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Integrating socioecological suitability with human–wildlife conflict risk: Case study for translocation of a large ungulate

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Abstract

1. Translocations are essential for re-establishing wildlife populations. As they sometimes fail, it is critical to assess factors that influence their success pre-translocation.
2. Socioecological suitability models (SESMs) integrate social acceptance and ecological suitability to enable identification of areas where wildlife populations will expand, which makes it likely that SESMs will also be useful for predicting translocation success.
3. To inform site selection for potential elk *Cervus canadensis* reintroduction to north-eastern Minnesota, United States, we developed broadscale maps of social acceptance from surveys of local residents and landowners, animal use equivalence (AUE) from forage measured in the field and empirical conflict risk from geospatial data. Resulting SESMs integrated social acceptance favourability scores, AUE and conflict risk, and weighted SESMs showed the relative influences of acceptance and conflict.
4. Social acceptance was positive for local residents and landowners (mean ≥ 5.4; scale of 1–7). AUE (scaled to an elk home range) ranged between 1 and 9 elk/16 km^2 during winter, and from 14 to 83 elk/16 km^2 during summer. Human–elk conflict risk was low (mean ≤ 0.10; scaled 0–1), increasing from north to south. Geographical distributions differed for social acceptance, AUE and conflict risk, and weighted SESMs revealed unsuitable areas that were otherwise obscured.
5. *Synthesis and applications*. Integrating human–wildlife conflict risk into SESMs shows where social acceptance of translocated species is likely to erode, even where viewed favourably pre-translocation, to inform translocation planning by highlighting interactions between key factors. Such integrated models supplement existing reintroduction biology frameworks by supporting decision-making and knowledge development. In north-eastern Minnesota, natural resource managers who are considering elk reintroductions are using SESMs reported here to identify where human–elk conflict is unlikely to result in an isolated elk population and where addressing concerns for area residents about conflict risk is essential.

KEYWORDS

Animal use equivalence, *Cervus canadensis*, elk, human–wildlife conflict, reintroduction, social acceptance, socioecological suitability model, translocation

1 | INTRODUCTION

Translocations are important for re-establishing wildlife populations but are complex and costly (Berger-Tal et al., 2019). Before wildlife translocations occur, it is critical to assess factors that influence success, including source populations, number of animals released and habitat quality (Armstrong & Seddon, 2008; Fischer & Lindenmayer, 2000). Public support is also important for translocations, though it is often underappreciated (Brichieri-Colombi & Moehrenschlager, 2016). Research that uses social surveys to study public support has illuminated understanding of people's acceptance, beliefs about risk and benefit and trust in management agencies (Bruskotter & Wilson, 2014; Schroeder et al., 2018), and has shown how experiences and deep-seated beliefs shape perceptions of wildlife conservation (Dickman, 2010; Teel & Manfredo, 2010).

Integrating social acceptance and ecological suitability supports coexistence in multiuse landscapes by identifying areas where wildlife can thrive (Carter et al., 2020), thereby underscoring the importance of interdisciplinary research (Dickman, 2010). Spatially explicit maps show where ecosystem services overlap with social valuation using proxies (e.g. road proximity) and social surveys, enabling identification of habitat suitability (Behr et al., 2017; Bryan et al., 2011; Treves et al., 2011). As they predict range expansion for existing wildlife populations (Behr et al., 2017), it is likely that such socioecological suitability models (SESMs) will be useful for predicting translocation success (Carter et al., 2020), but we were unable to find any study that has demonstrated this. Although one translocation study integrated leopard *Panthera pardus* habitat maps with maps where people reported killing carnivores (Lemeris, 2013), killings incompletely reflect intolerance (Bruskotter & Wilson, 2014).

Incorporating spatially explicit estimates of human–wildlife conflict (threats to human interests; Treves & Karanth, 2003) into SESMs should provide a better understanding of where post-translocation conflict will erode social acceptance. Conflict ranges in severity, from crop depredation to human fatalities (Nyhus, 2016; Walter et al., 2010), and shapes acceptance of wildlife (Struebig et al., 2018). Attacks on livestock and people, for example, reduced acceptance of resident tigers *Panthera tigris sumatrae* (Struebig et al., 2018). Additionally, acceptance of translocated otters *Lontra canadensis* declined following media reports about public perceptions of reduced fish populations (Serfass et al., 2014) and acceptance of translocated leopards declined following livestock kills (Lemeris, 2013).

Forecasting human–wildlife conflict when translocating a species to an area from which it was extirpated (reintroduction; Fischer & Lindenmayer, 2000) is a special situation as it requires information from other locations where the species still occurs. Wildlife conflict experience partly shapes social acceptance and perceived risk but is absent prior to reintroduction when the species is novel to people at the reintroduction site. Additionally, value orientations, basic beliefs and social processes independent of direct experience influence attitudes, perceived risk and acceptance of wildlife (Fulton et al., 1996; Landon et al., 2020). Such complexity presents a challenge when deciding where to reintroduce wildlife, as perceived risk correlates negatively with management preferences (Schroeder et al., 2018) and, along with perceived benefit, is more influential when forming acceptability judgements than other factors (Bruskotter & Wilson, 2014); however, perceived risk may not reflect empirical risk (Dickman, 2010). As a result, reintroduction sites will sometimes not correspond with low empirical conflict risk and for this reason, social acceptance erodes after conflict occurs post-reintroduction (Weise et al., 2015). This suggests that it will be useful to integrate empirical conflict risk surfaces into SESMs pre-reintroduction to avoid translocation to risky areas, as has been done for naturally expanding wildlife (Coz & Young, 2020; Treves et al., 2011).

