Feasibility of restoring elk to northeastern Minnesota: habitat availability and social acceptance

A cooperative study conducted by:
Department of Fisheries, Wildlife and Conservation Biology, University of Minnesota

Minnesota Cooperative Fish and Wildlife Research Unit

Fond du Lac Band of Lake Superior Chippewa

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Acknowledgements

We thank the many private landowners and local residents who allowed us to access their land, participated in focus groups, and responded to questionnaires. M. Ditmer and J. Berini assisted with developing field methods. G. Barnas, A. Erickson, M. Kaminski, C. Minnick, M. Mullin, J. Wagner, T. Weise, and S. Wesche collected field data. L. Cornicelli, L. McInenly, C. Balzar, G. Bernu, G. Beck, S. Olson, T. Rusch, N. Hansen and J. Meyer assisted with study design and questionnaire development. L. Cornicelli, B. Keller, A. Landon, and L. McInenly reviewed drafts of this report. W. Bartsch, C. Beal, J. Erb, W. Hakala, C. Humpal, J. Kelash, D. Ryan, M. Swigen, M. Westphal, and D. Wilson provided GIS data. D. Beyer, H. Campa III, B. Dhuey, J. Gilbert, and C. Williamson provided habitat suitability model information. The Environment and Natural Resources Trust Fund, Minnesota Cooperative Fish and Wildlife Research Unit, Fond du Lac Resource Management Division, Great Lakes Restoration Initiative, and Rocky Mountain Elk Foundation provided funding and other support for this project.

Suggested Citation

McCann, N.P., Walberg, E.M., Forester, J.D., and Schrage, M.W. 2019. Feasibility of restoring elk to northeastern Minnesota: habitat availability and social acceptance. University of Minnesota, Minnesota Cooperative Fish and Wildlife Research Unit, Department of Fisheries, Wildlife and Conservation Biology, University of Minnesota, St. Paul, Minnesota.

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Executive summary

Elk (*Cervus canadensis*) once ranged across most of Minnesota but were functionally extirpated by the early 1900s. Three groups occur in northwestern Minnesota but are managed at low levels (Figure S-1). This study examines the feasibility of restoring elk to northeastern Minnesota. It provides information for determining where elk restoration will be successful, should it occur, including information about habitat suitability, social acceptance, and human-elk conflict.

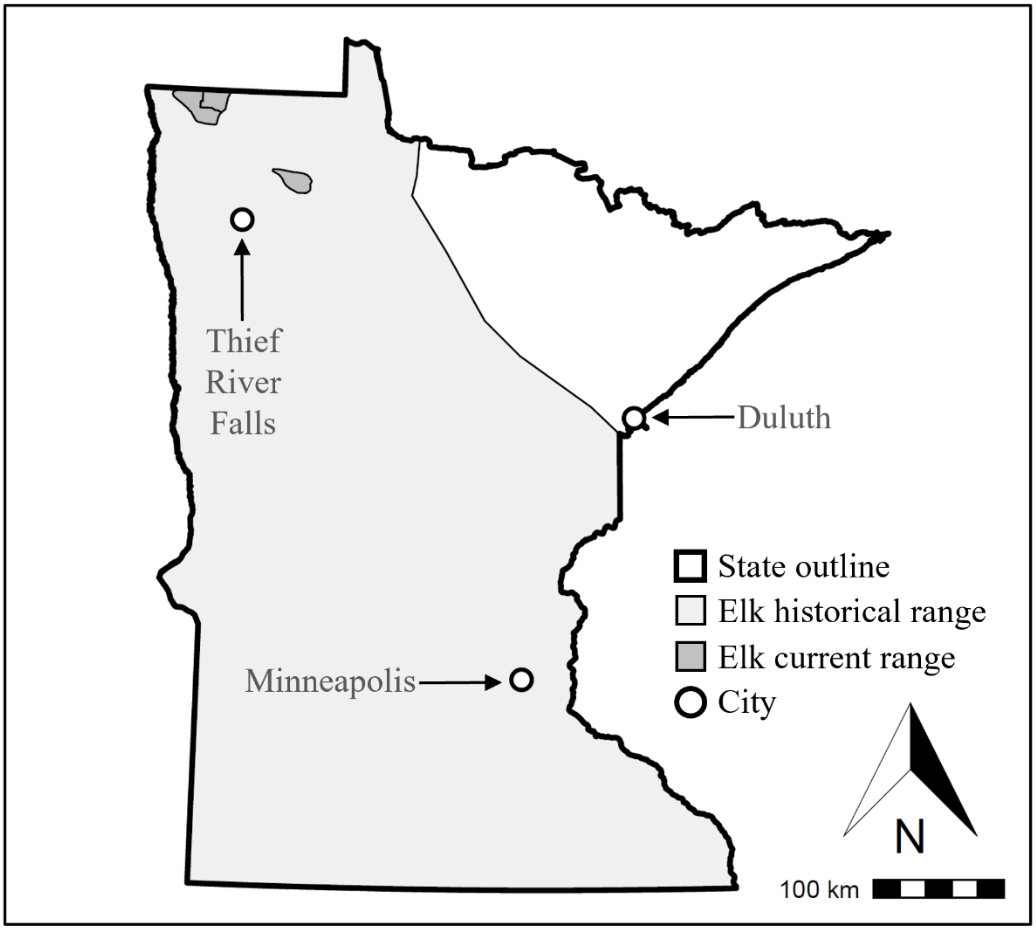

Figure S-1. Historical and current elk range in Minnesota. Some larger cities are included to serve as references. Scalebar: 100 km = 62 mi.

We studied habitat suitability and public support for elk on and near 3 study areas in northeastern Minnesota (Figure S-2). The Cloquet Valley study area was 1,764 km^2 (681 mi^2), the Fond du Lac study area was 766 km^2 (296 mi^2), and the Nemadji study area was 963 km^2 (372 mi^2). Study areas were comprised mostly of public land (60 to 75%) and had low road densities (0.96 km/km^2; 1.55 mi/mi^2) that are suitable for elk (< 2 km/km^2; 3.22 mi/mi^2).

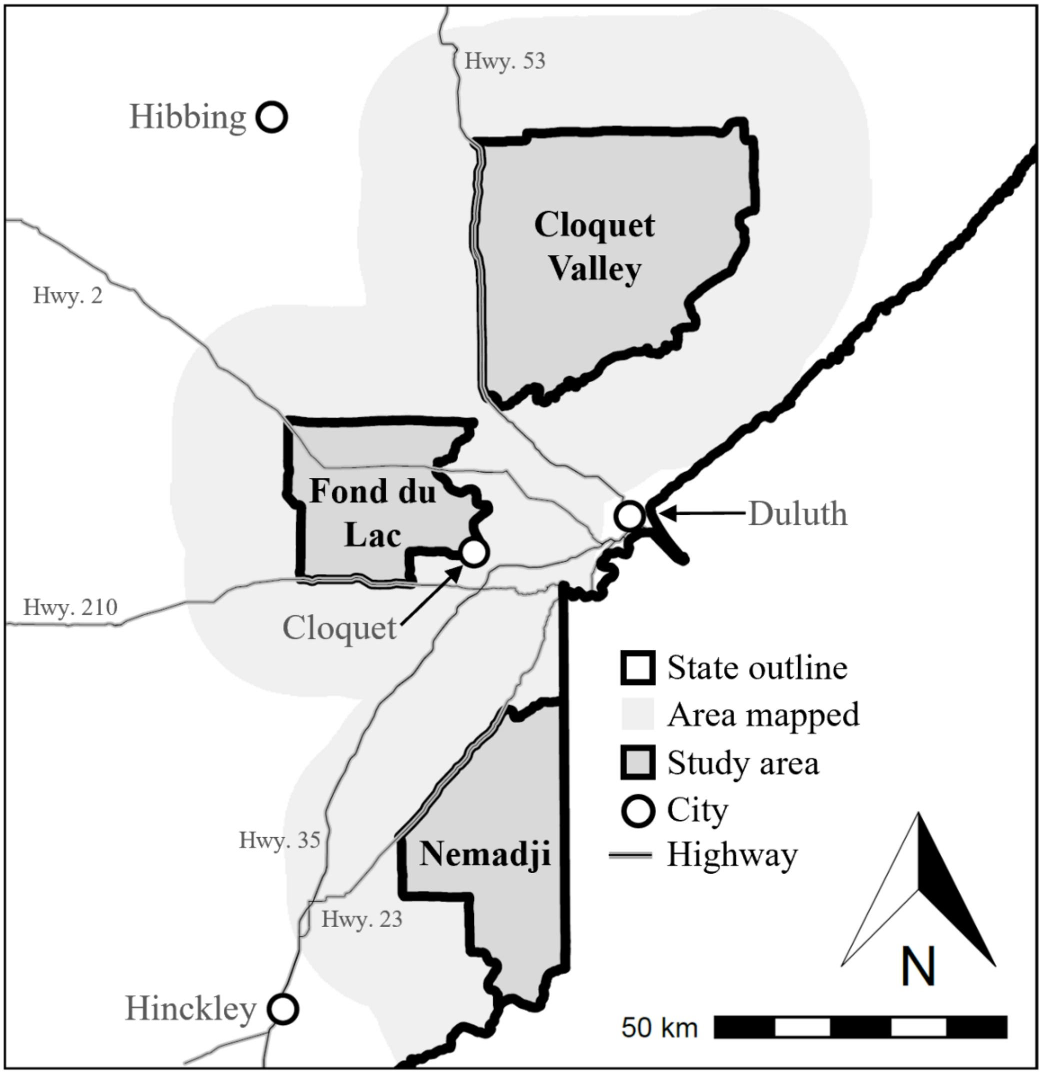

Figure S-2. Study areas in northeastern Minnesota where we studied the feasibility of restoring elk. Larger cities and highways included to serve as references. Maps we created were for the 3 study areas and the 20 km (12 mi) surrounding them. Scalebar: 50 km = 31 mi.

We used multiple methods to evaluate and map elk habitat suitability and social support. We measured potential summer (leaf-on) and winter (leaf-off) forage in the field and combined forage data with remotely sensed data to estimate the number of elk likely to be supported by each study area. We mapped habitat suitability index scores and a resource selection function, each developed in Wisconsin. Data from roads, feedlots[1], row crops, and hay and pasture fields enabled us to create a risk map for human-elk conflict, and data from mail-in questionnaires

[1] Feedlots, as defined by the state of Minnesota, are open land without maintained vegetation and buildings where producers hold animals for feeding. Pastures are not feedlots but the 2 often occur together (MPCA 2007).

enabled us to map support for elk restoration by landowners and local residents. In the end, we ranked study areas and tested the influence of considering some factors as being more important than others. Our findings show that habitat suitability and landowner support are not limiting factors for restoring elk to northeastern Minnesota.

Key finding 1: Mean summer forage at field plots exceeded amounts elk prefer and winter forage matched amounts where elk occur in Wisconsin.

We sampled 186 field plots: 63 plots on the Cloquet Valley study area, 69 on the Fond du Lac study area, and 54 on the Nemadji study area. Public land had more winter forage than private land, forested shrub wetlands had more winter forage than grasslands, and grasslands had more summer forage than coniferous forests and mixed forests. Mean summer forage at field plots was 0.130 kg/m^2 (0.426 oz/ft^2) and mean winter forage was 0.017 kg/m^2 (0.056 oz/ft^2).

Key finding 2: Our estimates of how many elk are likely to be supported during winter indicate that northeastern Minnesota can support densities similar to Wisconsin and Michigan.

Mean estimates of how many elk likely to be supported during winter on each study area ranged from 5 to 8 elk/16 km^2 (5 to 8 elk/6 mi^2; Figure S-3). These estimates correspond well with elk densities in Wisconsin's Black River Herd and in Michigan, and they are higher than Wisconsin's Clam Lake Herd. Estimates of how many elk likely to be supported during summer were much higher, ranging from 14 to 83 elk/16 km^2 (14 to 83 elk/6 mi^2) across the study areas. We focused on winter when determining how many elk can be supported, however, as it is the limiting season for wildlife population growth, including for elk.

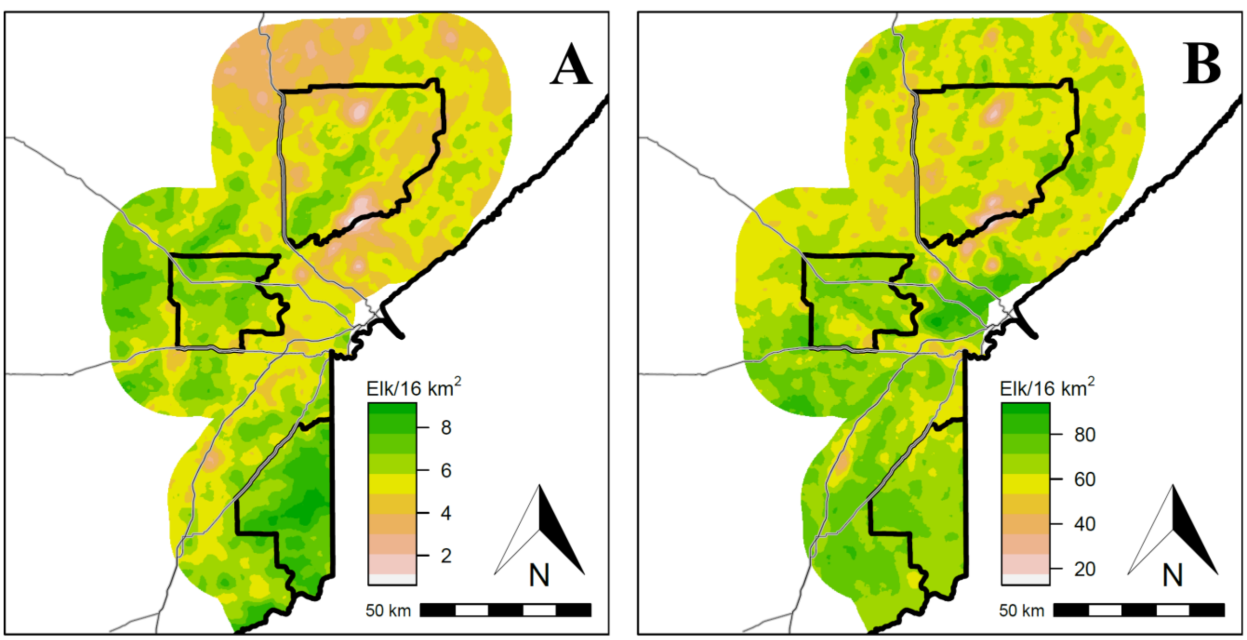

Figure S-3. Estimates of how many elk likely to be supported during winter (A) and summer (B) in northeastern Minnesota. Scalebar: 16 km^2 = 6 mi^2 and 50 km = 31 mi.

Key finding 3: Estimates of biological carrying capacity ranged from 287 on the Fond du Lac study area to 551 elk on the Cloquet Valley study area.

Carrying capacity estimates based on how many elk are likely to be supported during winter are in Table S-1. These estimates probably underestimate biological carrying capacity as we assumed elk consume only a small proportion of available forage. Estimates also do not account for cultural carrying capacity, which may be different.

Table S-1. Winter biological carrying capacity estimates for 3 study areas in northeastern Minnesota. Estimates are for the 3 study areas but not the surrounding areas.

Study area	Area (km^2)	Area (mi^2)	Carrying capacity (range)
Cloquet Valley	1,764	681	551 (335 to 768)
Fond du Lac	766	296	287 (193 to 381)
Nemadji	963	372	481 (364 to 599)

Key finding 4: Each of the 3 study areas had large amounts of habitat with suitability scores similar to where elk occur in Wisconsin.

Suitability maps of winter forage (Figure S-4A), spring forage (Figure S-4B), and winter cover (Figure S-4C) resulted in a map of overall suitability (Figure S-4D), ranging from 0 to 0.68 (higher values are better). The Cloquet Valley study area contained about 4-times more suitable habitat than the Black River Herd's core area in Wisconsin, while the Nemadji study area contained about 2-times more, and the Fond du Lac study area contained about the same amount.

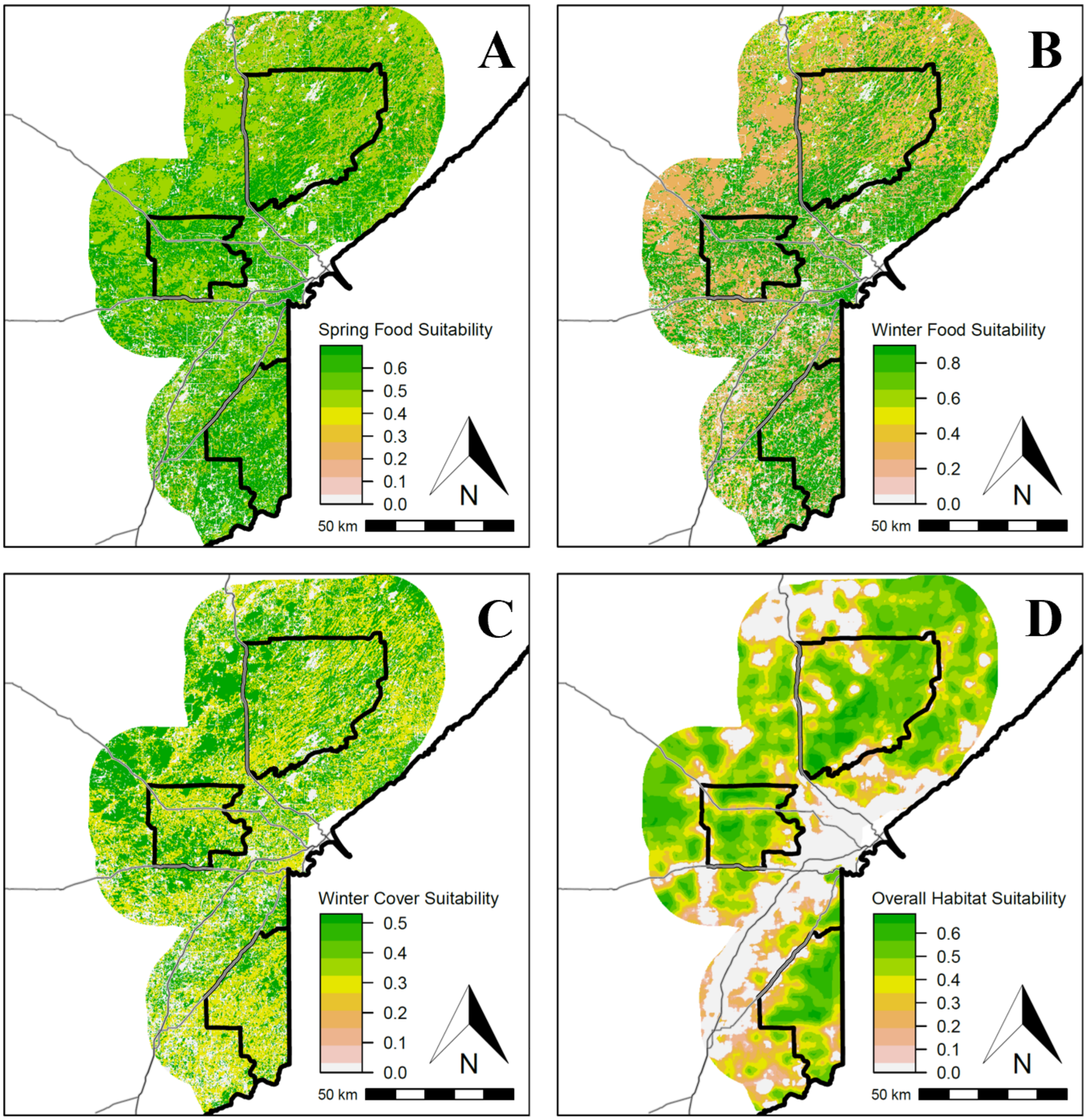

Figure S-4. Spring food (A), winter food (B), winter cover (C), and overall suitability indices (D) for elk in northeastern Minnesota when a habitat suitability model from Wisconsin was used. Higher values indicate better suitability. Scalebar: 50 km = 31 mi.

