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Are boreal forest mammals good indicators of cumulative effects?

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Executive Summary: Rapid development in the boreal forest of western Canada has led to an awareness that new ways of assessing cumulative effects are necessary if we are to set thresholds on levels of human activity that are ecologically and socially defensible. While various structural indicators have been proposed for setting thresholds, a need exists to link landscape level indicators more directly with the biological resources at risk. We used a dose-response approach to assess how different mammalian indicators respond to an index of cumulative effects in northeastern Alberta. For an area of 1452 townships centered on the Alberta Pacific Forest Management Area, we developed a ranking system (CEI) that integrated information on the cumulative effects of energy sector, forestry, and agricultural development at the township level. We then tested whether a series of biological indicators based on the richness and abundance of forest mammals responded to the CEI. Mammal data was collected in 165 townships by counting the number of tracks left in the snow along 9 km triangular transects. A total of 23 species were detected. Biological indicators based on the abundance & richness of different functional groupings of these species were created and tested using regression to determine whether they were correlated with the CEI. Out of 28 biological indicators examined, 15 responded to cumulative effects. The indicators most strongly correlated with the CEI were those associated with non-native species, particularly deer and coyote. However, several indicators based on the relative abundance of fisher, marten, and lynx were also significant. A method of integrating information from multiple indicators was developed called an Index of Mammalian Integrity (IMI). The IMI was more robust than individual species indicators, being less influenced by natural variation and survey variance. Most metrics were highly scalable being correlated with the CEI at the township (100 km^2) and landscape (>2500 km^2) scales. Our approach is the first of its kind in Canada to try and integrate information from multiple species

into an operational indicator of ecological integrity. With further refinement and integration of other biological indicators such as birds and plants, this approach could provide the necessary information to help managers set ecologically relevant thresholds for cumulative effects assessment. At the same time, our approach suggests that winter tracking is a cost-effective and relatively efficient technique of gathering information on a group of species that are of significant societal concern.

1. Introduction

Societal concern that human activity is causing long-term and possibly irreversible changes to natural ecosystems has resulted in various policy instruments to ensure ecosystems do not lose their “integrity”. In Canada, cumulative effects assessment (CEA) is a regulatory concept derived from these concerns that places limits or thresholds on levels of human activity during the approval phase of project development. Implicit in CEA is the assumption that decision makers understand how human activity affects ecosystem integrity, know where thresholds should be placed to avoid undesirable changes, and have a way of accurately assessing cumulative impacts.

Advances in computer technology, geographic information systems (GIS), and remote sensing have given managers charged with CEA the ability to document the footprint of human activity like never before. These technologies have resulted in the development of indicators based on landscape structure (i.e. fragmentation metrics, land cover types, road density, etc) that are easily obtained, can be used at various scales, are relatively cost-effective, and provide easily defined thresholds (Lausch and Herzog 2002). Structural metrics often resonate poorly with decision makers however, because they do not directly measure the biological resources at risk (Lindenmayer et al. 2002). In addition, structural indicators cannot measure all of the potential anthropogenic impacts that might affect ecosystem integrity and thus run the risk of missing a signal that something within an ecosystem is not functioning as expected (i.e. pollution, effects of invasive species, poaching, etc.).

For society to widely accept the need for thresholds in CEA, information relating altered landscape patterns to changes in biological composition or function is necessary. The challenge for science is to search for consistent relationships between the degree of structural change in an ecosystem and the resultant impact of that pressure on the biological components. Karr (1987) use the term “dose-response curves” to describe these relationships. The dose-response concept has a long history as a tool for setting limits on levels of risk deemed acceptable. In pharmacology, different dosages are applied to research subjects in an effort to identify drug concentrations that cause predefined effects (i.e. lethal dose 50). These results are used to make recommendations on the amount of a drug that should be given to maximize benefits for patients while minimizing risks of adverse side effects. Conceptually, the same principle can be applied for setting thresholds in CEA. By comparing the ecological condition of relatively undisturbed areas to sites along a gradient of cumulative effects, science can provide guidance on the level of risk to ecosystem integrity imposed by different levels of human activity. These risks can then be balanced against the economic rewards gained from human disturbances to aid in defining socially acceptable thresholds.

Which response indicators should be used to set thresholds is a question that has been hotly debated. A common approach in the management of terrestrial ecosystems has been to evaluate the status of charismatic species (i.e. woodland caribou (*Caribou rangifer*) or spotted owl (*Strix occidentalis*)). While criticism has been leveled at the focal or indicator species approach (Landres et al. 1988), the debate has centered less around whether these species are impacted by human activity and more on whether these species effectively encapsulate trends for other valued ecosystem components (Canterbury et al. 2000). An alternative approach that is gaining favor is to combine a number of ecological attributes into a holistic assessment of the state of the ecosytem called an Index of Biotic Integrity (IBI). The underlying tenant of the IBI is that no one species or process is likely to provide a complete picture of ecosystem integrity as single elements often show sensitivity to human influence over a relatively restricted range of activities in a narrow range of landscape elements (Karr 1987). To effectively monitor the condition of an ecosystem, an IBI should contain attributes that cover a wide range of ecological levels, potentially including individual health, population dynamics, community composition, and ecosystem processes.

The rapid development of the boreal forest, particularly in northern Alberta, has raised concerns about the integrity of this ecosystem. These concerns led to the development of the Alberta Biodiversity Monitoring Program (ABMP). The ABMP seeks to track changes in boreal forest integrity caused by multiple types of resource extraction, provide early-warning signals of emerging problems, to determine the scale at which thresholds should be placed, and to provide a reporting tool for assessing the efficacy of alternative management strategies. A wide variety of structural, compositional, and functional elements will be monitored in ABMP including vascular and non-vascular plants, beetles, forest songbirds, mammals, forest structure, soil carbon, and dead wood. Ultimately, data from the ABMP will be incorporated into a detailed IBI that incorporates indicators from multiple taxa. Key to the success of this initiative is assessing the suitability of the proposed biological indicators. Barbour et al. (1995) suggested 5 criteria be used to assess the value of a biological indicator before it is integrated into an IBI: 1) is it relevant to the ecosystem under study and the program’s objectives; 2) is measuring the indicator environmentally benign; 3) is it cost effective to measure; 4) is it sensitive to anthropogenic change; 5) does it have sufficient signal to noise ratio to allow human disturbance effects to be separated from natural variation?

Societal concerns over their welfare, economic value to hunters and trappers, large home range requirements, and a general perception of low ecological resilience to human activities are often cited as evidence for the value of mammalian carnivores as ecological indicators, fulfilling criteria 1 (Carroll et al. 2001). However, these same attributes can make it difficult to assess patterns of carnivore abundance/ diversity in a cost-effective and non-intrusive manner. In contrast, mammalian herbivores are relatively easy to sample and are extensively monitored for setting harvest levels. Less attention has been focused on the effectiveness of mammalian herbivores as indicators of cumulative effects. Our objective was to evaluate which mammalian species and/or functional groupings of mammals fit Barbour et al.'s (1995) criteria as good biological indicators by assessing how these indicators changed along a gradient of human disturbance in northeastern Alberta using a relatively cost-effective and non-intrusive snow tracking technique.

2. Methods

2.1) *Measuring cumulative effects*

Although the ABMP will eventually provide the data necessary to develop an IBI for the entire boreal ecozone of Alberta, a lack of remote sensing data forced us to constrain our study to the boreal plains ecoregion centered on the Alberta Pacific (ALPAC) Forest Management Area (Fig. 1). Except for a few scattered hill systems, the area has minimal topographic relief. Pure and mixed stands of aspen (*Populus tremuloides*) and white spruce (*Picea glauca*) are the most common species in upland sites, although jack pine (*Pinus banksiana*) predominates in drier areas (Schneider et al. 2003). Lowland sites are characterized by open stands of black spruce (*Picea mariana*) and tamarack (*Larix laricina*) and by extensive peatland complexes (Schneider et al. 2003). In the ALPAC FMA, there is 23,842 km^2 of potentially merchantable forest as well as extensive oil and gas deposits. The oil deposits include conventional liquid oil, heavy (low viscosity) oil, and oil sands (a mixture of semisolid oil and sand). Industrial activity within the study area was minimal in the first half of the 20th century (Schneider et al. 2003) but expanded rapidly thereafter. The forestry industry clears a total of 16,000 ha/yr in their FMA, compared with 11,000 ha/yr by the petroleum sector. In areas zoned for agriculture, beef cattle grazing and grain farming are the most common activities.

Given the wide variety of human disturbances occurring in our study area, our goal was not to determine if our indicators respond to a particular type of disturbance (i.e. clearcutting vs. energy sector activity). In fact, designing a study to separate such effects at a landscape scale is nearly impossible due to significant overlap in land tenure between different resource sectors. Instead, we chose to develop a Cumulative Effects Index (CEI) that integrated information about several different types of human disturbance into a single measure (Carver 1996). Because our biological response variables were mammals, we focused on integrating human impacts that we felt would have the greatest effect on these species. Programs designed to measure other biological responses may need to use different measures of human disturbance (i.e. configuration of a particular habitat element, pesticide use etc.). We settled on 5 measures of human disturbance: 1) area converted to agriculture; 2) area clearcut (AVI Mod1 Class = “CC”); 3) total density of roads (km per km^2); 4) total density of other linear features (seismic lines, powerlines, pipelines); and 5) oil/gas well density (wells per km^2). We chose the township as our base unit of measurement (10 km by 10km) for deriving our CEI and assessing mammal response to it. While arbitrary, the township is a convenient descriptor that is often used in resource management planning. At the same time it is sufficiently large to encapsulate the home range of multiple individuals of most of the mammal species we considered, making it realistic that we might be able to detect changes in relative abundance at this scale of resolution. Our study region consisted of 1452 townships.

