TIMBER-MANAGEMENT AND NATURAL-DISTURBANCE EFFECTS ON MOOSE HABITAT: LANDSCAPE EVALUATION

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ABSTRACT / We used 16 years of survey data for a moose population, and 3 Landsat satellite scenes, spanning 19 years, to evaluate the hypotheses that Ontario's Moose Habitat Guidelines for timber harvest: (1) mitigate the effects of unmodified clearcuts on moose populations, and (2) create enhanced habitat with greater interspersion of forage with cover and higher habitat suitability indices than areas dominated by unmodified clearcuts. The 5 study landscapes compared were 16,000 - 91,000 ha, and included landscape disturbance from timber-management and wildfire-burn, and landscapes with and without hunter access. Moose density differed among landscapes, but while neither main effects of hunter access ($P = 0.083$), nor landscape disturbance ($P = 0.31$) were significant, their interactions were ($P = 0.003$), with density increasing if disturbance occurs without hunter access. The habitat suitability index in the wildfire burn was similar (0.80) to both the modified and unmodified clearcut (0.85 and 0.83), and population rate of increase was positive in both the burn ($B = 0.153$, $P < 0.0001$) and the unmodified clearcut ($B = 0.127$, $P < 0.0001$). The population did not increase in the modified clearcut ($B = -0.016$, $P = 0.9907$) because hunter access increased as a consequence of high road density.

Key words: adaptive management, *Alces alces*, boreal forest, GIS, HSI, habitat ecology, hunting, landscape ecology, moose, natural disturbance, policy evaluation.

Scientific evaluation of resource management planning and practices is an essential component of adaptive resource management (Macnab 1983, Walters and Holling 1990, Sinclair 1991). Forestry practices affect wildlife habitat use, but interpretations of habitat use can vary with spatial scale (Holling 1992, Forbes and Theberge 1993). Thus, it is important to ensure that the scale at which habitat use is measured matches the spatial scale at which the habitat is altered. In the boreal forest, where plants and animals have adapted to disturbance caused by wildfire and insect outbreak (Hunter 1993), studies using broad spatial scales may be particularly important for contrasting the effects of forestry practices with natural systems.

Our hypothesis is that application of Ontario's Moose Habitat Guidelines for timber harvest results in a landscape structure that mitigates the effects of unmodified clearcuts on moose populations. Clearcuts are modified by restricting maximum cutover-size to ensure that forage provided by early successional plant communities is close to the protective cover provided by semi-mature or mature conifer stands. To test the hypothesis, we: (1) determine if landscape disturbance, hunter access, or the interaction of these factors explains differences in moose density among landscapes, and (2) if landscapes logged by modified clearcutting have greater interspersion of forage with cover and higher habitat suitability indices (HSI) than areas logged with unmodified clearcutting. We use a landscape disturbed by wildfire to contrast landscape structure, and to compare trends in population density.

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STUDY AREA

The study area was about 75 km northeast of Fort Frances, Ontario, Canada (Fig. 1), centered at about 92°45'W, 49°15'N. The area, on the northern border of the Quetico Great Lakes St. Lawrence Forest Region (Rowe 1972), was characterized by stands of eastern white and red pine (*Pinus strobus, P resinosa*), pure and mixed stands of jack pine (*P banksiana*), trembling and large tooth aspen (*Populus tremuloides*, *P grandidentata*), white birch (*Betula papyrifera*), balsam fir (*Abies balsamea*), and white and black spruce (*Picea glauca, P. martana*). Soils were shallow and coarse in texture as a result of heavy glaciation (Rowe 1972).

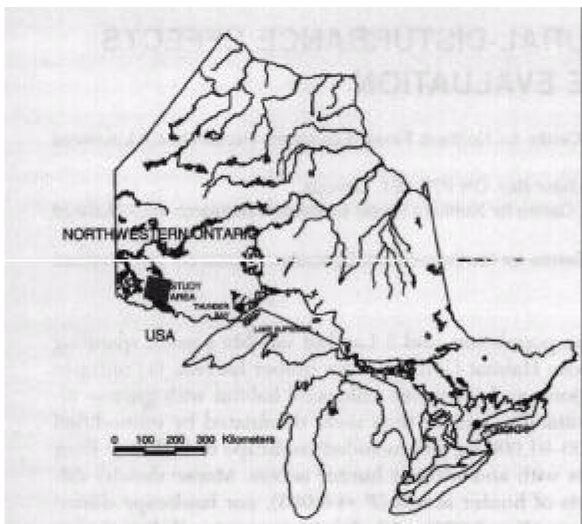

Fig. 1. Location of study area in Northwest Ontario, 1992.