Key finding 5: Resource selection function maps showed the greatest amount of summer elk habitat on the Nemadji study area.

Summer resource selection function scores reflected increasing relative probability of selection (1 = low probability and 4 = high probability). When we included known wolf territories in relative selection calculations, the Nemadji study area had mean selection scores 2.6- and 3-times higher than the Cloquet and Fond du Lac study areas (Figure S-5A). Selection was high on the Nemadji study area because wolf pack territory location influenced selection scores, but we were missing wolf territory data from packs that occur there. When we excluded the influence of wolf territories in selection calculations, mean selection score differences were smaller (Figure S-5B); the Nemadji study area had mean selection scores 1.5- and 1.3-times higher than the Cloquet Valley and Fond du Lac study areas.

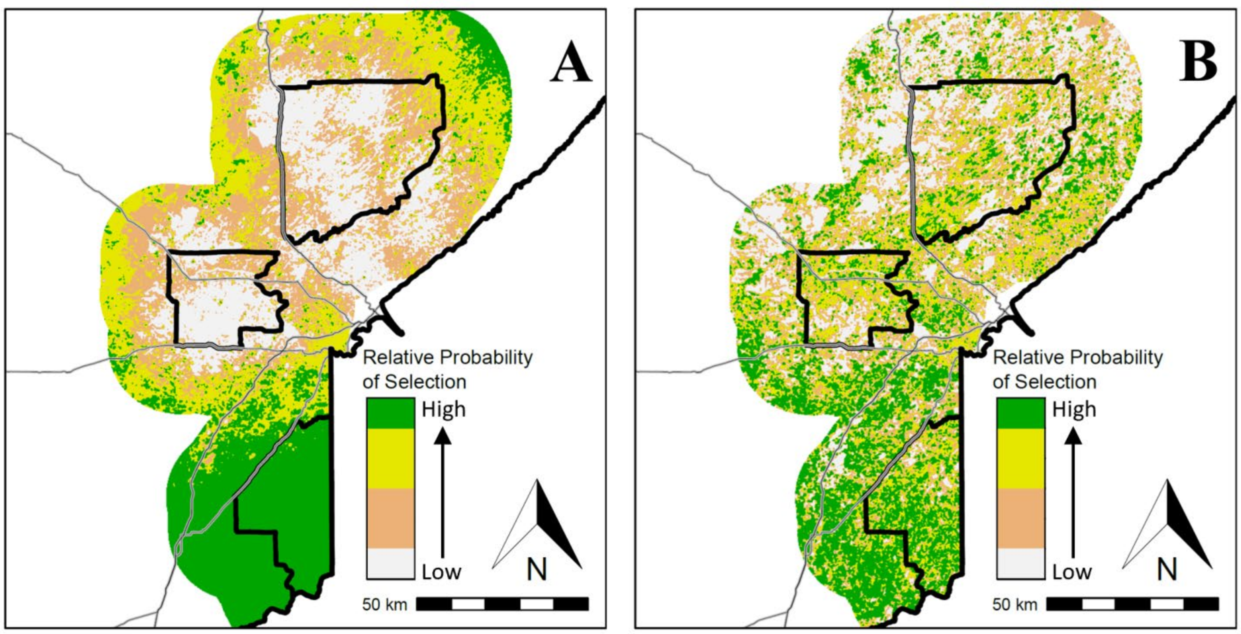

Figure S-5. Relative probability of resource selection by elk in northeastern Minnesota with (A) and without (B) influence of monitored wolf packs. Recent wolf territory data were unavailable for the Nemadji study area. Scalebar: 50 km = 31 mi.

Key finding 6: Aspen was more abundant in the Cloquet Valley study area than in the other study areas, while grassland was distributed similarly across the study areas.

Abundance of aspen (a selected elk forage) was 4.3 times greater on the Cloquet Valley study area (mean proportion = 0.17 aspen) than on the Fond du Lac study area (0.04 aspen), and 5.7 times greater than on the Nemadji study area (0.03 aspen; Figure S-6A).

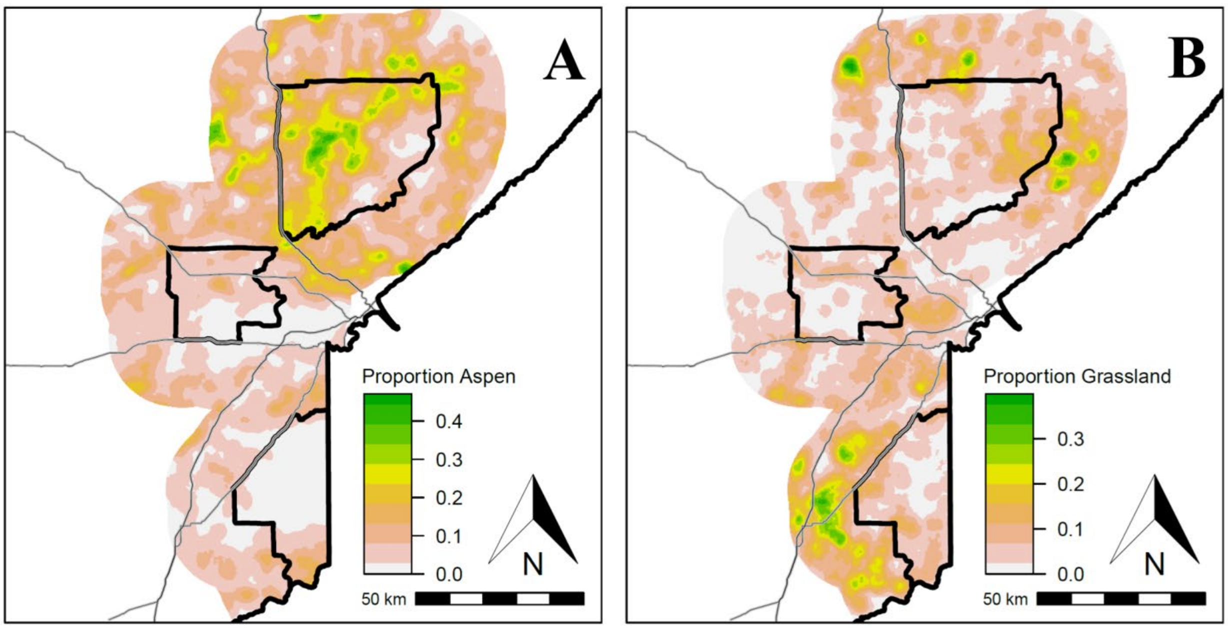

Figure S-6. Proportion of area that was aspen (A) and grassland (B) in northeastern Minnesota. Scalebar: 50 km = 31 mi.

Key finding 7: Public land made up the majority of all 3 study areas and was most abundant on the Cloquet Valley study area than in the other study areas.

The Cloquet Valley study area was 0.75 public land, while the Fond du Lac study area was 0.61 public, and the Nemadji study area was 0.60 (Figure S-7).

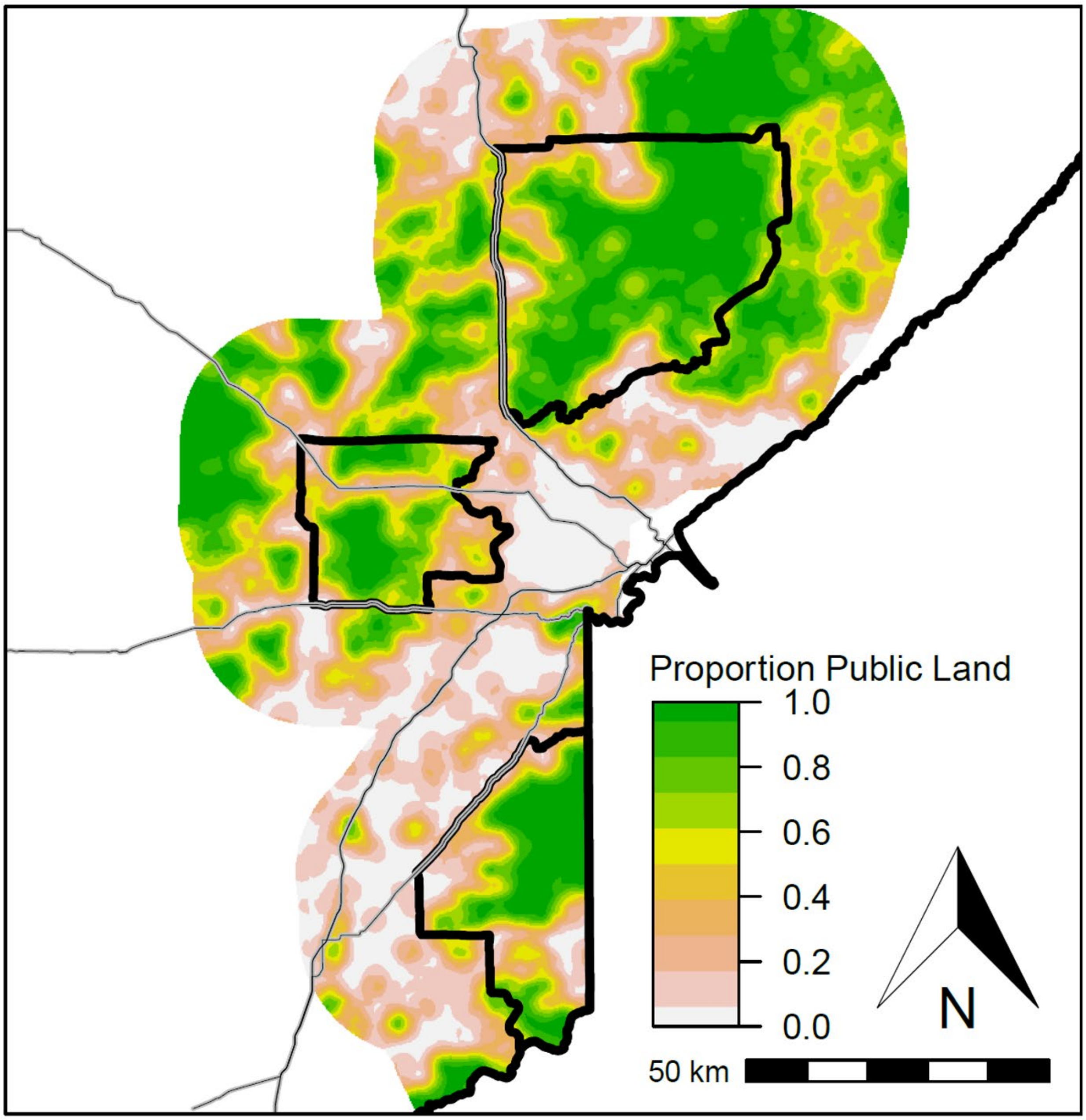

Figure S-7. Proportion of public land in northeastern Minnesota. Scalebar: 50 km = 31 mi.

Key finding 8: Most landowners and local residents supported elk restoration, and support was similar across study areas.

Overall, 82% of landowners (people who owned ≥ 4 ha of land) inside the 3 study area boundaries and 86% local residents (owned < 4 ha of land) with addresses inside the 3 study area boundaries expressed favorable attitudes toward elk restoration (Table S-2). Landowners and local residents had about the same level of acceptance for elk restoration on each study area. Our results correspond well with those from a companion study that focused on social acceptance in greater detail by Walberg et al. (2019). That study included additional statistical analysis of landowners and local residents from inside and outside the study areas.

Table S-2. Acceptance scores for landowners and local residents inside the boundaries of 3 northeastern Minnesota study areas. A companion study by Walberg et al. (2019) that included additional analysis of questionnaire responses from landowners and local residents from inside and outside study area boundaries had similar results.

	Landowners: count (proportion) of acceptance scores [a]							
Study area	1	2	3	4	5	6	7	Sum
Cloquet Valley	24 (0.07)	9 (0.03)	8 (0.02)	23 (0.07)	37 (0.11)	90 (0.27)	142 (0.43)	333
Fond du Lac	16 (0.08)	4 (0.02)	4 (0.02)	22 (0.11)	17 (0.08)	69 (0.33)	77 (0.37)	209
Nemadji	20 (0.05)	11 (0.03)	7 (0.02)	26 (0.06)	45 (0.11)	103 (0.25)	196 (0.48)	408
Sum (landowners)	60 (0.06)	24 (0.03)	19 (0.02)	71 (0.07)	99 (0.10)	262 (0.28)	415 (0.44)	950

	Local residents: count (proportion) of acceptance scores							
Study area	1	2	3	4	5	6	7	Sum
Cloquet Valley	6 (0.08)	1 (0.01)	0 (0.00)	6 (0.08)	13 (0.17)	18 (0.23)	34 (0.44)	78
Fond du Lac	0 (0.00)	1 (0.05)	0 (0.00)	1 (0.05)	4 (0.18)	8 (0.36)	8 (0.36)	22
Nemadji	1 (0.05)	0 (0.00)	0 (0.00)	1 (0.05)	2 (0.09)	8 (0.36)	10 (0.45)	22
Sum (local residents)	7 (0.06)	2 (0.02)	0 (0.00)	8 (0.07)	19 (0.10)	34 (0.28)	52 (0.43)	122

[a] 1 = low acceptance, 4 = neutral, 7 = high acceptance

Using questionnaire responses from 2,585 landowners and 1,521 local residents from inside and outside the study areas, we mapped social acceptance scores (ranging from 1 = unfavorable toward restoration to 7 highly favorable; 4 = neutral) and found landowner and local resident acceptance was high (Figure S-8). Landowner acceptance ranged from 5.5 on the Fond du Lac study area to 5.8 on the Nemadji study area, and local resident acceptance ranged from 5.4 on the Fond du Lac study area to 5.7 on the Nemadji study area.

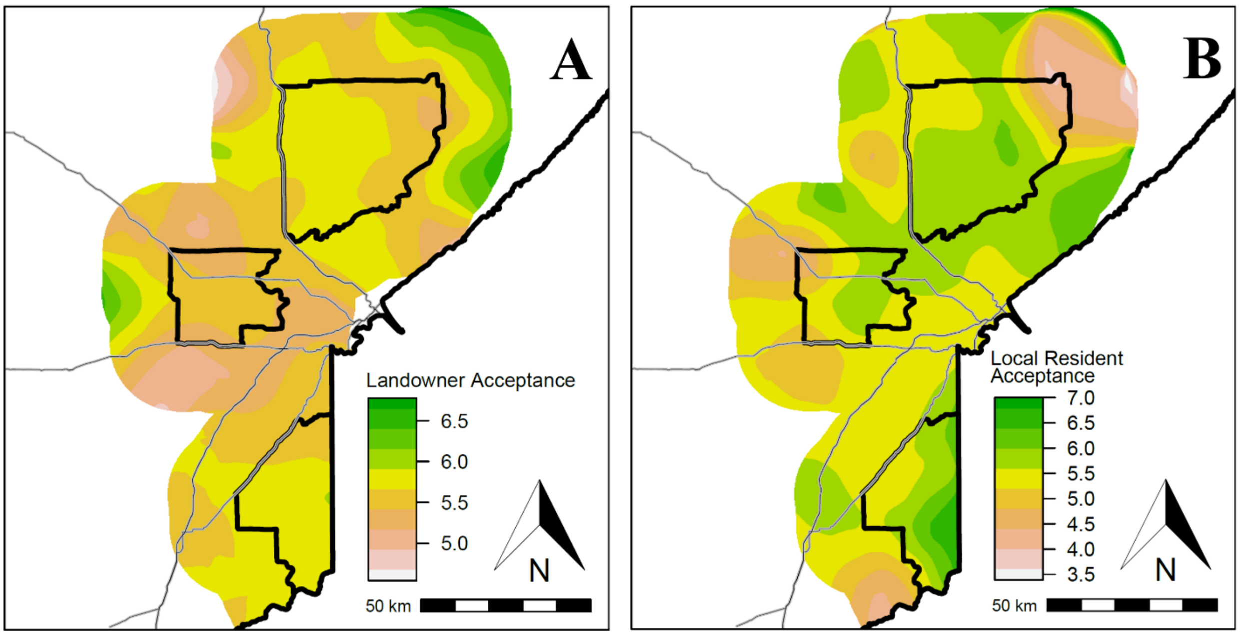

Figure S-8. Social acceptance of elk restoration by landowners (A) and local residents (B) on and near 3 study areas in northeastern Minnesota. Acceptance ranges from 1 (low) to 7 (high), with 4 being neutral. The scale bars start at values > 0 as minimum mean acceptance was 4.5 for landowners and 3.4 for local residents. Scalebar: 50 km = 31 mi.

Key finding 9: Human-elk conflict risk was low on the 3 study areas, but risk adjacent to our study areas may influence public support for elk population expansion.

Human-elk conflict risk (proportion of area made up of roads, feedlots, row crops, and hay/pasture fields) was low (mean risk ≤ 0.10) across each of the 3 study areas (Figure S-9). It increased from north to south, with the Nemadji study area having mean risk 5-times greater than for the Cloquet Valley study area. Low conflict risk in all directions adjacent to the Cloquet Valley study area may enable elk population expansion without eroding public support. The same is true to the west, north, and east of the Fond du Lac study area, but areas outside the Nemadji study area (in all directions within the state), had higher risk of human-elk conflict.