Elk *Cervus canadensis* are an excellent species for SESM development because their ecology, reintroduction history and conflict risk are well studied. Millions of elk once ranged over most of North America, but their numbers declined with overexploitation and habitat loss following European colonization (Murie, 1951). Remnant elk populations in western North America became sources for numerous reintroductions (Popp et al., 2014).

Multiple factors influence elk reintroduction success, including predation, disease and habitat quality (Popp et al., 2014). Human–elk conflict also influences reintroduction success. Elk use road right-of-ways with high amounts of forage (Anderson et al., 2005) and habituation to roads results in elk–vehicle collisions (Walter et al., 2010). Elk also damage fences; depredate row crops, grain and hay bales; and exchange disease with livestock, leading producers to implement costly strategies to minimize livestock–elk contact at feedlots (Hegel et al., 2009; Walter et al., 2010).

Vegetation also influences reintroductions (Popp et al., 2014) as elk select areas with abundant forage biomass (Anderson et al., 2005; Hebblewhite et al., 2008; Merrill et al., 2020). In addition to identifying areas elk select, forage biomass also enables computation of potential population density (animal use equivalence, AUE; Kuzyk & Hudson, 2007). Such estimates are essential because they correspond with post-reintroduction persistence (not only establishment; Armstrong & Seddon, 2008).

In this study, we forecasted elk reintroduction success to north-eastern Minnesota, United States. Once ranging throughout most of the state, elk are now only in north-western Minnesota and are managed at small numbers by statute (Minnesota Statute 97B.516) to minimize conflict with agriculture. This statute does not apply to north-eastern Minnesota where there is interest in reintroducing elk. To inform site selection for potential elk reintroduction, we developed SESMs that integrated empirical human–elk conflict risk estimates. Using social acceptance questionnaires and forage availability field measurements, we developed broadscale spatially explicit maps of social acceptance and AUE, and then integrated them with human–elk conflict risk surfaces. We then weighted social acceptance and conflict risk to estimate their relative influence on reintroduction success.

2 | MATERIALS AND METHODS

2.1 | Study area

We studied elk reintroduction to north-eastern Minnesota, United States, where managers are deciding whether to reintroduce elk to one of three study areas [Cloquet Valley (CLV), Fond du Lac (FDL) and Nemadji (NEM)] that have abundant public land and low road density (mean = 0.96 km/km^2) suitable for elk (Lyon, 1983; Figure 1a). In the northern lakes and forests ecoregion (Level III Region 50), the area was characterized by rolling topography, nutrient-poor soils, scattered lakes and rivers (Omernik & Griffith, 2014) and mixed plant vegetative communities. Forests were coniferous and northern hardwood (often mixed). Maps we developed buffered study areas 20 km (elk dispersal distance; Ryckman et al., 2010).

2.2 | Social acceptance

We used local resident and landowner surveys to map social acceptance for elk. We mailed 4,000 local residents a questionnaire (stratified random sampling; stratum levels were census blocks, highways and rivers; see Walberg et al., 2019) about attitudes towards translocating elk and measured attitudes using returned questionnaires (*N* = 1,521) scored from very unfavourable (1) to very favourable (7). To develop a social acceptance model, we mapped mean attitude scores within a circular moving window [size needed for continuous surface; four townships (372 km^2)].

We used the same methods for landowners (owners of ≥4 ha properties located ≤8 km from a study area). Landowner properties were near likely reintroduction sites (outside residential areas and near public lands), making them areas where post-reintroduction human–elk interactions were most likely. We used a stratified random sampling approach to select 4,500 landowners, where stratum was property size (levels: 4–16 ha and >16 ha). Mapped scores were from 2,585 returned questionnaires (including 35 from local residents who met landowner criteria).

2.3 | Forage

We measured trees, shrubs and understorey vegetation between 14 June and 8 August 2017 (season 1) and 6 June and 8 August 2018 (season 2). Sites we sampled were distributed throughout the study areas in a Geographic Information System (ESRI, 2018; R Core Team, 2019; Any use of trade, firm or product names is for descriptive purposes only and does not imply endorsement by the U.S. Government). During season 1, we randomly distributed points on roads that abutted public land, and then randomly sampled one point in each vegetated land cover type (Rampi et al., 2016) within 50–500 m from the road point (to improve logistics). To achieve sampling that was even through space (with respect to roads) and time (date), we systematically sampled study areas and four rectangular quadrants overlaid on them. We repeated this process during season 2 while sampling private properties from landowner questionnaire respondents. As elk in eastern North America select regenerating aspen *Populus tremuloides* as winter forage (Jenkins et al., 2007), we expanded sampling to include randomly selected points in aspen stands harvested ≤10 years ago (*N* = 4 plots per study area; M.P. Westphal, Carlton County; D. Ryan, US Forest Service; J. Kelash, Pine County; and B. Hakala, Saint Louis County; unpubl. data).

