The Influence of Habitat on Woodland Caribou Site Fidelity

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Abstract

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Site fidelity is the behaviour of individuals to return to the same location; for female woodland caribou it may reflect reproductive success and depend on habitat quality. I investigated the influence of landscape and disturbance conditions on fidelity among three populations in Manitoba and Ontario, Canada. Habitat classifications were based on Forest Resource Inventory (FRI) and Landsat TM landcover maps. A total of 261 sites were ground-truthed to determine mapping accuracy. An amalgamated map incorporating FRI and Landsat TM data was estimated from field measurements to have an overall accuracy of 69.0%. Site fidelity was expressed as the distance between consecutive-year locations of individuals and was investigated during five week-long periods representing calving, early and late post-calving, winter, and breeding. Site fidelity was strongest during the post-calving seasons and weakest during the winter. Habitat had little influence on site fidelity in all seasons, excepting winter, even under highly disturbed conditions, suggesting maintenance of fidelity may be a maladaptive trait. Individual variation proved a strong predictor and cursory mapping indicated that caribou may return to sites visited two or more years earlier. Conservation management and policy should recognize that site fidelity may represent an ecological trap.

Keywords: *Rangifer tarandus caribou*, site fidelity, habitat, disturbance, conservation, movement, habitat use, calving, Forest Resource Inventory Maps, Landsat TM Thematic Imagery Maps, ecological trap, Far North Landcover

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Chapter 1: General Introduction

Woodland Caribou

Caribou (*Rangifer tarandus*) are an ancient species of deer (Jermann et al. 1995; Geist 1998; Randi et al. 1998) located throughout the Arctic and Subarctic regions of the Holarctic (Flerov 1933; Williams and Heard 1986; Ma 1986; Valdez et al. 1995; Mallory and Hillis 1998; Aastrup 2000; Nellemann et al. 2001; Kojola et al. 2004). In Canada there exist four subspecies of caribou; their ranges stretch across such disparate environs as the windswept islands of the Arctic Archipelago in the north of their range, to the boreal forest in the south (Environment Canada 2012). The most southerly subspecies, the woodland caribou (*Rangifer tarandus caribou*), has itself been broken down into six distinct populations based both on geography and behaviour (COSEWIC 2002). The boreal woodland population is a widespread and sedentary caribou (Festa-Bianchet et al. 2011) that is endemic to the Canadian boreal forest (Environment Canada 2012). In 2000 this population was listed as “threatened” in Canada (COSEWIC 2002). The boreal woodland caribou (or sedentary ecotype) is distinguished from the migratory woodland caribou population through their movement behaviour especially during calving where the sedentary populations use a “spacing out” strategy to avoid predators while migratory animals use a “space away” strategy (Bergerud 1985; Bergerud and Page 1987; Bergerud et al. 1990; Bergerud 1996). “Spacing out” means that pregnant female caribou space themselves across the landscape in a diffuse arrangement to attempt to avoid detection by predators, while animals that “space away” tend to aggregate in large herds beyond the tree line on the tundra in an attempt to remove themselves from predators (Bergerud 1996). This distinction is somewhat nebulous, with sedentary and migratory animals occasionally mating together where they cooccur (Boulet et al. 2007). Despite the disparate calving strategies, both of these ecotypes exhibit a predictable behaviour known as “site fidelity” during the calving and post-calving period, tending to return to the same site year after year (Schaefer at al. 2000).

Site Fidelity

Female boreal woodland caribou (hereafter referred to as woodland caribou) have often been thought to exhibit a high degree of site fidelity (Brown et al. 1986, Schaefer et al. 2000), a behaviour common among many different species (Greenwood 1980; Shields 1984; Lanyon and Thompson 1986; Switzer 1997; Matthiopoulos et al. 2005). Site fidelity is typically distinguished using two different definitions: intra-year site fidelity is the tendency of an animal to remain in a specific portion of its range for a length of time, while inter-year site fidelity is the tendency for an animal to return to the same location at particular times in subsequent years (White and Garrott 1990; Switzer 1993; Faille et al. 2010). The return to a specific location or area is thought to be advantageous because those individuals have a prior knowledge about the quality of the habitat, and therefore can return to locations that are known to be beneficial while avoiding those that may be of a poorer quality. This fidelity is often related to reproductive success (Burger 1982; Badyaev and Faust 1996; Hoover 2003). Woodland caribou habitat selection, at the daily area scale and at coarser scales, is driven principally by an avoidance of predators (Stuart-Smith et al. 1997; Gustine et al. 2006), apparent competitors (Seip 1992), and human activity while selecting habitat types that would likely maximize this avoidance (Rettie and Messier 2000; Ferguson and Elkie

2005; Dussault et al. 2012). These “ideal” habitats include old-growth conifer forests, treed muskegs (Rettie and Messier 2000), islands and shorelines (Bergerud 1985). Site fidelity would therefore maintain this avoidance from year to year (Greenwood 1980).

Site fidelity is commonly thought to occur in many different populations of woodland caribou with the strongest occurring during the calving and post-calving seasons (Schaefer et al. 2000; Ferguson and Elkie 2004; Wittmer et al. 2006; Tracz et al. 2010; Popp et al. 2011; Schaefer and Mahoney 2013). A degree of site fidelity has been noted during the breeding season with winter commonly considered the period of time where fidelity is weakest (Schaefer et al. 2000; Schaefer and Mahoney 2013). Calving and post-calving is the most limiting time of year for a woodland caribou (Dussault et al. 2012) with mother caribou and young calves highly susceptible to predators and the stressors associated with feeding and interactions with humans (Calef et al. 1976; Gauthier and Theberge 1985; Bergerud and Elliot 1986; Murphy and Curatolo 1987; Parker et al. 1990; Whitten et al. 1992; Roffe 1993; Mahoney et al. 1990; Maier et al. 1998). Indeed, many species have been known to show increased site fidelity after successfully recruiting young into the population the previous year (Burger 1982; Hoover 2003; Sedgwick 2004). Early studies have shown that this assumption appears to be true in woodland caribou as well (Faille et al. 2010).

Northern Development in Canada

Canada is a country heavily indebted to its North with its natural resources shaping our economy both historically and presently (Hayter and Barnes 2001). In regions as inaccessible as the Far North of Ontario, permanent industrial development began nearly a century ago, during the early part of the 20th century, with the creation of the Canadian National Railroad (Bowman et al. 2010). While the North has been inhabited for centuries (Hamilton and Molto 2006), large-scale modifications associated with development and resource extraction are relatively recent (Carleton 2000) and chiefly attributed to the forest industry and logging activities (Pinto and Romaniuk 2004). Between 1951 and 1995 approximately 6.6 million hectares of forests were harvested in Ontario through clearcutting alone (Perera and Baldwin 2000). While some argue that northern development and resource extraction in some regions has slowed in recent years (Hanlon and Halseth 2005; Statistics Canada 2013a; Statistics Canada 2013b), profits are increasing and development continues (Timilsina et al. 2005; Islam 2008; Mining Association of Canada 2012; Natural Resources Canada 2013). The Alberta oil sands portend substantial growth, increasing production from 1.61 to 5.02 million barrels a day by 2030 and jobs from 75 000 to 905 000 in 2035, with a third of those directly involved in the oil sands (Mining Association of Canada 2012). Non-metal mines, such as quarries, have also increased between 2011 and 2012 (Mining Association of Canada 2011, 2012). This is in addition to the “Ring of Fire”, a large chromite deposit in northern Ontario, that is currently in an exploratory and planning phase, but which is estimated to be worth $50 billion (Lazenby 2013). Estimates state that 600 new resource-based projects are planned for development within the next 10 years in Canada (Office of the Auditor General of Canada 2013). All development requires a myriad of infrastructure: roads, rail, energy, and a work force (Dybas 2012), often cutting deeply into previously unreachable old growth boreal forest. As an example, the Ring of Fire chromite deposit will likely require a 329 km railway to link potential mines to the nearest railroad (Dugan et al. 2012). This development is mirrored in current population trends, showing a general increase throughout the north (Statistics Canada 2012) that is likely to continue (Statistics Canada 2010).

These modifications of the landscape leave a legacy of lasting environmental change, from their own physical footprint to alterations in the physical and biological composition of floral and faunal communities (Pinto and Romaniuk 2004). While development provides a host of benefits to Canada's economy, the influence it has on the environment should not be underestimated. Woodland caribou in Canada often come into contact with northern development, and have experienced a worrying range retraction and population loss throughout Canada since European colonisation (Dymond et al. 1928; Bergerud 1974; Schaefer 2003).

Impacts of Human Development on the Woodland Caribou – Historical and Present

Caribou populations worldwide are showing signs of decline and movement towards extinction (Dymond et al. 1928; Bergerud 1974; Ceballos and Ehrlich 2002; Schaefer 2003, Vors and Boyce 2009). In Ontario caribou ranges have receded northward at the expense of half of their historic range (Schaefer 2003). During an early survey into the fauna of the Lake Nipigon region of Ontario, Dymond et al. (1928) found caribou numbers at much lower levels than historically present, with moose (*Alces alces*) displacing caribou as the dominant large deer species in areas where moose were previously rare. Another faunal survey of the Lake Abitibi region in northern Ontario was conducted by Snyder (1928) who interviewed residents and found that caribou had been numerous less than a decade previously, but at the time of the survey the range appeared to have receded over 15 km north. By the time of Snyder's study, approximately 50 years after the first moose sighting in the region, moose were "plentiful and generally distributed." At the time, the cause of this local disappearance of caribou was attributed to the incoming moose and white-tailed deer, as well as an increase in forest fires (Snyder 1928; Dymond et al. 1928); however, Dymond et al. (1928) noted that the retraction coincided approximately with the construction of the National Transcontinental Railway. Other observations corroborate the early 20th century initiation of caribou range retraction in Canada (de Vos and Peterson 1951; Bergerud 1974) often being coincident with the construction of railroad and its opening up of the North (Bowman et al. 2010). Observations from the continental United States place this loss slightly earlier with the last observation from some states occurring before the turn of the 20th century (Jackson 1922) while others recorded observations until the early 20th century (Wood 1917; Johnson 1922).

Three decades ago there were two competing hypotheses as to why the woodland caribou distribution was retreating northward; as a result of poor range quality potentially caused by human or natural disturbances (Scotter 1964; Klein 1968; Darby and Duquette 1986) or the result of pressures from predators and overhunting (Bergerud 1974). Today a strong consensus exists that this range retraction is due to some combination of both these causal factors (Darby and Duquette 1986) with disturbances, anthropogenic and natural, causing a loss of the preferred forest types which in turn can cause an increase in predators (Dalerum et al. 2007; McCarthy et al. 2011; Wasser et al. 2011; Dussault et al. 2012; Fortin et al. 2013). Arguably the largest form of anthropogenic disturbance in the Canadian boreal forest is logging (Pinto and Romaniuk

2004; Bowman et al. 2010). Logging in Canada has traditionally used and still does use clearcutting on mature coniferous forests for paper production (Darby and Duquette 1986; Fourrier et al. 2013). When a mature coniferous forest is lost through logging, the forest goes through a period of regrowth where the plant species richness and the level of productivity are both higher than before the disturbance (Peltzer et al. 2000; Reich et al. 2001) with deciduous trees dominating (Carleton and MacLellan 1994). Caribou are known to avoid regenerating (Dalerum et al. 2007; Pinard et al. 2012) and deciduous forests (Hirai 1998; Minaskuat Inc 2009; Proulx 2013) as calf mortality is higher in this habitat (Dussault et al. 2012). This is thought to be because early successional and deciduous forests are the preferred habitat for apparent ungulate competitors such as moose (*Alces alces*) and white-tailed deer (*Odocoileus virginianus*) as well as black bears (*Ursus americanus*), an opportunistic, and sometime significant, predatory of woodland caribou calves (Dussault et al. 2005; Mosnier et al. 2008; Bastille-Rousseau et al. 2011; Dussault et al. 2012). Wolves (*Canis lupus*) also benefit from industrial development, using roads and linear corridors to travel faster through the forest than they otherwise would be able to (James 1999; Latham et al. 2011a; Whittington et al. 2011). The potential increase in the predatory ability of wolves is further compounded by the increased moose populations, which in turn support higher wolf densities than historically present (Bergerud 1974; Bergerud and Elliot 1986; Seip 1992; Latham et al. 2011b). Thus disturbances created through development cause a decrease in old-growth conifer habitat, an increase in apparent competitors, and an increase in both the population and the efficiency of the caribou’s primary predator, the wolf. This is in addition to adverse effects caused through direct interactions with humans such as harvest and vehicle collisions (Calef et al. 1976; Murphy and Curatolo 1987; Maier et al. 1998; Dyer et al. 2001; Wasser et al. 2011).

Governmental Conservation Strategies for Woodland Caribou

A recent Canadian federal recovery strategy for the conservation of woodland caribou was released in 2012 (Environment Canada 2012). The overarching goal of the strategy is to achieve self-sustaining status for all identified populations of woodland caribou in Canada. The strategy recognized 51 ranges in Canada, with only fourteen currently self-sustaining. Despite the 37 populations that are not self-sustaining, the strategy states that a recovery is “technically and biologically feasible” for all populations (Environment Canada 2012). To achieve this, the report lays out a guideline of ensuring that a minimum of 65% of all woodland caribou ranges be left undisturbed. This value is stated as a minimum due to the fact that the odds of achieving self-sustainability at this level of disturbance are 60%, leaving a 40% chance of failure. Similar efforts are being maintained in individual provinces with the 2005 Manitoba recovery strategy mimicking the goals of the federal strategy closely (Manitoba Conservation 2005). In Ontario and Quebec, the recovery strategies include province-specific issues, namely the isolation of herds cut off from the continuous populations: the coastal population along Lake Superior in Ontario and the Val d’Or and Charlevoix populations in Quebec (Équipe de rétablissement du caribou forestier du Québec 2008; Environment Canada 2012; OMNR 2012b).

One of the crucial stepping stones to woodland caribou conservation in Canada is filling in knowledge gaps on caribou population dynamics, habitat use, and the influence of human-caused disturbances on caribou (Environment Canada 2012). Much of the effort has been focused on monitoring caribou movement and habitat use as well as estimating population size and health. The purpose of my thesis is to investigate the influence that habitat, including both natural and anthropogenic features, has on site fidelity throughout key periods in the annual cycle of adult female caribou. This will provide a more complete understanding of a key behaviour in an imperilled species.

Objectives

This study investigates the relationship between inter-annual site fidelity and habitat using 72 female woodland caribou, equipped with GPS collars, from three populations in Ontario and Manitoba. I first conducted an investigation of the classification accuracy of the landcover maps used to represent the woodland caribou's boreal forest habitats (Chapter 2). By using a classification scheme that maximized accuracy as well as incorporating updated disturbance information, I investigated how the landscape, in the form of landcover, water, and disturbances, both natural and anthropogenic, influenced site fidelity during vital periods of the year for female woodland caribou (Chapter 3). I focused on three populations of woodland caribou in Canada. The Owl-Flintstone population in southern Manitoba represented a population historically subject to high disturbance levels while maintaining a relatively stable population (Environment Canada 2012). The Atikaki-Berens population, also in southern Manitoba, is an apparently self-sustaining, stable, population that has been subjected to little disturbance (Environment Canada 2012). The Kesagami population straddling the Ontario and Quebec border south of James Bay, in a range with a latitudinal gradient of disturbances, is listed as declining (Environment Canada 2012). Ultimately the results of this study are intended to further our understanding of how habitat influences site fidelity and to provide management recommendations based off my findings.

Chapter 2: Classification Accuracy of Forest Resource Inventory and Landsat TM Thematic Imagery for Boreal Forests in Ontario and Manitoba, Canada

Introduction

Conservation of many species, especially wide-ranging species such as the woodland caribou (*Rangifer tarandus caribou*), requires understanding their habitat and how they interact with it (Hulbert and French 2001; Rushton et al. 2004; Shafer et al. 2012). Modelling this interaction is often the first step in developing conservation strategies (Johnson et al. 2004) and studies have shown that management plans incorporating habitat use increase the potential for the continued occurrence of the target species (Brown et al. 2007). While there have been many recent technological advances in wildlife telemetry resulting in more frequent and accurate animal location points (Aarts et al. 2008; Tomkiewicz et al. 2010), inferences made from these locations are only as accurate as the habitat maps used to determine the relationships (Maxie et al. 2010). With the increased accuracy in telemetry comes increased reliance upon these data as the backbone of many studies (Bowman et al. 2010; Faille et al. 2010; Northrup et al. 2012; Stewart et al. 2013). Despite the repeated suggestion that the accuracy of these maps needs to be investigated (Dussault et al. 2001; Johnson and Gillingham 2008), there continue to be many studies that fail to report an accuracy of any kind (Metsaranta and Mallory 2007; Bowman et al. 2010; Burdett et al. 2010; McClure et al. 2011; Shafer et al. 2012; Leopold and Hess 2013). This is especially common in the woodland caribou's boreal forest habitat where few published studies on map accuracy exist (Thompson et al. 2007).

There are many different techniques used in creating habitat maps, including visual interpretation of aerial and satellite photography or the isolation of specific light wavelengths, modified radar, and the analysis of the reflected light caused by satellite bound lasers (Collis 1965; Carter 1969; Eberhart 1978; Innes and Koch 1998). Forest Resource Inventory (FRI) maps are based on aerial photograph interpretation; these maps are created by and for the forest industry in order to plan and implement timber harvesting (Leckie and Gillis 1995). FRI maps represent one of the most common map types used in wildlife-habitat studies (Rettie et al. 1997; Poole et al. 2004; Koen et al. 2007; Metsaranta and Mallory 2007; Bowman et al. 2010). As an FRI map's primary role is to measure the amount of merchantable timber in each forest stand, the stands are not described by predefined classes, but rather as a composition of the main canopy tree species (Leckie and Gillis 1995). For non-merchantable stands they are classed into broader, structurally-based classifications (e.g.: muskeg, water, rock). Other variables included are stand age, stocking, and management status. Standard FRI maps are interpreted from aerial images usually at a 1:20 000 scale, with 20 years between updates not uncommon (Thompson et al. 2007; Pinto et al. 2007). In short, FRI maps allow for ecologically meaningful classifications at a finer spatial and classification resolution than most other options (Brown et al. 2006). However, as FRI maps are created for the forest industry, their range is limited to areas of economic interest and they often extend only as far as harvest remains profitable, frequently excluding northern and more remote areas that are often required for studies of wide-ranging animals such as the woodland caribou (Brown et al. 2006). As a result, other data sources with coarser spatial and classification resolution must be used.

One such commonly used alternative to FRIs is landcover mapping interpreted from Landsat thematic mapper (TM) imagery. Landsat TM data are satellite imagery composed of multiple bands of light wavelengths, some in the visible spectrum and some in the infrared, allowing for the differentiation of vegetation variety and density (Crist and Cicone 1984; Stewart et al. 2012). Landsat TM imagery maps have a coarse spatial resolution of 30 metres and as a result most habitat classifications are based more on structural differences rather than species composition (Spectranalysis Inc. 2004; OMNR 2012a). The benefit of this imagery is that it extends north beyond the extent of FRI mapping.

Within all mapping techniques there exists classification error. Common errors associated with map creation include interpreter error (Powell et al. 2004) and difficulties distinguishing tree species in mixed-wood stands (Thompson et al. 2007). Additionally the effects of habitat change and disturbance over time can change map accuracy after its creation (Pinto et al. 2007). It was suggested by Thomlinson et al. (1999) that, to be appropriate for use in wildlife-habitat studies, maps should have an overall accuracy of greater than 85% with no single class less accurate than 70%. Nevertheless, a great many studies have published accuracies below this threshold (Johnson et al. 2003; Wickham et al. 2004; Pinto et al. 2007; Maxie et al. 2010; Thompson et al. 2007; Koen 2006; Wickham et al. 2013) including two studies in the boreal forest of Canada with accuracy levels of 46% (Thompson et al. 2007) to 68% (Dussault et al. 2001).

An understanding and acknowledgement of the accuracy and reliability of these maps are vital to the ability to interpret the results of any study (Maxie et al. 2010). Without acceptable levels of accuracy in the habitat representation, inferences from studies employing these maps in support of land-use and conservation policies may be incorrect or biased and lead to potentially damaging policy. Herein I quantified the accuracy of both FRI and Landsat TM maps in two study areas in the boreal forest of Ontario and Manitoba for the purpose of incorporating them into a wildlife-habitat interaction study of woodland caribou.

