSurvey. Bob also modifies the software as needed, updates specific maps, provided refresher training for survey observers using DNRSurvey, and now also assumes Barry Sampson's GIS survey responsibilities. The efforts of all of these people contribute to survey improvements, ensure the survey's rigor and the comparability of long-term results. This report has been reviewed by Paul Telander, Lou Cornicelli, Mike Larson, John Giudice, Mike Schrage, and Morgan Swingen. This project was funded in part by the Wildlife Restoration (Pittman-Robertson) Program.

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MINNESOTA WOLF POPULATION UPDATE 2019

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 2018-19 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 km^2; Erb et al. 2018) and use that in our population calculations for winter 2018-19.

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.

Twenty-five wolves have also been captured with the use of live-restraining neck snares, 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, while 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=7; mean number of radiolocations = 36), we multiplied each estimated territory size by 1.37 as in the past. For packs with > 100 radiolocations (n = 31; mean number of radiolocations = 3,040), 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 = ((km^2 of occupied range/mean scaled territory size)*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.

RESULTS AND DISCUSSION

Pack and Territory Size

A total of 39 packs were monitored during all or part of the survey period (April 2018 to April 2019). We obtained territory and winter pack size data from 26 radio-marked wolf packs (Figure 1). Twelve 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 1 pack for which there was insufficient data to delineate a territory.

Similar to winter 2017-18, a land cover comparison using the 2011 National Land Cover Database suggests that the location of collared packs this winter led to some over-representation of habitat classified as woody wetlands and under-representation of deciduous forest (Table 1), likely a combined result of slight over-representation of packs (with large territories) near Red Lake and fewer collared packs in our southwest study area. In addition, collared pack territories under-represented, as is typically the case, areas in occupied range classified as hay/pasture/cropland, largely a result of these areas being on private land where less wolf collaring is undertaken. (Table 1). Using spring 2018 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 10 deer/mi^2 (pre-fawn) in territories of radio-marked packs at the beginning of the biological year in which the survey was conducted. In comparison, 2018 spring deer density for the entirety of occupied wolf range (weighted by permit area) in Minnesota was approximately 13 deer/mi2.

The point estimate for average territory size this winter declined 7% from last winter. However, this change was not significant, and with possible exception of the 2014-15 estimate, average territory size has not fluctuated notably from 2003 to the present (Figure 2). After applying the territory scaling factors, average estimated territory size for radio-marked packs during the 2018-19 survey was 148 km^2 (range = 27 – 561 km^2).

Though the point estimate for average winter pack size declined by 5% from last winter, the confidence interval widely overlaps those from the previous 5 surveys, suggesting no significant changes. Average winter pack size in 2018-19 was estimated to be 4.6 (range = 2 – 8, Figure 3).

Wolf Numbers

Given an average territory size of 148 km^2 and assuming occupied range has not changed since the 2017-18 survey (73,972 km^2; Erb et al. 2018), we estimated a total of 500 wolf packs in Minnesota during winter 2018-19. Although also influenced by the estimated amount of occupied range, trends in the estimated number of packs (Figure 4) are generally the inverse of trends in estimated territory size (Figure 2).

After accounting for the assumed 15% lone wolves in the population, we estimated the 2018-19 mid-winter wolf population at 2,699 wolves, or 3.65 wolves per 100 km^2 of occupied range. The 90% confidence interval was approximately +/- 675 wolves, specifically 2,046 to 3,430. Given the nearly complete overlap with the 2017-18 confidence interval, we conclude that the 2018-19 statewide wolf population was unchanged from the previous winter.

Although local variability occurred, from spring 2018 to spring 2019 the overall average deer density within wolf range remained stable. Over the past 5 years, wolf population estimates have been positively correlated with average deer density within wolf range (Figure 6).

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 % Area	Radio-collared Wolf Territories % Area
Woody Wetlands	32.6	38.0
Deciduous Forest	23.6	19.6
Emergent Herbaceous Wetlands	9.9	12.1
Mixed Forest	7.2	8.6
Evergreen Forest	7.0	8.7
Open Water	5.4	4.6
Shrub/Scrub	4.5	4.8
Pasture/Hay/Grassland/Crops	7.7	2.1
Developed, All	2.2	1.5