We established a circular plot (field plot) centred on each land cover point. Field plots consisted of nested circles (Figure 1b). In the largest circle (*r* = 11.3 m), we collected ground cover vegetation from 10 150 cm × 10 cm quadrats. In each quadrat, we clipped, dried and massed woody (height <15 cm) and herbaceous vegetation (classes: grass, forb, sedge, rush, fern and woody). In three medium circles (*r* = 2.8 m) radiating 5.5 m from the plot centre (120° increments), we used a stepped diameter gauge (Paul et al., 2017) to count trees and shrubs with DBH 2.54 cm to 10 cm. In a small circle (*r* = 1.8 m)

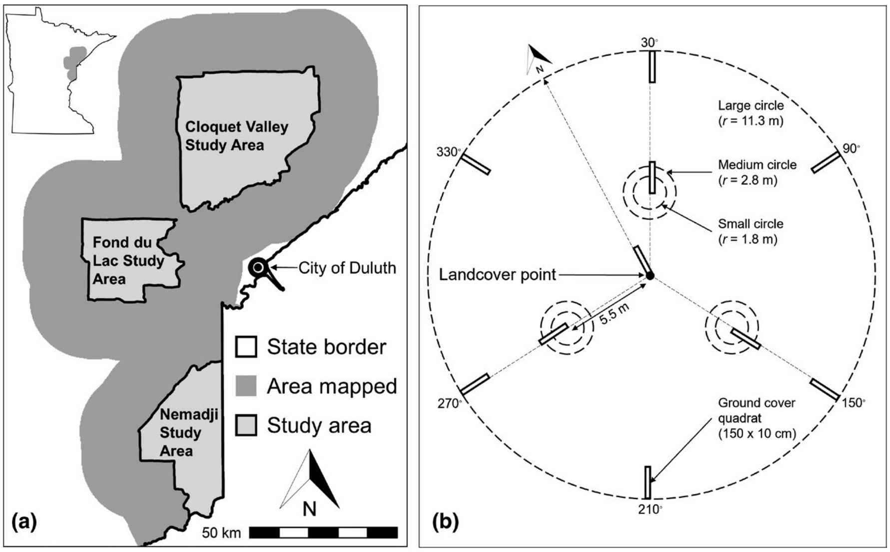

FIGURE 1 Study areas in northeastern Minnesota, USA (a) where we measured vegetation in nested circles and rectangular quadrats around a land cover point (b)

centred within each medium circle, we used a stepped gauge to count trees and shrubs (≥15 cm tall and <2.54 cm diameter at 15 cm height; D15).

During season 2, we sampled open areas near roads, railroad tracks and pipelines (right-of-ways; ROWs). We used a stratified random sampling approach to select ROWs; strata were study area and road class (MNDOT, 2017). Railroad and pipeline ROWs were adjacent to tracks and pipeline service roads, and 50–500 m (randomly selected) from a road intersection (MNDOT, 2015). At each ROW, we established a 200-m^2 rectangular plot (ROW plot). We measured the distance between the road/railroad edge and the nearest tree/shrub line to calculate plot width (maximum of 30 m). Plot length was plot width divided by 200. Once we established an ROW plot, we clipped, dried and massed ground cover vegetation from five quadrats.

We estimated forage biomass (non-avoided species) at each field plot during leaf-on (summer) and leaf-off (winter) because it correlates positively with elk habitat selection across multiple spatial scales (Anderson et al., 2005; Hebblewhite et al., 2008; Merrill et al., 2020). Summer forage was deciduous shrub and tree leaves, forbs and grasses (not sedges, rushes and ferns; Lupardus et al., 2011; Schneider et al., 2006). To estimate shrub and sapling leaf biomass, we summed woody stem counts at field plots and used diameter-specific allometric equations (Jenkins et al., 2004; Perala & Alban, 1993; Smith & Brand, 1983). Winter forage was deciduous twigs (not grasses; see Jenkins et al., 2007). We used allometric equations to estimate total above-ground deciduous shrub and tree biomass (Jenkins et al., 2004; Perala & Alban, 1993; Smith & Brand, 1983) and estimated twig biomass as the product of total biomass and the proportion that is current year growth consumed by ungulates in Minnesota (0.07; Ohmann et al., 1974, 1976). Forage biomass was for shrubs and trees ≤2.54 cm in diameter (D15), corresponding with mean elk foraging height (1.5 m; Rounds, 2006; Gehring et al., 2008; VanderSchaaf, 2013). We used ANOVAs to determine if forage (square root transformed) differed by land cover and ownership (public or private; STATS package in R) and to test for differences in ROW type summer forge. Tukey tests followed significant ANOVAs, α was 0.05 and the variance inflation factor assessed collinearity.

We used random forest (RF) analysis to model field plot forage biomass (Breiman, 2001; Cutler et al., 2007). RF fits a large number of regression trees, with each tree constructed using a random subset of data and predictors. This results in estimates that are essentially cross-validated, making it unnecessary to divide data into training and test sets (Chen et al., 2017; Prasad et al., 2006). RF results in predictions of the dependent variable, and measures of accuracy and variable importance (Liaw & Wiener, 2002). Frequently used in geospatial modelling (Evans & Cushman, 2009; Rodriguez-Galiano et al., 2012), RF models nonlinear relationships and interactions without error distribution assumptions (Cutler et al., 2007) and is robust to missing data (Rodriguez-Galiano et al., 2012) and overfitting (Breiman, 2001), yielding more accurate results than other methods (Chen et al., 2017; Prasad et al., 2006).