Methods

Study Areas and Populations

Three populations were included in this analysis: the Atikaki-Berens, Owl-Flintstone, and Kesagami populations (Figure 1). Each was chosen to represent a range of disturbances and sustainability and for which there was a large extant dataset of woodland caribou location information from global positioning system (GPS) equipped collars deployed on females. Each population's range was defined as a minimum convex polygon (MCP) surrounding all GPS locations points for each population plus an additional 15 km buffer to account for habitat and disturbances outside of the MCP (Smith et al. 2000; Schaefer and Mahoney 2007; Vors et al. 2007). As a result of this buffer the Atikaki-Berens and Owl-Flintstone population ranges overlapped along Highway 304 near the community of Bissett (Figure 2).

The Atikaki-Berens population (represented by 23 collared females [Table A1]) is located in southern Manitoba and is considered self-sustaining and has been subjected to little disturbance (Environment Canada 2012). The Atikaki-Berens range encompasses an area of approximately 30 000 km^2 including the 15 km buffer (Figure 2). The majority of the range is located in Manitoba with the eastern edge of the range extending approximately 20 km into Ontario. The range is bounded in the south by Highway 304 and the Poplar/Nanowin Rivers Park Reserve in the north with Lake Winnipeg serving as a hard western border. The range contains the entirety of Atikaki Provincial Park in Manitoba and part of Woodland Caribou Provincial Park in Ontario. Mean daily temperatures range from -19 ^{o}C in January to 18.3 ^{o}C in July. Average yearly rainfall is 430 mm and snowfall is 134 mm (Environment Canada 2013). Located on the south-western part of the Precambrian shield the Atikaki-Berens range is largely characterised by a landscape of uplands with thin soils dominated by black spruce (*Picea mariana*) and jack pine (*Pinus banksiana*) with exposed rock ridges separating depressions often forming into narrow lakes, poorly drained peat-filled bogs, and black spruce and tamarack (*Larix larcina*) lowlands (Rowe 1972; Schaefer 1996). Nearer to the shore of Lake Winnipeg the species composition changes; with increased drainage into Lake Winnipeg allowing for more jack-pine dominated forests and more deciduous forests of trembling aspen (*Populus tremuloides*; Rowe 1972).

The Owl-Flintstone population (represented by 27 collared females), also located in southern Manitoba, represented a population historically subjected to high disturbance levels while maintaining a relatively stable population (Environment Canada 2012). This population is located to the south of the Atikaki-Berens range. Its total area is just over 5000 km^2 including the 15 km buffer (Figure 2). The range is bounded by Highway 304 in the north and west and contains the majority of Nopiming Provincial Park. The far eastern edge of the range enters into Ontario where a small part of Woodland Caribou Provincial Park is contained in its far north-eastern section. Temperature and precipitation conditions are similar to the Atikaki-Berens population as well as both the landscape characteristics and species composition. However, in the western portion of the range deciduous forests composed of balsam poplar (*Populus balsamifera*) and aspens (*Populus* spp.) with white spruce (*Picea glauca*) appear in greater frequency as drainage increases and the landscape stabilizes into a less varied relief (Rowe 1972).

The Kesagami population (represented by 22 collared females) straddling the Ontario and Quebec border south of James Bay contains a latitudinal gradient of disturbances and is listed as declining (Environment Canada 2012). This population is located predominantly in north-eastern Ontario with its eastern edge located in Quebec. This range is the largest of the three, representing an area totalling nearly 60 000 km^2 including the 15 km buffer (Figure 3). The range stretches from nearly the southern tip of James Bay in the north to Highway 11 east of Kapuskasing in the south. Mean daily temperatures range from -18.4 ^oC in January to 16.8 ^oC in July. Mean yearly rainfall is 583.2 mm and snowfall is 296.8 mm (Environment Canada 2013). Structurally differentiated as an area of sedimentary rock covered with marine clay from glacial depositions, it appears as a vast landscape of poorly drained flats and small clay-lined streams (Rowe 1972; Carleton and Maycock 1978) that slowly drain northward into James Bay (Bergeron 2000). Sphagnum-dominated bogs and muskegs, fens, and black spruce forests cover the land with the occasional hardwood or mixed-wood stand in areas of increased drainage (Rowe 1972). In stark contrast to the Owl-Flintstone and the Atikaki-Berens population ranges, the Kesagami population range features few lakes and exposed rock is exceedingly rare (Rowe 1972).

Habitat Maps

The size of these woodland caribou ranges and the distance between each study population required the use of several habitat maps. In the Owl-Flintstone population range Forest Resource Inventory (FRI) maps were obtained from Manitoba Conservation and the Ontario Ministry of Natural Resources (OMNR; Figure 4). In Manitoba these maps represent an interpretation of aerial photographs from 1997 at a scale of 1:15 840 and account for 4017 km^2 of the total 5010 km^2 range. In Ontario, FRI maps from the Kenora district were used and accounted for 733 km^2. The remaining 258 km^2 remained unmapped in the north-eastern corner of the range. In the Atikaki-Berens population range FRI maps were used for the majority of the range (Figure 5). The south-western portion (6 556 km^2) of the range was represented with the same FRI as the Owl-Flintstone population while 18 438 km^2 of the range was represented by an older FRI, using imagery dating from 1984. As the Atikaki-Berens range extended beyond the extent of the forestry industry activity, and therefore beyond the FRI maps, I used Landsat TM imagery classed by the OMNR as the Far North Landcover Classification to represent these areas. This map was an interpretation of Landsat TM imagery recorded between 2005 and 2011 at a resolution of 30 m (OMNR 2012b). In the Atikaki-Berens population range the Landsat TM landcover represents 16 137 km^2 with 12 079 km^2 overlapping the FRI mapped area. A total area of 873 km^2 remained unmapped, including the far south east portion of the range as well as portions of the far western shore of Lake Winnipeg that was included by the buffer.

In the Kesagami population range I obtained FRI maps covering the Abitibi and Gordon Cosens forest districts from the OMNR (Figure 6). These maps had been interpreted from aerial photographs taken in 1991 and account for 24 188 km^2 of the range. The population extended beyond these FRI maps both to the north in Ontario and to the east in Quebec and here the Far North Landsat TM landcover was used. The Landsat TM accounted for 58 366 km^2 of the nearly 60 000 km^2 range of the Kesagami population, completely overlapping the FRI mapping area.

Stand Classification

FRI forest stands were classified according to the Standard Forest Units as outlined by Elkie et al. (2009) for the north-eastern region (Table 1). Forest stands were assigned the first class whose conditions were met while moving sequentially through the ordered classification (see Table 1). The definitions remained unchanged from Elkie et al. (2009) with the exception of the combination of deciduous stands into a single type whose sole criterion was that they have at least 20% deciduous canopy cover. I maintained the original Landsat TM classification scheme as defined in the original Far North Landcover classification with ecologically similar classes being combined to create a more parsimonious classification of landcover (Table 2). Some FRI maps included forest stands with no species composition information aside from a classification of the dominant canopy species. These stands were infrequent and were classed as if the site contained only their dominant canopy species.

Sampling Design

Determining the accuracy of the maps required comparing the predicted classification, or the map classification, against the “true”, or ground-truthed, classification. These ground-truthed classifications were obtained through extensive vegetation sampling surveys within each map extent, sampling only within the three population ranges. I further restricted stand selection to stands that were at least partially within 1 km of either a driveable road or a water body that intersected a road and was navigable by canoe. Rettie et al. (1997) noted that roads are not built at random; however, they suggested that prior selection of stand types for sampling should remove any potential bias associated with roads or lakes. I further restricted sites to where I could establish a straight 200 metre long transect that did not come within 10 metres of the stand edge. In this restricted subset, stand selection was based on stratified random sampling (Table 1 and 2). Transect start and end locations were determined prior to entering the field and habitat classification labels were removed to prevent bias. In order to maximize sampling efficiency in areas where the two map types (FRI and Landsat TM) overlapped, sites were selected where transects fit entirely in a single stand in both maps. This allowed us to sample both map types with a single transect. Pairings were not restricted to similar class combinations (e.g. deciduous with deciduous) and selection was based on a single map type, with the classification scheme from the second map unknown during selection and sampling. Water and rare landcover classes (<1% of the population range) were not sampled.

Field Methods

A handheld GPS unit was used to locate predetermined sampling locations. Ground measurements were made along a 140 metre transect, stopping every 20 metres to quantify vegetation at eight points per transect. In order to reduce the effects of edge and potential positional inaccuracies from both the map and the GPS, each transect began 10 metres within the forest stand perimeter as defined by the habitat maps. I ensured that the transect remained at least 10 metres from the polygon edge for its entire length. At each of these eight sampling points I marked a 1 x 1 metre plot centered on the transect (Figure A1). In this plot the ground cover for all plant material below 1 metre tall was visually estimated to the nearest percentage point. This included both biotic and abiotic features including all herbaceous plants, grasses, shrubs, trees, lichens, bryophytes, rock, water, bare soil, litter, and fallen woody debris. Fungi were not recorded. Plants were identified to species whenever possible and to genus when not. In certain cases the classification was made to major growth form. This was the case with several mosses and lichens (growth forms provided by Johnson et al. [1995]), ferns, and grasses. Mid-story and canopy composition was estimated using a circular plot with a 4 metre radius centred on the transect. Percentage cover for all shrubs above 1 metre was estimated with growth below 1 metre excluded. Canopy cover composition was determined by counting and identifying all living trees partially or wholly contained within the 4 metre circle. Diameter at breast height (dbh) of each living tree was categorized into size classes to estimate age: 0-10, 10-20, 20-30, and >30 cm. Tree density for each species was calculated using a forester’s prism from the centre of the plot. Standing dead trees were recorded separately from those still living. A densiometer was used to calculate the percentage of canopy cover attributed to coniferous and deciduous trees and tall shrubs. An example datasheet is provided in Appendix C.

In certain cases the selected site was deemed unsafe to survey and a new site was selected. This occurred when water in the site appeared more than waist deep for the entirety of the transect. In certain cases stinging insects were found within or very near the centre of a plot. These plots were shifted 10 metres along the same transect, thus making the transect 10 metres longer while still following the same trajectory.

Accuracy Measurements and Common Map Creation

Accuracy analyses for FRI maps were conducted by province; therefore sites sampled in the Kesagami range were used to represent “Eastern” habitats while sites sampled in both the Atikaki-Berens and the Owl-Flintstone ranges were combined into a larger dataset used to represent “Western” habitats. Sites sampled in the Landsat TM range were combined as the Landsat TM was common across all ranges.

The field results were classified according to the same standard forest unit definitions used to classify the habitat maps. When classification definitions were not provided (as was the case with non-merchantable FRI stands) or were inadequate for a complete classification (as with the Landsat TM classifications), I created definitions that took into account components used in the map classifications as well as features I noted as appearing definitive while in the field (Tables A2 and A3). Therefore, each transect location was assigned both a field classification, based upon characteristics found on the ground, and a predicted classification, based on the information contained within the maps. These classifications were compared and accuracy was assessed using a confusion matrix. Confusion matrices were computed using the mda package (Hastie and Tibshirani 2011) from the statistical program *R* v. 3.0.1 (R Core Team 2013). Kappa coefficient values (agreement rate between the two classification types with a random chance correction [Congalton and Green 1999], which can be viewed as a measure of precision [Viera and Garrett 2005]) were calculated for each confusion matrix using the irr package (Gamer et al. 2012). The Landsat TM and the Eastern and Western FRI map accuracies were investigated individually. Commonly misclassified habitat types were amalgamated into new, broader classifications to increase accuracy. The confusion matrices were computed again to reassess the accuracy of this new classification scheme. I ensured that the creation of the new, broader classifications remained biologically relevant to caribou (Bergerud 1985; Rettie and Messier 2000; Pinard et al. 2012; Proulx 2013). As fewer habitat classifications will always increase map accuracy, restricting the amalgamated classes to biologically relevant classes provided a counterbalance to an assessment based on accuracy alone.

As no single map covered all three caribou ranges in their entirety it was necessary to amalgamate the two map types (FRI and Landsat TM) into a common classification scheme. Using the broader, combined classifications determined above and the area of overlap between the FRI and Landsat TM maps (Figures 5 and 6), I overlaid both maps to determine which class pairings most often described the same forests and represented the largest geographic area. This was used to create a new classification system that allowed for reclassing both the Landsat TM and the FRI maps into equivalent classes and combining them to create a single map with common classifications that encompassed all three caribou population ranges. Just as earlier, this classification scheme also restricted the new classes to biologically relevant ones to provide a counterbalance to an assessment based on accuracy alone. In order to determine how the common classes derived from the FRI maps corresponded with those from the Landsat TM maps a confusion matrix was computed. Where both FRI and

Landsat TM maps overlapped, I created a dataset of random location points, one per square kilometre (8021 points in the Atikaki-Berens range and 24 276 points in the Kesagami range). Using those locations I compared their common classification as derived from the original FRI classes and compared it against those derived from the original Landsat TM classes using a confusion matrix. An additional confusion matrix was computed to reassess the accuracy of the final common map that accounted for all three caribou ranges comparing the ground-truthed sites against their new, common, map classifications. A total of 261 sites were sampled across all three ranges. All 261 sites represented a habitat class type in the FRI with 140 also representing a mapped Landsat TM class (Tables A4 and A5). Thirteen habitat types were sampled for the FRI maps in Manitoba, totalling 136 sites; eleven were sampled in the Kesagami range totalling 125 sites (Table A4). All habitat stands sampled in Ontario represented both an FRI and a Landsat TM habitat type. Fifteen additional Landsat TM stands were sampled in the Atikaki-Berens range surrounding Lake Sasaginnigak. In total, ten habitat types were sampled for the Landsat TM maps, including a single water class that was paired with an open muskeg FRI class.

Figure 1. Location of the three woodland caribou study populations in Canada based on a minimum convex polygon around all caribou GPS locations, plus a 15 km buffer. Figures 2 and 3 show each population in greater detail. (Sources: Esri, DeLorme, NAVTEQ, TomTom, Intermap, increment P Corp., GEBCO, USGS, FAO, NPS, NRCAN, GeoBase, IGN, Kadaster NL, Ordnance Survey, Esri Japan, METI, Esri China [Hong Kong], swisstopo, and the GIS User Community)

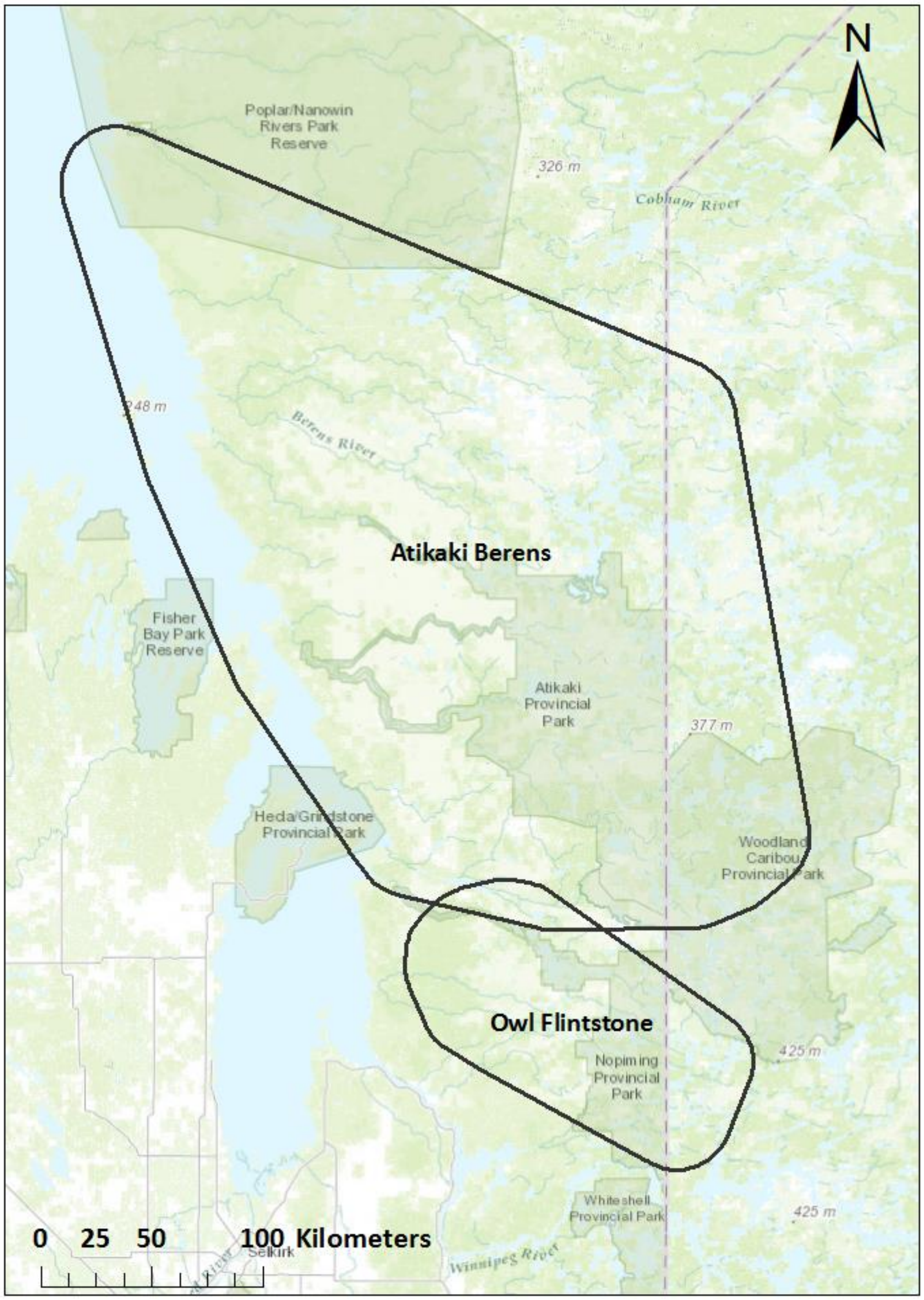

Figure 2. The Atikaki-Berens and Owl-Flintstone population ranges based on a minimum convex polygon around all caribou GPS locations plus a 15 km buffer. (Sources: ESRI-Canada, Natural Resources Canada, Agriculture and Agri-Food Canada, TANA)

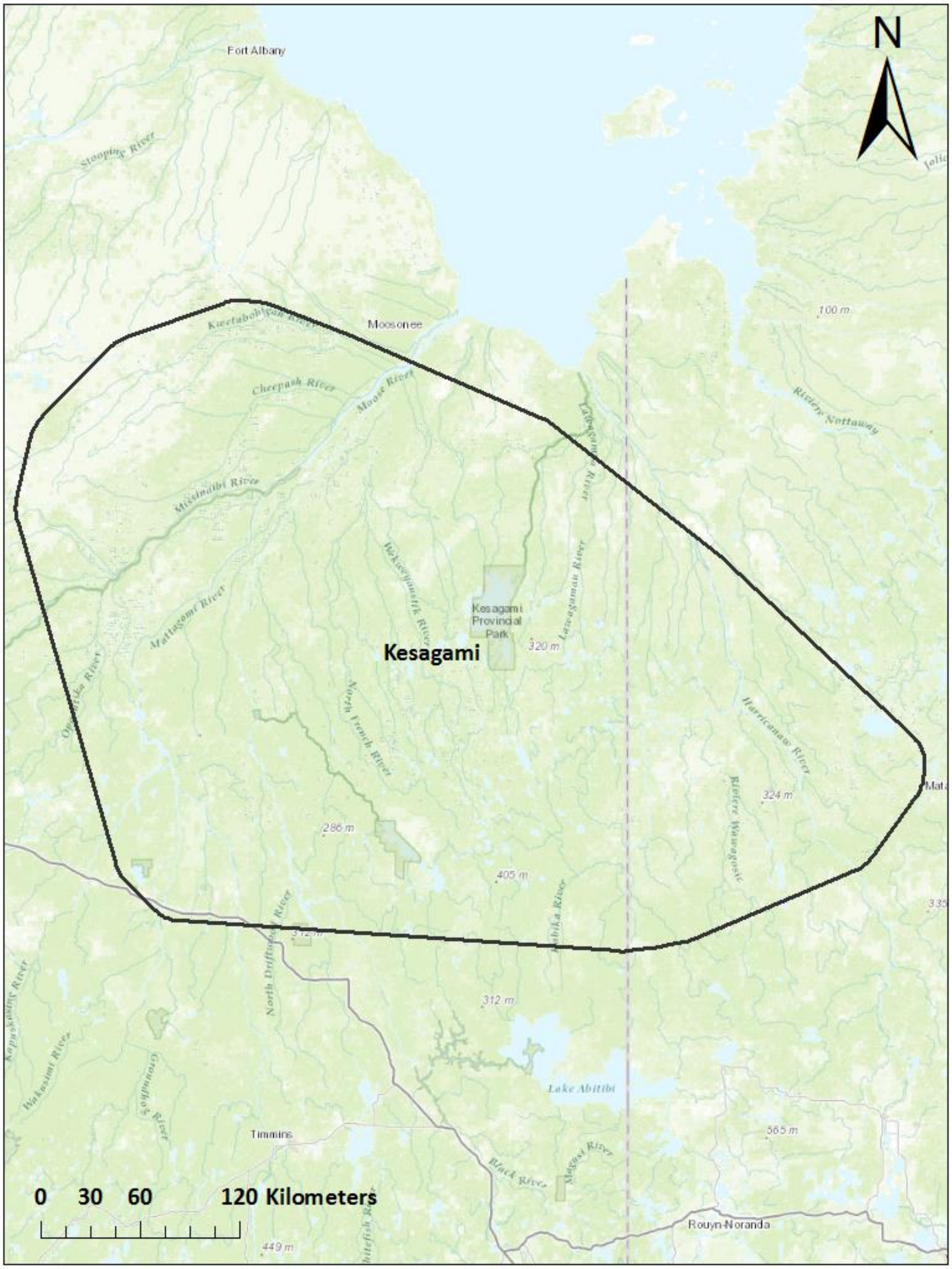

Figure 3. The Kesagami population range based on a minimum convex polygon around all caribou GPS locations plus a 15 km buffer. (Map credit – Sources: ESRI-Canada, Natural Resources Canada, Agriculture and Agri-Food Canada, TANA)