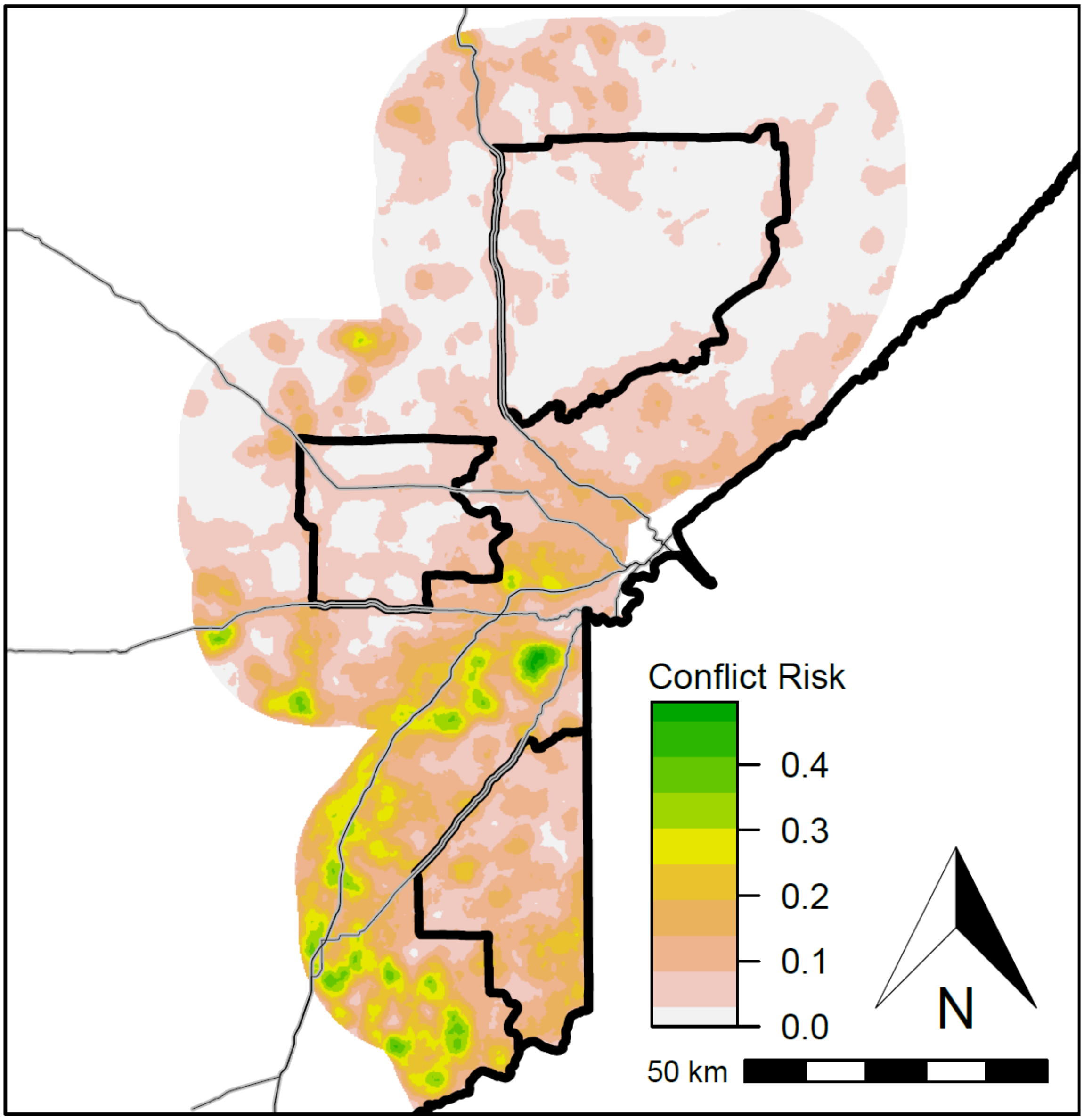

Figure S-9. Risk of human-elk conflict (proportion of area that is roads, feedlots, and row crops). Scalebar: 50 km = 31 mi.

Key finding 10: Considering factors we assessed to be equally important (evenly weighing them) did not result in statistically different study area rankings (on average, all 3 study areas were about the same). Some study areas ranked better than others, however, when we weighted factors (considered some factor to be more important than others).

It required weighting factors about 6 times to arrive at different ranks for each study area (Figure S-10), which means that a factor has to be considered to be 6 times more important before any 1 study area is found to be better than another. The Cloquet Valley study area ranked best most often (after weightings), followed by the Nemadji study area and the Fond du Lac study area.

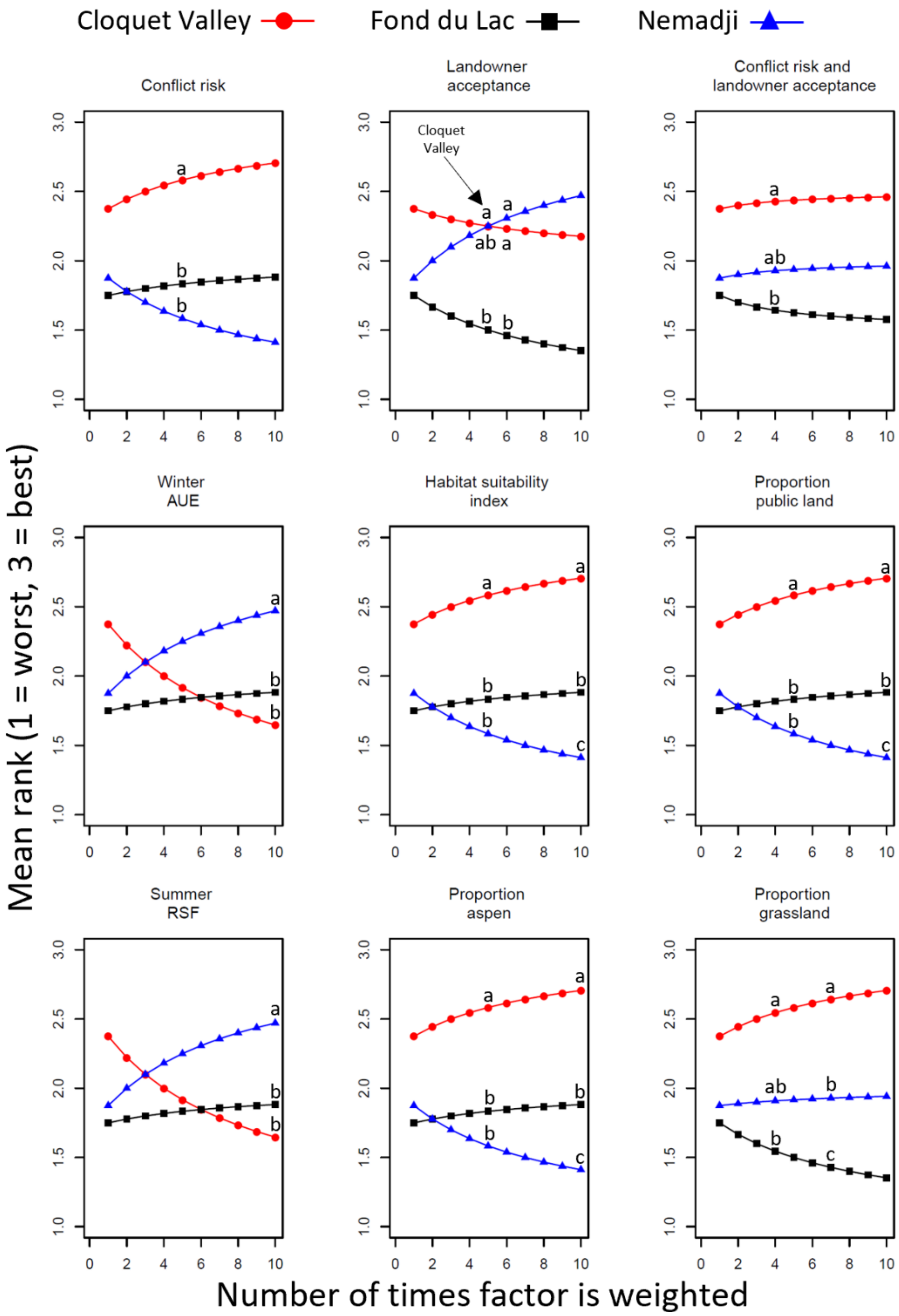

Figure S-10. Mean rank of 3 northeastern Minnesota study areas after weighting factors between 1 (even weights) to 10 (factor counted 10 times). Plot titles indicate weighted factor(s). Letters a, b, and c within plots are first instances of statistically significant pairwise differences that continue to be significant at greater weights. For example, the Cloquet Valley rank was first statistically greatest at conflict risk weight = 5 and continued to be greater at ranks ≥ 5. There were 2 sets of differences for panels that have >1 set of letters. AUE = animal use equivalence. RSF = resource selection function.

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Introduction

Elk (*Cervus canadensis*) historically ranged over most of North America and numbered in the millions, but their numbers declined with overexploitation and habitat loss following European colonization (Murie 1951). Remnant populations in western North America became sources for restorations, and at least 24 elk restorations occurred in eastern North America (Popp et al. 2014). Restoration success varied but has improved (Popp et al. 2014) along with maturation of restoration science (Seddon et al. 2007, Armstrong and Seddon 2008).

Multiple factors influence the success of animal restorations, including the source population (wild or captive), number of animals released, genetics, and competition (Fischer and Lindenmayer 2000). Releasing animals into suitable habitat also influences success (Armstrong and Seddon 2008), as it provides cover from predators and weather, and forage. In eastern North America, winter elk forage is deciduous shrub and tree current year growth (twigs; Jenkins et al. 2007). In an Ontario snow-tracking study, elk selected quaking aspen (*Populus tremuloides*; hereafter aspen) as forage and aspen was the most abundant winter diet item (16% of forage species at feeding stations; Jenkins et al. 2007). Sugar maple (*Acer saccharum*) was also important (11% of forage species), but elk did not select it consistently. Other prominent winter forage species were choke cherry (*Prunus virginiana*), willow (*Salix* spp.), beaked hazel (*Corylus cornuta*), and roundleaf dogwood (*Cornus rugosa*), but elk rarely consumed coniferous species and almost never cratered (excavated snow) to consume grass (Jenkins et al. 2007). The elk diet is different in summer, consisting of forbs, grasses, and deciduous shrub and tree leaves in eastern North America (Schneider et al. 2006, Lupardus et al. 2011). Pellet analyses showed that forbs, grasses, ferns, and legumes were 85% of the elk diet in summer in Tennessee (Lupardus et al. 2011) while >84% was forbs, grasses, and deciduous leaves in Kentucky (Schneider et al. 2006). We did not find diet studies from wild elk in eastern North America located closer to Minnesota.

Forage availability is assessed by field studies that estimate forage biomass. Because it is difficult to measure the mass of shrubs and trees in an area large enough to characterize natural heterogeneity, however, allometric equations estimate biomass using diameter measurements from the main stem of a shrub or tree (at breast height for larger stems and 15 cm for smaller stems). Equations are from species-specific regressions that correlate stem diameter with biomass after harvesting, drying, and massing above ground biomass (Jenkins et al. 2004). In addition to equations for total above ground biomass (for the entire tree or shrub), studies calculate equations for leaves (Smith and Brand 1983, Perala and Alban 1993) and twigs (Grigal et al. 1976). Forage estimates are linked to GIS landcover types, resulting in maps that estimate forage across broad areas (Anderson et al. 2005*a*, Coe et al. 2011).

Maps of broad-scale forage are used to calculate resource selection functions and estimates of how many animals can be supported in an area. Resource selection functions quantify the probability of an animal using 1 area instead of another area by combining data about animal space use (radio telemetry and GPS collar location data) with habitat data (Anderson et al. 2005*b*). Once calculated, these functions predict areas animals will use disproportionately (Anderson et al. 2005*a*, Coe et al. 2011). Estimates of how many animals an area can support combine forage and energetics data (Kuzyk and Hudson 2007). Feeding trial studies, for example, reveal how much forage elk require to maintain body condition (Christianson and Creel 2009). This information enables calculations of the number of animals supported by available forage, termed animal use equivalence (AUE; Kuzyk and Hudson 2007).

Another common method for assessing habitat suitability is developing habitat suitability indices. Such indices assessed feasibility of restoring elk in many locations, including Arkansas, New York, North Carolina, and Ohio (Didier and Porter 1999, Telesco et al. 2007, Karns et al. 2015, Williams et al. 2015). Habitat suitability indices assign scores to GIS landcover types that range from 0 (unsuitable) to 1 (highly suitable). Grassland, for example, receives a high score for summer elk forage while coniferous forest receives a low score. Habitat suitability indices also assign scores to landcover maps related to human land-use. Road surfaces, for example, are unsuitable due to elk-vehicle collisions. Once landcover maps are scored, a moving window quantifies suitability. Moving windows calculate suitability in the surrounding area for each point on a map. The value of each point on the resulting moving window map represents that location and the area around it, thereby reflecting the fact that while a given point may be good or bad, the suitability of that location is related to its surroundings. This makes sense because elk use big areas, so the quality of a given location is related to both the location and its surroundings. A location in a small gravel pit, for example, is poor habitat. After applying a moving window, however, that location scores higher when surrounded by good habitat. Conversely, a small island with good habitat scores poorly when surrounded by a big lake (open water is poor elk habitat). Moving window size approximates areas elk typically use (home range, 16 km^2; Van Deelen et al. 1997; Didier and Porter 1999; O'Neil and Bump 2014).

In addition to biological considerations such as habitat suitability, public support is important when restoring wildlife as restorations are more successful when people accept restored species (Fischer and Lindenmayer 2000). A common way to assess acceptance is with questionnaires (Walberg et al. 2017). Researchers score questionnaire responses on a scale, with scores corresponding with acceptance (Schroeder et al. 2018). For example, questionnaires are scored from 1 to 7, with 1 equaling low support, 4 equaling a neutral position (not in opposed or supportive), and 7 indicating high support (Walberg et al. 2019). Mapping scores facilitates assessment of where social acceptance is greater and where it is lower (Behr et al. 2017).

Even when public support is strong before restoration, it erodes when human-wildlife conflicts occur afterwards, making it important to assess conflict risk. For elk, multiple factors reduce public support (Hegel et al. 2009, Walter et al. 2010). Elk select road right of ways with high amounts of forage (grasses and forbs; Anderson et al. 2005*b*), which likely increases risk of vehicle collisions. Elk are costly to producers and agencies that institute compensation programs when they damage fences, and depredate row crops, hay bales, grain, and silage (Hegel et al. 2009, Walter et al. 2010, MNDNR 2017). Disease transmission with domestic livestock is another concern for producers. Accordingly, management strategies minimize elk-livestock contact at livestock feedlots (Byrne 1989, MNDNR 2017), which are open land (without maintained vegetation) and buildings where producers hold animals for feeding (pastures are not feedlots but the 2 often co-occur; MPCA 2007). Assessing conflict risk with habitat suitability and social acceptance will improve understanding of where elk restoration will be successful.

This study examines the feasibility of restoring elk to northeastern Minnesota. Elk once occupied most of Minnesota but were functionally extirpated by the early 1900s (MNDNR 2017; Figure 1). There are currently 3 groups of elk in northwestern Minnesota. The state manages them at low levels by statute[2] as elk have damaged fences and depredated agricultural crops. This state statute does not apply to northeastern Minnesota, where the likelihood of human-elk conflict is lower. Along with a companion study that focused specifically on social acceptance (Walberg et al. 2019), this study provides information for determining where elk restoration will be successful in northeastern Minnesota, should it occur. It assesses habitat suitability, social acceptance, and human-elk conflict.

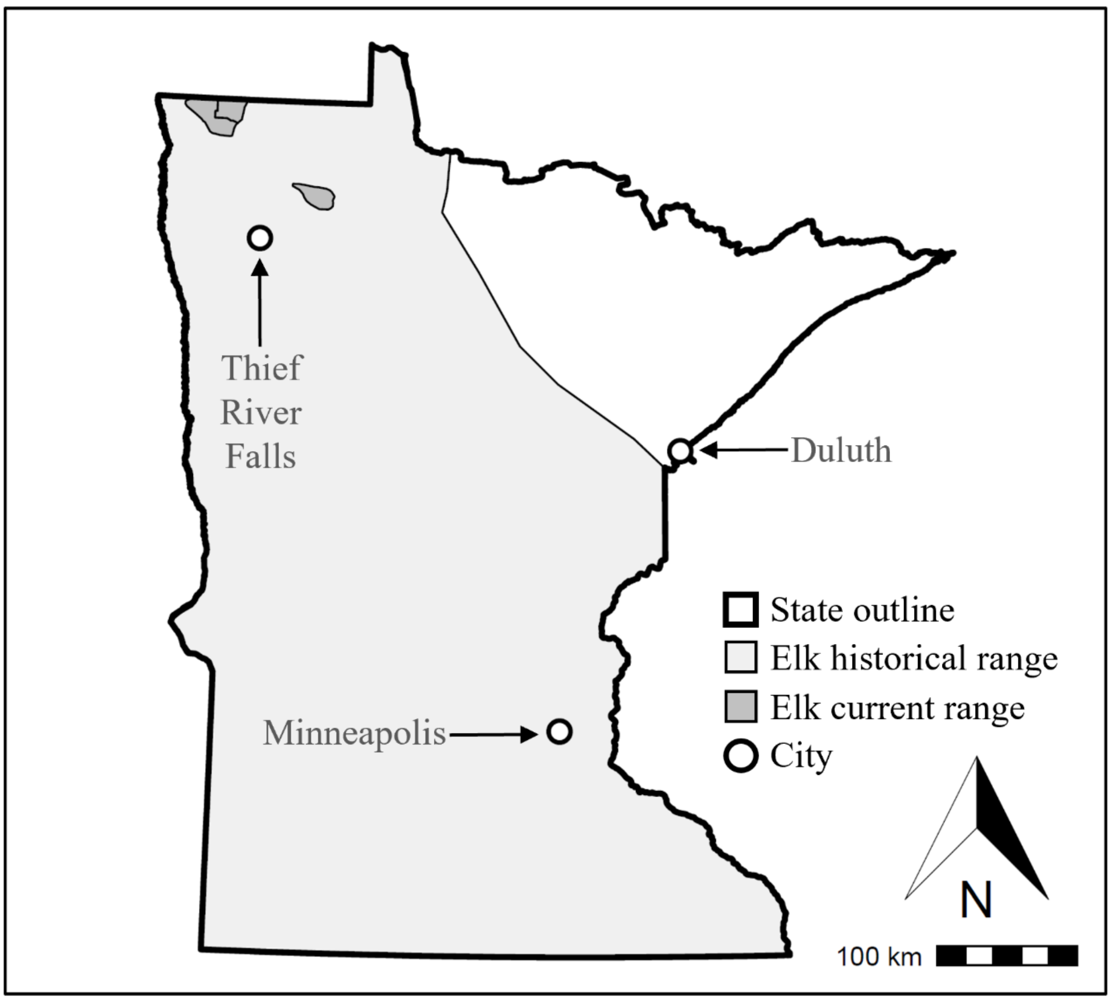

Figure 1. Historical and current elk range in Minnesota. Three cities are included to serve as references.

[2] Minnesota Statute 97B.516 does not allow for an increase in elk population size in Kittson, Roseau, Marshall, or Beltrami Counties unless the commissioner of agriculture verifies that crop and fence damages paid under section 3.7371 and attributed to the herd have not increased for at least two years.

Methods

Study area

We studied habitat suitability social support for elk on and near 3 study areas in northeastern Minnesota (Figure 2). Study areas were comprised mostly of public land (Table 1) and had low road densities (mean = 0.96 km/km^2, SD = 0.19 km/km^2, N = 3) that are suitable for elk (< 2 km/km^2; Lyon 1983; Beazley et al. 2004). The area was in the northern lakes and forests ecoregion (Level III Region 50), with often rolling topography, relatively nutrient-poor glacial soils, and scattered lakes and rivers (Omernik and Griffith 2014). Forests were coniferous and northern hardwood types, and forest stands were often mixed. Maps we created (described below) were for the 3 study areas and the surrounding 20 km (corresponding to elk dispersal distance in Ontario, near the Minnesota border; Ryckman et al. 2010).