To estimate field plot forage biomass, we implemented RF (RANDOMFOREST package; Liaw & Wiener, 2002) using biologically relevant predictors extracted from 15-m resolution rasters (Table 1). We eliminated correlated independent variables (Millard & Richardson, 2015; Spearman correlation coefficient $|r_s| > 0.5$; STATS package in R) by keeping only the variable that resulted in greater predictive accuracy (Gustafson et al., 2003). RF models predicted leaf or total biomass by growing 1,000 regression trees. Using RF forage estimates (not spatially autocorrelated; Moran's I test, $p > 0.25$; SPDEP package in R; Bivand & Wong, 2018; Millard & Richardson, 2015), we estimated summer and winter forage across the mapped area (Hijmans & van Etten, 2019). ROW forage was overlaid after assigning mean forb and grass biomass (field measurements). Mean pipeline forage was used for unsampled high-voltage power lines (similarly sized openings and vegetation; Minnesota Geospatial Information Office, 2016).

2.4 | Animal unit equivalence

To estimate AUE, we used RF forage estimates summed within 16-km^2 circular moving windows to match an elk home range (O'Neil & Bump, 2014), using: AUE = $(F \times C)/(S \times M \times D)$, where F is forage, S is dry forage (as % elk mass) required to sustain one elk of mass M for 1 day during a season lasting D days and C is a correction factor reflecting how much forage elk consume. AUE was for cow elk (M = 250 kg; Bender et al., 2006), consuming 2.1% of M daily (Christianson & Creel, 2009) during a 200-day winter and 2.2% of M daily (Kuzyk & Hudson, 2007) during a 165-day summer. To account for shrubs and trees not consumed within their use areas, we estimated that elk consume the same proportion of available forage as do moose (*Alces*; C = 0.03 of forage; Edenius et al., 2002; Peek et al., 1976). Resulting maps estimated elk supported by the surrounding 16 km^2, a scale that is biologically significant to elk (home range; O'Neil & Bump, 2014). We reported the mean and standard deviation (*SD*) of raster map values in each study area but did not develop statistical comparisons as the large number of raster cells (CLV: 7,841,931 cells; FDL: 3,402,931 cells; NEM: 4,279,849 cells), made p-values uninformative (Lin et al., 2013).

2.5 | Human–elk conflict risk

To estimate empirical human–elk conflict risk, we developed a surface reflecting the proportion of area (16-km^2 circular window) that was the sum of row crop, hay/pasture, feedlot and road.

Surface values were expected probabilities of elk traversing these features. Crop and hay/pasture were from land cover data (Rampi et al., 2016). Road surface was centrelines (MNDOT, 2017) buffered by mean road widths (field measurements by class). We estimated feedlot area by buffering cow, horse and pig feedlot locations by 0.12 km^2 (median of 20 random feedlots; MPCA, 2007) and added all four captive cervid operations within the mapped area (Minnesota Board of Animal Health, unpubl. data). Nearly all 923

TABLE 1 Variables used to model elk forage in northeastern Minnesota

Independent variable	Description
Cumulative topographic index	Topography-based wetness index
Elevation	Bare earth[a]
EVI for spring	April 2018[b] vegetation index
EVI for summer	August 2018[b] vegetation index
Height	Height above-ground (any object)[a]
Insolation	Solar radiation (WH/m^2)[a]
NDMI for spring	April 2018[b] moisture index
NDMI for summer	August 2018[b] moisture index
Ownership	Private or public
Phenology	Days into summer when sampled
Study area	CLV, FDL, or NEM
Years since disturbance	Years post-disturbance (e.g. timber harvest)[b]

[a] From Lidar.

[b] From Landsat 8.