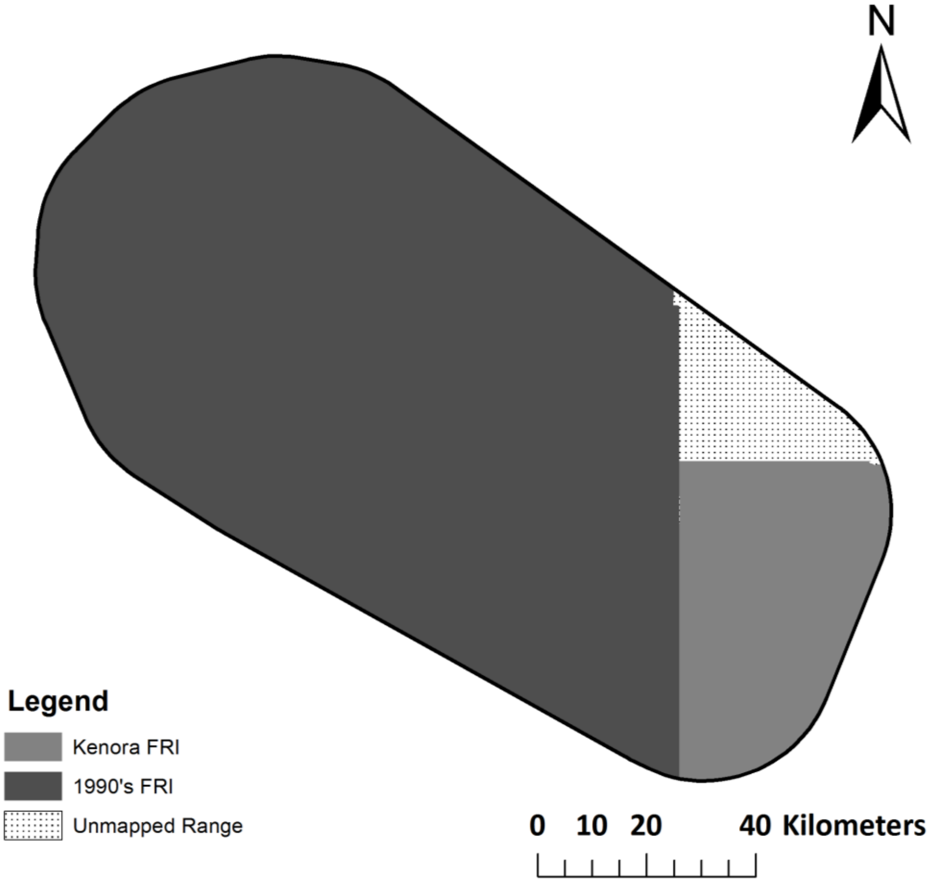

Figure 4. Contribution of each map type used to depict the landcover of the Owl-Flintstone caribou range.

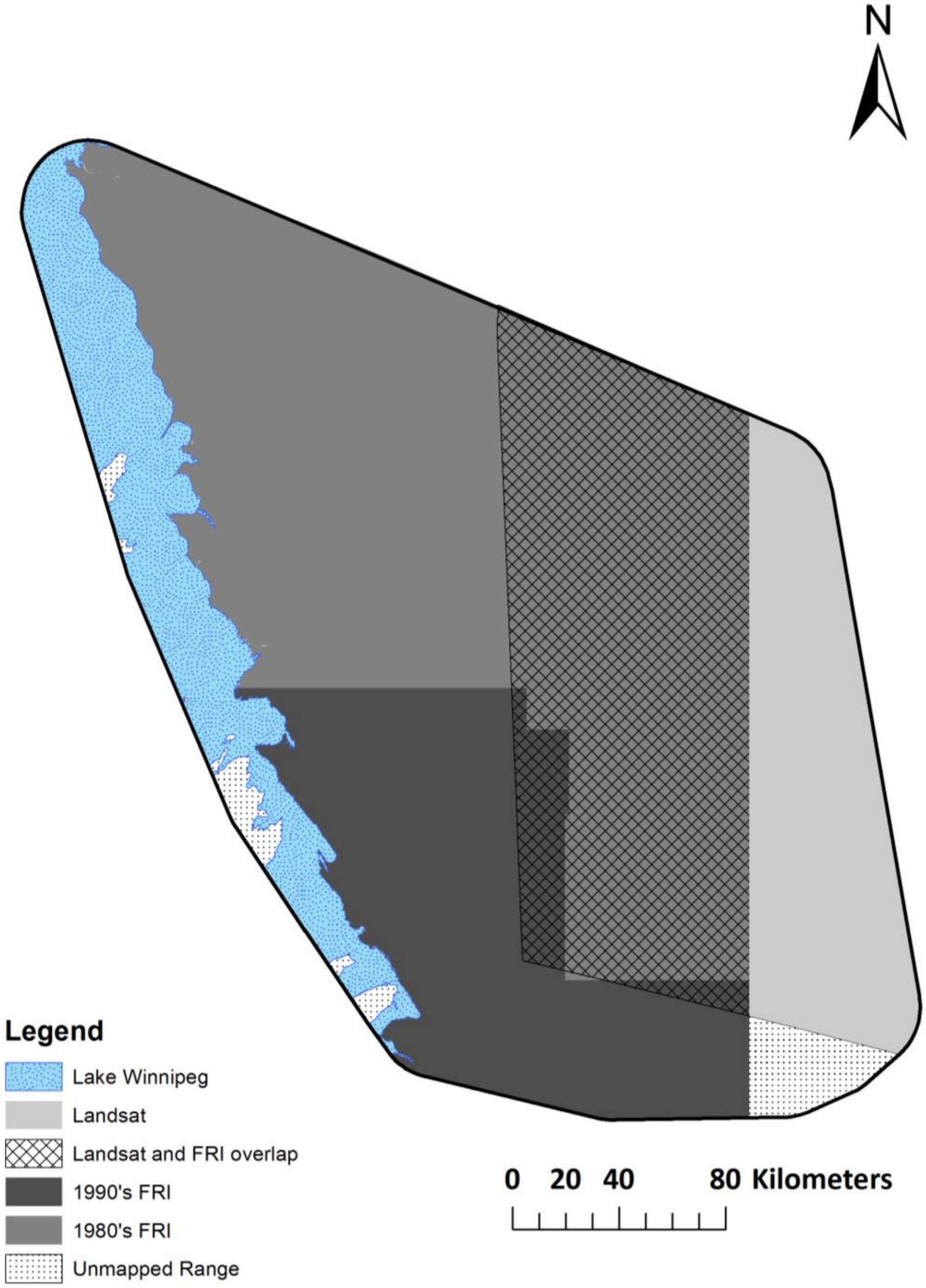

Figure 5. Contributions of each map type used to depict the landcover of the Atikaki-Berens caribou range.

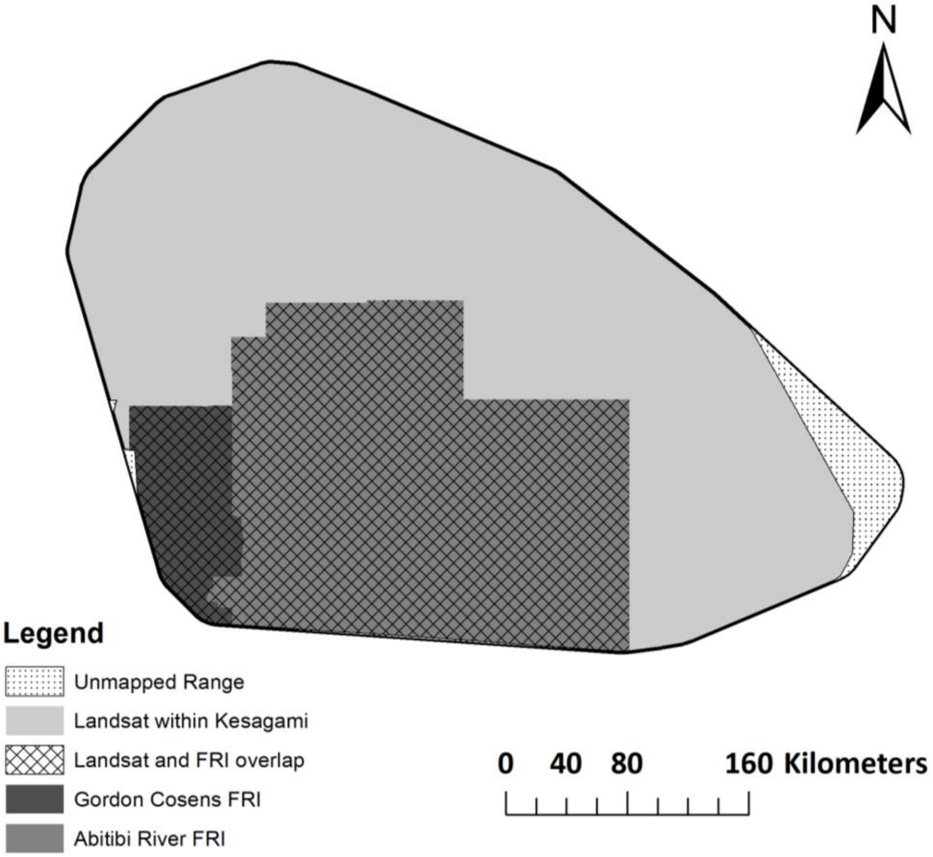

Figure 6. Contributions of each map type used to depict the landcover of the Kesagami caribou range.

Table 1. Modified classification scheme for forest stands in the Forest Resource Inventory maps for Manitoba and Ontario (Elkie et al. 2009).

Classification	**Criteria***	**Classification Order**
Clearcut	Forest stand identified as disturbed through forest harvest within 40 years	1
Disturbed/Young	Forests disturbed within 40 years through natural means (forest fires, disease, insect damage, weather)	
Rock	Classified as Rock within FRI	
Marsh-shrub	Classified as Shrub dominated habitats within FRI	
Treed Muskeg	Classified as Treed Muskeg within FRI	
Open Muskeg	Classified as Open Muskeg within FRI	
Deciduous	Any deciduous >= 20%	
Upland Conifer	RP >= 70% **OR (**(WP +RP + WS + HE) >= 40% **AND** WP > 30%) **OR** ((WP + RP) >= 40%) **OR** ((BS+WS+BF+CE+LA+WP+JP+RP+HE) >= 70% **AND** (BF+CE+WP+LA+WS+HE) >= 20%)	2
Open Muskeg	(BS+LA) >= 70% **AND** SC = 4	3
Black Spruce	(BS >= 80% **AND** (HM + YB + EL + OA + BW + RP) = 0%)	4
Jack Pine	JP > 70%	5
Lowland Conifer	((CE+LA+BS) >= 80% **AND** (HM+YB+EL+OA+BW+RP) = 0%	6
Jack Pine	((JP+BS+RP)>70% **AND** (JP+BS+BF+WS+HE+WP+RP+CE+LA)>=70% **AND** (BF+WS+HE+WP+CE+LA) <=20% **AND** (JP> BS)) **OR** ((JP >= 50% AND (JP+BS+BF+WS+HE+WP+RP+CE+LA)>=70% **AND** (BF+WS+HE+WP+CE+LA)<= 20 % **AND** (JP >= SB))	7
Upland Conifer	(JP >= 30% **AND** (BS+WS+BF+CE+LA+WP+JP+RP+HE) >=70%)	8
Mixed Conifer	(BS+WS+BF+CE+LA+WP+JP+RP+HE) >= 70%	9

*JP = Jack Pine (*Pinus banksiana*), BS = Black Spruce (*Picea mariana*), RP = Red Pine (*Pinus resinosa*), BF = Balsam Fir (*Abies balsamifera*), WS = White Spruce (*Picea glauca*), HE = Hemlock (*Tsuga* spp.), WP = White Pine (*Pinus strobus*), CE = Eastern White Cedar (*Thuja occidentalis*), LA = Tamarack (*Larix laricina*), HM = Maples (*Acer* spp.), YB = Birches (*Betula* spp.), EL = Elms (*Ulmus* spp.), OA = Oaks (*Quercus* spp.), SC = Site Class

Table 2. Amalgamated classes from the Far North Landcover Landsat TM Classification (OMNR 2012a)

Original Landsat TM	**Amalgamated Landsat TM**
Clear Open Water*	Water
Turbid Water*	
Freshwater Marsh*	Marsh-Shrub
Thicket Swamp	
Coniferous Swamp	Coniferous Swamp
Open Fen	Open Wetland
Freshwater Marsh*	
Open Bog	
Treed Fen	Treed Wetland
Treed Bog	
Sparse Treed	Sparse
Bedrock*	
Deciduous Treed	Deciduous
Mixed Treed	
Coniferous Treed	Conifer
Disturbance – Non and sparse woody	Disturbance
Disturbance – Treed and / or shrub	
Sand / Gravel / Mine Tailings*	
Community / Infrastructure*	
Agriculture*	
Cloud/Shadow*	Not included within analysis
Other*	

* Habitat types not ground truthed.

Results

The results of all confusion matrices are summarized in Table 3 and the full matrix for each comparison can be found in the appendix (A6 to A21). The confusion matrices indicated low accuracies when comparing the original habitat map classifications to the classifications derived from the field observations. The FRI maps exhibited a combined accuracy of 46.4% across both provinces (kappa = 0.4; Ontario: 44.0%, kappa = 0.4; Manitoba: 48.5%, kappa = 0.4; Tables A6 to A8). The Far North Landcover Landsat TM classification was slightly less accurate across both provinces with an accuracy of 43.2% (kappa = 0.3; Table A9). When commonly misclassified sites were amalgamated into a coarser classification scheme (Table A10) accuracy increased yet still remained poor. FRI map classification accuracy increased to 63.84% (kappa = 0.6) overall (Table A11; 66.0% in Ontario kappa = 0.6 and 60.3% in Manitoba kappa = 0.6; Tables A12 and A13). After amalgamation (Table A14) Landsat TM map classification accuracy remained weaker than FRI but increased to 61.9% (kappa = 0.5; Table A15).

The creation of the common map resulted in a substantial reduction of site types from thirteen FRI and ten Landsat TM habitat types to six, including Upland Coniferous/Rock, Lowland Coniferous, Deciduous, and Young and two site types that were not ground-truthed (Water and Disturbance; Table 4 and 5). The common map allowed for the correct prediction of an FRI stand classification using the Landsat TM classification 72.0% (kappa = 0.6) of the time (Table A16). This coarse classification also increased mapping accuracy to 69.0% (kappa = 0.6) for FRI maps (Table A17) (68.80% in Ontario kappa = 0.5 and 69.1% kappa = 0.6 in Manitoba; Tables A18 and

A19) and 74.1% (kappa = 0.6) for Landsat TM maps (Table A20). This achieved a correct classification for 69.0% (kappa = 0.6) of the sites ground-truthed.

Table 3. Amalgamated accuracies and kappa values for each confusion matrix in this study.

"Predicted" Observations	"True" Observations	Region	Sites Sampled	Accuracy (%)	Kappa	Number of Classes	Confusion Matrix Location
FRI	Field Observations	East	125	44.0	0.4	12	Table A7
		West	136	48.5	0.4	13	Table A8
		All	261	46.4	0.4	13	Table A6
FRI (only forested sites)			122	51.6	0.4	6	Table A21
Landsat TM Landcover			140	43.2	0.3	9	Table A9
Amalgamated FRI		East	125	66.0	0.6	6	Table A12
		West	135	60.3	0.6	8	Table A13
		All	261	63.8	0.6	8	Table A11
Amalgamated Landsat TM Landcover			140	61.9	0.5	6	Table A15
Amalgamated FRI	Amalgamated Landsat TM Landcover		32297	72.0	0.6	6	Table A16
Common Map derived from Amalgamated Landsat TM Landcover	Field Observations		140	74.1	0.6	4	Table A20
Common Map derived from Amalgamated FRI			261	69.0	0.6	5	Table A17
		East	125	68.8	0.5	4	Table A18
		West	136	69.1	0.6	6	Table A19

Table 4. Common Map Classes and the original FRI classifications that were amalgamated together to create them. The two letter coded stands were present only in Manitoba and were treated as though they consisted of only their title species.

Original Classification*	**Common Map Classification**
Deciduous	Deciduous
Mixed Conifer	
Agriculture	Disturbance
Grassland	
Unclassified	
WS	
Meadow	
Black Spruce	Lowland Conifer
Lowland Conifer	
Marsh-Shrub	
Open-Muskeg	
TL	
Treed Muskeg	
Upland Conifer	
BS	Upland Coniferous/Rock
Jack Pine	
JP	
Rock	
TA	
Small Islands	Water
Water	
Young Conifer	Young
Young Deciduous	

*WS = White Spruce, TL = Tamarack, BS = Black Spruce, JP = Jack Pine, TA = Aspen species)

Table 5. Common Map Classes and the original Landsat TM classifications that were amalgamated together to create them. A number of original classifications existed in the landscape that were present on the landscape at very low quantities. These were included in the final classification but were not ground-truthed and therefore are not present in previous classifications.

Original Classification	**Common Map Classification**
Deciduous Treed	Deciduous
Mixed Treed	
Thicket Swamp	
Agriculture	Disturbance
Community	
Sand/Gravel/Mine Tailings	
Coniferous Swamp	Lowland Conifer
Coniferous Treed	
Open Bog	
Open Fen	
Treed Bog	
Treed Fen	
Treed Peatland	
Bedrock	Upland Coniferous/Rock
Sparse Treed	
Freshwater March	Water
Turbid Water	
Clear Open Water	
Disturbance – Non and Sparse Woody	Young
Disturbance – Treed and/ or shrub	

Discussion

Forest Resource Inventory maps using slightly modified standard forest unit definitions as defined by the OMNR (Elkie et al. 2009) and the Far North landcover classification of Landsat TM maps appear to be less than 50% accurate in both provinces and all areas sampled. When forest classifications were merged based on both common biological features and commonly misclassified habitats, accuracy increased substantially to a moderate level of agreement with ground-truthed sites (69.0%). Similar levels of agreement were shown when using either map to predict their counterpart (i.e. the FRI derived common map compared against the Landsat derived common map and vice versa). Kappa values for the final classification scheme (0.6) also indicated a moderate level of agreement (Viera and Garrett 2005) which I considered acceptable.

Few published studies have investigated the accuracy of either FRI maps or Landsat TM landcover classifications in the boreal forest of Canada. To my knowledge there have been two published reports of FRI map accuracy. Thompson et al. (2007) investigated FRI maps in the boreal forest of Ontario and reported that 36% of all stands were correct with respect to species composition and 70% of those were correctly classified in the much broader forest categories of coniferous, deciduous, or mixed. Dussault et al. (2001) found similar accuracy levels in the FRI when they sampled 186 boreal forest stands in Jacques-Cartier Park in Quebec. When investigating species composition of broad classifications of coniferous, deciduous, and mixed, accuracies ranged from 40 to 74%**.** It should be noted, however, that an investigation of both FRI and Landsat TM maps by Brown et al (2006) indicated that these maps proved successful in accounting for variation in vegetation communities that are considered valuable to caribou. While not representing an accuracy measurement per se, it appears that both FRI and Landsat TM maps accounted for approximately 80% of the variation in vegetation communities.