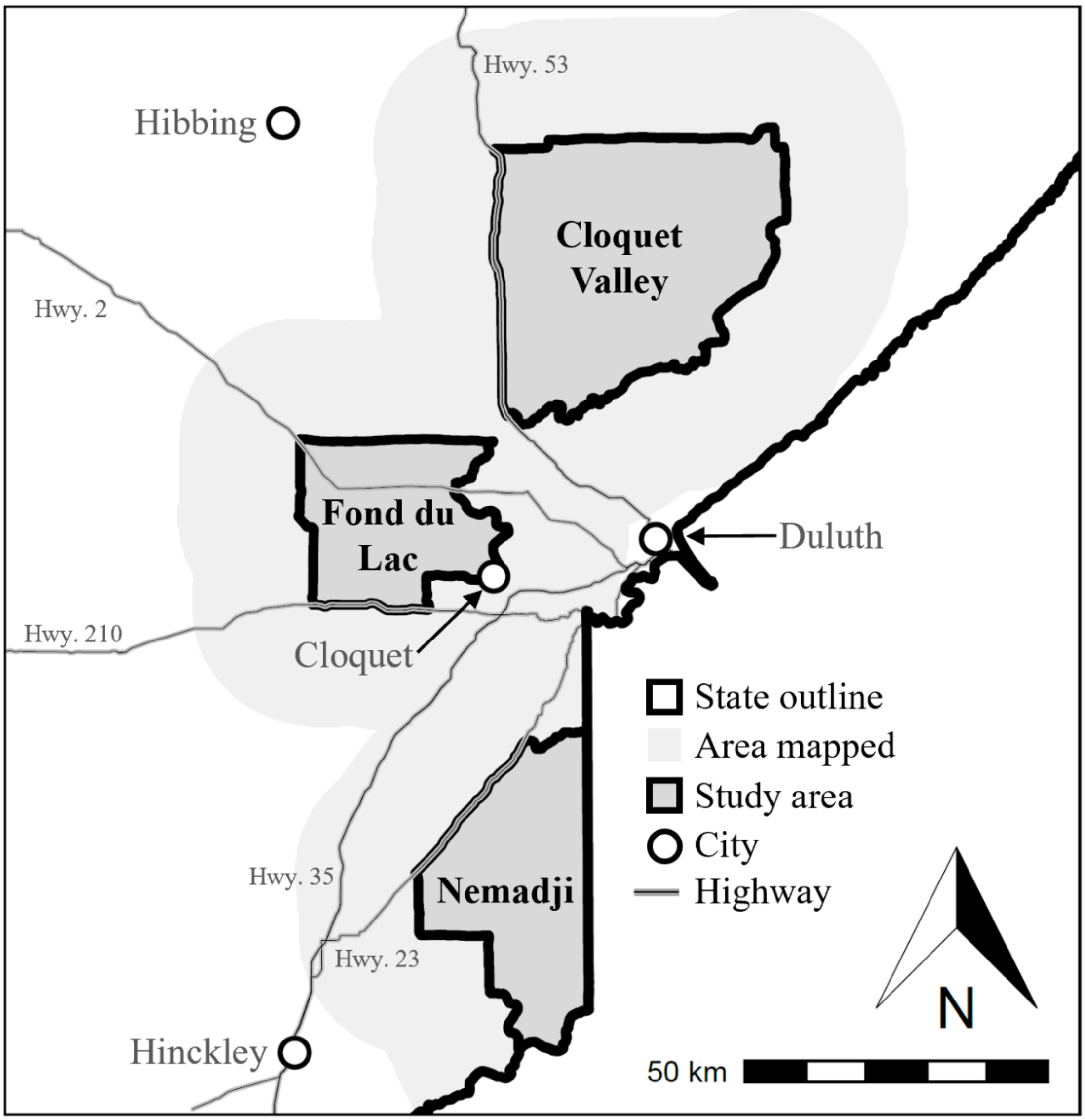

Figure 2. Study areas in northeastern Minnesota where we studied the feasibility of restoring elk. Larger cities and highways are included to serve as references.

Table 1. Ownership areas and proportions on 3 study areas (but not surroundings) in northeastern Minnesota.

	Area in each ownership classification (km^2) [a]								
Study area	Federal	Tribal	State [b]	County [b]	Private	Private industrial	Total	Total public (proportion)	Total private (proportion)
Cloquet Valley	153	0	1162	5 [c]	316	128	1764	1320 (0.75)	444 (0.25)
Fond du Lac	17	69	379	8	272	21	766	473 (0.62)	293 (0.38)
Nemadji	0	1	579	2	372	8	963	582 (0.60)	380 (0.40)

[a] Private non-industrial and private conservancy classifications not included; each was 0 km^2.

[b] State totals appear large and county totals appear small as many properties managed by counties are state-owned (tax-forfeited).

[c] Includes 2 km^2 of classification: other public.

Forage estimation

<u>*Field plots*</u>

We measured trees, shrubs, and understory vegetation at sites distributed throughout the 3 study areas between June 14 and August 8, 2017 (hereafter, season 1) and June 6 and August 8, 2018 (season 2). Sampling occurred on public land during season 1 and on private land during season 2. During season 1 we randomly distributed points on roads that abutted public land. We then randomly distributed 1 point in each vegetated landcover type (Rampi et al. 2016) that was within 50 to 500 m of the road point (to improve logistics) and randomly selected cover type points to sample (with periodic adjustments for even sampling of cover types). During season 2 we randomly distributed 1 point within each cover type that was within 50 to 500 m of a road that abutted private properties and randomly selected cover type points to sample. We distributed sampling points in R (R Core Team 2019) and ArcMap 10.5 (ESRI 2018).

To achieve sampling that was even through space (with respect to roads) and time (with respect to field season duration), we selected road points or properties using a stratified-random design, whereby strata were study area and rectangular quadrants (equal in area) overlaid on each study area in a GIS. We sampled study areas and quadrants systematically. For example, we selected the Nemadji study area and then selected road points or properties within quadrant 1. The next day we sampled Nemadji quadrant 2, and so forth until we sampled all 4 Nemadji quadrants. We repeated the same process in the other study areas before returning to the Nemadji study area.

Before season 2 we selected private properties to sample from the population of landowners who responded to a mail questionnaire (Walberg et al. 2019). We used a stratified random approach to select private properties to sample, whereby strata were study area and rectangular quadrant. We randomly selected landowners from each study area and contacted them by email ($N = 45$) and phone ($N = 41$), resulting in access from 47 landowners to 66 private properties (Cloquet Valley study area, $N = 16$; Fond du Lac study area, $N = 28$; Nemadji study area, $N = 22$). We then randomly selected properties from study area quadrants that we sampled systematically. During season 2 we also stratified sites for sampling in aspen regeneration on public lands by randomly selecting county-managed forest stands where harvest occurred within the last 10 years (M.P. Westphal, Carlton County; D. Ryan, US Forest Service; J. Kelash, Pine County; and B. Hakala, Saint Louis County; *Unpublished data*).

We established a circular plot centered on each cover type point (hereafter referred to as a field plot). Each field plot comprised nested circles within which we sampled trees, shrubs, and herbaceous vegetation. The largest circle was 401 m^2 (11.3 m radius). In this circle we measured diameter at breast height (DBH; 1.4 m) of trees > 10 cm DBH with a diameter tape. Three medium circles (25 m^2 plots; 2.8 m radius) radiated 5.5 m from the plot center at azimuths of 30°, 150°, and 270°. In these circles we used a stepped diameter gauge (*Sensu* Paul et al. 2017) to count trees and shrubs that were 2.54 cm to 10 cm DBH. We centered a small circle (10 m^2; 1.8 m radius) within each medium circle. In small circles we used a stepped diameter gauge to count trees and shrubs that were ≥ 15 cm tall and < 2.54 cm diameter (measured at 15 cm height, D15). Counts for stems < 2.54 cm were in 0.5 cm increments (e.g., number of stems between 0.5 cm and 1 cm) while counts for stems 2.54 cm to 10 cm were in 1 cm increments.

In addition to measuring and counting trees and shrubs at each field plot, we collected ground cover vegetation from 10 1.5 m^2 rectangular (150 cm x 10 cm) quadrats. One quadrat was at the center of each plot, 1 was at the center of each medium circle, and 1 was at each 60° increment (starting at an azimuth of 30°) along the border of the large circle. We clipped (from 1.5 cm above the ground) woody vegetation that was < 15 cm tall and herbaceous vegetation (all heights) within each quadrat. We classified clipped vegetation as: grasses, forbs, sedges, rushes, ferns, and woody vegetation. We then dried clipped vegetation at 40 °C (Isotemp Gravity and Convection Oven; Fisher Scientific, Pittsburgh, Pennsylvania) for 48 h and recorded mass for each group. Other data collected at plots included canopy cover (using a densiometer), aspect, slope, and visually estimated height and percent ground cover at vegetation quadrats.

Right of way plots

We measured forage adjacent to paved and gravel roads, railroad tracks, and pipelines (hereafter, right of ways) in season 2. There were 4 road classes sampled: county, federal, state, and township. Pipelines were underground and the area above them was open (managed to remove trees and shrubs) with a lightly maintained 2-track service road. Areas adjacent to roads and railroads were also open.

We used a random-stratified approach to select right of way locations for sampling, where strata were study area and road class (for sampling road right of ways). We selected road right of way locations by plotting points randomly on roads (MNDOT 2017). We randomly selected railroad (MNDOT 2015) and pipeline locations (traced in Google Earth) that intersected roads (to ensure access) and randomly selected a distance 50 to 500 m from the intersection for sampling. As each right of way feature (e.g., road) was bordered by 2 open areas (1 on either side), we randomly selected a side of the right of way for measuring forage. We supplemented road right of way data by sampling right of ways that abutted private properties we accessed.

At each right of way sampling site, we established a 200 m^2 rectangular plot (hereafter, right of way plot). We measured the distance between the edge of the road, pipeline service road 2-track, and railroad and the nearest tree line or shrub line. This measurement was the width side of the plot. The length side of the plot paralleled the road or railroad. We calculated the plot length side by dividing 200 by the width side measurement. We used a plot width of 30 m where a tree line or shrub line was > 30 m from the road or railroad edge. Once we established a right of way plot, we clipped ground cover vegetation from 5 quadrats: 1 placed at the plot center and 1 placed at each plot corner. This resulted sampling intensity that was similar to field plots (1 quadrat/40 m^2). Quadrat dimension, vegetation classifications, and drying methods were the same as described for field plots. In addition to clipping vegetation at right of way plots, we measured the distance between the tree lines or shrub lines that bordered each side of the right of way and the width of the railroad bed or road using a laser range finder. This resulted in a total right of way width (for both sides of the right of way) including the railroad bed and road surface.

Forage estimates

We estimated forage available to elk at each field plot during 2 periods: leaf-on (hereafter, summer) and leaf-off (hereafter, winter). Summer forage at each field plot was the sum of biomass from leaves, forbs, and grasses, as these are the most frequently consumed items by elk in eastern North America during summer (Schneider et al. 2006, Lupardus et al. 2011). To estimate forage from shrubs and sapling leaves, we summed woody stem counts from medium and small circles at each field plot (for each diameter class separately) and used allometric equations to estimate leaf biomass (Smith and Brand 1983, Perala and Alban 1993). All forage estimates were converted to kg/m^2. We estimated forb and grass forage by summing the masses of forbs and grasses that we collected, dried, and massed at each field plot.

As elk in eastern North America forage on deciduous shrub and tree current year growth (twigs) during winter (Jenkins et al. 2007), we estimated current year growth using allometric equations like those used to estimate leaf biomass. Instead of estimating leaf biomass (equations unavailable for most species), however, we estimated total above ground biomass (Smith and Brand 1983, Perala and Alban 1993) at each field plot and estimated current year growth as the product of biomass and proportion of biomass that is current year growth in Minnesota (0.07; (Ohmann et al. 1974, Ohmann et al. 1976).

We estimated forage biomass for shrubs and trees that were ≤ 2.54 cm in diameter (D15) as this diameter corresponds with mean height at which elk forage (1.5 m; Rounds 2006; Gehring et al. 2008; VanderSchaaf 2013). It strikes a balance in estimating available forage by excluding some forage that is within reach of elk (from taller trees and shrubs with low-hanging crowns) while including some forage that is out of reach to them (from ≤ 2.54 D15 trees and shrubs with crowns extending above the reach of elk; VanderSchaaf 2013). We estimated summer forage at right of ways by summing forb and grass biomass at each right of way plot.

Forage comparisons

We used analysis of variance (ANOVA) to determine if forage differed by study area, cover type, and ownership (public or private; stats package in R). Forage was the dependent variable in separate analyses for summer and winter. We used ANOVA to test for differences in summer forge between road right of way types (e.g., county roads vs. state roads) and forage between all right of way types and followed significant ANOVAs with Tukey tests. We transformed forage biomass by the square root for all statistical tests to meet model assumptions. We set $\alpha = 0.05$ and assessed collinearity using the variance inflation factor (for all statistical tests hereafter).

Forage maps

We used random forest analysis to model summer and winter forage at field plots (Breiman 2001, Cutler et al. 2007). Random forest fits a large number of regression trees (a forest), with each regression tree constructed by recursive partitioning of data into smaller groups at binary splits based on a single predictor variable that maximizes homogeneity of the resulting groups (minimizes residual sum of squares). Whereas classic regression tree analysis uses all data to construct a tree, random forest analysis constructs each tree with a random subset of predictors and then combines the results from all trees to yield predictions that are robust to outliers or small changes in data and unbiased out-of-bag (OOB) error rates that make dividing data into training and test sets and cross-validation unnecessary (Prasad et al. 2006).

Random forest models result in predictions of the dependent variable that are means from the ensemble of multiple trees. It also calculates measures of accuracy and variable importance based on mean squared error (MSE) of OOB data (Liaw and Wiener 2002). A pseudo-R^2 ($1 - MSE_{OOB} / \sigma^2_y$) summarizes model accuracy. The importance of each predictor variable is computed by comparing prediction error (standardized MSE) on the OOB portion of data with prediction error after permuting predictor variable values (Liaw and Wiener 2002).

Random forest is frequently used in geospatial modeling (Rodriguez-Galiano et al. 2012, Karlson et al. 2015) as it models nonlinear relationships and interactions without error distribution assumptions (e.g., normality; Cutler et al. 2007), is robust to missing data (Rodriguez-Galiano et al. 2012), and does not overfit (Breiman 2001). Results from random forest models are often more accurate than those from other methods, including regression trees and linear models (Prasad et al. 2006, Chen et al. 2017).

To estimate potential forage biomass at our field sites, we implemented random forest analysis in R (randomForest package; Liaw and Wiener 2002). Analysis included biologically relevant predictor variables that we extracted from 15 m resolution GIS raster maps spanning the three study areas (Table 2). Before analysis, we screened and eliminated correlated independent variables (Millard and Richardson 2015; Spearman correlation coefficient $|r_s| > 0.5$; stats package in R), including slope, aspect, precipitation, and temperature. We kept only the variable that resulted in greater predictive accuracy (assessed using pseudo-R^2) when variables were correlated (Gustafson et al. 2003). Each random forest model predicted leaf or total biomass (square root transformed) by growing 1,000 regression trees, each using 33% of predictor variables (Liaw and Wiener 2002).

Table 2. Independent variables used to model elk forage in northeastern Minnesota.

Independent variable	Description
Cumulative topographic index	Wetness index based on topography
Elevation	Lidar based elevation (bare earth)
Enhanced vegetation index for spring	Landsat 8 based vegetation index for April 2018
Enhanced vegetation index for summer	Landsat 8 based vegetation index for August 2018
Height	Lidar based height of above ground for any object
Insolation	Lidar based solar radiation (WH/m^2)
Normalized difference moisture index for spring	Landsat 8 based moisture index for April 2018
Normalized difference moisture index for summer	Landsat 8 based moisture index for August 2018
Ownership	Private or public ownership
Phenology	Number of days into summer when plot was sampled
Study area	Cloquet, Fond du Lac, and Nemadji study areas
Years since disturbance	Landsat based number of years since harvest, fire, and other disturbance

Using random forest model results and corresponding GIS maps, we predicted potential summer and winter elk forage across the 3 study areas and the surrounding 20 km (raster package; Hijmans 2019). Forage estimates were not spatially autocorrelated (Moran's I test, $P > 0.25$; spdep package in R; Millard and Richardson 2015, Bivand and Wong 2018).

We included right of way forage estimates when estimating summer forage. To do so, we estimated forb and grass forage biomass at right of ways distributed throughout the study areas by extrapolating measurements made in the field. For each right of way type (e.g., county highway) we calculated mean forb and grass biomass and mean right of way width (distance between shrub lines and tree lines that bordered the right of way). To extrapolate to the study area, we buffered right of ways (linear features in GIS) with mean right of way widths and deleted corresponding mean road widths. We rasterized the resulting map and assigned mean biomass to each cell. The resulting map depicted forage bordering right of ways.

When extrapolating, we used grand mean biomass and right of way width for road classes we did not sample (2 classes: other and municipal). We used forage estimates from pipelines to estimate forage at powerline right of ways (Minnesota Geospatial Information Office 2016), as we did not sample powerline right of ways. Powerline right of ways were only from high voltage (69 to 500 kilovolt) lines that were managed similarly to pipelines (maintained to be open), had similar widths (measured using Google Earth), and similar biomass characteristics (qualitatively assessed using Google Earth).

Animal unit equivalence

We used forage estimates to estimate animal use equivalence (Kuzyk and Hudson 2007). After predicting potential forage using random forest analysis, we summed forage using a 16 km^2; circular moving window. Each cell in resulting maps depicted available forage in the surrounding 16 km^2. Using forage maps, we calculated animal use equivalence (AUE) for elk as:

$$AUE = \frac{F \times C}{S \times M \times D};$$

where F was potential seasonal (winter or summer) forage available in the surrounding 16 km^2, S was dry forage (expressed as % elk body mass) required to sustain an elk of mass M for 1 day during a season lasting D days, and C was a correction factor reflecting how much forage elk consume in their use areas. AUE was for cow elk with a mass of 250 kg (median cow elk mass in Michigan; Bender et al. 2006), consuming 2.1% of body mass per day during a winter (Christianson and Creel 2009) lasting 200 days, and 2.2% of body mass per day during a summer (Kuzyk and Hudson 2007) lasting 165 days. To account for the presence of shrubs and trees not consumed within their use areas, we assumed elk consume the same proportion of available forage as do moose (*Alces alces*; 0.03 of available forage; Peek et al. 1976, Edenius et al. 2002). Each cell in resulting maps estimate the number of elk supported by the surrounding 16 km^2.

For AUE (and other maps we developed) we report the mean and standard deviation (*SD*) of raster map cell values in each study area, as well as the relative differences between study area means. We did not conduct statistical analyses (e.g., ANOVA) as map values are from a large number of cells (≥ 3,402,931), making statistical test *P*-values uninformative (Lin et al. 2013).

Biological carrying capacity

We used winter AUE to estimate carrying capacity for elk in each study area. It made sense to use AUE from winter as it is the limiting season for many animals at high latitudes and is when elk aerial surveys occur. We calculated carrying capacity (K_w) using:

$$Kw = \frac{AUE_w \times A}{16};$$

Where AUE_w was study area specific mean winter animal use equivalence and A was the area of each study area (km^2). For a range of potential K_w we made additional calculations after substituting AUE_w with $AUE_w \pm 1.96 \times SD$. These estimates likely underestimate biological carrying capacity as we assumed when calculating AUE that elk consume a small proportion of available forage (as observed for moose; Peek et al. 1976, Edenius et al. 2002). They also do not account for cultural carrying capacity, which may differ from biological carrying capacity (Minnis and Peyton 1995).

Habitat suitability

We used a habitat suitability index developed in Wisconsin (Gilbert et al. 2010) to map habitat suitability. The habitat suitability index used landcover types to estimate spring forage, winter forage, winter cover, and compatibility with people (hereafter, social suitability). It combined these indices to create a map of overall suitability.

Following the methods of Gilbert et al. (2010) we developed habitat suitability index maps by scoring landcover classes (Rampi et al. 2016) from 0 to 1 (0 = unsuitable and 1 = highly suitable) for spring food, winter food, and winter cover (Table 3). We used the same scores as Gilbert et al. (2010). This resulted in 1 map each for spring food, winter food, and winter cover.

Table 3. Suitability index scores for landcover classes representing spring food, winter food, and winter cover for elk. Index scores based on Gilbert et al. (2010).