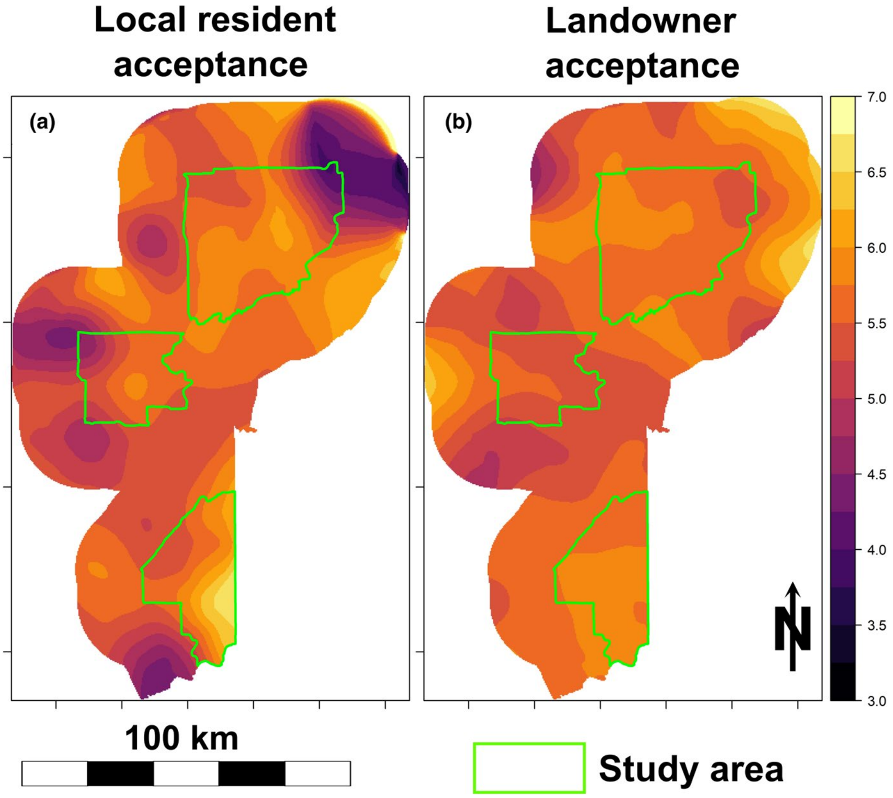

FIGURE 2 Social acceptance for local residents (a) and landowners (b) in northeastern Minnesota. Scale bar starts at minimum mean acceptance

feedlots had areas where elsewhere in elk range livestock–elk contact occurs and elk raid forage (98% had open lots and pasture and 84% had holding areas; MPCA, 2007).

2.6 | Socioecological suitability

To develop integrated socioecological suitability maps, we calculated mean AUE (summer and winter, weighted equally), social acceptance (residents and landowners, weighted equally) and empirical conflict risk after normalizing map values between 0 and 1 (min–max scalar). We subtracted conflict risk from 1 before normalization to ensure all values corresponded positively with suitability. The resulting map weighted AUE, acceptance and conflict equally. To assess their relative influence, we developed maps that weighted acceptance and conflict (each weighted two and three times of baseline map value). Using continuous variables enabled us to assess an array of interacting conditions, thereby obviating potentially influential thresholds where continuous data are discretized into categorical datasets (Behr et al., 2017).

3 | RESULTS

3.1 | Social acceptance

Acceptance was high across the mapped area, but local resident acceptance was lower in the very west, northeast and south (Figure 2). Local resident and landowner acceptances were highest on CLV (resident mean = 5.6, *SD* = 0.40; landowner mean = 5.7, *SD* = 0.20) and NEM (resident mean = 5.7, *SD* =0.38; landowner mean = 5.8, *SD* = 0.14), but lower on FDL (resident mean = 5.4, *SD* = 0.30; landowner mean = 5.5; *SD* = 0.15).

3.2 | Forage

We sampled forage at 186 field plots (CLV $N = 63$; FDL $N = 69$; NEM $N = 54$). Mean summer forage 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 land cover ($F_{5,177} = 2.65$, $p = 0.02$). Public land had more winter forage than private (Tukey $p < 0.05$; Figure 3a) and forested shrub wetland had more than grassland (Tukey $p < 0.05$; Figure 3b). Summer forage differed by land cover ($F_{5,177} = 3.20$, $p < 0.01$) but not ownership ($F_{1,177} = 0.24$, $p = 0.62$). Grassland had more summer forage than coniferous (Tukey $p < 0.05$) and mixed forest (Tukey $p < 0.05$; Figure 3c). All interactions were non-significant ($p > 0.05$).

The winter forage RF model estimated ≤0.043 kg/m^2 across study areas, with 50% of predictions within 0.007 kg/m^2 of field observation and 75% within 0.015 kg/m^2 (pseudo-$R^2 = 0.19$). The most important variables in the winter RF were phenology and August and April EVI. Summer forage was an order of magnitude greater than winter, ranging up to 0.433 kg/m^2, with 50% of predictions within 0.050 kg/m^2 of field observation and 75% were within 0.080 kg/m^2 (pseudo-$R^2 = 0.30$). April EVI, phenology and August NDMI were the most important summer variables.

Mean ROW forage was 0.122 kg/m^2 ($SD = 0.077$, $N = 6$) for railroads, 0.136 kg/m^2 ($SD = 0.117$, $N = 21$) for roads and 0.409 kg/m^2 ($SD = 0.127$, $N = 2$) for pipelines. Forage differed by ROW (all roads combined; $F_{2,26} = 4.53$, $p = 0.02$), as pipelines had more forage than roads (Tukey test $p = 0.02$) and railroads (Tukey test $p = 0.03$), but forage did not differ by road type ($F_{3,17} = 0.63$, $p = 0.61$).

3.3 | Animal unit equivalence

Winter AUE increased from north to south (Figure 4a); it was lowest on CLV (mean = 5 elk/16 km^2; *SD* = 1), intermediate on FDL (6 elk/16 km^2, *SD* = 1) and highest on NEM (8 elk/16 km^2, *SD* = 1). Summer AUE also increased from north to south but was less variable than winter AUE (Figure 4b). Summer AUE was lowest on CLV (mean = 58 elk/16 km^2; *SD* = 8), intermediate on FDL (67 elk/16 km^2; *SD* = 6) and highest on NEM (69 elk/16 km^2; *SD* = 4).

3.4 | Human–elk conflict risk

Mean empirical human–elk conflict risk increased from north to south (Figure 4c). Risk on CLV was 0.02 (*SD* = 0.02) and was two times greater on FDL (0.04, *SD* = 0.03) and five times greater on NEM (0.10, *SD* = 0.05). Riskiest areas were southwest of NEM and between NEM and FDL.