These low accuracies were not limited to studies in the boreal forest of central Canada. Maxie et al. (2010) investigated both FRI maps and Landsat TM classifications in two areas of south-central Ontario. Their results showed FRI accuracies that were similar to my study ─ from 44% to 48% for forested sites. Including wetlands, the majority of which were sampled from fixed wing aircraft, and non-merchantable stands the accuracy increased to 63% and 77%. Pinto et al. (2007) estimated accuracy at approximately 55% for their classification in the Nipissing forest in Ontario, located directly south of the Kesagami study range.

I found no reports of ground-truthing the Far North Landcover Landsat TM classification, likely as a result of the relatively recent [2012] creation of the classification. Studies into older Landsat TM classifications show a wide range of accuracies, from 54% to 89% in a variety of habitats in Canada. These studies investigated aerial photographs from British Columbia (Johnson et al. 2003), Algonquin Provincial Park (Maxie et al. 2010) and the Bruce Peninsula (Coady 2005) in Ontario. Large-scale studies of the National Landcover Database in the United States using high resolution imagery instead of field ground-truthing as a reference resulted in accuracies of 38% to 85% dependent on the region, habitat class, and map vintage (Wickham et al. 2004; Wickham et al. 2010; Wickham et al. 2013).

The accuracies that I found for both map types were far lower than the minimum overall accuracy goal of 85% suggested by Thomlinson et al. (1999). However, Foody (2008) suggested that the arbitrary decision that accuracies below 85% may not be suitable in all instances and that common methods of accuracy assessments are unnecessarily harsh. Foody (2008) also suggested that map accuracy is often, in truth, likely higher than presented in the literature. Particularly pertinent to my study was the suggestion that ground-truthing should not be taken as error-free as it usually is in studies of this type and that results should be strongly scrutinized. Additionally the choice of experimental design and sampling protocol may strongly influence results. Comparing my accuracies to Basterfield (2012) shows the likely influence of experimental design on accuracy assessments. Basterfield (2012) reported an accuracy of 88% for sites sampled in the Owl-Flintstone population range in southern Manitoba. This is in stark contrast to the accuracies I reported. This project's method of ground-truthing and its habitat definitions differed significantly from Basterfield (2012) yet both studies investigated the same range and the same FRI maps within 2 years of each other. Such a marked difference in accuracies, presumably as a result of field sampling methodology, should serve as a warning to future researchers to create a field protocol that best matches the original interpreter's map classification methods to ensure accurate representation. Other studies, employing substantially different methodology, included Van Beest (2010) whose study was concerned only with stand cutting class (tree height) and dominant tree species. They obtained an accuracy of 94.8% which is far greater than I found. However, I felt this level of detail in the habitat maps was inappropriate for my study. Similarly high accuracy rates are often reported when using higher resolution aerial photography to ground-truth coarser habitat maps. Wickham et al. (2010, 2013) as well as Fent et al. (1995) reported accuracies higher than most reported for ground-truthed studies with the lowest accuracy of 67.6%. The possible influence of study protocol on the accuracy assessment is strong and indicates a need for researchers to consider how their sampling methods could influence their results and ensure that the protocol is in line with both the creation methods of the map they are investigating, and how they wish to use the maps in their own studies.

Despite the apparent low accuracy of the maps, the rates I reported here are comparable to many of the cited studies. Thompson et al. (2007) suggested that researchers using FRI maps should expect an error rate of 30-60% depending on the classification scheme. Wilkinson (2005) investigated 15 years of accuracy assessments of satellite imagery classifications from 500 different studies and reported a mean kappa coefficient of 0.7 which, while not strictly a measure of accuracy per se, is comparable to the kappa coefficient of these maps (0.6). This seems to suggest that the accuracies found in my study are comparable to other studies that used similar maps.

While all attempts were made to control error and bias, I have determined six potential sources of error in my design, some avoidable and some not. They were: distance to edge of forest stand, age of FRI maps, a complicated classification scheme, classification of non-merchantable stands, small sample size, and a poor measure of forest age. I set 10 metres as the distance from the stand edge to each transect start location. This was based on the precision of the FRI maps (OMNR 2009) and personal experience with the positional accuracy of handheld GPS units. However, it became obvious during field work that increasing this distance would limit the impact of slight mapping errors in the FRI and Landsat TM maps, as well as reduce the influence of edge effects (Harper et al. 2005) or the difficulty in maintaining a straight transect line due to the uneven terrain or thick forest brush. Maxie et al. (2010) used a distance to edge of 50 metres. Although increasing the distance would reduce the number of potential sites that meet selection criteria, it would likely have increased accuracy across all maps and study areas. Moreover the impacts from this short distance to edge would likely have been compounded by the increasing age of the FRI maps. As the time between map creation and ground truthing increases, there is a greater potential for structural change in the habitat (Dussault et al. 2001; Pinto et al. 2007; Thompson et al. 2007). It has been said that boreal forest species composition changes little beyond 40 years of age (Thompson et al. 2007); however, a forest would be more likely to change at its edge before its central core (Harper et al. 2005).

Difficulties in classifying non-merchantable stands also may have increased error. Forested stands were classified using canopy species composition and according to the standard forest units for eastern Ontario (Elkie et al. 2009). However, as the species composition of non-merchantable stands was not provided in FRI maps, the field data classification of habitats such as open muskeg, treed muskeg, and marsh-shrub were based on classifications of my own creation from personal experience. By removing these non-merchantable stands from the original classification, stand accuracies increased slightly to just above 50% (kappa = 0.4; Table A21). Original Landsat TM classifications in the Far North Landcover, while created through methods impossible to emulate in the field, provided descriptions of what each class should represent and allowed for an on-the-ground classification scheme creation (OMNR 2012a). While classifying these non-merchantable FRI sites was difficult, this study was the first to my knowledge that rigorously sampled non-merchantable stands in an identical manner to forested sites in central Canada to determine their classification accuracy (Dussault et al. 2001; Pinto et al. 2007; Thompson et al. 2007). Brown et al. (2006) sampled peatlands in their assessment of FRI and Landsat TM maps' ability to account for variation in community vegetation while Maxie et al. (2010) sampled a small portion of what they referred to as "wetland sites" by ground but stated that the majority of these sites were sampled by air.

Leckie and Gillis (1995) and Potvin et al. (1999) showed that sites that had a simple species composition (one or two species) were most often correctly classified and that incorrect classification increased as the complexity of the species composition increased. Standard forest units have the potential for complicated stands, incorporating multiple species in each classification type. In my study 31.5% of stands contained a mixed canopy of more than two species representing greater than 10% of the trunk density. Additionally, this study used diameter at breast height as an approximation of tree age. Species grow and age differently (Gutsell and Johnson 2002) and growth rates of individual species will vary in different environments (Tappeiner et al. 1997) and, as such, using standardized diameter measurements is likely too rough to accurately represent stand age. However, as Dussault et al. (2001) showed, the age of forest stands is often the most accurate feature of FRI maps. Therefore the likely inaccuracies of my ageing methodology may be at least partially ignored. Furthermore, despite the scope of this study, there is potential that I did not sample enough sites, and that the accuracy reported may be influenced by this lack of precision. Congalton (1991) suggested that a minimum number of sampled sites for large areas should be no fewer than 75 for each class. I sampled far fewer than this and as a result the accuracy may not truly reflect the map's ability to predict the landscape composition.

Despite these potential limitations in my field work, the accuracies that I estimated for both Landsat TM and FRI maps were not much different from those reported in other studies (Dussault et al. 2001; Johnson et al. 2003; Coady 2005; Pinto et al. 2007; Thompson et al. 2007; Maxie et al. 2010). It can therefore be assumed that the majority of the error exists in the maps themselves or in commonly accepted methods of ground-truthing maps, rather than errors specific to my study. Multiple reasons for inaccuracies in the maps themselves have been suggested. The most likely causes of within-map error include the fact that canopy cover is not uniform across species, with certain species representing a greater percentage of the canopy than their stem density would suggest (Thompson et al. 2007). As a result the density and quantity of those species may be over-estimated. This would also at least partially explain why studies that use high-resolution photography to ground-truth maps achieve higher levels of accuracy, as their ground-truthing methods would still suffer the same inability to count tree stems (Fent et al. 1995; Wickham et al. 2010; Wickham et al. 2013). In addition, trees that are often not in the canopy, such as balsam fir and eastern white cedar may be underrepresented in estimations as they would not show up in aerial photographs as often as they are actually present (Pinto et al. 2008). As a result, complex forests with multiple species of trees are more easily misclassified (Leckie and Gillis 1995; Potvin et al. 1999). It has also been suggested that age, stress, and tree orientation can also influence correct identification (Ciesla 1990 in Thompson et al. 2007). Accuracy can be further reduced by technological limitations (Fent et al. 1995) and as some of the FRI maps are nearly 30 years old, they were created with technology less powerful than that which is available today and which was used to create the Far North Landcover Landsat TM maps.

Both FRI and Landsat TM maps are popular data sources that are often used in habitat studies (Rempel et al. 1997; Hillis et al. 1998; Schindler 2006; Brown et al. 2007; Dyke 2008; Bowman et al. 2010; Kuemmerle et al. 2010) and an understanding of their strengths and limitations is vital. The accuracy of all maps involved within this study fall well below the 85% accuracy mark suggested by Thomlinson et al. (1999). Other remote sensing technology appears to provide accuracy levels that are similar, including 61-74% for synthetic aperture radar (Wolldersheim et al. 2011), 75% for compact airborne spectrographic imager (Franklin et al. 2001) and from 64-77% for LiDAR (Suratno et al. 2009; Brandtberg 2007; Moffiet et al. 2005).

This study shows that the accuracy of both FRI and Landsat TM maps appears low, regardless of their date of creation, and demonstrates the importance of accuracy assessments. As there exists large differences in accuracy based on both classification scheme and assessment methods (Basterfield et al. 2012) future studies using either newly created maps or untested classifications should endeavour to ground-truth and identify an accurate yet relevant classification scheme. While these data will likely not prove to be ground-breaking, understanding the accuracy of the landcover maps will serve to strengthen the core assumptions on which these studies are based. Futhermore, as there appears no single mapping technique that is consistently more accurate than others, future studies may consider map accuracy secondary to the required features when selecting their maps. The fine-scale resolution of the FRI maps may be best suited for habitat selection studies despite their age. Basing habitat classifications on simple yet biologically relevant combinations of features will likely result in more reliable representation of the landscape.

Chapter 3: The Influence of Habitat on Woodland Caribou Site Fidelity

Introduction

Boreal woodland caribou are considered one of the most threatened populations of caribou in Canada (Mallory and Hillis 1998) having experienced extensive range retraction in North America since the turn of the 20th century (Wood 1917; Jackson 1922; Johnson 1922; Snyder 1928; Dymond et al. 1928; de Vos and Peterson 1951; Bergerud 1974; Schaefer 2003). This range retraction is thought to be caused by increased pressure by predators and habitat loss facilitated by human disturbances coupled with a history of overhunting (Bergerud 1974; Dalerum et al. 2007; McCarthy et al. 2011; Wasser et al. 2011; Dussault et al. 2012; Fortin et al. 2013). Modifications to the landscape, both anthropogenic and natural, alter it in ways that are often to the detriment of woodland caribou. These modifications can increase predator numbers (Bergerud 1974; Bergerud and Elliot 1986; Latham et al. 2011b) and variety (Brodeur et al. 2008) often as the result of apparent competition (Maier et al. 2005) caused by changes in the amount of preferred landcover (Hébert and Weladji 2013). They can also directly influence caribou mortality through stress and behavioural changes (Harrington and Veitch 1992). In 2000, woodland caribou were listed as threatened by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC 2002) and with continued boreal forest development (Timilsina et al. 2005; Islam 2008; Mining Association of Canada 2012; Natural Resources Canada 2013) the future viability of many populations remains unclear (Dymond et al. 1928; Bergerud 1974; Ceballos and Ehrlich 2002; Festa-Bianchet et al. 2011).

Behavioural strategies, such as site fidelity, that are hypothesized to improve fitness may be affected by habitat alteration (Schaefer et al. 2000; Faille et al. 2010). Site fidelity is a common behaviour among many species (Greenwood 1980; Shields 1984; Lanyon and Thompson 1986; Switzer 1997; Matthiopoulos et al. 2005) and refers to the tendency for an animal to return to approximately the same location, often in consecutive years, and may be related to habitat condition (White and Garrott 1990; Switzer 1993; Faille et al. 2010). Woodland caribou habitat selection is driven by an avoidance of predators (Stuart-Smith et al. 1997; Gustine et al. 2006), apparent competitors (Seip 1992), and human activity (Fortin et al. 2013) while selecting habitats capable of providing adequate food and cover (Dussault et al. 2012) that also allow for predator escape (Ferguson and Elkie 2005). These preferred habitat types include old-growth conifer forests, treed muskegs (Rettie and Messier 2000), islands, and shorelines (Bergerud 1985), while caribou avoid disturbed or young forests (Dalerum et al. 2007; Fortin et al. 2013) and deciduous forests (Frelich and Reich 1995; Festa-Bianchet et al. 2011). Site fidelity is therefore thought to maintain habitat selection from year to year (Greenwood 1980) with animals returning to locations that maximize their individual fitness and likelihood of successful calf recruitment and avoiding those of poorer quality (Burger 1982; Badyaev and Faust 1996; Hoover 2003).

Inter-annual site fidelity has commonly been shown in many different populations of woodland caribou with the strongest fidelity occurring during the calving and post-calving season (Schaefer et al. 2000; Ferguson and Elkie 2004; Wittmer et al. 2006; Tracz et al. 2010; Popp et al. 2011; but see Rettie and Messier 2001). A degree of site fidelity, albeit weaker, has been noted during the breeding season with winter commonly considered the period of time when fidelity is weakest (Schaefer et al. 2000). Calving and post-calving periods are the most limiting for a female caribou (Dussault et al. 2012) and many other species show increased site fidelity when they reproduce successfully and recruit young into the population (Burger 1982; Hoover 2003; Sedgwick 2004). At least one study shows that this appears to be true in woodland caribou as well (Faille et al. 2010).

When a caribou does not return to the same location as the previous year, the possibility exists that the caribou may shift their location to more appropriate habitat. Caribou move away from poorer habitats in terms of individual fitness and calf recruitment (Fortin et al. 2013), such as young or deciduous forests (Frelich and Reich 1995; Dalerum et al. 2007; Festa-Bianchet et al. 2011; Fortin et al. 2013) and move to preferred forest types such as mature conifer forests (Rettie and Messier 2000). I would anticipate, therefore, that the distribution of caribou on the landscape would result from fewer caribou near disturbances with higher concentrations in the more intact forests (Fortin et al. 2013). There, however, exists the possibility that caribou may show a directional change in habitat use with caribou showing consistent habitat selection, and returning to areas of similar environmental condition. While this behaviour would include caribou that are showing strong site fidelity, it would also include animals that are showing weak site fidelity but are returning to areas of similar composition. This means that a caribou who calves 50 km away its previous calving location but that again chose to calve on a small island or in peatland would show weak site fidelity, but strong fidelity to specific environmental conditions.

To my knowledge an investigation of the effect of habitat and human disturbances on site fidelity at the fine scale of daily or weekly use has yet to be undertaken. This study will enhance our understanding of site fidelity and how the landscape influences its strength during key periods throughout the year. I analyzed site fidelity and directional changes in habitat use based on GPS tracking of 72 adult females, in relation to habitat conditions for three populations of woodland caribou in Ontario and Manitoba.

Hypothesis and Predictions

The goal of my study was to investigate three main aspects of woodland caribou site fidelity. The first aspect was to determine the relative strength of site fidelity across seasons. I anticipate that site fidelity will vary seasonally with the changing needs of the caribou and that site fidelity will be strongest during the calving and post-calving season when compared to the winter period (Schaefer et al. 2000; Ferguson and Elkie 2004; Wittmer et al. 2006; Tracz et al. 2010; Popp et al. 2011). Second, I anticipate that the strength of site fidelity is directly related to the surrounding landscape. As such, I hypothesize that disturbances, both anthropogenic and natural, will reduce the strength of site fidelity where caribou occur in close proximity to them, whereas I believe that proximity to intact preferred forest types, such as mature old growth conifer forests (Rettie and Messier 2000), will serve to maintain or increase site fidelity. Thirdly I anticipate that female caribou will show a tendency for consistent directional, year-to-year changes in habitat use, moving away from disturbances and remaining in, or selecting beneficial intact habitats, regardless of the strength of their site fidelity.

Methods

Study Areas and Populations

Each of the three caribou ranges were defined as a minimum convex polygon (MCP) surrounding all GPS location points for each population plus an additional 15 km buffer to account for habitat and disturbances outside of the MCP (Smith et al. 2000; Schaefer and Mahoney 2007; Vors et al. 2007).

In Manitoba I investigated the Atikaki-Berens and Owl-Flintstone populations (Manitoba Conservation 2005) and in Ontario a northern segment of the Kesagami population (Environment Canada 2012). The three populations each represent a different assessment of self-sustainability: Atikaki-Berens is “likely self-sustaining”, Owl-Flintstone is “as likely as not”, and the Kesagami population is “very unlikely” (Environment Canada 2012).

Atikaki-Berens Population and Range

Atikaki-Berens boasts the largest population of woodland caribou in Manitoba with an estimated 300 to 500 animals (Manitoba Conservation 2005). I obtained Global Positioning System (GPS) locational data from 23 females between 2000 and 2010.

The Atikaki-Berens range encompassed an area of approximately 30 000 km^2 including the 15 km buffer (Figure 2). The majority of the range occurs in Manitoba with the far eastern edge extending approximately 20 km into Ontario. The range is characterized by uplands with thin soils dominated by black spruce and jack pine with exposed rock-ridges separating depressions often forming into narrow lakes and poorly drained peat-filled bogs with black spruce and tamarack lowlands (Rowe 1972; Schaefer

1996). Nearer Lake Winnipeg the occurrence of deciduous forests increases as drainage increases (Rowe 1972).

Anthropogenic disturbances are limited, with much of the disturbance located at the southern edge, which includes the community of Bissett (Figure 7). Logging is heavily concentrated along Highway 304 with some occurring along the Rice River road which runs north, parallel to Lake Winnipeg. Forests disturbed by logging in the past 40 years account for less that 1% of the total range (151 km^2). A transmission line follows the shore of Lake Winnipeg with a branch bisecting the range. In total it's estimated that 5.4% of the range is disturbed through anthropogenic causes (Callaghan et al. 2011).

Forest fires are common in Manitoba and burns younger than 40 years old account for nearly a quarter of the Atikaki-Berens range, at nearly 7 000 km^2. This is shown on the ground by the common occurrence of jack pine, a species intrinsically linked to fires (Chrosciewicz 1988; Schaefer and Pruitt 1991).

Owl-Flintstone Population and Range

The Owl-Flintstone population is the most southern woodland caribou population in Manitoba (Callaghan et al. 2011). Despite its high concern conservation status, Owl-Flintstone has often been remarked upon due to its apparent stability despite the relatively high levels of both anthropogenic and natural disturbances on the landscape (Hummel and Ray 2008; Environment Canada 2012). The estimated population levels have remained stable since the early 1980s (Schindler 2006) with current estimates between 71-85 animals (Callaghan et al. 2011). I obtained GPS locational data from 27 females between 1995 and 2010.

The Owl-Flintstone population range is located south of the Atikaki-Berens range. It totals just over 5000 km^2 including the 15 km buffer and extends a short distance into northwestern Ontario (Figure 2). It is compositionally very similar to the Atikaki-Berens population range featuring a landscape of uplands with thin soils with black spruce and jack pine stands interspersed with exposed rock separating poorly drained black spruce and tamarack forests and sphagnum filled boggy lowlands and long narrow lakes (Rowe 1972). In the west of the range, deciduous forests become increasingly common as drainage increases as the landscape flattens (Rowe 1972).