Our logic in choosing the aforementioned descriptors of human impact was that most of the species we examined are either hunted or trapped. Increased access through linear features is assumed to increase harvesting pressure (Schnedier and Wasel 2000). Conversion of forest to different seral stages by forestry and different vegetation types by agriculture or energy sector development affects the quality of food resources for herbivores, which in turn influences the distribution of predators (Rempel et al. 1997). Mammals may avoid areas of high human activity due to noise or traffic while others may be attracted to such areas as a refuge from predation or as a source of carrion (Forman and Deblinger 2000). Some species such as fisher and marten may require particular habitat types (i.e. old growth forest) that are targeted for forest harvesting (Carroll et al. 1999). However, we did not use measures of fragmentation or amount of a specific forest type in our CEI because so little is known about the habitat selection patterns of most of the species we surveyed in our study area.

To assess the amount of agriculture in our study region, we used the Alberta Ground Cover Classification (AGCC). The AGCC is a Landsat-based classification developed in 2001 that divides the landscape into 28 distinct habitat categories. To assess the level of energy sector development in the region we used the Alberta Base Features Layer. This is a vector-based GIS product that is the official provincial record of the number of seismic lines, pipelines, power lines, roads, well pads, industrial sites, and mines. This product is continually being updated so it is difficult to know exactly what period of time the data is accurate to. Our receipt date for this data was Feb. 2003. Our GIS layer for wells recorded all holes drilled, not the actual wellpad area. Therefore, we defined a well as any legal subdivision where at least a single hole had been drilled. To assess the level of forest harvesting we used the Alberta Vegetation Inventory from

ALPAC current to the 2002 harvest year. For areas outside the FMA we derived our measure of harvesting intensity from the AGCC.

The key to developing a CEI is determining how to integrate data that uses different units or scales of measurement. Our approach was to rank the level of each human disturbance variable, standardize these values by the maximum, and sum the standardized ranks across the 5 metrics of human disturbance. The beauty of this approach is that it allows human impact indicators measured using different methods to be scaled in a way that allows them to be combined in a consistent and rigorous fashion. This approach assumes that each type of disturbance is equivalent in terms of its effects on the biological indicator. Based on Spearman rank tests, all of the human disturbance metrics were significantly correlated with one another (Table 1). Our CEI potentially could range from a minimum of 0 to a maximum of 5. The observed maximum was slightly greater than 4 because few townships had high forest harvesting and high agricultural development.

2.2) A*ssessing mammal responses to cumulative effects*

To assess the relative abundance of mammals, we sampled all individuals that left tracks in the snow by surveying 9 km triangular (3 km a side) transects or “triangles”. Each triangle was centred on a township as much as possible and oriented with the apex of the triangle pointing north. Hence, triangles were placed randomly with respect to habitat type. If any transect intersected >3km of open water, the position of the triangle was moved as little as possible to reduce its exposure to open water. UTM coordinates were generated using GIS and downloaded to Global Positioning Systems (GPS) to allow the snow-tracking technicians to accurately moved around the triangle. Each transect was divided into 9, 1 km segments for purposes of data recording. In each segment of each transect we recorded species and track type and tallied the number of “hits”. Hits were defined as tracks that intercepted the transect and included individuals that had moved back and forth across transects. Scores were then totalled for the entire triangle and used as our measure of abundance for the township. All surveys were done between December 15 and March 15 over a 3-year period (2001-2003).

A major source of variation in our data was the time over which tracks were known to have accumulated (hereafter DSS). We minimized this variation by surveying transects between 2 to 15 DSS. A track-obliterating snow was used to reset the DSS clock and was defined as one that made track identification difficult or impossible. Our working rule was to consider any snowfall > 1 cm as track obliterating. In the winter of 2001-2002, we found that snowfalls < 1 cm generally did not deter trackers from collecting reliable data. When any given township received a snowfall > 1 cm, the DSS count for that township was “reset” at zero. DSS was controlled for in all of our analyses.

To determine how mammals would respond to the CEI, we selected 141 townships (24 of which we resampled in subsequent years) across the range of the CEI (Fig. 2). Selection was done in a stratified random manner with townships sampled in the same year never being directly adjacent, the amount of peatland being < 25%, and the spatial distribution of CEI being maximized to the greatest extent possible. While our objective was to get a balanced design across the CEI, there are relative few pristine townships that we could access economically so the distribution was slightly left skewed.

2.3) *Indicators considered*

Although the utility of individual species as indicators of cumulative effects has been questioned, there is great interest in understanding how large mammals and furbearers respond to human disturbance. Therefore, we modeled how the relative abundance of each species changed with the CEI and various other sources of natural variation. Overall, we detected 23 species of mammals. Of these, 10 were relatively uncommon occurring in < 15% of samples. These included domestic cat (*Felis cattus*), cow (*Bos taurus*), domestic dog (*Canis familaris*), horse (*Equus caballus*), domestic sheep (*Ovis aries*), wolverine (*Gulo luscus*), caribou (*Rangifer caribou*), beaver (*Castor canadensis*), mink (*Mustela vison*), elk (*Cervus canadensis*), and fox (*Vulpes fulva*). Due to a lack of data we chose not to create models for these species. Although otters (*Lontra canadensis*) occurred in 18% of samples, we chose not to model this species due to its strong association with water. We also chose not to develop models for the tracks of humans (*Homo sapiens*). The remaining 11 terrestrial species were common, occurring in 42 to 100% of our samples (Table 2). We pooled the tracks of fisher (*Martes pennanti*) and marten (*Martes americana*) due to their similar ecological requirements.

Although indicators based on individual species can be useful for developing support for a monitoring program or eliciting a public response to an environmental issue, the reliability of individual species as indicators can be problematic if there is considerable variation in the estimate of abundance. Variation in abundance can be caused by many factors including natural population cycles, sampling error, annual movements, and natural variation in response to habitat structure. A good indicator should be reasonably robust to these sources of variation. This has led many IBI proponents to use functional groups (i.e. guilds) as indicators. Diversity measures such as species richness are often proposed as good ecological indicators because of their reduced variance relative to abundance measures. Therefore, we tested whether total species richness, % of species consisting of native species, % of species that were carnivores, and % of species of concern were correlated with the CEI. The coyote (*Canis latrans*), deer (*Odocoileus* spp.), humans, and domestic animals were considered non-native species. Wolverine, wolf (*Canis lupus*), caribou, fisher, marten, and lynx (*Lynx canadensis*) were listed as species of concern. The difficulty with diversity indicators for boreal forest mammals however, is the limited number of species. Therefore, we also generated indicators based on the proportional abundance of different groups. We tested the % of all tracks left by species native to the boreal forest, % of tracks left by carnivores, % of tracks of species of concern, % of carnivore tracks left by coyotes, and % of herbivore tracks left by deer. Human impact may alter the competitive advantage of generalist species over more specialized native species (Gompper 2002). To test whether ratios of particular native to non-native species were good predictors of the CEI we examined the lynx to coyote ratio, moose to deer ratio, and fisher/marten to weasel ratio. Finally, systems where top-level predators have been extirpated by human activity can have increased densities of prey that the predator formerly relied on (Schmitz et al. 2000). Thus, we tested whether various predator to prey ratios were correlated with CEI (total predator to prey ratio, native predator to prey ratio, lynx to hare ratio, fisher/marten to squirrel ratio).

The principle of IBI is that no one species or process is likely to provide a complete picture of ecosystem integrity. Although some of our indicators were strongly correlated, the majority only showed weak correlations with each other suggesting they incorporated different information. Therefore, we generated an IMI (Integrated Mammal Index) to determine if a composite indicator would be more robust by reducing the effects of variation seen in the individual indicators. We combined four of our strongest individual indicators into the IMI (see results). Indicators that were predicted to respond positively to the CEI were given values of 1 for their highest values, 2 for above median values, 3 for below median values, and a 4 for low values. This scoring system was reversed for those indicators that that were predicted to respond negatively to CEI. In other words, indicators with a score of 4 were assumed to be indicative of a system with high ecological integrity. Cutoff values for these categories were set based on the median and quartile values for each indicator after they were corrected for DSS. The derived scores from the four indicators were summed to derive the IMI.

2.4) *Statistical analysis*

Almost with exception, our raw measures of mammalian abundance and diversity violated the assumption of normality in their raw form, tending to be right skewed. This was not surprising as the data were generated from a count type of process (Cameron and Trivedi 1998). However, rather than use a model such as Poisson or negative binomial regression we chose to normalize the data using a Box-Cox maximum likelihood transformation and analyze the data using standard ordinary least squares (OLS) regression. This approach normalized the mammal data and minimized heteroscedasticity in most cases. Our logic in using this approach was that many counts were extremely high (into the hundreds) making count models suspect. Preliminary analyses using negative binomial regression demonstrated poor model fit particularly at low count values. In addition, each track we detected was unlikely to represent a unique observation from a single individual. Instead our measure combines information on both abundance and activity level making it difficult to say exactly how many animals per township were detected. A more useful measure is the relative difference between townships occurring at different ends of the CEI adjusted for other parameters. These results were best portrayed using transformed data, OLS regression, and partial residual plots. To account for a lack of statistical independence between townships sampled in multiple years, we clustered our analysis using the robust cluster option in the program STATA (Hardin and Hilbe 2001). The robust cluster accounts for the lack of independence by reducing total degrees of freedom on which statistical tests are based to the level of the cluster, which in this case was the township (n = 142).