3.5 | Socioecological suitability

Integrating AUE, social acceptance and empirical human–elk conflict risk with equal weights resulted in high suitability for elk across study areas (Figure 5). Mean relative suitability was 0.68 (*SD* = 0.05) on CLV, 0.69 (*SD* = 0.03) on FDL and 0.71 (*SD* = 0.04) on NEM. Increasing social suitability weights resulted in reduced suitability on FDL and northeast CLV. Increasing conflict risk weights highlighted a band of unsuitable area between FDL and NEM.

4 | DISCUSSION

Reintroductions are important for rewilding and mitigating impacts of global climate change (Armstrong & Seddon, 2008; Jarvie

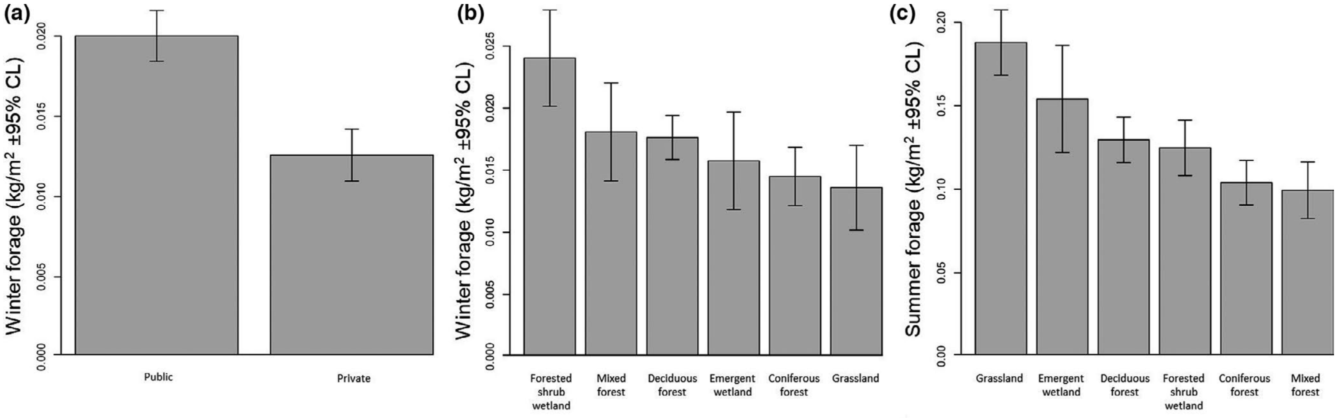

FIGURE 3 Winter forage by land ownership (a) and land cover (b), and summer forage by land cover (c) in northeastern Minnesota