The Owl-Flintstone range represents a far more disturbed habitat than the Atikaki-Berens range (Figures 8). Forest harvest is common, with an area of 256 km^2, or approximately 5% of the range harvested between 1970 and 2010. As a result, much of the range is also criss-crossed by harvest roads, as well as Highway 304 that loops through the western half of the range. Due to this ease of access, many of the large lakes that are accessible by road are frequented by tourists and can be quite busy during the summer months (personal observations 2012). Little anthropogenic disturbance occurs on the far eastern edge in Ontario as well as in the centre of the range where harvest and roads are infrequent. In total it is estimated that 23.8% of the total range is disturbed through anthropogenic causes (Callaghan et al. 2011).

Fire patterns in the Owl-Flintstone range are similar to those of the Atikaki-Berens range and burns younger than 40 years account for greater than 25% of the range, representing 1 270 km^2.

Kesagami Population and Range

The newly designated Kesagami population of woodland caribou has an estimated population of 492 individuals (Environment Canada 2012). The Kesagami population has the largest range of those currently assigned in Ontario and experiences widely variable levels of disturbances along a latitudinal gradient (OMNR 2012b). Snyder (1928) commented on the extirpation of woodland caribou from the region around Lake Abitibi as early as 1921, while Cumming and Beange (1993) estimated the minimum number of caribou in the commercial forest surrounding Cochrane to be 50 animals. I obtained GPS locational data from 22 females between 2010 and 2013 which represented a portion of the northern part of the Kesagami population.

The Kesagami range represents nearly 60 000 km^2 including the 15 km buffer, the majority of which falls in Ontario, with the far eastern edge occurring in Quebec (Figure 3). Structurally the Kesagami range is significantly different from both Manitoba populations. It is an area of sedimentary rock covered with marine clay from glacial depositions, appearing as a vast landscape of poorly drained flats (Rowe 1972; Carleton and Maycock 1978) that slowly drain northward through small clay lined streams (Rowe 1972) into James Bay (Bergeron 2000). Sphagnum-dominated bogs and muskegs, fens, and black spruce forests cover the land with the occasional hardwood or mixed-wood stand in areas of increased drainage. Exposed rock is exceedingly rare (Rowe 1972).

Animals in the Kesagami population experience a latitudinal gradient of disturbances. Dense aggregations of roads and logged forests span the southern third of the range (Figure 9) with a strip of disturbance bisecting the range, following the railway connecting Cochrane with Moosenee. Anthropogenic disturbances appear more commonly in Quebec than in Ontario; however, anthropogenic disturbance is essentially non-existent in the northern portion of the range in both provinces. In total logged forests represent nearly 10% of the range with 5 821 km^2 having been logged in the past 40 years.

Forest fires are rare in the Kesagami range compared to the two Manitoba populations with only 4% (2 411 km^2) having burnt in the past 40 years.

Telemetry Data Preparation

In total, 467 692 GPS location points were obtained from all three populations (Table 1A). These points were filtered for accuracy using Keating's ξ, a method of using animal movement distances and directionality to identify unrealistic and inaccurate locations (Keating 1994). Rejecting locations where $\xi \geq 5$ km (following Rettie and Messier [2001]) resulted in discarding approximately 16% of the data, or 29 761 individual points.

Location points were further filtered to one point every 25 hours. This ensured an approximately equal sampling schedule as well as not restricting data collection to a single time of day (as one point every 24 hours would). The final number of points used in the analysis was 42 783.

Seasons

I investigated fidelity in five biologically relevant, week-long periods that corresponded with important phases of a caribou year: Winter, Calving, Early Post-Calving, Late Post-Calving, and Breeding.

Using movement data estimated from GPS collar locations, I applied known movement behaviours around calving to determine the date when each caribou most likely gave birth. Ferguson and Elkie (2004) noted that caribou show a reduction in movement coincident with the timing of calving. I defined the calving event as the first consecutive 3-day period where each day's movement was less than 500 m between calendar days 121 and 180, the traditional calving period of May and June (Wilson 2013). If a calving event could not be determined, or the individual was lacking a significant number of points (fewer than 35 points for the entire period), I assigned them a calving date from the remainder of the population, in most cases using the average calving date for that calendar year. If no calving events were determined for that year, the animal was assigned the average of its own individual calving events. If neither of these options were possible, the average value across all animals across all years for that population was used. When assigned an averaged date for the calving event, the location point from the nearest day (± 5) was used. This resulted in a total of 2003 locations across all populations for all five seasons.

Calving – Calving was determined as the period of one week immediately following the calving event. This was set to correspond to the expectation of increased calving site fidelity (Schaefer et al. 2000), a reduction in movement (Ferguson and Elkie 2004), and an increase in mortality associated with calving (Whitten et al. 1992; Dussault et al. 2012).

Early Post-Calving – Early Post-Calving was the period of one week beginning four weeks after the calving event. It was set to correspond to high calf mortality rates associated with the first month of life (Stuart-Smith et al. 1997; Gustine et al 2006;

Pinard et al. 2012). It also corresponds to the evidence that woodland caribou exhibit the strongest fidelity to post-calving grounds (Schaefer et al. 2000; Schaefer and Mahoney 2013).

Late Post-Calving – Late Post-Calving was the period of one week beginning seven weeks after the calving event. It was set to correspond to a period of decreased calf mortality (Pinard et al. 2012) and increased independence and movement rates (DeMars et al. 2013).

Breeding – Breeding was determined as the period of one week centred on 229 days before the calving event. This was based upon the average gestation period of woodland caribou (Bergerud 1975). This period was selected to represent an important period of life history for the woodland caribou.

Winter – Winter was represented by a set period based strictly on fixed calendar dates. It was meant to represent the core of winter, during which caribou aggregate into small groups and habitat selection is different than during the summer at least in part due to snow conditions (Stardom 1975; Stuart-Smith et al. 1997; Brown 2005; Wittmer et al. 2006). It was a period of one week between days 32 and 38. (1-7 February)

Site Fidelity

Fidelity was determined for each individual by calculating the distance between all consecutive-year pairs of locations during each season and determining the average distance. Each season contained a maximum of seven days; each date was numbered based on the time since the start of the season rather than its order. Therefore a season may contain only two days, but may represent day two and seven depending on the time since season initiation. A day was determined to be absent if no location point existed within 25 to 30 hours after the previous point. Only points with the same day assignment were compared and seasons with fewer than two comparisons were removed from later analyses. Fidelity for that season was expressed as the average of all the distance measurements contained within that season.

Site fidelity was also assessed during the biological year. Yearly site fidelity profiles were created for each animal beginning at breeding and ending 300 days later (to account for variability in breeding dates between years). This “biological year” tied fidelity directly to the biology of the caribou instead of strict calendar dates. I expressed site fidelity as the mean distances between all pairs of locations during each 7-day period in consecutive years. I omitted animals with fewer than two points during a seven day period from the analysis.

Habitat Classifications

Woodland caribou home ranges are often very large and with three study areas this required the use of several habitat maps. All three populations were, at least in part, represented by Forest Resource Inventory maps (FRI) including the entirety of the Owl-Flintstone range and large portions of the Atikaki-Berens and Kesagami ranges (Figures 4to 6). These maps were provided by Manitoba Conservation and the Ontario Ministry of Natural Resources (OMNR). Classifications of the FRI maps were based on a modification of the standard forest units as outlined by Elkie et al. (2009) for the north-eastern region (Table 1) and were determined using canopy cover species composition. In order to increase mapping accuracy, the forest stands were combined into three undisturbed habitat classes (Table 4): Lowland Conifer, Upland Coniferous / Rock, and Deciduous. The remaining area of the Atikaki-Berens and the Kesagami ranges were derived from Landsat Thematic Mapper (TM) landcover provided from the OMNR. Classifications in the Landsat TM were based upon provided classifications and custom combinations to represent vital caribou habitat classes while also maximizing accuracy. Classifications comparable to those defined in the areas covered by FRI maps were used: Lowland Conifer, Upland Coniferous/Rock, and Deciduous (Table 5).

Landscape Disturbances

Disturbance information was provided by the provincial governments for any portion of the range contained in their jurisdictions. All attempts were made to create a common set of classifications across all three ranges. For certain disturbances such as roads, this required both advice from provincial contacts and first-hand experiences in the field (Table B1).

In total I considered ten different disturbances: Railroads, power lines, three categories of roads (based on permanence and use), communities, mines, forest harvesting, forest fires, and natural non-fire disturbances such as disease and insect damage.

Mines, communities, roads, railroads, and power lines were considered permanent once created and they were retained on all maps after their creation. Temporary, non-linear disturbances such as forest fires, forest harvesting, and natural non-fire disturbances were considered as disturbed for 40 years after their occurrence. After 40 years these forests were considered to have reached a level of structural similarity with mature forests and were deemed suitable as caribou habitat (Schaefer and Pruitt 1991; Rettie and Messier 2000). At that point their inclusion in a disturbance category was no longer merited (Racey et al. 1991; Dunford et al. 2006; Thompson et al. 2007). These forest stands were then treated as intact forests and classed according to the information provided in the original maps.

Due to differences in both caribou and wolf use, rivers and lakes were considered as separate classes (Bergerud 1985; Latham et al. 2011a). Lakes were determined as water bodies larger than 5 hectares (Ferguson and Elkie 2005) with a shoreline to area ratio of greater than 0.012 m/m^2 (determined through trial and error). In the Kesagami range, several large rivers, such as the Moose River, were initially identified as lakes but were re-classified by hand.

Certain rare features were combined with other, more common features to create a new variable such as combining power lines and railroads with roads to create a linear features variable. I additionally created amalgamated features that represented similar features. For example, each road category was considered on its own, but was also amalgamated within an “All Roads” category (Table B2).

In order to determine the environmental conditions for each caribou location point, I determined the distance from each point to the nearest representative habitat feature on the map (Connor et al. 2003).

Fidelity across Seasons

Fidelity was compared among seasons using a one-way analysis of variance for each population. It was expected that animals in all populations during the Breeding and

Winter seasons would show weaker fidelity than during Calving, Early Post-Calving, and Late Post-Calving. Comparisons were therefore investigated further using paired one-tailed t-tests for each season-pairing where differences were determined to likely occur. The t-tests were paired by animal to ensure that comparisons were restricted to the same animal within the same year. These comparisons were: Breeding and Winter against Calving, Early Post-Calving, and Late Post-Calving. I adjusted α levels using a Bonferroni correction (Quinn and Keough 2002).

Site Fidelity and Habitat Conditions

Principal Component Analyses (PCA) were conducted using prcomp (R Core Team 2012) run individually for each population and each season in order to identify correlated environmental variables. The variables used in the PCA were the distance measures (averaged during each season) from each caribou location to each landscape feature. For features that were combined or amalgamated (Table B2) the distance to the nearest feature was used. Using the proportion of variance explained (>10%), I selected variables to represent habitat features that commonly clustered together along the same PCA axis and repeated this for the first three principal component axes. These clusters were largely consistent across all three populations for each season and the representative variable functioned as a replacement for that PCA axis while reducing multi-collinearity between variables.

I created models that tested for a link between habitat characteristics (distances to landcover types) and site fidelity (inter-year distances of the locations of female caribou). These models used a penalized quasi-likelihood mixed-effects generalized linear model approach (GLM: with log link function) estimated with the MASS package (Venables and Ripley 2002). To account for pseudo-replication created through the use of multiple years of data from the same animal, I included animal as a random effect in all models (Gillies et al. 2006; Moreau et al. 2012).

I selected models based on a method of reverse stepwise selection investigating parameter estimates, using parameters estimates and standard error to assess model fit, removing those variables that contributed the least to each model and assessing their influence on the other variables and their fit (Nocera et al. 2008). If the removal of the habitat variable had a positive influence on the remaining parameters (their coefficients increase) or if the removal did not influence the other model parameters in either direction, it was deemed uninfluential or surplus and removed. The influence of the random effects was determined by comparing the standard deviation of the intercept-only model and the residuals of the full nested random effect model. The random effects were necessary to account for pseudoreplication and were retained in all models (Nocera and Ratcliffe 2010).

Model fit was determined using the method of calculating R^2 values from Nakagawa and Schielzeth (2013). This methods allows for the development of two R^2 values: the marginal R^2 which represents the amount of variation that can be explained by the model without the influence of the random effects, and the conditional R^2 which represents the predictive ability of the entire model including random effects. This method was specifically developed for use in generalized linear mixed models (Nakagawa and Schielzeth 2013).

Directional Changes in Habitat Use

To test for directional changes in habitat use, I used a longitudinal repeated measures multivariate analysis of variance (MANOVA) using Statistica version 10 (Statsoft, Tulsa, OK, USA) to test environmental habitat use in each population. Using the same habitat distances derived from the PCA, acting as indicators of environmental conditions and as surrogates for the principal component scores, I conducted a MANOVA on the first two years of complete data for each animal to determine if there was a tendency to move towards or away from areas with specific environmental compositions. Distances were both heteroscedastic and non-normal and were log-transformed to meet the assumptions of the MANOVA (Quinn and Keough 2002). Year and population were included as independent variables. Only the first and second year location points, and their associated habitat conditions, were used to avoid pseudo-replication and to maintain sample balance. Any relationships between variables (both the response and predictors) were investigated using paired t-tests, adjusting the α-value to 0.017 (i.e. $\alpha = 0.05/3$ tests) by applying a Bonferroni correction (Quinn and Keough 2002).

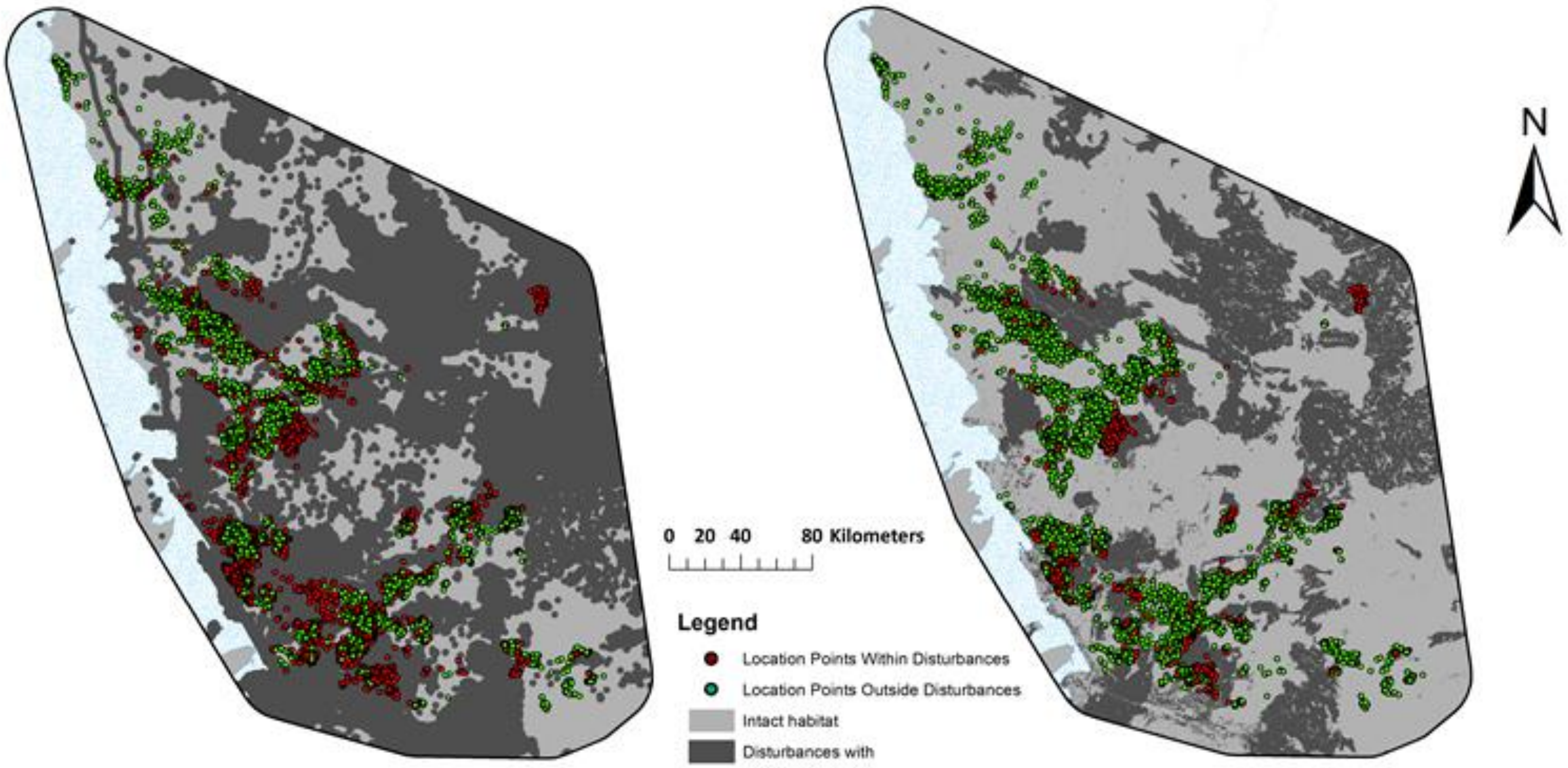

Figure 7. Locations of female caribou in all disturbances (red points) with a 1 km buffer (left) and 10 m buffer (right) and in intact habitat (green points) on the Atikaki-Berens range, Manitoba, 2000-2010.

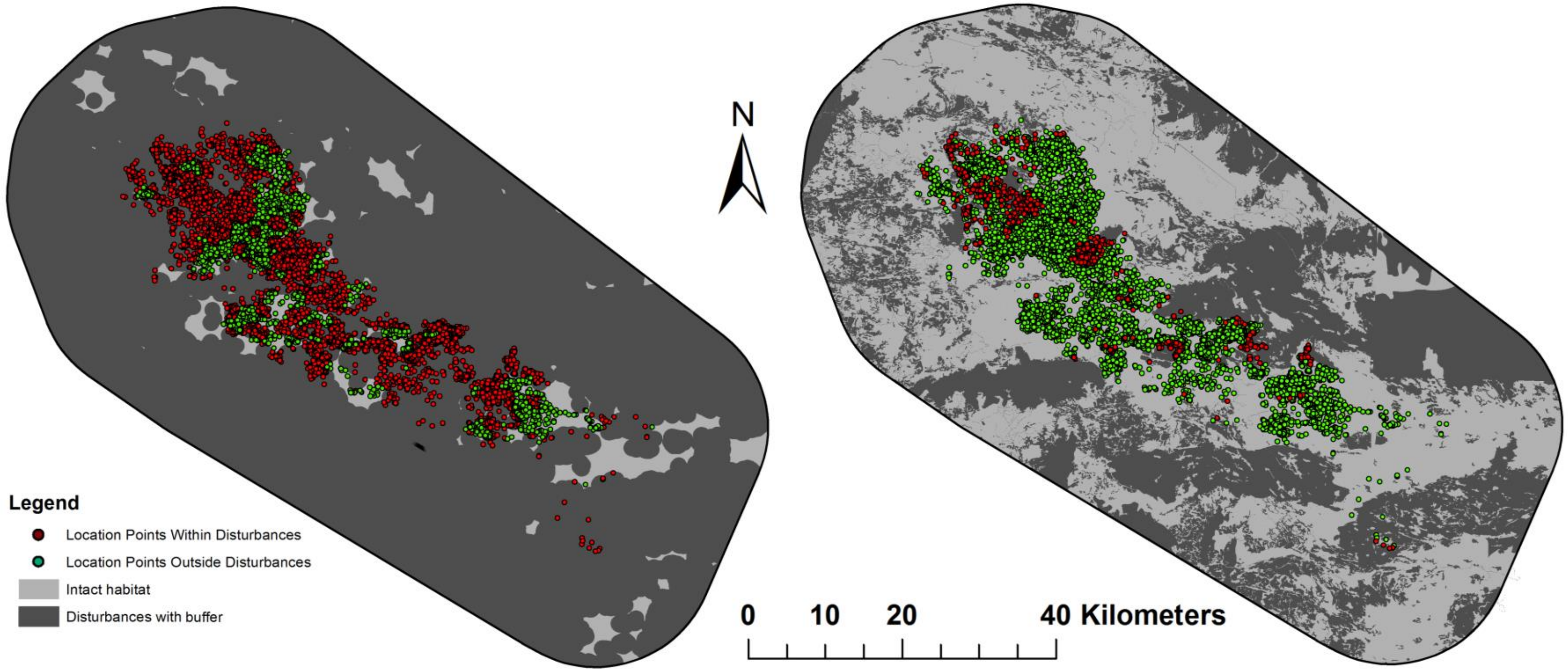

Figure 8. Locations of female caribou in all disturbances (red points) with a 1 km buffer (left) and 10 m buffer (right) and in intact habitat (green points) on the Owl-Flintstone range, Manitoba, 1995-2010.