[a] Land cover data derived from the 2011 National Land Cover Database

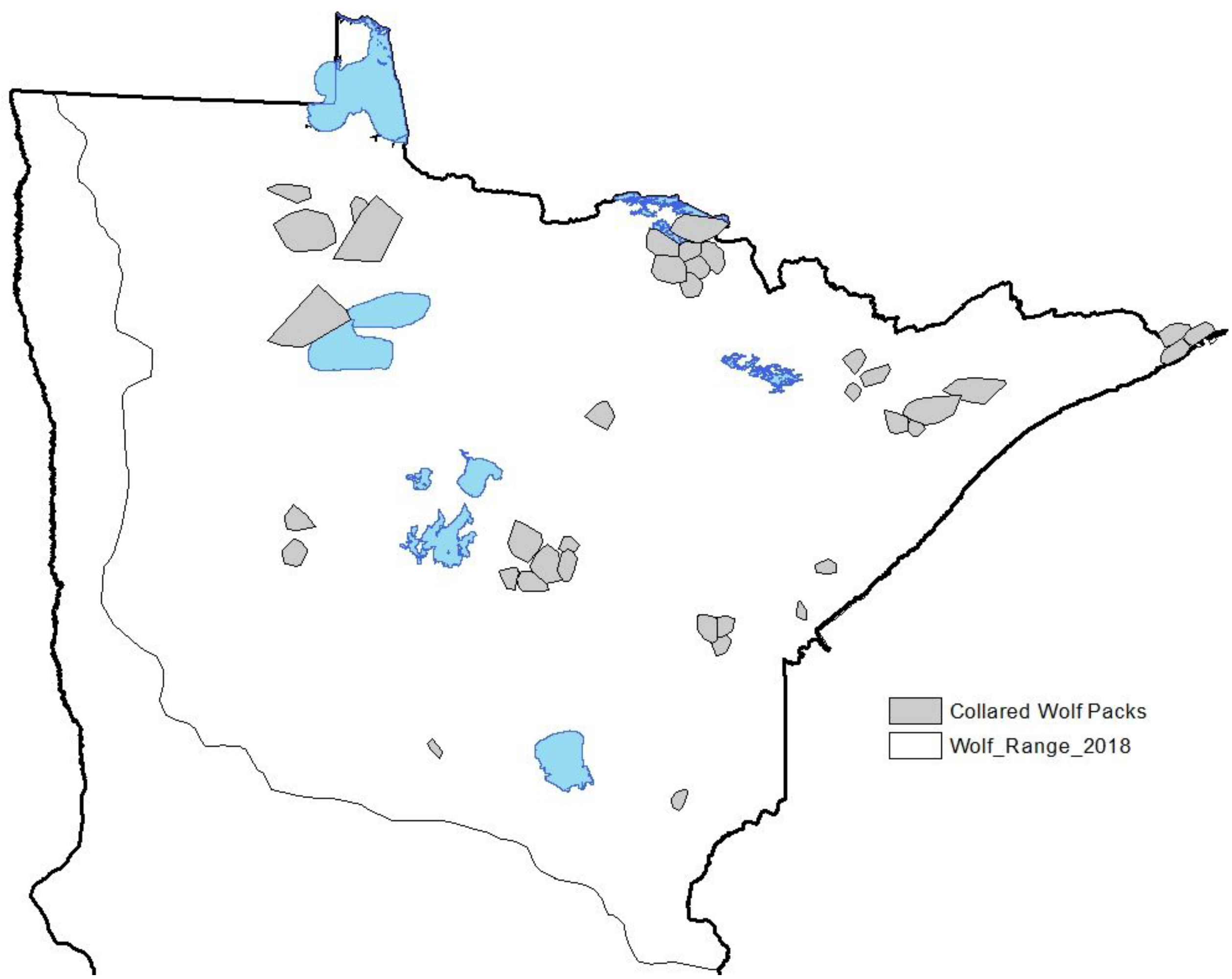

Figure 1. Location of radio-marked wolf packs during the 2018-19 survey.