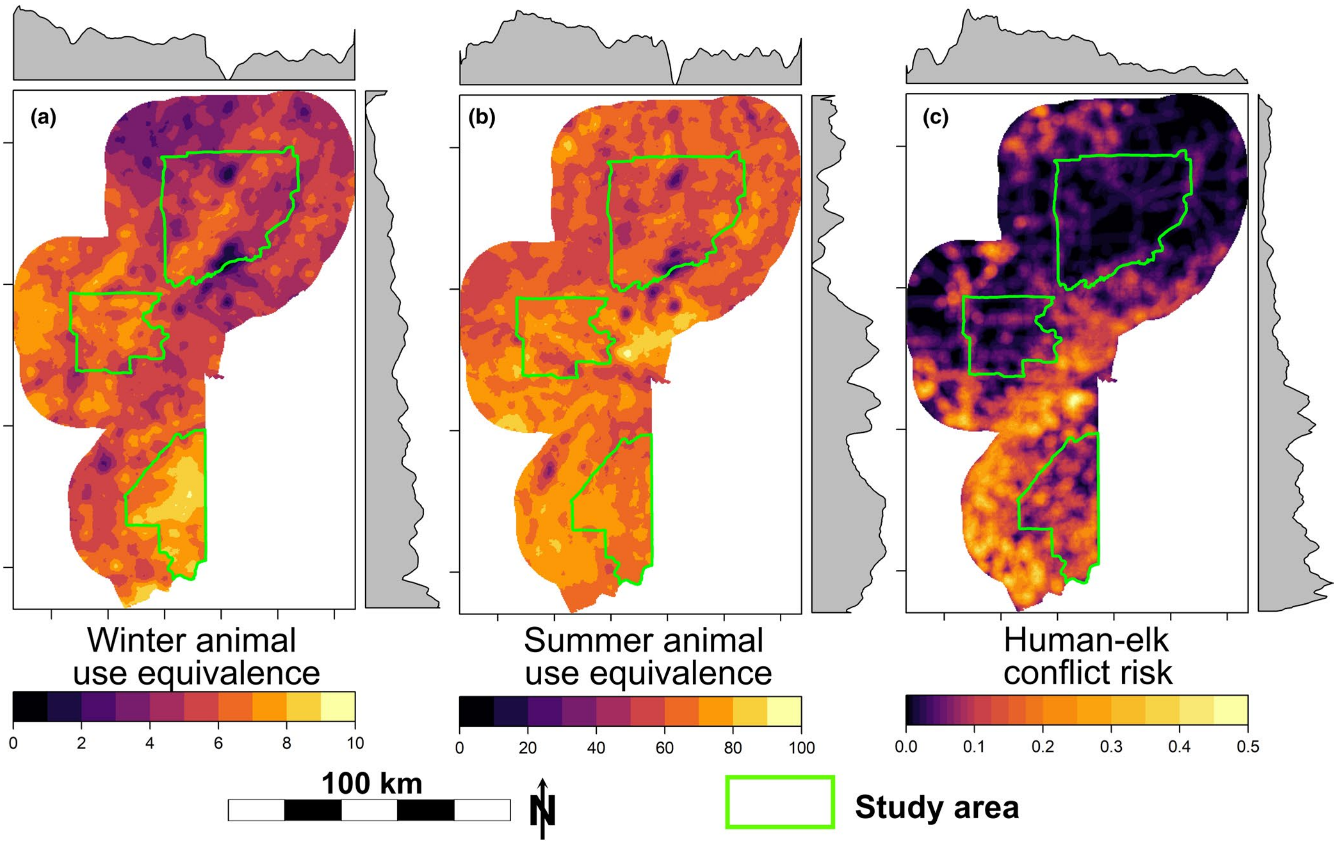

FIGURE 4 Animal unit equivalence (elk/16 km^2) during winter (a), summer (b) and human–elk conflict risk (c) in northeastern Minnesota

& Svenning, 2018); however, forecasting reintroduction success is complex as it is influenced by multiple often interacting factors including social acceptance, habitat quality and human–wildlife conflict (Berger-Tal et al., 2019). Our results show that integrating these factors informs translocation planning by illuminating interactions that are otherwise obscured.

Among many challenges associated with reintroductions, identifying optimal reintroduction sites is difficult because multiple factors influence reintroduction success; as such, selecting sites based only on one factor can reduce reintroduction success (Berger-Tal et al., 2019). Focusing only on habitat, for example, reduced reintroduction success for otters and leopards (Serfass et al., 2014; Weise et al., 2015). As was found for naturally expanding wolves *Canis lupus* (Behr et al., 2017), we found that SESMs that integrated social acceptance refined understanding of where reintroduced elk will be most successful. For example, despite both areas having relatively high AUE that was similar to regional elk densities (McCann et al., 2019), SESMs predicted higher reintroduction success in the south-western CLV relative to the north-eastern CLV due to variable resident acceptance.

Integrating empirical conflict risk into SESMs enabled identification of areas where social acceptance of an expanding elk herd is expected to erode. This finding highlights the importance of integrating conflict risk into decision-making for reintroductions, as it does when developing management plans for naturally expanding wildlife populations, such as wolf and beaver *Castor fiber* (Treves et al., 2011; Coz & Young, 2020). When we increased the weight of conflict risk, an unsuitable band around NEM stood out, suggesting elk reintroduced to NEM will be isolated if conflict results in management actions that limit elk numbers (as occurred in north-western Minnesota). Such isolation increases the risk of reintroduction failure by negatively influencing population dynamics and genetic structure (Armstrong & Seddon, 2008). Conversely, our results suggest that human–elk coexistence is more likely in other study areas because suitability remained high, even after increasing the weight of conflict risk.

To increase the likelihood for coexistence, our findings suggest that addressing concerns about conflict risk is most important on FDL. Weighted SESMs showed areas of low acceptance on FDL where landowners and some local residents perceived elk to pose greater risk to economics and safety, and less overall benefit (Walberg et al., 2019). Although most important for FDL, any reintroduction site will benefit from effective communication with the local community that focuses on risk and benefit, while recognizing that communicator credibility influences receptivity (Bruskotter & Wilson, 2014). By engaging the community, managers will better understand social barriers to reintroduction, thereby identifying key behavioural changes to enhance elk–human coexistence (van Eeden et al., 2021).

Accurately forecasting coexistence will likely benefit from including greater social and ecological complexity when developing

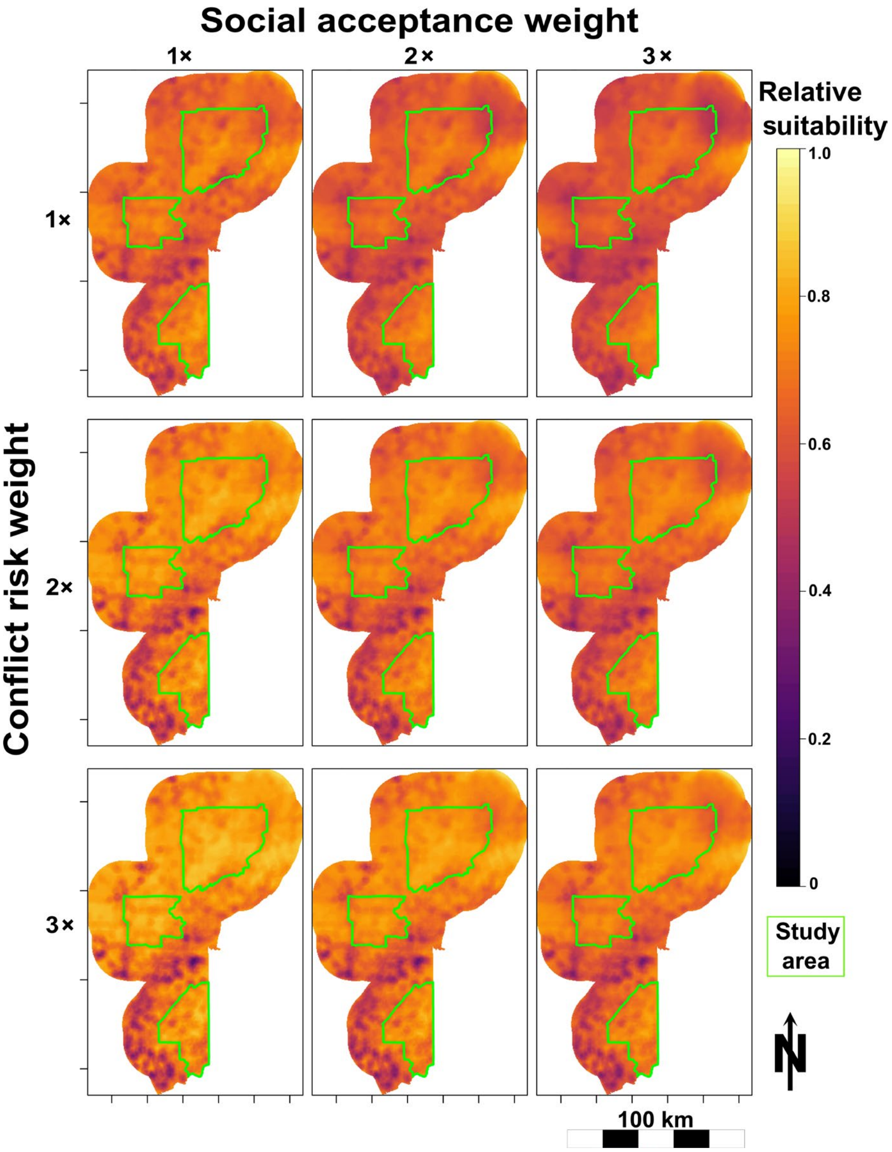

FIGURE 5 Relative suitability for elk in northeastern Minnesota. Maps are mean of animal use equivalence social acceptance and human–elk conflict risk (scaled 0–1). AUE weighted 1. Acceptance and conflict weighted 1–3

conflict risk surfaces. Our conflict surfaces predicted negative consequences for reintroduction success due to reduced social acceptance post-conflict. While conflict factors we integrated negatively influence perceptions of wildlife (Dickman, 2010; Leong, 2010), the relationship is not straightforward. Roads, for example, are associated with wildlife–vehicle collisions that reduce social acceptance (Leong, 2010); however, increased wildlife tolerance corresponds with positive meaningful events like wildlife viewing that often occurs from roads (Marino et al., 2021). Likewise, while crop depredation results in negative perceptions of wildlife (Dickman, 2010), such perceptions vary among landowners and crop types (Hinton et al., 2020; Walberg et al., 2019). Moreover, the influence of conflict factors on biological success is complex. Roads, for example, reduce population performance by facilitating hunter access and elk–vehicle collisions (Frair et al., 2007), but roadsides with abundant forage are expected to benefit elk because forage correlates positively with population performance (Cook et al., 2004). This suggests that improved decision-making will result from a better understanding of the post-reintroduction net influence of conflict factors on social acceptance and population performance, with additional research needed to determine optimal frequencies of post-reintroduction social acceptance measurements, the results of which we predict will vary by wildlife species and population characteristics (e.g. human and wildlife density).

Although we integrated key factors associated with reintroduction success (Berger-Tal et al., 2019), other factors will likely improve future SESMs. For example, additional mapping of conflict factors and fine-scale participatory mapping are expected to improve estimates of conflict risk (McInturff et al., 2021), while predation, disease and global climate change will improve understanding of ecological factors (Berger-Tal et al., 2019; Jarvie & Svenning, 2018). Predation, for example, influences reintroduction success (Popp et al., 2014), though the influence of predation on SESM accuracy is likely to decrease over time as elk learn to avoid predators (Frair et al., 2007).

By integrating key ecological and social factors, SESMs support key elements of reintroduction biology frameworks, including decision-making and knowledge development (Armstrong & Seddon, 2008). Tools that enable managers to select among alternative management actions with a goal of long-term persistence (population density) in addition to short-term establishment, while also developing knowledge by testing a priori hypotheses, are essential to reintroduction biology (Taylor et al., 2017). Accordingly, our SESMs integrated long-term persistence (AUE), thereby enabling elk managers to compare study areas and specific locations within them while accounting for the broader landscape when selecting from among alternative reintroduction sites. Additionally, weighted SESMs were de facto predictions about influences of social acceptance and conflict risk on reintroduction success that are testable post-reintroduction. Thus, in addition to complementing structured decision-making approaches (Martin et al., 2009) by enabling visualization of the relative influence of key-weighted factors, our results suggest that integrating social acceptance and conflict risk supplements existing reintroduction biology frameworks.

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CONFLICT OF INTEREST

The authors do not declare any conflict of interest.

AUTHORS' CONTRIBUTIONS

D.C.F., J.D.F., M.W.S. and N.P.M. conceived the ideas; E.M.W. and N.P.M. led data collection; N.P.M. led data analysis and writing. All authors designed methodology and contributed to drafts and approval of publication.

DATA AVAILABILITY STATEMENT

Data available via the Dryad Digital Repository https://doi.org/10.5061/dryad.4b8gthtdc (McCann et al., 2021).

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