FOREST WILDLIFE POPULATIONS

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STATUS OF MINNESOTA BLACK BEARS, 2017

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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 were used to reconstruct a minimum statewide population through time. 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

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 (Figure 1). This was scaled upwards (to include bears that died of other causes), using 4 statewide tetracycline mark–recapture estimates as a guide (Figure 2). Whereas both the tetracycline-based and reconstructed populations showed a “humped” trajectory, with an increase during the 1990s, followed by a decline during the 2000s, the shapes of the 2 trajectories differed somewhat (the reconstructed population curves were less steep). Therefore, it was not possible to exactly match the curve from the reconstruction to all 4 tet- based estimates.

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), ensuing population estimates will be biased low, so it is possible that the curve for the most recent years should be higher.

Harvests were intentionally reduced in the quota zone when it was surmised (in the mid-2000s) that the population was declining. Since 2013, quotas were maintained at a low and fairly consistent level (Table 1), although harvests varied with food.

Population reconstruction does not provide reliable estimates for the 2 most recent years, so the most recent estimate is pre-hunt 2015. This estimate shows an increase of about 10%, following the very low harvest of 2014. Both quota and no- quota zones increased by about the same percent. However, the unexpectedly high harvest of 2016 (in both quota and no-quota zones) is not yet reflected in the model estimates.

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 (≥7 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 4). Harvest rates since 2014 have been significantly less than what they were in the early 1980s, when the bear population was increasing (Figure 2).

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 5). Number of complaints declined in 2017, despite a higher number of DNR personnel recording complaints. A new recording system was instituted in 2017 whereby Wildlife Managers recorded all bear complaints online as they were received, instead of submitting reports at the end of each month (thus, unlike previous years, Managers who had no complaints were not counted in the number of personnel participating). Conservation Officers continued to use the monthly reporting system (and recorded zero when they had no complaints). Six Wildlife Managers and 2 Conservation Officers received 20 or more (up to 40) nuisance bear reports in 2017. The number of nuisance bears killed in 2017 was less than half that of 2015 and 2016. Conservation Officers recorded 4x the number of bears killed than Wildlife Managers. A new effort to target nuisance bears through an “area 88” quota hunting license resulted in only 1 bear being killed. No bears were killed by permittees.

Food abundance

The composite range-wide, all-season abundance of natural bear foods (fruits and nuts) in 2017 was similar to 2016; this was lower than 2013 and 2014 (both good food years) and above 2015 (a poor food year). Regionally in 2017 (Figure 6), more summer foods were below than above the long-term (33-year) average (Figures 4, 5, 6). The statewide fall food index (productivity of dogwood+oak+hazel), which helps predict annual harvest after accounting for hunter effort (Figure 7), was equivalent to 2013 and 2014, and considerably higher than 2015 and 2016. Dogwood and hazelnut production were low in the north-central and northeast, but high in east-central. Oak was above average in the northwest and north-central, and average elsewhere.

Predictions of harvest from food abundance

The 2017 statewide harvest was close to what was expected, based on regression of harvest as a function of hunter numbers and the fall food productivity index (Figure 7). This regression is particularly strong (and has accurately predicted previous harvests) when only the past 15 years are considered. However, for the quota zone, the actual harvest in 2017 was higher 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, 2012–2017. Highlighted values show a change from the previous year. BMUs 26 and 44 were divided into 27/28 and 46/47, respectively, in 2016.

BMU		2012	2013	2014	2015	2016		2017
						Before BMU split [a]	After BMU split	
12		300	200	200	150	150	150	125
13		400	250	250	250	250	250	225
22		100	50	50	50	50	50	50
24		300	200	200	200	200	200	175
25		850	500	500	500	500	500	400
26		550	350	350	350	325		
	27						250	225
	28						75	60
31		900	550	550	550	550	550	500
41		250	150	150	150	125	125	125
44		700	450	450	450	450		
	46						400	350
	47						50	40
45		200	150	150	150	250	250	175
51		1450	900	900	900	1000	1000	900
Total		**1650**	**3750**	**3750**	**3700**	3850	**3850**	**1465**

[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.

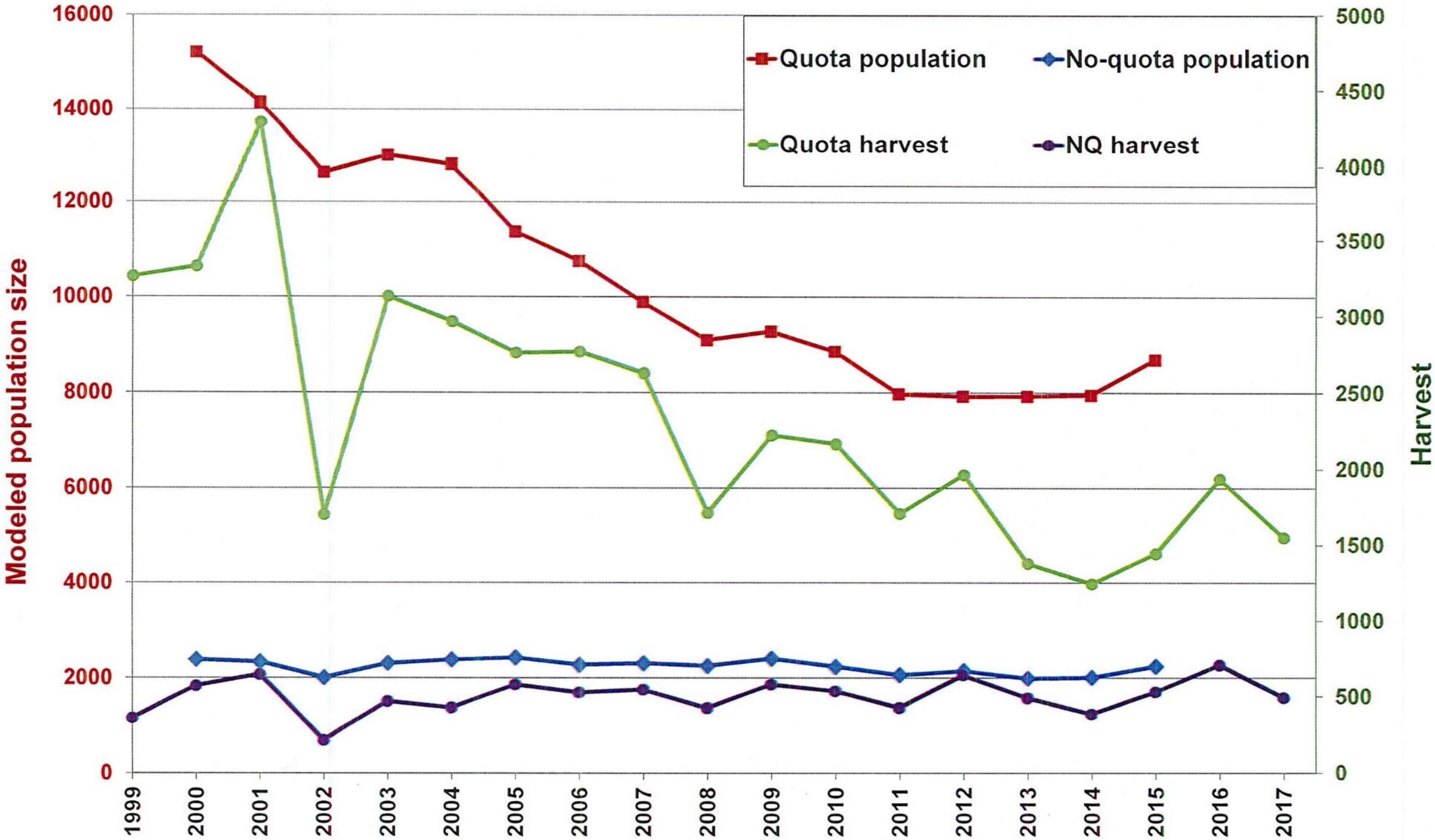

Figure 1. Population trends during 2000s derived from Downing reconstruction for quota and no-quota zones compared to respective harvests. Reconstruction-based estimates <2 years from the most recent harvest age data are unreliable (hence curves terminate in pre-hunt 2015). Population curves were scaled (elevated to account for non-harvest mortality) to match the lower curve in Figure 1 (i.e., the actual scale of the population estimates is not empirically-based).

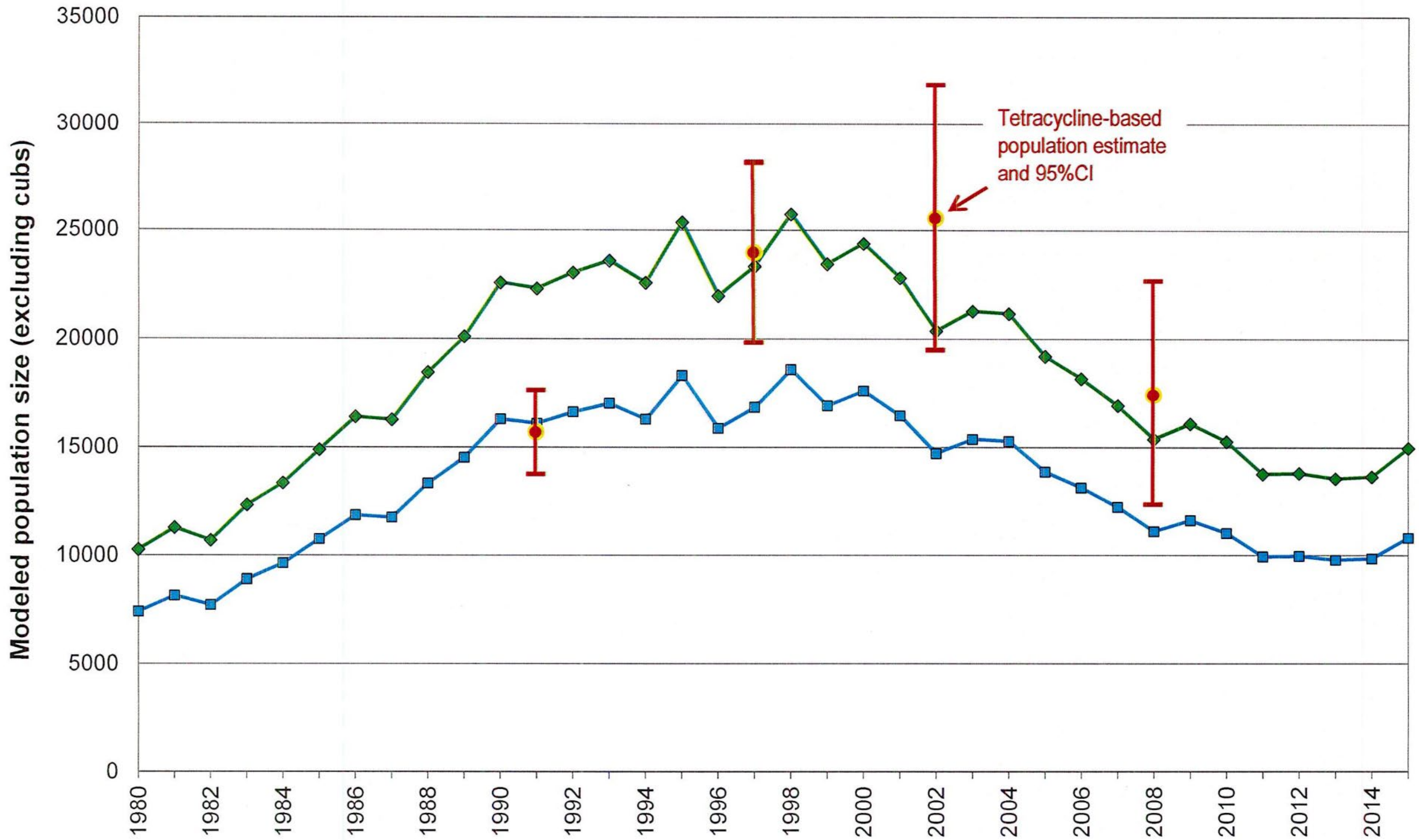

Figure 2. Statewide bear population trend (pre-hunt) derived from Downing reconstruction using harvest age structures, 1980–2017. Curves were scaled (elevated to account for non-harvest mortality) to various degrees to attempt to match the tetracycline-based mark–recapture estimates (2 such curves shown here). Estimates beyond 2015 are unreliable.

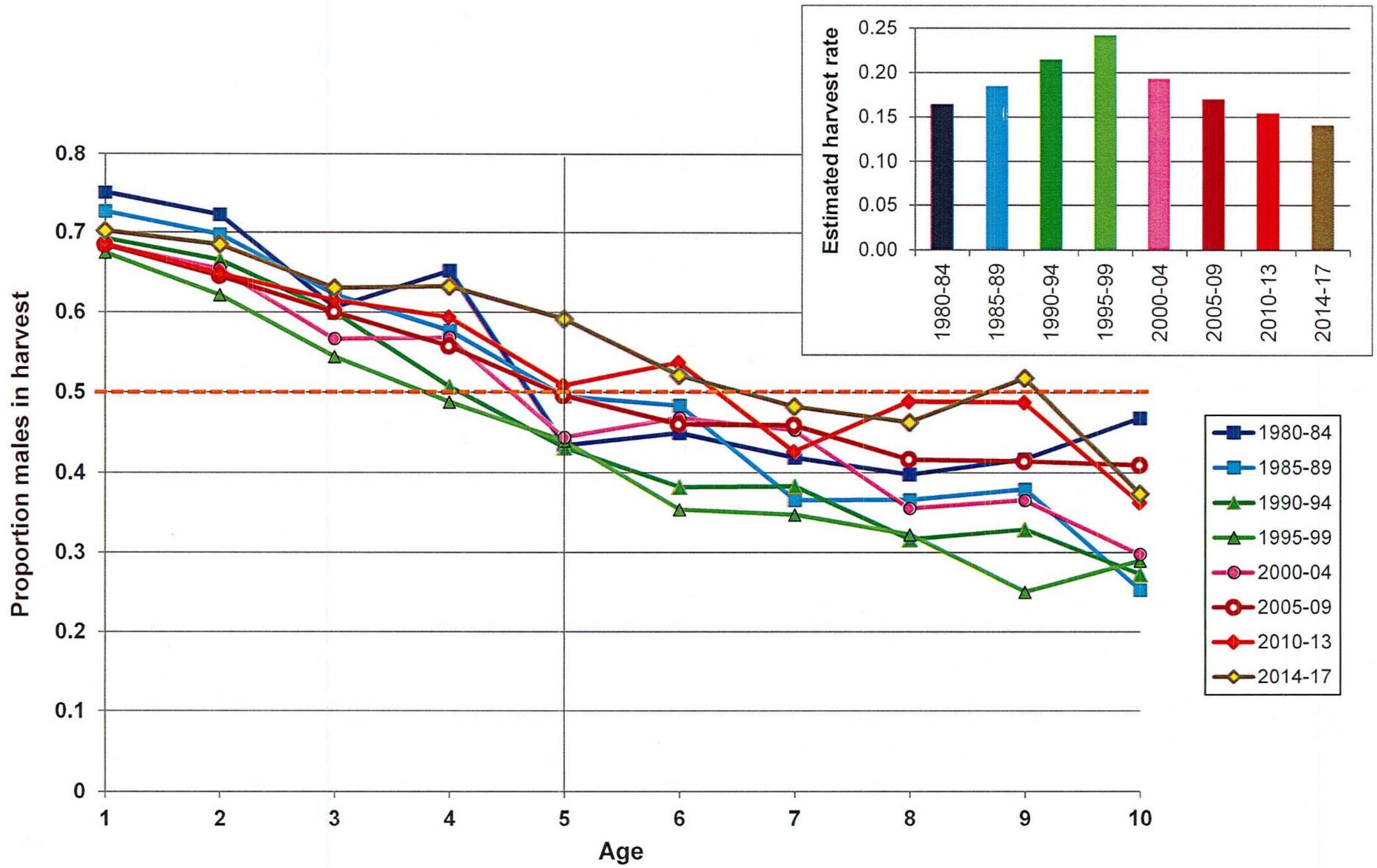

Figure 3. Trends in proportion of male bears in statewide harvest at each age, 1–10 years, grouped in 5-year time blocks, 1980–2017 (last 2 intervals are 4 years). Higher harvest rates result in steeper curves because males are reduced faster than females. Fitting a line to the data for each time block 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).

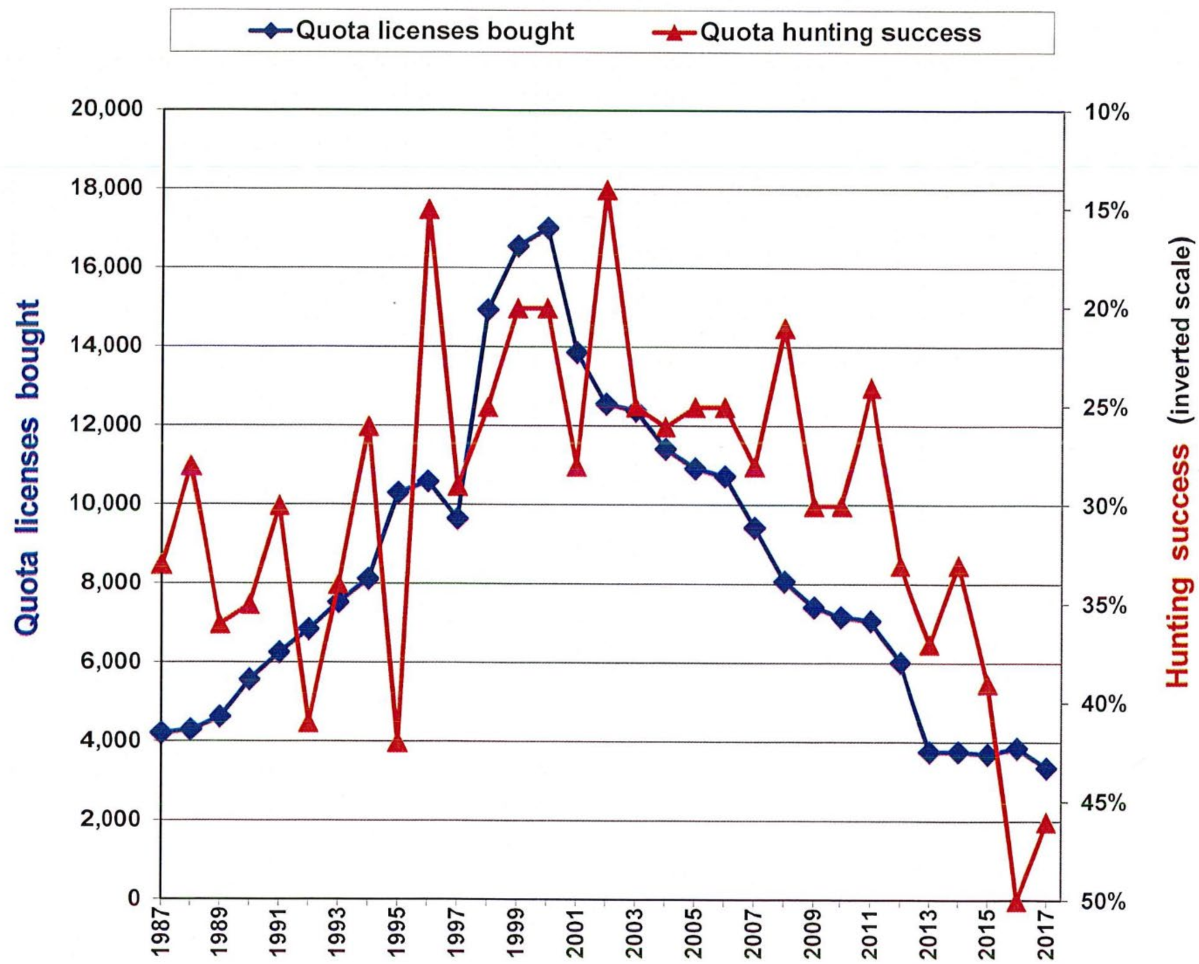

Figure 4. Relationship between licenses sold and hunting success (*note inverted scale*) in quota zone, 1987–2017 (no-quota zone first partitioned out in 1987). Number of licenses explains 48% of variation in hunting success during this period. Large variation in hunting success is also attributable to food conditions.

Table 2. Number of nuisance bear complaints registered by Conservation Officers and Wildlife Managers during 1997–2017, including number of nuisance bears killed and translocated, and bears killed in vehicular collisions.

	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017 [i]
Number of personnel participating in survey [a]	69	71	52	60	54	50	39	34	42	46	46	37	51	40	34	56	63	64	61	55	86 (51,35)
Complaints examined on site	661	226	189	105	122	75	81	75	61	57	63	59	65	70	37	113	69	79	97	118	71 (22,49)
Complaints handled by phone [b]	2196	743	987	618	660	550	424	507	451	426	380	452	535	514	396	722	623	570	840	780	644 (450,194)
Total complaints received	2857	969	1176	723	782	625	505	582	512	483	443	511	600	584	433	835	692	649	937	898	715
• % Handled by phone	77%	77%	84%	85%	84%	88%	84%	87%	88%	88%	86%	88%	89%	88%	91%	86%	90%	88%	90%	87%	90%
Bears killed by: • Private party or DNR	93	31	25	25	22	12	13	25	28	11	21	22	23	22	9	16	24	26	45	53	22 (4, 18)
• Hunter before season [c]																					
– from nuisance survey	32	23	5	7	4	0	3	3	6	2	18	3	4	3	3	11	0	0	1	13	1
– from registration file	35	31	24	43	20	11	8	4	13	6	25	5	15	10	5	12	0	1	4	6	3
• Hunter during/after season [d]	4	3	0	1	1	0	0	0	1	0	0	0	0	0	0	0	1	0	1	1	1
• Hunter by Area 88 license [e]																					1
• Permittee [f]	7	11	7	2	6	4	6	1	5	4	5	1	3	5	0	0	1	0	3	0	0
Bears translocated	115	24	29	1	6	3	1	3	3	3	1	3	2	2	2	0	3	2	0	0	0
• % bears translocated [g]	17	11	15	1	5	4	1	4	5	5	2	5	3	3	5	0	4	3	0	0	0
Bears killed by cars [h]	52	61	60	39	43	26	25	16	22	18	20	27	18	28	15	33	32	28	47 h	27	9 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.

[i] 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).

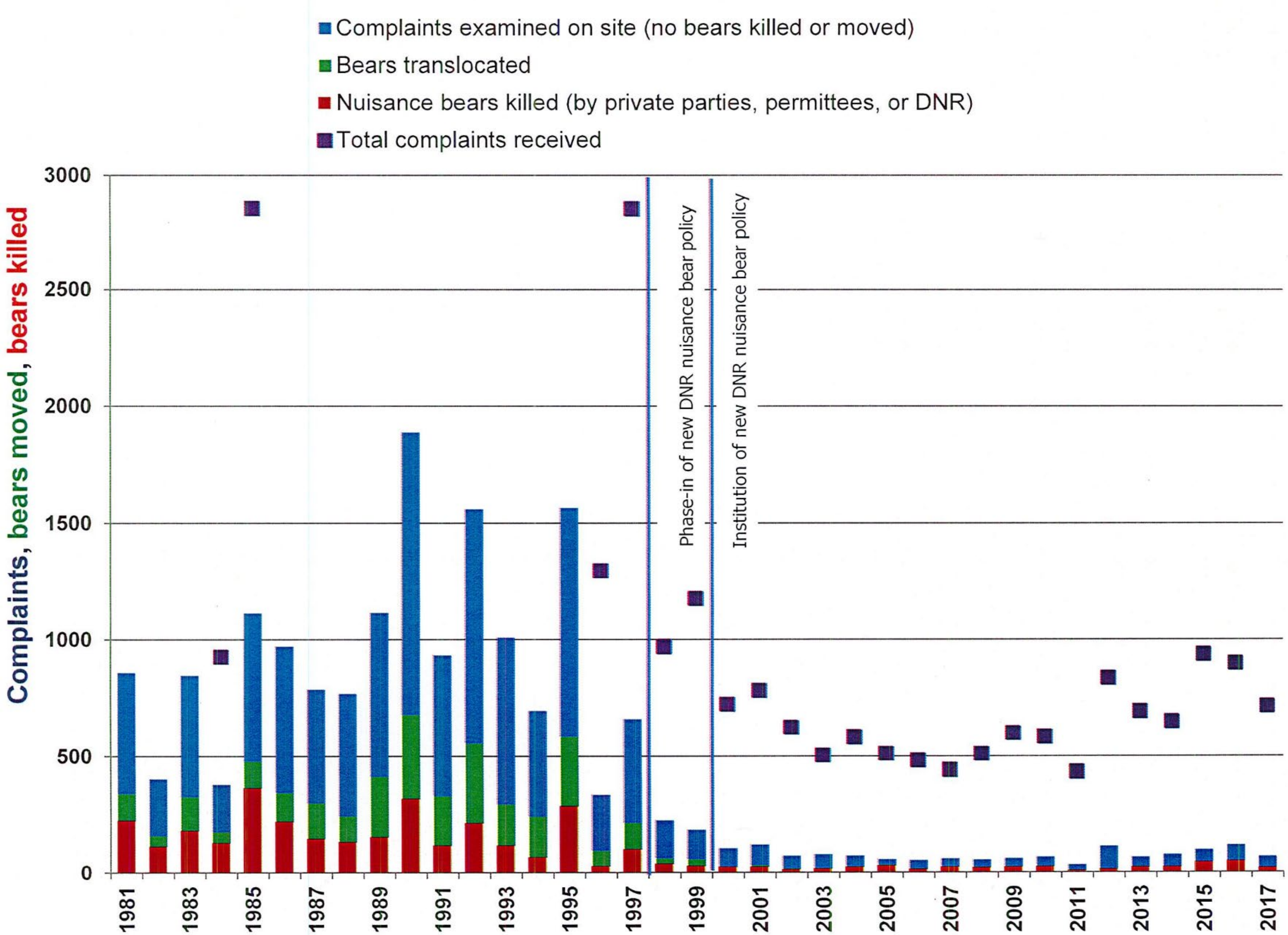

Figure. 5. Trends in nuisance bear complaints, and nuisance bears killed and moved, 1981–2017, showing dramatic effect of change in nuisance bear policy, and slight increasing trend over past decade.

Table 3. Number of bear hunting quota area permits available, 2012–2017. Highlighted values show a change from the previous year. BMUs 26 and 44 were divided into 27/28 and 46/47, respectively, in 2016.

BMU	2012	2013	2014	2015	2016 Before BMU split [a]	2016 After BMU split	2017
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22	100	50	50	50	50	50	50
24	300	200	200	200	200	200	175
25	850	500	500	500	500	500	400
26	550	350	350	350	325		
27						250	225
28						75	60
31	900	550	550	550	550	550	500
41	250	150	150	150	125	125	125
44	700	450	450	450	450		
46						400	350
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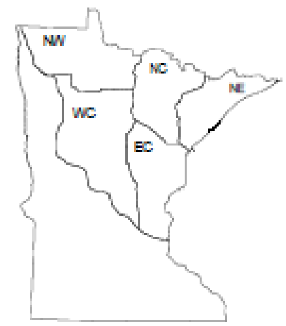
51	1450	900	900	900	1000	1000	900
Total	**1650**	**3750**	**3750**	**3700**	3850	**3850**	**1465**

[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 4. Regional bear food indices[a] in Minnesota's bear range, 1984–2017. Shaded blocks indicate particularly low (<45; pink) or high (≥70; green) values.

Year	NW	NC	NE	WC	EC	Range wide
	Survey Area					
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	64.4	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	47.1	56.3	41.3	64.8	45.5	48.7
2016	71.9	60.3	73.8	53.7	57.0	60.3
2017	57.2	55.7	52.7	62.2	68.3	58.9



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

Table 5. Regional mean index values[a] for bear food species in 2017 compared to the previous 33-year mean (1984-2016) in Minnesota's bear range. Shading indicates particularly high (green) or low (pink) fruit abundance relative to average (≥1 point difference for individual foods; ≥5 points difference for totals).

FRUIT	NW		NC		NE		WC		EC		Rangewide	
	33yr mean	2017 (*n* = 10[b])	33yr mean	2017 (*n* = 11)	33yr mean	2017 (*n* = 6)	33yr mean	2017 (*n* = 9)	33yr mean	2017 (*n* = 8)	33yr mean	2017 (*n* = 37)
SUMMER												
Sarsaparilla	4.5	4.3	5.7	4.9	5.2	2.7	4.5	3.3	5.3	5.9	5.0	4.3
Pincherry	3.3	3.1	4.5	3.9	4.2	4.3	3.7	3.1	3.7	4.3	3.8	3.7
Chokecherry	5.7	5.5	5.5	5.1	4.5	5.3	5.4	4.7	4.6	4.3	5.2	5.0
Juneberry	5.1	4.4	4.9	4.3	5.0	4.3	3.6	3.0	4.0	3.0	4.3	3.8
Elderberry	1.6	0.8	3.0	1.7	3.6	3.1	3.1	0.8	3.3	2.3	3.1	1.9
Blueberry	5.0	3.0	5.4	5.8	4.9	5.7	3.5	3.3	3.6	5.0	4.4	4.4
Raspberry	6.4	6.0	8.0	5.8	7.9	7.8	7.0	8.2	7.0	7.3	7.0	6.8
Blackberry	1.3	1.1	2.4	2.2	1.2	1.0	3.5	4.7	4.4	4.9	2.9	2.7
FALL												
Wild Plum	2.1	3.4	1.8	1.0	1.1	1.8	2.7	4.4	2.4	2.4	2.2	2.7
HB Cranberry	5.3	5.1	4.5	4.2	3.9	2.4	3.9	3.3	3.7	4.9	4.2	3.8
Dogwood	6.0	8.2	5.7	4.7	4.9	4.0	6.0	6.5	5.8	6.9	5.7	6.3
Oak	3.6	4.8	3.1	4.1	1.9	1.3	5.9	6.1	5.6	4.9	4.4	4.4
Mountain Ash	1.5	1.1	2.5	1.8	4.6	2.7	1.6	2.9	2.2	2.5	2.5	1.8
Hazel	6.3	6.4	7.3	6.2	7.0	6.3	7.9	7.9	7.5	9.7	7.2	7.3
TOTAL	57.9	57.2	64.3	55.7	59.7	52.7	62.3	62.2	63.3	68.3	61.9	58.9

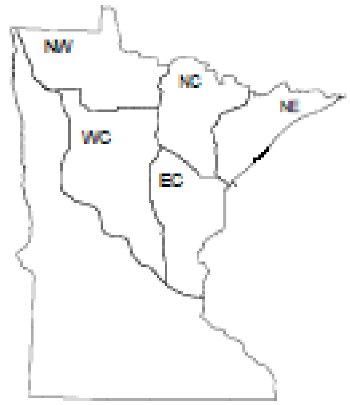
[a] Food abundance indices were calculated by multiplying species abundance ratings x fruit production ratings.

[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 6. Regional productivity index[a] for important fall foods (oak + hazel + dogwood) in Minnesota's bear range, 1984–2017. 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	5.3	5.3
2017	6.8	5.6	5.1	7.1	7.4	6.5



[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.

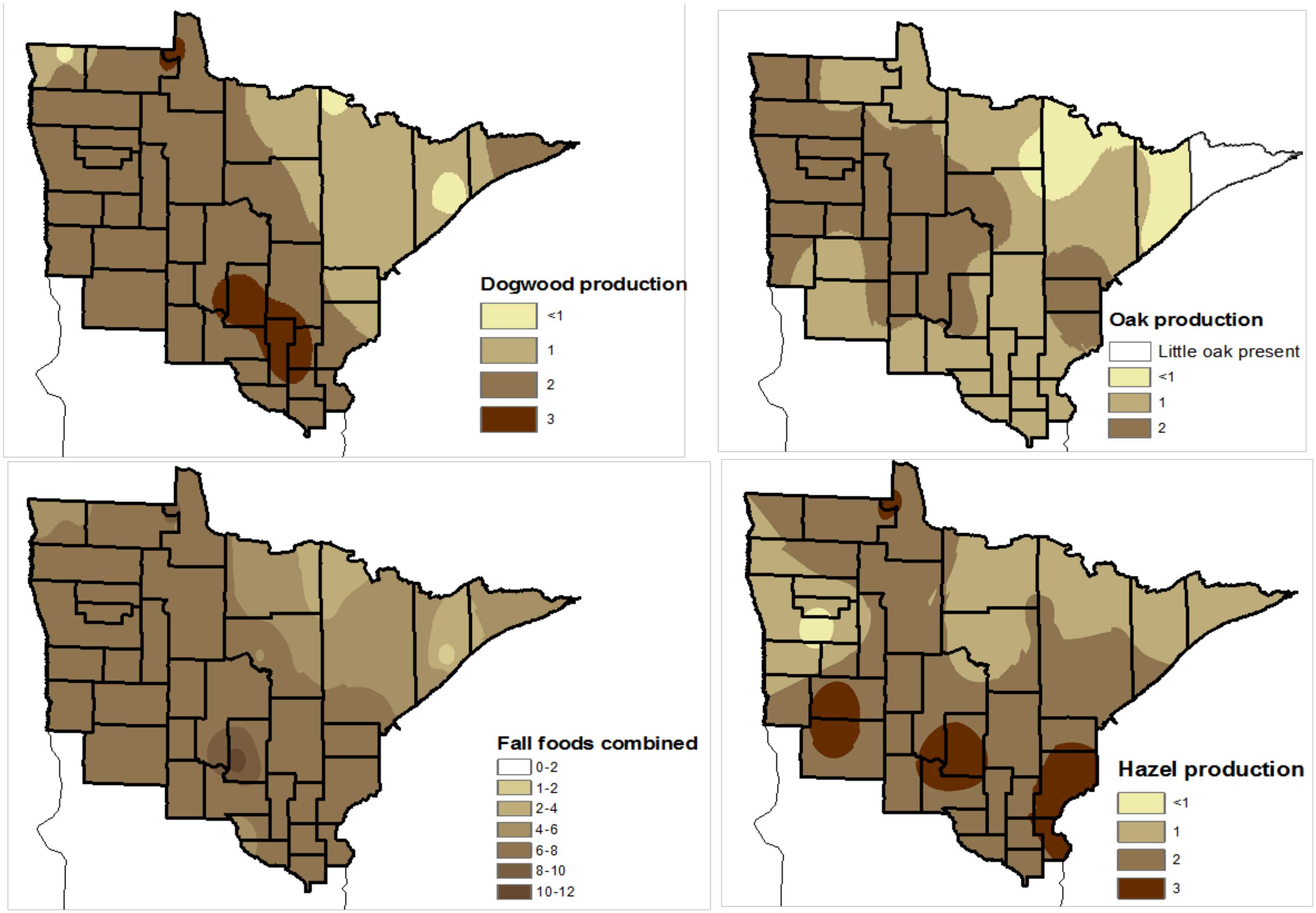


Figure 6. Production of fall bear foods (dogwood, oak, hazel) across Minnesota, 2017.

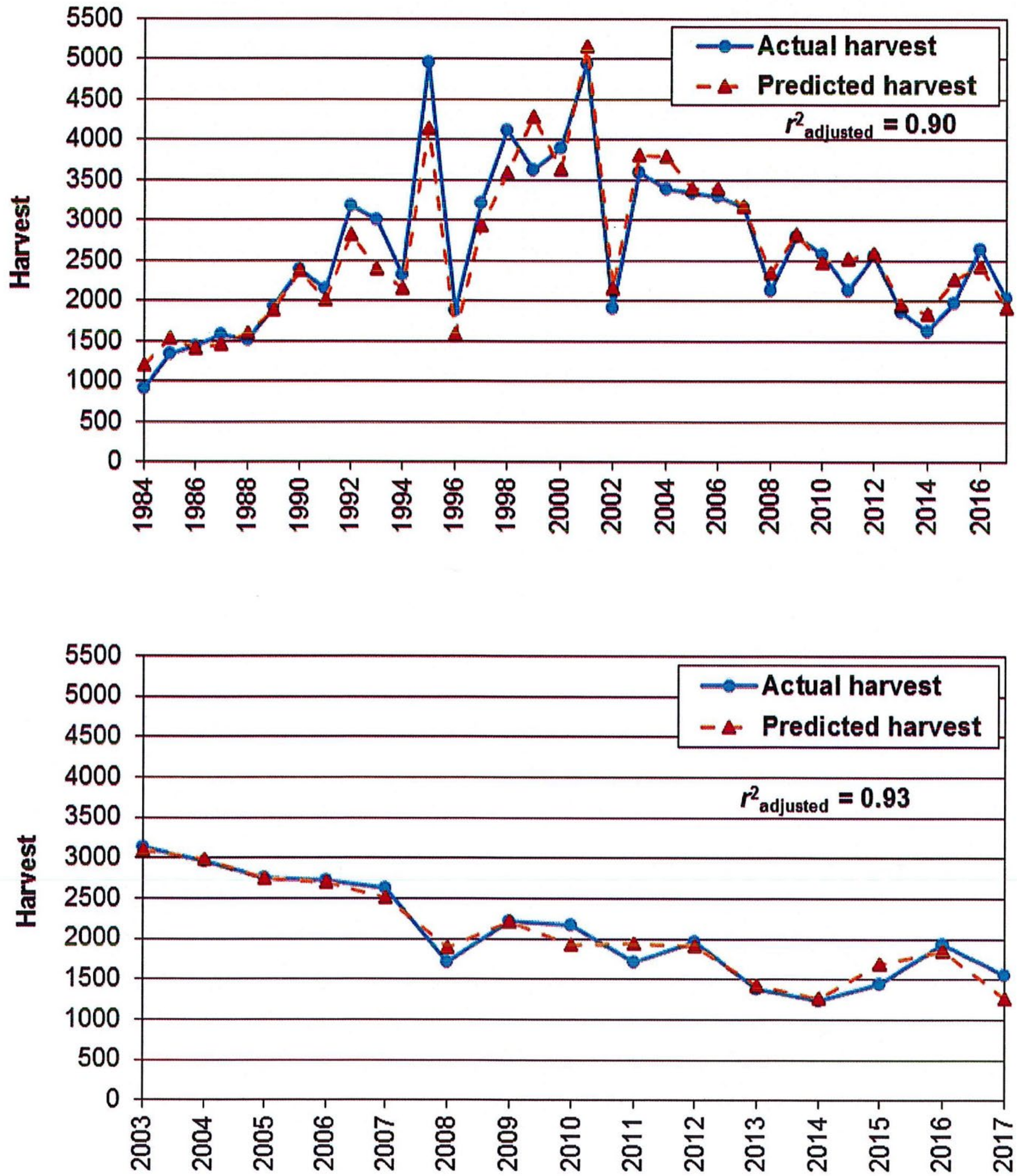


Figure 7. Number of bears harvested vs. number predicted to be harvested based on number of hunters and fall food production — top panel: statewide 1984–2017; bottom panel: quota zone only, most recent 15 years. Regression for the full dataset included an interaction term between food and hunters to better predict the drastic changes in harvest when fall foods were extremely high or low.