Five landscapes totaling ca. 240,000 ha were delineated in the study area: (1) a modified clearcut (MCC), following the Moose Habitat Guidelines; (2) an unmodified, progressive clearcut landscape (PCC); (3) a wildfire burn (WFB); (4) a mature, uncut wilderness landscape, without road access (UCW); (5) a mature, uncut, road-accessible landscape (UCR) (Fig. 2, Table 1). The burn occurred on the WFB in 1981, and timber harvest began in MCC in 1981 and in PCC in 1978. Temporal rate of disturbance differed among landscapes, and by 1985 ca. 30% of logging was completed for MCC, 60% completed for PCC, and 80% of total disturbance completed in WFB (Table 2). The additional 20% of disturbance in WFB was due to lowdown and logging after 1985.

Hunter access was restricted in parts or all of PCC between 1978 and 1989. In 1978 ca.11,000 ha (25%) was closed to hunting for worker safety on the western side of PCC. Between 1979 and 1986, all logged areas in PCC were closed to hunting to study effects of hunting closure after clearcutting, resulting in ca. 45,000 ha (100%) closed to hunting in 1986. Between 1987 and 1989, hunting closures were progressively removed from the northern side of PCC, with ca. 22,000 ha in 1987 (50%) and 10,000 ha (25%) in 1989 closed to hunting on the southern side. There were no hunting closures in PCC from 1990 to 1992.

Road density affects hunter access and differs among landscape treatments, with high densities in MCC, PCC, and UCR, and lower densities in WFB and UCW (Fig. 2, Table 1). Heavy blowdown after the burn also severely restricted hunter access in WFB.

Post-harvest silvicultural activities included herbicide application in both MCC and PCC, and aerial seeding and subsequent thinning of jack pine in PCC. Between 1987 and 1991, 9.0 and 9.3% of logged MCC and PCC were treated with herbicide.

METHODS

Landscape disturbance (disturbed, undisturbed) and hunter access (restricted, unrestricted) formed the 2 classification levels of landscape treatment for the study (Table 1). Disturbance occurred in MCC, PCC, and WFB, and hunter access was restricted in PCC (by hunting closure) and UCW and WFB (by absence of roads and woody debris). The 5 landscapes were delineated on the 1991 landcover map (Fig. 2). Convex polygons were digitized to delineate the extent of each landscape unit, with the lines extending 50-200 m past the disturbance boundary for MCC, PCC, and WFB.

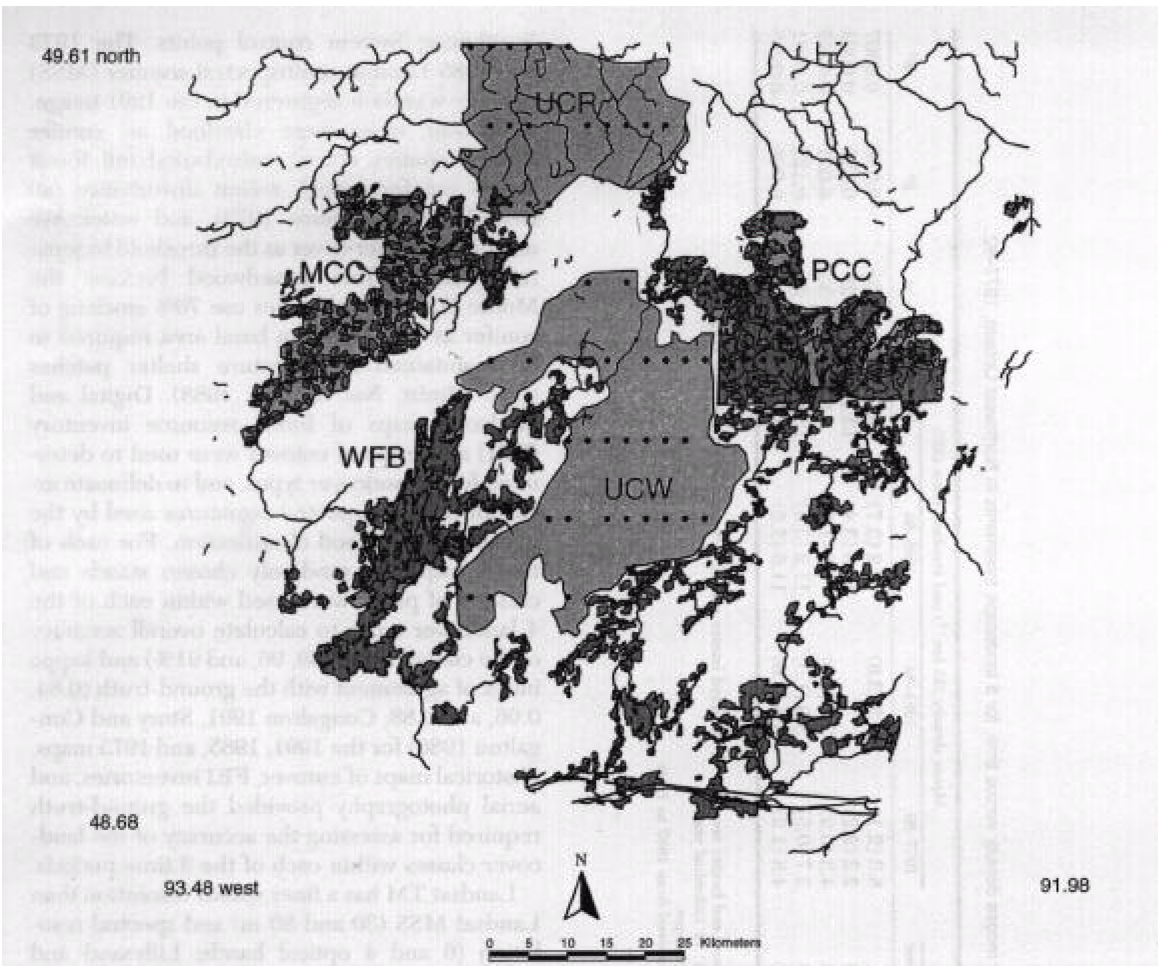

Fig. 2. Roads, center points of moose survey plots, and landscape disturbance (dark shading), as determined by satellite image analysis. Outlines and light shading indicate identity and boundaries of the 5 landscape treatment areas: modified clearcut (MCC). wildfire burn (WFB), uncut wilderness (UCW), progressive clearcut (PCC), and uncut road-accessible landscape (UCR).

Moose Population Analysis

The center point of each survey plot (2.5 X 10 km) from early-winter aerial moose surveys (1977-92) was registered to landscape maps. Sightability was calculated as 79% based on analysis of resurvey data, where a random selection of plots was reflown at a higher intensity immediately after the standard survey (Bisset and Rempel 1991). Surveys conducted between 1977 and 1991 were standard OMNR district stratified-random surveys (Bisset 1991), with sampling emphasis proportional to expected moose density, and ca. 20% of the land area surveyed. Within strata, sample design is sampling with replacement, where every year plots are randomly selected, and each plot has an equal chance of being selected. The 1992 survey, designed specifically for this study, used a systematic placement of plots, with every other plot location sampled and 50% of the land area surveyed.

We tested the hypothesis that neither landscape disturbance, hunter access, or the interaction of these factors affected moose densities using analysis of variance with data pooled for 1985-92. Most of the landscape disturbance had occurred by 1985. Two-way ANOVA (SAS Inst. Inc. 1989) was conducted with hunter access (0,1) and presence of landscape disturbance (0,1) as the classification variables (Table 1), and $\log_e$ (N + 1) as the response variable, where N is the number of moose seen per 25km^2 plot. Curvilinear trends in the normal distribution plot disappeared, and estimates of normality (Wilke-Shapiro's W) increased from 0.85 to 0.92 following $\log_e$ transformation. Consequently, statistical inference was based on the F-distribution and the log-linear transformed data. Regression residuals from analysis of trends in year-to-year population density are not independent, so we tested the hypothesis that moose density did not increase over time with autoregression analysis. Density for each year over the period 1977-92 was transformed to $\log_e$ (N + 1). Regression coefficients and probability $B = 0$ were estimated with the exact maximum-likelihood method (SPSS Inc. 1993).

Table 1. Experimental Design, moose density, and regression coefficients for moose density versus time, for 5 landscape treatments in Northwest Ontario, 1977-92.

				Design Martix		Moose Density (25km -2) and standard error (SE)						
Landscape Acronym	Area (ha)	Road Density [a]	Landscape Treat no	Disturb	Harvest	1977-80	1981-84	1985-88	1989-92	n	B[c]	P[d]
MCC	38,601	8.1	1	1	1	8.3 (2.1)	10.5 (3.7)	8.2 (3.7)	5.6 (1.0)	20	-0.016	0.9907
WFB	15,514	0.3	2	1	0	2.2 (2.2)		12.3 (3.4)	13.2 (2.4)	28	0.153	0.0001
UCM	91,255	>0.01	3	0	0	4.3 (0.9)	3.7 (0.9)	7.8 (1.5)	8.8 (1.1)	53	0.061	0.0042
PCC	46,913	6.3	4	1	0	2.7 (0.5)[e]	5.9 (1.5)[e]	11.2 (3.8)[e]	12.4 (1.7)	55	0.127	0.0001
UCR	46,844	4.9	5	0	1	4.8 (1.0)	11.6 2.2	11.6(1.8)	8.9 (1.8)	30	0.068	0.0295

[a] Expressed as m · ha^{-1}

[b] design matrix specifies 2 levels of disturbance (0=no disturbance, 1=disturbance) and hunter access (0=restricted hunter access, 1=unrestricted hunter access).

[c] autoregression coefficients, based on $\log_e$ transformed densities (no. seen/plot + 1) vs. individual year

[d] Approximate probabilities for B = 0, estimated with exact maximum-likelihood technique

[e] Hunting was closed for these treatments, with hunting restrictions progressively removed from 1987 to 1989

Landscape Disturbance and HSI Analysis

Landcover maps were derived from Landsat satellite imagery for 1973, 1985, and 1991. The 1991 thematic mapper (TM) imagery was georegistered with 1: 50,000 mapsheets and Global Positioning System control points. The 1973 and 1985 Landsat multispectral scanner (MSS) imagery was then registered to the 1991 image. Landcover types were classified as: conifer (>=70% conifer cover), mixedwood (all forest <= 70% conifer cover), recent disturbance (all new disturbance since 1973), and water. We used 70% conifer cover as the threshold to separate conifer from mixedwood because the Moose Habitat Guidelines use 70% stocking of conifer as the minimum basal area required to be maintained in immature shelter patches (Ont. Minist. Nat. Resour. 1988). Digital and hardcopy maps of forest resource inventory (FRI) and maps of cutover were used to determine forest landcover types, and to delineate areas for deriving spectral signatures used by the maximum likelihood classification. For each of the 3 maps, 40 randomly chosen stands and clusters of pixels were used within each of the 4 landcover types to calculate the overall accuracy of the classification (89, 96, and 91%) and kappa index of agreement with the ground-truth (0.84, 0.96, and 0.88; Congalton 1991, Story and Congalton 1986) for the 1991, 1985, and 1973 maps. Historical maps of cutover, FRI inventories, and aerial photography provided the ground-truth required for assessing the accuracy of the landcover classes within each of the 3 time periods.

Landsat TM has a finer spatial resolution than Landsat MSS (30 and 80 m) and spectral resolution (6 and 4 optical bands; Lillesand and Kiefer 1987). Each map was re-sampled to 57 m resolution within the SPANS GIS (Intera Tydac 1993). Wetlands and open jack pine/rock caused misclassification problems so they were determined from the 1991 TM image and fixed in the 1973 and 1985 imagery. Roads were manually digitized on-screen with the 1991 TM imagery as a geo-referenced background.

Changes in land area of disturbance and vegetation landcover classes were calculated for the 3 landscape treatments with forest disturbance (MCC, PCC, and WFB) during the 1973-85 and 1985-92 time intervals. Disturbance area, patch size, forage/cover edge density, patch interspersion, core disturbance area, interpatch distance, and adjacency of disturbance to mature forest were determined for the 1985 and 1991 landcover maps with FRAGSTATS (McGarigal and Marks 1993). Gap-phase disturbance of finer grain caused by succession was not included in the experimental design, but was measured by calculating change in overstory composition from pure conifer to mixedwood with the landcover maps from satellite data.

Table 2. Landscape metrics derived from satellite imagery for modified clearcut (MCC), wildfire burn (WFB) and progressive clearcut (PCC), for 1985 and 1991, in Northwest Ontario.

	1985			1991		
Landscape feature	MCC	WFB	PCC	MCC	WFB	PCC
Disturbance area (ha)	4,600.0	7,630.0	16,022.0	13,870.0	9,602.0	26,309
Mean disturbance patch area (ha)	73.0	191.0	280.0	121.0	223.0	1,184
Mean core area of disturbance patch (ha) [a]	39.0	113.0	174.0	63.0	124.0	806
Density of disturbance patches (no./100 ha)	1.4	0.5	0.4	0.8	0.5	.01
Mean inter-patch (disturbance) distance[b] (m)	528.0	188.0	249.0	178.0	79.0	133
Edge density (m/ha)	6.4	26.7	16.8	23.6	36.5	23.0
Interspersion and juxtaposition index[c] (%)				18.5	48.0	29.3
Available forage[d] (ha)	1,712.0	3,192.0	5,939.0	6,665.0	4,251.0	8,711
% disturbance as available forage	37.2	41.8	37.1	48.1	44.3	33.1

[a] Core disturbance patch area is >=100 m from closest edge.
[b] Nearest neighbor distance of like patches, based on patch edge-to-edge distance.
[c] Interspersion and juxtaposition index is a percentage relative to the highest possible interspersion, given the total no. of patch types.
Index calculated only for the 1991 landscape map which had the greater no. of landcover classes.
[d] Area of recent disturbance (ha) within 100 m of mature, uncut, forest cover

The vegetation maps from the 1991 satellite imagery and the Lake Superior Model II HSI for moose (Allen et al. 1987) were used to calculate HSI values for preferred forage, forage and cover, winter cover, and combined HSI for each landscape treatment unit. The landcover maps were first reclassified into 3 moose habitat classes, where (1) recent disturbance is preferred forage, (2) mixed wood is forage and cover, and (3) conifer is late winter cover. The model describes (1) preferred forage as % area in shrub or forested cover types <20 years old, (2) forage and cover as % area in upland deciduous or mixed forest >=20 years old, and (3) late winter cover as % area in spruce/fir forest >=20 years old. Percentage area of each habitat class within a landscape treatment was calculated and assigned an HSI value (Allen et al. 1987). The combined HSI value is a geometric mean of the separate HSI components, and provides an overall estimate of habitat suitability (Allen et al. 1987):

$$HSI_c = (HSI_{pf} \cdot HSI_{wc} \cdot HSI_{fc})^{1/3},$$

where *c* is combined, *pf* is preferred forage, *wc* is winter cover, and *fc* is forage and cover.
An alternative HSI model can be formulated by assuming that moose require late-winter dense conifer cover, but that forage derived from clearcuts, burns, mixedwood forest, or decadent forest are of equal value, i.e.:

$$HSI_{mc} = (HSI_{wc} (\text{max. } [HSI_{pf}, HSI_{fc}]))^{1/2}$$

where *mc* is the modified combined HSI. The factor HSI_{mc} considers the combination of late winter cover and food, where food is the maximum of either the preferred forage (HSI_{pf}) or mixed food and cover (HSI_{fc}) indices.

RESULTS

Moose Population Abundance

During 1977-92, moose density increased in the 2 landscapes that had both large-scale forest disturbance (logging or burn) and reduced hunter access (B = 0.153, P < 0.001, and B = 0.127, P < 0.0001, for WFB and PCC; Table 1). Moose density did not increase in the MCC landscape where both disturbance and hunter access occurred (B = - 0.016, P = 0.991). Landscapes with no logging and little disturbance provide a measure of population change that is independent of timber harvest, and in both UCW and UCR moose density increased slightly (B = 0.061, P = 0.004, and B = 0.068, P = 0.030) for UCW and UCR. Gap-phase disturbance and vegetation succession occurred in both of these landscapes, as percentage mixedwood increased from 27 to 46% in UCW, and decreased from 70 to 60% in UCR between 1973-91.
Neither hunter access (P = 0.083), nor landscape disturbance (P = 0.311) accounted for the observed differences in moose density for the period 1985-92, but the interaction of hunter access with disturbance was significant (P = 0.003; 1, 114 df; F = 9.05), and may in part explain changes in moose density among the 5 landscapes. Without landscape disturbance, regardless of the level of hunter access, moose densities remained relatively similar, although the degree of similarity varies over time (Table 1). Analysis of the pooled 1985-92 data suggests that if disturbance occurs concurrently with hunter access, then the least-square mean of moose density decreases, but if disturbance occurs without hunter access, then moose density increases. Even though hunting pressure in the PCC was apparently high immediately after its reopening in l988 (W. May, Ont Minist. Nat. Resour., pers. commun.), moose numbers did not decrease.

Landscape Disturbance and HSI Analysis

Edge density was higher in WFB (36.5 m ha^{-1}) than in both MCC (23.6 m ha-l) and PCC (23.0 m ha-l), which were similar in edge density (Table 2). Mature forest/disturbed forest patch interspersion was highest in WFB (48%), and was higher in PCC (29.3%) than MCC (18.5%). Although PCC had one large clearcut patch, there were numerous smaller patches of mature forest interspersed throughout the cut, resulting in PCC having higher interspersion than MCC. Shoreline reserves around lakes and rivers also contributed to the interspersion in PCC. Percentage of total disturbance area located within 100 m of undisturbed forest in WFB and MCC was similar (44.3 and 48.1), and higher than in PCC (33.1). This was an expected consequence of the small and dispersed blockcuts in MCC positioning a greater proportion of the clearcut adjacent to mature forest. Landscapes MCC, WFB, and PCC had similar HSI_{mc} values (0.85, 0.80, and 0.83; Table 3), whereas UCW and UCR had values of about half those of the disturbed landscapes (0.45 and 0.54). Both forage and winter cover HSI values were lowest, and mixed food and cover highest in the undisturbed landscapes.

Table 3. Percent cover of moose habitat classes, as defined in Lake Superior moose HSI (Allen et al. 1987), and derived from satellite imagery based landcover maps, for northwest Ontario, 1991. HSI values derived from relations described in Model II of Lake Superior moose HSI (Allen et al. 1987), where 1.0 represents highest value possible.

		% cover			HIS				
Landscape treatment unit		Forage	Winter cover	Food and cover	Forage	Winter cover	Food and cover	Combined	Combined modified
MCC	1	40.72	38.90	7.00	1.00	0.73	0.20	0.527	0.85
WFB	2	68.20	13.40	3.14	0.64	1.00	0.09	0.380	0.80
UCW	3	0.03	59.70	15.49	0.01	0.48	0.43	0.128	0.45
PCC	4	61.80	24.20	2.50	0.78	0.89	0.06	0.342	0.83
UCR	5	1.21	59.40	21.14	0.03	0.48	0.60	0.194	0.54

Landscape pattern in the MCC treatment was compared with specifications given in the Moose Habitat Guidelines (Ont. Minist. Nat. Resour. 1988). Mean size of cut block in the MCC (121 ha; Table 2) was within the range (80-130 ha) of acceptable clearcut size specified in the Moose Habitat Guidelines, and 64% of uncut forest exceeded the requirement to leave 50% uncut. Buffer zones of 200 m between cuts are suggested, but mean interpatch distance in the MCC was only 178 m. Cut blocks are interspersed with wide road corridors in the MCC, which had the effect of decreasing inter-patch distance.

DISCUSSION

A principal goal of the Moose Habitat Guide lines is to help increase Ontario's moose population by enhancing habitat through timber management processes, including forest access, harvest operations, site preparation, regeneration, and maintenance (Ont. Minist. Nat. Resour. 1988). Of these, the most obvious, costly, and controversial component involves harvest operations, where block size of clearcut is generally restricted to <130 ha. Increased interspersion of mature uncut forest with forage producing clearcuts is expected to improve cover quality and forage availability (Euler 1981), and therefore mitigate negative effects that conventional clearcutting has on moose populations (Ont. Minist. Nat. Resour. 1988). This expectation is predicated on the assumption that moose populations are limited because cover and forage are not interspersed sufficiently in large clearcuts.

Our results do not support the hypothesis that application of modified cutting with smaller clearcut blocks results in higher moose densities than application of progressive clearcutting. Moose population increased within PCC yet remained unchanged within MCC, and indeed the interspersion of cover with forage was greater in PCC. Combined HSI was highest in MCC, but this was countered by a high density of roads, which is a direct result of the harvest strategy of dispersed block cuts. The resultant increase in hunter access may be the cause of lower moose density within MCC.

The Ontario government's 1980 wildlife policy instructs resource managers to create moose habitat that approximates habitat created by a large forest fire of medium intensity, and allows for road closures where “logging operations have recently been completed and bush roads make moose easy to locate” (Ont. Minist. Nat. Resour. 1980). Our results suggest that the 1980 policy was not in error, rather the implementation of the policy through the 1988 Moose Habitat Guidelines, which resulted in a fragmented forest, caused the failure of moose populations to increase as expected. In contrast, landscape patterns created by cutting in PCC more closely resemble large burns, although a more comprehensive, multiscale comparison of patch shape and size in PCC and WFB identifies important differences (Gluck and Rempel 1996).

Cutting within PCC occurred over 1977-91, and consequently the cutover is of nonuniform age classes. The high level of interspersion occurred in part because of the temporal progression of the cut, aerial seeding and subsequent thinning of jack pine, and inclusion of wildlife corridors and aquatic buffers. In this sense, PCC was not a contiguous cutover, but rather a mosaic of vegetation age classes. To a lesser degree, the residual forest stands in WFB also created a mosaic of vegetation age classes (Gluck and Rempel 1996).

Alternative explanations that Moose Habitat Guidelines were not properly implemented in MCC and that regional trends in density overpowered landscape-specific trends were also considered. Our comparison of patterns in the MCC to specifications in the Moose Habitat Guidelines indicate planners of the MCC landscape did indeed follow rules of the guidelines. The alternative hypothesis that moose density did not increase in MCC because of a general downward trend in regional moose population levels, perhaps driven by climate or predation, is also unsupported. Moose density increased slightly in those landscapes not disturbed by logging or fire, possibly because of the fine scale disturbance regime caused by forest succession.

Comparable observations of hunter-habitat interaction found in this study have been reported elsewhere. Euler (1983) found that the moose population increased following reduction of hunting pressure in 2 wildlife management units, and that this change in population appeared unrelated to levels of natural predation. Eason (1985) reported that moose numbers declined in recently clearcut areas if hunting was unrestricted, and then rebounded when restrictions were reinstated. Timmerman and Gollat (1983) found that declines in numbers of moose could be attributed in part to hunter access provided by logging roads.

Hunters have the potential to regulate moose populations under the selective harvest system used in Ontario because the number of tags issued is a function of estimated prey density from previous population surveys. Consequently, the numerical response of hunters is density dependent, and hunters can theoretically both limit and regulate moose populations.

MANAGEMENT IMPLICATIONS

Our results suggest Moose Habitat Guidelines alone are insufficient for increasing moose density, and that some form of hunting restriction, such as closure in the early years following harvest, is required to increase density. With the modifications suggested for analysis of forage, the Lake Superior HSI (Model II) is a suitable landscape-level tool for characterizing habitat quality; however, our results indicate that high quality habitat alone will not result in higher densities of moose if hunting is unmanaged. The effect of hunting on moose density explains why relations between the HSI and moose density are often weak, and illustrates the importance of evaluating effects of resource management policies at the landscape scale.

Our analysis suggests that application of Moose Habitat Guidelines is inappropriate for the objective of patterning landscape structure after natural disturbance; the disturbance pattern mimics neither broad-scale burn nor fine-scale gap disturbance. Our evidence suggests that if landscapes are managed to emulate the structure of natural burns, and if this is applied concurrently with restrictions on hunter access, the policy objective of increased moose density can be achieved.

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