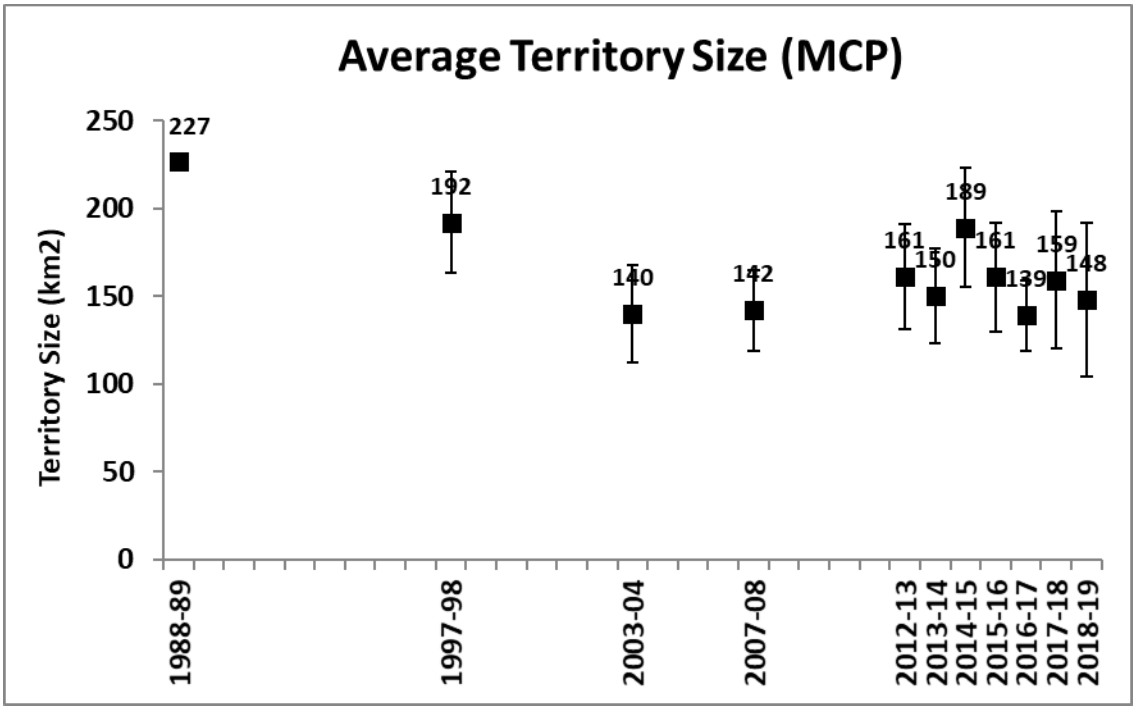

Figure 2. Average scaled territory size for radio-marked wolf packs in Minnesota from 1989 to 2018.

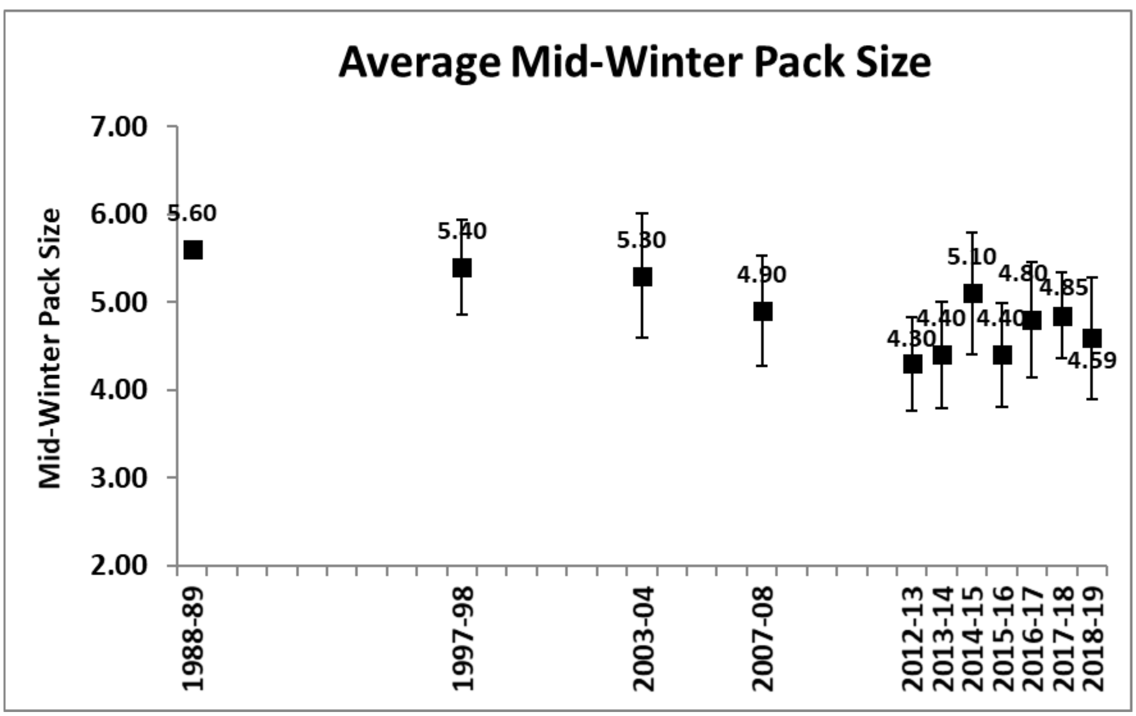

Figure 3. Average mid-winter pack size for radio-marked wolf packs in Minnesota from 1989 to 2018.

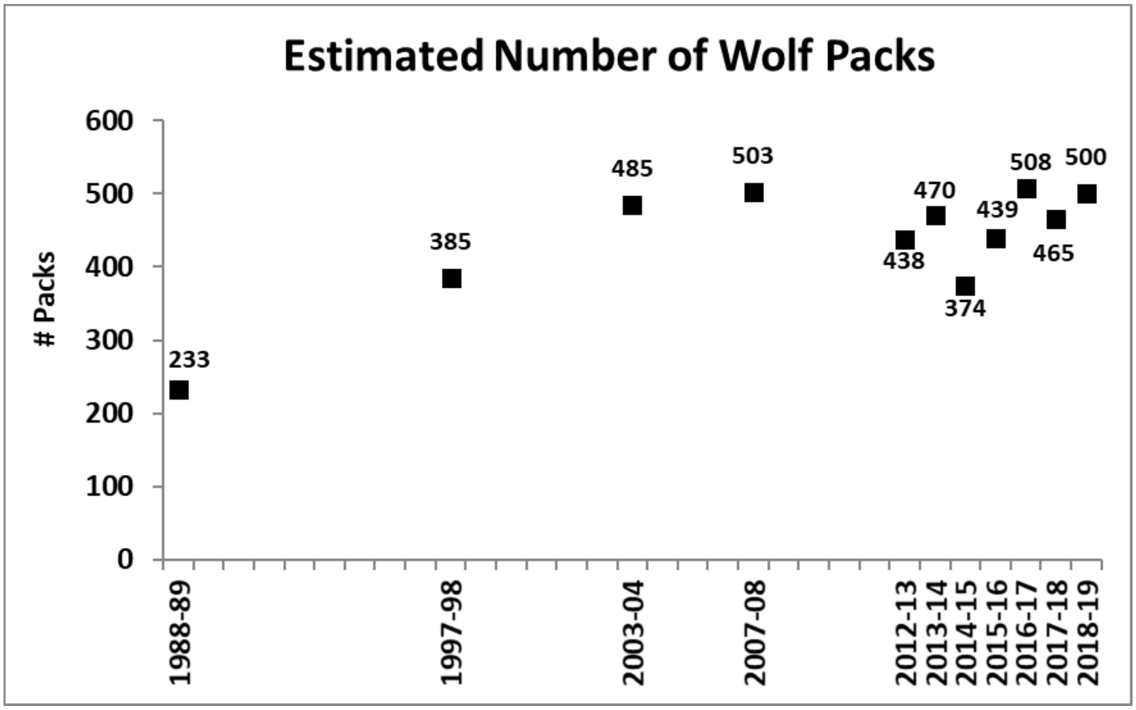

Figure 4. Estimated number of wolf packs in Minnesota at periodic intervals from 1989 to 2018.

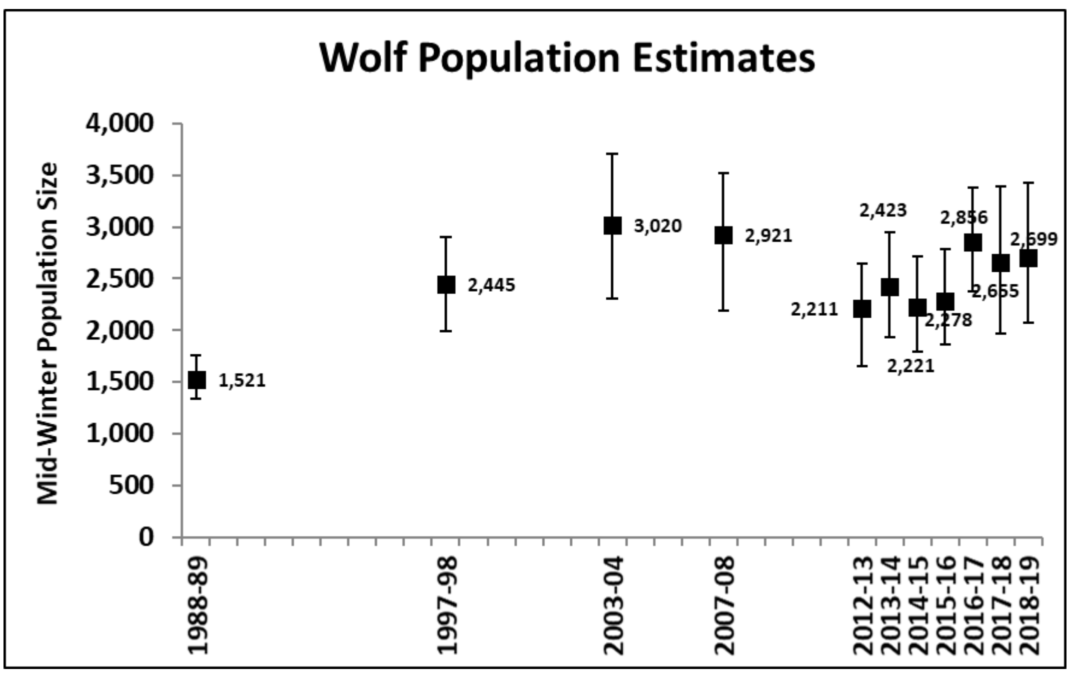

Figure 5. Wolf population estimates from periodic standardized surveys in Minnesota from 1989 to 2018.

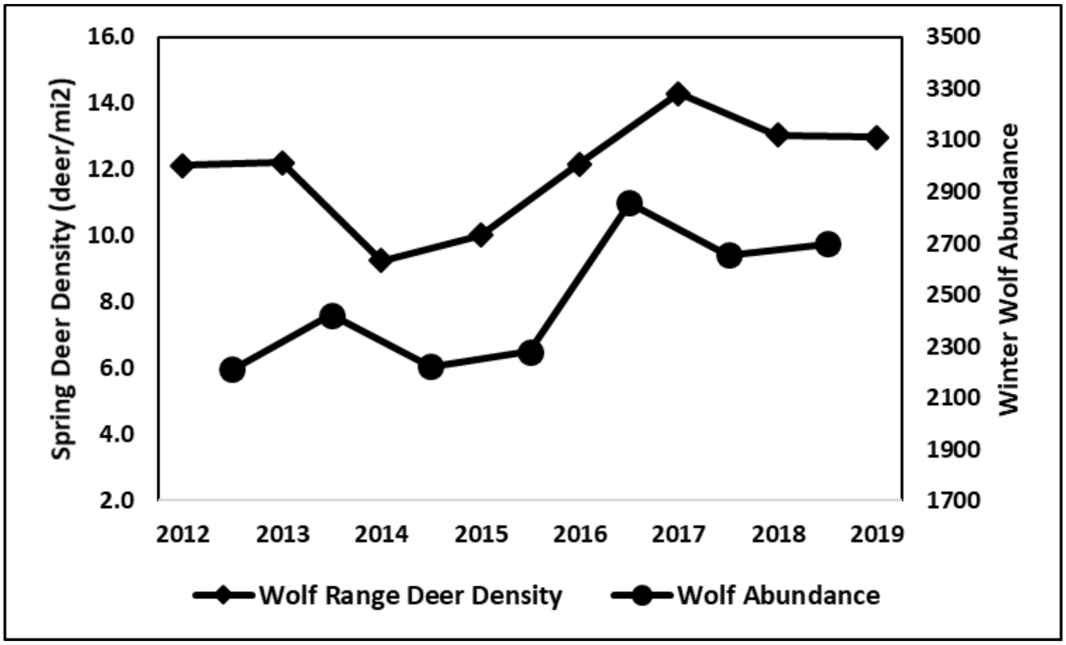

Figure 6. Comparison of estimated spring (pre-fawn) deer density and winter wolf abundance in Minnesota, 2012-2019.

ACKNOWLEDGMENTS

We thank staff with the USDA Wildlife Services program (John Hart, Kevin Fuller, and Jeff Grabarkewitz) and Barry Sampson for assistance with capturing and radio-collaring wolves. We are grateful for the critical contributions of DNR pilots Jason Jensen, John Heineman, Chris Lofstuen, Tom Buker, and Bob Geving during wolf telemetry and winter pack counts. Special thanks to numerous collaborators for their assistance or sharing of radio-telemetry data utilized in this survey, including Dave Mech and Shannon Barber-Meyer (USGS), Steve Windels, Tom Gable, Austin Homkes, and Sean Johnson-Bice (Voyageurs National Park), Jay Huseby, Dave Price, and Sarah Ruffing (Red Lake Band of Chippewa), Seth Moore and Yvette Ibrahim (Grand Portage Band of Chippewa), Mike Schrage, Lance Overland, Terry Perrault, and Matt Weske (Fond du Lac Resource Management Division), Morgan Swingen (1854 Treaty Authority), and Brian Dirks and Nancy Dietz (Camp Ripley Military Reservation). This project was funded in part by the Wildlife Restoration Program (Pittman-Robertson).

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