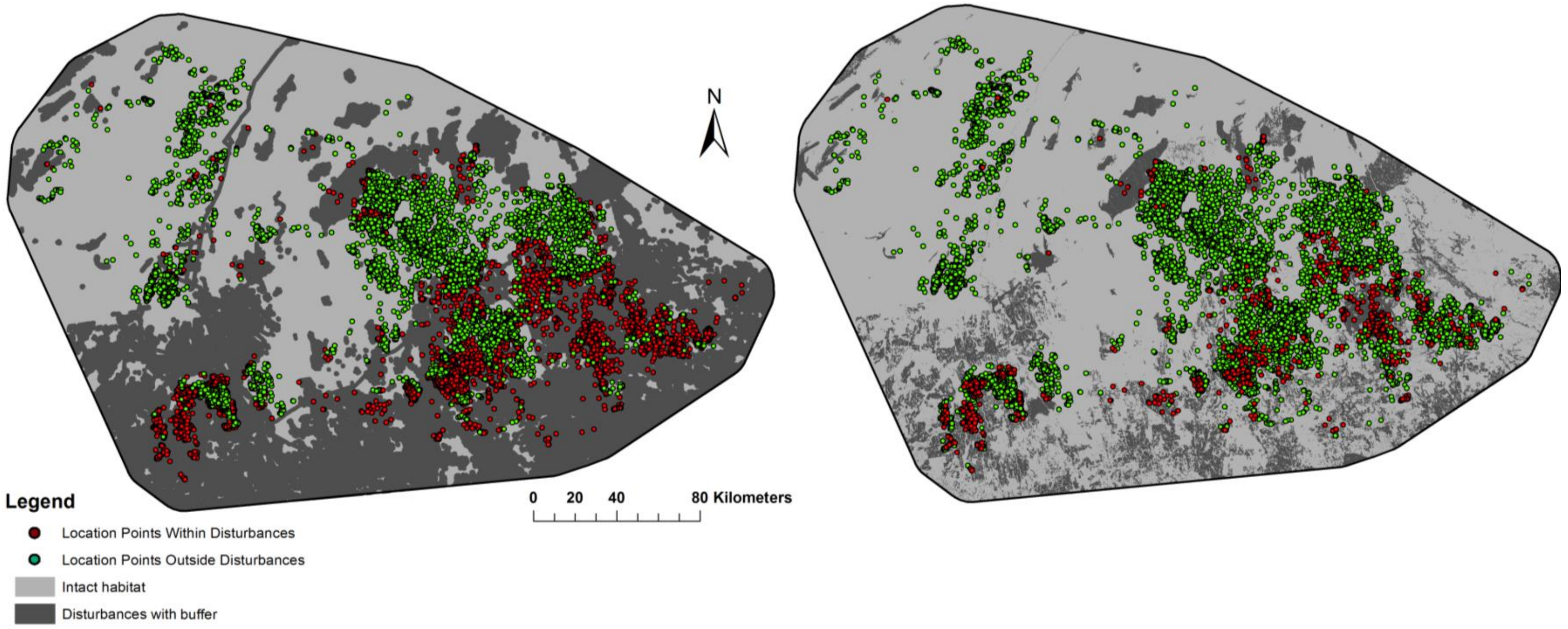

Figure 9. Locations of female caribou in all disturbances (red points) with a 1 km buffer (left) and a 10 m buffer (right) and in intact habitat (green points) on the Kesagami range, Ontario, 2010-2013

Results

Seasonal Site Fidelity

Biological year profiles for each population (Figures 10 to 12) show obvious maxima in fidelity strength (minima in distance) during the Early and Late Post-Calving periods as anticipated (Atikaki-Berens: 8.6 km and 4.9 km, respectively; Kesagami: 14.7 km and 9.0 km; Owl-Flintstone: 9.7 km and 4.1 km [Table 6]). In all three populations, the average Calving site fidelity was markedly weaker (Atikaki-Berens: 9.6 km, Kesagami: 17.8 km, Owl-Flintstone: 11.0 km) than during the Late Post-Calving period and either stronger than, (Kesagami: 37.8 km) or approximately equivalent (Atikaki-Berens: 15.8 km, Owl-Flintstone: 10.3 km) to that of the Winter season. Breeding site fidelity in Kesagami was remarkably strong (11.9 km) compared to the remainder of the year, equivalent to the post-calving periods. In both the Kesagami and Atikaki-Berens populations there was a marked spike in distances (weaker fidelity) during a period of approximately 1-2 months prior to Calving. This spike was not during a period included in my habitat analysis but bears highlighting due to its consistency and strength.

Minimum distances for any single animal were similar across Calving, and both post-calving periods (Table 6); however, the Calving and Early Post-Calving seasons also contained the two weakest fidelity values at over 104 km and 105 km respectively (both from the same animal between 2010 and 2011). Across the entire biological year period (until day 300 after breeding) the minimum distance for any one animal across all three populations occurred in the Kesagami population during mid-May at only 126 metres (Table 6) with the maximum distance, over 136 km, occurring in the Kesagami population during late April.

The one-way ANOVAs indicated strong differences in fidelity among the five seasons in all three populations: Owl-Flintstone ($F_{4,96} = 3.0$, $p = 0.02$), Atikaki-Berens ($F_{4,60} = 4.1$, $p = 0.005$), and Kesagami ($F_{4,142} = 9.1$, $p < 0.001$) (Figures 13 to 15). Additionally, in all three populations Late Post-Calving (Atikaki-Berens $p <0.001$, Owl-Flintstone $p < 0.001$, and Kesagami $p < 0.001$ [Table 7]) had significantly stronger fidelity than during the Winter period. During the Early Post-Calving period only the Kesagami population ($p < 0.001$) had significantly stronger fidelity strength than during the Winter while during the Calving period, fidelity was stronger than during the Winter period in both Kesagami ($p < 0.001$) and Owl-Flintstone ($p = 0.013$). Breeding period fidelity strength was weaker than only a single period in one population, Late Post-Calving in the Owl-Flintstone population ($p = 0.006$). In all other periods and populations Breeding site fidelity was not significantly weaker.

Table 6. Summary statistics for site fidelity expressed as the distance (km) between consecutive-year locations of individuals for female caribou in three populations, Manitoba and Ontario, 1995-2013

Population and Season		Average Distance	Maximum Distance	Minimum Distance	Median Distance	Standard Error	Number of Animals	Number of Animal-Years
Atikaki-Berens	Breeding	17.8	27.6	5.9	18.7	4.5	2	4
	Calving	9.6	33.5	0.5	7.6	2.0	13	19
	Early Post-Calving	8.6	33.0	1.9	7.1	2.0	13	15
	Late Post-Calving	4.9	19.6	0.5	4.3	1.2	13	15
	Winter	15.8	47.9	5.2	12.2	3.6	9	12
	Full Year	12.1	56.1	0.4	9.7	0.2	22	53
Kesagami	Breeding	11.9	33.3	1.5	10.5	1.6	18	29
	Calving	17.8	104.8	0.1	9.6	4.4	18	29
	Early Post-Calving	14.7	105.1	0.5	7.2	4.0	18	31
	Late Post-Calving	9.0	46.0	0.7	4.3	2.1	18	30
	Winter	37.8	98.4	4.1	34.3	5.0	17	28
	Full Year	26	136.4	0.1	17.6	0.3	20	68
Owl-Flintstone	Breeding	7.8	15.1	1.9	5.5	1.4	5	11
	Calving	11.0	51.8	0.3	4.3	3.0	14	25
	Early Post-Calving	9.7	51.7	1.1	5.6	2.9	12	23
	Late Post-Calving	4.1	9.9	1.4	3.3	0.6	11	22
	Winter	10.3	21.3	3.2	11.3	1.0	12	20
	Full Year	9.9	55.3	0.4	7.8	0.1	20	54

Table 7. Results of the one-tailed t-tests investigating relationships of site fidelity across seasons with expected differences. Significance adjusted to a Bonferroni corrected α value of 0.017.

Population	Season of Weak Fidelity	Season of Strong Fidelity	Mean difference* (log-transformed)	Degrees of Freedom	t value	p value
Kesagami	Winter	Calving	1.33	24	3.80	**<0.001**
		Early Post-Calving	1.38	26	5.30	**<0.001**
		Late Post-Calving	1.62		6.96	**<0.001**
	Breeding	Calving	-0.17	25	-0.56	0.290
		Early Post-Calving	-0.23	27	-0.81	0.490
		Late Post-Calving	-0.52		-1.87	0. 036
Owl-Flintstone	Winter	Calving	0.79	18	2.41	**0.013**
		Early Post-Calving	0.60		2.02	0.029
		Late Post-Calving	1.02		4.95	**<0.001**
	Breeding	Calving	-0.22	10	-0.36	0.382
		Early Post-Calving	0.03		0.07	0.526
		Late Post-Calving	-0.70		-3.04	**0.006**
Atikaki-Berens	Winter	Calving	0.54	5	1.20	0.145
		Early Post-Calving	0.91		2.59	0.024
		Late Post-Calving	1.61		3.40	**<0.001**
	Breeding	Calving	-0.89	3	-1.03	0.190
		Early Post-Calving	-1.33		-1.97	0.072
		Late Post-Calving	-1.18		-1.87	0.079

* Difference between fidelity in the "Season of Strong Fidelity" column minus that in the "Season of Weak Fidelity" column.

Site Fidelity in Relation to Habitat Conditions

Variable selection achieved through the Principal Component Analysis consistently identified road and naturally generated young forest across all seasons and populations (Figures B1 to B15) as representatives of PC1, PC2, or PC3. In each season I selected a third variable representing the natural landscape: mature upland conifer for the Winter, Calving, and Late Post-Calving periods, and water for the Breeding and Early Post-Calving periods.

The random effect of individual animal was influential in all cases except during Winter (Residual SD = 0.6, Intercept SD = $3.4e^{-05}$) which was also the only season where the full model was selected. During Winter, fidelity strongly associated with population (Atikaki-Berens|Kesagami: $t_{35} = 2.4$; $p = 0.02$; Atikaki-Berens|Owl-Flintstone: $t_{35} = -1.39$; $p = 0.2$ [Table 8]) and was weakly positively associated (stronger site fidelity in close proximity to the habitat feature) with proximity to roads ($t_{19} = 1.8$; $p = 0.09$), naturally generated young forests ($t_{19} = 1.2$; $p = 0.3$), and weakly negatively associated (weak site fidelity in close proximity to the habitat feature) with upland conifer ($t_{19} = -0.8$; $p = 0.4$). The predictive ability of the marginal Winter model was the strongest of all seasons ($R^2 = 0.5$) while the conditional Winter model had an equivalent predictive ability ($R^2 = 0.5$; Table 9) as a result of the inconsequential influence of the random effect of animal.

During Calving the final model showed a strong negative association of site fidelity with proximity to naturally generated young forests ($t_{27} = -2.3$; $p = 0.03$; Figure 16) and a weak relationship with population (Atikaki-Berens|Kesagami: $t_{42} = 1.8$; $p = 0.07$, Atikaki-Berens|Owl-Flintstone: $t_{42} = -1.0$; $p = 0.3$). The predictive ability of the fixed effect-only model was weak (marginal $R^2 = 0.1$) while the influence of the random effect of animal was influential (Residual SD = 0.7; Intercept SD = 1.1) and greatly increased the predictive ability of the final model (conditional $R^2 = 0.8$). The Late Post-Calving model also included naturally generated young forests ($t_{23} = 2.8$; $p = 0.009$); however, proximity to naturally generated young forests showed a strong positive relationship with fidelity while proximity to roads showed a weak negative relationship ($t_{23} = -0.3$; $p = 0.8$). The fixed effect-only model had the poorest predictive ability (Marginal $R^2 = 0.09$) of all five seasons however, the inclusion of the random effect of animal was influential (Residual SD = 0.6; Intercept = 0.5) and its inclusion in the final model increased predictive ability to the second best predictive model (Conditional $R^2 = 0.6$). The final model for the Early Post-Calving site fidelity included a weak negative relationship with proximity to naturally generated young forests ($t_{24} = -1.6$; $p = 0.1$) and water ($t_{24} = -1.0$; $p = 0.3$). The effect of population was determined to be weak (Atikaki-Berens|Kesagami: $t_{40} = 0.9$; $p = 0.4$, Atikaki-Berens|Owl: $t_{40} = -1.2$; $p = 0.2$) while the influence of the random effect of animal was strong (Residual SD = 0.7; Intercept SD = 0.8) and increased predictive ability of the model greatly (Marginal $R^2 = 0.1$; Conditional $R^2 = 0.5$). The final model representing Breeding season site fidelity included all three habitat variables but excluded the fixed effect of population. Site fidelity was weakly and positively associated with proximity to roads ($t_{16} = 1.78$; $p = 0.09$) and weakly negatively associated with proximity to naturally generated young forests ($t_{16} = -1.2$; $p = 0.3$) and water ($t_{16} = -1.6$; $p = 0.1$). The influence of the random effect of animal was determined to be influential (Residual SD = 0.4; Intercept SD = 0.6) however, the predictive ability of the final model was weak in comparison to the other four seasons of investigation (Marginal $R^2 = 0.1$; Conditional $R^2 = 0.4$). Plots of predicted versus actual values for each model can be found in the appendix (Figures B16 to B20).

The Importance of Individual Variation

In four of the five seasons of investigations (excluding the Winter season) the inclusion of the random effect of animal greatly increased the predictive ability of each model. Originally included to account for psuedoreplication, the predictive power of animal identity can be interpreted as caribou that show strong site fidelity in the past tend to show strong site fidelity in the future; caribou showing weak fidelity tend to continue to show weak fidelity. This is best shown during Calving (Figure 16), where site fidelity strength in subsequent years appears well predicted by site fidelity in the first two consecutive calving seasons (the four other seasons are presented in the appendix B21 to B24). This also, however, means that animals with weak fidelity tend to show fidelity of approximately equivalent strengths and might indicate a return to the location occupied two or more years ago, spending the intervening years in other areas. This behaviour appears to be present in all three populations with nearly 50% of the animals (n = 20) appearing to return to near the same location they visited two or more years previously (in comparison to the fidelity in the intervening year; Figures B25 to B29). Of the 20 animals with at least 3 years of data during the calving season, three animals appear to exhibit strong site fidelity during all years (Figures B30 and B31), two exhibit no returns (Figure B32), six show a mix of strong and weak fidelity (Figures B33 to B35), and nine appear to return to areas visited two or more years ago (Figures B25 to

B29). It should be noted that site fidelity was not quantified in this study as “strong” or “weak” despite the common use of these terms. Instead it must be considered in relation to the fidelity distances of the other animals, seasons, or different spatial scales. The counts obtained above and presented in figures B25 to B35 took fidelity distances common in the season, population, and individual animal into account but there are no guidelines in the literature (e.g. strong fidelity = 1% of maximal straight line distance of yearly range).

Table 8. Parameters of the final predictive models of site fidelity. Bold p-values represent "strong" relationship between the variable it represents and site fidelity during that season.

Season	**Variable**	**Coefficient Value**	**Standard Error**	**T value**	**p Value**
Calving	Natural Young	-1.22e-04	5.24e-05	-2.33	**0.027**
	Pop: A-K*	9.68e-01	5.29e-01	1.83	0.074
	Pop: A-O*	-5.04e-01	4.90e-01	-1.03	0.310
Early Post-Calving	Natural Young	-8.80e-05	5.39e-05	-1.64	0.114
	Water	-1.82e-04	1.77e-04	-1.03	0.314
	Pop: A-K*	4.09e-01	4.59e-01	0.89	0.378
	Pop: A-O*	-5.25e-01	4.38e-01	-1.20	0.238
Late Post-Calving	Roads	-3.00e-06	1.15e-05	-0.29	0.778
	Natural Young	9.60e-05	3.39e-05	2.84	**0.009**
Winter	Roads	1.60e-05	9.03e-06	1.78	0.091
	Natural Young	3.50e-05	2.97e-05	1.19	0.251
	Upland Conifer	-3.00e-05	3.53e-05	-0.84	0.411
	Pop: A-K*	7.03e-01	2.97e-01	2.37	**0.023**
	Pop: A-O*	-3.36e-01	2.43e-01	-1.39	0.175
Breeding	Roads	3.3e-05	1.87e-05	1.78	0.094
	Natural Young	-4.30e-05	3.55e-05	-1.20	0.248
	Water	-3.45e-04	2.22e-04	-1.56	0.139

* A = Atikaki-Berens, K = Kesagami, O = Owl-Flintstone

Table 9. R^2 values of the final predictive models of site fidelity for each season of investigation.

Season	**Marginal R^2**	**Conditional R^2**
Breeding	0.105	0.384
Winter	0.488	0.488
Calving	0.124	0.771
Early Post-Calving	0.141	0.468
Late Post-Calving	0.092	0.606

Directional Changes in Habitat Use

I applied a MANOVA to test the idea that, in consecutive years, female caribou might select more favourable habitat conditions and show movement away from disturbances towards intact, preferred forest types. In four of the five seasons, only population and the three habitat variables had a significant relationship with each other (Breeding: $F_{4,44} = 10.2$, $p < 0.001$ Calving: $F_{4,84} = 8.2$, $p < 0.001$ Early Post-Calving: $F_{4,78} = 8.4$, $p < 0.001$ Late Post-Calving: $F_{4,80} = 8.5$, $p < 0.001$). This was expected, mainly as a result of the differences in disturbance and landscape composition of the three populations and was of little interest within this study. In these four seasons there appeared to be no predictive ability associated with year in determining distance to any of the habitats (Breeding: $F_{1,22} = 1.8$, $p = 0.2$ Calving: $F_{1,42} = 0.6$, $p = 0.4$ Early Post-Calving: $F_{1,39} = 0.4$, $p = 0.6$ Late Post-Calving: $F_{1,40} = 0.04$, $p = 0.83$). The Winter period alone had a significant three-way interaction among population, year, and distance to habitat types ($F_{4,70} = 7.2$, $p < 0.001$). The Winter season was investigated further using paired t-tests with a Bonferroni-corrected α level (0.017). The t-tests compared the distance from each habitat feature in the first year collared compared with distances in the second year, for all three populations (Table 10). In Owl-Flintstone and Kesagami there appeared a significant difference in the distance to mature upland conifer sites between the first two years of collaring data ($p = 0.009$; $p < 0.001$ respectively) with the relationship appearing to show caribou moving towards upland conifer stands in Kesagami while moving away from them in Owl-Flintstone (Figure 17).

Table 10. Results of paired t-tests investigating the three way interaction of year collared, population, and distance to habitat feature observed within the MANOVA results for the Winter season. The tests compared the distance from all three habitat features in year 1 to year 2.

