

**Monitoring Landscape and Spatial Behavioural Outcomes of Large Scale
Forest Management for Boreal Caribou (*Rangifer tarandus caribou*) in Upland
and Lowland Forests of Ontario, Canada**

by

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Abstract

Habitat management is a key tool used to support species conservation and sustainable timber harvest on managed forests; however assessments of long-term management outcomes are rare. My study is a qualitative assessment of the outcomes of over 15 years of large scale forest management for boreal caribou in two unique Ontario, Canada landscapes: the northwestern Boreal Shield and the northeastern James Bay Lowlands. I assessed: (1) the realized outcomes of forest management direction with respect to the landscape scale structure and configuration of caribou habitat and harvested areas, and (2) changes in caribou spatial behaviour in response to forest management. I found patterns opposite to those predicted based on management recommendations in terms of harvest configuration and size, and observed no detectable changes in caribou habitat continuity or area that could be related to management application. Similarly, I observed behavioural patterns suggestive of maladaptive behavioural response to management outcomes, which could be potentially detrimental to caribou population viability. Overall, these results suggest that further investigation into the management of habitat for boreal caribou in Ontario is required. Moreover, these results highlight the importance of long-term monitoring for management practices in order to ensure management success.

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Table of Contents

Abstract.....	iii
Acknowledgements	iv
List of Figures.....	viii
List of Tables	x
1.0 General Introduction	1
1.1 Literature Cited.....	6
2.0 Large scale forest management planning for boreal caribou: are guidelines meeting predicted harvest and habitat outcomes?	9
2.1 Summary	10
2.2 Introduction	11
2.3 Methods	14
2.3.1. Study areas.....	14
2.3.2. Forest harvest assessment.....	16
2.3.3. Caribou habitat assessment.....	19
2.4 Results	23
2.4.1. Forest harvest.....	23
2.4.2. Caribou habitat	27
2.5 Discussion.....	33
2.5.1 Forest harvest.....	33
2.5.2 Caribou habitat	34
2.6 Conclusions	37
2.7 Literature Cited.....	38
Appendix 2.1	44
3.0 Spatial behavioural patterns as indicators of long-term management outcomes: a case study of boreal caribou in Ontario, Canada.....	45
3.1 Summary	45
3.2 Introduction	46
3.3 Methods	50
3.3.1 Study areas.....	50
3.3.2 Data processing.....	52
3.3.3 Home range size	53
3.3.4 Proximity index	54
3.3.5 Fidelity Index.....	56
3.3.6 Statistical assessments	57
3.4 Results	58
3.4.1 Home range size	58
3.4.2 Proximity index	60
3.4.3 Fidelity Index.....	63
3.5 Discussion.....	65
3.6 Literature Cited.....	69
Appendix 3.1	75
4.0 General Conclusions and Recommendations	79
Consideration 1: Spatial scales of observation	79
Consideration 2: Seasonal sensitivity of spatial behaviours.....	80

Consideration 3: Different behaviours may be sensitive to different aspects of disturbance 82
4.1 Literature Cited..... 85

List of Figures

- Figure 2.1.** Northeastern and northwestern study areas, created using the 100% Minimum Convex Polygons of documented boreal caribou (*Rangifer tarandus caribou*) radio-locations that overlapped between pre- and post-introduction of forest management guidelines for caribou.... 15
- Figure 2.2.** a) The change in harvest Mean Patch Size (Harvest MPS) through time in both the western (grey, dashed line) and eastern (black, solid line) study areas. Guideline introduction occurred in 1999 (black, dotted line). b) The cumulative sum of deviations (CUSUMs) from the overall MPS in the western study area through time. c) The cumulative sum of deviations from the overall MPS in the eastern study area through time. 24
- Figure 2.3.** a) The change in Patch Size Standard Deviation (Harvest PSSD) through time in both the western (grey, dashed line) and eastern (black, solid line) study areas. Guideline introduction occurred in 1999 (black, dotted line). b) The cumulative sum of deviations (CUSUMs) from the overall mean PSSD in the western study area through time. c) The cumulative sum of deviations from the overall mean PSSD in the eastern study area through time. 25
- Figure 2.4.** a) The change in Clumpiness Index (Harvest CI) through time in both the western (grey, dashed line) and eastern (black, solid line) study areas. Guideline introduction occurred in 1999 (black, dotted line). b) The cumulative sum of deviations (CUSUMs) from the overall mean CI in the western study area through time. c) The cumulative sum of deviations from the overall mean CI in the eastern study area through time..... 26
- Figure 2.5.** a) The change in harvest Mean Shape Index (Harvest MSI) through time in both the western (grey, dashed line) and eastern (black, solid line) study areas. Guideline introduction occurred in 1999 (black, dotted line). b) The cumulative sum of deviations (CUSUMs) from the overall MSI in the western study area through time. c) The cumulative sum of deviations from the overall MSI in the eastern study area through time. 28
- Figure 2.6.** The change in habitat metrics through time in eastern and western Ontario, fitted with segmented or linear regressions, where panel (a) shows total core area (TCA), panel (b) shows mean core patch size (MCA), and panel (c) shows area-weighted mean proximity index (AWMPI). Error bars represent standard errors. 29
- Figure 2.7.** The relationship between total harvest depletion and the difference between consecutive years (i.e., the TCA value from 1996 subtracted from the TCA value in 1995) of mean core patch size (MCA) and total core area (TCA) in the eastern study region. Data points that fell on or after the indicated breakpoint for each metric are labelled with their corresponding date..... 31
- Figure 2.8.** The total harvested area within both eastern and western study regions through time. The black vertical line indicates 2007, the time period at which area metric values were found to decrease in slope. 32

Figure 3.1. The study areas in eastern and western Ontario, Canada based on boreal caribou (*Rangifer tarandus caribou*) radio-locations (black dots) that occurred in pre- and post-guideline time periods. Only caribou who had locations that occurred below the northern limit of forest management units (FMU) were assessed..... 51

Figure 3.2. A comparison of least square mean home range size of boreal caribou (*Rangifer tarandus caribou*) between the eastern and western study areas in the early period (black) and late period (grey). Error bars represent standard error. 59

Figure 3.3. Comparison of the least square mean proximity index of boreal caribou (*Rangifer tarandus caribou*) between the eastern and western study areas in the early period (black) and late period (grey) at the population and individual annual home range scale during summer and winter seasons. Decreasing proximity index values suggest increasing caribou proximity to harvests when compared to expected distances. Error bars representing standard error..... 61

Figure 3.4. A comparison of the mean fidelity index of boreal caribou (*Rangifer tarandus caribou*) to summer ranges between the eastern and western study areas in the early period (black) and late period (grey). Increasing fidelity index values suggest increases in fidelity behaviour relative to expected distances. Error bars represent standard error. 64

List of Tables

Table 2.1. Class-based metrics calculated for forest harvests within the eastern and western study areas between 1991 and 2011. The descriptions of patch metrics are modified from McGarigal and Marks (1995) and McGarigal et al., (2012).	18
Table 2.2. A description of expected cumulative sum control chart (CUSUMs) output interpretations for measured landscape metrics (Berthouex and Hunter, 1981).....	19
Table 2.3. Class-based metrics calculated for caribou habitat within eastern and western study areas between 1995 and 2011. Descriptions of each metric are alterations of those of McGarigal and Marks, (1995) and McGarigal et al., (2012).	21
Table 3.1. Predictions and their rationale for changes in each measured spatial behavioural metric in relation to management application.....	49

1.0 General Introduction

Anthropogenic disturbances within forested regions can pose a conservation challenge for many forest-dependent wildlife species. Wildlife habitat management is a widely adopted tool used to reconcile competing demands for economic resource development and the conservation of biodiversity, under the assumption that the provision of habitat will sustain wildlife populations. Knowledge of the outcomes and effectiveness of management actions is essential to inform further management decisions and ultimately protect wildlife species; however, monitoring programs to assess management outcomes are rare.

An iconic species of Canada's boreal forest is the boreal caribou (*Rangifer tarandus caribou*), which has experienced significant declines across Canada, leading to their designation as "threatened" under Canada's Federal Government's Species at Risk Act (2004; Mallory and Hillis, 1998; COSEWIC, 2011; Festa-Bianchet et al., 2011). In the province of Ontario, range recession of boreal caribou has been associated with large scale forest conversion caused by advancing forest harvesting (Schaefer 2003). Forest harvests target the old-growth conifer forests on which caribou depend (Rettie and Messier, 2000; Brown, 2005; O'Brien et al., 2006; Hins et al., 2009), leading to a shift in vegetation composition that reduces caribou habitat availability, while creating deciduous and early successional habitat ideal for the alternate prey of caribou's predators, such as moose (*Alces alces*) and white-tailed deer (*Odocoileus virginianus*; Potvin et al., 2005; Brown, 2011; Latham et al., 2011). This creates an increased availability of prey for wolves (*Canis lupus*), leading to increased predator abundance within caribou range and larger wolves that specialize on moose (Lesmerises et al., 2012; Wiwchar and Mallory, 2012). Early successional forests associated with forest harvesting also attract black bears (*Ursus americanus*), a predator of caribou calves (Brodeur, 2008; Dussault et al., 2012; Pinard et al.,

2012). Increased abundances of predators and alternate prey are not limited to the boundaries of a harvested area, meaning harvests surrounding a profitable habitat patch can decrease that patch's likelihood of use and cause boreal caribou to use potentially sub-optimal habitat where there is a lower abundance of forest harvests (Hins et al., 2009; Lesmerises et al., 2013). As such, habitat loss from forest harvesting often extends well beyond harvest block limits (Dyer et al., 2001; Schaefer and Mahoney, 2007; Vors et al., 2007; Bowman et al., 2010; Moreau et al., 2012).

Considerable effort has been placed on habitat management to address caribou conservation concerns in Ontario, which introduced forest management guidelines for the conservation of boreal caribou in 1999 for use in the northwest (Racey et al., 1999). The managing principles of these guidelines were also applied to northeastern Ontario, though altered due to differences in landscape ecology and local research into caribou ecology (Wilson, 2000; Proceviat, 2003; Brown, 2005; Brown et al., 2007). Guidelines were intended to sustain caribou populations within managed areas by maintaining large tracks of continuous caribou habitat at large landscape scales using a mosaic approach. Habitat mosaics reflect a variety of large habitat patches of varying stand age, which are distributed among forest management areas, meant to mimic the stand structure created by natural disturbance fluctuations experienced during natural fire cycles (Racey et al., 1999). Within the habitat mosaic, certain areas were designated for harvest, while other areas which contained or would soon contain ideal caribou habitat remained unaltered. For instance, when considering winter habitat, older, unused habitat that had declined or would soon decline in habitat value was recommended to be allocated for harvest, while high quality habitat or younger forest, which would provide high quality habitat in the near future, was retained.

Boreal caribou employ a “spacing out strategy” used to help separate themselves from predators, alternate prey and conspecifics (Bergerud & Page, 1987; Siep, 1991) leading to very large individual ranges (Brown et al., 2003; Ferguson and Elkie, 2004). In order to maintain large tracts of habitat required by caribou within the mosaic, guidelines suggested that both forest harvests and maintained tracts of old growth forests be greater than 10,000 ha in area based on the average forest fire size within northwestern Ontario (Racey et al., 1999). Similarly, management planning for caribou should be based over large temporal (≥ 80 years) scales to ensure continuous caribou habitat supply throughout each planning rotation. Because caribou can range over multiple Forest Management Units (FMUs), guidelines suggest coordination of FMU management plans in order to ensure caribou habitat connectivity across managed areas. This suggests that a post-guidelines harvested landscape should reflect large harvest patches aggregated on the landscape as well as large interconnected patches of caribou habitat.

Because wildfires are a common disturbance within the boreal forest (Johnson and Rowe, 1975), forest managers are encouraged to emulate natural disturbance patterns (Government of Ontario, 1994). As such, caribou guidelines recommend that managers coordinate management planning based on the instruction of the natural disturbance emulation guide released in 2001 (Racey et al., 1999; OMNR, 2001). This guide directs managers on stand and site recommendations for forest harvests, suggesting that managers emulate residual stand structure of wildfire disturbances within the boreal forest. Reflective of this, forest harvests should increase in size variation as well as shape complexity, with harvest edges following the natural contours of the landscape and large insular patches left to assist with forest regeneration.

Although the inclusion of caribou management guidelines are assessed at the forest management planning level, landscape outcomes of management practices have yet to be evaluated. I conducted a qualitative assessment on 15 years of forest management application on landscape configuration and boreal caribou spatial behaviours following guideline introduction in 1999. Differences in the functional response of animals to varying local environmental conditions are commonly observed (e.g., Watson et al., 2005; Environment Canada, 2011) and may reveal differences in the impacts of disturbance and forest management practices between regions. Therefore, I conducted my assessments in two separate study areas in northeastern and northwestern Ontario. Ontario is divided by two ecozones: the Hudson Plain region and the Boreal Shield. The study area in northwestern Ontario is comprised solely of the Boreal Shield ecozone, while northeastern Ontario is comprised of both the Boreal Shield and Hudson Plain region, creating two unique landscapes.

My thesis is divided into two chapters. The first chapter evaluates the landscape outcomes of forest management by assessing changes in the shape, size and configuration of forest harvests and caribou habitat using landscape metrics to detect changes in pattern indicative of management application. Based on guideline recommendations, I predicted that forest harvests would increase in size, size variation, and shape complexity in order to better mimic natural disturbances within the boreal forest (Racey et al., 1999; OMNR, 2001). Following the mosaic approach, I anticipated that harvest patches would increase in aggregation on the landscape, while caribou habitat would increase in patch size and connectivity (Racey et al., 1999).

In the second chapter, I assess the direct outcomes of guidelines on caribou by observing changes in caribou spatial behaviours between pre- and post-guideline introduction periods. Changes in spatial behaviours were predicted to be early warning indicators of potential changes in caribou population viability in response to changing management within caribou range (Tuomainen and Candolin, 2011). Boreal caribou have been shown to space away from disturbance (Vistnes and Nellmann, 2008); however, as the level of fragmentation increases on the landscape, caribou can become ‘trapped’ in smaller patches of habitat surrounded by harvests, decreasing their distances to disturbance and home range size (Smith et al., 2000; Courtois et al., 2007; Lesmerises et al., 2013; Beauchesne et al., 2014). Due to the high level of disturbance already associated with my study landscapes, I predicted that caribou would increase their distance from harvests and caribou home range size would not decrease, if management was effective at separating disturbance and habitat patches on the landscape. Further, increasing levels of disturbance have been associated with decreased caribou fidelity to summer ranges (Courtois et al., 2007; Faille et al., 2010); thus, I predicted that the caribou fidelity would increase if guidelines were effective at reducing the level of disturbance to caribou.

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2.0 Large scale forest management planning for boreal caribou: are guidelines meeting predicted harvest and habitat outcomes?

2.1 Summary

Sustainable forest management is used to meet wildlife habitat targets in many jurisdictions. Although a growing trend in forest management is to emulate natural disturbances during planning, monitoring of the application and effectiveness of this practice is lacking. I evaluated changes in landscape characteristics in both northwestern and northeastern Ontario over a 17-year period to assess the landscape outcomes of forest management planning in relation to management guidelines for boreal caribou (*Rangifer tarandus caribou*). Characteristics of harvest blocks and sparse and dense conifer forests, representative of caribou habitat, were measured for each year using a Landsat land cover classification updated with annual disturbance information. In both locations, forest harvest blocks were expected to increase in size, size variation, aggregation and shape complexity following guideline implementation to mimic natural disturbances within these regions. Additionally, caribou habitat patches were expected to increase in size and landscape continuity was expected to increase. Cumulative sum control charts and segmented regressions were used to assess temporal trends in landscape characteristics relevant to caribou. Results indicated that in both eastern and western Ontario, mean harvest block area, mean area variation and harvest aggregation declined following guideline introduction. This was contrary to the outcomes I expected from guide implementation, which was to create large harvests which would mature into large even-aged blocks of caribou habitat. In eastern Ontario, the decrease in habitat patch size through time declined and was associated with lower total forest harvest and decreased lumber exports. No changes in caribou habitat patch proximity were detected. These results suggest that guidelines employed during forest management planning in my study period did not have their expected outcomes on harvest

block structure or caribou habitat at large spatial scales. Disturbance from fire and the general downturn of the forest industry were suspected to be more influential drivers of the patterns I observed in habitat conditions. Furthermore, these results highlight the importance of long-term monitoring to evaluate forest management outcomes to support biodiversity conservation.

2.2 Introduction

In recent years, the goals of forest management direction have transitioned from a focus on sustainable wood supply to a broader range of forest resource values, including social, cultural, economic and ecological integrity of forests (Lindenmayer et al., 2000; Perera and Euler, 2000; Siry et al., 2005). This shift in management attention has led to the creation of novel, multi-objective management strategies in many jurisdictions; however, the application and success of these strategies are rarely monitored, particularly over the long-term (Simberloff, 2001; Schulte et al., 2006).

Forest management often focuses on the management of wildlife habitat as a means to support sustainability objectives for wildlife. Goals for habitat management can vary from preserving or creating diverse habitat types (Buffum et al., 2011; Hargreaves-Allen et al., 2011) to focusing on the habitat requirements of specific species as a way to maintain ecological complexity (Bro et al., 2004; Fuentes-Montemayor et al., 2010; Bergman et al., 2014; Ferreira et al., 2014). Umbrella species, whose conservation is anticipated to apportion protection to other co-occurring species, are commonly suggested as focal points for management efforts (Roberge and Angelstam, 2004; Branton and Richardson, 2011).

Across Canada, a common umbrella species is the boreal caribou (*Rangifer tarandus caribou*), which has received a great deal of attention due to its vulnerability to extirpation following forest landscape alterations that lead to shifts in forage availability and predator-prey balance (Seip 1992, Mallory and Hillis, 1998, Brown et al., 2007; Courtois et al., 2007; Vors et al., 2007; Wittmer et al., 2007; Festa-Bianchet et al., 2011; Environment Canada, 2011; Wiwchar and Mallory, 2012). Early successional forests, including those associated with forest harvesting, favour ungulates such as white-tailed deer (*Odocoileus virginianus*) and moose (*Alces alces*), causing increased predator abundance within caribou range (Seip, 1992; Potvin et al., 2005; Latham et al., 2011; Lesmerises et al., 2012). Forest regeneration also provides habitat for black bears (*Ursus americanus*), a major predator of caribou calves (Brodeur et al., 2008; Pinard et al., 2012). Within Ontario, Canada, boreal caribou have lost nearly half of their historical range, with the current southern limit corresponding to the northern limit of commercial timber harvest (Mallory and Hillis, 1998; Schaefer, 2003), leading to increasing demands for management considerations of caribou (Armstrong, 1998).

Typical habitat management practices involve affecting the physical structure and successional dynamics of vegetation to favour a particular habitat type (Ausden, 2007). Across North America, a common objective is to emulate natural disturbance regimes during management planning (Perera and Euler, 2000; Long, 2009). The primary goal is to maintain historical diversity by mimicking the forest ecosystem structure and function which follow natural disturbances (Bergeron et al., 2002; Perera and Buse, 2004; Brown and Mallory, 2007; Long, 2009). Forest management guidelines for Ontario boreal caribou recommended following a mosaic landscape design based on natural fire cycles during my study period (Racey et al., 1999). This approach is intended to maintain large, interconnected patches of mature conifer

forest (caribou habitat), shown to be strongly selected for by caribou (O'Brien et al., 2006; Brown et al., 2007; Hins et al., 2009). Forest management selectively assigned areas for intensive harvest, creating aggregations of harvest patches meant to mimic large scale natural fires. Later, these patches were intended to regenerate into large even-aged blocks of caribou habitat. At the same time, other areas which already contained, or would soon contain, ideal patches of caribou habitat would remain unharvested for caribou use. Over an extended temporal scale, this tactic attempts to ensure that large interconnected patches of conifer forest are maintained at all times during each management rotation, while creating disturbances which mimic natural forest fires.

Habitat guideline inclusion within forest management plans is assessed and approved by the Ontario Ministry of Natural Resources and Forestry (OMNRF) prior to management plan application (Government of Ontario, 1994); however the actual landscape outcomes of forest management plans over the long-term have yet to be evaluated. My study aims to assess the application of forest management guidelines by observing temporal trends in caribou habitat and harvest block metrics. Assessments were made in Ontario's northwestern Boreal Shield and northeastern James Bay Lowland regions, allowing me to assess the differences in management outcomes in areas with different natural disturbance regimes. Selecting a broad area spanning multiple management units, I was able to evaluate large landscape changes at a scale consistent with population range use by caribou, as well as assess the cumulative outcomes of management application among multiple forest management planning units. I anticipated that implementation of caribou management guidelines would result in larger mean harvest block areas, larger mean harvest block area variations, greater shape complexity of harvest blocks, and increased aggregation of harvests over the landscape following guideline introduction (in 1999; Table 2.1).

Furthermore, I anticipated an increase in the protection of caribou habitat patch core areas and proximity following 1999 (Table 2.2), as outlined in caribou forest management guidelines.

2.3 Methods

2.3.1. Study areas

Forest management practices were assessed in two unique landscapes within caribou range in Ontario. A study area of 28,668 km² was selected in the northeastern James Bay Lowlands Region, while a study area of 103,683 km² was selected in the Northwestern Boreal Shield Region (Figure 2.1). Study area boundaries were defined using 100% Minimum Convex Polygons (MCPs) from the telemetry locations of boreal caribou (52 individuals for northeastern Ontario and 91 individuals for northwestern Ontario) collected by the Ontario Ministry of Natural Resources and Forestry (OMNRF) between 1995 and 2013. Telemetry locations, rather than FMUs, were used to designate study areas to support subsequent analysis of animal spatial behaviours in relation to management landscape outcomes.

The northeastern Ontario study area is flat, with little relief and a mean elevation of 250 m (Boissonneau, 1996; Belleau et al., 2011). Poorly drained soils and humid boreal climate make the area susceptible to paludification leading to an abundance of peatlands (Jeglum and He, 1995; Brown et al., 2003; Belleau et al., 2011). Mono-specific black spruce (*Picea mariana*) stands are a dominant habitat characteristic throughout the region (Brown et al., 2003; Belleau et al., 2011). The influence of James and Hudson Bays creates an altered humid continental climate regime displaying many maritime climate characteristics (Baldwin et al., 2000). Fire cycles in this region are estimated at 398 years and mean stand age is estimated at 148 years (Bergeron et

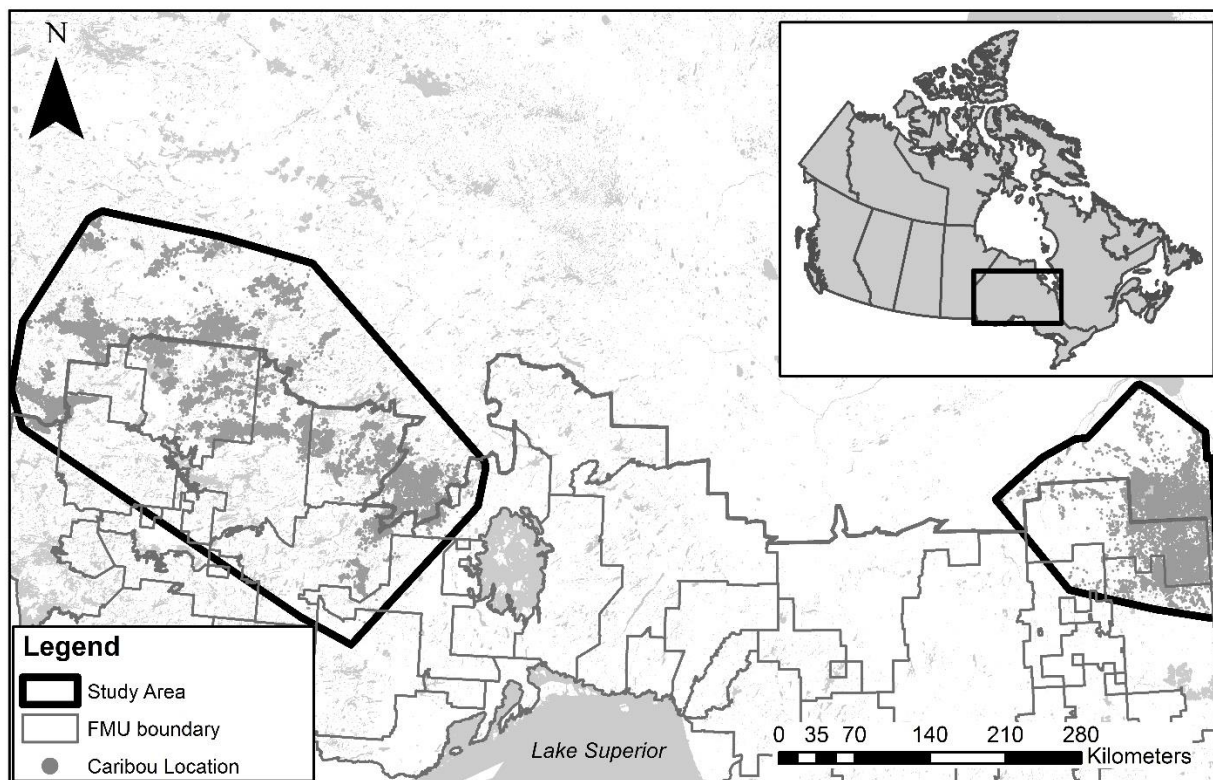


Figure 2.1. Northeastern and northwestern study areas, created using the 100% Minimum Convex Polygons of documented boreal caribou (*Rangifer tarandus caribou*) radio-locations that overlapped between pre- and post-introduction of forest management guidelines for caribou.

al., 2004).

The northwestern study area in the Boreal Shield ecozone contains rolling hills, staggered by depressions (Sims et al., 1995). Soils tend to be well drained, although peatlands and bogs are still common between hills (Zoladeski and Maycock, 1990; Watson 1997). Differing from northeastern Ontario, the dominant stand type is Jack pine (*Pinus banksiana*; Ferguson and Elkie, 2004) and the climate regime matches the majority of Ontario as humid continental (Baldwin et al., 2000). The natural wildfire cycle in this region is 74 years (Li and Perera, 1997) and a relatively low mean stand age of 99 years (Zoladeski and Maycock, 1990).

2.3.2. Forest harvest assessment

Harvest blocks made each year between 1991 and 2011 were assessed as polygon shape files in ArcGIS v. 10. Harvests made prior to 1999 were included to allow comparisons of the structure of post-guideline introduction harvests to those made prior to guideline release. As forest management in Ontario uses 5-year operating plans for forest harvests (OMNR, 2001; OMNR, 2009), this selected time range allowed for the inclusion of approximately two operational periods before and after guideline introduction. Along with annual assessments, annual harvest polygons were also aggregated into a 5-year grouping on a sliding scale (e.g. the 1991 to 1995 aggregated grouping would represent 1995 harvest in my analysis). This was assumed to give a more accurate representation of the planned outcomes of harvest operations, as the intended area of a harvest patch may have only been realized after multiple years of harvesting within one 5-year management plan. Further, this would remove extraneous annual variation in the data set due to short-term operational decisions. Only harvest polygons that were within 50 m of each other were aggregated within 5-year groupings. This distance was the added

distance of the standard size of a forestry road (approximately 20 m in width based on OMNRF forestry road guidelines; OMNR 2000a) and the distance that may have potentially been lost along harvest patch edges during conversions between raster and polygon files in ArcGIS (30 m of width).

Patterns in forest harvest associated with the 1999 introduction of caribou forest management guidelines were assessed using FRAGSTATS 4.2 (McGarigal et al., 2012). FRAGSTATs offers a wide variety of metrics that can be assessed at the patch, class or landscape level. I selected metrics at the class level to observe the configuration of harvest patches in relation to one another. A brief description of class metrics, as well as the reasoning behind their use, is provided in Table 2.1.

Changes in harvest block metrics were graphically assessed for temporal trends. Cumulative sum control charts (CUSUMs) were used to visually amplify the changes in trend of each plotted statistic using the cumulative sum of deviations from the overall mean of each metric during my study period. CUSUM calculations followed the equation:

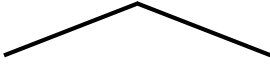
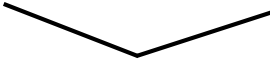

$$S_t = (Q_t - k) + S_{t-1}$$

where S_t represents the cumulative sum at time t , Q_t represents the value of the plotted statistic at time t , and k represents the overall mean of statistic Q . It is important to recognize that the outputs seen in CUSUM plots are not a direct representation of temporal trends, but rather how these trends relate to a target value (Table 2.2).

Table 2.1. Class-based metrics calculated for forest harvests within the eastern and western study areas between 1991 and 2011. The descriptions of patch metrics are modified from those of McGarigal and Marks (1995) and McGarigal et al., (2012).

Metric	Abbrev.	Description/Predicted Responses
Mean Patch Size (ha)	MPS	Mean harvest size in hectares. Harvest size was expected to increase following 1999 based on guideline recommendations intended to decrease the level of habitat fragmentation created by many small harvests as well as to better mimic natural disturbances (OMNR, 2001; Racey et al., 1999)
Patch Size Standard Deviation	PSSD	Harvest size variation. Mean patch size standard deviation was expected to increase following 1999 to better mimic natural disturbances within Ontario (OMNR, 2001)
Mean Shape Index	MSI	Mean shape complexity of harvests. MSI is similar to mean perimeter-area ratio, however, is standardized to remove size bias. An output equal to 1 indicates a square shape, while values increasing from 1 indicate increasing shape complexity. MSI was expected to increase following 1999 based on recommendations to mimic natural disturbances by adding insular patches within harvests and increasing harvest shape irregularity (OMNR, 2001).
Clumpiness Index	CI	Harvest aggregation on the landscape. CI uses a normalized index which depicts deviations from a normal distribution. Values range from -1 to 1, where a value of -1 indicates maximum disaggregation, 1 indicates maximum aggregation, and 0 indicates a random distribution. CI was anticipated to increase following 1999 as harvests would be contained within areas designated areas within the landscape mosaic (Racey et al., 1999).

Table 2.2. A description of expected cumulative sum control chart (CUSUMs) output interpretations for measured landscape metrics (Berthouex and Hunter, 1981).

CUSUM Output	Interpretation
	<p>Metric values were initially above the overall mean, and then shifted to below the overall mean at the time period indicated by the peak output value. This indicates a negative slope through time in the measured metric.</p>
	<p>Metric values were initially below the overall mean and then shifted to above the overall mean at the time period indicated by the peak output value. This indicates a positive slope through time in the measured metric.</p>
	<p>Metric values remain equal to the overall mean through time. This indicates zero slope in the measured metric.</p>

2.3.3. *Caribou habitat assessment*

Caribou habitat was assessed using a 25 x 25 m pixel Landsat thematic land cover classification for Ontario (OMNR, 2000b). I chose a Landsat thematic land cover instead of Forest Resource Inventory (FRI) data, as FRI data sets did not cover the full span of my study areas. Research contrasting FRI and Landsat classifications have found that both methods provide adequate detail to identify caribou habitat (Brown et al., 2006). Annual changes in habitat were assessed from 1995 to 2011 using ArcGIS v. 10 to update the 1995 land cover classification with yearly harvest blocks and burns supplied by the OMNRF and the Canadian Forest Service (2011). Boreal caribou habitat was designated as sparse and dense conifer forests, based on evidence from previous habitat selection studies that highlighted the importance of these cover types to caribou (Rettie and Messier, 2000; Ferguson and Elkie, 2004; James et al., 2004; Brown, 2005; McLoughlin et al., 2005; Brown et al., 2006, Hins et al., 2009). Both cover types were merged in ArcGIS in order to create one class representative of caribou habitat.

Caribou habitat was characterized using multiple class-based metrics, summarized in Table 2.3, using 'FRAGSTATS' version 4.2 (McGarigal et al., 2012). Two different sampling approaches were initially used to calculate metrics to ensure results were not sensitive to the method employed: 'no sampling', meaning that each metric was measured for the entire uninterrupted study area and 'uniform tile sampling', meaning that each metric was measured at smaller sub-landscapes or sampling windows within each study area.

Table 2.3. Class-based metrics calculated for caribou habitat within eastern and western study areas between 1995 and 2011. Descriptions of each metric are modified from those of McGarigal and Marks, (1995) and McGarigal et al., (2012).

Metric	Abbrev.	Description/Predicted Responses
Area-Weighted Mean Proximity Index	AWMPI	Indicates the aggregation of patches of the same class within a specified search radius. In this case, the search radius was set at 10 km, based on recommendations from Racey et al., (1999). A value of zero indicates there are no patches of habitat within the search radius. Increasing values indicate a greater number of neighbours and less fragmentation within the search radius, with larger neighbouring habitat patches being weighted more heavily. I predicted that the AWMPI rate of decline would decrease following 1999, based on recommendations to increase habitat connectivity, suggesting harvesting should remove small, isolated patches of habitat while maintaining large patches in close proximity (Racey et al., 1999).
Mean Core Patch Size	MCA	Indicates habitat core area, where depth-of-edge effect was 100 m based on the same rationale used in a previous caribou study (Hansen et al., 2001). I predicted that MCA rate of decline would decrease following 1999, based on recommendations that larger caribou habitat patches should be maintained, while smaller patches left by previous harvests should be removed (Racey et al., 1999).
Total Core Area	TCA	The total area of the landscape comprised of core area. I anticipated that the rate of decline in TCA would decrease following 1999, based on recommendations that large caribou habitat patches should be maintained, while smaller habitat patches with low core area should be removed (Racey et al., 1999).

Uniform tiling is used to reduce bias introduced by wide variation in patch sizes across the landscape, which can influence mean calculations of each class (McGarigal et al., 2012). Each sampling window is a block from a uniform grid laid out across the landscape. I selected a sampling window size to approximate caribou annual range size by using the 100% Minimum Convex Polygon (MCP) of 143 individuals' annual locational data collected over my study period. The square root of the average MCP area was then used as the side length for each square sampling window. Spatial extent (sampling window size) for landscape metrics has been shown to bias the outcomes of landscape assessments (Saura and Martinez-Millan, 2001). Therefore, although average annual ranges varied between study areas, an overall average for both regions was used to maintain comparability between each location's measured metrics.

Limitations in the data set meant that I was unable to account for forest maturation and successional dynamics that may have resulted in the generation of new caribou habitat, meaning degradation of caribou habitat was cumulative through time without being balanced by the addition of newly regenerated habitat. However, my habitat metrics' abilities to detect change should be dependent on what type of patches are being removed from the landscape, rather than habitat patch sample size. I therefore tested for changes in rate of decline through time in each metric to determine the impacts of caribou forest management guides introduced in 1999. I fit segmented regressions to my data using 'Joinpoint' software (Joinpoint Regression Program, 2014), which follows the methods of Kim et al., (2000). A single break point was expected to indicate a shift in slope, potentially suggestive of guideline implementation. Breakpoint location was limited so that breakpoints could not occur if there were fewer than two data points on either side of a potential breakpoint location.

Permutation tests were used to assess whether a segmented or linear regression best represented my data set, with a null hypothesis of zero breakpoints and an alternative hypothesis of one breakpoint. Autocorrelation in data was assessed by plotting autocorrelation functions using ‘acf’ in ‘R’ software environment (R version 3.0.1) and using the Durbin-Watson test from the ‘lmtest’ package (Zeileis and Hothorn, 2002). If autocorrelation was detected, an autocorrelation parameter was included in the piece-wise regression models in ‘Joinpoint’.

2.4 Results

2.4.1. Forest harvest

Because of strong similarities between annual and 5-year sliding scale harvest outputs, only results from the 5-year sliding scale assessments will be presented.

Harvest MPS, CI and PSSD in both the eastern and western study areas declined through time (Figs. 2.2, 2.3, 2.4). Short-term increases in these metrics associated with the time period of guideline introduction occurred in both study areas; however, this trend was not maintained in any case for extended periods. Similar trends between PSSD and MPS indicated that declining values in MPS were not due to increases in size variation, a possible confounding factor with guideline recommendations. Changes in the mean were also confirmed through observing similar patterns in median and area-weighted mean values for harvest patch size. Greater MPS was maintained in the eastern study area compared to the western study area following guideline introduction, while relatively similar PSSD and CI values were maintained in both study regions. However, while CI showed a decreasing trend for much of the study period, an increasing trend in the eastern study area was observed following 2008.

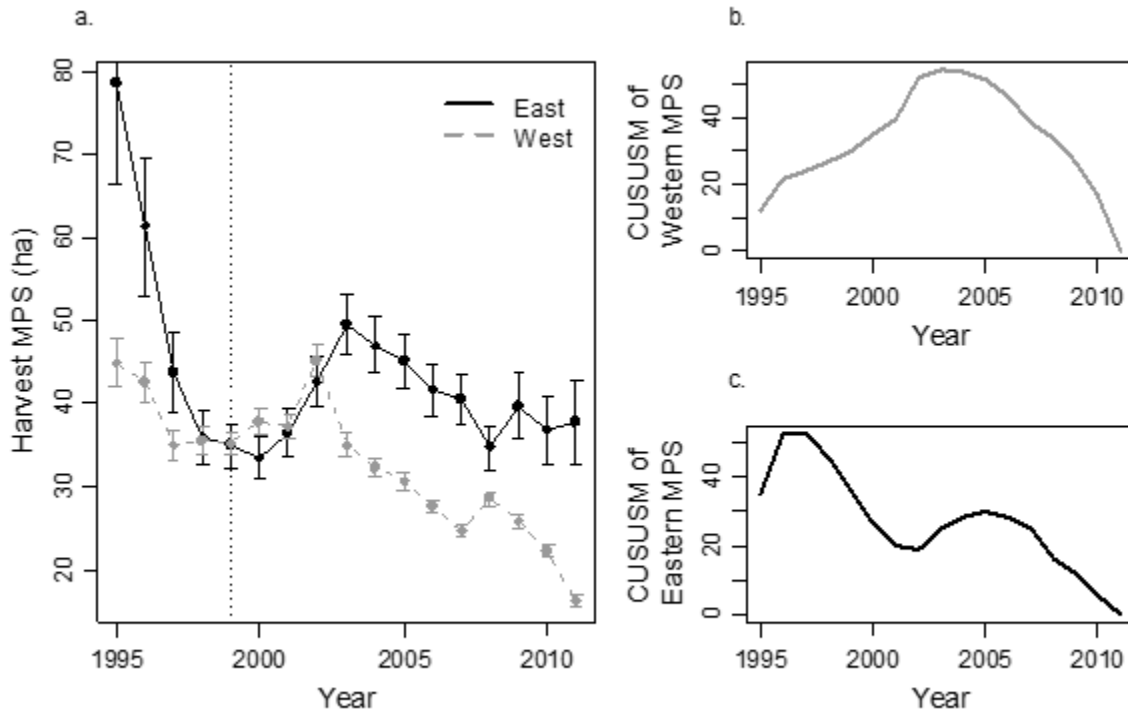


Figure 2.2. a) The change in harvest Mean Patch Size (Harvest MPS) through time in both the western (grey, dashed line) and eastern (black, solid line) study areas. Guideline introduction occurred in 1999 (black, dotted line). b) The cumulative sum of deviations (CUSUMs) from the overall MPS in the western study area through time. c) The cumulative sum of deviations from the overall MPS in the eastern study area through time.

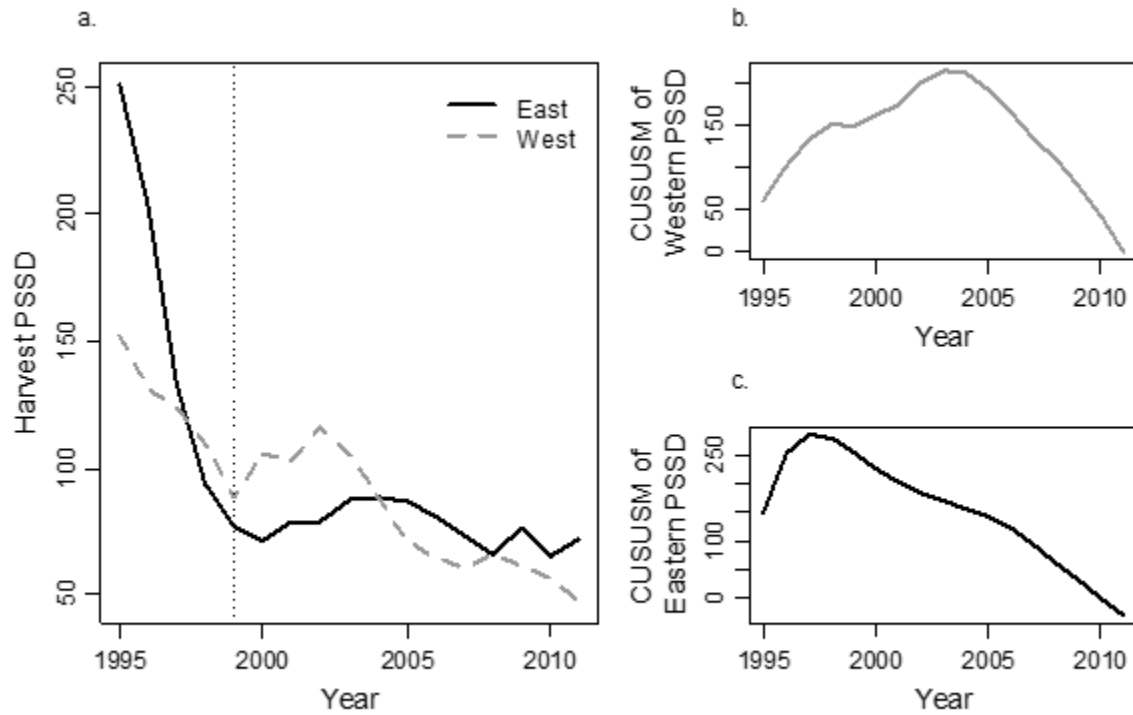


Figure 2.3. a) The change in Patch Size Standard Deviation (Harvest PSSD) through time in both the western (grey, dashed line) and eastern (black, solid line) study areas. Guideline introduction occurred in 1999 (black, dotted line). b) The cumulative sum of deviations (CUSUMs) from the overall mean PSSD in the western study area through time. c) The cumulative sum of deviations from the overall mean PSSD in the eastern study area through time.

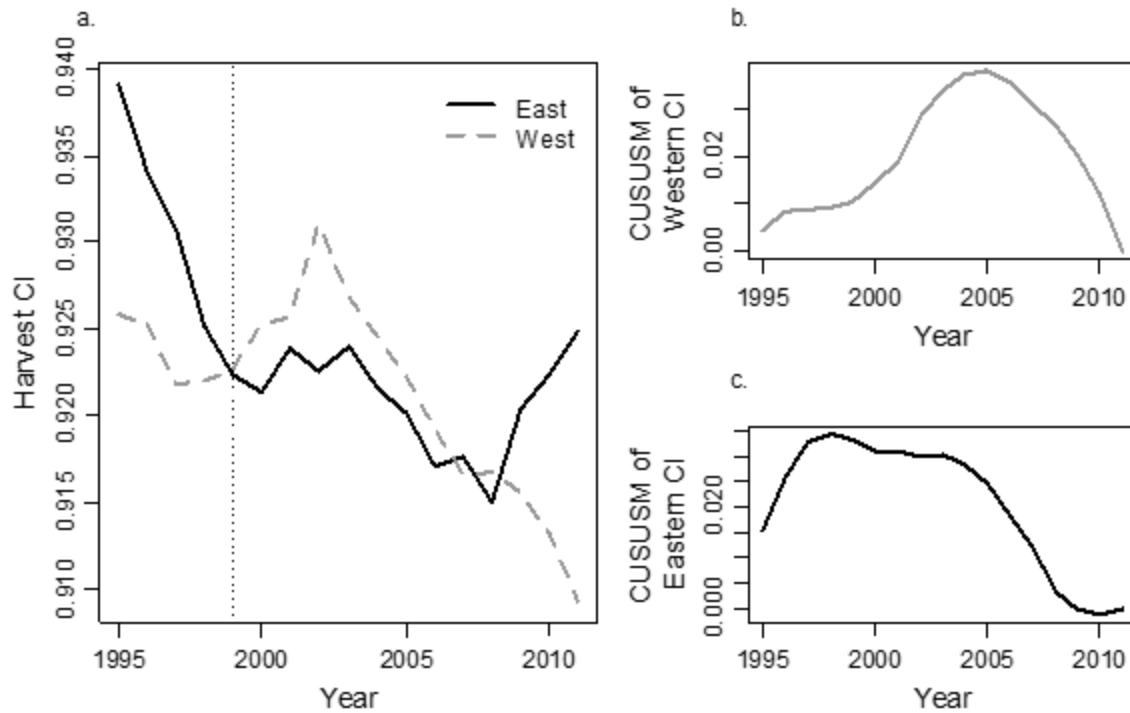


Figure 2.4. a) The change in Clumpiness Index (Harvest CI) through time in both the western (grey, dashed line) and eastern (black, solid line) study areas. Guideline introduction occurred in 1999 (black, dotted line). b) The cumulative sum of deviations (CUSUMs) from the overall mean CI in the western study area through time. c) The cumulative sum of deviations from the overall mean CI in the eastern study area through time.

Temporal harvest MSI values differed greatly between eastern and western study areas. The western study area's MSI values steadily declined through time (Figure 2.5), whereas in the eastern study area, MSI values increased following 2001.

2.4.2. *Caribou habitat*

Based on strong similarities in outputs between the two selected sampling methods, only 'no sampling' results will be presented.

In both MCA and TCA, a segmented regression modeled the data significantly better than a linear regression in the eastern study area ($p < 0.01$ in both cases), with breakpoints indicated in 2007 (95% CI between 2004, 2009, Figs. 2.6 a,b). MCA and TCA are both likely to be impacted by the total amount of forest harvest depletion. This aspect of forest harvesting was not included in caribou guideline recommendations, which instruct on changes in harvest configuration pattern rather than the volume of available wood (Racey et al. 1999), and as such needed to be considered a confounding factor for both metrics. Plotting the differences between succeeding years of both metrics against total forest harvest depletions for the eastern study area revealed an outlier in the first data point (the difference between 1995 and 1996). As this data point could have no relationship to changes in management practice as it occurs before guideline implementation, it was removed from analysis. The relationships between total forest harvest depletion and declines in habitat metric values were strong (MCA: $R^2=0.91$, $p < 0.01$, $t=11.49$; TCA: $R^2=0.90$, $p < 0.01$, $t=11.03$; Figure 2.4). As suggested by the timing of each indicated breakpoint, the differences between succeeding metric values after 2007 were associated with

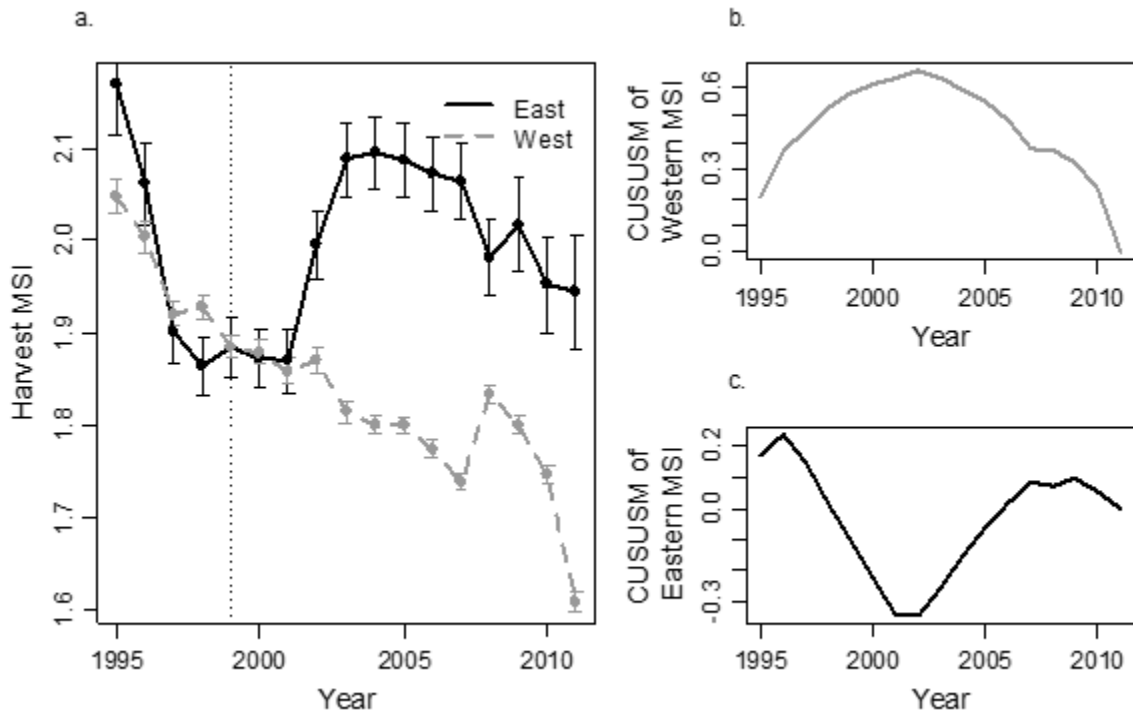


Figure 2.5. a) The change in harvest Mean Shape Index (Harvest MSI) through time in both the western (grey, dashed line) and eastern (black, solid line) study areas. Guideline introduction occurred in 1999 (black, dotted line). b) The cumulative sum of deviations (CUSUMs) from the overall MSI in the western study area through time. c) The cumulative sum of deviations from the overall MSI in the eastern study area through time.

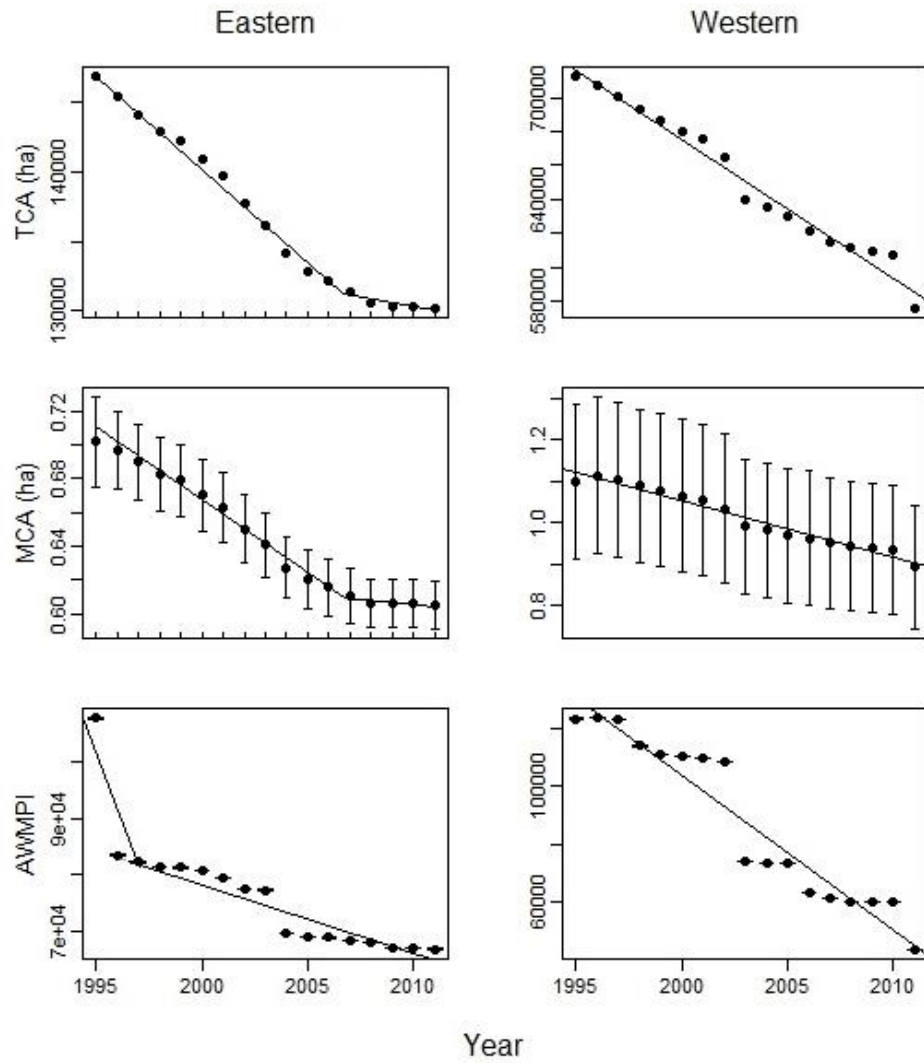


Figure 2.6. The change in habitat metrics through time in eastern and western Ontario, fitted with segmented or linear regressions, where panel (a) shows total core area (TCA), panel (b) shows mean core patch size (MCA), and panel (c) shows area-weighted mean proximity index (AWMPI). Error bars represent standard errors.

lower total harvested area (Figure 2.7). Assessment of the residual variation of this relationship revealed a random distribution, suggesting that decreases in total harvested area following 2007 were adequate to explain the observed declines in slope. This is supported by the decreased values of total harvested area following this period (Figure 2.8).

A linear model fit MCA, TCA and AWMPI data better than a segmented model in the western study area ($p=0.35$, $p=0.91$, $p=0.54$, respectively). Large decreases in metric value between consecutive years were consistently observed between 2002 to 2003 and 2010 to 2011 (Figure 2.6). These years were associated with large fire disturbances which occurred in large habitat patches (Appendix 2.1). Removing 2011 to try and reduce the impact of these sharp declines on breakpoint identification yielded no difference in results. However, upon visual assessment of temporal metric trends, a general decrease in slope corresponding with 2007 was observed, though no significant breakpoints were identified.

I found no relationship between total forest harvest depletion and western area metrics, (MCA: $p=0.86$, $t=0.18$; TCA: $p=0.76$, $t=-0.32$), though I did observe declines in total forest harvest depletion corresponding to roughly the same period as in my eastern study region (Figure 2.8).

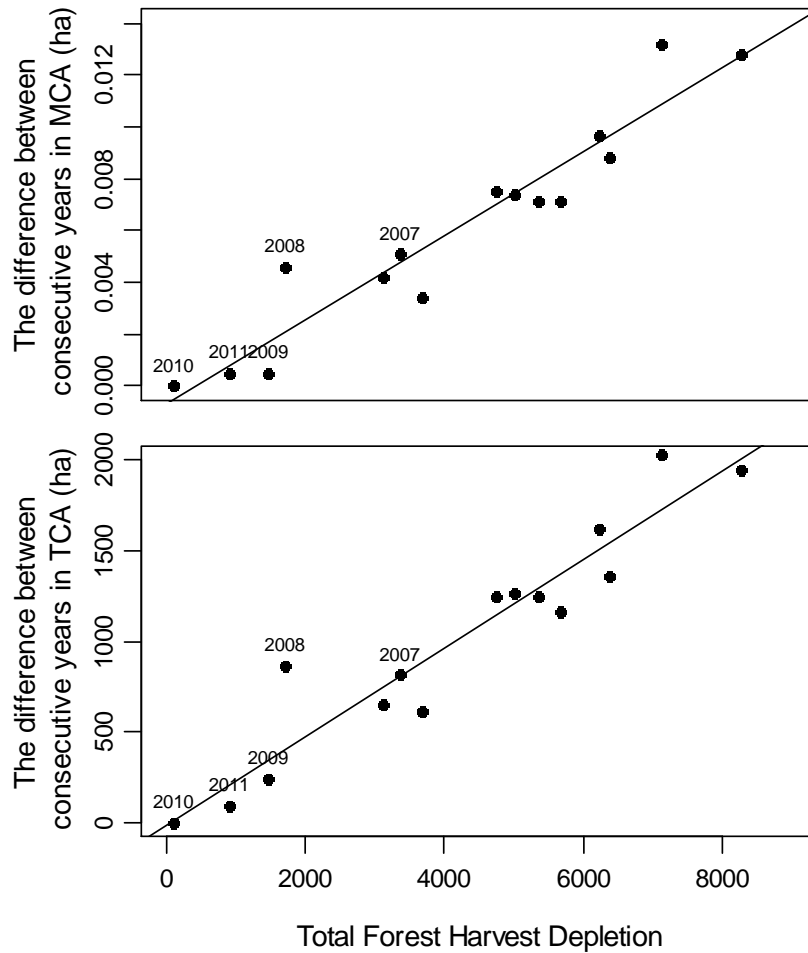


Figure 2.7. The relationship between total harvest depletion and the difference between consecutive years (i.e., the TCA value from 1996 subtracted from the TCA value in 1995) of mean core patch size (MCA) and total core area (TCA) in the eastern study region. Data points that fell on or after the indicated breakpoint for each metric are labelled with their corresponding date.

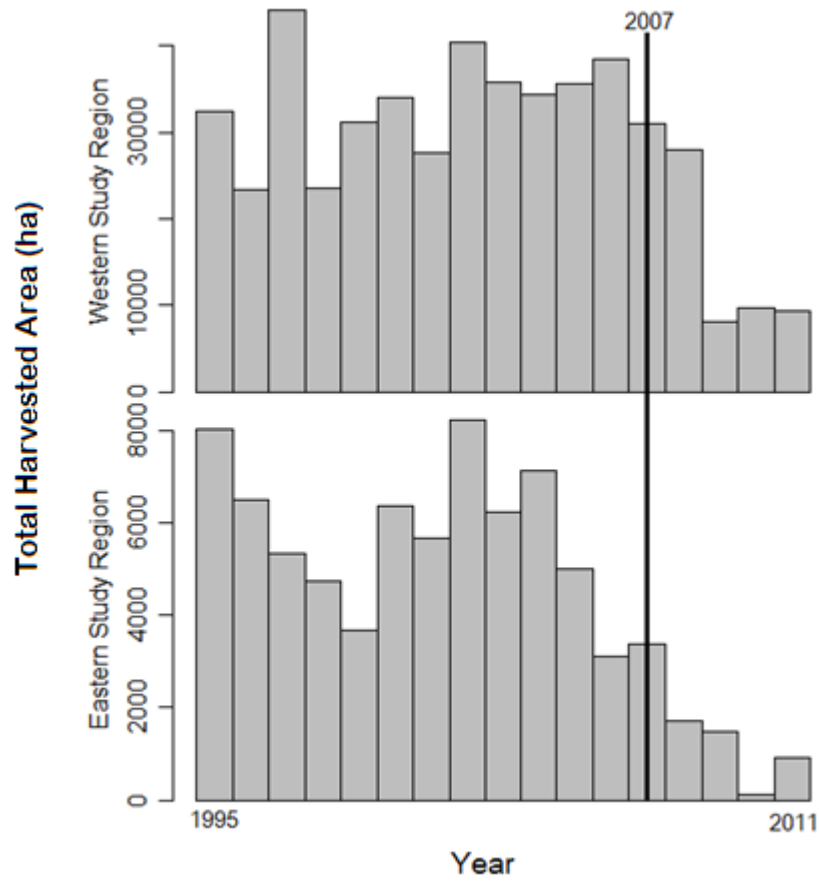


Figure 2.8. The total harvested area within both eastern and western study regions through time. The black vertical line indicates 2007, the time period at which area metric values were found to decrease in slope.

A segmented regression modeled AWMPI data in the eastern study area significantly better than the linear model ($p < 0.01$, Figure 2.6c), with a breakpoint indicated in 1997 (95% CI between 1997, 1999). This was caused by the relatively large drop in metric value between 1995 and 1996. These years are associated with relatively large fire disturbances for this study region.

2.5 Discussion

2.5.1 Forest harvest

My results support the conclusion that although some improvements in harvest block configuration were evident in the eastern study area, overall, trends during the monitored timeframe were not consistent with my predicted outcomes of guide implementation. In fact, many harvest configuration metrics indicated patterns opposite to those expected.

Trends in harvest block configuration were more consistent with my predicted landscape outcomes in the eastern study area than the west. For instance, the eastern study area maintained much greater MPS values than the western study area, although both showed an overall decline in MPS through time. Similarly, trends in MSI in the eastern study area were increasing, consistent with my predicted outcomes of guideline recommendations, while I observed decreasing trends in the west. I suggest that a possible reason for these landscape differences is a greater abundance of obstructing landscape features in the form of fens and bogs in the east (Jeglum and He, 1995; OMNR, 2000b), which would result in different distributions of harvestable wood and could thus influence harvest pattern. Similarly, obstructing landscape features could have made road building more limited in the eastern lowland area, meaning it was likely more economical to focus harvests in one location. In the western study area, access routes could be more freely distributed, meaning movement to more marketable timber was likely less

restrained. Moreover, multi-objective management strategies associated with sustainable forest management practices have to consider a wide range of social, cultural and economic perspectives, which, like environmental factors, likely differ between these areas. Long (2009) suggests that socio-economic barriers can inhibit ecologically driven forest management goals based on lack of social acceptance for mimicking high severity disturbance. Similarly, Schulte et al. (2006) noted that managers and forestry planners often found that social constraints hindered their ability to apply certain aspects of management guides. However, any difference between study areas in societal pressures or variation among forest companies in harvesting targets and constraints were beyond the scope of my study.

Although my results suggest that guidelines were not having the intended outcomes at a large scale, it is possible that changes in harvest patterns that were consistent with guidelines did occur at smaller scales. Guidelines were intended to have a large landscape level outcome; however, companies within separate FMUs operate independently, meaning that individual FMU's may meet intended management outcomes for the configuration of harvest blocks without desired outcomes being realized at a broader scale. This may also suggest why I saw results following more closely with predicted outcomes in the east when compared to the west. The western study area was made up of considerably more FMUs, meaning that FMU level outcomes would have been more diluted within the western landscape when compared to the east. Future investigation into management application at the FMU level is required.

2.5.2 Caribou habitat

Overall, my findings suggest that forest management guidelines did not have the effect on caribou habitat that I anticipated during the period of assessment. At the large scale of

measurement employed in this study, observed changes in caribou habitat were more consistent with stochastic dynamic natural disturbances (fire events) and changes in industry productivity than expected outcomes of management application.

Total forest harvested each year appears to be the major driver which led to observed decreases in the slope of metric values following 2007. In Ontario, the percentage of available wood approved in management plans that was actually harvested dropped from 73% in 2004 to 41% in 2008 (OMNR, 2012). Similarly, thousands of forestry sector jobs were lost due to mill closures across Ontario between 2006 and 2007 and the profitability of wood and paper products dropped substantially through this period (OMNR, 2012). Along with global economic recession, this reduced forest industry productivity coincides with the Canada-U.S. Softwood Lumber Agreement in 2006 and a large economic downturn in the United States housing market that occurred in 2007 (OMNR, 2012; Couture and Macdonald, 2013; Nagubadi and Zhang, 2013). As the U.S. receives the largest portion of Ontario's international lumber exports, these events markedly damaged the Canadian forestry sector, greatly impacting total forest depletions in Ontario during this period (Duncun, 2010; OMNR, 2012; Couture and Macdonald, 2013). It is likely, therefore that the economy was the major driver of the observed decreases in metric slope through time, rather than any changes in forest management practice.

My results suggest that forest fires are another major driver of the landscape patterns observed within my study areas, particularly in the west. Fires likely influenced my ability to detect gradual changes in slope expected to be representative of guideline application using segmented regressions within this study area. Large fires that occurred in sizable patches of caribou habitat consistently coincided with sharp declines in metric values between consecutive

years. The higher sensitivity of western study area metrics to these disturbances is likely related to fire cycle, as western Ontario is prone to much more frequent and large scale fires than eastern Ontario (Li and Perera, 1997; Perera and Baldwin, 2000; Bergeron et al., 2004). Evidently, there should be strong consideration for differences in disturbance processes during management planning at larger scales. This also highlights the importance of considering climate change prediction models during future management planning; fires are predicted to increase in size and frequency in Ontario's boreal forest in the near future according to future climate projections (Wotton et al., 2003; Flannigan et al., 2008). This could greatly influence the amount and configuration of available caribou habitat.

Management guidelines suggest planning over 80 years or more, and my analyses included approximately 10 years of post-guideline management (Racey et al., 1999). Although the goal of management is to maintain the integrity of remaining caribou habitat throughout all stages of this 80 year planning period, habitat changes may have been more difficult to detect during my shorter study period, as the full regeneration of harvests made during previous management strategies will take upwards of 80 years (Hins et al., 2009; Racey et al., 1999). Monitoring over a larger temporal scale should be conducted in the future to reduce the influence of large scale stochastic dynamic processes on overall temporal trends and to allow for further habitat regeneration from pre-guideline management practices. That being said, shorter term studies such as this one should act as an early warning indicator for the effectiveness of long-term management, and indicate whether more thorough investigations into management application are required.

2.6 Conclusions

Although consideration for caribou guidelines is taking place during forest management planning, my results suggest that landscape outcomes did not match those predicted based on guideline recommendations in maintaining caribou habitat or changing harvest block structure during management application at the temporal and spatial scales that I assessed. In contrast, fires and overall harvest rates associated with the downturn of the forest industry were highly influential. At a much broader scale, the results of my study underline the strong importance of monitoring management outcomes for effective wildlife management, particularly when management plans are aimed at meeting large landscape scale objectives across independent planning units. I recommend that forest management include long-term monitoring of management guideline application in meeting conservation targets for wildlife.

2.7 Literature Cited

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Appendix 2.1

Table A2.1. The relationship between large differences in habitat metric value between consecutive years (jumps) and burn and cut features on the landscape. The size of caribou habitat patches where burns occurred was divided into three classes: S (patch<50 ha), M (patch size between 50-100 ha), L (patch>100 ha). The top five largest burns were examined, and the largest habitat patch to intersect any of these burns was used for classifications. Data jumps are highlighted in bold under metric values, where metric values represent the difference between metric values recorded between consecutive years (ie metric value in 2003 represents 2002 metric value – 2003 metric value). Boxed years represent the location of consistent data jumps between all western metrics.

Year	Total Fire Area in West (ha)	Mean Burn Area	Mean Burn Area SE	Habitat Size in Relation to Large Burns	Total Cut Area in West	The Difference Between Consecutive Habitat Metric Values		
						MPS	AWMPI	TCA
1995	17,246.11	18.33	4.97	NA	32,402.10	NA	NA	NA
1996	118,640.15	181.68	41.75	S	23,408.36	0.0862	-316.57	5163.2
1997	4,690.08	17.97	6.77	M	44,163.34	0.0345	521.33	6287.7
1998	13,893.40	105.25	58.53	M	23,599.80	0.0388	9242.29	7350.7
1999	14,973.57	65.10	29.68	M	31,235.74	0.037	2804.63	7550.2
2000	3,248.85	98.45	55.80	S	33,983.52	0.0432	920.21	5932.2
2001	776.29	38.81	12.48	M	27,561.64	0.038	192.27	4794.1
2002	13,220.53	155.54	87.52	L	40,335.24	0.0811	1478.73	10655
2003	97,878.47	221.95	95.81	L	35,833.53	0.1574	34522.39	24349.1
2004	123.25	4.40	1.48	S	34,408.81	0.0416	694.57	4553.5
2005	3,815.35	65.78	40.92	S	35,576.42	0.0531	-495.53	5805.3
2006	44,176.29	271.02	116.80	S	38,400.49	0.0081	10273.13	8277.6
2007	16,012.65	165.08	2846.58	S	31,047.23	0.0498	1870.79	6811.3
2008	1,040.69	24.20	79.07	L	28,003.29	0.0345	1173.11	3508.4
2009	6,367.49	167.57	382.11	L	8,132.91	0.0166	128.99	2134.7
2010	6,457.04	169.92	357.40	S	9,692.02	0.0185	261.96	1807.8
2011	199,992.29	200.59	3095.17	M	9,392.04	0.166	16163.87	31637.7

3.0 Spatial behavioural patterns as indicators of long-term management outcomes: a case study of boreal caribou in Ontario, Canada

3.1 Summary

Monitoring long-term forest management outcomes is essential to conciliate species conservation efforts and the use of natural resources for human benefit. Gauging changes in wildlife behavioural response offers a direct method for assessing species directed management outcomes before there is strong risk to population viability. I employed hypothesis-based testing to assess changes in boreal caribou spatial behaviour patterns before and after management implementation to assess the feasibility of using changes in spatial behavioural response as an indicator of long-term management outcomes. Caribou behaviours were assessed over a 19 year period in two unique Ontario, Canada landscapes: the Boreal Shield and James Bay Lowlands. Changes in caribou proximity to forest harvests, home range size, and summer fidelity were used as behavioural indicators. Results suggest a negative behavioural response to forest management in large scale caribou proximity, with increasing caribou proximity to harvests suggest increasing predation risk for caribou. At smaller scales, caribou decreased their proximity to forest harvests; however their behaviours remained maladaptive. No changes in caribou home range size were observed. Caribou fidelity increased in the eastern study area following predictions but decreased in the western study area. Behavioural metrics varied in their sensitivity to changes in management implementation at different spatial and temporal scales. The results of my study suggest that monitoring changes in spatial behaviours provides a useful indicator of the impacts of forest management outcomes when a diversity of metrics and species-specific scales are employed. Using an animal's spatial behavioural response to monitor management outcomes will be most effective when assessed in relation to observed changes in habitat features which are directly managed through forestry practices.

3.2 Introduction

The management of wildlife habitat is used in many jurisdictions as a way to maintain wildlife populations; however, there is limited knowledge of the effectiveness of this approach and long-term monitoring of habitat management outcomes remains rare. Monitoring efforts that are conducted tend to be indirect measurements assumed to be associated with species conservation outcomes, such as landscape metrics used to measure changes in wildlife habitat (e.g. Fauth et al., 2000; Hansen et al., 2001; Schindler et al., 2008). However, indirect management assessments should be complemented with direct measures of animal response. Of particular importance are early warning indices that facilitate the understanding of species response to management outcomes before there is a strong risk to population viability.

Spatial ecology, the practice of monitoring an animal's geographical patterns in relation to ecological phenomena, has become commonplace in the study of wildlife (Horne, 2007). Spatial behaviours, in particular, have been used to evaluate ecological patterns in a range of taxa, with different behavioural strategies representing different adaptations used to meet life history requirements (White and Garrott, 1990; Bowler and Benton, 2005). With increasing public interest in the influences of anthropogenic disturbance on the environment, spatial behaviours have been used to gauge for the level of disturbance on a landscape (e.g. Smith et al., 2000; Scillitani et al., 2009; Gaulton et al., 2011), where changes in behaviour can be used to indicate the amount of ecological stress (Sutherland, 1998; Tuomainen and Candolin, 2011). Indicator species tend to be a common focus of these studies, as they are assumed to represent biota or environmental change within their ecosystem, providing a biological warning of ecosystem disturbance (Lindenmayer and Likens, 2011). Spatial behavioural assessments in regards to disturbance tend to be short term studies following disturbance activities. Here, I use

spatial behaviours as indicators of the outcomes of long-term species directed forest management application by contrasting the behavioural response of boreal caribou (*Rangifer tarandus caribou*) before and after guideline implementation time periods.

Because of their large individual home ranges and high sensitivity to anthropogenic disturbances, boreal caribou are considered an indicator of boreal forest ecosystem health (OMNR, 2009; Festa-Bianchet et al., 2011). Forest harvests are a suggested cause of caribou range declines (Schaefer et al., 2003; Vors et al., 2007; Wittmer et al., 2007), as they deteriorate caribou habitat while simultaneously increasing the habitat of predators and their alternate prey, leading in turn to increased predator abundance and predation on caribou (Potvin et al., 2005; Boan et al., 2011; Latham et al., 2011; Lesmerises et al., 2012). Northward expansion of natural resource development has led to a large number of anthropogenic disturbances within caribou range (Metsaranta and Mallory, 2007). As such, recent research has documented an array of changes in caribou spatial behavioural responses attributed to varying levels of anthropogenic landscape changes, including caribou avoidance behaviours, movement patterns and space use (e.g. Smith et al., 2000; Courtois et al., 2007; Schaefer and Mahoney, 2007; Hins et al., 2009; Moreau et al., 2012). Evidence from existing literature on caribou's reactions to disturbance allows for the creation of informed predictions on their spatial behavioural responses towards predicted management outcomes, making behaviour an opportune measure for the long-term assessment of forest management practices.

I used hypothesis-based testing and comparison of caribou behaviour patterns between pre- and post- management implementation periods to assess the outcomes of forest management guidelines for caribou in Ontario, Canada. Guidelines are intended to sustain caribou populations

on landscapes managed for timber harvest. Caribou select for large, interconnected patches of mature, conifer forests (O'Brien et al., 2006; Lesmerises et al., 2013). As such, guidelines intend to increase the size and aggregation of forest harvest patches over multiple Forest Management Units (FMUs) to reduce caribou habitat fragmentation and maintain large, interconnected blocks of caribou habitat (Racey et al., 1999; OMNR, 2001). I compared (1) the proximity of caribou to harvests, (2) home range size and (3) seasonal fidelity before and after 15 years of habitat management implementation as a method to assess management outcomes. Because wildlife have been found to show high levels of variability in behaviour in relation to diverse environments (e.g. Ferguson et al., 1999; Watson et al., 2005; Environment Canada, 2011), I tested whether there were differences in the sensitivity of behavioural responses to management application between two unique landscapes that differed in fire cycle, forest community structure, and climate. Predictions for each metric were based on evidence from existing literature of behavioural responses to expected outcomes of forest management to habitat (Table 3.1).

Table 3.1. Predictions and their rationale for changes in each measured spatial behavioural metric in relation to management application.

Spatial Behaviour	Prediction and Rationale
Proximity Index	Boreal caribou have been shown to space away from forest harvests (Vors et al., 2007; Vistnes and Nellmann, 2008); however, as the level of fragmentation increases on the landscape, caribou can become ‘trapped’ in smaller patches of habitat surrounded by harvests, decreasing their distances to disturbance (Hins et al., 2009; Lesmerises et al., 2013). As such, I predicted that caribou would become farther from harvests (increase their proximity index), if management was effective at separating disturbance and habitat patches on the landscape.
Annual Home Range Size	Studies have found a negative relationship between home range size and increasing levels of disturbance, as caribou will initially attempt to space away from disturbance, but eventually become ‘trapped’ in smaller fragments of habitat as fragmentation increases (Smith et al., 2000; Courtois et al., 2007; Beauchesne et al., 2014). Due to the high level of disturbance already associated with these landscapes, I predicted that caribou home ranges would not decrease if management was effective at separating disturbances and habitat patches on the landscape.
Fidelity	Boreal caribou have been shown to display strong fidelity to particular habitats within the summer season (Schaefer et al., 2000; Metsaranta and Mallory, 2007; Popp et al., 2011), however, increasing levels of disturbance and fragmentation have been associated with decreased fidelity (Courtois et al., 2007; Faille et al., 2010). I predicted that caribou fidelity would increase or remain the same if management was effective at separating disturbance and habitat patches on the landscape.

3.3 Methods

3.3.1 Study areas

Caribou were monitored in two Ontario landscapes: northwestern Ontario, comprised primarily of the Boreal Shield ecozone, and northeastern Ontario, dominated by the Hudson Bay Plain ecozone. The northeastern study area is flat with a mean elevation of 250 m (Boissonneau, 1996; Belleau et al., 2011). The altered humid continental climate, which displays many maritime climate characteristics as well as poorly drained soils, leads to high levels of paludification, making peatlands and monospecific black spruce stands dominant habitats throughout the region (Jeglum and He, 1995; Baldwin et al., 2000; Brown et al., 2003; Belleau et al., 2011). As such, fire cycles are long, estimated at 398 years, while mean stand age is estimated at 148 years (Bergeron et al., 2004).

In contrast, the northwestern study area is dominated by well drained soils and rolling hills (Zoladeski and Maycock, 1990; Sims et al., 1995; Watson, 1997). The climate regime in this region matches the majority of Ontario as humid continental (Baldwin et al., 2000). Jack pine (*Pinus banksiana*) is the dominant stand type, with a mean stand age of 99 years and a relatively short fire cycle of approximately 74 years (Zoladeski and Maycock, 1990; Li and Perera, 1997; Ferguson and Elkie, 2004).

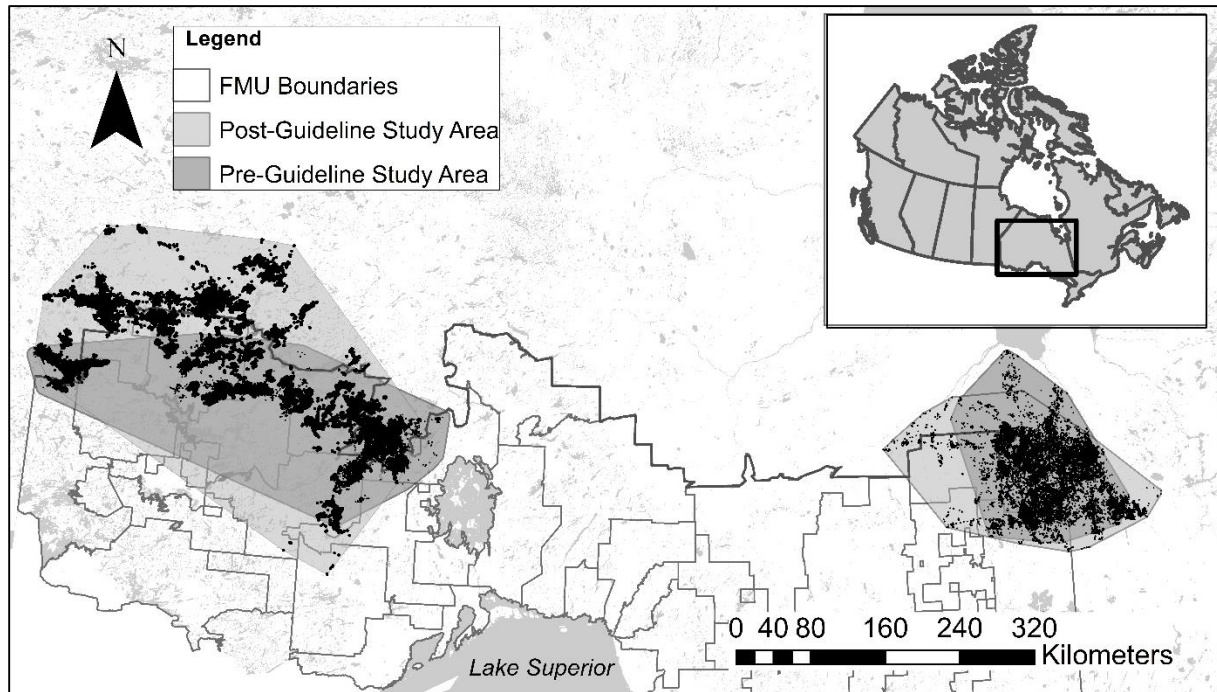


Figure 3.1. The study areas in eastern and western Ontario, Canada based on boreal caribou (*Rangifer tarandus caribou*) radio-locations (black dots) that occurred in pre- and post-guideline time periods. Only caribou which had locations that occurred below the northern limit of forest management units (FMU) were assessed.

3.3.2 Data processing

Data sets included approximately one million telemetry locational points from 143 adult female boreal caribou recorded between 1995 and 2013. Only caribou whose cumulative ranges overlapped with managed forests were analysed. Caribou locations were divided into four sub-categories during analysis based on time period and study area (eastern and western) (Figure 3.1). Time was composed of early and late classes based on data availability. Early classes fell between 1995 and 2001 (pre-guideline implementation) and late classes fell between 2009 and 2013 (post-guideline implementation). Early data were composed solely of Service Argos telemetry data, while late data were composed of GPS data.

I defined each biological year as May 1 to April 30 the following year, consistent with the approximate start of the calving season. This period also allowed me to maximize use of available data as collaring of animals occurred in February or March. Seasonal periods included: Winter (November 16th to February 15th), Spring (February 16th to April 30th), Summer (May 1st to September 15th), and Fall (September 16th to November 15th). These designations were based on current caribou literature within and surrounding each study area (Ferguson and Elkie, 2004; Courbin et al. 2009; Hazell and Taylor, 2011). Calving season was not separated from summer because of a low number of samples within this season in the early data set.

ARGOS data sets use the quality of satellite reception to grade each calculated location using Location Classes (LCs), with 3 being the highest, followed by 2, 1, 0, A, B, and Z. All ARGOS datasets were preprocessed by removing data with a $LC < 1$, as well as any aberrant data found to be at unrealistic distances from other locations. Some studies suggest using Kalman's filtering methods to reduce the error in ARGOS locations (Patterson et al., 2010; Hoenner et al.,

2012); however, these studies were focused on marine animals, with data sets composed largely of $LC < 1$. Hoenner et al., (2012) found that adding $LC > 1$ data points following Kalman's filtering back into the data set yielded the best spatial behaviour estimates. As such, I did not filter my data further.

3.3.3 Home range size

I used fixed 90% kernel utilisation distributions to calculate annual home range size, using the reference smoothing factor (h_{ref}). This method was chosen over the least squares cross-validation smoothing (h_{scv}) method because of a large level of failure for my samples, assumed to be caused by tight clusters of my locational points (Gitzen et al., 2006). Further, studies have suggested that the h_{scv} method can result in a large level of variability between estimates at lower sample sizes, a limitation of my early data set (Seaman et al., 1999; Kernohan et al., 2001; Borger et al., 2006). All home range size estimates were calculated using the 'AdehabitatHR' package in R software (Calenge, 2006).

Only animals with a minimum of 50 locational points per year were used to estimate annual home ranges, as this number has been shown to be the point at which range size estimates stabilize (Seaman et al., 1999, Kernohan et al., 2001). This created a total sample size of $n=124$ caribou, with $n=28$ in the east early period, $n=17$ in the east late period, $n=15$ in the west early period and $n=64$ in the west late period. I found range estimates fit my data set well during the early period in both study areas. However, during the late period, a large level of over smoothing was observed, with travel corridors generally being eliminated from range estimates. As such, I applied a bootstrapping method, where I calculated ranges for 65 randomly selected sub-sampled GPS locations over 1000 iterations for each individual. Sub-sampling within large GPS data sets

has been shown to have high comparability with lower quality data sets of smaller sample sizes (Pellerin et al., 2008). I then averaged all home range estimate rasters from the 1000 iterations for each individual within each year to create my late period home range size estimates. Following sub-sampling, home range fit was greatly improved.

The influence of forest harvests on home range size was quantified by measuring the percentage of annual home range area that was composed of harvests made within approximately 10 years of the measured caribou locations. Because it is likely that elements outside of an animal's selected home range also influence home range size, I measured the percent of harvest area within varying buffer distances from each individual's home range, with only the buffer area, and not the home range area, included. Different buffer distances were used for each study area (37 km for eastern Ontario and 21 km for western Ontario) due to large differences in mean home range sizes between areas. Both distances were calculated using the square root of the average annual home range size within each study area, roughly providing a buffer distance matching the width of the average home range area. I chose to keep the percent harvest area within home range and within home range buffers as separate variables, as it is likely that each could influence home range contraction versus home range expansion differently.

3.3.4 Proximity index

Caribou proximity to harvests, determined by taking the ratio of observed to expected distances from harvests, was calculated for cuts made within approximately 10 years of each recorded caribou location. Caribou have been shown to avoid harvest blocks beginning at their first year of creation (Smith et al., 2000; Hins et al., 2009), with increasing levels of harvests <10 years of age being a strong predictor associated with caribou extirpation (Vors et al., 2007). The

'near tool' in ArcGIS was used to measure the distance of the closest forest harvest block to each caribou location. Expected distances were measured using the systematic approach outlined by Benson (2013), where the mean distance to harvest was calculated using ArcGIS distance rasters at both population and individual annual home range scales. Population scales were expected to be representative of caribou's ability to space away from harvests within my designated population range, while individual annual home range scales were expected to be representative of an individual's ability to space away from harvests within its home range. The population range was created by using the 100% Minimum Convex Polygon (MCP) of all caribou locations in each data group (e.g. east early, west early etc.) and adding a 7.5 km buffer to each range. Annual ranges were created for each individual using methods described in the previous section, with expected annual distances calculated for each individual within each year. The number of individuals used within proximity analysis matched those of home range analysis.

Unlike home range, the proximity index is measured at each individual location, and therefore does not face the same limitations with sample size as do home range size estimators. As such, I was able to isolate separate seasonal measures for proximity. To increase the degrees of freedom within each model, I ran each analysis separately for summer and winter seasons (summer population level proximity, winter population level proximity etc.). These are the seasons most emphasized in previous caribou literature due to higher caribou vulnerability to predation during these periods (e.g. Ferguson and Elkie, 2004; O'Brien et al., 2006; Pinard et al., 2012; Beauchesne et al., 2013).

I applied the same harvest measures used for home ranges (the percent harvest within each 90% kernel home range, and the percent harvest within designated buffer distances

surrounding home ranges) to assess the influence of forest harvests within the landscape matrix on caribou proximity to harvests. It is likely that caribou will change their location in relation to harvests based on the habitat features that surround each harvest patch. Thus, I also measured the percent harvest area within a 7.5 km buffer surrounding the closest harvest to each caribou location. This distance was based on data that suggest that caribou will make decisions about a habitat patch based on the surrounding landscape matrix up to 7.5 km away (Lesmerises et al., 2013).

3.3.5 Fidelity Index

The fidelity index was assessed using two average distances: paired and expected. Locational points were put into pairs, where each pair consisted of two locational points which were recorded on the same day during consecutive years (ie, July 1st 1998 and July 1st 1999). The distance between each pair of points was then recorded. Paired distances were calculated for each individual by taking the average distance between each pair of locational points measured for each caribou (Schaefer et al., 2000; Popp et al., 2011). Measuring fidelity using individual locations rather than home range overlap estimators allowed me to isolate enough data to assess fidelity during the summer season, where the strongest displays of fidelity are generally observed (Schaefer et al., 2000; Metsaranta and Mallory, 2007; Popp et al., 2011). Distance calculations between paired points were done following the equation outlined in Popp et al., (2011). Only individuals with a minimum of 10 locational points spread across each month of the summer season were included in the analysis. A total of n=53 individuals were used for fidelity index assessments, with n=14 in the east early period, n=12 in the east late period, n=11 in the west early period, and n=16 in the west late period.

Expected distances for each data grouping were derived using the average distance between all possible pairs of locations for all collared caribou within each annual summer season (Schaefer et al., 2000). These values were then averaged over all measured years within each data grouping (east early, west early, etc.). I then created a ratio of expected to paired distance values to represent fidelity index in subsequent analysis. By creating a ratio, I hoped to control for the possibility of range contraction causing the appearance of increased fidelity due to high levels of fragmentation.

To account for the influence of harvests on changes in the fidelity index, I created a buffer around each locational point included within the fidelity index analysis. I selected buffers which would represent the area used by caribou within each year. Two different buffer distances were used: a 12 km buffer distance on western study area caribou locations and a 21 km buffer distance on eastern study area caribou locations. Buffer distances were calculated by taking the square root of the average home range size divided by π , in order to attain a relative radius value of caribou home range. Different buffer distances were used in each study area due to the large differences between average home range sizes, as indicated in Section 3.4.1. The average percent harvest area within each buffer was then calculated for all locational points collected for each individual within consecutively occurring years.

3.3.6 Statistical assessments

Because my data were nested, linear mixed effects models (LMEs) from the ‘nlme’ package in ‘R’ software environment (R version 3.0.1) were used for the analysis of proximity and home range size (Pinheiro et al., 2015). Each spatial metric was used as a dependent variable, while data grouping (west early, east early, etc.) and different combinations of harvest

measures, were included as fixed effects. Individual was used as the grouping factor for random effects. In order to meet model assumptions, I applied an array of relevant correlation and variance structures to remove autocorrelation and heterogeneity. Top correlation and variance structures were selected using AICc (Akaike's Information Criterion for small sample sizes) from the package "AICcmodavg" in R (Mazerolle, 2015) following the process outlined by Zuur et al., (2009). For fidelity, I modeled changes using linear regressions, as fidelity measures were averaged over all years for each individual, with the same covariates as the fixed effects described for LMEs.

In all analyses, AICc was used to select between candidate models with varying harvest covariates. If the top model was found to have a model weight below 0.90, then model averaging was applied using "MuMIn" in R (Barton, 2014). Once the top model was selected, post-hoc tests were conducted where necessary using the 'glht' function from the package "multcomp" in R (Hothorn et al., 2008).

3.4 Results

3.4.1 Home range size

The best model included time period and study area (i.e. east early, west early), the percentage of harvest area within annual home ranges and buffered zones around home ranges (Table A3.2). There was no significant change in home range in either study area (east: $p=0.92$, $\beta=9.96$, $SE=15.60$, 95% CI [-29.95, 49.88], west: $p=0.99$, $\beta=1.25$, $SE=15.05$, 95% CI [-37.36, 39.75]); however, eastern Ontario was found to have significantly larger home ranges than

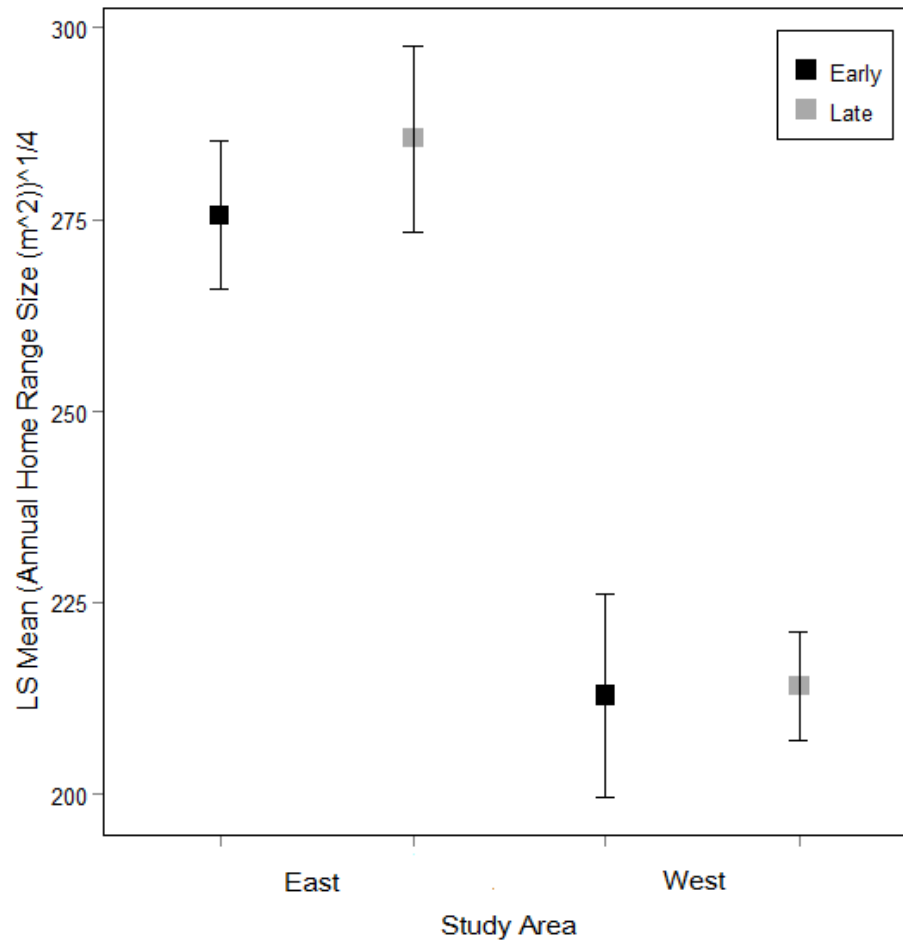


Figure 3.2. A comparison of least square mean home range size of boreal caribou (*Rangifer tarandus caribou*) between the eastern and western study areas in the early period (black) and late period (grey). Error bars represent standard error.

western Ontario in both the early and late time periods (early: $p < 0.01$, $\beta = -62.81$, $SE = 16.31$, 95% CI [-104.56, -21.07], late: $p < 0.01$, $\beta = -71.52$, $SE = 14.02$, 95% CI [-107.38, -35.66]).

Home range size decreased with increasing harvest area surrounding a caribou's home range ($p < 0.01$, $\beta = -27.59$, $SE = 7.98$, 95% CI [-27.59, -11.58]). When the percent harvest area increased within caribou home ranges, so did home range size ($p < 0.01$, $\beta = 47.63$, $SE = 10.09$, 95% CI [47.63, 67.87]).

3.4.2 Proximity index

At the population scale, I found that the best model to explain variation in the caribou proximity index contained the percent harvest area within a 7.5 km buffer of the nearest harvest to each caribou location, along with data grouping (e.g. east early, west early, etc.) in both seasons (Table A3.3). The same results were found during summer at the individual annual home range scale. In winter at the individual annual home range scale, the percent harvest area within each animal's annual home range was also included in the top model (Table A3.3).

There was no change in caribou proximity index between early and late time periods during the winter season at the population scale in eastern Ontario ($p = 0.47$, $\beta = -0.07$, $SE = 0.05$, 95% CI [-0.18, 0.05]); however, there was a significant decline in the proximity index in the west, suggesting caribou became closer to harvests following guideline introduction ($p < 0.01$, $\beta = -0.21$, $SE = 0.04$, 95% CI [-0.32, -0.09]; Figure 3.3). During summer at the population scale, I did not detect any changes in the proximity index between early and late periods in either study area (east: $p = 0.46$, $\beta = -0.13$, $SE = 0.09$, 95% CI [-0.37, 0.10]; west: $p = 0.13$, $\beta = -0.22$, $SE = 0.10$, 95% CI [-0.22, 0.04]; Figure 3.3). In contrast, at the individual annual home range scale during

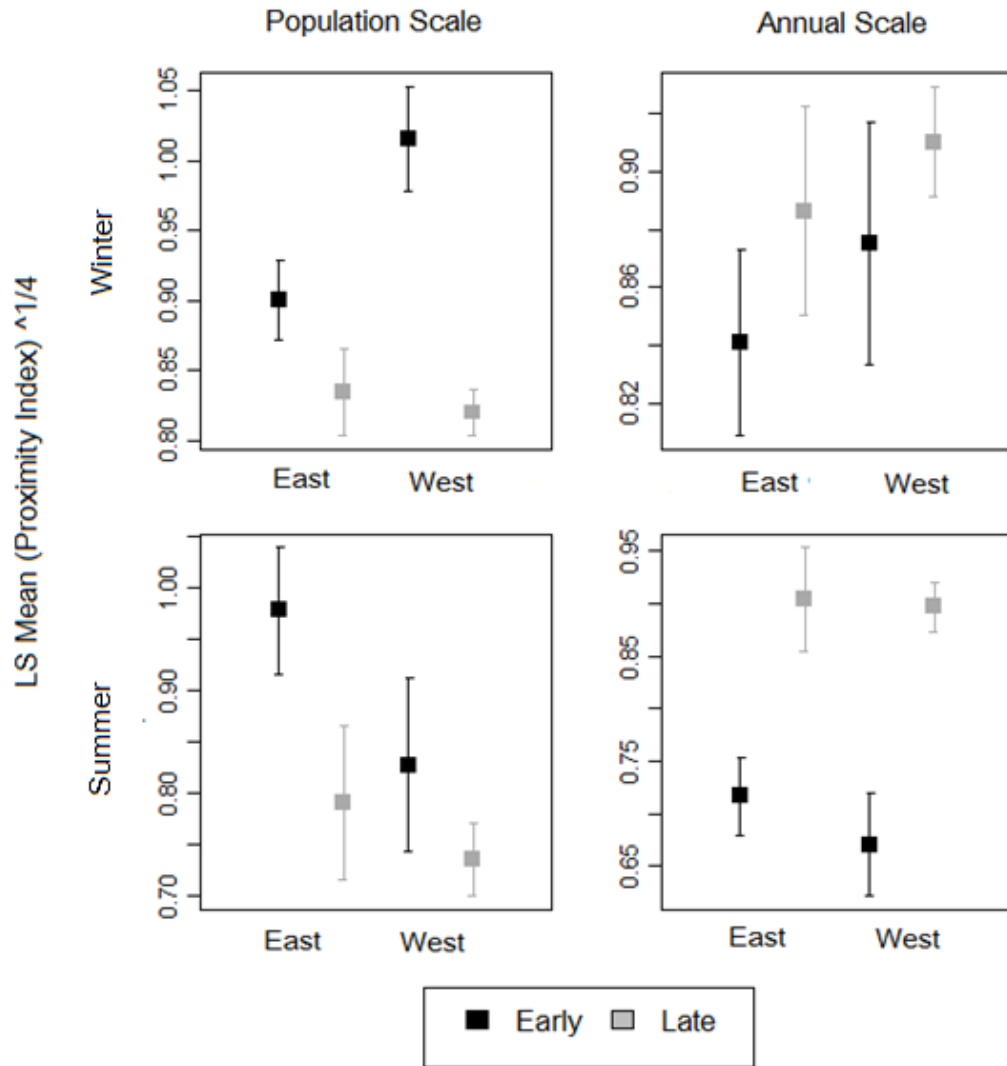


Figure 3.3. Comparison of the least square mean proximity index of boreal caribou (*Rangifer tarandus caribou*) between the eastern and western study areas in the early period (black) and late period (grey) at the population and individual annual home range scale during summer and winter seasons. Decreasing proximity index values suggest increasing caribou proximity to harvests when compared to expected distances. Error bars represent standard error.

summer, I observed a significant increase in proximity index in both study areas (east: $p=0.01$, $\beta=0.19$, $SE=0.06$, 95% CI [0.03, 0.35]; west: $p<0.01$, $\beta=0.22$, $SE=0.05$, 95% CI [0.08, 0.36]; Figure 3.3). That being said, caribou proximity values never increased to a ratio higher than 1, suggesting caribou still remained closer to harvests than expected. During winter, there were no changes in either eastern or western study areas at the individual annual home range scale ($p=0.71$, $\beta=0.05$, $SE=0.04$, 95% CI [-0.06, 0.15]; $p=0.91$, $\beta=0.03$, $SE=0.04$, 95% CI [-0.08, 0.13] respectively; Figure 3.3).

Caribou moved closer to areas where there was increased harvesting. In both seasons and at both scales, there was a significant decrease in caribou proximity index with increasing levels of percent harvest area within a 7.5 km buffer surrounding the harvest nearest to caribou (Winter, population: $p<0.01$, $\beta=-0.05$, $SE<0.01$, 95% CI[-0.06, -0.05]; Winter, annual: $p<0.01$, $\beta=-0.06$, $SE<0.01$, 95% CI[-0.07, -0.05]; Summer, population: $p<0.01$, $\beta=-0.02$, $SE<0.01$, 95% CI [-0.03, -0.02]; Summer, annual: $p<0.01$, $\beta=-0.17$, $SE<0.01$, 95% CI[-0.18, -0.16]). There were no significant differences in proximity index between eastern and western Ontario in either pre- or post-guideline periods during both seasons at the population scale (Winter, pre-guidelines: $p=0.09$, $\beta=0.11$, $SE=0.05$, 95% CI [-0.01, 0.24]; Winter, post-guidelines: $p=0.86$, $\beta=-0.03$, $SE=0.04$, 95% CI [-0.13, 0.07]; Summer, pre-guidelines: $p=0.99$, $\beta=0.03$, $SE=0.12$, 95% CI [-0.27, 0.33]; Summer, post-guidelines: $p=0.85$, $\beta=-0.05$, $SE=0.07$, 95% CI [-0.24, 0.13]) or the individual annual home range scale (Winter, pre-guidelines: $p=0.88$, $\beta=0.03$, $SE=0.05$, 95% CI [-0.09, 0.15]; Winter, post-guidelines: $p=0.90$, $\beta=0.02$, $SE=0.03$, 95% CI [-0.07, 0.11]; Summer, pre-guidelines: $p=0.87$, $\beta=-0.04$, $SE=0.06$, 95% CI [-0.20, 0.11]; Summer, post-guidelines: $p=0.99$, $\beta=-0.01$, $SE=0.05$, 95% CI [-0.15, 0.13]; Figure 3.3).

3.4.3 Fidelity Index

Multi-model inference provided harvest and grouping factor covariate importance values of 0.31 and 1.00 for my fidelity index assessments (Table A3.4).

I found a significant decline in caribou fidelity to summering areas in the western study area following guideline introduction (Figure 3.4; $p < 0.01$, $\beta = -0.37$, $SE = 0.12$, 95% CI [-0.58, -0.15]). In the eastern study area, I found a significant increase in fidelity to summering areas (Figure 3.4; $p < 0.01$, $\beta = 0.33$, $SE = 0.11$, 95% CI [0.54, 0.11]); however, a mean fidelity index value below one prior to guideline release indicates that caribou were farther from previously used locations than expected, suggesting that no fidelity behaviour was displayed during the early period in the east. Western caribou displayed much stronger fidelity to summer ranges than eastern caribou in both early and late time periods (Figure 3.4; $p < 0.01$, $\beta = 1.20$, $SE = 0.11$, 95% CI [0.99, 1.43]; $p < 0.01$, $\beta = 0.51$, $SE = 0.12$, 95% CI [0.30, 0.72], respectively). There was no significant impact of the measured harvest value on fidelity behaviour ($p = 0.70$, $\beta < -0.01$, $SE = 0.01$, 95% CI [-0.04, 0.01]).

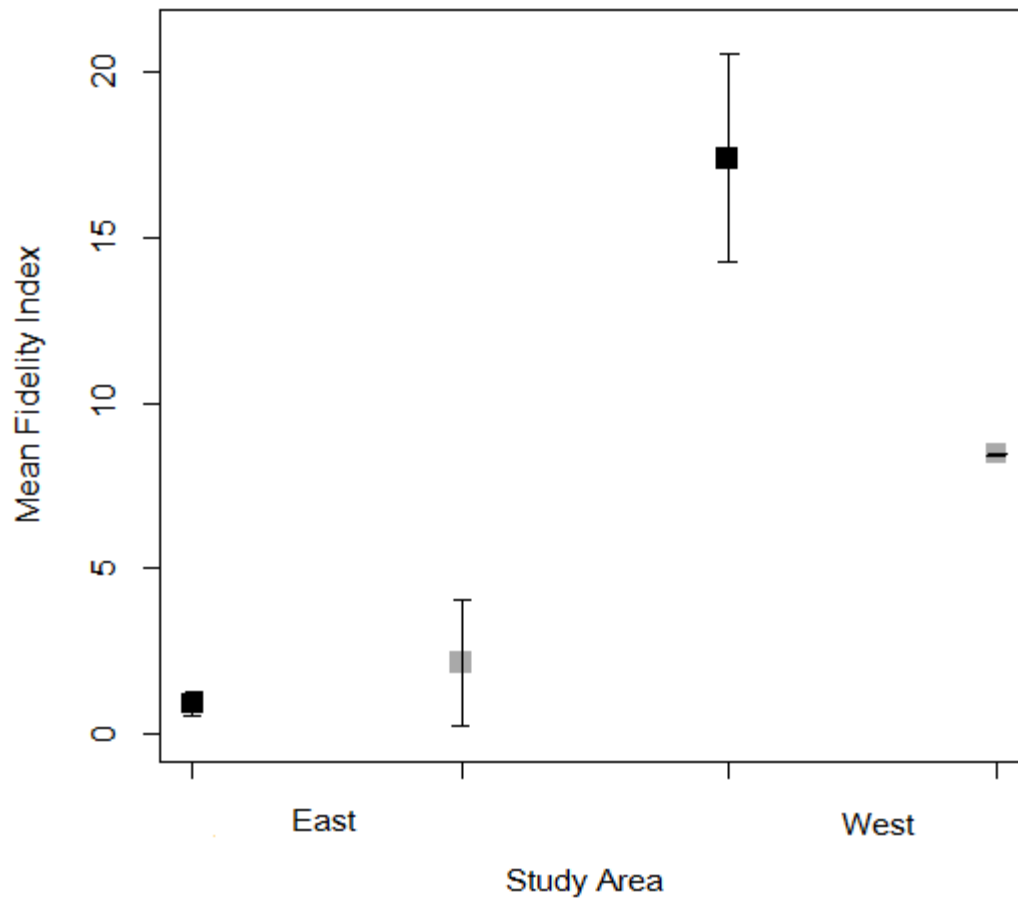


Figure 3.4. A comparison of the mean fidelity index of boreal caribou (*Rangifer tarandus caribou*) to summer ranges between the eastern and western study areas in the early period (black) and late period (grey). Increasing fidelity index values suggest increases in fidelity behaviour relative to expected distances. Error bars represent standard error.

3.5 Discussion

Despite the large scale motivations underlying habitat management direction, caribou demonstrated behavioural changes consistent with an increased negative influence of disturbance at larger scales in the caribou proximity index. Increases in caribou proximity to harvests relative to harvest distribution in the population range is suggestive of a maladaptive response to forest management, as animals increase their risk from predation by moving closer to disturbances (Courtois et al., 2007; Latham et al., 2011; Lesmerises et al., 2012). Research elsewhere suggests that intensive disturbance associated with increased levels of habitat fragmentation can create an ecological trap for caribou, where caribou become restricted to small patches of high quality habitat close to forest disturbances making them a more predictable target for predators like wolves (*Canis lupus*; Battin et al., 2004; Lesmerises et al., 2013).

At smaller scales, caribou were found to increase their distance from harvests relative to what was expected between early (pre-guidelines) and late (post-guidelines) periods. This suggests a positive influence of forest management on caribou at the individual annual home range scale, opposite to my results at the population scale. Differences in caribou selection and avoidance of harvests in relation to spatial scale have been previously linked to strong associations between high quality caribou habitat and forest harvests (Hins et al., 2009). Although I saw increases in caribou distance from disturbance, caribou still remained closer to harvests than expected based on the distribution of harvests within individual annual home ranges. This suggests that caribou maintained a maladaptive behavioural response to harvesting regardless of the scale measured, suggesting caribou were likely still exposed to elevated predation risk (Lesmerises et al., 2013).

In the western study area, I observed decreased caribou fidelity between time periods, suggesting increased levels of disturbance to caribou during the summer season at smaller scales (Courtois et al., 2007; Faille et al., 2010). In the eastern study area, I did not detect the display of fidelity behaviour during the early period of investigation; however, fidelity was observed following guideline implementation. This may suggest decreasing disturbance to caribou following guideline implementation in the eastern study area. However, because not all caribou display fidelity behaviour (e.g. Rettie and Messier, 2001), observed increases could also be related to maladaptive behaviours associated with fragmentation, with caribou becoming trapped in smaller patches of habitat (Battin et al., 2004).

Unlike in other measured behavioural metrics, I did not detect any changes in caribou home range size between time periods in either study area, suggesting that caribou home range size was not influenced by changes in management. However, I observed strong differences in range size between study areas, with western Ontario caribou having much smaller home range areas than eastern Ontario caribou. Similarly, caribou displayed much stronger fidelity behaviour in the west than in the east. Differences in spatial behaviours between these regions can be observed when comparing other studies (e.g. Brown et al., 2003; Racey et al., 1997), and is likely related to a number of interacting factors, including differences in habitat distribution and abundance within each landscape and greater levels of natural and anthropogenic disturbances within the western study region (Li and Perera, 1997; Armstrong, 1998; Perera and Baldwin, 2000; Bergeron et al., 2004). These results highlight the importance of considering the inherent complexity associated with caribou behavioural response within varying landscapes when planning management and monitoring efforts.

The maintained maladaptive response of caribou to forest harvests suggest that caribou were still exposed to elevated predation threat following guideline introduction (Wittmer et al., 2007; Latham et al., 2011). Particularly in the western study area, where increasingly maladaptive behavioural responses were observed in large scale caribou proximity measures, as well as decreased fidelity behaviour, results suggest that should management outcomes continue in this manner, there may be detrimental impacts to caribou population viability (Tuomainen and Candolin, 2011). Further, as an indicator species, the spatial behavioural response of boreal caribou to forest management during my study period could suggest negative outcomes for boreal forest health and biodiversity (Rodrigues and Brooks, 2007). These results are consistent with landscape outcomes observed in Section 2.0, which suggest harvest distributions moving in directions opposite to guideline recommendations, most strongly in the western study region. As such, I recommend further investigation into the outcomes of caribou directed forest management be conducted.

I have demonstrated the use of hypothesis-based testing on spatial behaviours as an indicator of long-term habitat management outcomes. Monitoring using spatial behaviours indicates the direct impacts of management on focal species, providing more detailed information on management outcomes than can be achieved solely through habitat assessments. As such, I recommend that managers utilize spatial behaviours as a monitoring method during future management assessments. This being said, researchers must consider the inherent complexity of behavioural decisions when interpreting results. I observed large differences in spatial behaviour response based on spatial and temporal scale as well as the metric used. As such, multiple behavioural metrics measured at spatial and temporal scales relevant to management targets and focal species should provide the best understanding of management

outcomes by accounting for the varying levels of sensitivities associated with different populations in different landscapes.

3.7 Conclusions

I observed maintained maladaptive responses of caribou to forest management during my study period, particularly in the western study region, suggesting potentially negative impacts for boreal caribou population viability. These results are consistent with harvest assessments for these study areas (Section 2.0). My study demonstrates that hypothesis-based testing on caribou behavioural responses to changes in habitat management application can provide a valuable early indicator of the impacts of management on wildlife populations. Similarly, monitoring spatial behaviours provides more direct information on the impacts of forest management outcomes than is detectable solely based on indirect methods such as landscape metrics. Due to the limitations associated with historical telemetry data sets as well as limited knowledge on behavioural thresholds between populations, hypothesis-based testing for behavioural metrics is likely best used in association with other monitoring methods.

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Appendix 3.1

Table A3.1 A description of model covariates used in candidate models run in Akaike's Information Criterion to explain variation in annual home range size, the proximity index and summer fidelity.

Covariate	Definition
Group	A categorical variable representing the four groupings that data was sorted into based on study area and time period: Early East, Early West, Late East, Late West.
CutinHR	The percent area composed of forest harvests within an individual's 90% fixed kernel home range
CutHRBuffer	The percent area composed of forest harvests within a 21 km (east) or 37 km (west) buffer region surrounding an individual's 90% fixed kernel home range
CutBuffer	The percent area composed of forest harvest within the 7.5 km buffer region surrounding the forest harvest nearest to a measured caribou location
CutPoint	The average percent area composed of forest harvest within a 21 km (east) or 37 km (west) buffer region surrounding each locational point included in fidelity measures
C1	Autoregressive process of order 1 correlation structure (Pinheiro et al., 2015)
C2	Linear spatial correlation structure with a constant variance function structure (Pinheiro et al., 2015)

Table A3.2 The top models explaining variation in home range size (HR) for caribou exposed to changing management in Ontario: the degrees of freedom of each model (df), the natural logarithm of maximum likelihood for each model (LogLik), the Akaike's information criterion adjusted for small sample size bias (AICc), the change in AICc (Δ AICc) and the Akaike weight for each model (W).

Model	df	LogLik	AICc	Δ AICc	W
HR~Group+CutinHR+CutHRBuffer	8	-921.10	1859.06	0	1.00
HR~Group+CutinHR	7	-929.33	1873.33	14.27	0
HR~Group+CutHRBuffer	7	-932.49	1879.65	20.59	0
HR~Group	6	-935.96	1884.42	25.36	0

Table A3.3 The top five models explaining variation in caribou proximity to harvests (Prox) at population and annual scales during winter and summer periods: the degrees of freedom of each model (df), the natural logarithm of maximum likelihood for each model (LogLik), the Akaike's information criterion adjusted for small sample size bias (AICc), the change in AICc (Δ AICc) and the Akaike weight for each model (W).

Model	df	LogLik	AICc	Δ AICc	W
Winter, Population					
Prox~Group+CutBuffer+C1	8	22862.69	-45709.37	0	0.98
Prox~Group+CutBuffer+CutinHR+C1	9	22859.74	-45701.47	7.90	0.02
Prox~Group+CutHRBuffer+C1	8	22776.14	-45536.28	173.09	0.00
Prox~ Group+CutinHR+CutHRBuffer+C1	9	22774.61	-45531.22	178.15	0.00
Prox~ Group+CutinHR+C1	8	22768.96	-45521.91	187.46	0.00
Winter, Annual					
Prox~ Group+CutBuffer+CutinHR+C1	9	18449.34	-36880.68	0.00	1.00
Prox~ Group+CutBuffer+C1	8	18440.43	-36864.86	15.82	0.00
Prox~ Group+CutBuffer+CutHRBuffer+C1	9	18438.19	-18438.19	22.30	0.00
Prox~ Group+CutinHR+C1	8	18363.87	-36711.74	168.94	0.00
Prox~ Group+C1	7	18359.11	-36704.22	176.86	0.00
Summer, Population					
Prox~ Group+CutBuffer+C2	9	51316.99	-102616.0	0.00	1.00
Prox~ Group+CutinHR+C2	9	51300.41	-102582.8	33.17	0.00
Prox~ Group+Cutin+CutHRBuffer+C2	10	51220.35	-102420.7	195.29	0.00
Prox~ Group+CutBuffer+CutinHR+C2	10	51209.24	-102398.5	217.50	0.00
Prox~ Group+C2	8	51206.82	-102397.6	218.35	0.00
Summer, Annual					
Prox~ Group+CutBuffer	7	18647.23	-37280.45	0.00	0.98
Prox~ Group+CutBuffer+CutinHR	8	18644.24	-37272.48	7.97	0.02
Prox~ Group	6	17779.99	-35547.98	1732.47	0.00
Prox~ Group+CutinHR	7	17776.74	-35541.03	1740.97	0.00
Prox~ Group+CutHRBuffer	7	17777.52	-35532.67	1747.78	0.00

Table A3.4 The top models explaining variation in fidelity for caribou exposed to changing management in Ontario: the degrees of freedom of each model (df), the natural logarithm of maximum likelihood for each model (LogLik), the Akaike's information criterion adjusted for small sample size bias (AICc), the change in AICc (Δ AICc) and the Akaike weight for each model (W).

Model	df	LogLik	AICc	Δ AICc	W
Fidelity~Group	5	-4.66	20.59	0	0.69
Fidelity~Group+CutPoint	6	-4.19	22.21	1.61	0.31

4.0 General Conclusions and Recommendations

Overall, predicted management outcomes intended to alter habitat and harvest structure do not appear to have been achieved at the measured landscape scale. Similarly, caribou behavioural response to changes in management appeared to be maladaptive, with behaviours in the western study area changing in the opposite direction of what I predicted based on guideline recommendations suggesting increased levels of disturbance to caribou. This suggests that forest management may not be having outcomes expected to minimize the impacts of harvesting to caribou populations on managed landscapes. That being said, the downturn of the forestry industry explained changes found in a large portion of tested habitat metrics, and therefore, needs to be considered as a potential confounding factor of my analysis. Results of landscape metrics and spatial behaviour assessments were relatively consistent and therefore, both methods can serve as useful diagnostic tools to evaluate habitat management outcomes. However, each method was sensitive to different aspects of management outcomes as well as varying characteristics of each observed landscape. Most predominately, I observed patterns within and between my observed spatial behavioural metrics, which appeared to be dependent on the type of metric used and spatial and temporal scale. As such, I have outlined three considerations that managers should take into account when planning monitoring assessments using spatial behaviours:

Consideration 1: Spatial scales of observation

I found differences in caribou response to management practices at different spatial scales of testing, most clearly demonstrated by the proximity metric. Dissimilar selection and avoidance responses of wildlife to landscape features at different scales is commonly reported (e.g. Herfindal et al., 2009; Hins et al., 2009; Pedersen et al., 2010). When interpreting differences in

responses at varying scales, it has been argued that coarser scales tend to be most relevant, as these constrain decisions made at smaller scales (Johnson, 1980). For my study, management goals generally pertain to large scale habitat management (Racey et al., 1999; OMNR, 2001), as caribou have large home ranges relative to the scales of forest harvest planning. However, the relevance of each scale when interpreting management outcomes for other studies should be dependent on management targets as well as the ecology of the focal species being observed (e.g. Levin, 1992). Furthermore, it is clear that the spatial scale at which managers choose to monitor their focal species' behavioural response can greatly influence the outcome of their management assessment. As such, assessing multiple relevant scales may be needed to understand the influence of management application.

Consideration 2: Seasonal sensitivity of spatial behaviours

Many species display differences in movement patterns, habitat selection, and sensitivity to disturbance depending on season (McLoughlin et al., 2002; Moreau et al., 2012; Shapiera et al., 2014). My results for caribou were consistent with these findings. Further, I found a potential relationship between the seasonal effects of disturbance on behaviour and my scales of assessment. My results show that caribou distance to harvest blocks only increased significantly between time periods during the summer at the individual annual home range scale. At the population scale, the opposite was observed, with caribou becoming significantly closer to harvests in the later time period during winter. Observing significant positive responses to management outcomes only during summer at one scale, and significant negative responses only during winter at the other, may suggest that caribou alter their behavioural response to varying levels of disturbance within different seasons.

Seasonal sensitivity of a spatial behavioural metric is likely related to the way an animal prioritizes life history requirements. It has been suggested that caribou will prioritize habitat selection decisions based on food requirements during winter and predator avoidance to minimize calf predation during summer (McLoughlin et al., 2005; Gustine et al., 2006; Leclerc et al., 2012), which likely influences changes in the level of caribou proximity to harvests. Because of caribou's tendency to prioritize predation avoidance, and therefore spatial segregation from harvests during summer, the proximity metric may have been more sensitive to detecting changes in behaviours in relation to decreased fragmentation by harvest disturbance during this season. During winter, caribou have been observed to decrease their distances to anthropogenic disturbances when compared to summer, possibly explained by their need to fulfill necessary nutritional requirements, though caribou 'avoidance' of harvests is still observed (Beauchesne et al., 2013). If there are increased levels of fragmentation on the landscape, caribou will likely be closer to these disturbances during winter. It is important to note that suggesting that available forage and predation are the two defining factors associated with my observed changes in caribou proximity between seasons likely over-simplifies the complexities associated with other behaviours, such as strong summer fidelity displayed even with increasing disturbance levels and predation threat (e.g. Faille et al., 2010; Tracz et al., 2010). Clearly, investigation into the cause of differences in seasonal sensitivity of behavioural metrics in detecting changes in management outcomes is required, as this is beyond the scope of this study. However, regardless of the underlying cause, my results provide valuable information for future monitoring efforts looking to assess management application using spatial behavioural metrics on species who display seasonality in their behaviour, as season can clearly influence a metrics ability to detect management outcomes.

Consideration 3: Different behaviours may be sensitive to different aspects of disturbance

Animal behaviours may vary in response to differing aspects of disturbance (e.g. Thurber et al., 1994; Semeniuk et al., 2014), and in accordance, my results revealed that certain behavioural metrics may be more sensitive to detecting specific management changes. For instance, declining fidelity in the west suggests negative impacts of management on caribou, opposite to findings for proximity at the individual annual home range scale in this same study area. Unlike proximity, harvests were not found to be as important in explaining variations in the fidelity metric, suggesting that fidelity may be influenced by other drivers. For instance, Faille et al., (2010) found that roads had an important influence on caribou fidelity, and this would be a useful area for future study. This being said, assessments of caribou fidelity to annual ranges, as well as proximity measurements for seasonal ranges, would be beneficial to distinguish differences associated with scale in interpreting metric sensitivity. Unfortunately, limitations in the sample size given the comparisons being made within my data set meant that I was unable to investigate contrasts in metric outputs further. Regardless, my results suggest that monitoring management outcomes based on one behavioural metric may not be adequate to understand the impacts of management implementation. Instead, a range of metrics should be applied in order to more thoroughly assess the impacts of management outcomes.

In summary: (1) behavioural response to management varies with the spatial scale at which it is tested, (2) seasonal differences in behavioural response may influence the sensitivity of behavioural metrics to changing management and (3) some behavioural responses are more sensitive to different aspects of management outcomes than others. I recommend that managers note the considerations that I have listed with respect to their focal species and incorporate each into the planning, application and interpretation of spatial behaviour monitoring assessments.

Based on these complexities, I recommend that managers employ both landscape and behavioural assessments to attain the most effective management assessments of landscape outcomes.

Based on the qualitative results of my study, further investigation into the application of forest management with respect to boreal caribou is required within Ontario. I recommend that future activities investigating the effectiveness of management guidelines should include:

- 1) Investigating changes in landscape metrics at the FMU level to determine if FMUs demonstrate outcomes consistent with guideline recommendations.
- 2) Assessing harvest allocation in relation to the mosaic plan within each FMU to determine the level of consistency between mosaic planning objectives and landscape outcomes
- 3) Assessing the distribution of habitat as a result of the mosaic plan to determine its effectiveness at reaching habitat continuity targets that span multiple FMU's.
- 4) Assessing caribou use of protected known calving and wintering habitat within each FMU.
- 5) Assessing caribou selection of regenerated habitat following forest harvesting to determine the timeframes, patch composition and landscape connectivity required to ensure caribou use of previously occupied portions of their range following forest harvest.
- 6) Re-assessing caribou spatial behavioural changes over an extended time period and determining animal behavioural thresholds within each Ontario region in order to better understand the direct impacts of management on caribou response.

- 7) Investigating management's impacts on caribou vital rates, as well as caribou population persistence within managed areas to determine the success of caribou guides at maintaining caribou.

My study underlines the importance of monitoring to support assessment of management success. It is crucial that managers establish monitoring as an integral component of management planning so that long-term monitoring is consistently applied following management application. My findings on the influence of natural disturbance regimes on landscape configuration, and differences in behavioural responses to spatial and temporal scale, will be informative to managers monitoring habitat management for a range of species spanning multiple biogeographic gradients. Future studies should investigate the use of these methods for other focal species, as the sensitivity of these monitoring methods will likely differ for different management targets.

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