We had five key questions: 1) what is the nature of the relationship between each indicator and the CEI; 2) how sensitive was the indicator / CEI relationship to natural sources of variation (latitude, year, habitat composition, or food abundance); 3) did the indicators respond to CEI consistently in space and time; 4) was CEI the best way to test for human impacts on boreal forest mammals; and 5) did these relationships change with scale? To test question 1, we compared the fit of a Generalized Additive Model (GAM) to a simple linear function based on a model with all main effects. To address question 2, we built a series of models that varied the natural sources of variation (Table 3). We then calculated Akaike weights for each model, multiplied these weights by the parameter estimate and SE for CEI, and summed the product across all models (Burnham and Anderson 1998). This resulted in a model-averaged parameter estimate and confidence intervals for the CEI that was robust to model uncertainty. Question 3 was tested by comparing the fit of a model containing all main effects relative to models that also contained the interaction between spatial location*CEI and year*CEI. Model fit was compared using Akaike's Information Criteria corrected for small sample size (AIC*c*). To address question 4, we replaced CEI with each of the human impact variables and compared these models to the CEI model using AIC*c*.

Given that our triangles were randomly located with respect to habitat, we hypothesized that some species might be affected by the composition of the remaining undisturbed habitat.

We summarized the data from the Alberta Ground Cover Classification into 11 classes and calculated the proportion of undisturbed habitat comprised of these classes. A Principal Components Analysis was used to reduce 11 variables to 5 orthogonal factors. Although habitat is often used in developing predictive models of species abundance, habitat can be a surrogate for other resources (i.e. prey abundance). Thus, we developed models for each carnivore species that used different prey species to predict carnivore abundance. The prey species used for each predator were chosen *a priori* based on what the literature speculated were the major food items for these species. Finally, we chose to model latitude. Latitude could represent a natural gradient related to climate, soils, subcanopy vegetation attributes, or some combination thereof. However, in our study area, latitude was highly correlated with the CEI ($r^2 = 0.55$). This occurs because almost all of the agriculture is at the southern periphery of the study area where road density and energy sector activity is highest. We made the *a priori* assumption that the CEI was the more important predictor of our indicators and used the residuals from a regression model (Latitude = CEI) as our measure of spatial location independent of CEI (hereafter North).

Although the township is often used for some types of resource planning, the forestry sector in particular, often plans on much larger (1000's km^2) scales. To determine whether the indicators responded to cumulative effects at such scales, we generated 11 *a posteriori* landscapes. Landscapes were derived by selecting groups of 12 spatially clustered townships and then using a minimum convex polygon to delineate the boundary of each landscape. This post hoc approach resulted in 11 landscapes ranging in size from 2500 to 7000 km^2. For each landscape, we derived the same human disturbance metrics and created a new CEI^{Land}. With a small sample size such as this, modeling a lot of variables was not possible. We chose to examine whether North, CEI^{Land}, and prey abundance for carnivores were significantly correlated with each indicator. DSS was not controlled for in this analysis because average DSS at this scale was nearly identical among landscapes. Because of small sample size we had to assume that the data did not violate the assumptions of OLS regression. We did not transform these data.

3. Results

3.1) *Indicator species*

Of the 10 species modelled, 4 responded to our metric of cumulative effects (Table 2). None of the species showed strong evidence of non-linear responses (all $P > 0.05$). Deer and coyote were positively correlated with CEI while lynx and fisher/marten were negatively correlated with CEI (Fig. 3a,b,c). These patterns were consistent in space and time for most species as the interaction between CEI*Year and CEI*Space were not significant. The exception was deer for which the model with the interaction between CEI*Year was the most likely candidate. Deer response to CEI was positive and significant at $P < 0.001$ within each year. The significant interaction was caused by the fact that the relationship between CEI and deer was extremely pronounced in 2002 with the relationship being weaker in 2001 and 2003. Wolves did not respond to CEI. For predators, the abundance of prey was always a significant predictor. North was significant for 7 species with fisher/marten and lynx being more common in the north and deer and coyote more common in the south. Mouse, squirrel, and wolf showed a slight response to North. No other single human disturbance factor generated a better fitting model except for the coyote. Coyote abundance was better predicted by well density than CEI (Table 3). Mouse and squirrel showed highly variable relationships with the CEI varying from positive to negative within different years.

3.2) *Species groups*

Species richness was positively correlated with the CEI (Table 4). The proportion of species detected that were native species or species of concern was greater in landscapes with lower cumulative impacts (Table 4). The proportion of all species detected that were predators was the only diversity indicator not correlated with CEI. All of the diversity indicators except for total species richness increased the further north you went. Although most of the richness metrics were correlated with the CEI, the majority of the relationships were weak ($r^2 < 0.25$). All of the metrics except % native species met the assumptions of normality and homogeneity of variance.

The proportion of carnivore tracks made by coyotes and the proportion of all herbivore tracks caused by of deer were positively correlated with CEI, while the proportion of all tracks caused by individuals of native species was negatively correlated with CEI. The proportion of tracks caused by predators and the proportion of tracks caused by species of concern were related to CEI, but inconsistently as the interaction between CEI and latitude provided better model fit. The fit of most abundance measures was reasonable ($r^2 > 0.16 < 0.44$). The assumptions of OLS regression were violated for total # individuals, % carnivores that were coyotes, and % herbivores that were deer. These violations were minor however ($P > 0.01$). The ratios of lynx/coyote, fisher & marten/weasel, and moose/deer were all negatively correlated with the CEI, but again all ratios other than the fisher & marten/weasel ratio violated one or more assumptions of OLS ($P > 0.01$ in all cases however). None of the predator/prey ratios were significantly correlated with CEI except for fisher&marten/squirrel. The interaction between CEI and year was significant for the lynx/coyote ratio although the response to CEI was always negative when each year was examined separately.

3.3) *Which indicators were the "best"?*

We found that 15 of the 28 indicators we examined were significantly correlated with the CEI. Only 12 were stable in space and time (no evidence for significant year*CEI or latitude*CEI interactions). Ranking these indicators by the amount of change predicted by the regression model for low vs. high CEI landscapes, the best indicators were: proportion of herbivore tracks created by deer (Fig. 4) > proportion of tracks made by non-native species > proportion of carnivore tracks made by coyotes (Fig. 5) > ratio of moose to deer tracks > number of fisher/marten tracks > number of lynx tracks > number of coyote tracks > ratio of fisher/marten to squirrel > total species richness > % of total richness comprised of species of concern > ratio of fisher/marten to weasel > % of total richness comprised of native species. Proportion of herbivore tracks created by deer and proportion of tracks created by native species were highly correlated (r = -0.98). All of the indicators that utilized fisher/marten or coyote as numerators were significantly correlated to fisher/marten or coyote abundance, respectively (r > 0.8).

3.4) *Integrating different indicators*

We combined the proportion of herbivore tracks created by deer, the proportion of carnivore tracks made by coyotes, the ratio of fisher/marten to weasel tracks, and % of total richness comprised of native species to derive the IMI. We chose these indicators because they had the best fit to CEI, were not perfectly correlated with each other, and were influenced by the fewest other parameters (natural variation variables). The IMI could range from a 4 to 16. The

IMI was negatively correlated with CEI (index of integrity declined as cumulative effects increased: $\beta = -1.55 \pm 0.19$, $r^2 = 0.28$: Figure 6). Including north in the model dramatically improved the variance in the IMI that was explained ($\beta = 0.144 \pm 0.02$, $r^2 = 0.42$). Addition of other variables such as habitat or year did little to improve model fit.

3.5) *Issues of scale*

At the landscape scale, we found that the majority of species that were significantly correlated with CEI at the township were also correlated with CEI at the landscape scale. Coyote ($r^2 = 0.52$) and deer ($r^2 = 0.87$) were positively correlated with CEI and negatively correlated with North (Fig. 7a). Lynx ($r^2 = 0.55$) and fisher ($r^2 = 0.51$) responded negatively to CEI and positively to their primary prey (Fig. 7b). Marten was not correlated with CEI but there was a significant outlier in one moderately impacted landscape. Removal of this outlier resulted in a highly significant negative correlation with CEI and a positive correlation with North ($r^2 = 0.78$: Fig. 7c). Weasels were positively correlated with the abundance of mice ($r^2 = 0.55$) while wolves were positively correlated with north ($r^2 = 0.59$). Hare, moose, and mouse were not correlated with any variable.

Richness increased with CEI. The % all species that were predators increased with North, while % of all species that were species of concern and % species that were native were not correlated with either north or CEI. The total proportion of tracks left by native species and species of concern were negatively correlated with CEI while the proportion of carnivore tracks left by coyotes and proportion of herbivore tracks left by deer were positively correlated with CEI. The proportion of carnivore tracks caused by coyotes was negatively correlated with North. Finally, the lynx/coyote ratio and moose/deer ratio were negatively correlated with CEI. The fisher/weasel and fisher/squirrel ratio were positively correlated with North. Predator/prey, native predator/prey and lynx/hare ratios were all not significant.

4. Discussion

4.1) *The patterns*

Human activity is rapidly increasing in the boreal forest of North America. As our study shows, this activity is correlated with changes in the abundance of individual mammalian species and the overall composition of that community. No one human disturbance factor seems to be responsible for these changes, as individual measures of human disturbance (i.e. road density) were rarely better predictors of the biological indicators than was the CEI. Regardless of the mechanisms causing these changes, our results suggest many of the indicators we considered fit Barbour et al. (1995)'s criteria as suitable for an IBI as: 1) the indicators are directly relevant to the goals of the ABMP; 2) snow tracking is a passive measure of collecting abundance and diversity data for mammals with low ecological impact; 3) snow tracking is inexpensive to collect relative to other survey methods; and 4) many of the indicators considered respond to human disturbance at various spatial scales.

Demonstrating that these indicators responded to human disturbance was important. Equally important however was whether the indicators had sufficient signal to noise ratio to allow human disturbance effects to be separated from natural variation. Using an information theoretic approach we were able to demonstrate the mean response to CEI was robust to natural variation for at least 10 of our indicators and the IMI overall. Indicators based on the relative proportions of certain groups or species were particularly useful as they were less influenced by survey variance (i.e. DSS and year) than were individual species. These types of indicators tended to require fewer number of model parameters to explain similar levels of variance.

An uncertainty in our indicators is how they respond to latitude. North was often just as important as the CEI in explaining the patterns in abundance we observed. The confounded nature of cumulative effects and latitude is not unique to Alberta's boreal forest. Over most of the boreal forest of North America, gradients in human disturbance are typically strongest in the south and increase as you move northward due to declining quality of agricultural soils, distance to market for wood products, and accessibility of petroleum resources (Hobson et al. 2002). *A priori* we did not expect there would be such a strong spatial pattern in mammal abundance with latitude because we tried to sample townships along the CEI in different spatial locations. We could not do this for agriculture nor for the relatively pristine townships because these were spatially clustered. While our approach to dealing with collinearity between latitude and CEI was robust, it does give precedence to finding significant effects of the CEI over latitude (Villard et al. 1999). That the corrected latitude term remained an important predictor for most of our indicators suggests the spatial pattern is extremely robust. What is unclear is whether this spatial pattern has always been present in the boreal forest (i.e. lynx were never common in the south) or whether this represents a slow deterioration of the southern fringe of the boreal forest caused by human activity directly. Alternatively, human-mediated expansion of deer and coyotes may be playing a role in shifting the distribution of native species such as lynx and fisher northward through direct or indirect competition.

4.2) *What could an IMI be used for?*

Once constructed, traditional IBI are used by managers to monitor the ecological condition of individual units (Wallace et al. 1996). Despite the fact that the mean response of our indicators to the CEI was significant, their predictive capability at the level of the individual township was poor. The 95% prediction intervals for highly disturbed vs. low impact townships overlapped in all cases. The same issue exists for the IMI. Does the fact that our predictive ability at the township level is low mean that boreal forest mammals do not make good biological indicators? The answer depends on the intended use of the indicator. For a manager wanting to assess whether the condition of a particular township has deviated from expected, our indicators may be of limited utility because the predicted value of a single pristine township can by chance be similar to that of a highly degraded township. With more data the degree of uncertainty in our indicators will decline but whether the level of precision will be sufficient for explicitly defining the ecological condition of individual townships is uncertain.

In contrast, if the indicator is intended as a reporting tool to assess the overall ecological integrity of a specific area then our indicators seem to be robust. Regional level assessments using these types of indicators could be done in several ways. One approach would be to calculate the average value and variance for the IMI or other indicator for *x* number of townships in a specific area and compare that to other regions or other time periods to assess ecosystem condition or management effectiveness. Based on our landscape level analyses however, pooling data from multiple townships to generate a regional score could be more effective as fewer variables are required to explain the same amount of variation with many of the details (i.e. DSS, habitat type) having less of an effect.

4.3) *Issues in implementing* the *IMI as an operational tool*

The indicators with the best predictive power were those based on what we considered non-native species. The coyote originally was a predator of the Great Plains ecosystem until about 1800 when they began to expand their range. Now they occur across most of North America but are highly variable in abundance (Gompper 2002). A similar pattern has been observed for white-tailed deer expansion. Use of deer, and to a lesser extent coyotes, as an index of negative impacts on the boreal forest ecosystem may be met with some resistance. Deer are important to people for both food and aesthetic value. Thus, increasing and expanding deer populations may to some be a desirable condition for the boreal forest. In many areas of the world, healthy deer populations are used as positive indicators of ecological condition (Hanley 1996). We do not dismiss the value of deer and coyotes to society nor their ecologically important role in their native prairie and aspen parkland ecosystems. However, our data suggests that human disturbance is allowing these animals to expand their range in the boreal forest. Whether this expansion is having negative effects on native boreal species is unclear. However, the potential for altered predator-prey dynamics, new competitive interactions, and changes in of herbivory patterns means the possibility of negative impacts from deer and coyotes should be considered, particularly given the difficulty of reversing the dominance of these animals in a system once they become established (Gompper 2002).

The potential reluctance of society to accept deer and coyotes as indicators of negative ecological effects reflects a criticism of the dose-response and IMI concept. Finding a reference condition that suits all interested parties can be problematic. Should the reference condition be the most pristine places in the current landscape as we have done, the landscape prior to any industrial development, the landscape prior to European settlement, or the landscape prior to humans existing in the boreal forest? Each of these reference conditions has its pros & cons. Many government agencies (i.e. Parks Canada) have maintaining the landscape within the natural range of variation prior to European settlement as their stated target. While laudable, each step back in time makes the level of uncertainty and possibility of obtaining defensible data on the "natural range of variation" more difficult. In the boreal forest, the number of reference "landscapes" with no to low human impact are rapidly disappearing but can still be found suggesting that the current landscape may provide a suitable reference condition. Regardless of whether we use historical or current landscapes, what is needed is a decision on the type of baseline condition that is desired. Without a clearly defined target condition to use as a reference, society will suffer from what Pitcher (2001) calls ''Pauly's ratchet''. This is the psychological tendency for people to relate changes in an ecosystem to what things were like at the time they started their jobs, their youth, or some other specific time period. While not wrong, such perceptions become a moving target that results in slow environmental changes becoming widely accepted as the normal condition for that system (i.e. coyotes in the boreal forest).

The IBI / dose-response approach has also been criticized because sites are selected based on whether or not the observer perceives them to be degraded or pristine and then seeks indicators to justify this designation (Lopez and Fennessy 2002). While such a criticism is valid on its face, repeated rigorous documentation of biological responses to similar gradients of human disturbance will increase our confidence in the generality of the patterns and the potential utility of these indicators. If the models we created can be used to predict the average condition in other areas of the boreal forest then we can be confident that these indicators have merit as management tools. Of particular interest, is whether similar patterns would occur if our gradient of human impact spanned zones of longitude rather than latitude. At the same time, 13 of our indicators did not respond to the CEI. Are these indicators any less important for assessing ecological integrity? Certainly, these species are crucial to ecosystem functioning and integrity and play important ecological roles. That they have not changed with increasing cumulative effects could be caused by a variety of factors including survey error, inappropriate scale, and the strong possibility that they are resilient to cumulative impacts. Some would argue that these indicators should also go into any index intended to track how boreal forest biodiversity responds to human impacts. This was not our goal however. Our goal was to provide a measure of ecological integrity that was highly sensitive to cumulative effects.

Although the dose-response concept provides the scientific framework for establishing how biological indicators respond to increasing human disturbance, science cannot provide “the” acceptable threshold of human activity for CEA using this or any other approach. Thresholds must be set by society and will depend on the tradeoff between economic growth and the level of ecological risk we are willing to accept. Future work should examine how dose-response curves and IBI can be linked with economic indicators. By doing so, science will provide society a better set of tools for making informed decisions about the type of environment they desire and the costs & benefits of these decisions.

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Table 1 - Spearman rank correlations between five measures of human disturbance and the overall cumulative effects index (CEI).

Human Disturbance Variable	Road Density	Linear Feature Density	Well Density	Area Harvested	Area of Agriculture	CEI
Road Density	1	0.46	0.61	0.12	0.70	0.85
Linear Feature Density	.	1	0.56	0.26	0.21	0.72
Well Density	.	.	1	0.12	0.44	0.78
Area Harvested	.	.	.	1	-0.24	0.38
Area of Agriculture	.	.	.	.	1	0.63
CEI	.	.	.	.	.	1

Table 2 - Model averaged coefficients and 95% confidence intervals for individual species/ CEI relationship. The overall fit of the model with the lowest AIC*c* score is reported as r^2. Parameters from the model with best fit that were statistically significant at $P < 0.05$ are reported. Parameters are listed from highest to lowest explanatory power based on standardized beta coefficients. Predicted values based on least-squares means for landscapes with low vs. high CEI scores are given with 95% prediction intervals shown in brackets. The difference in AIC*c* scores for the best fitting model based on CEI vs. the other measures of human impact are also listed. The number of townships in which a species was detected is also listed as Occ.