Landcover class	Spring food [a]	Winter food [a]	Winter cover [a]
Conifer	0.50	0.70	0.50
Deciduous	0.70	0.90	0.30
Emergent wetland	0.70	0.00	0.00
Forested and shrub wetland [b]	0.43	0.27	0.53
Grassland	0.70	0.30	0.00
Mixed forest	0.30	0.50	0.30

[a] Extraction, hay field, row crop, impervious, and water were 0.

[b] Mean of values for lowland shrub, forested wetland, and shrub from Gilbert et al. (2010).

Additional maps assigned social suitability according to land ownership (from 0 to 1; Table 4) and as a negative function of road density (Gilbert et al. 2010, MNDOT 2017). After developing each map, we combined them to create a map of overall suitability. The overall suitability map reflected the geometric mean of all map scores, but with a suitability score of 0 assigned to row crops, hay/pasture fields, and urban areas. After scoring each raster cell in each map, we applied a circular moving window to calculate mean suitability within the surrounding 16 km^2.

Table 4. Suitability index scores for landowner classes. Based on Gilbert et al. (2010).

Owner	Suitability
County	0.80
Federal	0.80
Other public [a]	1.00
Private	0.00
Private (non-industrial)	0.50
Private conservancy [b]	1.00
Private industrial	0.50
State	1.00
Tribal	1.00

[a] Used value from Gilbert et al. (2010) for parks, trails, and riverways.
[b] Used value from Gilbert et al. (2010) for Scenic Natural Areas.

Our methods were similar to those from the Wisconsin habitat suitability index, but while the Wisconsin habitat suitability index used a 100 km^2 moving window to estimate statewide suitability, we used 16 km^2 to be consistent with other habitat maps we developed and other elk habitat suitability studies (Van Deelen et al. 1997, Didier and Porter 1999, Karns et al. 2015).

Resource selection

We used a resource selection function developed from elk radio telemetry locations collected during summer in Wisconsin (large extent function; Anderson et al. 2005*b*) to map the relative probability elk would select areas in and near our study areas. The resource selection function predicted the relative likelihood of home range selection by elk (second order selection; Johnson 1980). It is possible that these resource selection functions will predict relative elk habitat selection well on our study areas as they originate from the same ecoregion (Level III Region 50; Omernik and Griffith 2014).

Following the methods of Anderson et al. (2005*b*), we used field plot data to calculate mean biomass of forbs and grasses, woody browse, and sedges in each landcover type (e.g., conifer; Rampi et al. 2016). We then assigned these biomass values to each landcover type raster cell and used a circular moving window to calculate mean biomass in the surrounding 0.3 km^2 (300 m radius). Two other maps contained the distance from each map raster cell to the nearest road (MNDOT 2017) and wolf territories (data from packs monitored during 2015 to 2018; Erb et al. 2017, M. Swingen, 1854 Treaty Authority; *Unpublished data*). We used biomass, road distance, and wolf distance to calculate selection scores using parameters from Anderson et al. (2005*b*). We then classified scores using quartiles. Resulting scores ranged from 1 to 4, reflecting increasing probability of selection.

The distance from nearest wolf territory was influential for estimating elk resource selection (Anderson et al. 2005*b*), but we did not know the location of some wolf pack territories, including all recent territories on the Nemadji study area. To account for missing territories, we developed a second resource selection function map after setting the influence of wolf territories to 0. The resulting map reflected selection scores (1 to 4) without the influence of wolves.

Aspen, grassland, and public land

We used a moving window (16 km^2) to calculate aspen, grassland, and public land densities in a GIS. Aspen was from a forest inventory layer for public land (C. Beal, United States Forest Service; *Unpublished data*). Grassland was from the landcover data used for other analyses (Rampi et al. 2016; excluded hay/pasture landcover class). Land ownership data combined multiple landowner databases (D. Wilson, University of Minnesota; *Unpublished data*).

Social acceptance

We used data from 2 surveys to map social acceptance for elk across the 3 study areas: 1 of landowners and 1 of local residents. These data, further details, and additional analyses are presented in a companion report by Walberg et al. (2019).

<u>Landowners</u>

Landowners owned ≥ 4 ha of land located ≤ 8 km of the 3 study areas. It made sense to include an analysis that focused on landowners as (all else being equal) elk are more likely to use a large property than a small property and elk are more likely to be restored to areas with large tracts of land than to areas with smaller tracts. Additionally, producers are often landowners and are more likely to experience property damage and concerns about livestock-elk disease transmission.

From the population of landowners in northeastern Minnesota, we used a random-stratified approach to select 4,500 landowners, where the stratum was hectares owned (2 levels: 4 to 16 ha and >16 ha). We mailed selected landowners a questionnaire (up to 2 additional questionnaires to nonrespondents). The questionnaire asked landowners about their attitudes toward elk restoration in the area. We measured landowner attitudes toward elk restoration using 2,550 returned questionnaires scored using a scale ranging from very unfavorable (1) to very favorable (7). We included 35 additional surveys from respondents of the local resident survey (see below) who owned ≥ 4 ha of land located ≤ 8 km from the study areas, bringing our sample to 2,585 surveys.

We determined if landowner attitudes were more likely to be favorable on any of the 3 study areas by comparing attitudes of landowners who owned property inside the boundaries of the 3 study areas. While this analysis did not include landowners from outside the study area boundaries, mapping of attitudes (see below) and the companion report by Walberg et al. (2019) did. Focusing on landowners with property within the study area boundaries made sense as restored elk will affect these landowners before others. We assigned landowner attitude scores to points at the center of properties (polygon representing land ownership) they owned and randomly selected a property when a single landowner owned > 1 property. Residuals from linear models of acceptance scores were not normally distributed so we categorized scores ≤ 3 as unfavorable to elk restoration (0) and scores ≥ 5 as favorable to elk restoration (1), and tested for different attitude scores in each study area using logistic regression and Wald tests for pairwise comparisons (stats package in R; acceptance scores = 4 were not used for analysis).

To better understand how attitudes were distributed in space, we mapped mean attitude scores. This mapping process included questionnaire responses from landowners with land inside and ≤ 8 km away from the study area boundaries to provide a better understanding of attitudes in the broad area where elk populations are likely to expand if restored. It used a circular moving window with an area equaling 4 townships (372 km^2, 10.9 km radius). This window was larger than others we used to ensure > 20 respondents in most calculations. It created a map surface that filled in intervening areas with opinions from multiple landowners. Smaller moving windows we developed resulted in what amounted to a less informative point map, with isolated 15×15 m raster cells representing a single landowner's attitude. For example, the 16 km^2 window's radius is 2.3 km but landowner properties were usually > 2.3 km apart, resulting in isolated raster cells (1 for each land parcel) separated by large areas without acceptance estimates.

<u>Local residents</u>

In addition to considering landowner support, we examined data from a local resident survey (details in Walberg et al. 2019). We selected 4,000 local residents using a random-stratified approach, with a geographic stratum containing 4 levels: (1) St. Louis County south of the St. Louis River; (2) Carlton County; (3) Pine County north of Minnesota Highway 48; and (4) Duluth and surrounding suburbs. These areas matched census blocks corresponding to county boundaries and major landmarks (e.g., roads).

As we did for landowners, we determined if attitudes were more likely to be favorable on any of the 3 study areas by comparing attitudes of members of the general public who had mailing addresses within the boundaries of the 3 study areas. As was the case for the landowner statistical analysis, we did not include residents from outside the study area boundaries, but mapping of attitudes (see below) and the companion report by Walberg et al. (2019) did. As above, we categorized scores ≤ 3 as unfavorable to elk restoration (0) and scores ≥ 5 as favorable (1), and tested for different attitude scores by study area using logistic regression and Wald tests for pairwise comparisons (stats package in R; acceptance scores = 4 were not used for analysis).

To better understand how local resident attitudes were distributed in space, we mapped mean attitude scores using methods described above for landowners (circular moving window the size of 4 townships). This mapping process included local resident respondents with addresses inside and outside of the 3 study areas, bringing our sample size to 1,521 local resident respondents.

Risk of human-elk conflict

To assess the potential for elk-human conflict we developed a risk map by calculating the proportion of area (16 km^2 moving window) that was row crop and hay/pasture, feedlot, and road surface. Each cell in the resulting map is the proportion of the surrounding 16 km^2 that has potential for conflict should an elk center its activities there. For example, if an elk use area is centered on a cell with conflict risk = 0.15, then 15% of its likely use area poses conflict risk.

Row crop and hay/pasture were from landcover data used in other analyses (Rampi et al. 2016). Road data came from buffering road centerlines with road widths measured for different road classes in the field (MNDOT 2017; see the <u>*Right of way plots*</u> section above). We estimated feedlot area by buffering feedlot locations 0.12 km^2 (median feedlot size). Locations were for cow, horse, and pig feedlots (MPCA 2007) and we added a dataset containing the 4 captive cervid operations within 20 km of the 3 study areas (locations from 2017; Minnesota Board of Animal Health, *unpublished data*). Nearly all (98%) of the 923 feedlots within 20 km of our study areas included open lots and pasture, and 84% had holding areas (MPCA 2007). These are locations where elk-livestock contact occurs and where elk raid dispersed and stored forage. To estimate feedlot size, we measured 20 randomly selected northeastern Minnesota (within 20 km of study areas) feedlots in Google Earth. Feedlot measurements included grassless areas and adjacent pastures with cow paths.

Winter suitability

We intersected maps to identify areas best suited for elk restoration using maps we developed (described above). We calculated mean: (1) winter AUE; (2) winter forage habitat suitability index; and (3) winter cover habitat suitability index. We then deleted areas with values less than the mean from each map and intersected resulting maps with areas where social acceptance for elk restoration was high (acceptance score ≥ 5). Resulting map depicted areas where winter conditions were better than mean conditions and restoration was most favorable to landowners.

Ranking study areas

We ranked the suitability of elk restoration for each study area using mean values from maps we developed. We calculated mean values from each study area (e.g., mean winter AUE for the Cloquet Valley study area) and ranked study areas from worst (1) to best (3). Means we ranked were: (1) winter AUE; (2) overall habitat suitability index from Wisconsin; (3) summer resource selection (excluding wolf territories because we lacked recent Nemadji study area data); (4) proportion grassland; (5) proportion aspen; (6) social acceptance (from questionnaire); (7) conflict risk; and (8) proportion public land. The best study areas had greatest means (e.g., mean winter AUE) and highest proportions (e.g., proportion aspen), while the worst study areas had the lowest means and proportions. We compared ranks using a one-way permutation test for ordinal data (Hothorn et al. 2008).

To measure the influence of considering 1 factor to be most important, we weighted ranks of each factor by including records for each factor more than once. For example, we included the set of conflict risk ranks 2 times in our dataset (weight = 2) but included only 1 set of ranks for other factors (e.g., 1 set for winter AUE). We then repeated this after including risk ranks 3 times in our dataset (weight = 3), and so forth until we included 10 sets of risk ranks (but only 1 set from other factors). For each of the 10 weighting iterations, we compared study area ranks using a one-way permutation test for ordinal data (as above). A pairwise permutation post-hoc test (Holm's method) followed significant one-way tests. In addition to weighting the factors above, we weighted conflict and social acceptance together by weighting both factors at the same time for each of the 10 weighting iterations.

Thresholds from other elk studies

We compared forage availability, AUE, and habitat suitability to threshold values from other studies. We calculated the area within each study area that had ≥ the amount summer forage preferred by elk (0.120 kg/m^2 of forage; Wilmshurst et al. 1995) and the area that had ≥ the mean amount of winter forage where elk occur in Wisconsin (Anderson et al. 2005*a*). Winter forage estimates from Wisconsin (0.025 kg/m^2) included grasses (Anderson et al. 2005*a*), while our winter forage estimates did not (as elk rarely crater for grass in winter in eastern North America; Jenkins et al. 2007). When we accounted for this difference by eliminating grass from Wisconsin estimates (using the proportion of grass on our field plots), adjusted winter forage in Wisconsin was 0.017 kg/m^2.

We calculated the area within each study area with winter AUE ≥ densities reported in Michigan and Wisconsin after multiplying density estimates (elk/km^2) from Michigan and Wisconsin by 16 to make them comparable to our AUE estimates. Elk density is about 7 elk/16 km^2 in Michigan (converted from 0.46 elk/km^2; MIDNR 2019), 5 elk/16 km^2 in Wisconsin's Black River Herd (0.33 elk/km^2 in core area; Stowell et al. 2012, WDNR 2019), and 3 elk/16 km^2 in the Clam Lake Herd (0.20 elk/km^2 in core zone; Stowell et al. 2012, WDNR 2019*b*). Area calculations we made used estimates from Michigan and the Black River Herd as threshold values (elk densities ≥ 7 elk/16 km^2 and 5 elk/16 km^2). Lastly, we calculated the area within each study area with habitat suitability ≥ 0.5. This calculation used the overall suitability score from the Wisconsin model (Gilbert et al. 2010). We selected 0.5 as our threshold as Wisconsin herds occur in areas with suitability index scores ≥ 0.5 (Gilbert et al. 2010, Stowell et al. 2012).

Results

Forage estimation

We sampled 186 field plots, including 63 plots in the Cloquet Valley study area, 69 in the Fond du Lac study area, and 54 in the Nemadji study area. Mean summer forage at field plots was 0.130 kg/m^2 (SD = 0.106, N = 186 plots) and winter forage was 0.017 kg/m^2 (SD = 0.016, N = 186 plots). Winter forage differed by ownership ($F_{1,177} = 17.08$, $P < 0.01$) and landcover type ($F_{5,177} = 2.65$, $P = 0.02$), but not by study area ($F_{2,177} = 1.50$, $P = 0.23$). Public land had more winter forage than private land (Tukey $P < 0.05$; Figure 3A) and forested shrub wetlands had more winter forage than grasslands (Tukey $P < 0.05$; Figure 3B). Summer forage differed by landcover type ($F_{5,177} = 3.20$, $P < 0.01$), but not by study area ($F_{2,177} = 1.16$, $P = 0.32$) and ownership ($F_{1,177} = 0.24$, $P = 0.62$). Grasslands had more summer forage than coniferous forests (Tukey $P < 0.05$) and mixed forests (Tukey $P < 0.05$; Figure 3C). Two way and 3-way interactions were nonsignificant for winter and summer forage models.

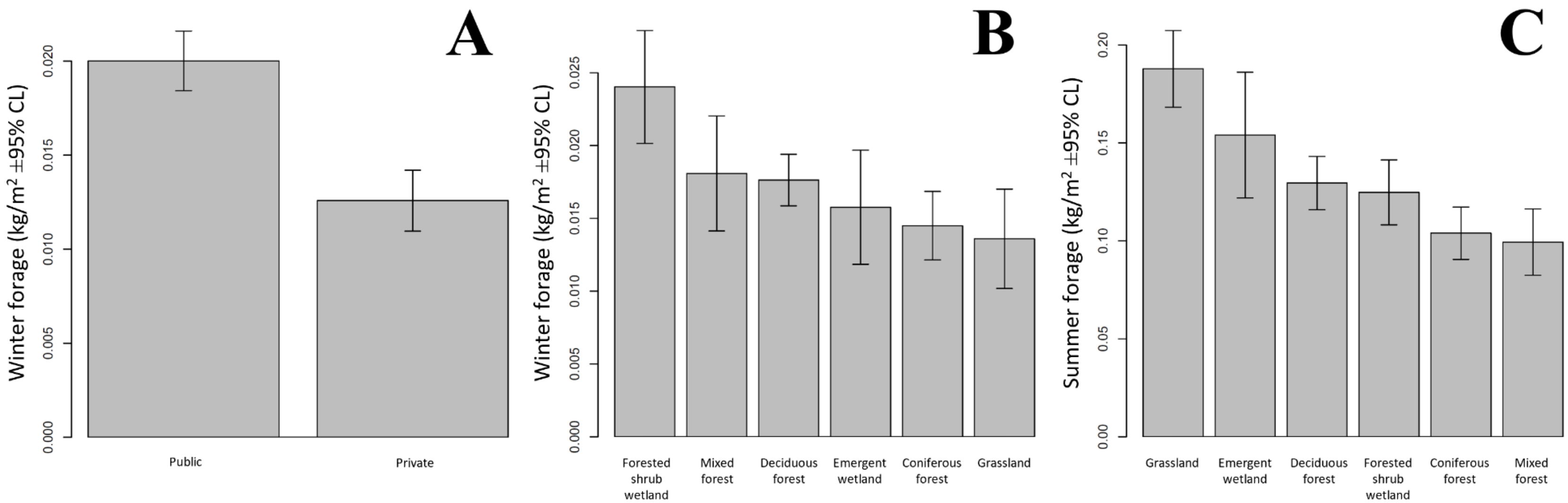

Figure 3. Winter forage on public and private land (A) and by landcover type (B), and summer forage by landcover type (C) in northeastern Minnesota. CL = confidence limits. Each panel is sorted in descending order by mean forage.

The random forest model for potential winter forage was 19% accurate (pseudo-R^2 = 0.19) and the 3 most important variables were phenology, August enhanced vegetation index, and April enhanced vegetation index (Figure 4A). Fifty percent of predictions were within 0.007 kg/m^2 of field observations and 75% were within 0.015 kg/m^2 (Figure 4B).

Summer forage estimates were an order of magnitude greater than winter estimates (Figures 4C). The random forest model for summer forage was 30% accurate and the 3 most important variables for predicting forage were April enhanced vegetation index, phenology, and August normalized difference moisture index. Fifty percent of predictions were within 0.050 kg/m^2 of field observation and 75% were within 0.080 kg/m^2 (Figure 4D).

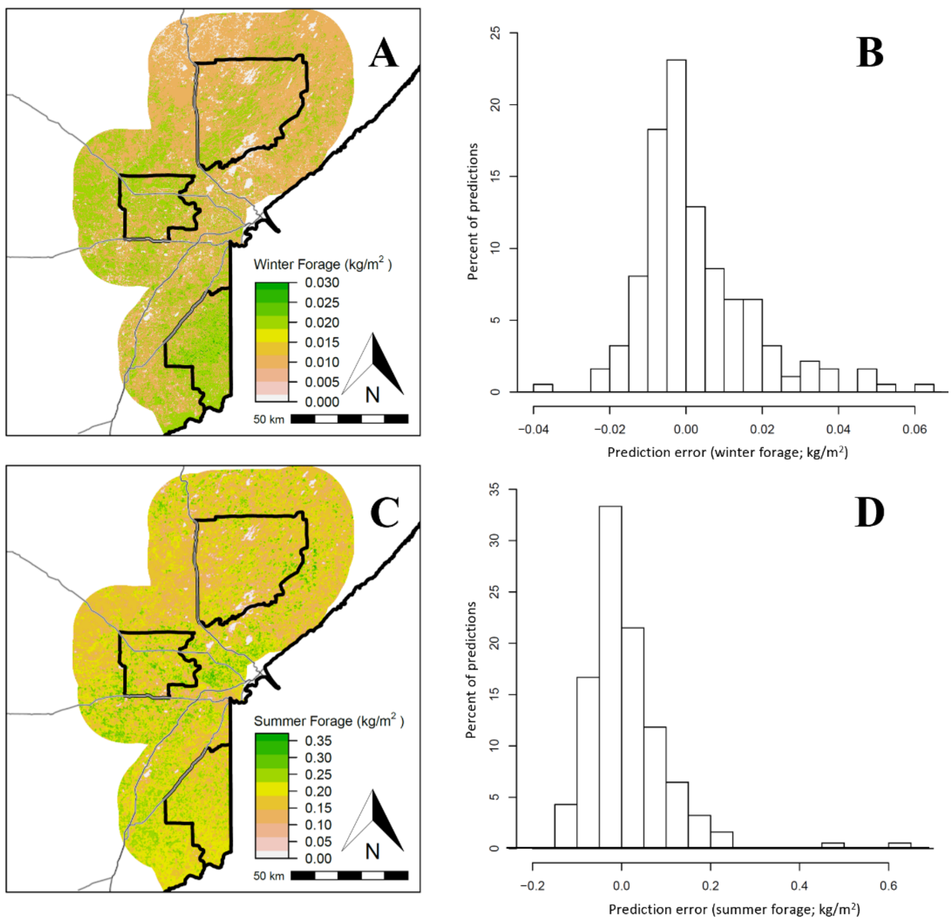

Figure 4. Estimated forage and prediction error (observed forage minus predicted forage) for winter (A and B) and summer (C and D) in northeastern Minnesota during summer 2017 and 2018. Summer forage includes forbs and grasses at right of ways.

We sampled summer forage at 29 right of way plots. Mean forage was 0.136 kg/m^2 (SD = 0.117, N = 21) along road right of ways, 0.122 kg/m^2 (SD = 0.077, N = 6) along railroads, and 0.409 kg/m^2 (SD = 0.127, N = 2) along pipelines (Table 5).

Table 5. Number of right of way plots sampled in northeastern Minnesota in summer 2018.

	Biomass (kg/m^2)		Width (m)		
Right of way type	Mean	*SD*	Mean	*SD*	*N*
All roads	0.136	0.117	44.07	39.69	21
County road	0.162	0.135	32.69	12.67	14
Federal road	0.089	0.051	53.00	22.63	2
State road	0.071	0.039	68.53	90.08	3
Township road	0.100	0.041	78.05	81.25	2
Pipelines	0.409	0.127	32.25	18.74	2
Railroads	0.122	0.077	20.06 [a]	5.96	6

[a] Based on $N = 5$

Forage along road right of ways did not differ by road type ($F_{3,17} = 0.63$, $P = 0.61$) but differed by right of way type (all road types combined, pipelines, and railroads; $F_{2,26} = 4.53$, $P = 0.02$; Figure 5) as pipeline right of ways had more forage than road (Tukey test $P = 0.02$) and railroad (Tukey test $P = 0.03$) right of ways.

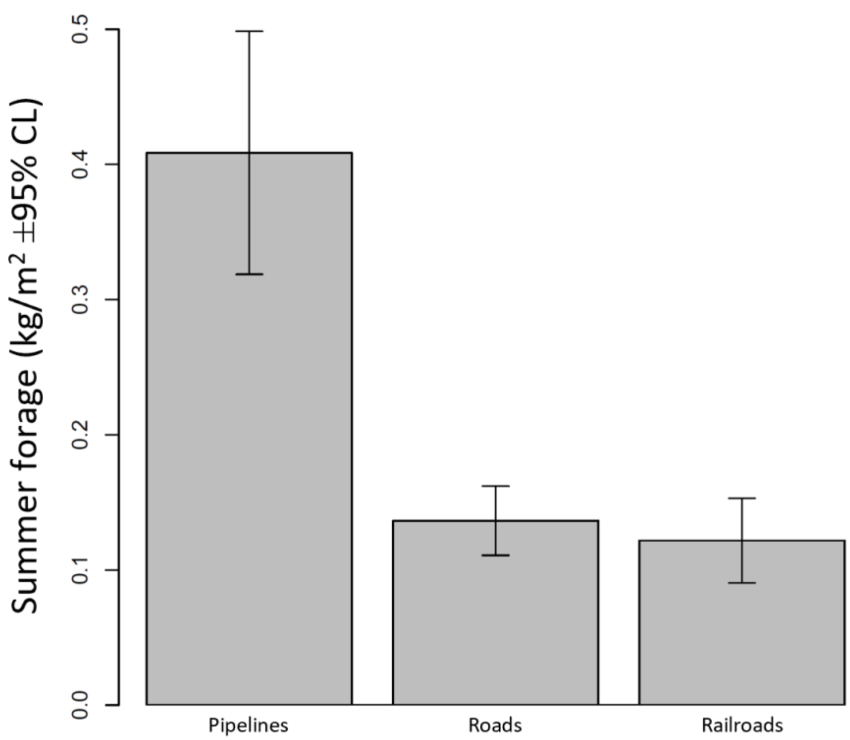

Figure 5. Summer forage along 3 right of way types in northeastern Minnesota.

Animal unit equivalence

Winter AUE ranged from 1 to 9 elk/16 km^2 across all study areas (Figure 6A). Mean winter AUE was 5 elk/16 km^2 (SD = 1 elk/16 km^2, N = 7,841,931 raster cells) on the Cloquet Valley study area, and was 1.2 times greater on the Fond du Lac study area (6 elk/16 km^2, SD = 1 elk/16 km^2, N = 3,402,931 raster cells) and 1.6-times greater on the Nemadji study area (8 elk/16 km^2, SD = 1 elk/16 km^2, N = 4,279,849 raster cells).

Summer AUE was greater than winter AUE, ranging from 14 to 83 elk/16 km^2 across the study areas (Figure 6B). Mean summer AUE was 58 elk/16 km^2 (SD = 8 elk/16 km^2, N = 7,841,931 raster cells) on the Cloquet Valley study area, and 1.2 time greater on the Fond du Lac study area (67 elk/16 km^2, SD = 6 elk/16 km^2, N = 3,402,931 raster cells) and the Nemadji study area (69 elk/16 km^2, SD = 4 elk/16 km^2, N = 4,279,849 raster cells).

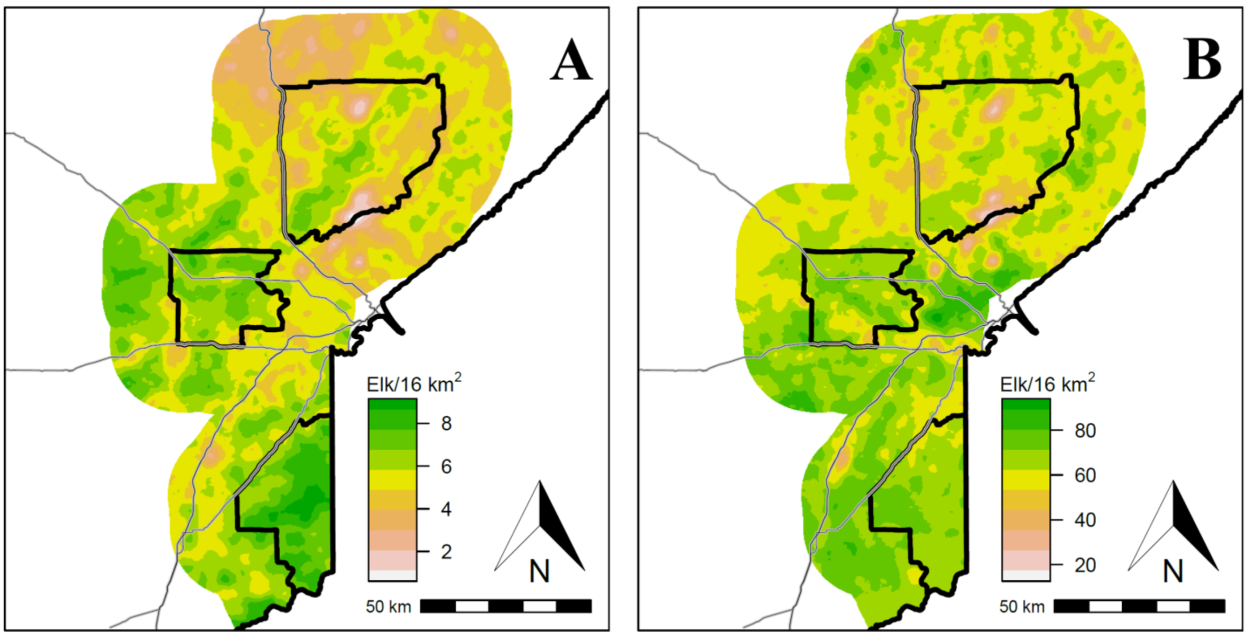

Figure 6. Winter (A) and summer (B) animal unit equivalence estimates for elk in northeastern Minnesota.

Biological carrying capacity

Winter carrying capacity (K_w) estimates based on mean AUE ranged from 287 on the Fond du Lac study area to 551 elk on the Cloquet Valley study area (Table 6). Estimates based on the SD of AUE ranged from 193 elk on the Fond du Lac to 768 elk on the Cloquet Valley study area.

Table 6. Winter biological carrying capacity estimates for 3 study areas in northeastern Minnesota. Estimates are for the 3 study areas but not the surrounding areas.

Study area	Area (km^2)	K_w (range) [a]
Cloquet Valley	1,764	551 (335 to 768)
Fond du Lac	766	287 (193 to 381)
Nemadji	963	481 (364 to 599)

[a] Carrying capacity estimate for winter; range is from 1.96 × SD of AUE.

Habitat suitability

Suitability maps of winter forage (Figure 7A), spring forage (Figure 7B), and winter cover (Figure 7C) resulted in a map of overall suitability (Figure 6D) after accounting for road density and ownership. Overall suitability within the 3 study areas ranged from 0 to 0.68. Mean overall suitability was similar across study areas (Figure 7D). Mean overall suitability was 0.45 (SD = 0.17, N = 7,831,119 raster cells) on the Cloquet Valley study area, 0.45 (SD = 0.16, N = 3,402,931 raster cells) on the Fond du Lac study area, and 0.43 (SD = 0.17, N = 4,279,849 raster cells) on the Nemadji study area.

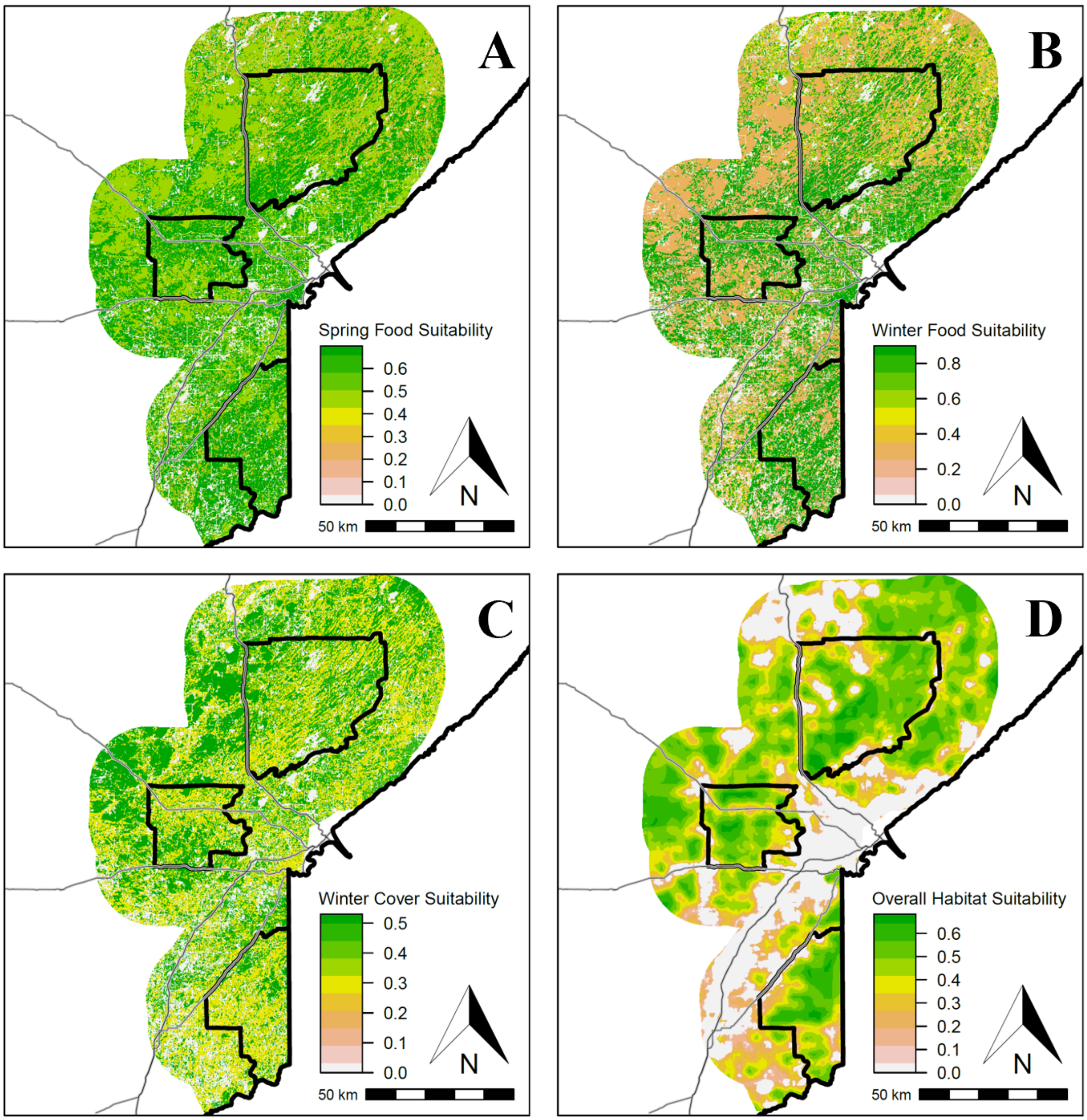

Figure 7. Spring food (A), winter food (B), winter cover (C), and overall suitability indices (D) for elk in northeastern Minnesota.

Resource selection

Summer resource selection function scores reflected increasing relative probability of selection (1 = low probability and 4 = high probability). When we included known wolf territories in relative selection calculations, the Nemadji study area had mean selection scores 2.6- and 3-times higher than the Cloquet Valley and Fond du Lac study areas (Figure 8A). The Cloquet Valley study area had a mean relative summer selection score of 1.55 (SD = 0.68, N = 7,841,931 raster cells), while the Fond du Lac study area had a mean of 1.34 (SD = 0.57, N = 3,402,931 cells), and the Nemadji study area had a mean of 3.99 (SD = 0.08, N = 4,279,849 cells). Selection was high on the Nemadji study area because wolf pack territory location influences selection scores (Anderson et al. 2005*b*), but we were missing wolf territory data from packs there.

When we excluded the influence of wolf territories in selection calculations, mean selection score differences were smaller (Figure 8B). The Nemadji study area had mean selection scores 1.5- and 1.3-times higher than the Cloquet Valley and Fond du Lac study areas. The Cloquet Valley study area had a mean relative summer selection score of 2.19 (SD = 1.02, N = 7,841,931 raster cells), while the Fond du Lac study area had a mean of 2.49 (SD = 0.97, N = 3,402,931 cells), and the Nemadji study area had a mean of 3.23 (SD = 0.88, N = 4,279,849 cells).

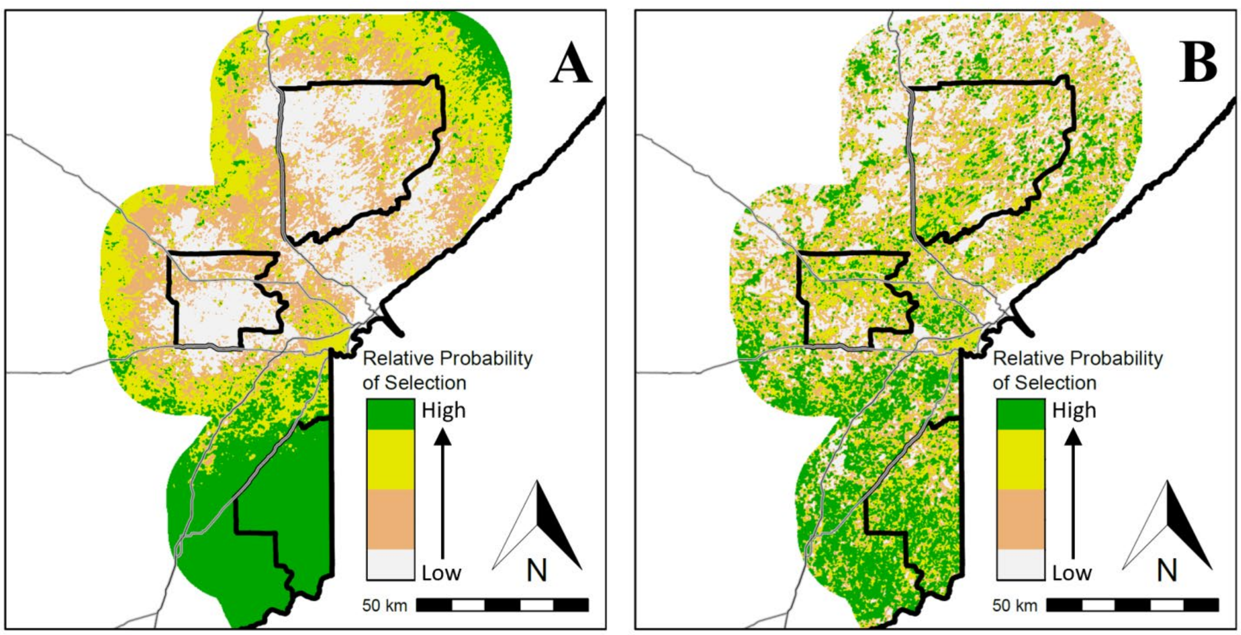

Figure 8. Relative probability of resource selection by elk in northeastern Minnesota with (A) and without (B) influence of monitored wolf packs. Wolf data were unavailable for the Nemadji study area.

Aspen, grassland, and public land

Aspen was most common on the Cloquet Valley study area. It was 4.3 times greater on the Cloquet Valley study area (mean proportion = 0.17 aspen, SD = 0.09 aspen, N = 7,841,931 raster cells) than on the Fond du Lac study area (0.04 aspen, SD = 0.03 aspen, N = 3,402,931 cells), and 5.7 times greater than on the Nemadji study area (0.03 aspen, SD = 0.04 aspen, N = 4,279,849 cells; Figure 9A).

Grassland distribution was similar across the 3 study areas. The Cloquet Valley study area was a mean of 0.054 grassland (*SD* = 0.04 grassland, *N* = 7,841,931 raster cells), while the Fond du Lac study area was 0.040 grassland (*SD* = 0.03 grassland, *N* = 3,402,931 cells), and the Nemadji study area was 0.048 grassland (*SD* = 0.03 grassland, *N* = 4,279,849 cells; Figure 9B).

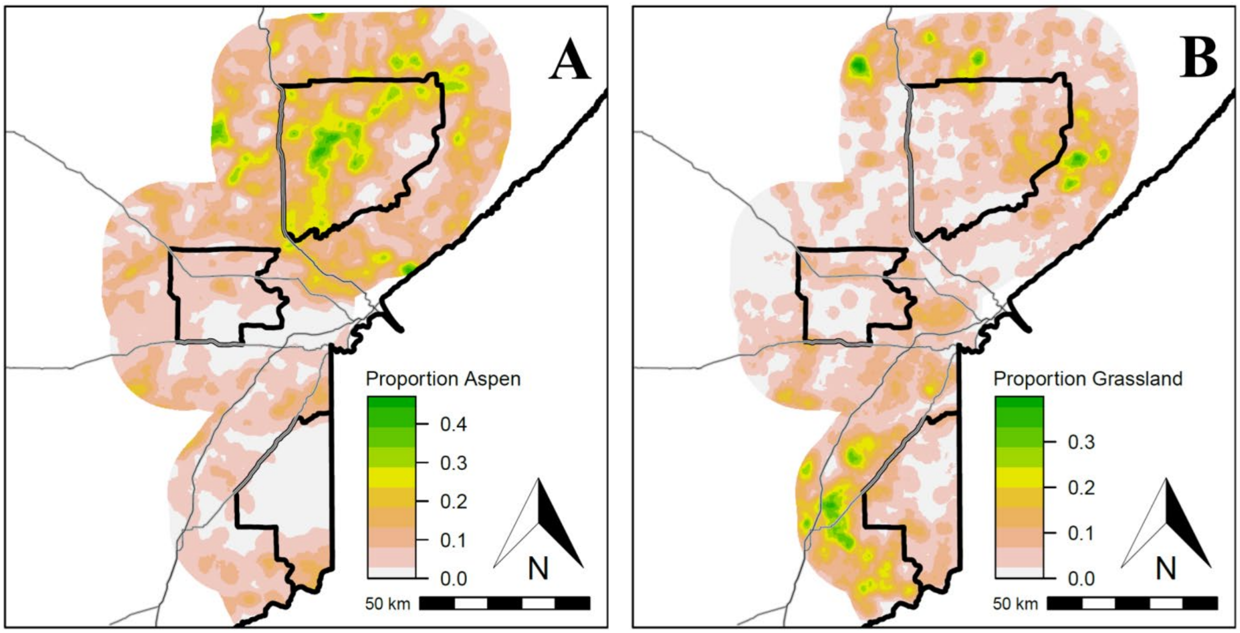

Figure 9. Proportion of area that was aspen (A) and grassland (not hay/pasture; B) in northeastern Minnesota.

Public land was in greatest abundance on the Cloquet Valley study area, and similar on the Fond du Lac and Nemadji study areas. The Cloquet Valley study area was a mean of 0.75 public land (*SD* = 0.26 public land, *N* = 7,841,931 raster cells), while the Fond du Lac study area was 0.61 public (*SD* = 0.27 public, *N* = 3,402,931 cells), and the Nemadji study area was 0.60 (*SD* = 0.34 public, *N* = 4,279,849 cells; Figure 10).

Social acceptance

<u>Landowners</u>

Most landowners supported elk restoration. Overall, 776 (82%) of 950 questionnaire respondents who owned ≥ 4 ha of land inside the 3 study area boundaries expressed favorable attitudes toward elk restoration (attitude scores ≥ 5; Table 7). Logistic regression analysis of unfavorable (scores ≤ 3) and favorable attitudes (scores ≥ 5) showed landowner support for elk restoration did not differ between study areas (Wald $|z| \leq 1.35$ and $P \geq 0.17$ for all pairwise comparisons).

Mapping acceptance scores from landowners who owned ≥ 4 ha of land inside and outside (≤ 8 km away) from the study areas (*N* = 2,585 questionnaire responses) using a moving window showed high mean acceptance across the 3 study areas (Figure 11A). Mean acceptance was 5.7 (*SD* = 0.20, *N* = 7,841,931 raster cells) on the Cloquet Valley study area, 5.5 (*SD* = 0.15, *N* = 3,402,931 cells) on the Fond du Lac study area, and 5.8 (*SD* = 0.14 *N* = 4,279,849 cells) on the Nemadji study area.

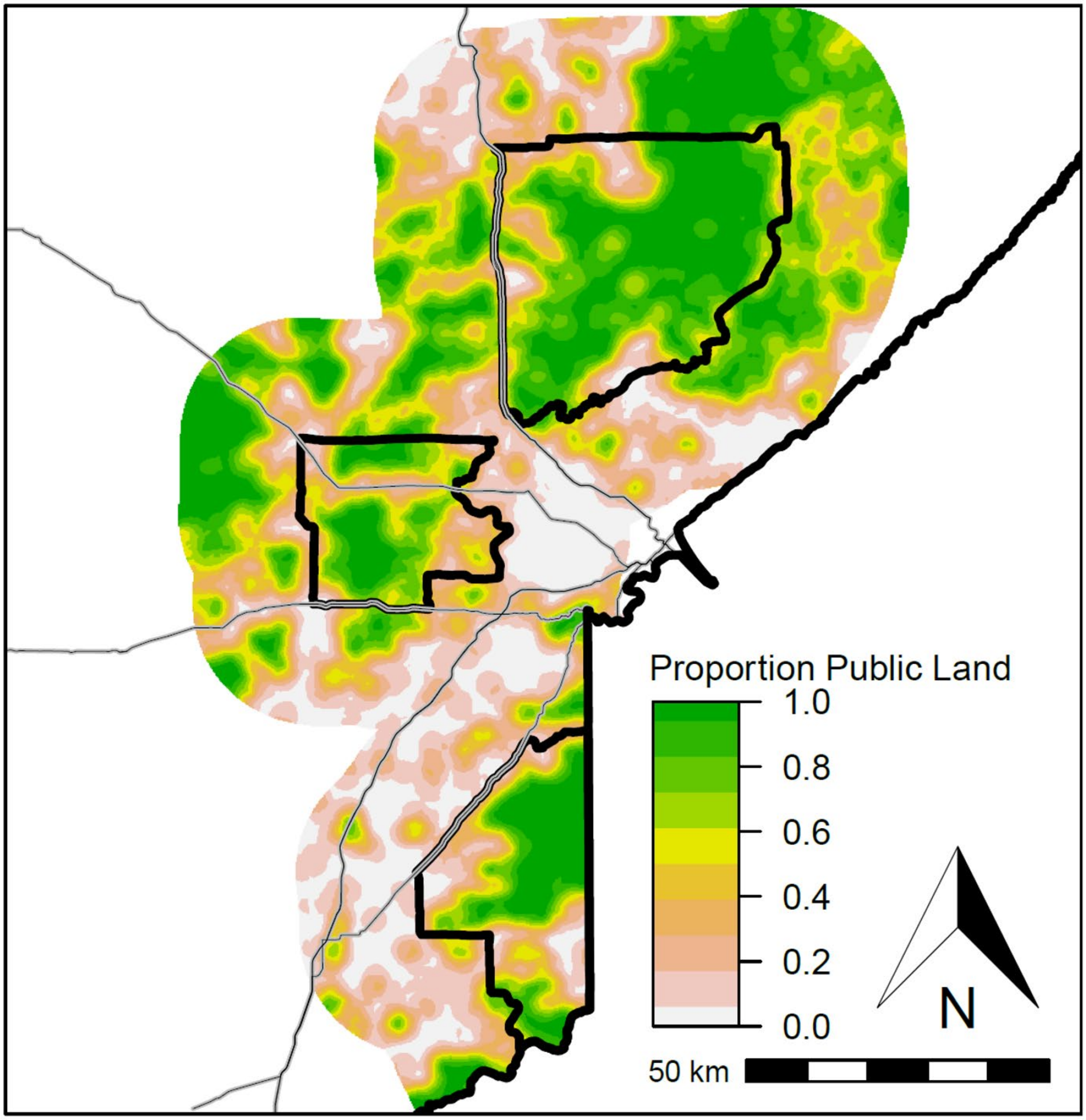

Figure 10. Proportion of land under public ownership in northeastern Minnesota.

<u>Local residents</u>
Most local resident survey respondents supported elk restoration. Overall, 105 (86%) of 122 local resident questionnaire respondents with addresses inside the 3 study area boundaries expressed favorable attitudes toward elk restoration (attitude scores ≥ 5; Table 6). Logistic regression analysis of unfavorable and favorable attitudes local resident attitudes did not differ between study areas (Wald $|z| \leq 2.23$ and $P \geq 0.49$ for all pairwise comparisons).

Table 7. Acceptance scores for landowners (owned ≥ 4 ha of land) and local residents (owned < 4 ha of land) in 3 northeastern Minnesota study areas. Scores presented here are for questionnaire respondents from inside the study area boundaries. See the results from our moving window mapping and the companion report by Walberg et al. (2019) for additional analyses that included landowners and local residents from inside and outside the study area boundaries.

	Landowners: count (proportion) of acceptance scores [a]							
Study area	1	2	3	4	5	6	7	Sum
Cloquet Valley	24 (0.07)	9 (0.03)	8 (0.02)	23 (0.07)	37 (0.11)	90 (0.27)	142 (0.43)	333
Fond du Lac	16 (0.08)	4 (0.02)	4 (0.02)	22 (0.11)	17 (0.08)	69 (0.33)	77 (0.37)	209
Nemadji	20 (0.05)	11 (0.03)	7 (0.02)	26 (0.06)	45 (0.11)	103 (0.25)	196 (0.48)	408
Sum (landowners)	60 (0.06)	24 (0.03)	19 (0.02)	71 (0.07)	99 (0.10)	262 (0.28)	415 (0.44)	950

	Local residents: count (proportion) of acceptance scores							
Study area	1	2	3	4	5	6	7	Sum
Cloquet Valley	6 (0.08)	1 (0.01)	0 (0.00)	6 (0.08)	13 (0.17)	18 (0.23)	34 (0.44)	78
Fond du Lac	0 (0.00)	1 (0.05)	0 (0.00)	1 (0.05)	4 (0.18)	8 (0.36)	8 (0.36)	22
Nemadji	1 (0.05)	0 (0.00)	0 (0.00)	1 (0.05)	2 (0.09)	8 (0.36)	10 (0.45)	22
Sum (local residents)	7 (0.06)	2 (0.02)	0 (0.00)	8 (0.07)	19 (0.10)	34 (0.28)	52 (0.43)	122

[a] 1 = low acceptance, 4 = neutral, 7 = high acceptance.

Moving window analysis of questionnaire responses from local residents with addresses inside and outside the 3 study areas (*N* = 1,521 responses) showed acceptance scores were high across the study areas (Figure 11B). Mean acceptance was 5.6 (*SD* = 0.40, *N* = 7,841,931 raster cells) on the Cloquet Valley study area, 5.4 (*SD* = 0.30, *N* = 3,402,931 cells) on the Fond du Lac study area, and 5.7 (*SD* = 0.38, *N* = 4,279,849 cells) on the Nemadji study area.

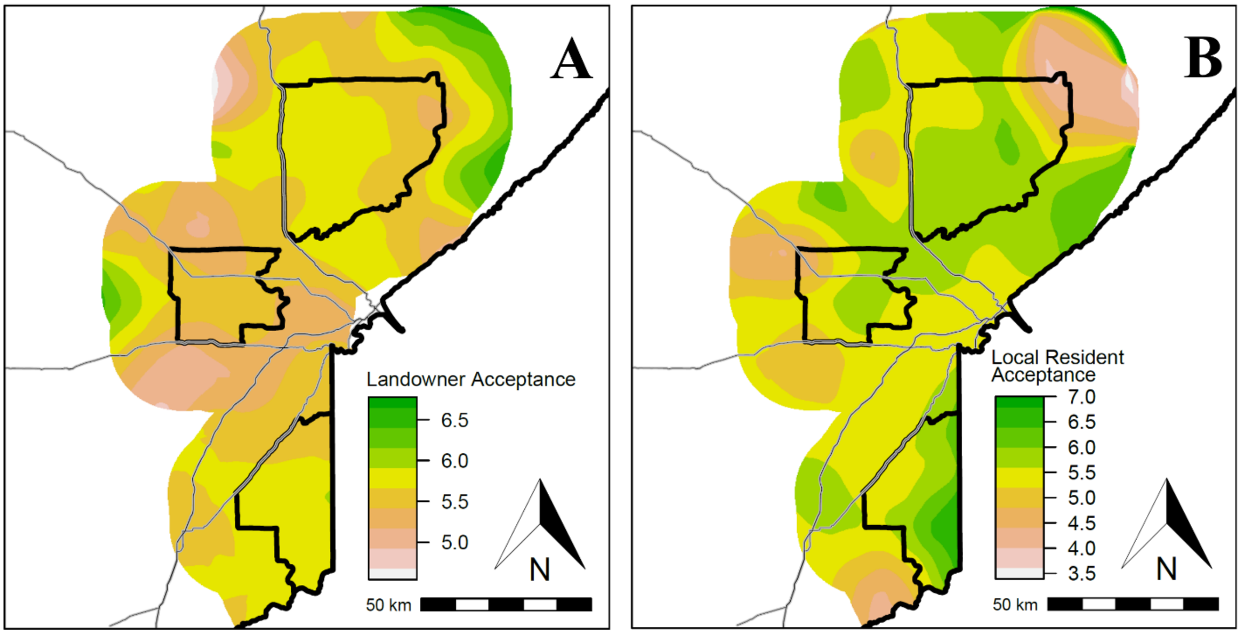

Figure 11. Social acceptance of elk restoration by landowners (A) and local residents (B) on and near 3 study areas in northeastern Minnesota. Acceptance ranges from 1 (low) to 7 (high). The scale bars start at values > 0 as minimum mean acceptance was 4.5 for landowners and 3.4 for local residents.

Risk of human-elk conflict

Conflict risk averaged ≤ 0.10 across the 3 study areas, increasing from north to south (Figure 12). Mean risk on the Cloquet Valley study area was 0.02 (*SD* = 0.02, *N* = 7,841,931 raster cells), while risk on the Fond du Lac study area was 2-times greater (mean = 0.04, *SD* = 0.03, *N* = 3,402,931 cells) and risk on the Nemadji study area was 5-times greater (mean = 0.10, *SD* = 0.05, *N* = 4,279,849 cells). Areas southwest of the Nemadji study area, between the Nemadji and Fond du Lac study areas, and south of the Fond du Lac study area had greater risk (Figure 12).

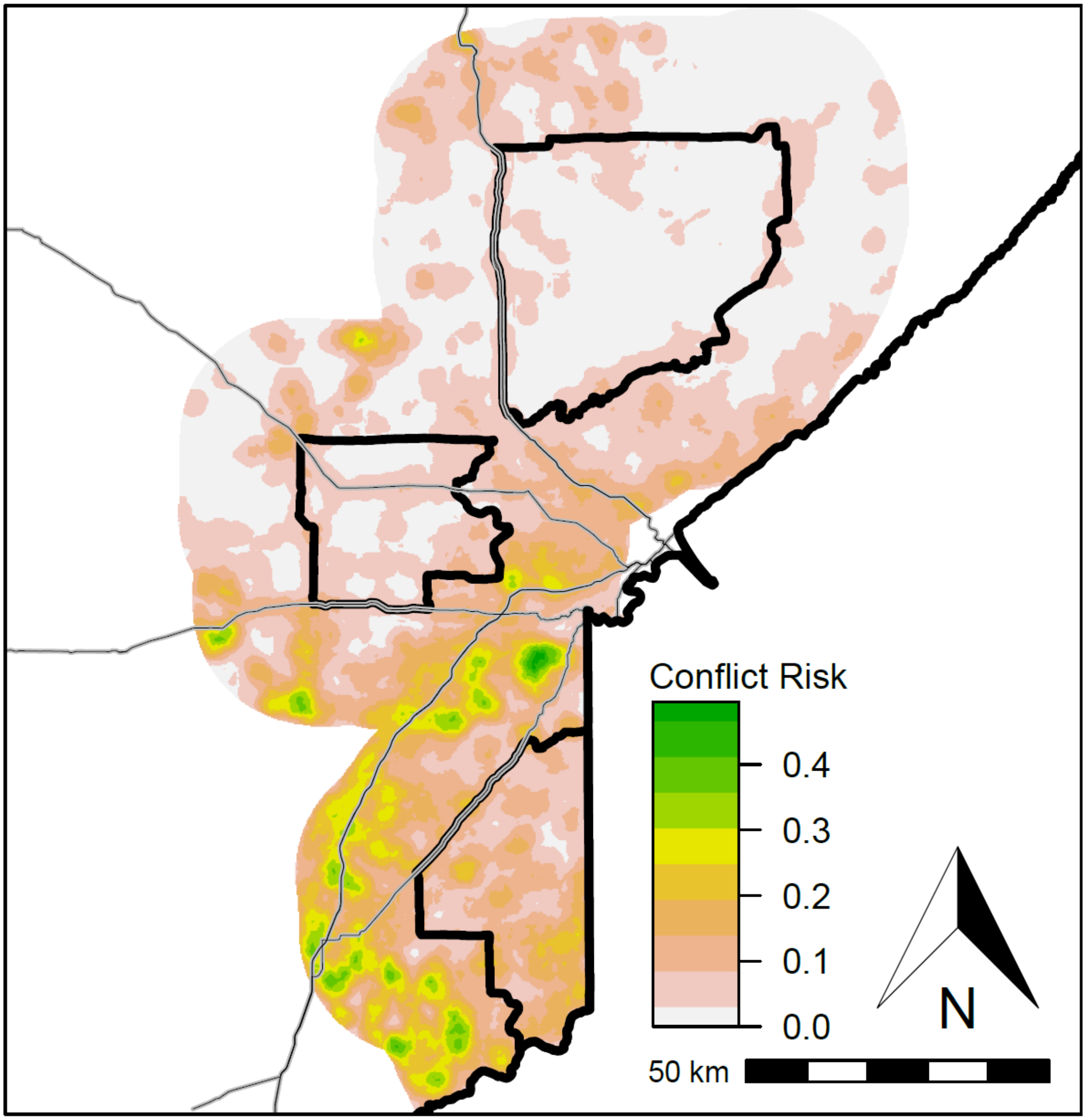

Figure 12. Risk of human-elk conflict in northeastern Minnesota. Conflict risk is the proportion of area that is roads, feedlots, hay/pasture, and row crops.

Winter suitability

By intersecting landowner acceptance scores ≥ 5 with the greater than average winter animal use equivalence and winter habitat suitability, we estimated that 443 km^2 of the Cloquet Valley study area (0.25 of the study area) had both high landowner acceptance and high winter habitat suitability (Figure 13). We estimated that the Fond du Lac study area had 234 km^2 (0.30 of the study area; 1.2 times more than the Cloquet Valley study area) and the Nemadji study area had 138 km^2 (0.14 of the study area) of high winter habitat suitability and landowner acceptance (0.6 and 0.5 times than on the Cloquet Valley and Fond du Lac study areas).

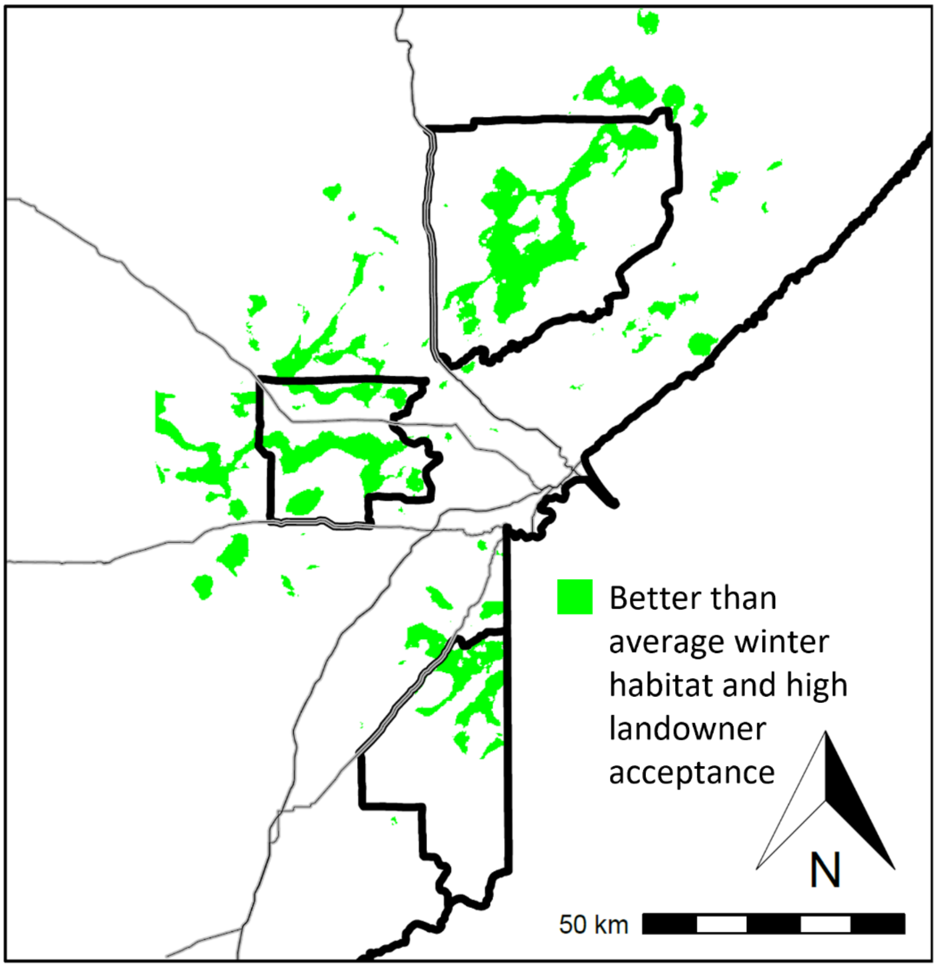

Figure 13. Areas where winter animal use equivalence and winter habitat availability were better than the mean, and landowner support was high (acceptance score ≥ 5).

Ranking study areas

Study areas ranks were similar when we weighted each factor equally. The Cloquet Valley study area had the highest mean rank (1 = worse and 3 = better; Table 8), but ranks were not significantly different (Permutation test; max $T = 1.56$, $P = 0.26$).

Weighting factors resulted in differences between study areas. It required weighting factors a mean of 5.9 times (range 4 to 10 times, $SD = 2.4$ times, $N = 9$ sets of comparisons) to obtain ≥ 1 statistically significant pairwise difference. All 9 sets of 10 weighted comparisons had at least 1 statistically significant pairwise difference (Figure 14). Pairwise post hoc tests that followed significant one-way permutation tests resulted in a mean rank for the Cloquet Valley study area that was greater than the rank for at least 1 other study area in 7 of the 9 sets of comparisons, and less than at least 1 other study area in 2 sets of comparisons (Figure 14). The Nemadji study area had a greater mean rank than at least 1 other study area in 4 of the 9 sets of comparisons, and less than at least 1 other study area in 5 of the 9 sets of comparisons. The Fond du Lac study area had a greater mean rank than at least 1 other study area in 3 of the 9 sets of comparisons, and a lower rank than at least 1 other study area in all of them.

Table 8. Study area ranks (1 = worst and 3 = best) for factors that influence elk restoration success.

Study area	Winter animal use equivalence	Wisconsin habitat suitability index [a]	Summer resource selection [b]	Proportion aspen	Proportion grassland	Proportion public land	Landowner acceptance	Conflict risk	Mean with equal weights
Cloquet Valley	1	3	1	3	3	3	2	3	2.4
Fond du Lac	2	2	2	2	1	2	1	2	1.8
Nemadji	3	1	3	1	2	1	3	1	1.9

[a] Overall suitability score.

[b] Excluded wolf data, as recent wolf territory data for Nemadji study area were not available.

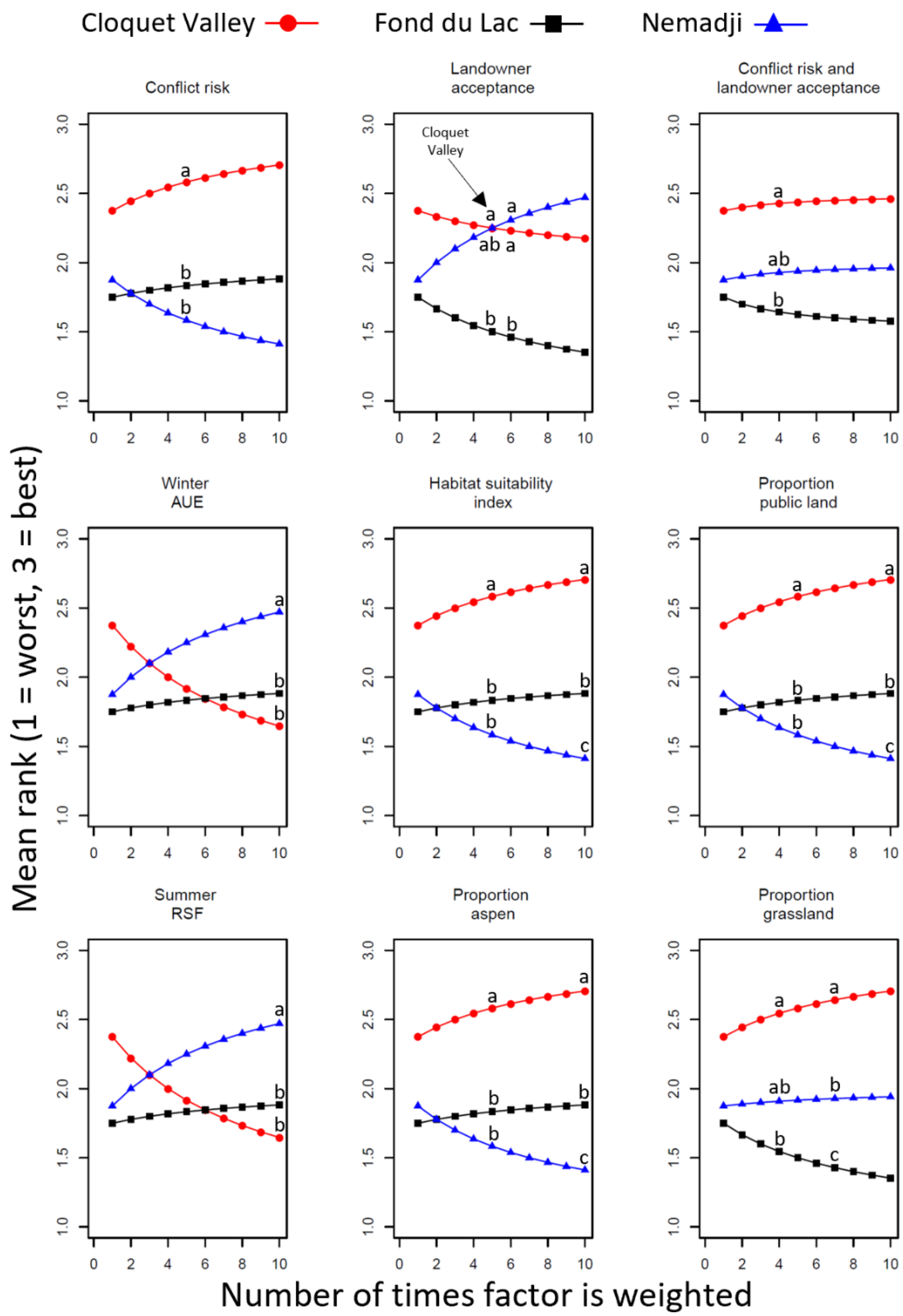

Figure 14. Mean rank of 3 northeastern Minnesota study areas after weighting factors between 1 (even weights) to 10 (factor counted 10 times). Plot titles indicate weighted factor(s). Letters a, b, and c within plots are first instances of statistically significant pairwise differences that continue to be significant at greater weights. For example, the Cloquet Valley rank was first statistically greatest at conflict risk weight = 5 and continued to be greater at ranks ≥ 5. There were 2 sets of differences for panels that have > 1 set of letters. AUE = animal use equivalence. RSF = resource selection function.

Thresholds from other elk studies

Each study area had ≥ 390 km^2 with preferred summer forage and ≥ 225 km^2 of winter forage that was ≥ forage available to elk in Wisconsin (Table 9; Figure 15A and 15B). Each study area had > 740 km^2 with winter AUE ≥ elk density in Wisconsin's Black River Herd (after converting Wisconsin density to elk/km^2; Figure 15C). Areas with winter AUE ≥ elk density in Michigan ranged widely, from a low of 31 km^2 in the Cloquet Valley study area to a high of 720 km^2 in the Nemadji study area. The Wisconsin habitat suitability index was ≥ 0.5 in > 270 km^2 of each study area, equaling 0.35 to 0.45 of each study area (Figure 15D).

Table 9. Amount of each study area ≥ threshold values from elk studies.

	Area (km^2) and proportion of study area above threshold				
Study area / Threshold	Summer forage ≥ 0.120 kg/m^2	Winter forage ≥ 0.017 kg/m^2	Winter AUE [a] ≥ 5 elk/16 km^2	Winter AUE ≥ 7 elk/16 km^2	Wisconsin Habitat Suitability Index ≥ 0.5
Cloquet Valley	642 (0.36)	225 (0.13)	908 (0.51)	31 (0.02)	800 (0.45)
Fond du Lac	390 (0.51)	240 (0.31)	743 (0.97)	152 (0.20)	271 (0.35)
Nemadji	586 (0.61)	449 (0.47)	963 (1.00)	720 (0.75)	403 (0.42)

[a] AUE = Animal use equivalence.

Discussion

Suitable environmental conditions and social acceptance are important for restoring wildlife populations (Fischer and Lindenmayer 2000, Behr et al. 2017). Our results show abundant suitable elk habitat in northeastern Minnesota, and high landowner and local resident support for restoring elk there. Our social suitability results correspond well with a companion report by Walberg et al. (2019) who examined the same landowner and local resident support data in additional ways using different statistical methods.

Forage availability at field plots suggests northeastern Minnesota can support elk. Summer forage on our study areas (0.130 kg/m^2) exceeded amounts elk prefer (0.120 kg/m^2; Wilmshurst et al. 1995). Winter forage on our study areas (0.017 kg/m^2) was the same as in Wisconsin when we excluded grass from Wisconsin estimates (0.017 kg/m^2; Anderson et al. 2005*a*), which made sense as elk rarely consume grass during winter in eastern North America (Jenkins et al. 2007).

AUE estimates indicate northeastern Minnesota can support densities of elk found in Wisconsin and Michigan. Winter AUE in our study (5 to 8 elk/16 km^2) corresponds well with elk densities in Wisconsin's Black River Herd (5 elk/16 km^2) and in Michigan (7 elk/16 km^2; Stowell et al. 2012, MIDNR 2019, WDNR 2019*a*). They were 1.7- to 2.7-times higher than densities in Wisconsin's Clam Lake Herd (3 elk/16 km^2; Stowell et al. 2012, WDNR 2019*b*).

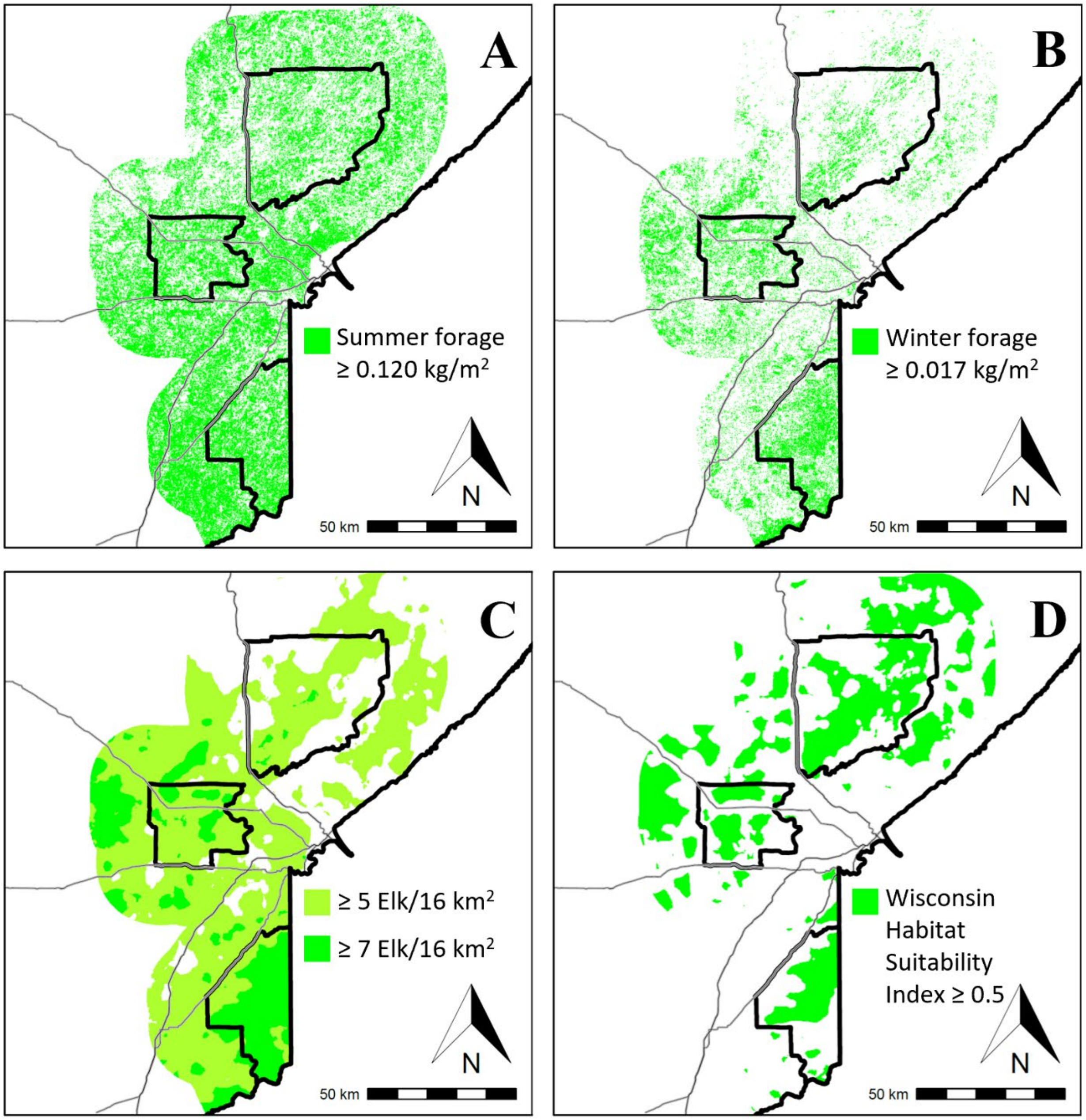

Figure 15. Areas with summer forage (A), winter forage (B), winter animal use equivalence (C), and habitat suitability (D) that are ≥ threshold values from elk studies.

Our study areas had large amounts of habitat with suitability scores similar to where elk occur in Wisconsin (habitat suitability index scores ≥ 0.5; Gilbert et al. 2010, Stowell et al. 2012). Using Wisconsin's Black River Herd's approximately 200 km^2 core area as a reference (Stowell et al. 2012, WDNR 2019*a*), the Cloquet Valley contained about 4-times more suitable habitat, the Nemadji study area contained about 2-times more suitable habitat, and the Fond du Lac study area contained about the same amount of suitable habitat.

Conflict risk adjacent to our study areas may influence public support for elk population expansion. It is likely that low conflict risk in all directions adjacent to the Cloquet Valley study area will enable elk population expansion without eroding public support. Low risk to the west, north, and east from the Fond du Lac study area will also enable elk population expansion. In contrast, areas in Minnesota in all directions from the Nemadji study area had high risk, making it likely that human-elk conflict will reduce public support for elk population expansion there.

Ranking the 3 study areas for elk restoration will be influenced by whether the factors considered are perceived as being equally important or if some factors are perceived as more important than others. Ranking study areas while considering each factor to be equally important (evenly weighted factors) suggests that the 3 study areas were equally suitable for elk restoration (study areas were not statistically different). The study areas were not equally suitable for elk restoration (statistically different study area ranks), however, when we considered some factors to be more important (unevenly weighted factors). Thus, the perceived relative importance of factors we assessed, and others, will influence study area selection if restoration moves forward.

The Cloquet Valley study area is more likely to be considered best for elk restoration when 1 or more of the factors we assessed is perceived to be ≥ 4 times more important than others. Compared to the other study areas, the Cloquet Valley study area had the highest rank most often (statistically greatest rank) when we weighted factors. It ranked best when we weighted each of 5 factors 4 to 7 times (conflict risk, Wisconsin's habitat suitability index, and proportions of public land, aspen, and grassland). By comparison, the Nemadji study area only had the best rank when we weighted each of 2 factors 10 times (winter AUE and summer RSF), and the Fond du Lac study area never rank best.

Multiple methods yielded high habitat suitability estimates, which strengthens our conclusions (Johnson 2007). It is important to note, however, that maps we developed have limitations. For example, although we developed summer AUE maps using forage models that were often within 0.05 kg/m^2 of field observations, these models were only 30% accurate, reflecting difficulty in estimating small diameter shrub and tree biomass over broad areas well.

Predation is likely to influence elk restoration success. Black bears (*Ursus americanus*) and wolves (*Canis lupus*) are present in northeastern Minnesota and both kill elk, thereby reducing restoration success (Frair et al. 2007, Popp et al. 2014, Keller et al. 2015). Resource selection functions we developed showed that wolves will influence elk distributions but did not estimate the influence of predation on population growth. Prior restoration efforts show bears and wolves kill restored elk (WDNR 2019*a*), but surviving elk reduce mortality rates by learning to avoid predators (Frair et al. 2007).

Disease transmission is an important factor when considering elk restoration, but we did not address it in this assessment. Brainworm (*Parelaphostrongylus tenuis*) and chronic wasting disease (CWD) infections, for example, are present in wild ungulates in Minnesota (MNDNR 2017, Carstensen et al. 2018) and kill elk (Keller et al. 2015). Brainworm spreads via consumption of intermediate hosts (snails and slugs), while CWD spreads via direct contact between animals and through exposure to materials contaminated by urine, saliva, feces, and carcasses of infected animals (Gillin et al. 2018). Brainworm is associated with white-tailed deer (*Odocoileus virginianus*) distributions in eastern North America, including in northeastern Minnesota. CWD, however, has not been detected in northeastern Minnesota and CWD transmission risk makes wildlife managers hesitant to relocate cervids (Gillin et al. 2018).

We did not address climate change in this assessment, as we expect elk to adapt to northeastern Minnesota's warming climate. Northeastern Minnesota is projected to have a climate similar to Iowa by 2069 (Galatowitsch et al. 2009), but the range, physiology, and foraging behavior of elk suggest they are not adversely affected by warm climates. The historic elk range included warm areas in North America and elk currently occur in warm climates of Arkansas, Iowa, Kentucky, North Carolina, and Texas (Kagima and Fairbanks 2013; Popp et al. 2014). Although climate warming has been implicated in moose population declines (Weiskopf et al. 2019), elk have a higher upper critical temperature than moose (about 10 °C higher during summer; Parker and Robbins 2009; McCann et al. 2013), suggesting elk will adapt to a warmer climate in northeastern Minnesota. Being mixed feeders that locate forage in many habitat types, elk will also likely adapt to projected habitat conversion to savanna and grassland.

Our findings show widespread suitable habitat and public support for elk restoration in northeastern Minnesota. Human-elk conflict risk is low on our study areas but is high in some nearby areas where a restored elk population might expand. Factors we assessed in this report, and factors we did not assess, require consideration when deciding whether to restore elk.

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