2018 MINNESOTA SPRING GROUSE SURVEYS

Charlotte Roy, Forest Wildlife Populations and Research Group

SUMMARY OF FINDINGS

The Minnesota DNR coordinates ruffed grouse (*Bonasa umbellus*) and sharp-tailed grouse (*Tympanuchus phasianellus*) surveys each spring with the help of wildlife staff and cooperating federal, tribal, and county agencies. In 2018, ruffed grouse surveys were conducted between 5 April and 15 May. Mean ruffed grouse drums per stop (dps) were 1.5 statewide (95% confidence interval = 1.3–1.7) and decreased (29%) from the previous year. High points in the population cycle occur on average every 10 years, and surveys this year indicate that the peak occurred last year, with counts similar to the previous peak in 2009. In more southern portions of ruffed grouse range, survey results were more similar to last year. Spring was very late in 2018, and it is possible that the drumming survey was conducted earlier than the peak in drumming this year. However, other factors likely also contributed to the decline in counts.

Sharp-tailed grouse surveys were conducted between 21 March and 20 May 2018, with 1,503 birds (males and birds of unknown sex) observed at 161 leks. The mean numbers of sharp-tailed grouse/lek were 7.3 (5.4–9.6) in the East Central (EC) survey region, 9.8 (8.8–10.9) in the Northwest (NW) region, and 9.3 (8.4–10.3) statewide. Comparisons between leks observed in consecutive years (2017 and 2018) indicated a 23% decline in birds/lek statewide ($t = 3.9$, $P = 0.0001$) and a 24% decline in the NW region ($t = 3.5$, $P = 0.0006$), but the 22% decrease in the EC region ($t = 1.8$, $P = 0.09$) was not statistically significant, likely due to the smaller number of leks surveyed in that region.

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, in combination with hunter harvest statistics, provide evidence that the ruffed grouse population cycles at approximately 10-year intervals.

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 16 cooperating organizations surveyed 122 routes between 5 April and 15 May 2018. Most routes (98%) were surveyed between 22 April and 15 May, with a median survey date of May 3, which is similar to last year (May 3) and the median survey date for the most recent 10 years. Excellent (58%), Good (35%), and Fair (7%) survey conditions were reported for 119 routes reporting conditions.

Statewide counts of ruffed grouse drums averaged 1.5 dps (95% confidence interval = 1.3–1.8 dps) during 2018 (Figure 3). Drum counts were 1.7 (1.5–2.1) dps in the Northeast (n = 101 routes), 1.0 (0.5–1.5) dps in the Northwest (n = 8), 0.9 (0.4–1.4) dps in the Central Hardwoods (n = 12), and 0.9 (0.5–1.4) dps in the Southeast (n = 7) regions (Figure 4a-d). Statewide drum counts decreased (29%) from last year. The ruffed grouse population was in the increasing phase of the 10-year cycle and was expected to peak this year or next year. Although peaks in the cycle average 10 years, they have occurred slightly before or after 10 years. Surveys this year indicate the peak occurred last year. Some portion of the decline might have been due to the very late spring in 2018, and a lack of synchrony between calendar date and the peak of drumming activity this year. However, poor synchrony is unlikely to entirely explain the drop in counts this year.

Sharp-tailed Grouse

A total of 1,503 male sharp-tailed grouse and grouse of unknown sex were counted at 161 leks (Table 1) during 21 March to 20 May 2018. The statewide index value of 9.3 (8.4–10.3) grouse/lek was centrally located among values observed since 1980 (Figure 5). In the EC survey region, 220 grouse were counted on 30 leks, and 1,280 grouse were counted on 130 leks in the NW survey region. One lek of 3 birds was counted just south of the NW region in Clearwater County. The grouse/lek index was similar statewide and in both survey regions compared to 2017 (Table 1). Leks with $\geq$2 grouse were observed an average of 2.1 times. Counts at leks observed during both 2017 and 2018 were 23% lower in 2018 statewide ($t = 3.9$, $P = 0.0001$) and 24% lower in the NW region ($t = 3.5$, $P = 0.0006$), but the 22% decline in the EC region was not significant ($t = 1.8$, $P = 0.09$; Table 2), likely because fewer leks were surveyed in that region (Figure 6).

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. Although decreases in the population index were not statistically significant in the EC region, 25% fewer leks were detected in the EC region this year, which diminishes statistical power to detect differences. Furthermore, a loss of small leks would tend to maintain the average lek size, while comparisons of leks surveyed in successive years would tend to decline. This is the pattern that was observed. No sharp-tailed grouse were observed in surveys in Kanabec County this year, which is the first time this has occurred in recent history, although the number of birds have dwindled slowly in this area. Sharp-tailed grouse rely on habitats that require ongoing management (e.g., prescribed fire, mowing, and shearing) for maintenance. Obstacles to successfully completing management include funding, staffing shortages, equipment, weather, and landowner permission.

In the NW region, the number of leks counted, average lek size, and comparisons between leks surveyed in successive years were all lower in 2018. Continued monitoring will document whether the NW population will continue to be a stronghold for sharp-tailed grouse in the state. During 2016–2018, the DNR allowed the capture and translocation of sharp-tailed grouse from the NW region to supplement a population of sharp-tailed grouse at Moquah Barrens in Wisconsin. The impact of this effort, if any, has not yet been examined, but only leks with $\geq$15 birds were trapped to try to safeguard against negative impacts.

ACKNOWLEDGEMENTS

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, Grand Portage, Leech Lake, Red Lake, and White Earth Reservations; 1854 Treaty Authority; Blandin Paper, Agassiz National Wildlife Refuge (U.S. Fish & Wildlife Service); 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, Cambridge, Cloquet, Crookston, Karlstad, International Falls, Tower, Thief River Falls, and Thief Lake work areas, staff and volunteers at Red Lake and Roseau River WMAs, and partners at Agassiz National Wildlife Refuge for participating in sharp-tailed grouse surveys. Pam Coy, Alex Elliott, and Dan Ruka 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

[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	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

[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.

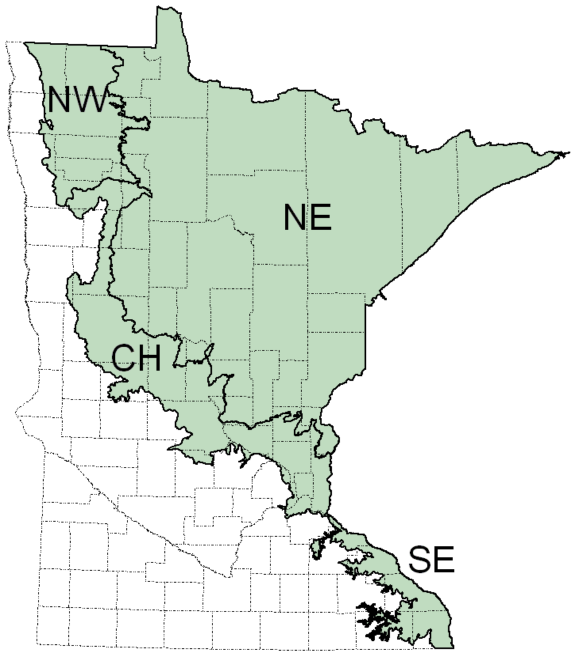


Figure 1. Survey regions for **ruffed grouse** in Minnesota. Northwest (NW), Northeast (NE), Central Hardwoods (CH), and Southeast (SE) survey regions are depicted relative to county boundaries (dashed lines) and influenced by the Ecological Classification System.

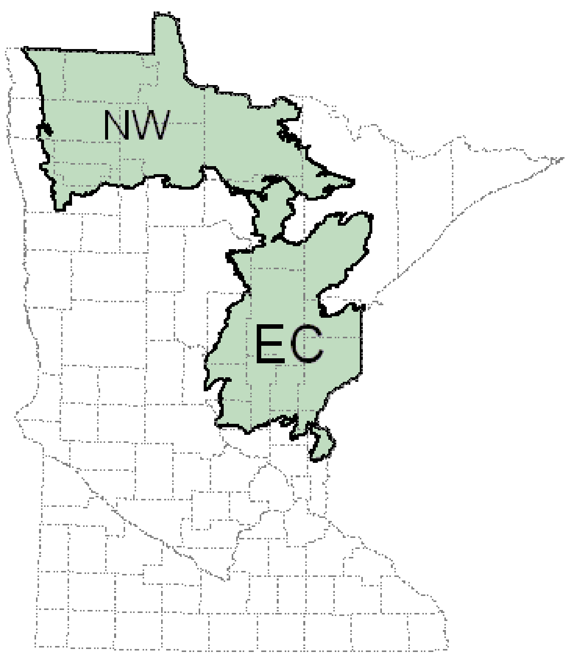


Figure 2. Survey regions for **sharp-tailed grouse** in Minnesota. Northwest (NW) and East Central (EC) survey regions are depicted relative to county boundaries (dashed lines) and influenced by Ecological Classification System Subsections boundaries.

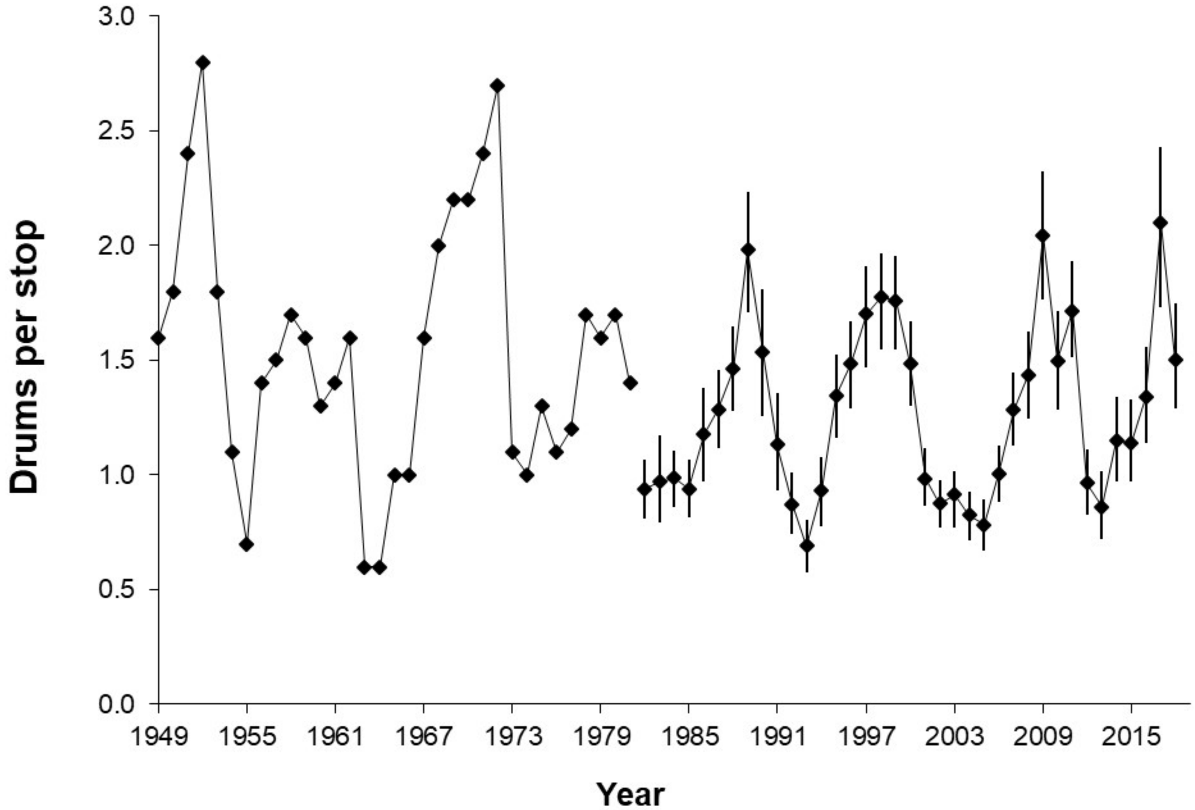
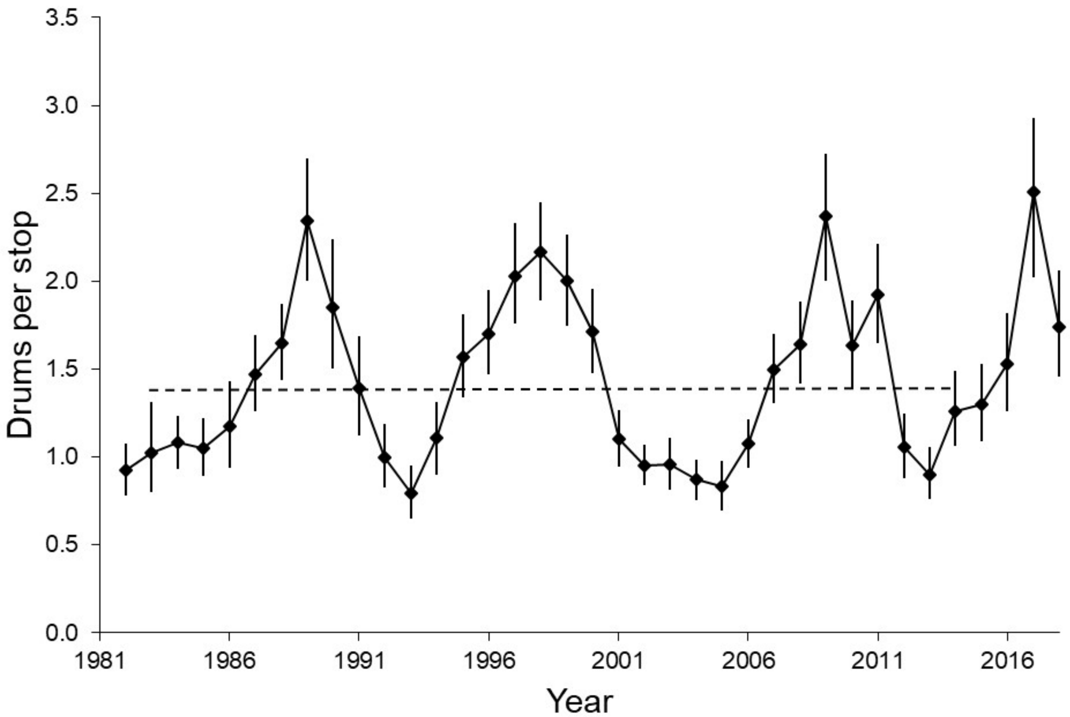
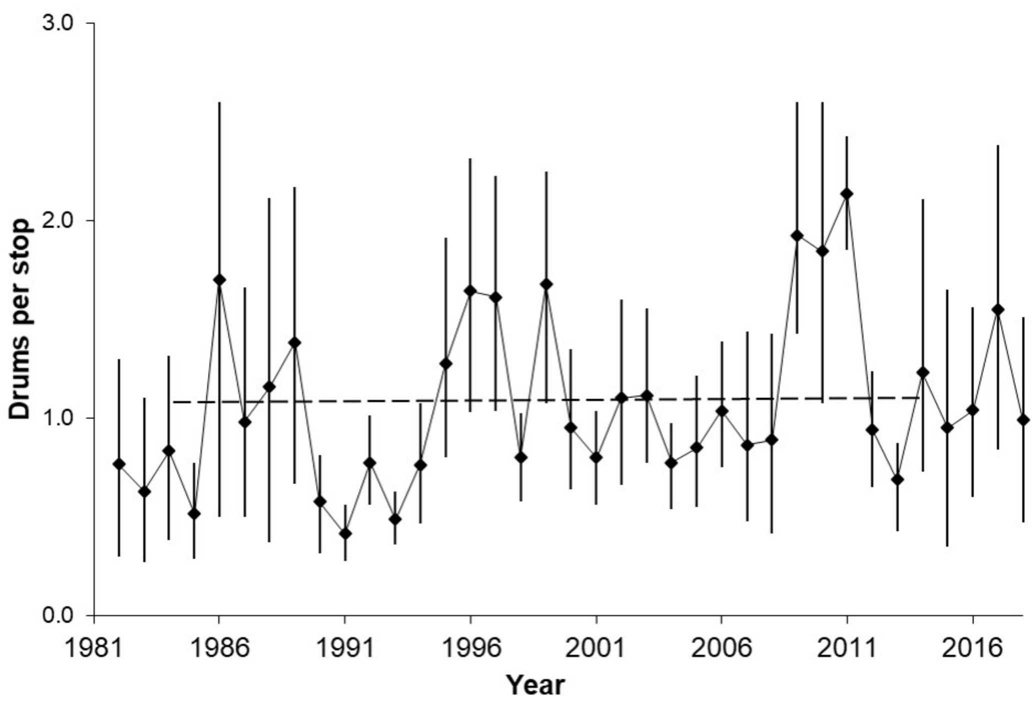


Figure 3. Statewide ruffed grouse population index values in Minnesota. Bootstrap (95%) confidence intervals (CI) are provided after 1981, but different analytical methods were used prior to this and thus CI are not available for earlier years. The difference between 1981 and 1982 is biological and not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

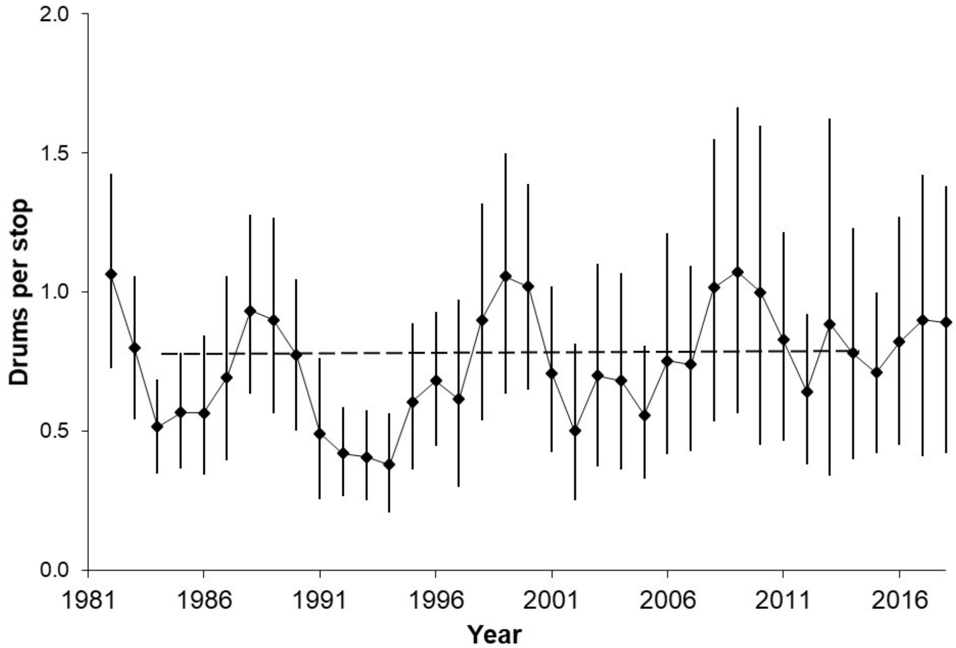
a.


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.


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

c.



d.

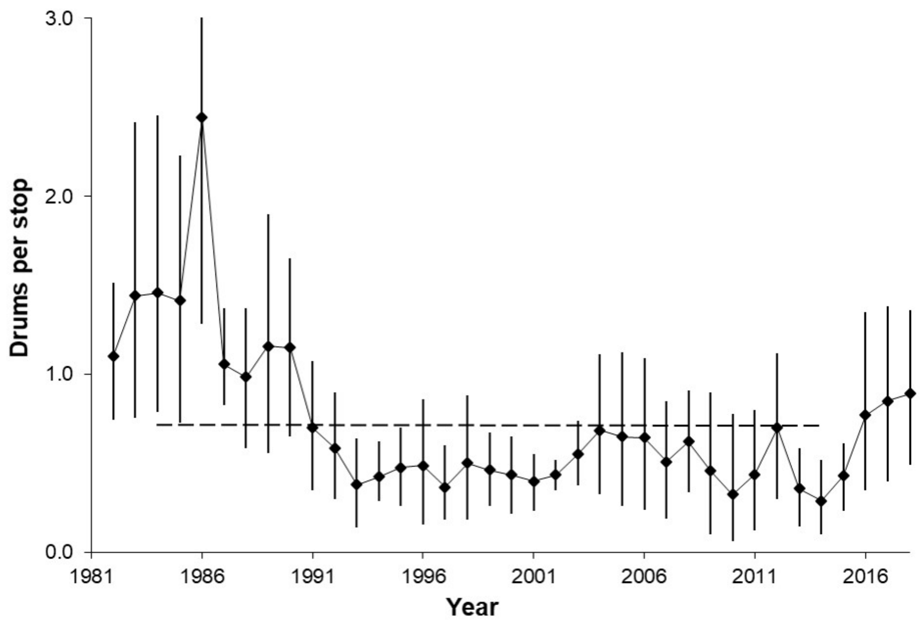


Figure 4a,b,c,d. Ruffed grouse population index values in the **Northeast** (a), **Northwest** (b), **Central Hardwoods** (c), and **Southeast** (d) survey regions of Minnesota. The mean for 1984-2014 is indicated by the dashed line. Bootstrap (95%) confidence intervals are provided for each mean. In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

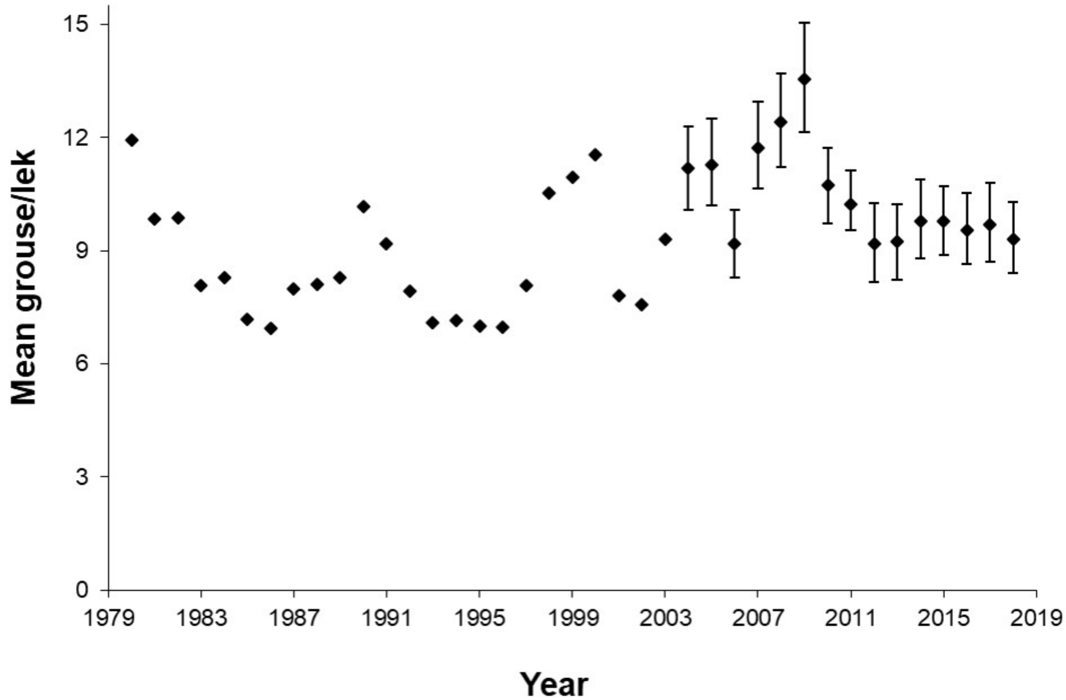


Figure 5. **Sharp-tailed grouse** counted in spring lek surveys statewide in Minnesota during 1980–2018. Bootstrap (95%) confidence intervals are provided for recent years. Annual means are not connected by lines because the same leks were not surveyed every year.

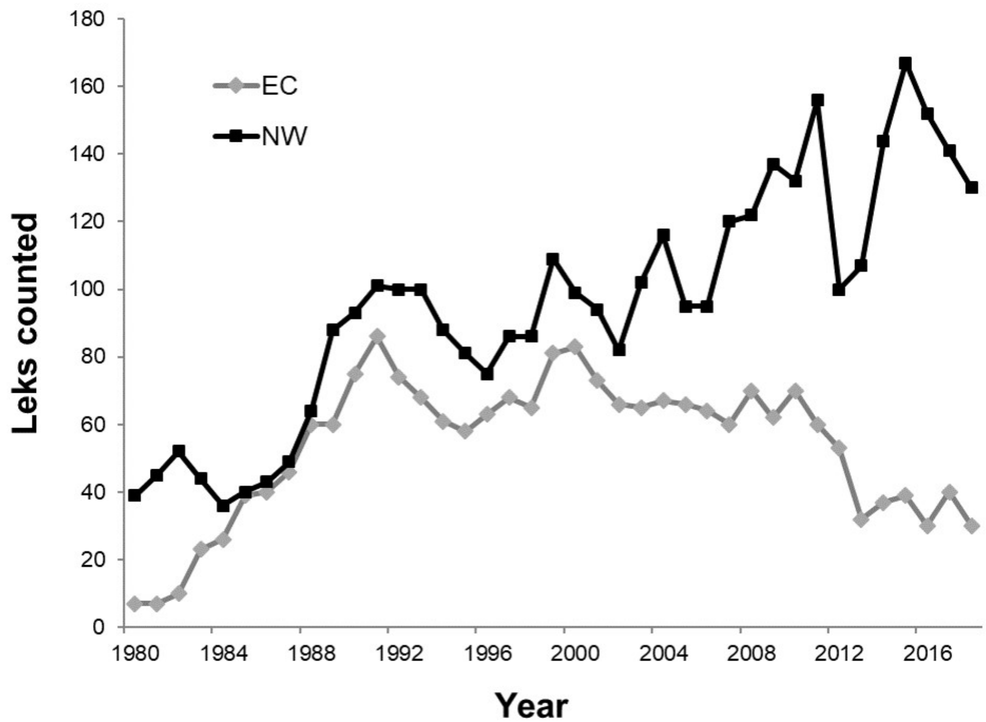


Figure 6. The number of **sharp-tailed grouse** leks counted in spring lek surveys in the Northwest (NW) and East Central (EC) survey regions of Minnesota during 1980-2018.

2018 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 2018. Observers located 59 booming grounds and counted 630 males and birds of unknown sex in the survey blocks. They located 148 booming grounds,1,354 male prairie-chickens, and 164 birds of unknown sex throughout the prairie-chicken range. Estimated densities of 0.09 (0.06–0.11) booming grounds/km^2 and 10.7 (8.6–12.8) 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), but have declined since the standardized survey began in 2004. 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, 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 1998-2006. Today, the beach ridges of glacial Lake Agassiz hold most of Minnesota's prairie-chickens, but their populations do extend southward (Figure 1). Hunting was re-opened using a limited-entry season in 2003, and 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 late-March through May. Each survey block was nonrandomly selected so that surveys would be conducted in areas where habitat was expected to be good (i.e., grassland was relatively abundant) and leks were known to occur. Each observer attempted to find and survey each booming ground repeatedly in his/her assigned block, which comprised 4 sections of the Public Land Survey (approximately 4,144 ha). Observers obtained multiple counts at each booming ground in the morning because male attendance at leks varies throughout the season and throughout the day.

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

In the analysis, I used counts of males and unknowns at each booming ground but not females. Leks were defined as having ≥2 males, so observations of single males were not counted as leks. Data were summarized by hunting permit area and spring survey block. The survey blocks were separated into a core group and a periphery group for analysis. The core group had a threshold density of approximately 1.0 male/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.

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 26 March and 14 May 2018. Observers located 148 booming grounds and observed 1,354 male prairie-chickens and 164 birds of unknown sex within and outside survey blocks (Table 1). These counts represent a minimum number of prairie-chickens in Minnesota during 2018, but because survey effort outside of survey blocks is not standardized among years, these counts should not be compared among years or permit areas.

Within the standardized survey blocks, 630 males and birds of unknown sex were counted on 59 booming grounds during 2018 (Table 2). These counts are the lowest since the standardized survey began in 2004 and 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 2.2 times (median = 2), with 39% of booming grounds observed just once. These counts should not be regarded as estimates of abundance because detection probabilities of leks and birds were not estimated. However, if 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.

Table 1. Minimum abundance of prairie-chickens within and outside hunting permit areas in Minnesota during spring 2018. Lek and bird counts are not comparable among permit areas or years.

Permit Area	Area (km^2)	Leks	Males	Unk[a]
803A	1,411	10	71	0
804A	435	5	19	0
805A	267	22	176	10
806A	747	14	85	40
807A	440	21	234	7
808A	417	20	303	0
809A	744	12	180	0
810A	505	9	64	27
811A	706	8	39	27
812A	914	7	13	35
813A	925	6	26	18
PA subtotal	7,511	134	1,210	164
Outside PAs[b]	NA[c]	14	144	0
Grand total	NA[c]	148	1,354	164

[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.

Densities of prairie-chickens in the 10 core survey blocks were 0.10 (0.07–0.13) booming grounds/km^2 and 11.9 (9.4–14.5) males/booming ground (Table 2, Figure 2). In the 7 peripheral survey blocks, densities were 0.07 (0.03–0.10) booming grounds/km^2 and 8.1 (4.4–11.7) males/booming ground. The density of 0.08 (0.06–0.11) booming grounds/km^2 in all survey blocks during 2018 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 10.7 (8.6–12.8) males/booming ground in all survey blocks during 2018 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 is underway in collaboration with researchers at the Minnesota Cooperative Fish and Wildlife Research Unit at the University of Minnesota.

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, Becky Ekstein, and Jason 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, Cody Townsend, Ben Walker, Erin Lentz, Traver Fields, and Stacy Salvevold; and numerous additional volunteers participated, including Dan Svedarsky, Doug Wells, Tom Kucera, Jon Voz, Ross Hier, Phil Doll, and Doug Hedtke. Bemidji State University faculty member, Brian Hiller, and students also assisted with surveys this year. This survey was funded in part by the Wildlife Restoration (Pittman-Robertson) Program W-69-S-13 Project #16. Mike Larson provided assistance and comments which improved this report.

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

			2018		Change from 2017[a]	
Range[b]	Survey Block	Area (km^2)	Booming grounds	Males[c]	Booming grounds	Males[c]
Core	Polk 1	41.2	4	35	-2	-22
	Polk 2	42.0	5	65	1	20
	Norman 1	42.0	1	8	-1	-7
	Norman 2	42.2	5	30	-1	-13
	Norman 3	41.0	7	50	3	14
	Clay 1	46.0	6	104	-1	4
	Clay 2	41.0	2	55	0	-21
	Clay 3	42.0	5	86	-1	25
	Clay 4	39.0	1	3	-2	-16
	Wilkin 1	40.0	4	41	0	-2
	Core subtotal	415.0	40	477	-2	-18
Periphery	Mahnomen	41.7	3	62	0	23
	Becker 1	41.4	7	48	1	-3
	Becker 2	41.7	2	5	-3	-18
	Wilkin 2	41.7	1	3	0	-2
	Wilkin 3	42.0	3	13	-1	-20
	Otter Tail 1	41.0	2	11	0	2
	Otter Tail 2	40.7	1	11	0	3
	Periphery subtotal	290.6	19	153	-3	-15
Grand total		705.5	59	630	-5	-33

[a] The 2017 count was subtracted from the 2018 count, so positive values indicate increases.

[b] Survey blocks were categorized as within the core or periphery of the Minnesotaprairie-chicken range based upon bird densities and geographic location.

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

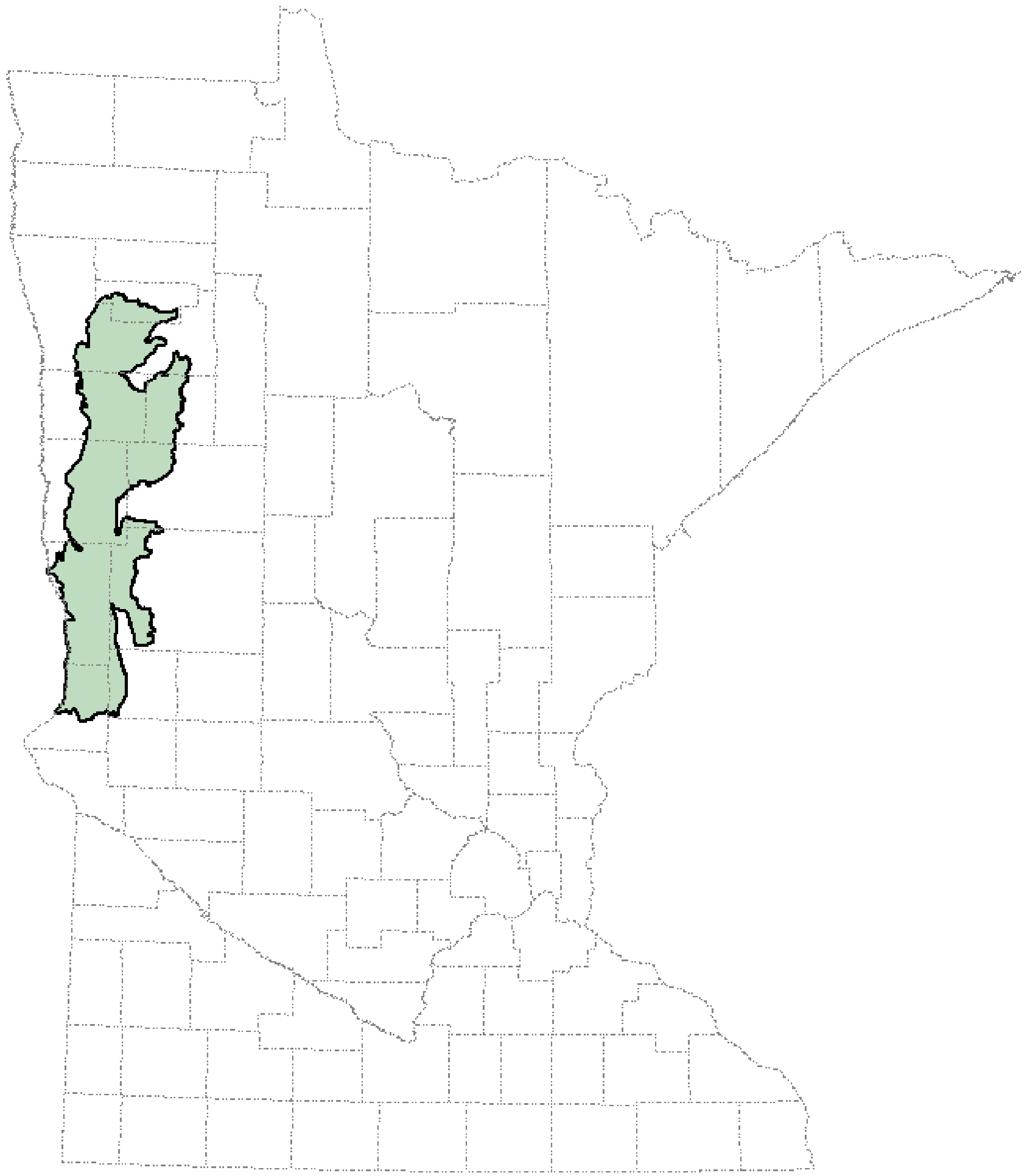


Figure 1. Primary greater prairie-chicken range in Minnesota (shaded area) relative to county boundaries. The range boundary was based on Ecological Classification System Land Type Associations and excludes some areas known to be occupied by prairie-chickens.

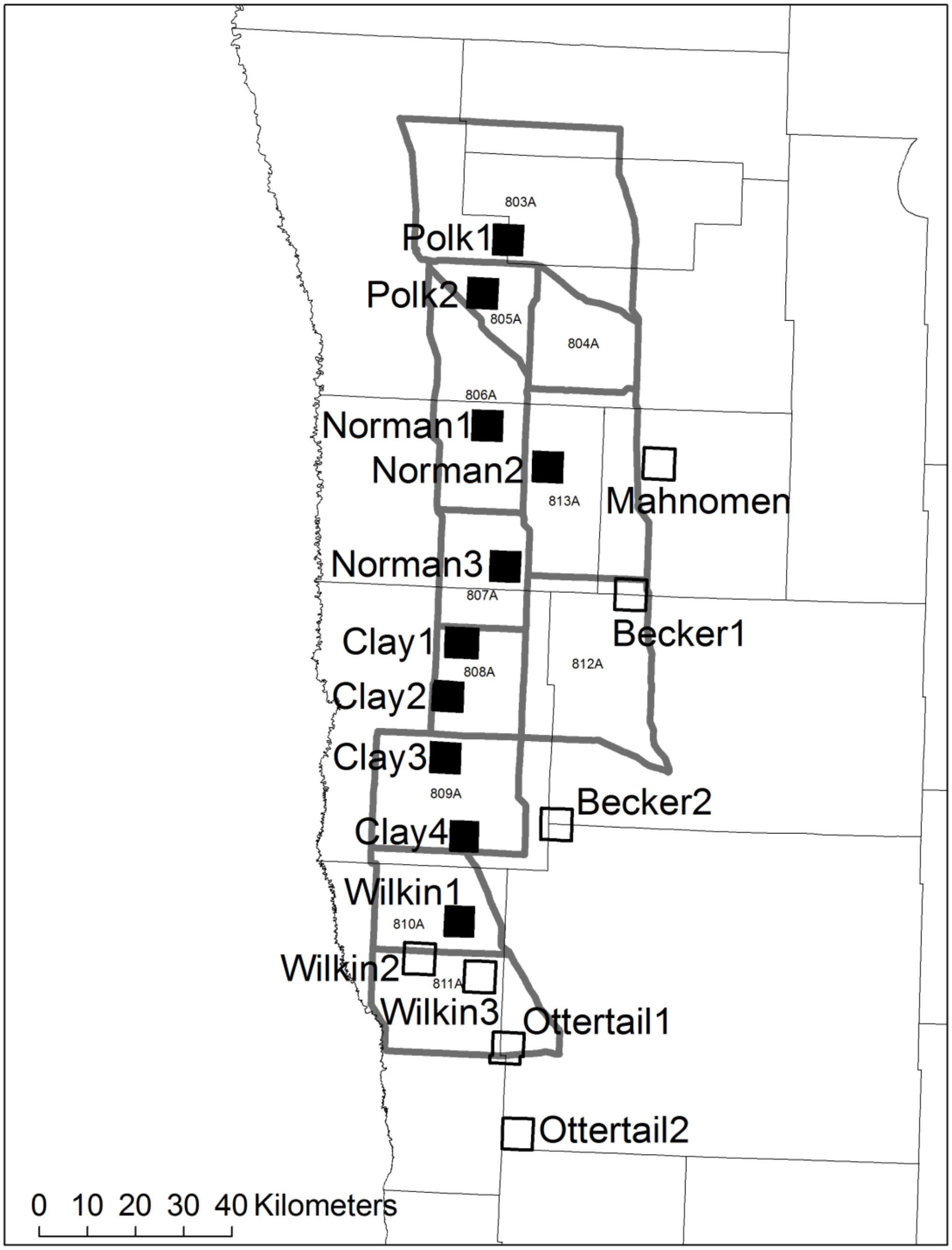


Figure 2. Prairie-chicken lek survey blocks (41 km^2, labeled squares) and hunting permit areas (thick grey lines) in western Minnesota. Survey blocks were either in the core (black) or periphery (white) of the range with a threshold of 1.0 male/km^2 in 2010, and were named after their respective counties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

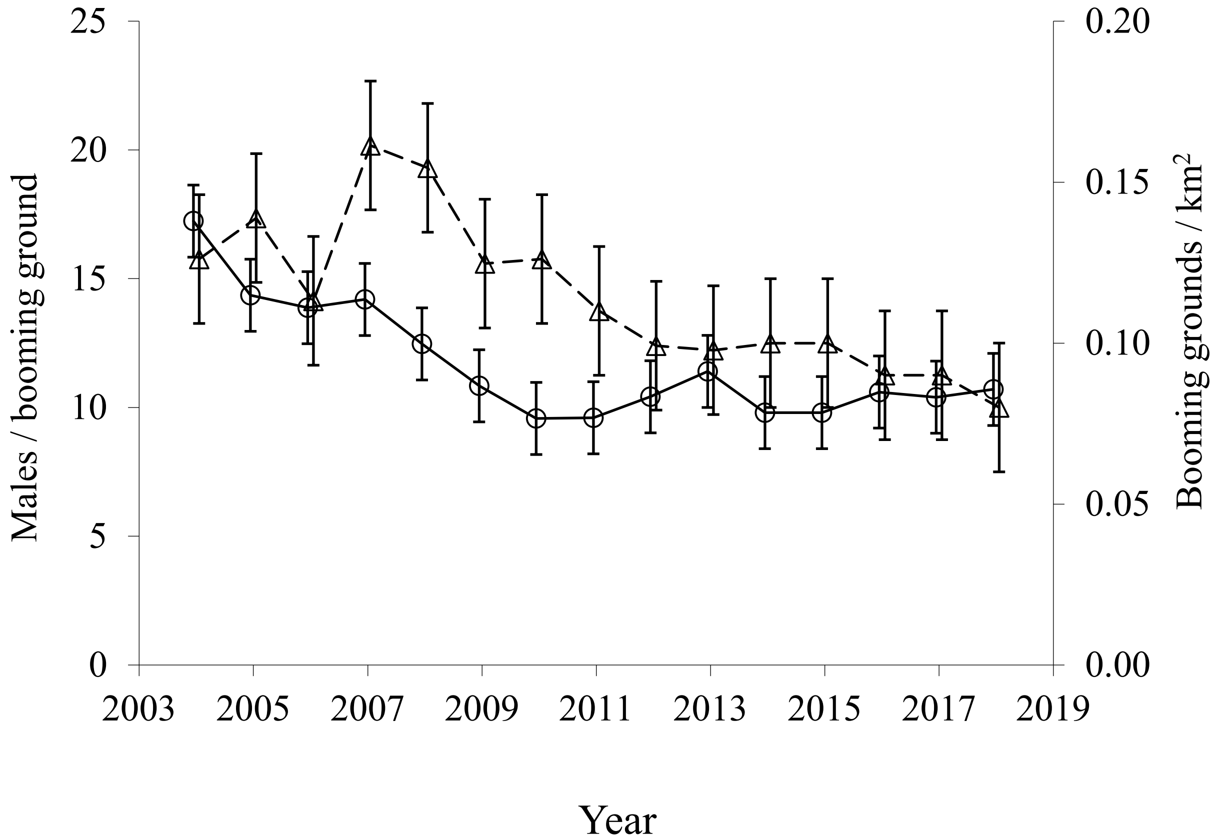


Figure 3. Mean prairie-chicken males/booming ground (circles connected by solid line) and booming grounds/km^2 (triangles connected by dashed line) in survey blocks in Minnesota with 95% confidence intervals.

DEPARTMENT OF NATURAL RESOURCES

2018 NW MINNESOTA ELK SURVEYS

Doug Franke, Area Wildlife Manager, Thief River Falls

INTRODUCTION

Minnesota DNR FAW staff used fixed-wing aircraft (Cessna 185 Skywagon) to conduct aerial elk surveys for the Lancaster and Grygla elk herds. We were also able to complete the Caribou-Vita survey again this year since Manitoba Wildlife secured funding to complete their aerial elk survey the same day on the Canadian side. We used MN DNR Forestry's Quest Kodiak turboprop airplane to complete the border survey. The MN DNR fixed-wing aircraft crew followed the same predetermined transects used in 2017—transects are 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. Antlered elk were recorded as either branch antlered or spike bulls.

The surveys were completed between February 5th and March 11th, 2018. Snow depths and conditions varied across the elk ranges. Snow conditions were considered good in the Grygla area and fair to good in the Lancaster and Caribou-Vita survey blocks. Snow depths ranged from 12 to 15 inches in the Grygla and Caribou-Vita survey blocks and 8 to 10 inches in the Lancaster survey block. Weather conditions were average for this time of the year with temperatures ranging from a low of -15°F to a high of 34°F and mostly clear skies. We had a weather delay of one day during the Lancaster survey.

We waited again this year to complete the Caribou-Vita survey block since Manitoba Wildlife staff indicated that they were also planning to survey elk on the Canadian side in late February to early March. The surveys for both the Canadian and US border areas were completed on March 11th 2018 within a two hour period of each other.

Grygla Survey Block

This survey started on February 5th and was completed on February 6th, 2018. The area surveyed was the same 133 mi^2 area that has been used the past two years. Total aircraft engine time to complete this survey (takeoff to landing) was 11.7 hours. The entire survey area received a light snowfall the day before which made for good survey conditions. The fixed-wing crew recorded elk at 5 separate locations within the survey boundary. Total elk observed was 15 and included: 7 antlerless (cows/calves) and 8 bulls (5 branch antlered and 2 spike bulls).

Thief Lake WMA staff believed three of the seven antlerless elk were calves based upon ground observations during the summer.

Lancaster Survey Block—Water Tower and Percy WMA herds

This survey started on February 12th and after a one-day weather delay was completed on February 14th, 2018. The area surveyed was the same 167 mi^2 area that has been flown the past several years. Total aircraft time to complete the survey was 15.2 hours (takeoff to landing). The fixed-wing crew recorded elk at 5 separate locations within the survey boundary. Total elk recorded within the Lancaster survey block was 75 and included: 57 antlerless (cows/calves) and 18 bulls (13 branch antlered and 5 spike bulls). The Water Tower group had 35 antlerless elk—there were 7 branch antlered bulls located in the same woodlot as the antlerless group. The Percy WMA antlerless herd (22 elk) along with 6 branch antlered bulls and 3 spike bulls were observed approximately four miles northwest of the Percy WMA. One spike bull was located on the western edge of the Percy WMA.

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

Minnesota DNR and Manitoba Wildlife staff successfully coordinated a joint aerial elk survey for the survey areas close to the US/Canadian border. This survey started and was completed on March 11th, 2018. The area surveyed in MN was the same 35.5 mi^2 area that has been surveyed the past few years. Manitoba also flew the same survey blocks as they did in 2017. Total aircraft time to complete the DNR survey was 3.0 hours (takeoff to landing). The fixed-wing crew recorded elk at one location (6 branch antlered bulls and 1 spike bull) within survey boundary. A majority of this herd was expected to be north of the Minnesota border—this assumption was as confirmed with the Manitoba aerial elk survey results. Manitoba completed an aerial survey for the Vita area the next day on March 12th.

Manitoba Wildlife staff used a Jet Ranger helicopter to fly north/south transects within predetermined survey blocks that covered a broad area along the border. They recorded 80 elk near the US/Canadian border and another 46 elk slightly north of Vita. Table 2 details the age/sex breakdown for these two populations in Canada.

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 2018 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 pilots Chris Lofstuen, Bob Geving, and Luke Ettl who provided safe flying for all of us. Observers this year included: Kyle Arola (Thief Lake Assistant 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 2014 and 2018 for the Lancaster, Caribou-Vita, and Grygla herds.

	Lancaster					Caribou-Vita (US side of border)					Grygla				
	2014	2015	2016	2017	2018	2014	2015	2016	2017	2018	2014	2015	2016	2017	2018
Spike bull	3	2	6	2	5	10	5	0	0	1	2	3	2	4	2
Branch antlered bull	14	16	12	14	13	7	17	6	1	6	4	6	9	6	6
Total bulls	17	18	18	16	18	17	22	6	1	7	6	9	11	10	8
Antlerless	20	16	34	45	57	34	57	4	0	0	14	9	10	7	7
Total elk	37	34	52	61	75	51	79	10	1	7	20	18	21	17	15

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.

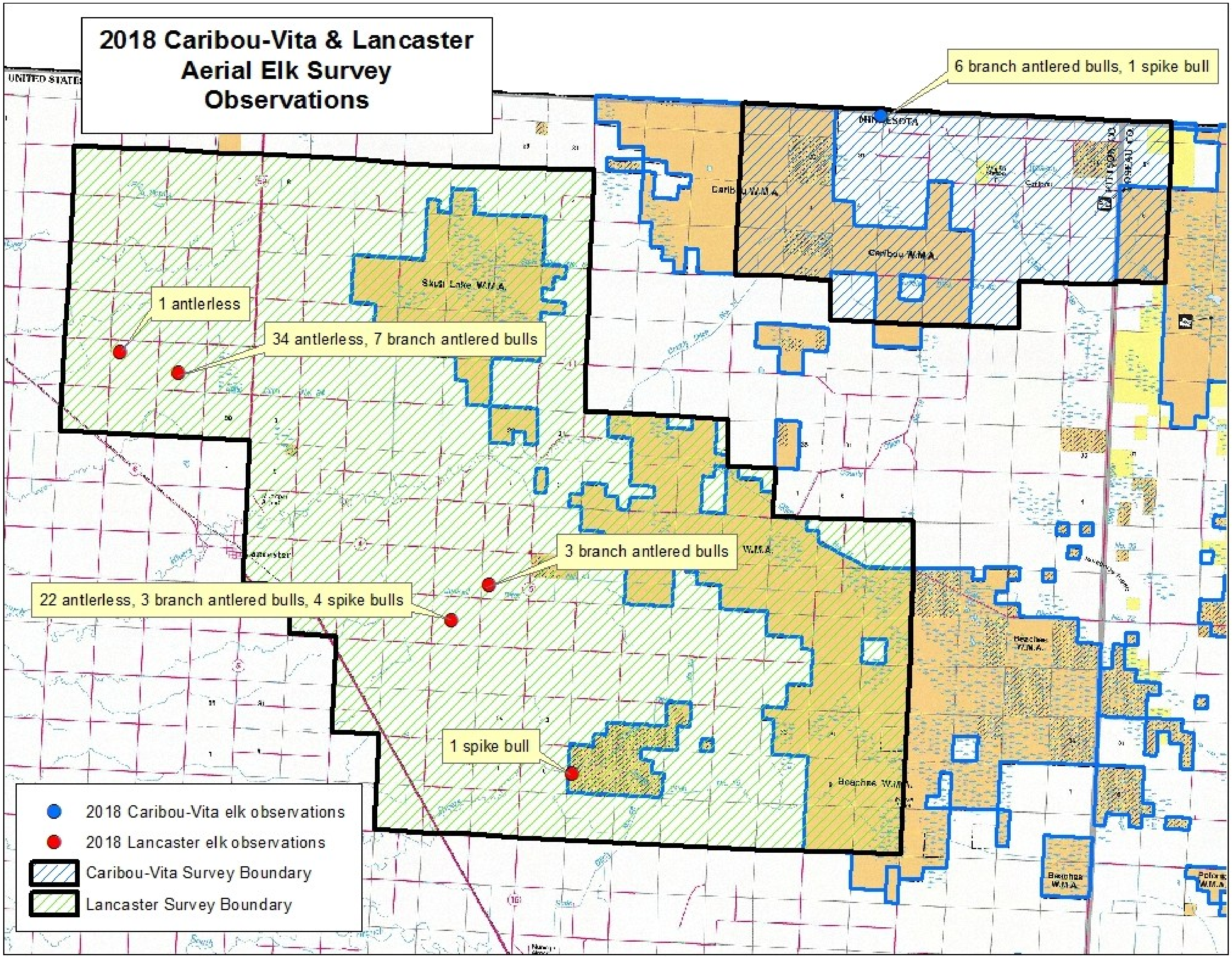


Figure 1. Locations of elk observed within the Caribou-Vita and Lancaster area survey blocks, 2018.

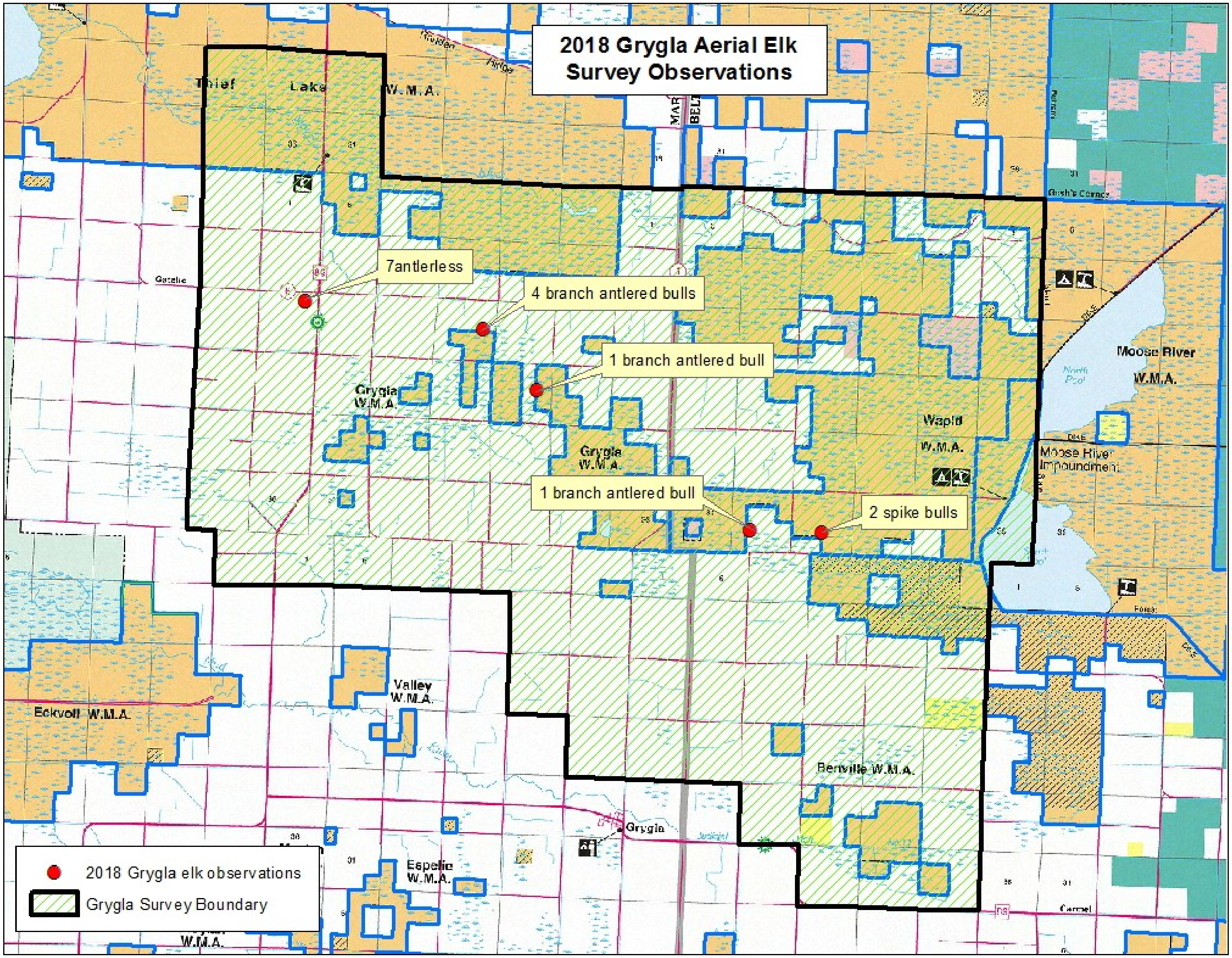


Figure 2. Locations of elk observed within the Grygla area survey blocks, 2018.

2018 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 americanus) 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 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 moose density classes are based on whether ≤ 2, 3–7, or ≥8 moose, respectively, would be expected to occur in a specific plot. The most recent re-stratification was conducted in November 2013 for the 2014 survey, but additionally, individual plots are 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).

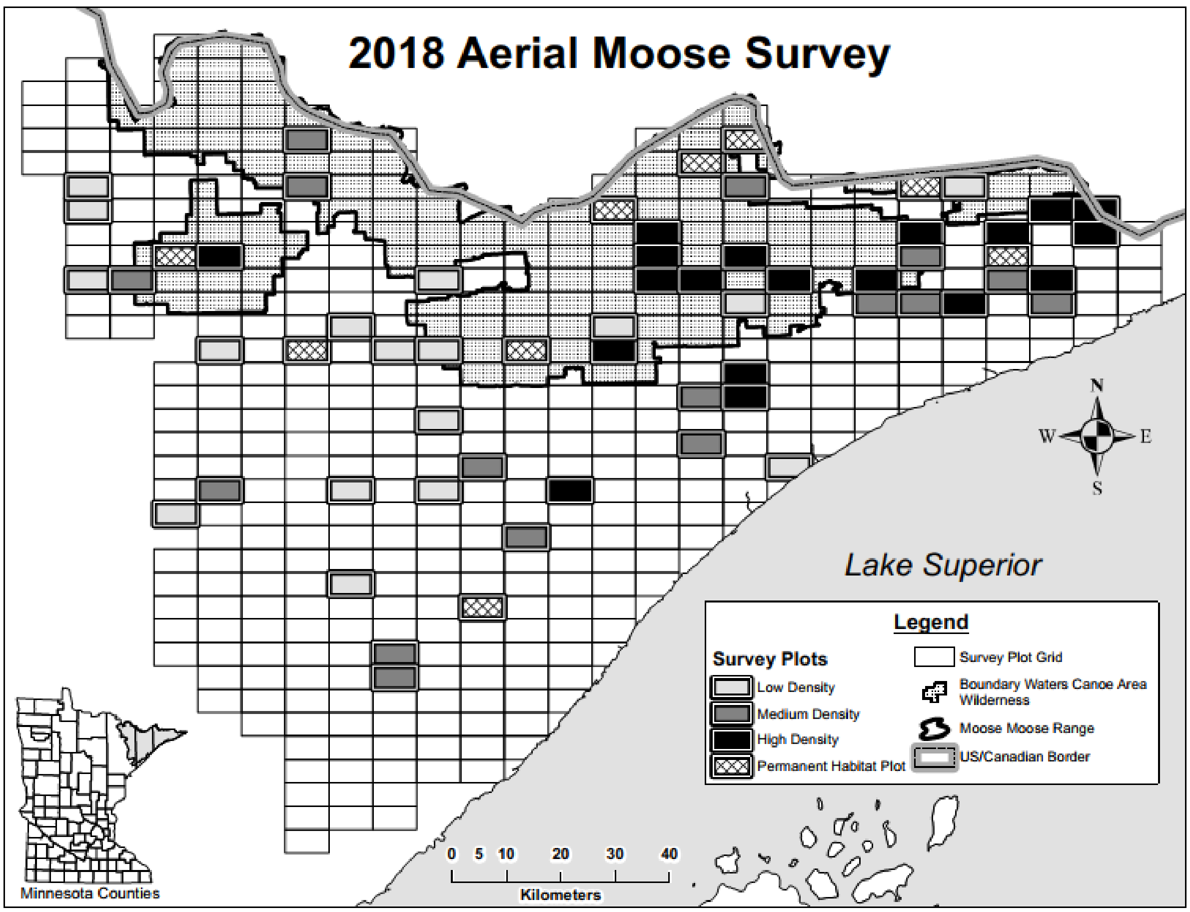


Figure 1. Moose survey area and 52 sample plots flown in the 2018 aerial moose survey.

RESULTS AND DISCUSSION

The survey was conducted from 3 to 13 January 2018. It consisted of 9 actual survey days, and as in 2014, 2015, and 2016, and 2017, it included a sample of 52 survey plots. This year, based on optimal allocation analyses, we surveyed 14 low-, 13 medium-, and 16 high-density plots, and the 9 permanent or habitat plots (Giudice 2018). Generally, 8" of snow cover is our minimum threshold depth for conducting the survey. Snow depths were 8–16" and >16" on 65% and 31% of the sample plots, respectively. Overall, survey conditions were rated as good for 98% and fair for 2% of the plots when surveyed. Average survey intensity was 48 minutes/plot (13.9 mi^2) and ranged from 40 to 60 minutes/plot (Giudice 2018).

This year a total of 415 moose were observed on 37 (71%) of the 52 plots surveyed (a total 723 mi^2), less than the 508 moose observed on 47 of 52 plots during the 2017 survey. An average of 11.2 moose (range = 1–31) were observed per "occupied" plot. Plot occupancy during the past 14 years averaged 82% (range = 65–95%) with a mean 11.8 moose observed per occupied plot. This year's 415 observed moose included 181 bulls, 170 cows, 63 calves, and 1 unclassified adult. Overall, estimated VO averaged 37% (range = 0−85%) and average estimated detection probability was 0.61 (range = 0.23−0.85); both were comparable to those of previous years.

After adjusting for sampling and sightability, we estimated the population in northeastern Minnesota at 3,030 (2,320–4,140, 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 2017). The current population estimate is 65% less than the estimate in 2006 and the declining linear trend during the past decade remains statistically significant (r^2 = 0.81, P< 0.001, Figure 2). However, the leveling since 2012 persists, and a piecewise polynomial curve indicates that the trend from 2012 to 2018 is not declining (Figure 3). While this recent short-term trend (7-year) is noteworthy, it applies only to the existing survey estimates, and does not forecast the future trajectory of the population (Giudice 2018).

The January 2018 calf:cow ratio of 0.37 is low but similar to the 13-year average since 2005 (0.35, Table 1, Figure 4). Calves were 15.1% of the total 415 moose actually observed and represented 15% of the estimated population (Table 1, Figure 4). Twin calves were observed with 6 of the 170 (4%) 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 January2018 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 sprint 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-collard 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 2018 estimated bull:cow ratio (1.25, Table 1; Figure 5) appears to be elevated compared to the long-term mean of 0.98 during 2005-2017, 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, 2005–2018.

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

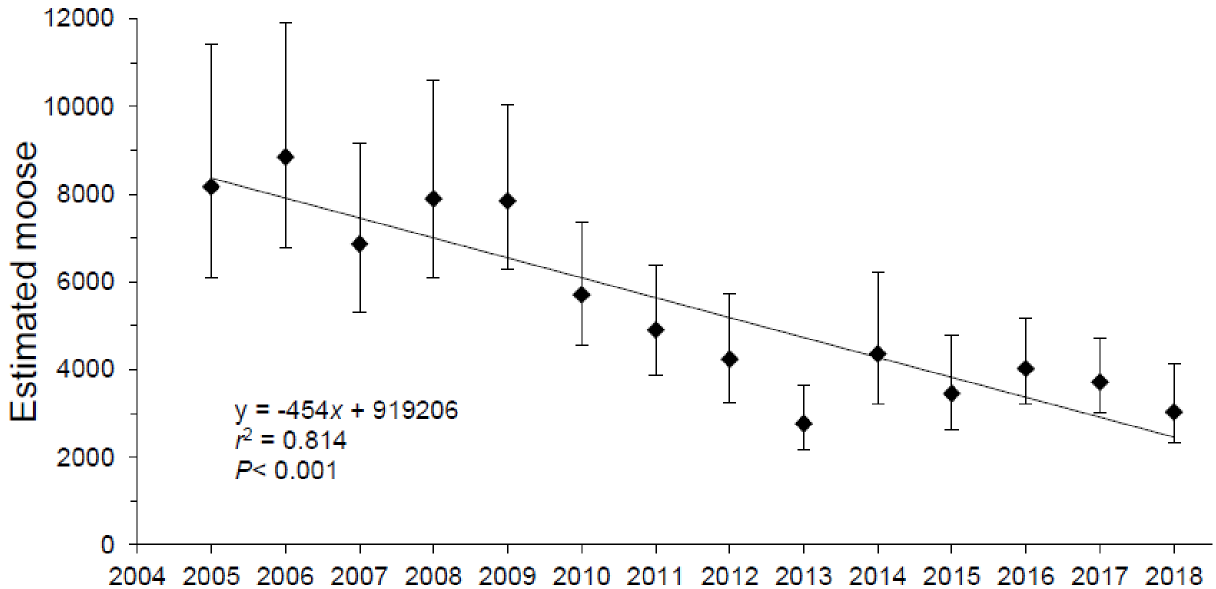


Figure 2. Point estimates, 90% confidence intervals, and a linear trend line of estimated moose abundance in northeastern Minnesota, 2005–2018. (Note: The 2005 survey was the first to be flown with helicopters and to include a sightability model and a uniform grid of east-west oriented rectangular 13.4-mi^2 plots).

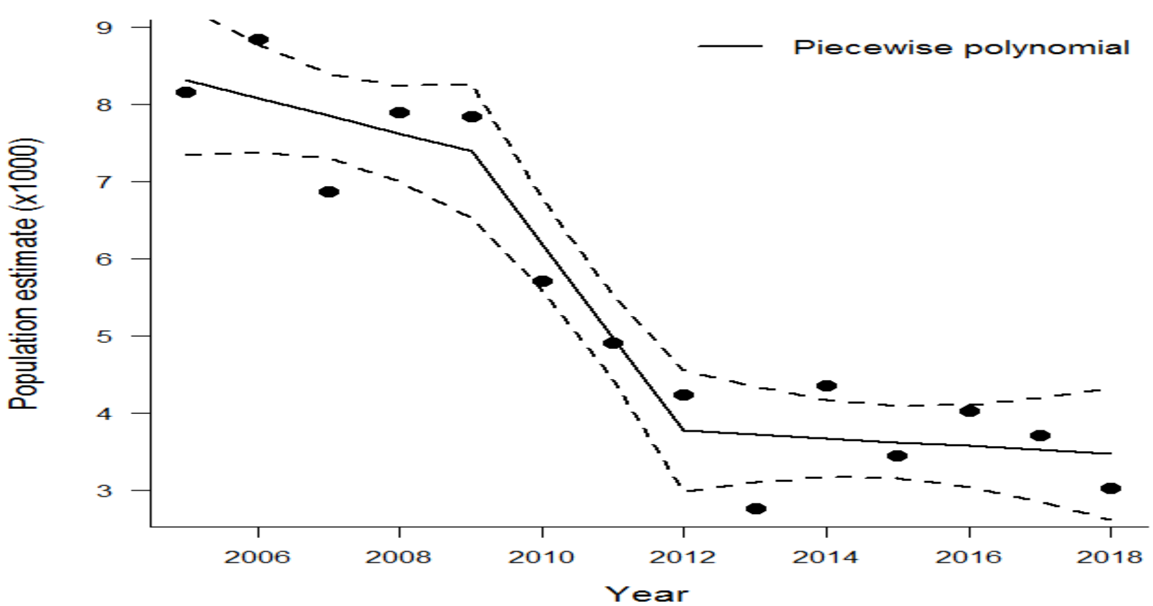


Figure 3. Point estimates, 90% confidence intervals, and a piecewise polynomial curve of moose abundance in northeastern Minnesota, 2005–2018. This curve shows a change in the short-term slope of the trend from 2012 to 2018 compared to 2009 to 2012.

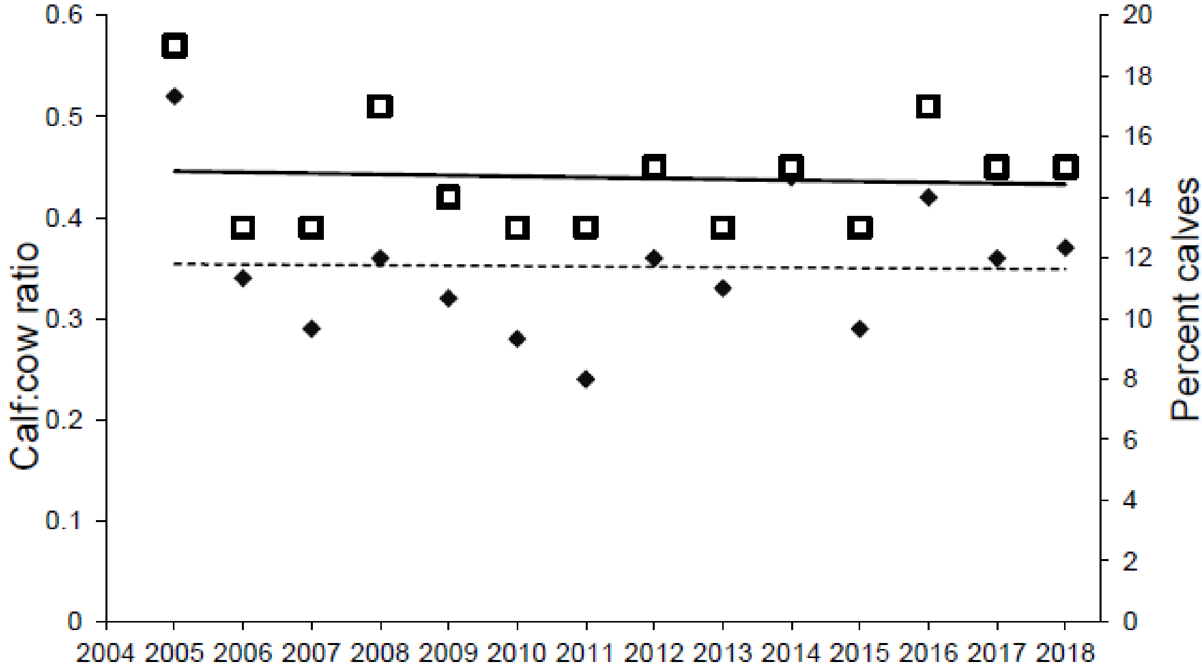


Figure 4. Estimated calf:cow ratios (solid diamonds, dashed trend line) and percent calves (open squares, solid trend line) of the population from aerial moose surveys in northeastern Minnesota, 2005–2018.

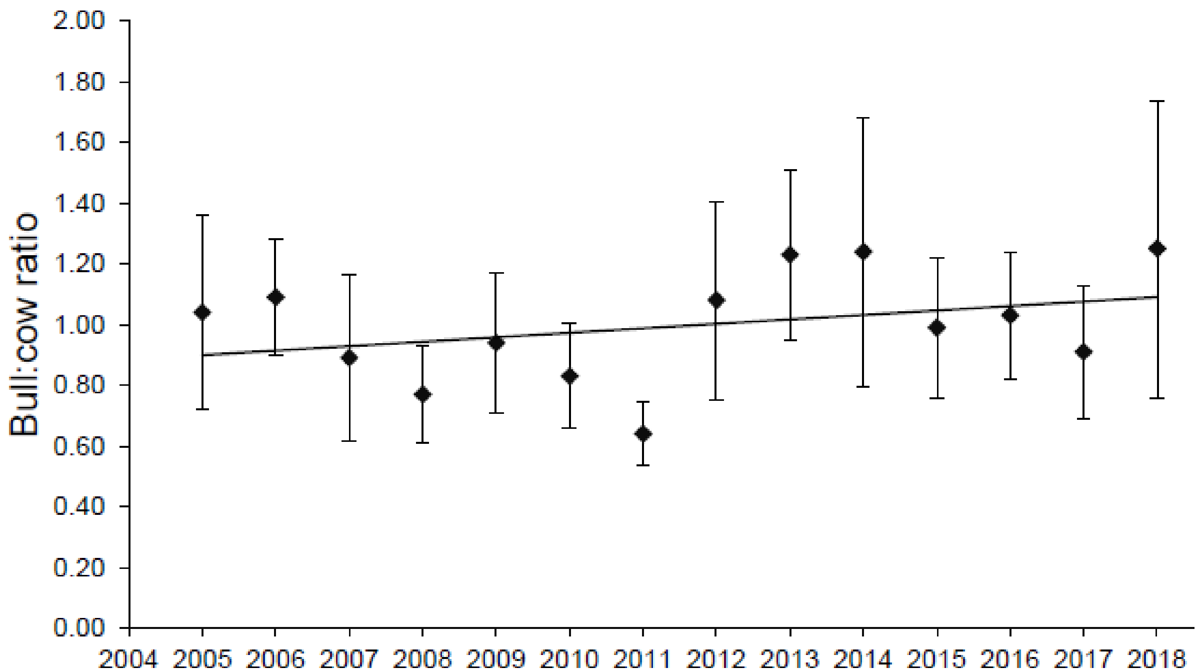


Figure 5. Estimated bull:cow ratios, 90% confidence intervals, and trend line from aerial moose surveys in northeastern Minnesota, 2005–2018.

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 Andy Edwards (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, Andy Edwards, Mike Schrage, and Nancy Hansen flew as our seasoned observers. Thank you to John Giudice who continues to provide critical statistical consultation and analyses, and to Barry Sampson 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, and provided refresher training for survey observers using DNRSurvey. 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 Andy Edwards. This project was funded in part by the Wildlife Restoration (Pittman-Robertson) Program.

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DISTRIBUTION AND ABUNDANCE OF WOLVES IN MINNESOTA, 2017-18

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At the time wolves were federally protected in the mid-1970s, Minnesota contained the only known reproducing wolf population in the lower 48 states, except for that on Isle Royale. Over the years, much attention has been focused on studying and monitoring Minnesota's wolves. Research efforts began in the mid-1930s (Olson 1938) and with few lapses continue to this day. Efforts to delineate wolf distribution and enumerate populations have also been made at various times over the last 50 years (Erb and DonCarlos 2009).

Early estimates of Minnesota's wolf population, often derived from bounty records and anecdotal information, were by necessity subjective. With the advent of radio-telemetry, geographic information systems (GIS), and global positioning systems (GPS), more detailed monitoring and mapping of wolf populations has been possible. However, financial and logistical considerations often limit intensive monitoring to small study areas.

Enumerating elusive carnivore populations over large areas remains a difficult task, particularly in forested landscapes (Kunkel et al. 2005). Complete territory mapping (Fuller and Snow 1988, Burch et al. 2005) is usually not possible over large areas, though various sampling designs can be considered (Potvin et al. 2005). Use of standard mark-recapture methods may not be practical given the difficulties of capturing and recapturing sufficient samples. However, genetic mark-recapture methods have recently been applied to wolves (Marucco et al. 2009) but may also be impractical over large areas.

Population estimation approaches based on prey or habitat assessments (e.g., Fuller 1989, Boyce and Waller 2003, Cariappa et al. 2011) may be useful for estimating potential abundance of large carnivores but may not always match realized abundance due to other time-varying factors that may limit populations (e.g., disease, weather, lagged responses to changes in prey). Newer aerial sampling methods exist (Becker et al. 1998, Patterson et al. 2004) but may be logistically challenging when applied to broad expanses of dense forest. Initial evaluation of these aerial snow-tracking methods in Minnesota was not promising (J.E., unpublished data). Further evaluation may be needed, including a cost-benefit analysis, but many assumptions of the method appear difficult to meet in Minnesota's forested landscape with moderate to high deer abundance.

Since the late-1970s 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 occupied by pack wolves. The methods employed have changed only slightly during this time. During 1978-1998, surveys were conducted at 10-year intervals. During 1998-2012, surveys were conducted at approximately 5-year intervals, in part for consistency with the survey timeline specified in the Minnesota Wolf Management Plan (first and fifth year after delisting; Minnesota Department of Natural Resources 2001). 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) and only slight geographic expansion between 2007 and 2012 (Erb and Sampson 2013). These results have been coarsely consistent with separate wolf population trend indicators in Minnesota (i.e., annual scent station survey, winter track survey, and number of verified depredations).

In 2012, wolves in the Great Lakes Distinct Population Segment were removed as a listed species under the federal Endangered Species Act (ESA). From 2012 to 2014, a regulated public harvest of wolves was allowed, with annual harvests in Minnesota ranging from 238 to 413 wolves. As a result of a court ruling in late-2014, ESA protections were reinstated on wolves in the Great Lakes and no public harvest of wolves has since occurred in Minnesota. Beginning in 2012, the Minnesota Department of Natural Resources (DNR), with assistance from numerous collaborators, began deriving wolf population estimates annually. However, one component of the surveys - delineation of total and occupied wolf range (defined below) - is still being re-assessed only at 5-year intervals, as was the case from 1998 to 2012. Winter 2017-18 marks the 5th year since wolf range has been re-assessed, and re-assessment of wolf range was included in this winter's wolf survey.

METHODS

The approach we used to delineate wolf distribution and estimate population size was essentially identical to the previous 5 wolf range surveys (Fuller et al. 1992, Berg and Benson 1998, Erb and Benson 2004, Erb 2008, Erb and Sampson 2013), and conceptually similar to the 1978-79 wolf range survey (Berg and Kuehn 1982). Primary cooperators were similar to previous wolf range surveys and included natural resources staff within: 1) DNR; 2) U.S. Forest Service; 3) U.S. Fish and Wildlife Service; 4) U.S. Department of Agriculture - Wildlife Services; 5) U.S. Geological Survey; 6) Tribal and Treaty resource authorities; 7) County Land Departments; 8) Camp Ripley Military Facility; 9) Voyageurs National Park; and 10) various University collaborators and research projects.

We mailed instructions to participants in October 2017 and asked them to record a location and group size estimate for all wolf sign (e.g., visual, track, scat) observed during the course of normal work duties from November 2017 until snowmelt the following spring (~ mid-May 2018). Participants could record locations on forms or maps then provided to us for later data entry, but most data were entered directly by participants in a web-based GIS survey application. As in previous wolf range surveys, we used the Public Land Survey township (~93 km^2, with some exceptions) as the spatial scale for classifying wolf observations.

Although recorded estimates of wolf group size are not used directly for population enumeration, the assessment of township-specific wolf occupancy, as discussed below, treats observations of single wolves differently than pack (>1 wolf) detections. We conservatively assumed group size to be 1 in situations where sign was recorded but no group size was noted. If group size was recorded as 'numerous', it was set to 2 (i.e., a pack). We then combined this database with wolf observations recorded on other wildlife surveys during 2017-18 (e.g., carnivore scent station survey, furbearer winter track survey, moose/deer/elk surveys, etc.). This combined database is hereafter referred to as 'WISUR18'. Locations of verified wolf depredations from 2013 to 2018, as well as locations of wolves harvested during the 2012-2014 regulated wolf seasons, were also consulted for purposes of delineating total wolf range, but they were not used in any assessment of townships currently occupied by wolf packs and are not part of the WISUR18 database.

Delineation of both total range and occupied range includes, but is not limited to, consideration of whether townships meet human and road density criteria defined by Fuller et al. (1992; i.e., townships within wolf range are presumed to be occupied by wolves if road density is <0.7 km/km^2 and human density is <4/km^2, <u>or</u> if road density is <0.5 km/km^2 and human density is <8/km^2; hereafter termed 'modeled' townships). As in previous surveys, human density was calculated using the most recent (i.e., 2010) U.S. Census Data as incorporated into the 2010 Minor Civil Divisions GIS layer produced by the Minnesota Legislative Coordinating Commission. Road density calculations are based on the Minnesota Department of Transportation's 1:24,000 GIS roads layer (excluding 'forest roads') and summarized within each township as the number of kilometers of road per km^2.

Delineation of total wolf range is intended to encompass those areas within the state where consistent or sufficient wolf detections occur (either singles or packs) more than might be expected from 'random' temporally-irregular dispersals. Total wolf range depicts the coarse distribution of wolves within the state and is useful for documenting larger-scale expansions or contractions of wolf range. Although Minnesota's wolf range has expanded south and west since the 1970s, it has remained essentially contiguous with the Canadian border to the north and Lake Superior and Wisconsin to the east.

Because systematic searches for wolf sign are not conducted and much of the southern and western periphery of wolf range in Minnesota is private land, there is some subjectivity in the approach used to delineate the south and west boundary. Using the previously delineated boundary as the reference point, we re-evaluated the south and west border based on the following data: 1) all WISUR18 observations; 2) modeled townships; 3) land use and cover; and 4) knowledge of wolf activities in the area since the last survey (e.g., wolf depredation sites, 2012-14 wolf harvest locations). While maintaining a contiguous total wolf range, the overall approach is designed to maximize inclusion of areas with periodic (since last survey) or recently abundant wolf observations and modeled townships, while minimizing inclusion of areas that neither fit the model nor contained numerous or consistent wolf observations.

We computed occupied range by subtracting from the total range all townships that neither contained current observations of a pack (defined as >1 animal) nor fit the human-road density model criteria. We also fully excluded lakes larger than 200 km^2 (n = 5) from calculations of both total and occupied range.

To radio-collar wolves for use in estimation of territory and pack sizes, 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. In addition, numerous wolves were captured using live-restraining neck snares during winter. 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 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 overly-excessive 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 the majority of territories were delineated using VHF radiolocations, territory sizes were increased 37% to account for the average amount of interstitial space between 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 are much more likely to be an artifact of territory underestimation when there are comparatively sparse radiolocations. Hence, for packs with <100 radiolocations (n = 9; mean number of radiolocations = 38) we multiplied the area of each estimated territory by 1.37 as in the past. For packs with >100 radiolocations (n = 36; mean number of radiolocations = 1,301), territories were assumed to be fully delineated and were not re-scaled.

To estimate the number of packs within occupied wolf range, the area of occupied range is divided 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 percentile method (Manly 1997), the 90% confidence interval for population size 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

A total of 1,601 opportunistic wolf sign observations were recorded during the 2017-18 wolf range survey (Figure 1). Observations consisted of 65% tracks, 15% visuals, 4% scats, and 16% other (howls, deer kills, depredation sites, etc.).

Distribution

We evaluated potential shifts in total wolf range by examining available information near the southern and western edge of the previously-delineated wolf range boundary. After considering the totality of information (see Methods), we concluded that sufficient data existed to extend the previous wolf range line in numerous areas along the southern and western periphery. Revised total wolf range was estimated to be 111,862 km^2, an increase of ~18% from 2012 (Figure 1, Table 1).

After removing townships within the revised total range that neither met human-road model criteria nor contained WISUR18 pack observations, estimated occupied range was 73,972 km^2 (Figure 1), a 4.8% increase from the 2012 survey (Figure 1, Table 1). Of the total estimated occupied range, 66% was confirmed to be occupied based on pack detection in the township, and 34% was presumed to contain packs because of low human and road density (i.e., modeled townships; Table 1). Of all the townships in wolf range that contained pack observations, 27% had higher human and/or road density than the thresholds in the road-human density model previously developed (Table 1).

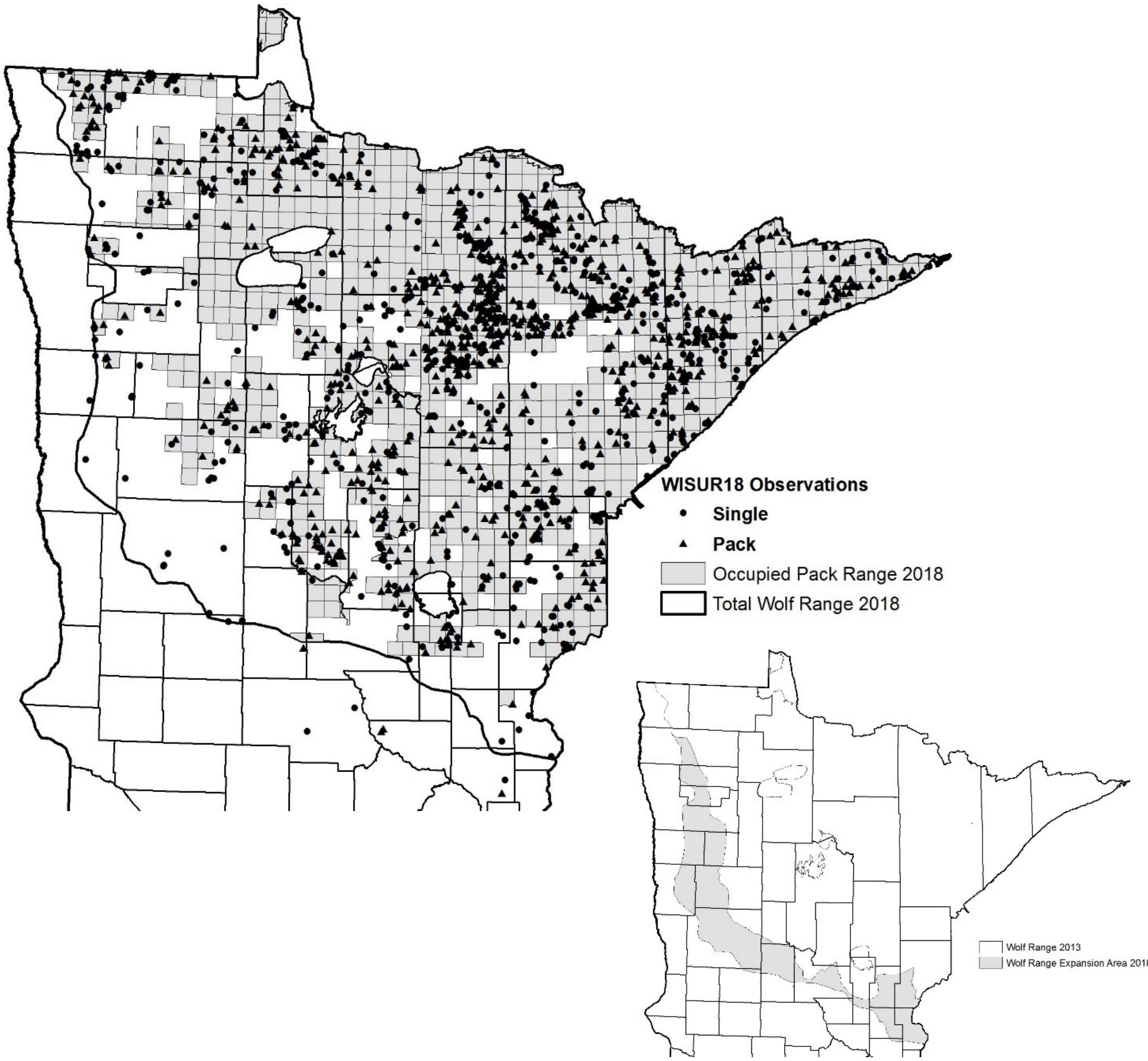

Figure 1. Wolf sign observations, total wolf range, and occupied townships delineated as part of the 2017-18 winter wolf survey in Minnesota. Small inset highlights area of range expansion since 2012.

Table 1. Comparison of Minnesota wolf range assessments, 1988 – 2018.

	1988/89	1997/98	2003/04	2007/08	2012/13	2017/18
Total Wolf Range (km^2)	60,229	88,325	88,325	88,325	95,098	111,862
Occupied Range (km^2)	53,100	73,920	67,852	71,514	70,579	73,972
% Occupied Range confirmed by pack detection in township	55	84	54	68	70	66
% occupied area with pack detection that exceeds human/road density thresholds [a]	11	17	19	20	30	27
Wolf Population Density (wolves/100 km^2)	2.86	3.31	4.45	4.08	3.13	3.59

[a] thresholds from Fuller et al. (1992)

Pack and Territory Size

We obtained sufficient location data to generate territories for 45 packs (Figure 2); their collective territory area represented 10% of occupied wolf range. Winter pack size counts were obtained for 41 packs, including 6 packs with insufficient location data for territory delineation.

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 2), likely a combined result of more collared packs (and 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. Average spring 2017 deer density in the larger deer permit areas within which the wolf territories were situated, weighted by the number of radio-marked wolf packs within a given permit area, was 11.1 deer/mi^2. In comparison, spring deer density for the forest zone of Minnesota, a close approximation of wolf range, was 13.3 deer/mi^2 in spring 2017. Considering this collective information, we suspect that the sample of collared packs this winter might be slightly biased towards areas of lower quality wolf habitat compared to last winter.

After applying the 'interstitial scaling factors' discussed in the Methods, average territory size for radio- marked packs was 158.97 km^2 (Figure 3). Average winter pack size was 4.85 wolves (Figure 4).

Wolf Numbers

Dividing estimated occupied range (73,972 km^2) by average territory size (158.97 km^2) results in an estimate of 465 wolf packs in Minnesota (Figure 5). Multiplying by average pack size (4.85) and accounting for an estimated 15% lone wolves yields a population point estimate of 2,655 wolves (Figure 6), or 3.6 wolves per 100 km^2 of occupied range (Table1). The 90% confidence interval ranges from 1,972 wolves to 3,387 wolves (Figure 6).

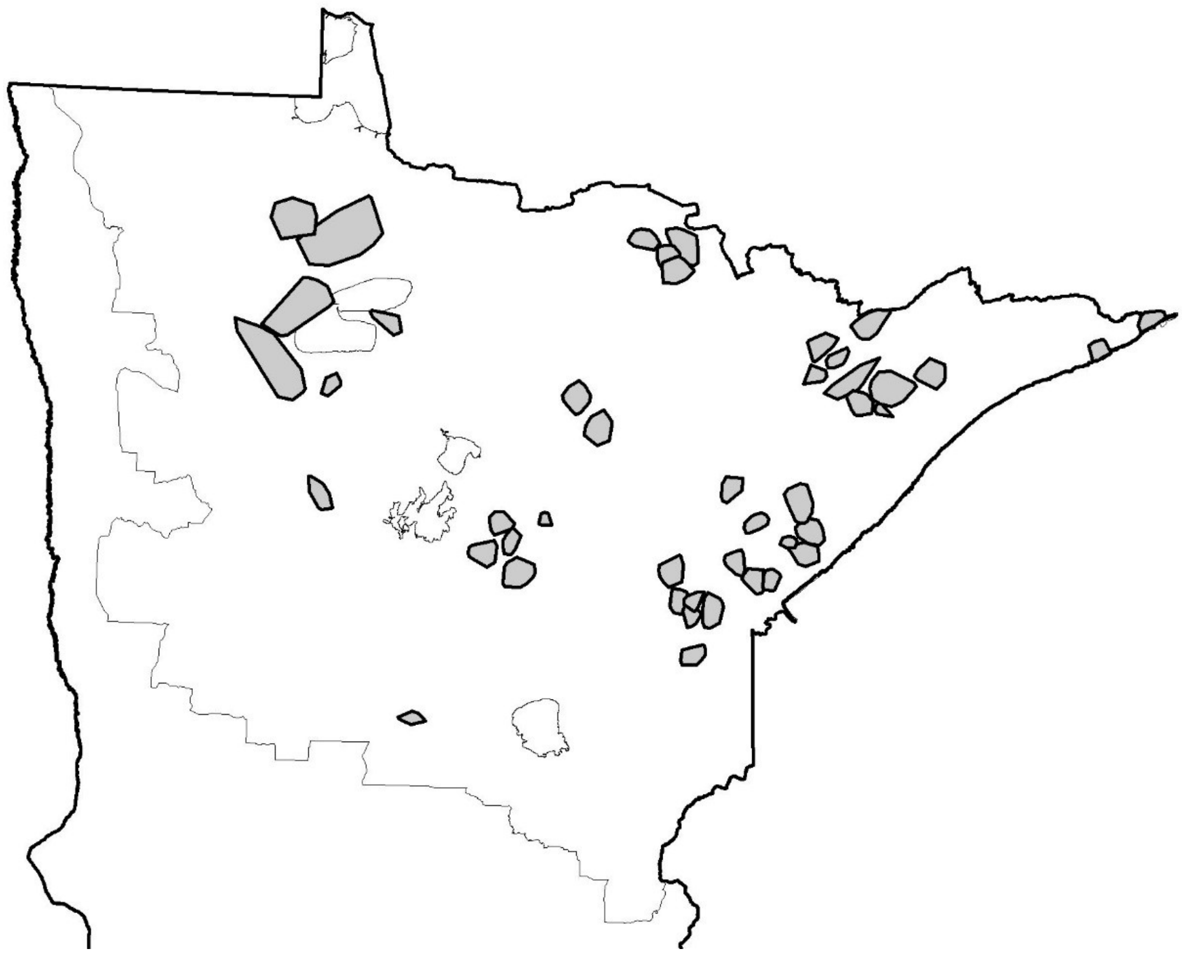

Figure 2. Location of radio-marked wolf packs in Minnesota from which data on territory and pack size were derived during the 2017-18 survey.

Table 2. Comparison of land cover[a] in territories of radio-collared wolf packs during winter 2017-18 with land cover in all of occupied wolf range in Minnesota.

	Overall Occupied Wolf range	Radio-collared Wolf Territories
Land Cover Category	% Area	% Area
Woody Wetlands	31.9	37.2
Deciduous Forest	23.1	18.1
Emergent Herbaceous Wetlands	10.1	13.2
Mixed Forest	6.9	7.1
Evergreen Forest	6.7	8.5
Open Water	5.2	4.5
Shrub/Scrub	4.5	4.8
Pasture/Hay/Grassland/Crops	9.3	4.9
Developed, All	2.4	1.8

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

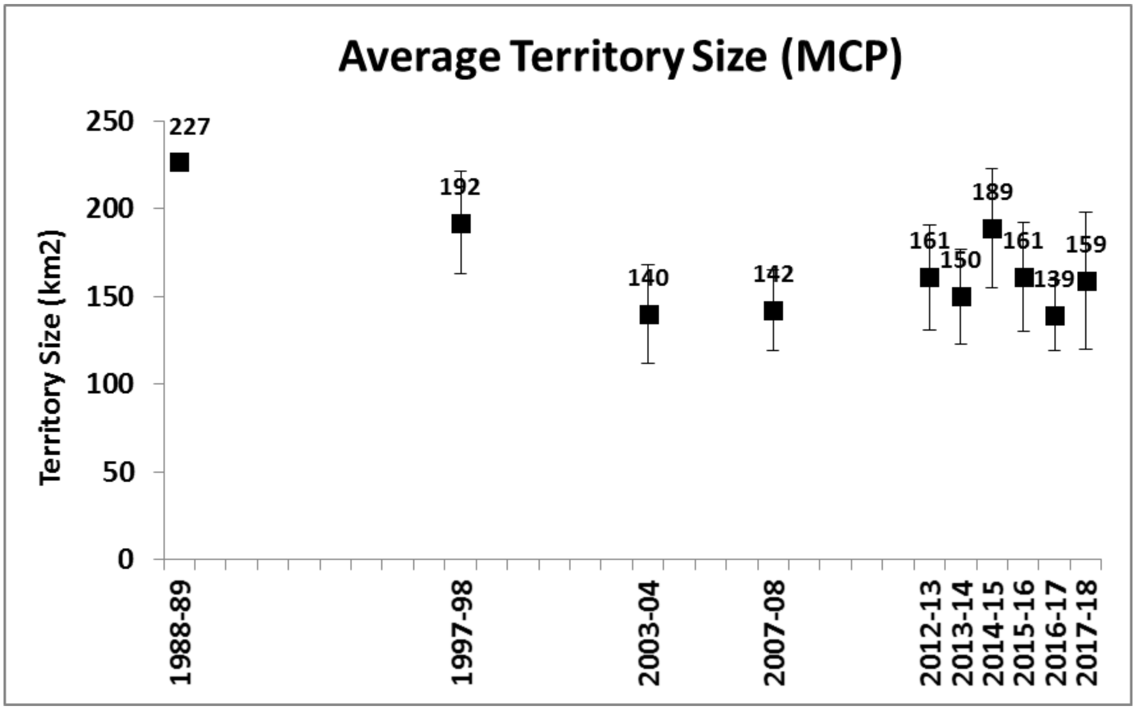

Figure 3. Average scaled territory size for radio-marked wolf packs in Minnesota from 1989 to 2018.

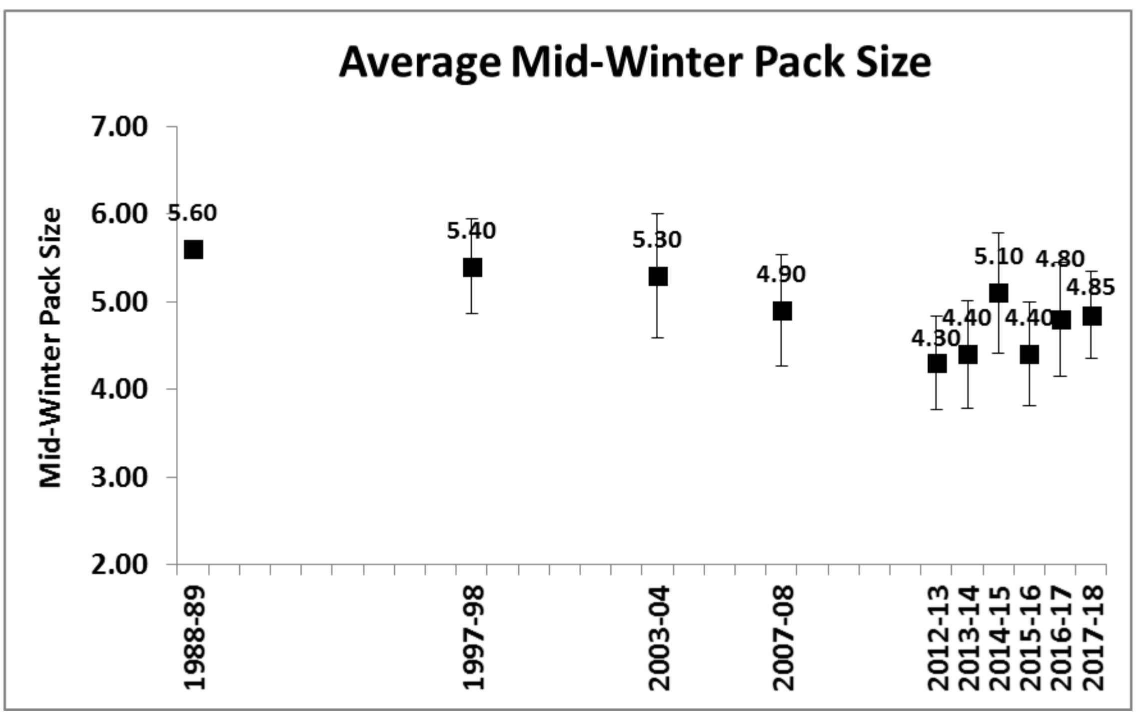

Figure 4. Average mid-winter pack size for radio-marked wolf packs in Minnesota from 1989 to 2018.

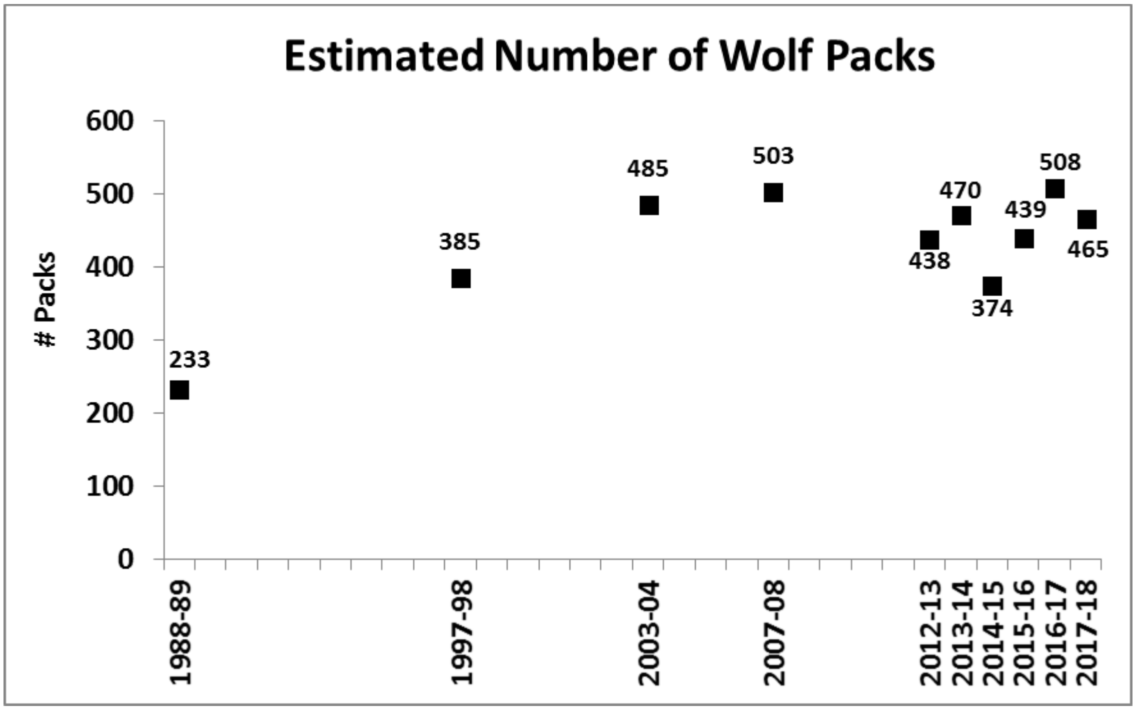

Figure 5. Estimated number of wolf packs in Minnesota at periodic intervals from 1989 to 2018.

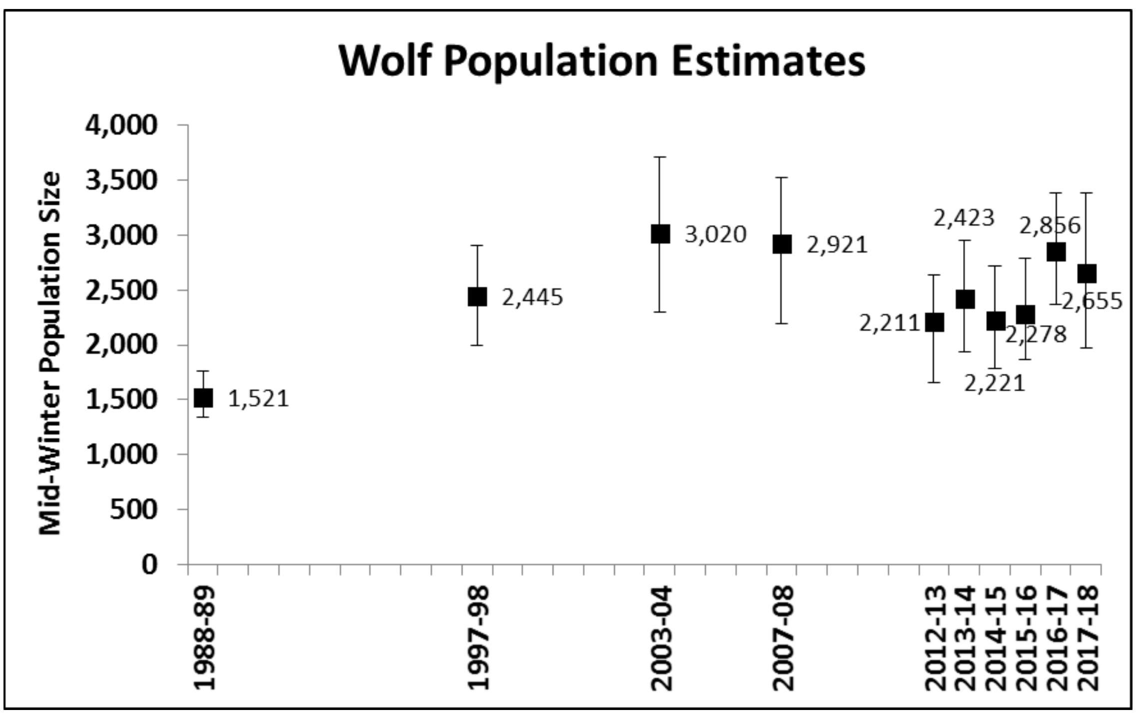

Figure 6. Wolf population estimates from periodic standardized surveys in Minnesota, 1989 to 2018.

DISCUSSION

Available information since the 2012 survey indicates that wolf range has expanded in several areas along the southern and western periphery. In these areas, we repositioned the wolf range line after considering multiple data sources, resulting in an 18% increase in wolf range to 111,862 km^2. Although much of the area of expansion was not concluded to be occupied by packs based on either the human- road density model or pack detections, and hence was not included in occupied range, we felt sufficient confirmations have occurred in these areas since the last survey to justify range expansion. It is also likely that wolves remain under-detected by survey participants in these areas during winter due to private ownership of much of the land.

Approximately two-thirds of total wolf range, or 73,972 km^2, was estimated to be occupied by wolf packs during winter 2017-18. This represents an approximately 5% increase in occupied range since the 2012- 13 wolf range survey; the 4 estimates of occupied wolf range since then have fluctuated between approximately 68,000 and 74,000 km^2. Because delineation of wolf range relies on opportunistic wolf sign observations, effort across surveys likely varies as a result of fluctuations in the number of personnel able to contribute wolf sign observations or the number of hours spent afield by survey participants. Hence, we can't rule out sampling variation as the cause of slight changes in the estimate of occupied range, though changes in wolf demographics likely contribute to the fluctuations as well. Since 1998, there has been no consistent increasing or decreasing trend in the amount of occupied range.

Because 34% of the townships were deemed occupied based only on 'low' human/road density (i.e., not via pack detections), it remains possible that occupied range could be overestimated. However, in a majority of cases a lack of pack detections likely reflects a lack of sampling effort rather than a lack of wolves. Some wolves occupy remote areas (e.g., the BWCAW) and are unlikely to be opportunistically detected, and notable amounts of private land, particularly in the southern and western portion of the range, are also unlikely to be opportunistically surveyed. Stated differently, pack detection probability is undoubtedly less than 1 in many areas. Finally, while prey- or habitat-based models have some potential to overestimate occupancy at any given time, the 1988-89 human-road density model (Fuller et al. 1992) utilized in our methodology has generally been a conservative descriptor of wolf 'habitat' in Minnesota.

The percentage of township area containing pack observations but exceeding the occupancy thresholds in the 1988-89 road-human density model had increased from 1988 (17%) to 2012 (30%), but may now have stabilized; results from the 2017-18 survey indicate that 27% of the townships in which wolf packs were confirmed have human-road densities that exceed the thresholds.

From 1988 to 2003, wolf pack territory sizes declined in Minnesota. Although numerous factors can influence territory size, we believe 2 largely explain this pattern. First, expanding wolf populations (or portions thereof) that compose a significant number of colonizing packs have been shown to exhibit declines in average pack territory size as the population becomes more established or available range more saturated (Fritts and Mech 1981, Hayes and Harestad 2000), a characterization that applies to the Minnesota wolf population from early recovery up to approximately 2003. Second, territory size is negatively correlated with prey density (Mech and Boitani 2003, Fuller et al. 2003), and Minnesota's deer population exhibited an increasing trend during much of wolf recovery in Minnesota. Since 2003, our estimates of average territory size have been comparatively stable, with fluctuations in point estimates likely driven by sampling variability and the direct or lagged influence of deer density fluctuations.

Average mid-winter pack size as estimated from radio-marked packs was approximately 4.9 and has generally exhibited only minor fluctuations over time. The correlation between winter pack size and prey density is not as strong as the correlation between prey density and wolf territory size, though prey density certainly has an influence on pack size, particularly via changes in pup survival (Fuller et al. 2003). Our estimates of winter pack size are highly likely to underestimate true pack sizes, though we suspect not substantially so. Underestimation results from the difficulties of obtaining counts at times when the full pack is together, and in locations and conditions in which all are detectable from the air or ground.

Accuracy in estimates of average territory and pack size is dependent, in part, on radio-collaring a representative sample of wolf packs. Because it is not feasible to identify and stratify all wolf packs to employ true random sampling, our efforts have focused on identifying study areas for radio-collaring that are believed to be collectively representative of overall wolf range, particularly with respect to land cover and deer density. Even so, annual capture success in those areas varies, some collared wolves die or disperse, and some radio-collars prematurely fail. This creates annual variability in the degree to which collared packs are representative of the entire population. Examination of land cover and deer density data from this past winter suggests that location of collared packs may have been somewhat biased towards less productive areas, with the potential result being a population point estimate biased low.

Nonetheless, confidence intervals for the past 2 surveys widely overlap (Figure 6), indicating no significant population change from last year.

We estimate the current population of wolves to be 2,655 (+/- ~ 700), or 3.6 wolves/100 km^2. We estimate total wolf range to have increased by an estimated 18% since 2012, while occupied range was estimated to have increased ~5%. Since wolf population estimates have been derived annually (2012– present), wolf population estimates appear to coarsely track changes in deer density (Figure 7), and wolves remain widely distributed throughout Minnesota's forest zone.

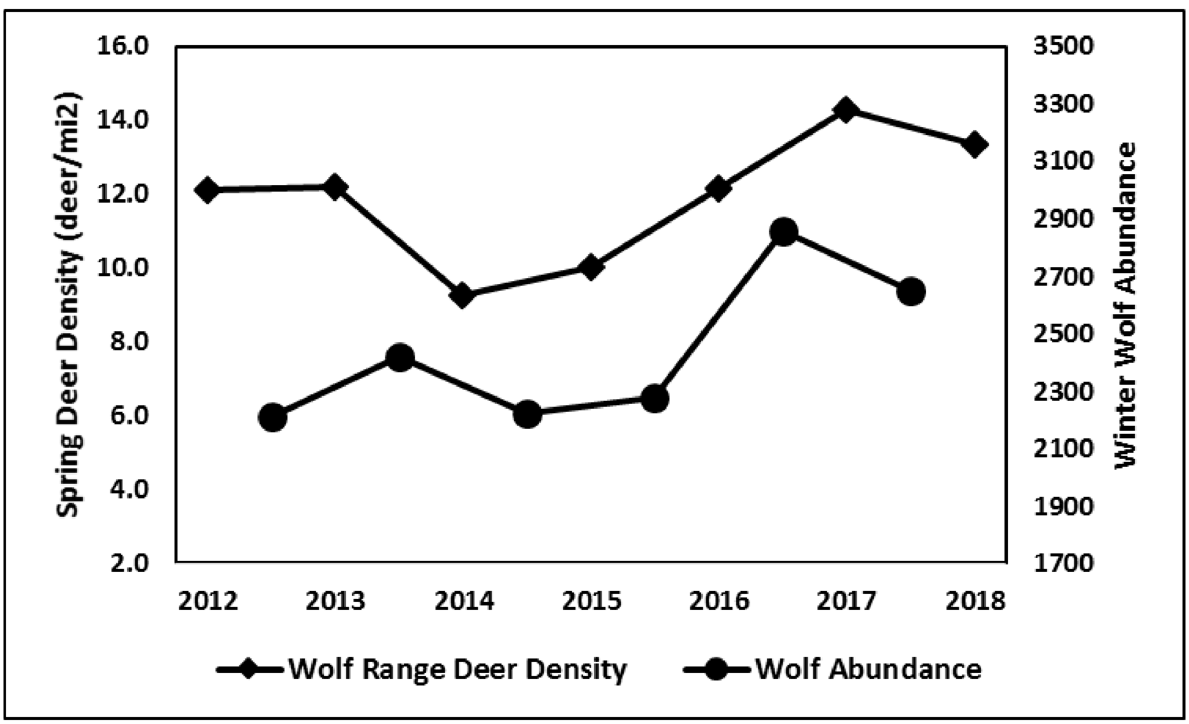

Figure 7. Comparison of estimated pre-fawn deer density in wolf range with winter wolf abundance in Minnesota, 2012-2018.

ACKNOWLEDGMENTS

We thank Craig Perrault for development of the web-based wolf observation application. We also acknowledge John Hart, Jeff Grabarkewitz, and Kevin Fuller (USDA Wildlife Services) for assistance with wolf capture and radio-collaring. We are grateful for the critical contributions of DNR pilots Jason Jensen, John Heineman, Chris Lofstuen, Bob Geving, Brad Maas, and Tom Buker during wolf telemetry and pack counts. We thank numerous collaborators for assistance or sharing of radio-telemetry data utilized in this survey, including Dave Mech and Shannon Barber-Meyer (USGS), Steve Windels and Tom Gable (Voyageurs National Park), Jay Huseby and Dave Price (Red Lake Band of Chippewa), Mike Schrage, Terry Perrault, Lance Overland and John Goodreau (Fond-du-Lac Band of Chippewa), Andy Edwards, Saranda Oestricher, and Christina Maley (1854 Treaty Authority), Seth Moore and Yvette Ibrahim (Grand Portage Band of Chippewa), and Brian Dirks and Nancy Dietz (Camp Ripley Military Reservation). Finally, special thanks to all natural resources field personnel throughout Minnesota that contributed wolf observations as part of the 2017-18 winter wolf survey.

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