Indicator	CEI Coefficient	R^2	Significant parameters	Predicted Value for Low CEI	Predicted Value for High CEI	Better Fit	ΔAIC	Occ.
Coyote	0.47 (0.27 to 0.67)	0.40	Year +CEI +Hare -North +Deer +Hab4 +DSS	1.5 (-0.7 to 3.7)	3.3 (1.1 to 5.4)	Wells	14.2	83%
Deer	3.62 (1.98 to 5.27)	0.52	CEI –North +CEI*Year +Hab2 +Hab1 +Year	4.7 (-5.4 to 14.7)	19.1 (4.9 to 28.8)	Harvest	3.4	87%
Hare	-0.30 (-1.91 to 0.31)	0.34	Year	.	.	Wells	1.1	100%
Lynx	-0.34 (-0.55 to –0.14)	0.57	Hare +DSS +North –CEI +Year -Hab5	1.8 (0.4 to 3.2)	0.8 (-0.6 to 2.2)	.	0	71%
Fisher/Marten	-0.26 (-0.40 to –0.12)	0.20	-CEI +Squirrel +North +Hab4 +Year	1.8 (0.08 to 3.5)	0.8 (-0.9 to 2.4)	.	0	74%
Moose	0.23 (-0.06 to 0.53)	0.12	Hab4	2.5 (-1.2 to 6.1)	3.4 (-0.2 to 7.0)	Harvest	0.55	85%
Mouse	-0.03 (-0.25 to 0.18)	0.22	Year + CEI*Year +CEI + Hab1+ DSS +Hab2 –Hab3	NC	NC	Other	0.92	88%

Squirrel	0.77 (-0.11 to 1.67)	0.18	CEI*Year +Year +CEI +DSS +North –Hab3 +Hab2 +Hab1 –Hab4	NC	NC	Other	1.59	95%
Weasel	0.06 (-0.35 to 0.47)	0.21	Year +Mouse +DSS -Hab3	2.1 (-1.2 to 5.5)	1.9 (-1.4 to 5.2)	.	0	84%
Wolf[A]	0.20 (-0.20 to 0.65)	0.14	Hab5 +Deer +North	NT	NT	.	0	42%

[A] – Insufficient number of detections to use OLS regression. Data were treated as presence/absence and analyzed using logistic regression.

Table 3 – Model set considered for each indicator.

Model #	Variables in model
1	CEI + Habitat (Hab1 + Hab2 + Hab 3 + Hab4 + Hab5) + DSS
2	CEI + Year
3	CEI + Northing
4	CEI + Year + North
5	CEI + Year + Habitat
6	CEI + Prey (For Carnivore Models Only)
7	CEI + Habitat + North
8	CEI + Habitat + Year + North
9	CEI + Habitat + Year + North + Prey
10	CEI + Only
11	Model 9 + North * CEI
12	Model 9 + Year * CEI
13	Model 9 (Replace CEI with Road Density)
14	Model 9 (Replace CEI with Other Linear Feature Density)
15	Model 9 (Replace CEI with Other Well Density)
16	Model 9 (Replace CEI with Area Harvested)
17	Model 9 (Replace CEI with Area Converted to Agriculture)

Table 4 - Model averaged coefficients and 95% confidence intervals for diversity/ CEI relationship. The overall fit of the model with the lowest AIC*c* score is reported as r^2. Parameters from the model with best fit that were statistically significant at $P < 0.05$ are reported. Parameters are listed from highest to lowest explanatory power based on standardized beta coefficients. Predicted values based on least-squares means for landscapes with low vs. high CEI scores are given with 95% prediction intervals shown in brackets. The difference in AIC*c* scores for the best fitting model based on CEI vs. the other measures of human impact are also listed.

Indicator	Coeff	R^2	Significant parameters	Predicted Value in Pristine	Predicted Value in Degraded	Better Fit	ΔAIC
Richness	3.71 (1.27 to 6.16)	0.18	CEI +Year +Hab5	29.6 (1.0 to 58.2)	45.3 (17 to 73.6)	Wells	1.98
% Spp. Natives	-0.027 (-0.044 to –0.01)	0.25	Year +North –Hab4 -CEI	0.87 (0.71 to 1.03)	0.74 (0.58 to 0.90)	.	0
% Spp. Predators	-0.058 (-0.134 to 0.0180)	0.05	Year +North	1.31 (0.51 to 2.12)	1.10 (0.31 to 1.90)	Agriculture	1.44
% Spp. of Concern	-0.038 (-0.069 to –0.009)	0.12	North +Year –CEI	0.48 (0.13 to 0.82)	0.32 (-0.02 to 0.66)	Agriculture	1.29

Table 5 - Model averaged coefficients and 95% confidence intervals for composition of mammalian community/ CEI relationship. The overall fit of the model with the lowest AIC*c* score is reported as r^2. Parameters from the model with best fit that were statistically significant at $P < 0.05$ are reported. Parameters are listed from highest to lowest explanatory power based on standardized beta coefficients. Predicted values based on least-squares means for landscapes with low vs. high CEI scores are given with 95% prediction intervals shown in brackets. The difference in AIC*c* scores for the best fitting model based on CEI vs. other measures of human impact are also listed.

Indicator	CEI Coefficient	R^2	Significant parameters	Predicted Value for Low CEI	Predicted Value for High CEI	Better Fit	ΔAIC
# Individuals	1.28 (-0.29 to 2.87)	0.44	Year +DSS +Hab2 +Hab5	35.2 (19.0 to 51.4)	39.6 (23.5 to 55.6)	.	0
% Individuals native	-1.15 (-1.41 to –0.88)	0.43	-CEI +North +DSS	6.2 (3.1 to 9.3)	1.6 (-1.5 to 4.7)	.	0
% Individuals predators	-0.002 (-0.007 to 0.002)	0.16	Year +CEI*North +Hab4	0.049 (-0.010 to 0.108)	0.041 (-0.018 to 0.099)	Roads	2.14
% Individuals species of concern	-0.005 (-0.007 to –0.003)	0.28	CEI*North +Year – CEI	0.024 (-0.001 to 0.005)	0.006 (-0.018 to 0.030)	Roads	1.43
% Carnivores coyotes	0.030 (0.018 to 0.042)	0.26	CEI +Year +Hab4 +Hab5	0.046 (-0.073 to 0.165)	0.160 (0.042 to 0.277)	.	0
% Herbivores deer	0.057 (0.043 to 0.072)	0.41	CEI – North - DSS	0.012 (-0.163 to 0.186)	0.242 (0.068 to 0.415)	.	0

Table 6 - Model averaged coefficients and 95% confidence intervals for specific ratios of predators & prey / CEI relationship. The overall fit of the model with the lowest AIC*c* score is reported as r^2. Parameters from the model with best fit that were statistically significant at $P < 0.05$ are reported. Parameters are listed from highest to lowest explanatory power based on standardized beta coefficients. Predicted values based on least-squares means for landscapes with low vs. high CEI scores are given with 95% prediction intervals shown in brackets. The difference in AIC*c* scores for the best fitting model based on CEI vs. the other measures of human impact are listed.

	CEI Coefficient	R^2	Significant parameters	Predicted Value for Low CEI	Predicted Value for High CEI	Better Fit	ΔAIC
Lynx to coyote ratio	-0.102 (-0.164 to –0.04)	0.34	CEI*Year – CEI +Year +North +DSS –Hab4	0.475 (0.114 to 0.836)	0.046 (-0.298 to 0.391)	.	0
Moose to deer ratio	-0.020 (-0.036 to –0.005)	0.23	North –CEI +DSS +Hab4 – Hab5	0.147 (0.022 to 0.272)	0.057 (-0.067 to 0.181)	.	0
Fisher/Marten to weasel ratio	-0.020 (-0.041 to –0.001)	0.25	Year +Hab3 – Hab4 –Hab1 – DSS -CEI	0.240 (0.021 to 0.460)	0.172 (-0.046 to 0.390)	Agriculture	4.2
Predator to prey ratio	-0.003 (-0.008 to 0.003)	0.16	CEI*North + Year + Hab4	.	.	Road	2.0
Native predator to prey ratio	0.001 (-0.005 to 0.007)	0.19	Year	0.03 (-0.04 to 0.11)	0.04 (-0.04 to 0.11)	Harvest	1.8
Lynx to Hare ratio	-0.002 (-0.005 to 0.001)	0.21	CEI*Year + Hab3 –Hab1	.	.	.	0
Fisher/Marten to Squirrel ratio	-0.013 (-0.023 to –0.004)	0.11	-Hab1 –CEI –Hab4	0.095 (-0.012 to 0.202)	0.050 (-0.056 to 0.156)	Other	0.5

Figure 1 – Location of study area within Canada and the Alberta Pacific Forest Management Area. Area shown in light gray is study area consisting of 1452 townships.

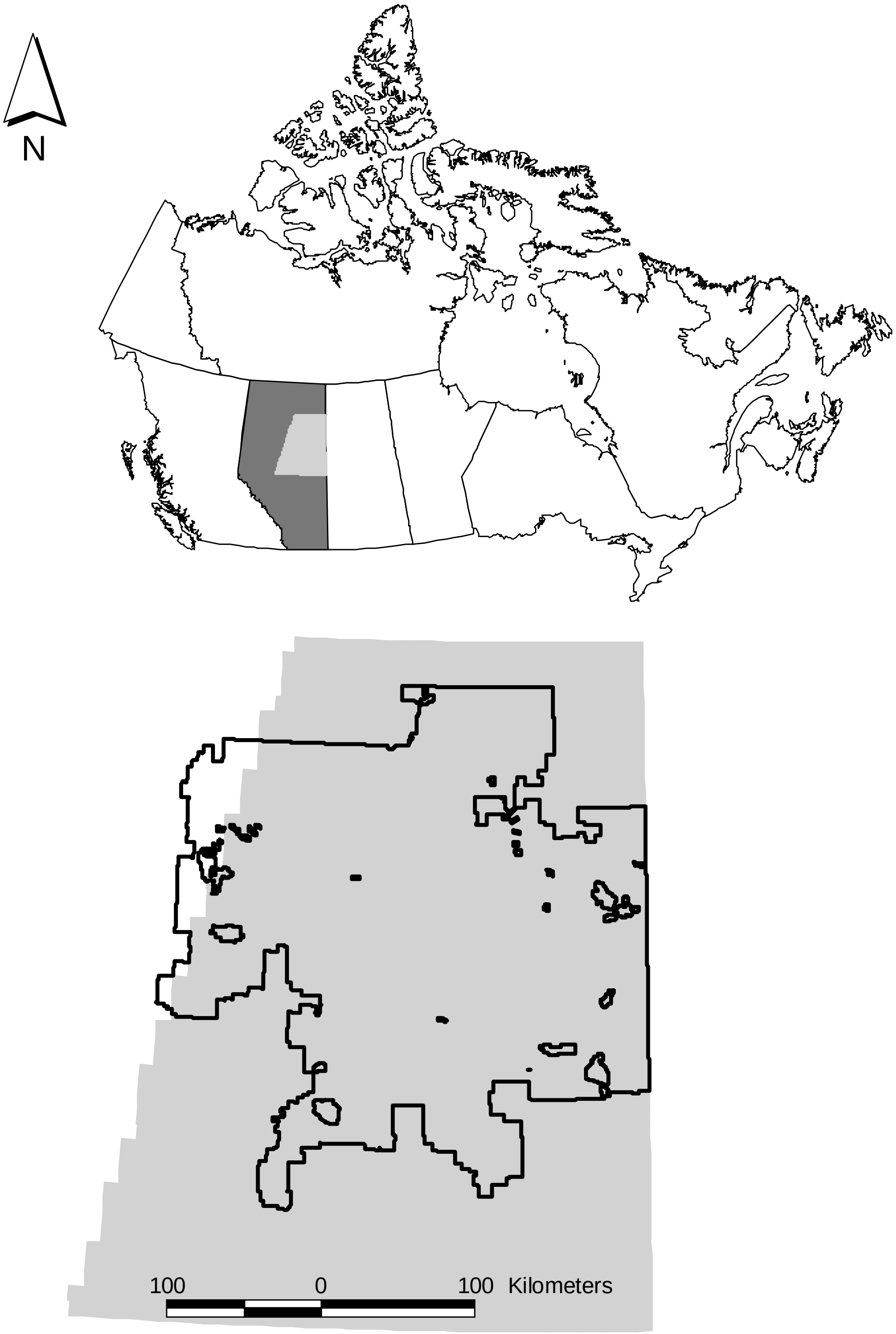


Figure 2 – Cumulative Effects Index Score (CEI) for all 1452 townships in study area. Those with dark black line surrounding them were sampled for mammals.

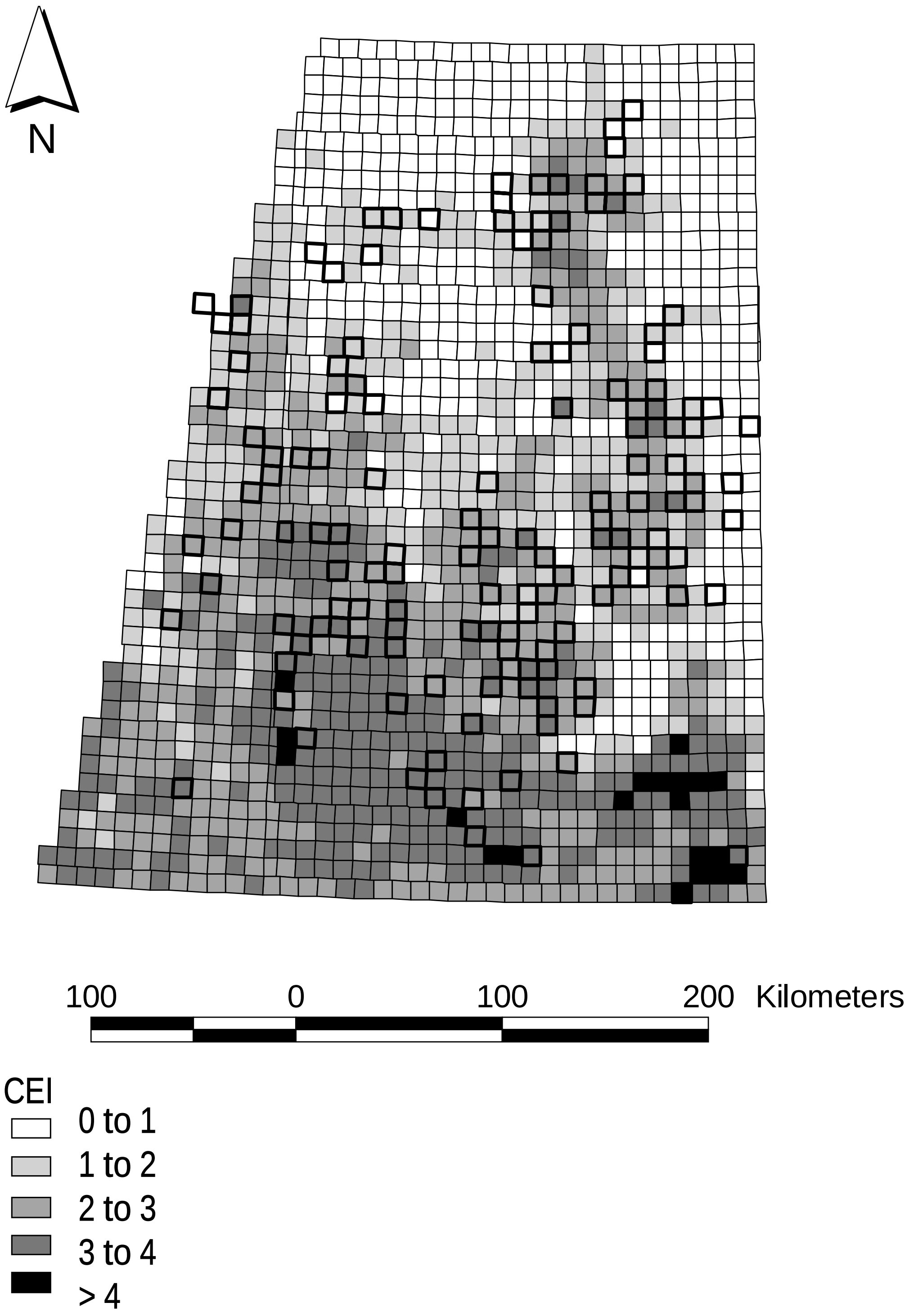


Figure 3 – Partial residual plots showing relationship between abundance of fisher, lynx, and coyote corrected for other variables within multivariable regression plot.

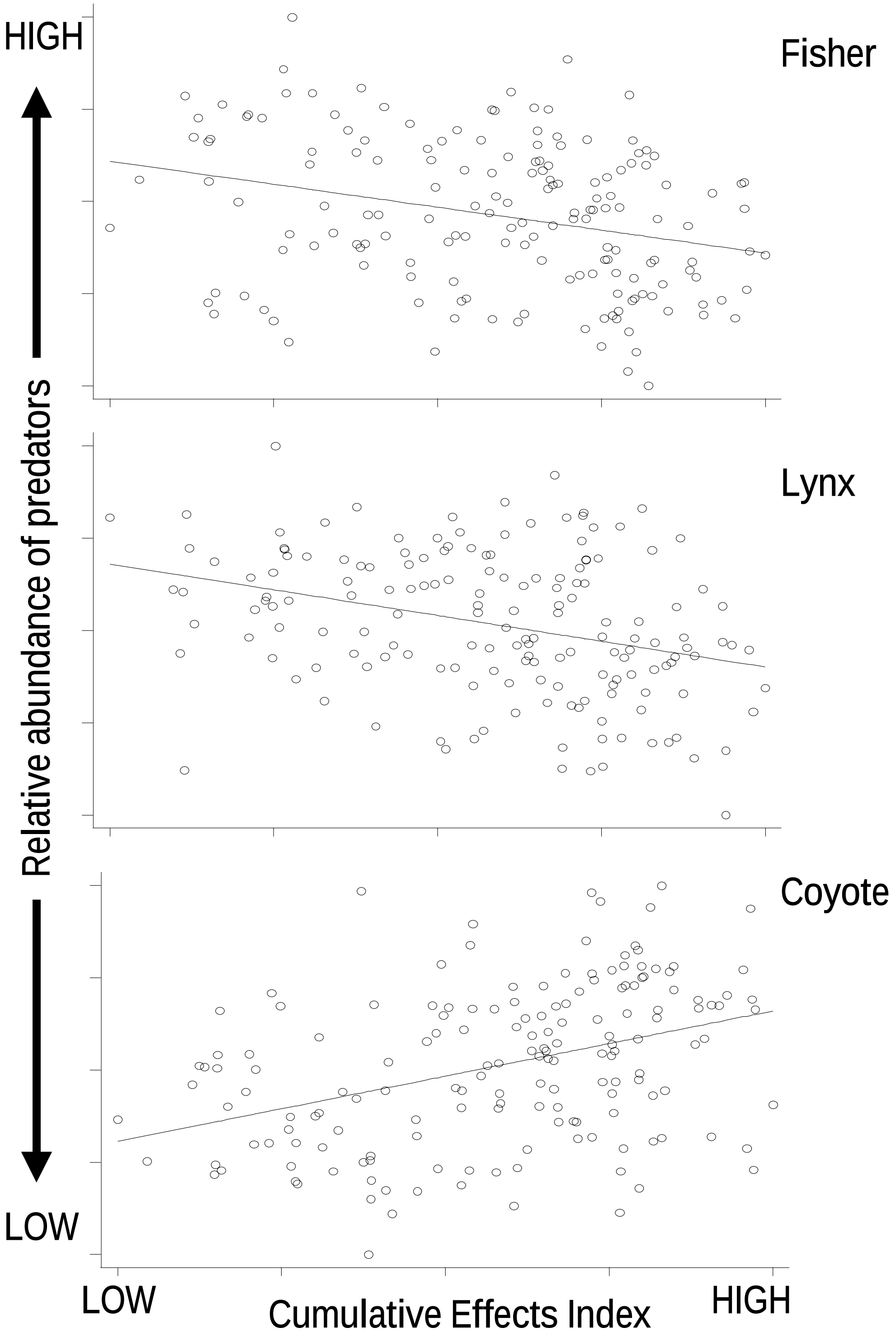


Figure 4 – Map showing predicted proportion of all herbivore tracks created by deer per township for entire study area.

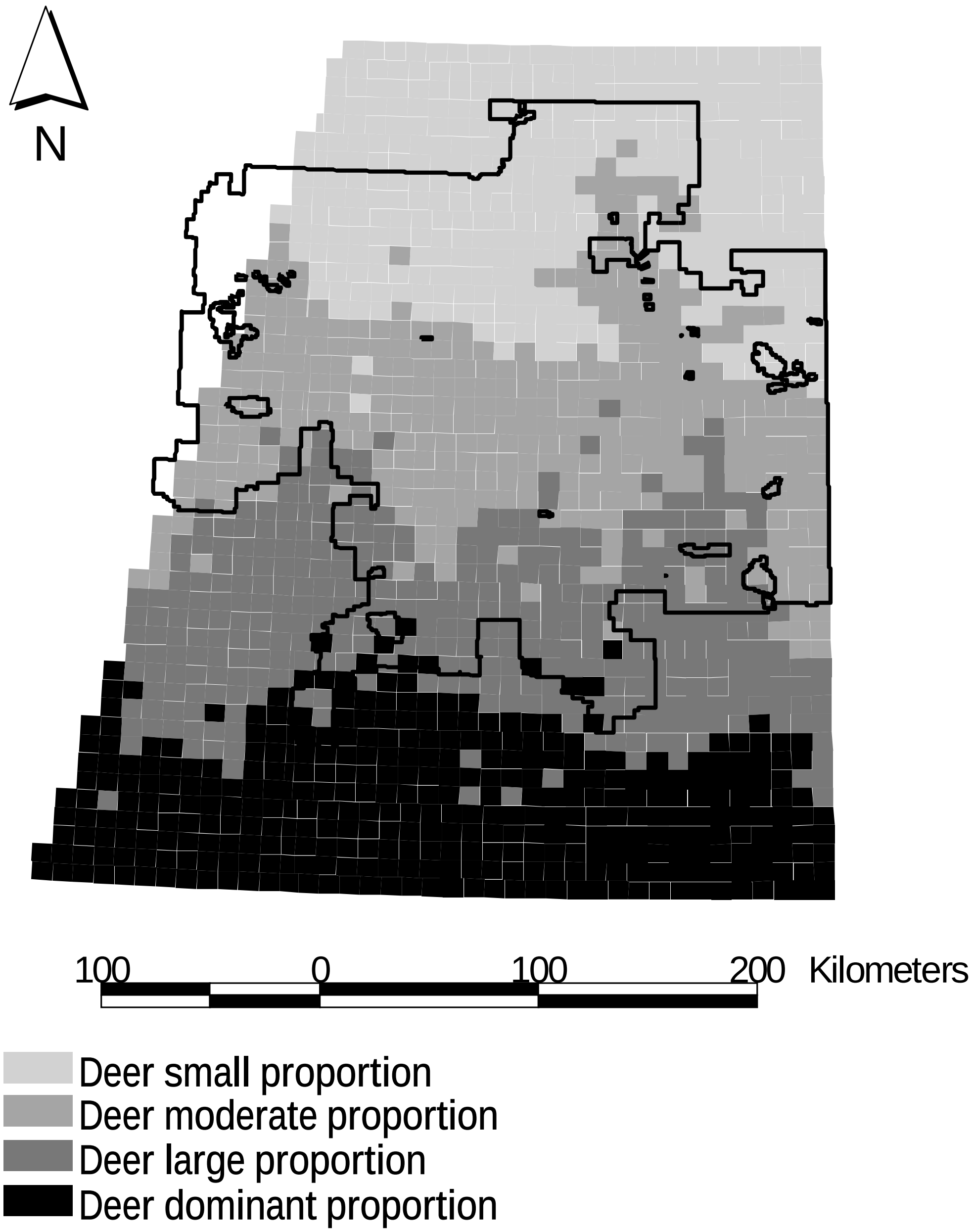


Figure 5 – Map showing predicted proportion of all carnivore tracks created by coyote per township for entire study area.

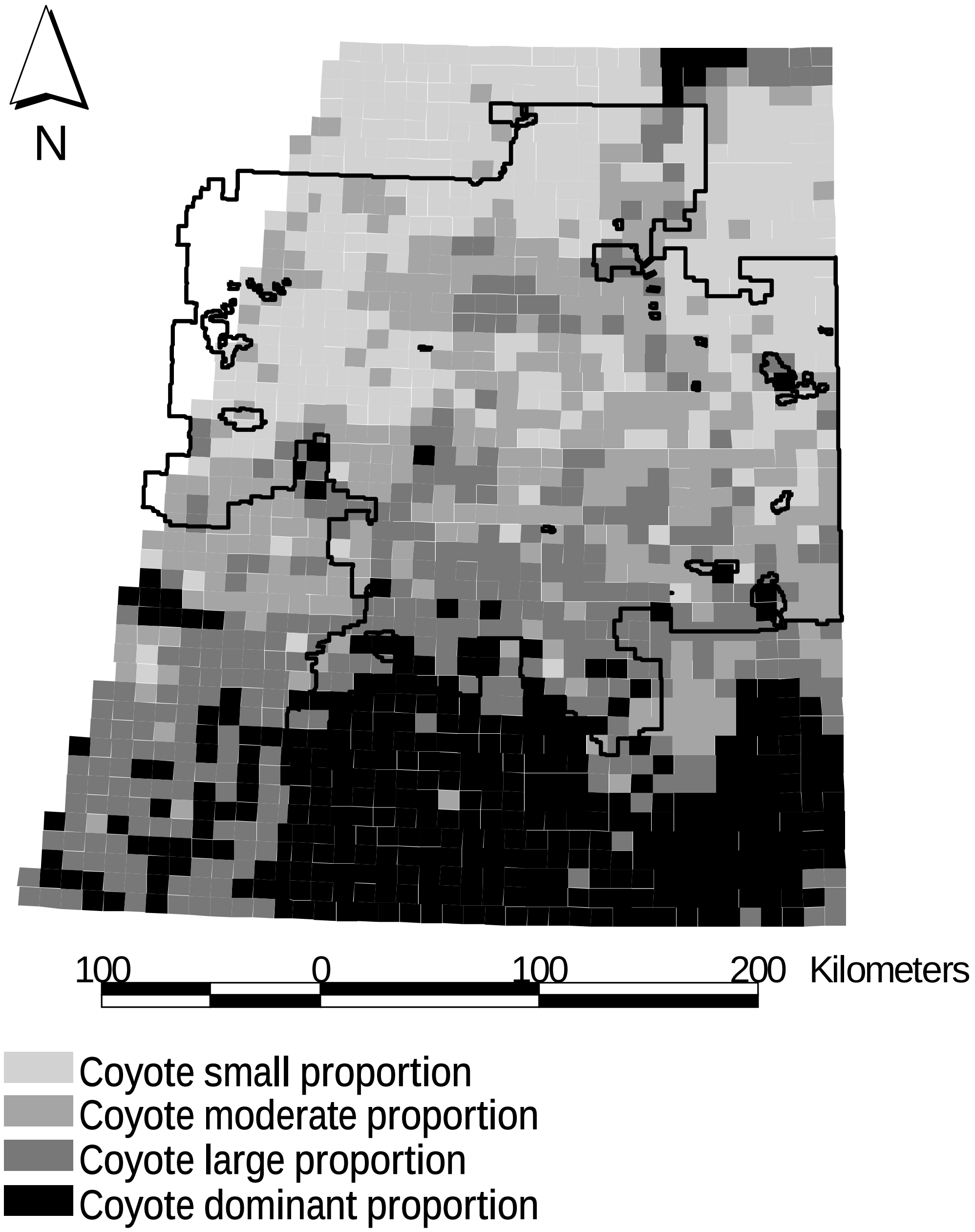


Figure 6 – Scatterplot showing relationship between IMI (Index of Mammalian Integrity) versus CEI (Cumulative effects index).

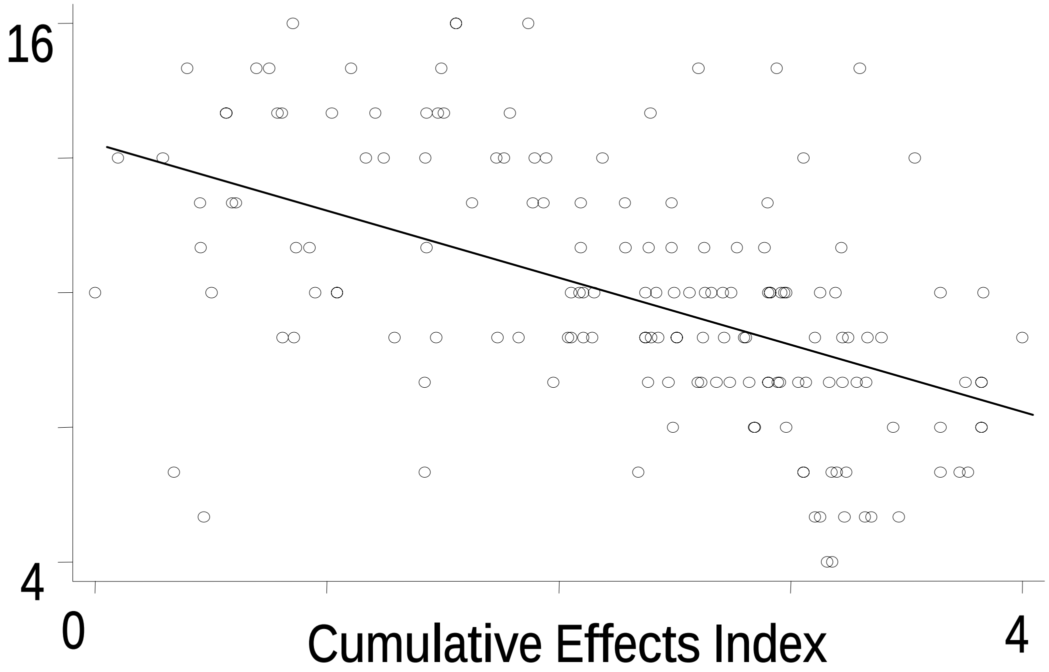
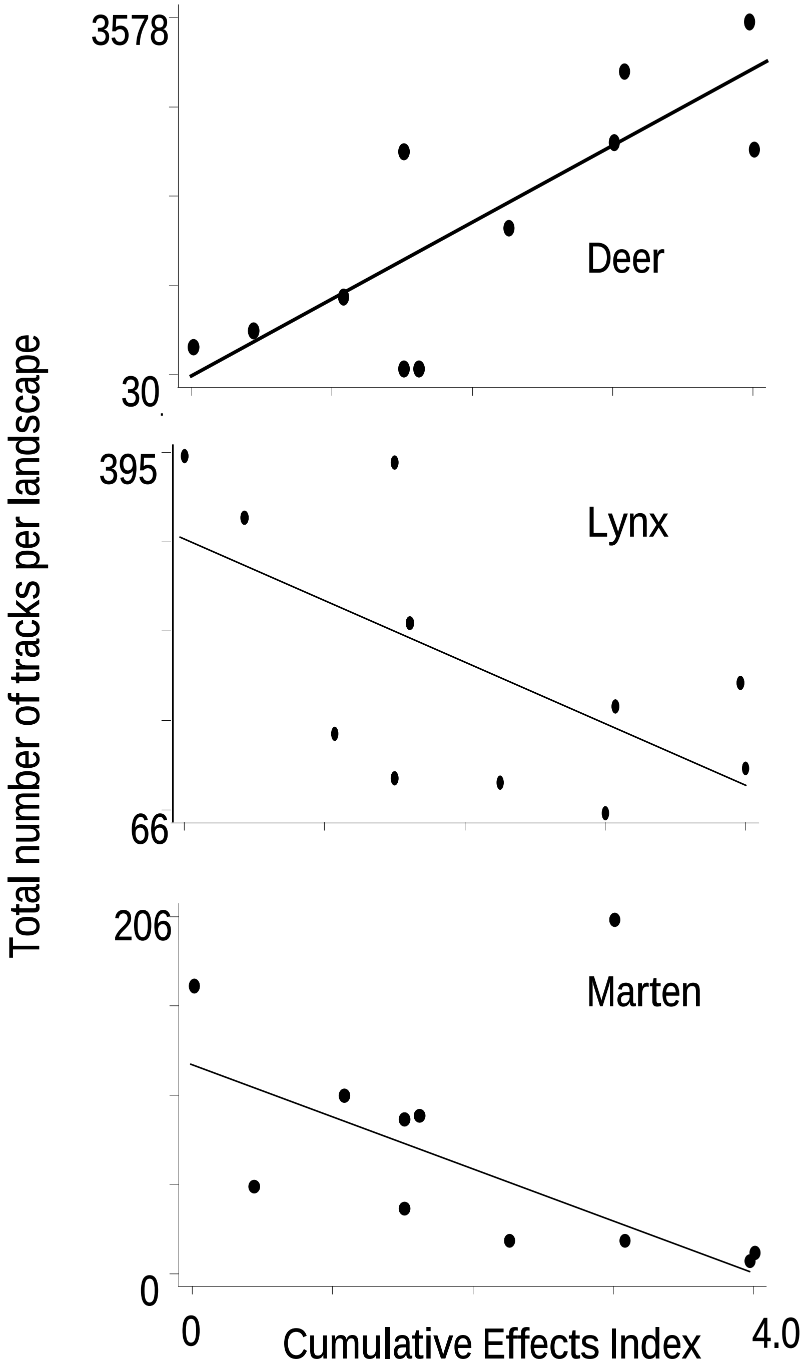


Figure 7 – Scatterplot showing relationship between raw abundance of deer, lynx, and marten versus CEI at landscape scale (12 townships pooled).



6.) **Literature Cited:**

Barbour, M.T. J.B. Stribling, and J.R. Karr. 1995. Multimetric approach for establishing biocriteria and measuring biological condition. In: Davis, W.S., and T.P. Simon (Eds.). Biological assessment and criteria: Tools for water resource planning and decision-making. Lewis Publishers, London. Pp 63-77.

Cameron, A.C., and P.K. Trivedi. 1998. Regression analysis of count data. Cambridge University Press. New York.

Canterbury, E.G., E.T. Martin, R.D. Petit, and J.L. Bradford. 2000. Bird communities and habitat as ecological indicators of forest condition in regional monitoring. Conserv. Biol. 14:544-558.

Carroll, C, R.F. Noss, and P.C. Paquet. 2001. Carnivores as focal species for conservation planning in the Rocky Mountain region. Ecol. Appl. 11: 961-980.

Carroll, C., W.J. Zielinski, and R.F. Noss. 1999. Using presence-absence data to build and test spatial habitat models for the fisher in the Klamath Region, U.S.A. Cons. Biol. 13: 1344-1359.

Carver, S. 1996. Mapping the wilderness continuum using raster GIS. in S.Morain & S.Lopez-Baros (eds) Raster imagery in Geographic Information Systems. OnWord Press, New Mexico, 283-288.

Forman, T.R., and D.R. Deblinger. 2000. The ecological road-effect zone of a Massachusetts (U.S.A.) suburban highway. Cons. Biol. 14:36-46.

Gompper, M.E. 2002. Top carnivores in the suburbs? Ecological and conservation issues raised by colonization of northeastern North America by coyotes. Bioscience 52:185-190.

Hanley. T.A. 1996. Potential role of deer (Cervidae) as ecological indicators of forest management. For. Ecol. Manage. 88: 201-214

Hardin, J. and J. Hilbe. 2001. Generalized linear models and extensions. Stata Press, College Station, Texas.

Hobson, K.A., E.M. Bayne, and S.L. Van Wilgenburg. 2002. Large-Scale conversion of forest to agriculture in the boreal plains of Saskatchewan. Conserv. Biol. 16:1530-1541.

Karr, J.R., 1987. Biological monitoring and environmental assessment: A conceptual framework. Environ. Manage. 11:249-256.

Landres, P.B., J. Verner, and J.W. Thomas. 1988. Ecological uses of vertebrate indicator species: A critique. Conserv. Biol. 2:316-329.

Lausch A. and F. Herzog. 2002. Applicability of landscape metrics for the monitoring of landscape change: issues of scale, resolution and interpretability. Ecol. Ind. 2: 3-15.

Lindenmayer, D.B., R.B. Cunningham, C.F. Donnelly, R. Lesslie. 2002. On the use of landscape surrogates as ecological indicators in fragmented forests. For. Ecol 159:203-216.

Lopez, R.D. and M.S. Fennessy. 2002. Testing the floristic quality assessment index as an indicator of wetland condition. Ecol. Appl. 12:487-497.

Pitcher, T.J. 2001. Fisheries managed to rebuild ecosystems? Reconstructing the past to salvage the future. Ecol. Appl. 11:601-617.

Rempel, R.S., P.C. Elkie, A.R. Rodgers, and M.J. Gluck. Timber-management and natural-disturbance effects on moose habitat: Landscape evaluation. J. Wildlife Manage. 61:517-524.

Schmitz, O.J., P.A. Hambaeck, and A.P. Beckerman. 2000. Trophic cascades in terrestrial systems: A review of the effects of carnivore removals on plants. Am. Nat. 155:141-153.

Schneider, R.R., and S. Wasel. 2000. The effect of human settlement on the density of moose in northern Alberta. J. Wildlife Manage. 64:513-520.

Schneider, R. R., J. B. Stelfox, S. Boutin, and S. Wasel. 2003. Managing the cumulative impacts of land uses in the Western Canadian Sedimentary Basin: a modeling approach. Conservation Ecology 7(1): 8. [online] URL: http://www.consecol.org/vol7/iss1/art8

Villard, M-A, M.K. Trzcinski, and G. Merriam. 1999. Fragmentation effects on forest birds: Relative influence of woodland cover and configuration on landscape occupancy. Conserv. Biol. 13:774-783.

Wallace, J.B., J.W. Grubaugh, and M.R. Whiles. 1996. Biotic indices and stream ecosystem processes: Results from an experimental study. Ecol. Appl. 6:140-151.