Population	**Habitat Feature**	**Year**	**Mean (log m)**	**Standard Deviation**	**Degrees of Freedom**	**t score**	**p value**
Kesagami	Roads	1	7.74	1.51	16	-0.92	0.370
		2	7.99	1.51			
	Naturally Young	1	8.94	8.94		1.97	0.067
		2	8.44	0.83			
	Upland Conifer	1	7.67	1.40		5.57	<0.001
		2	5.71	0.40			
Atikaki-Berens	Roads	1	8.82	0.86	8	-0.05	0.961
		2	8.83	0.81			
	Naturally Young	1	6.81	0.75		0.50	0.633
		2	6.61	0.88			
	Upland Conifer	1	4.58	1.67		0.32	0.757
		2	4.39	0.81			
Owl-Flintstone	Roads	1	7.53	0.46	11	-1.68	0.121
		2	7.78	0.40			
	Naturally Young	1	6.69	0.63		0.20	0.845
		2	6.65	0.46			
	Upland Conifer	1	2.94	1.38		-3.19	0.009
		2	3.79	0.91			

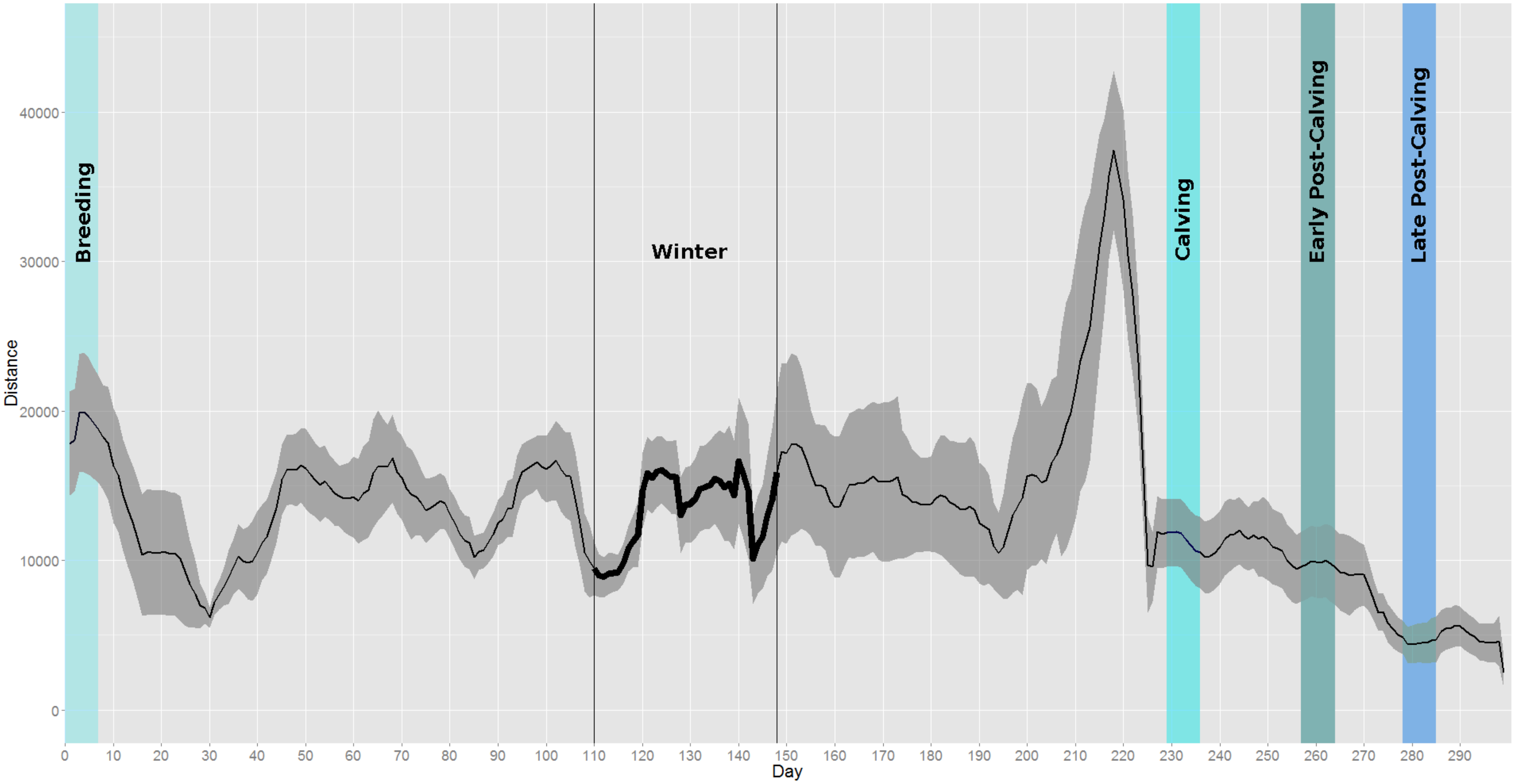

Figure 10. Site fidelity of female caribou from the Atikaki-Berens population expressed as the average distance (± SE) between locations of an individual in consecutive years (m) averaged across every 7 day period. The start of breeding is represented by day 0 and coloured bars represent the length of the labelled seasons. The start and end dates of Winter are denoted by the vertical black lines.

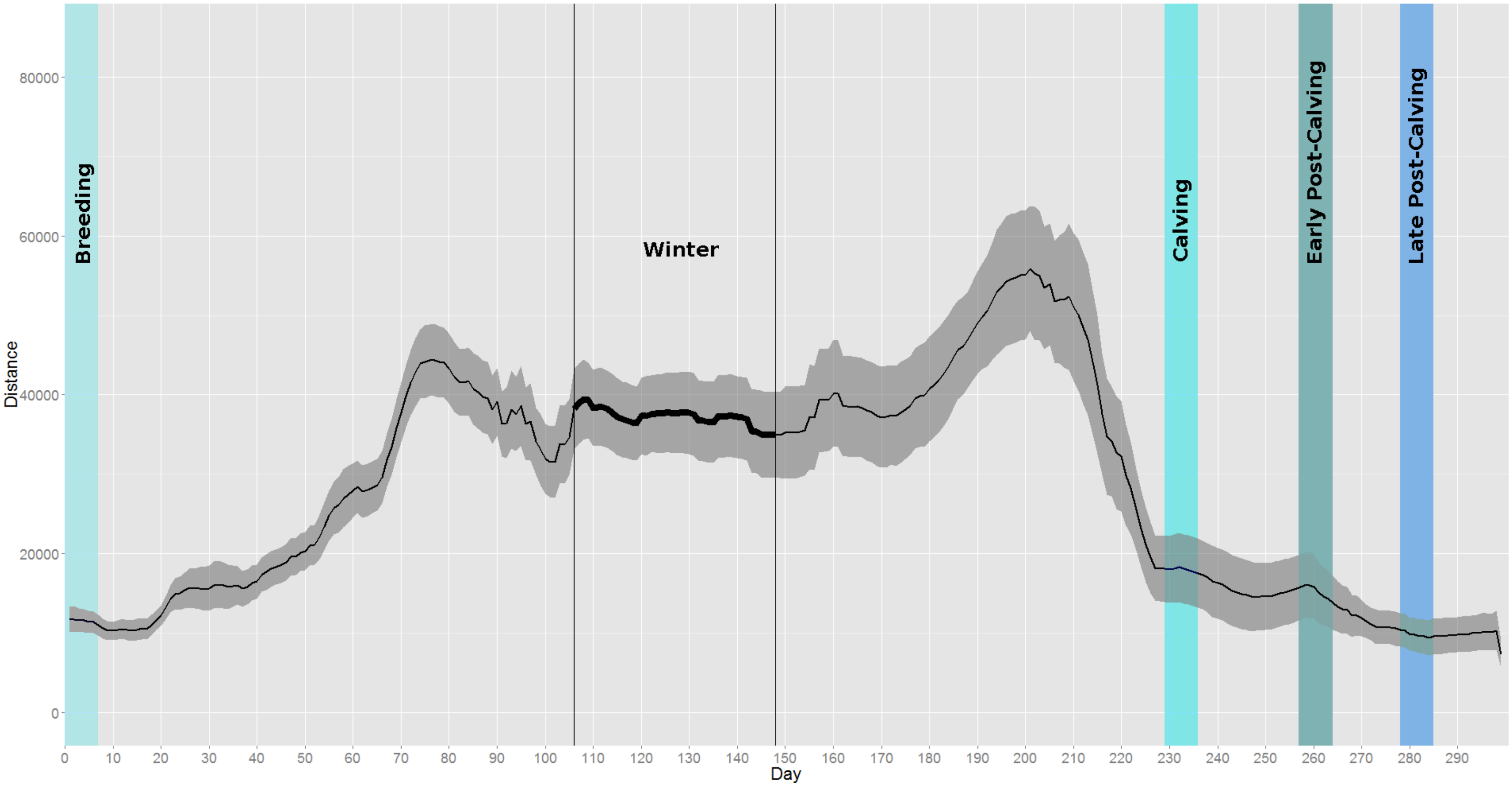

Figure 11. Site fidelity of female caribou from the Kesagami population expressed as the average distance (± SE) between locations of an individual in consecutive years (m) averaged across every 7 day period. The start of breeding is represented by day 0 and coloured bars represent the length of the labelled seasons. The start and end dates of Winter are denoted by the vertical black lines.

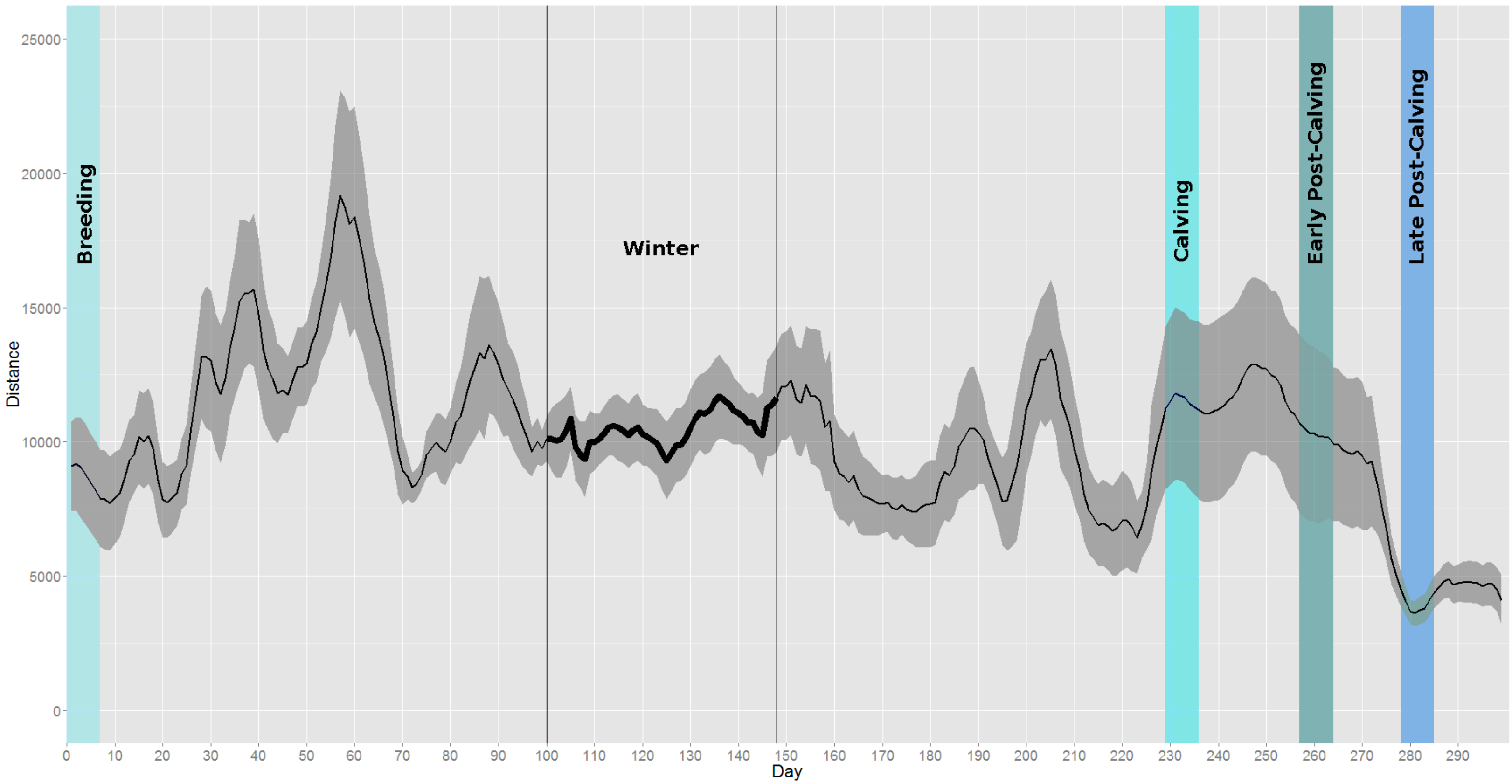

Figure 12. Site fidelity of female caribou from the Owl-Flintstone population expressed as the average distance (± SE) between locations of an individual in consecutive years (m) averaged across every 7 day period. The start of breeding is represented by day 0 and coloured bars represent the length of the labelled seasons. The start and end dates of Winter are denoted by the vertical black lines.

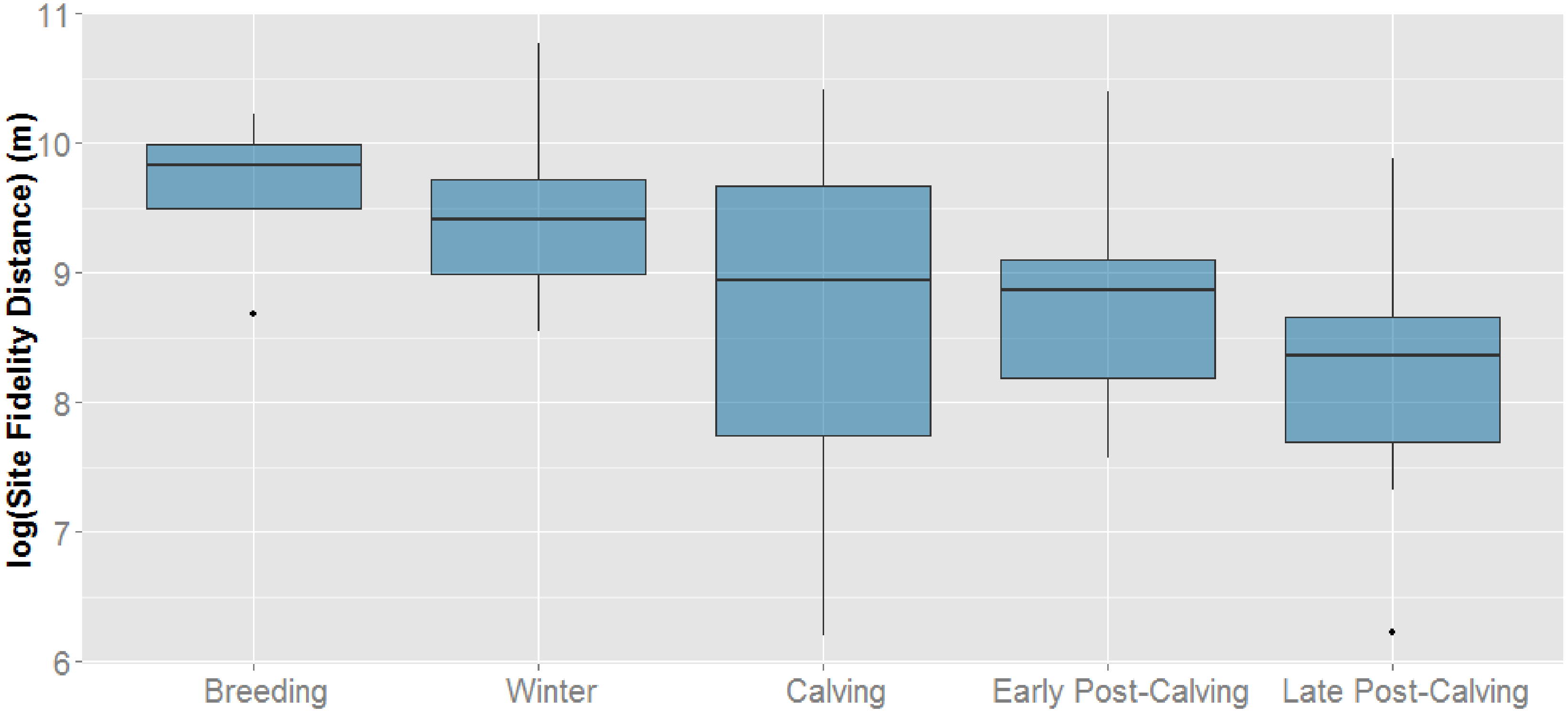

Figure 13. Boxplots of average site fidelity distances (m) during five seasons for the Atikaki-Berens population.

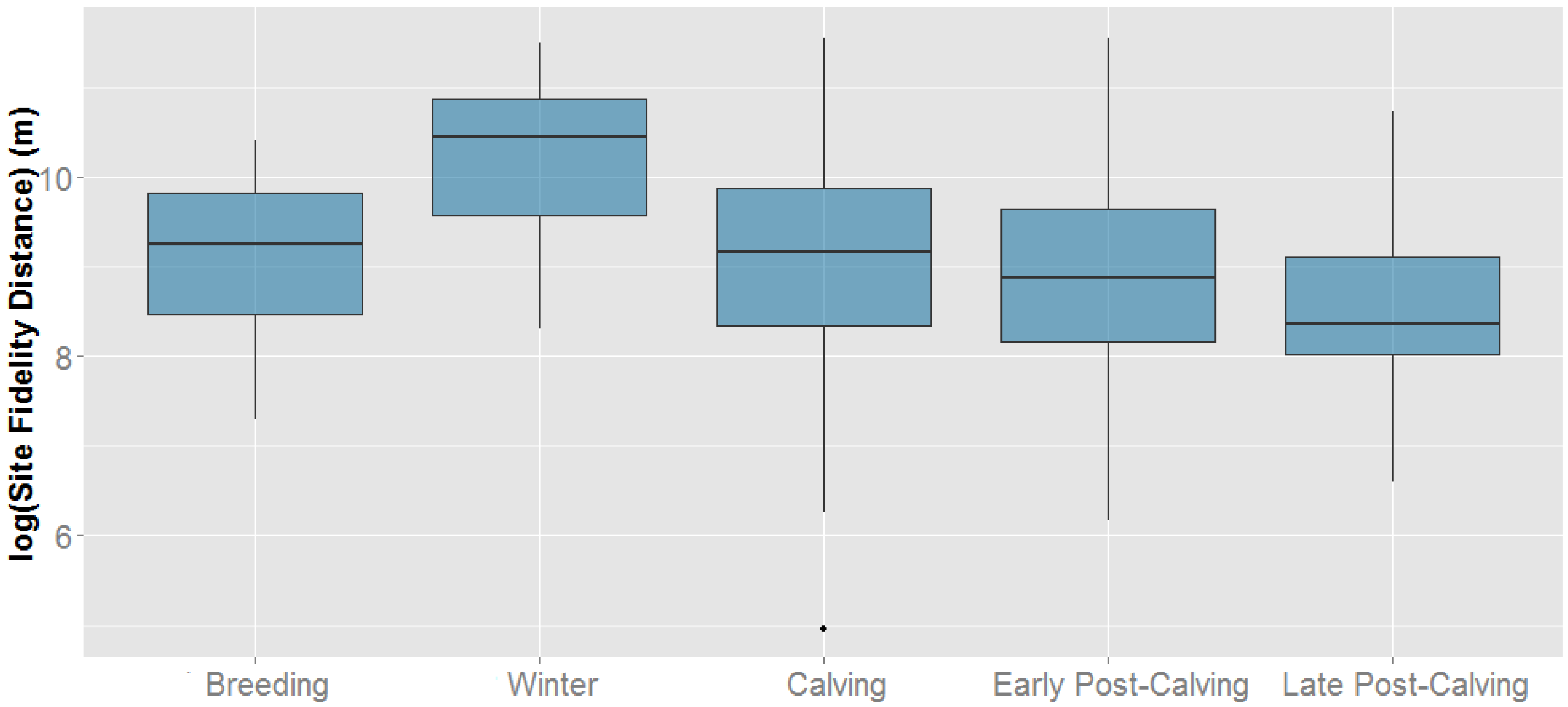

Figure 14. Boxplots of average site fidelity distances (m) during five seasons of investigation for the Kesagami population.

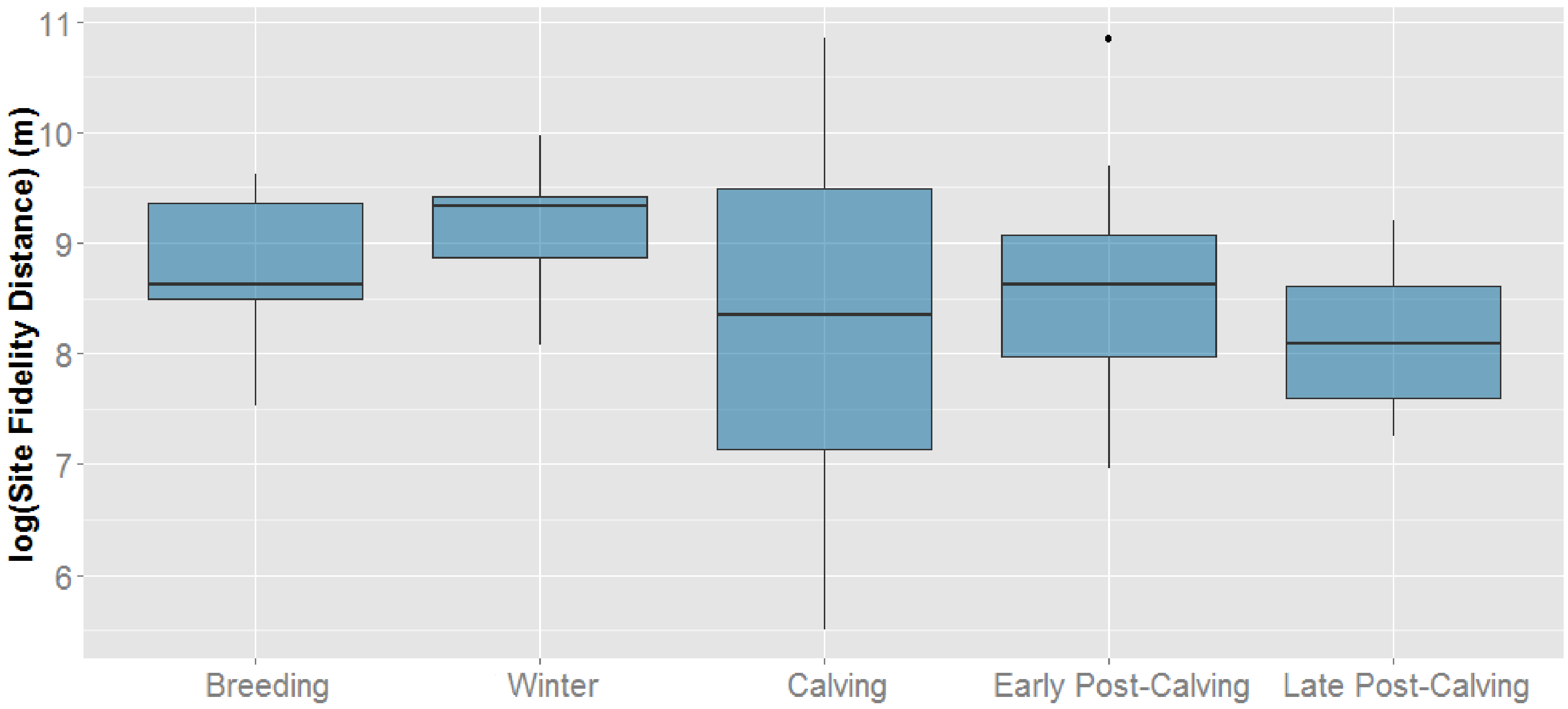

Figure 15. Boxplots of average site fidelity distances (m) during five seasons of investigation for the Owl-Flintstone population.

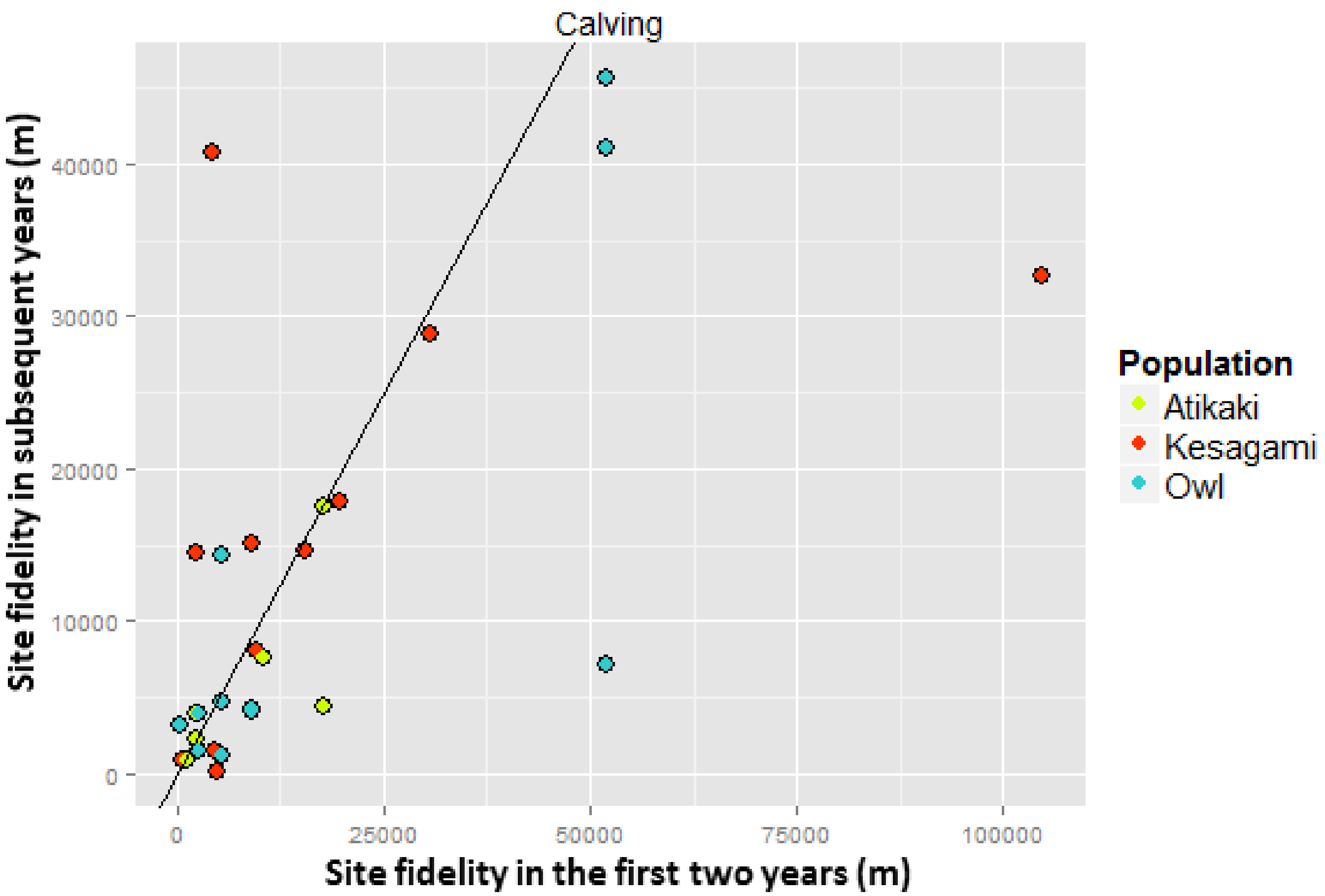

Figure 16. Calving site fidelity (m) of female caribou between the first 2 years compared against all subsequent years in the three populations.

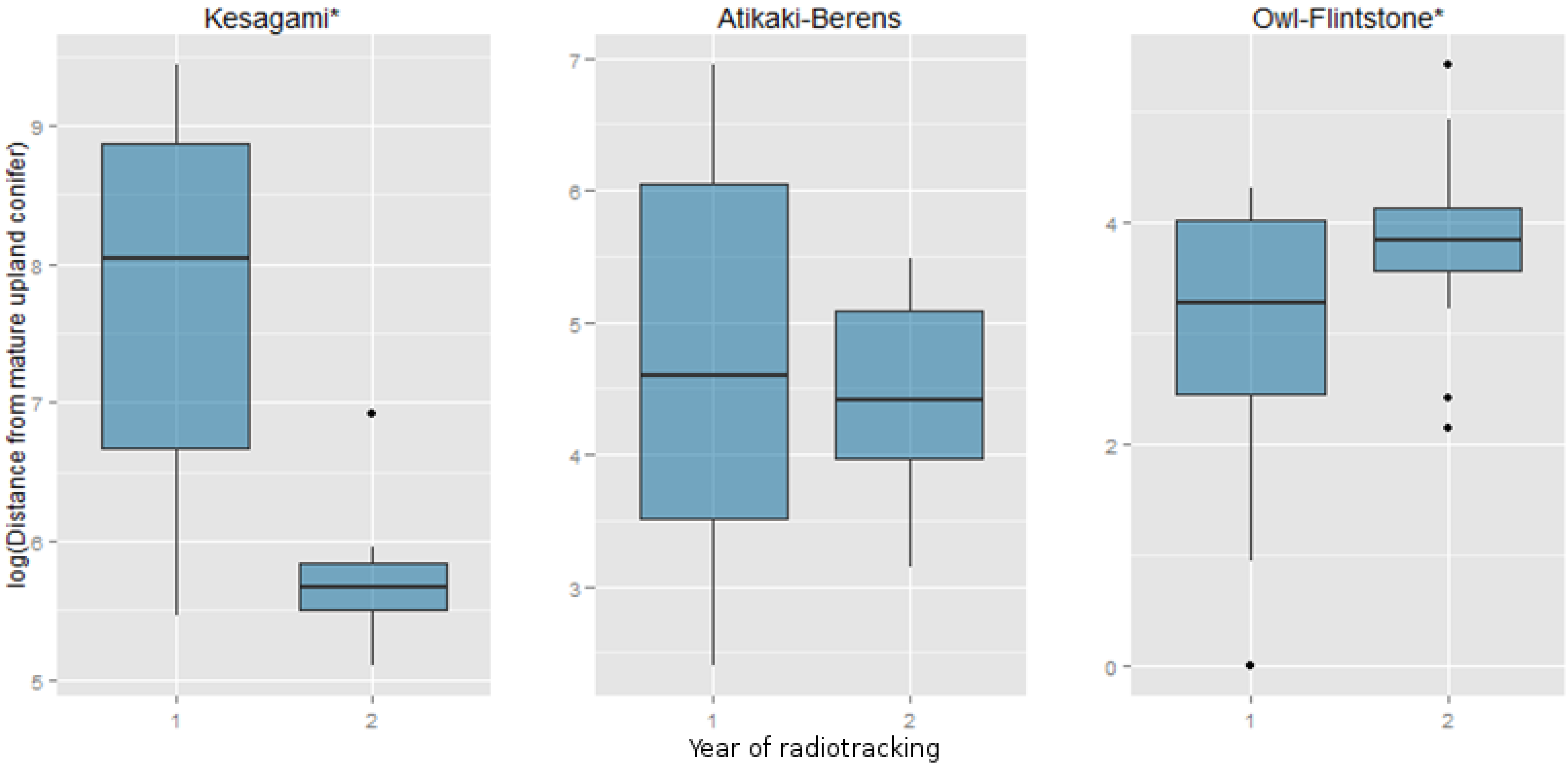

Figure 17. Log distances (m) from mature upland conifer forests in all three populations during the Winter season. Significant differences between years ($p < 0.017$) are represented by *

Discussion

Caribou site fidelity, counter to my original hypothesis, appears poorly predicted by the surrounding landscape. Site fidelity was strongest during the calving and post-calving periods as expected (Schaefer et al. 2000; Ferguson and Elkie 2004; Wittmer et al. 2006; Tracz et al. 2010; Popp et al. 2011). The four seasons with the strongest site fidelity (Calving, Early Post-Calving, Late Post-Calving, and Breeding) showed that the landscape variables had little effect on site fidelity, whereas, the effect of the landscape variables was greatest during the period of weakest site fidelity (Winter) (Table 8). Despite the weak influence of the landscape in four of the five seasons, the final models showed an exceedingly high predictive ability for site fidelity, apparently due to the inclusion of animal identity as a random effect (Figures 18 and B21 to B24). Its inclusion in the final model increased the marginal R^2 values of 0.09-0.1 to conditional R^2 values of 0.4-0.8 (Table 9). While animal identity was originally included to account for pseudo-replication (Koper and Manseau 2012), its strong predictive ability in four of the five seasons implies that site fidelity is far more controlled by the individual identity than the habitat in which they find themselves.

The Strong Influence of Past Fidelity in Predicting Future Fidelity

The apparent strength of animal identity in predicting site fidelity compared to that of the landscape is a surprising result and one that has not yet been identified. Site fidelity, especially during the calving and post-calving periods, is thought to minimize calf and cow mortality through the acquisition of food or the avoidance of predators (Greenwood 1980; Wittmer et al. 2006). Site fidelity theory implies that animals will return to a site that they perceive as high quality (good forage, safety from predators, proximity to conspecifics; Hoover 2003) and will leave once the cost of staying outweighs the cost of leaving (Tracz et al. 2010). The weak relationship between habitat variables and site fidelity for my three study populations is not consistent with this hypothesis. Instead it appears that the caribou that show strong site fidelity are more likely to continue to show strong site fidelity, while those that show weak site fidelity are more likely to continue showing weak site fidelity, regardless of the habitat. I propose an interpretation of caribou site fidelity that requires a slight shift from our current understanding. Caribou site fidelity, as it is currently understood, is the behaviour by which caribou return to the same location as in the previous year (Brown et al. 1986; Schaefer et al. 2000). My study shows that the temporal scale of caribou site fidelity could be expanded, with caribou maintaining fidelity across multiple years. This "multiannual" fidelity, however, is not restricted to consecutive years. Female caribou, instead, might return to the same location two or more years later after spending the intervening year(s) in a different location.

This behaviour appears prevalent across all three populations, occurring in nearly 50% of the animals with sufficient data (Figure B25 to B29). Behaviour such as this might result in weak site fidelity under current interpretations but strong fidelity if considered on a lifetime scale. This would also help explain the strong predictive ability of animal identity as animals shifting their locations would show approximately the same distance between year one to year two, and year two to year three despite the long distance travelled. This is shown well in Figure 16. It should also be noted that some animals initially thought to be displaying multiannual site fidelity, after closer inspection of the full set of locations (instead of the averaged location), instead appear to be exhibiting strong annual site fidelity. The confusion was due to using locations averaged across one week where caribou increased movement after their calving event resulting in an unrepresentative averaged location placement. This unrealistic averaged location may be the result of increased movement following the loss of a calf (DeMars et al. 2013). However, this is likely not a systemic issue as I was able to identify it in only two animals and previous studies have shown a similar technique identified the calving event with high accuracy (DeMars et al. 2013).

Multiannual site fidelity is not an unknown animal behaviour. Species known to exhibit this behaviour include bottlenose dolphins, Pacific white sharks, horseshoe crabs, and sea turtles (Kimmel 2007; Silva et al. 2008; Beekey and Mattei 2009; Nasby-Lucas and Domeier 2012). In these species researchers found individuals exhibiting either annual (returning to the same site in consecutive years) or multiannual site fidelity (returning to the same site with intervening years spent at a second site). In two of the studies researchers suggested that it was a result of either an 18-month gestation for Pacific white sharks, or a non-consecutive spawning event with the horseshoe crabs (Beekey and Mattei 2009; Nasby-Lucas and Domeier 2012). While caribou are known to show non-consecutive breeding events, or reproductive pauses, they occur at a rate of approximately one in four (Cameron 1994; Ropstad 2000), much lower than the non-consecutive, multiannual site fidelity rate I found, of one in two. As such, reproductive pauses are likely not the sole determining factor of this multiannual site fidelity. One potential factor may be predator avoidance, thought to drive habitat use during the spring, summer, and fall (Wittmer et al. 2006) but not during the winter (Schaefer et al.

2000). Showing site fidelity to more than one location during periods of increased predation risk such as calving (Dussault et al. 2012) may decrease predation rates as the consistent use of a single area may increase the potential of detection. It has also been noted that during the calving and post-calving periods woodland caribou space away from other caribou (Bergerud et al. 1990; Bergerud 1996). This behaviour may influence site fidelity as females attempting to return to the previous year's site may find it in use by another caribou, and, to maintain this diffuse pattern of low densities, will travel to a new calving location (Berglund et al. In Press). They might then return to the contested calving location the following year before another caribou has settled. Such an interaction might show the pattern of multiannual site fidelity. However, cursory plots of the onset date of calving compared against Calving site fidelity strength do not appear to support this explanation, as the trend expected (one of decreasing site fidelity with later calving events) is not seen (Figure 19). It should also be noted that site fidelity within this study was measured on the scale of kilometres whereas stand-level competition between caribou would likely happen on a smaller scale.

Technological limitations commonly prevent the collection of caribou locational data for more than 3 years and so we must infer life-long behaviours from a snap-shot of caribou life. Further investigations of similar data with longer term collar deployment would shed light on the tendency of caribou to show consistent fidelity strengths and would enable estimation of the proportion of caribou showing non-consecutive, multiannual, site fidelity. In the three study populations, only two animals had more than four years of locations and both appeared to show consecutive and multiannual year site fidelity during the Calving season (Figure B28). This behaviour is most apparent during the Calving season.

Influence of Disturbances on Fidelity

In the four non-winter seasons, the influence of the landscape was significant only during the Calving and Late Post-Calving periods. During the Calving period site fidelity was negatively influenced by proximity to naturally disturbed sites (weaker site fidelity when closer to naturally disturbed sites [Figure 18]) whereas during the Late Post-Calving period it was positively influenced (Table 8). Caribou select habitat away from naturally disturbed young forests (Dalerum et al. 2007; Fortin et al. 2013) for a number of reasons. Both the early seral stage forests resulting from forest fires and the mixed-aged mixed deciduous/coniferous forests created through disease, insect damage, and blow downs are beneficial moose and white-tailed deer habitat (Frelich and Reich 1995; Festa-Bianchet et al. 2011). Caribou are known to avoid habitat selected by moose as increased ungulate populations associated with disturbances, both natural and anthropogenic, have been shown to support higher wolf populations than areas of mature forest (Bergerud 1974; Bergerud and Elliot 1986; Seip 1992; Latham et al. 2011b). Additionally young forests are the preferred habitat of black bears, an opportunistic predator of woodland caribou calves (Dussault et al. 2005; Bastille-Rousseau et al. 2011) that have been shown to be a significant predatory threat in some populations (Mosnier et al. 2008; Dussault et al. 2012). The relative degradation of Calving site fidelity accounted for in the proximity to naturally disturbed forests (Figure 18) perhaps is due to the increased susceptibility of caribou mothers and calves to stress and predators during this period (Calef et al. 1976; Gauthier and Theberge 1985; Bergerud and Elliot 1986; Murphy and Curatolo 1987; Parker et al. 1990; Whitten et al. 1992; Roffe 1993; Mahoney et al. 1990; Maier et al. 1998; Dussault et al. 2012). However, the relative increase in Late Post-Calving site fidelity as a result of the proximity to naturally disturbed forests may be the result of mother caribou showing increasingly risky behaviour due to the demands of lactation (Gustine et al. 2006) and as the calf becomes more mobile and independent. While proximity does not always equate to use per se, it does increase accessibility and it may show that the mother is selecting young disturbed forests that are known to show remnant patches of suitable habitat and forage (Schaefer and Pruitt 1991) and have a large amount of young growth (Bergeron 2000; Reich et al. 2001) perhaps targeting very young forests (<5 years) which appear to be suboptimal habitats for black bears (Bastille-Rousseau et al. 2011). The benefit of using these habitats may outweigh the risks of predation (Dussault et al. 2012) and could serve to increase calf survival rates during this period (Gustine et al. 2006).

These relationships are counter to those presented by Faille et al. (2010) who found a positive influence of natural disturbances on site fidelity during both the calving and summer, or post-calving, periods. This disparity might be due to my use of a finer temporal scale based on individual caribou movement. Faille et al. (2010) used the same one-month period to represent the calving season for all animals and therefore their calving period may have included behaviours indicative of either pre-calving or post-calving (Pinard et al. 2012; DeMars et al. 2013; Figures 10 and 11). This may have masked the decline in calving site fidelity in the presence of naturally disturbed forests as is evident in my results.

While naturally disturbed forests account for a significant amount of variation attributable to habitat features during the Calving and Late Post-Calving periods, the variation accounted for is still low, and the remaining habitat features, including roads, mature upland forests, and water, show even less predictive ability (Table 8). Faille et al. (2010) found that the density of roads had a strong negative relationship to within-year range fidelity and inter-year fidelity during the winter but a positive relationship during the summer. A similarly confounding but inverse relationship was found in my study, with site fidelity negatively associated with roads during the Late Post-Calving period and positively associated during the Winter and Breeding seasons. This discrepancy in part may have been accounted for by Faille et al. (2010) noting that animals in more pristine environments reacted more strongly to disturbances than those animals in more disturbed populations. Because reactions to disturbance are influenced by the degree of disturbance, with human use patterns and resource extraction dependent on season and weather, and because caribou show behavioural plasticity in their habitat selection (Fortin et al. 2008), it is not surprising that results can differ among populations and studies and this has been noted before (Brown 2005). The general lack of influence of habitat, however, may be indicative of a more serious issue, that site fidelity in caribou may be a maladaptive strategy in the changing face of the North (Fortin et al. 2013). Mortality rates are known to be higher in highly disturbed landscapes (James and Stuart-Smith 2000; Courtois et al. 2007; Dussault et al. 2012), yet these disturbances appear not to elicit a significant response in caribou site fidelity. If caribou continue to select sites regardless of the impact on their survival, as my results suggest, then site fidelity can be truly considered maladaptive.

Several recent studies have shown similar results. Dalerum et al. (2007) found that range fidelity was maintained in areas that recently experienced large scale forest fires, with up to 76% of home ranges being burnt. Tracz et al. (2010) found a similar behaviour in Alberta with caribou maintaining range fidelity in the face of petroleum development, while DeCesare et al (2014) showed that habitat selection in woodland caribou appeared to be flawed, with caribou selecting habitats with higher risk of predation than anticipated. