HABITAT SELECTION BY MOOSE IN AN EMERGENT LOW-DENSITY EDGE POPULATION

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ABSTRACT: The Adirondack Park in northern New York contains about 700 moose (*Alces alces*) that persist as a low-density population (0.03 moose/km^2) that occurs along the periphery of the moose's southern range in the eastern United States. As part of a comprehensive effort to evaluate the status of the New York moose population, we fitted 26 moose with GPS collars during 2015–2017 and assessed summer (June–August) and winter (December–March) resource selection to understand moose space use and potentially limiting factors (e.g., climate, forage availability). Home ranges ($\bar{x}$ = 22 km^2) predominately contained deciduous forest, including managed forest stands recently harvested for timber. During summer moose did not exhibit variation in selection among years suggesting that adequate forage may be available across the landscape regardless of habitat type. Moose resource selection within home ranges was most variable during winter, and moose selected areas of managed timber during the most severe winters. Observed habitat selection highlights the potential of direct and indirect interactions with white-tailed deer (*Odocoileus virginianus*), given that deer in the Adirondack Park forage in areas selected by moose such as those with regenerative timber. Because white-tailed deer are an intermediate host for two fatal moose parasites (*Parelaphostrongylus tenuis* and *Fascioloides magna*), increase in habitat overlap between moose and deer could be detrimental to the long-term health of the New York moose population. Additionally, the dependence of both moose and white-tailed deer on regenerating forest stands for optimal forage could put commercial stands at risk of regenerative failure.

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Moose (*Alces alces*) occurred in the State of New York from the Pleistocene until the late 19th century, and prehistorical evidence suggest that moose were present in the northern part of the state, north of the Mohawk River (Fischer 1955, Ritchie 1969, Ritchie and Funk 1973), corresponding largely with the present-day Adirondack Park and Forest Preserve. Moose were extirpated from New York by 1861 due to intense forest management, timber extraction and unregulated hunting (Grant 1894). Several failed reintroductions occurred in northern New York between 1870 and 1902 (Colvin 1880, Wish 1902, Barnham 1909, Bump 1940). By the late 1950s, transient moose from neighboring states and provinces (Massachusetts, Quebec and Vermont) occasionally ventured into areas of New York (Severinghaus and Jackson 1970) with moose becoming a permanent resident of northern New York by the 1980s (Hicks 1986, Hicks and McGowan 1992, Garner and Porter 1990, Hickey 2008). Today, the Adirondack Park (hereafter the Park) population is estimated to be approximately 700 moose and persists as a low-density population (0.29 moose/km^2) along the moose's southern geographical extent (Hinton et al. 2022a). Generally, moose exist in clustered pockets of high-quality habitat in the northern region of the Park, resulting in localized area with moose densities of 0.09–0.12 moose/km^2 (Hinton et al 2022a).

Commercial timber harvests have long been identified as an important tool for creating optimal habitat for moose in North America (Raymond et al. 1996, Bergeron et al. 2011, Andreozzi et al. 2014, 2016, Peterson et al. 2020, 2022). Young regenerating forests created post-harvest (<20 years; Peek et al. 1976) often provide preferred forage that is both high in abundance and high in nutritional quality (Rea and Gillingham 2001, Peterson et al. 2020), and can be used as a means of thermoregulation (Renecker and Hudson 1986, Thompson et al. 2021). Though regenerating forests can provide most of the summer forage (Peterson et al. 2022), moose habitat selection can still vary seasonally. Moose have been documented to transition between regenerating deciduous and conifer forests in winter (Courtois et al. 2002, Andreozzi et al. 2016) and then increasing their use of wetlands and deciduous stands and decreasing their use of coniferous cover in warmer months (MacCracken et al. 1997, Laforge et al. 2016, Teitelbaum et al. 2021).

Despite the abundance of optimal foraging habitat via young forests, neighboring New England moose populations have experienced recent declines. These population declines have highlighted the importance of identifying resource use beyond the traditional lens of intraspecific resource competition, resource availability, and predation (Jones et al. 2017, 2019; Debow et al. 2021). Detrimental parasites, such as winter tick (*Dermacentor albipictus*), brainworm (*Parelaphostrongylus tenuis)* and liver fluke (*Fascioloides magna*), can potentially thrive in areas of optimal moose habitat. This shift in focus on parasite mediated population responses may be correlated with the northward expansion of white-tailed deer (*Odocoileus virginianus*) over the past few decades (Dawe and Boutin 2016, Ditmer et al. 2020). White-tailed deer are aptly suited to fill the ecological niche created by commercial harvest for the similar reasons as moose (Côté et al. 2004). Concerningly, white-tailed deer act as a primary host for two of the detrimental parasites (*P. tenuis, F. magna*; Vanderwaal et al. 2015, Vannatta and Moen 2016). This results in forage abundance and habitat quality mediating parasite dynamics through effects on moose and white-tailed deer host densities and space use (Lankester and Foreyt 2011, Healy et al. 2018). By identifying preferred habitat and environmental conditions, managers can focus monitoring in areas which may present the greatest likelihood of parasitic impacts through habitat niche overlap between deer and moose.

Our primary objective was to determine if moose in the Park select for early successional habitat created by forest management. Broadly, we hypothesized moose would preferentially use certain landcover types, and selection would vary seasonally. We defined four competing predictions: (1) moose do not select for recently managed forested habitat (i.e., early successional, or regenerating stands); (2) moose select for recently managed forest regardless of harvest regime; (3) moose selection of managed forest differs by harvest regime (intermediate or overstory removal); and (4) moose selection differs by both harvest regime and forest type (deciduous, conifer). We tested these hypotheses in winter and summer and in multiple years with variable winter severity. In addition, as a secondary objective we examined if moose selected for wetland areas due to the abundance of forage and utility for thermoregulation that those habitats provide.

STUDY AREA

We studied moose resource selection in the Adirondack Park (24,281 km2; Figure 1) in northern New York. Elevations range from 100 m in low-lying lake shores to about 1600 m. The Park consists of large glacial

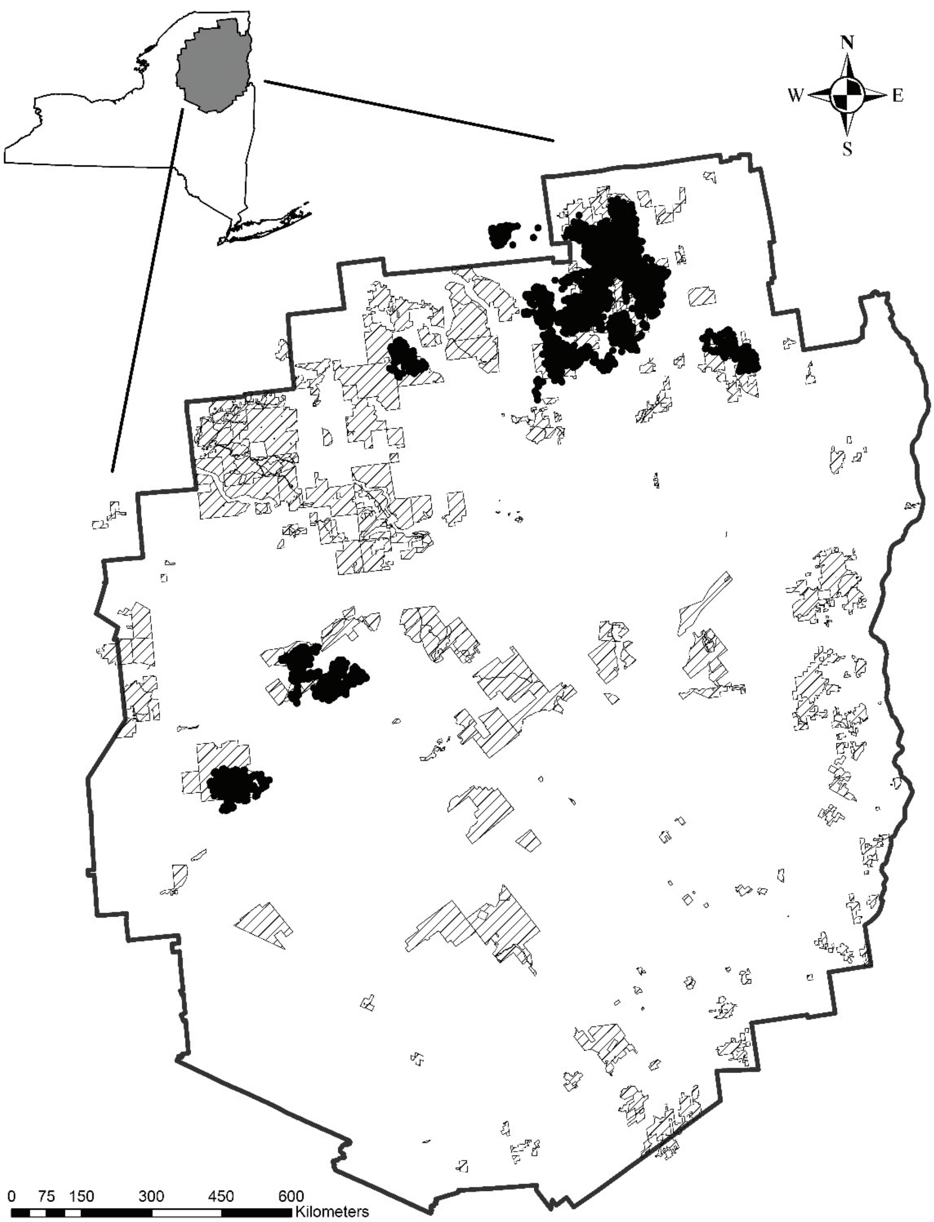

Fig. 1. Map of the Adirondack Park boundary in northern New York, USA. Simple hatch pattern denotes areas of conservation easements and black dots are locations of collared moose from 2015 to 2017. The inset map is a representation of the state of New York, USA.

valleys that gradually rise in elevation to the High Peaks region in the east-central part of the Park. Average monthly temperatures ranged from -9°C in winter to 18°C in summer. The region received an average of 100 cm of rainfall precipitation a year, with an additional 290 cm of snowfall annually (Jenkins and Keal 2004).

The Park included public (61%) and private (39%) lands, where all public lands were protected by Article XIV of the New York State Constitution as "forever wild forest" which prohibits any resource extraction (i.e., timber harvest) or development (N.Y. Const. art. XIV, § 1) on public lands regardless of purpose. The Park comprises a patchwork of the late seral stage northern boreal ecosystem interspersed with temperate deciduous forests and large peatland complexes. Lower elevations with fertile soils support diverse tree species dominated by American beech (*Fagus grandifolia*), yellow birch (*Betula allegheniensis*), paper birch (*B. papyrifera*), sugar maple (*Acer saccharum*) and red maple (*A. rubrum*). Higher elevations are more coniferous, dominated by species such as red spruce (*Picea rubens*), balsam fir (*Abies balsamea*), white pine (*Pinus strobus*) and eastern hemlock (*Tsuga canadensis*; Jenkins and Keal 2004, Peterson et al. 2020).

About 25% of forested private lands in the Park (13% of all Park land) are enrolled in the New York State Conservation Easement Program (NYSDEC 2022). Private properties enrolled in an easement participate in a structured forest management program, which allows for timber harvest and other associated activities. Lands subjected to timber management are predominately composed of marketable timber species such as sugar maple, red maple, red oak (*Quercus rubra*), white ash (*Fraxinus americana*), black cherry (*Prunus serotina*) and white pine; Peterson et al. 2020). Timber management methods include shelterwood removal, overstory removal, single tree selection and salvage thinning. Because public land acquisition continues within the Park, portions of public land could have been exposed to resource extraction immediately prior to acquisition by New York State.

METHODS

Moose Capture and Health Assessment

In 2015–2017, we captured 26 adult and subadult moose (3 male, 23 female) by net-gun fired from helicopters by Native Range Capture Services during January when snow conditions provided increased visibility and limited potential injury to moose. Given the overall low density of moose population within the Park (see Hinton et al. 2022), the majority of moose were captured in the northern portions of the Park where the population density was the greatest to increase the efficiency of capture efforts. We fitted captured moose with Iridium Global Positioning System (GPS) collars (BASIC Iridium Track M 3D, Lotek Wireless, Newmarket, ON; or TGW-4670-3, Telonics, Mesa, AZ) and estimated their ages based on a visual inspection of tooth wear and body size. Moose were released at capture sites. We programmed collars to attempt a GPS location every 2 hours for 2 years, and collars achieved a mean location rate of 98.7 ± 1.1% (Peterson et al. 2020). Animal capture and handling protocols met American Society of Mammalogists recommended guidelines (Sikes et al. 2016) and were approved by the State University of New York College of Environmental Science and Forestry Animal Care and Use Committee (Protocol #140901).

Data Analysis

We analyzed moose 3rd-order resource selection for years 2016—2019, comparing used and available locations within the home ranges of individual moose (Johnson 1980). Used locations were the GPS fixes described above, with data truncated to remove the first two weeks following capture or last two weeks prior to collar release or death (if applicable). We defined availability as the cumulative home range of each moose, calculated using all available GPS data from 2015-2019 and Brownian bridge movement models (Horne et al. 2007) using the R (R Core Team 2022) package `adehabitatHR` (Calenge 2006). We randomly sampled available locations for each moose equal to the number of used locations.

We used spatial rasters of the study area with the following land cover categories: deciduous forest, conifer forest, mixed forest, grassland, and wetland. Forested cells (e.g., deciduous, conifer and mixed) were further classified as either mature forest, intermediate removal, or overstory removal (Kramer et al. 2022), resulting in 11 landcover classes (Table 1). The land cover raster combined National Land Cover Database (NLCD) data (Yang et al. 2018), Landsat 8 satellite imagery, and timber management polygons obtained from timber companies (Kramer et al. 2022). We obtained rasters for years 2015 – 2018. We calculated the minimum distance from each used and available location to each of the land cover types using the R package `raster` (Hijmans 2021). We also calculated the distance to the nearest managed forest of either type, and the nearest intermediate removal and overstory, regardless of forest type. For each used and available point, we derived landcover covariates from the most recent raster layer available (e.g., some points were given values from the previous calendar year).

Table 1. Landcover classifications and source data used to assess habitat selection of moose in northern New York, USA. All canopy removals have occurred within the past 20 years.

Land cover class	Definition
Deciduous Forest (Mature)*#	Forest that is 75% or more of deciduous trees; <30% canopy removal
Deciduous Forest (Intermediate Removal)*#	Forest that is 75% or more of deciduous trees; 30%-60% canopy
Deciduous Forest (Overstory Removal)*#	Forest that is 75% or more of deciduous trees; >60% canopy
Conifer Forest (Mature)*#	Forest that is 75% or more of coniferous trees; <30% canopy removal
Conifer Forest (Intermediate Removal)*#	Forest that is 75% or more of coniferous trees; 30%-60% canopy
Conifer Forest (Overstory Removal)*#	Forest that is 75% or more of coniferous trees; <60% canopy removal
Mixed Forest (Mature)*#	Forest that is neither deciduous nor coniferous trees that are >75% of total tree cover; <30% canopy removal
Mixed Forest (Intermediate Removal)*#	Forest that is neither deciduous nor coniferous trees that are >75% of total tree cover; 30%-60% canopy
Mixed Forest (Overstory Removal)*#	Forest that is neither deciduous nor coniferous trees that are >75% of total tree cover; <60% canopy removal
Grassland*	Landcover dominated by upland grasses or forbs
Wetland+	Any land which is annually subject to periodic or continual inundation of water which are either (a) one acre or more in size or (b) located adjacent to a free-flowing body of water

Data Sources:*Yang et al. 2018, #Kramer et al. 2022, +Adirondack Park Agency 2004

To assess the annual variability in winter conditions we calculated an index of winter severity for white-tailed deer (Verme 1968) using snow depth (NOHRSC 2004) and daily mean temperatures (Daly et al. 1994). Winter severity was the sum of the total number of days (November 1–March 31) where snow depths were greater than 38 centimeters and total number of days where the average daily temperature was below -17.8°C. Winter severity for winter 2017-2018 was extended to April due to an abnormal late season extreme weather event, resulting in a winter severity measured from 1 November 2017–April 30 2018 (Table 2). We acknowledge the potential that extending the sampling window only for 2018 could induce a bias, however by not incorporating the abnormal late season blizzard into our calculations, we would have failed to accurately quantify the overall influence of climatic variation on moose selection.

We used logistic regression models to determine the effects of land cover, winter severity and forest management on moose habitat selection. We quantified selection for each season-year combination in separate analyses, and defined winter as December-March, and summer as June-August. Accordingly, we quantified habitat selection for winter and summer separately for December 2015-August 2019, resulting in 8 total analyses (4 years x 2 seasons). Within each analysis, we compared four candidate models, corresponding to our four competing hypotheses: (1) no forest management covariates (NULL); (2) distance to nearest forest management (intermediate or overstory removal) regardless of forest type (MAN); (3) distance to intermediate removal and overstory removal regardless of forest type (TYPE); and (4) distance to each possible forest management-forest type combination (TYPE x FOR). All four models included covariates for distance to natural cover types including grassland, wetland, and mature deciduous, conifer, and mixed forest. All four models also included random intercepts and slopes by moose ID to account for individual variation in selection (Muff et al. 2020). All models were fit using the R package glmmTMB (Brooks et al. 2017). Within each analysis we ranked the four candidate models using AIC (Burnham and Anderson 2002) and calculated the area under the receiver operating characteristic curve (AUC) on the full dataset as a measure of each model's predictive power (Cumming 2000).

RESULTS

We estimated home ranges for 26 moose during December 2015–August 2019 using an average of 7,532 GPS locations per moose (SD = 3,411; range = 1,824–5,986). Annual moose home ranges averaged 21.6 km^2

Table 2. The average winter severity index (WSI) and standard deviation (SD) for northern New York, USA from November 1–March 31, from four consecutive winters (2015-2019). The value is the total number of days where snow depths were greater than 38 centimeters and total number of days where the average daily temperature was below -17.8°C. Winter severity for winter 2017-2018 measured from 1 November 2017–April 30, 2018.

Year	WSI	SD
2015-2016	13	4
2016-2017	39	29
2017-2018	61	35
2018-2019	69	39

(SD = 11.0; range = 3.8–54.4). There was considerable spatial overlap between individual study animals, with only two collared individuals not exhibiting any range overlap with another collared moose. Composition of land cover within home ranges varied among moose (Figure 2). Among natural (i.e., non-management) land cover types, deciduous forest was the most common land cover type in home ranges, followed by wetland and coniferous forest. Managed/harvested forests made up a smaller portion of moose home ranges, with intermediate removal of deciduous forest being the most common.

We fit four winter season resource selection models (yearly 2016-2019; 21, 22, 10 and 6 moose, respectively) using an average

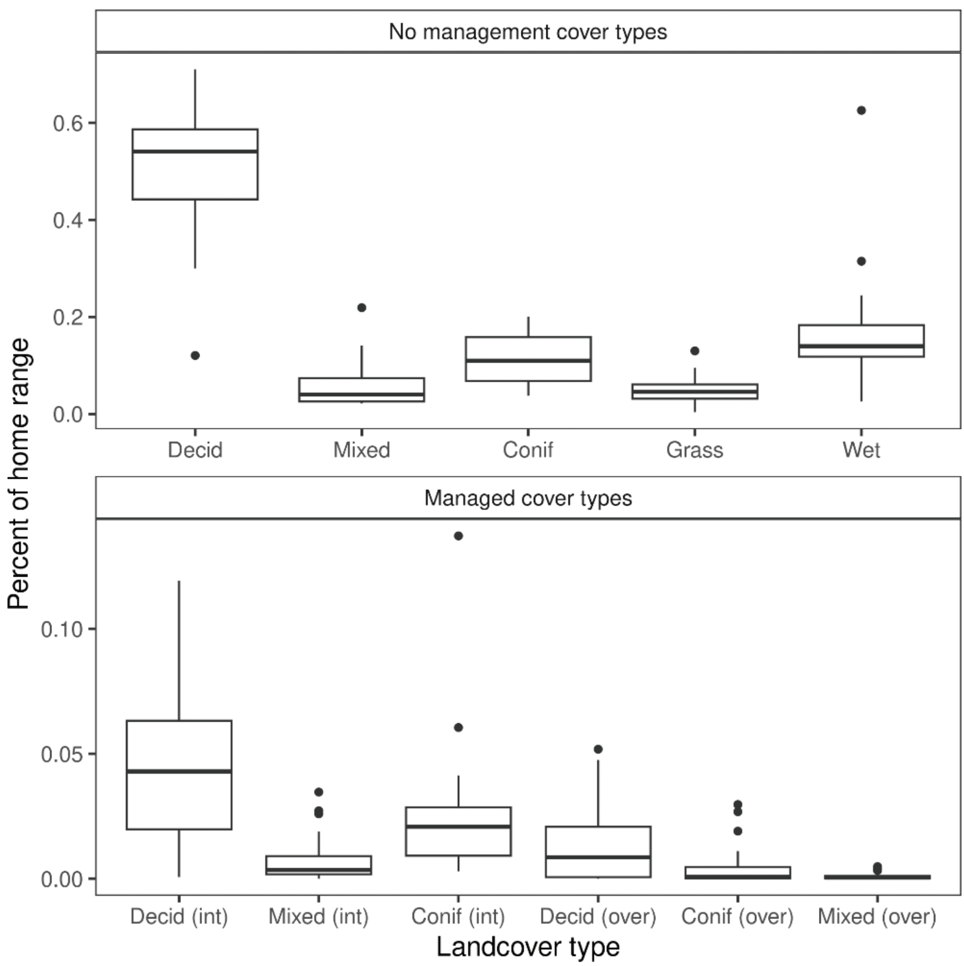

Fig. 2. Land cover composition in moose all-season 95% home ranges in Adirondack Park, NY, 2016-2019. The boxplots represent the interquartile interval, where 50% of the data is found within the bounds of the box, with a line representing the mean value. The vertical lines represent the upper and lower quartiles (>75% and <25% of the data, respectively), and dots represent outliers. The top panel shows non-management cover types (Decid = deciduous forest, Mixed = deciduous/conifer forest, Conif = conifer forest, Grass = open grass, Wet = wetlands) and the bottom panel shows forest cover with recent management (int = intermediate removal, over = overstory removal).

of 17,629 locations per moose (range 8,231 – 25,186). We fit three summer resource selection models (yearly 2016-2018; 19, 14 and 6 moose, respectively) using an average of 13,191 locations per moose (range 5,467 – 20,389). We did not fit a model in summer 2019 due to a small sample size (768 locations; 1 moose).

Across all year and season combinations, the top-ranked and clearly superior model was model TYPE x FOR which included effects of distances to individual forest type – management combinations (Tables 3,4). Top-ranked models had AUC values ranging from 0.83 to 0.97. Because these AUC values were calculated from the full dataset without subsetting, they likely overestimate the accuracy of the model predicting for new data, but still indicate adequate predictive power.

Moose selection for early successional forest habitat varied by season, year, and forest type. In winter, there was little evidence that moose selected for or against habitat close to forest stands with recent intermediate removal within their home ranges. The exception was 2019, when moose selected for habitat further from this management type in deciduous forest (Figure 3). In 2016 (a relatively mild winter), selection for overstory removal varied by forest type, with selection for areas close to overstory removal in mixed forest and selection against overstory removal in deciduous forest. In years 2018 and 2019 that were characterized by above average

Table 3. Candidate resource selection models for moose in winter (December–March) 2016-2019 in Adirondack Park, NY. Candidate models were NULL (no forest management parameters); MAN (distance to nearest forest management); TYPE (distance to nearest intermediate removal and nearest overstory removal); and TYPExFOR (distance to nearest intermediate and overstory removal by forest type). All models also included parameters for distance to nearest natural habitat types. K is the number of free parameters in the model, AIC represents the mathematical evaluation of model fit, ΔAIC is the difference between the best fit candidate model from those provided and AUC is the measure of how different a model prediction is from random chance.

Year	Model	K	AIC	ΔAIC	AUC
2016	TYPExFOR	24	113,063.8	0.0	0.88
	TYPE	16	126,463.4	13,399.5	0.83
	MAN	14	135,275.9	2,2212.0	0.79
	NULL	12	138,250.5	25,186.6	0.77
2017	TYPExFOR	24	92,642.4	0.0	0.92
	TYPE	16	114,179.5	21,537.0	0.87
	MAN	14	123,819.7	31,177.3	0.82
	NULL	12	130,198.6	37,556.2	0.80
2018	TYPExFOR	24	50,094.6	0.0	0.96
	TYPE	16	60,425.5	10,331.0	0.93
	MAN	14	64,744.9	14,650.3	0.91
	NULL	12	67,719.6	17,625.1	0.90
2019	TYPExFOR	24	34,672.0	0.0	0.97
	TYPE	16	37,896.5	3,224.5	0.96
	MAN	14	41,042.8	6,370.8	0.94
	NULL	12	41,358.2	6,686.2	0.94

Table 4. Comparison of candidate resource selection models for moose in summer (June – August) 2016-2018 in Adirondack Park, NY. Candidate models included NULL (no forest management parameters); MAN (distance to nearest forest management); TYPE (distance to nearest intermediate removal and nearest overstory removal); and TYPExFOR (distance to nearest intermediate and overstory removal by forest type). All models also included parameters for distance to nearest natural habitat types. K is the number of free parameters in the model, AIC represents the mathematical evaluation of model fit, ΔAIC is the difference between the best fit candidate model from those provided and AUC is the measure of how different a model prediction is from random chance.

Year	Model	K	AIC	ΔAIC	AUC
2016	TYPExFOR	24	111,194.4	0.0	0.83
	TYPE	16	117,819.3	6,624.8	0.78
	MAN	14	121,096.1	9,901.7	0.76
	NULL	12	123,677.3	12,482.9	0.74
2017	TYPExFOR	24	77,424.0	0.0	0.87
	TYPE	16	81,253.8	3,829.8	0.84
	MAN	14	84,990.9	7,566.9	0.81
	NULL	12	87,005.3	9,581.3	0.79
2018	TYPExFOR	24	32,915.8	0.0	0.94
	TYPE	16	34,690.9	1,775.1	0.92
	MAN	14	36158.4	3,242.6	0.91
	NULL	12	36621.8	3,706.1	0.90

winter severity, there was a general pattern of selection for habitat close to overstory removal regimes in all forest types. However, the associated parameter estimates were accompanied with large uncertainty, possibly due to high variability in selection among individual moose and smaller sample sizes in these years.

During summers, moose had a mixed response to intermediate removal harvest (Figure 4). In 2016, there was no strong selection within home ranges for or against intermediate removal. In 2017, moose selected habitat further from intermediate removal in coniferous forest, and in 2018, moose selected for habitat further from all three forest types but particularly deciduous forest. In 2017 and 2018, moose selected for habitat close to overstory removal harvests in most forest types.

As with managed forest habitat, there was no consistent yearly or seasonal pattern in moose selection for wetland habitat (Figure 5). In 2016-2018, moose did not select for or against areas close to wetland habitat in either season. In 2019, moose selected for habitat closer to wetlands during the winter.

DISCUSSION

The average home range size (21.6 km^2) for New York moose was similar to ranges observed in regional populations in Maine (Leptich and Gilbert 1989). This suggests that the population has not surpassed resource availability and will continue to persist in localized, higher-density sub-populations concentrated around suitable habitat. Home ranges predominately contained deciduous forest and included forested stands that had received recent timber harvest (i.e., shelterwood harvests, overstory removal) similar to home range compositions in neighboring states (Wattles and DeStefano 2013, Andreozzi et al. 2016). Observed

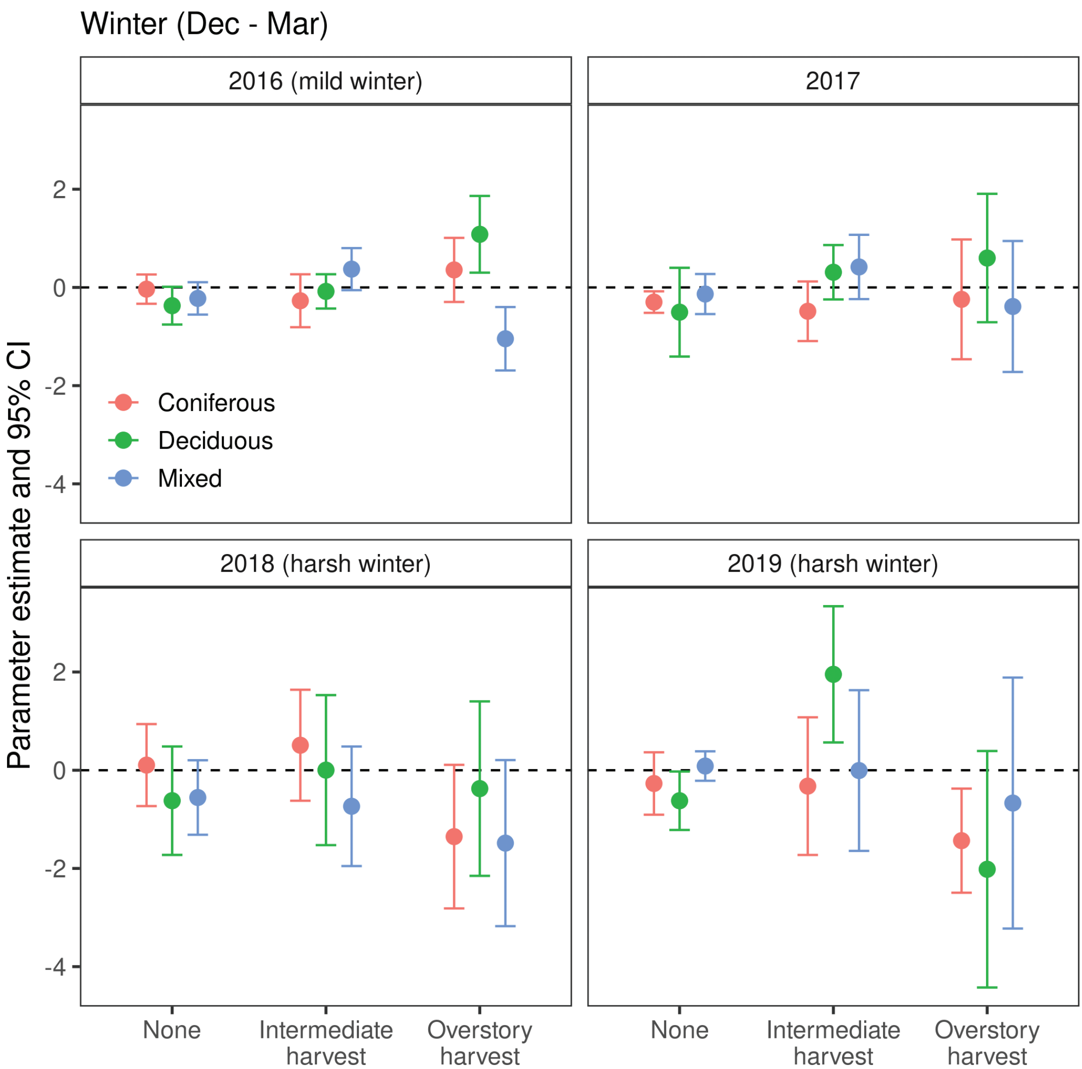

Fig. 3. Moose selection for combinations of forest and management type from the top-ranked moose resource selection models in winter (December – March) of 2016 – 2019 in Adirondack Park in New York, USA. Covariates are the minimum distance to the nearest forest/management type; therefore, positive logit-scale coefficient values indicate moose were selecting *against* (i.e., for areas further away from) that forest/management type, and vice-versa.

non-migratory behavior of New York moose, coupled with high spatial overlap among individuals and previous work on forage availability and nutrition (Peterson et al. 2020, Kramer et al. 2022, Peterson et al. 2022) suggest that moose in the Park are at or below the landscape-level population capacity.

Moose resource selection within home ranges varied during years with severe winter conditions, with moose selecting for overstory removals during the harsh winters in 2018-2019. Timber management treatments in the Park are often conducted at small scales ($\overline{x} = 4.2$ hectares) due to the limits on timber extraction on public land and the matrix of land ownership in the Park. Moose may use more heterogenous patches of managed and non-managed forest during

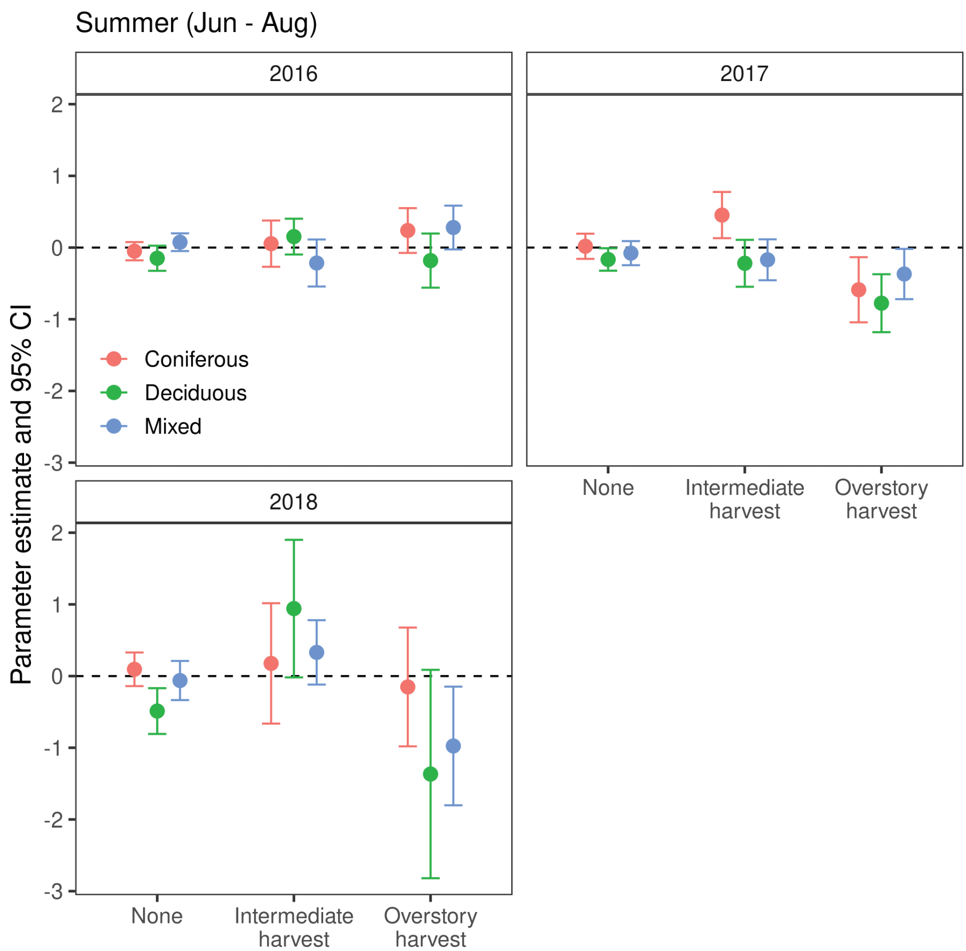

Fig. 4. Moose selection for combinations of forest and management type from the top-ranked moose resource selection models in summer (June – August) of 2016 – 2018 in Adirondack Park in New York, USA. Covariates are the minimum distance to the nearest forest/management type; therefore, positive logit-scale coefficient values indicate moose were selecting *against* (i.e., for areas further away from) that forest/management type, and vice-versa. There was no 2019 model due to a small sample size.

severe weather to reduce energetic demands during periods of thermal stress (summer) or impeded movements (winters) (Poole and Stuart-Smith 2005, Andreozzi et al. 2016). However, our measure for winter severity may not be ideal to measure the impacts of winter stressors on moose given that it was initially derived for white-tailed deer. It is possible to modify the index to meet moose thresholds more aptly, but it is unlikely that anywhere in New York would meet those conditions, given our location on the southern extent of moose range.

There was no indication that moose selected wetlands within their home ranges during summer, which was unexpected given that moose typically derive a portion of the summer diet from aquatic plants and that wetlands can facilitate thermoregulation during warm summer conditions, especially along the southern range extent for moose (Broders et al. 2012, Morris 2014, Teitelbaum

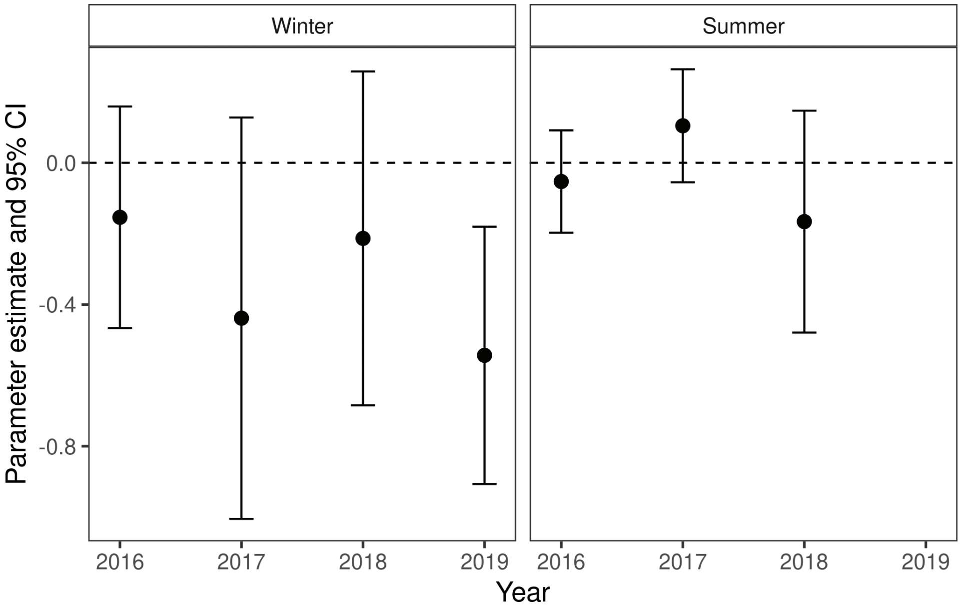

Fig. 5. Moose selection for wetland habitat from the top-ranked moose resource selection models in winter (December – March) and summer (June – August) of 2016 – 2018 in Adirondack Park in New York, USA. Covariates are the minimum distance to the nearest wetland; therefore, positive logit-scale coefficient values indicate moose were selecting *against* (i.e., for areas further away from) wetlands, and vice-versa. There was no summer 2019 model due to a small sample size.

et al. 2021, Tischler et al. 2022). It is likely that wetlands are not a limiting resource for moose in the region. Regional beaver activity is high (Zevin 2022), and the lack of selection may indicate that moose are randomly encountering wetlands at a rate that may be sufficient to meet their foraging and thermoregulatory needs.

Our findings align with the concerns identified by private landowners and timber managers that the current distribution and density of the New York moose population may be negatively impacting commercial timber harvest through disproportionate selection of managed forests (Connelly et al. 2020). When paired with previous research on browse selection and population distribution (see Peterson et al. 2020, Hinton et al. 2022a, Peterson et al. 2022), our findings indicate that the status of the current moose population is influenced by the availability of commercial working forests. Monitored moose in our study showed little selection for or against unmanaged forests, which largely occur on public lands and are the dominant habitat on the landscape, accounting for half of all moose home ranges. Alternatively, managed forests (i.e., private commercial forest) were selected for during summer and winter. The result is that private owners of working forests generate the habitat which supports the majority of the New York moose population, but they thus experience the negative effects of moose browsing (ex, damage to regenerating timber stands).

It is important to acknowledge the potential for overlap in the ecological niche and population distribution between moose and white-tailed deer (Whitlaw and Lankester 1994, Post and Stenseth 2002). Early successional habitat, which can be found in timber stands 5-10 years post-management, are important sources of forage for deer and often constitute a large portion their home range when available (Aycrigg and Porter 1997, Beier and McCullough 1990, Quinn et al. 2013, Lesser et al. 2019). The Adirondack landscape is predominately mature forest, with smaller pockets of commercial forest, creating a landscape where deer may be more likely to congregate in those commercial forests to access preferred forage. Although deer densities in northern New York are lower than other parts of the state, deer and moose co-occurrence is likely occurring (Whitlaw and Lankester 1994, NYSDEC 2021, Hinton et al. 2022b).

The amount of spatial overlap between moose and deer has likely increased with time in New York because milder winters have resulted in deer population expansion northward while moose population distribution has expanded (Teitelbaum et al. 2021; Hinton et al. 2022 a,b). Historically, severe winters influenced deer populations in upstate New York through winter die-off events or by encouraging deer to congregate in pockets of dense thermal cover (i.e., deer yards) at low elevations (Tierson et al. 1985, Hurst and Porter 2008). The increasing frequency of mild winters in the region due to climate change has allowed deer to persist year-round in areas that were typically seasonally inhospitable, while also lessening the frequency of mortality events. The result is a potential increase in year-round interactions between white-tailed deer and moose and an increased opportunity for parasitic transmissions via feces and gastropod vectors (Lankester 2002, Vanderwaal et al. 2015, Vannatta and Moen 2016, Hinton et al. 2022a). There are already indications that the frequency of parasitic infection in moose has increased over the past 10-15 years (Kevin Hynes, NYSDEC; personal comm.) in New York.

The shared preference for managed forest habitat creates a conundrum for wildlife managers responsible for managing moose and deer populations. Wildlife managers in New York have limited tools available to manage moose. Currently, the state has no regulated hunting of moose; predation of moose and deer is limited; and forest management is restricted to private lands, with public forests protected from commercial timber extraction. Managers have more tools for managing the white-tailed deer population through regulated hunting seasons and depredation permits. However, portions of the Park are remote and the region has some of lowest hunter and deer harvest densities in the state (NYSDEC 2021). There is potential for agency managers to work with commercial forest owners to develop long-term management plans that focus on the distribution and timing of forest management that would potentially lessen the impacts of moose browsing but also to mitigate the increasing overlap between white-tailed deer and moose.

In light of apparent moose population declines in various New England states (i.e., Maine, New Hampshire and Vermont), future moose research in New York should elucidate patterns of overlap between moose and white-tailed deer, examine landscape parasite prevalence, seek to identify potential management actions should the moose population in the Park begin to decline, and work with commercial foresters to develop long-term management plans for working forests.

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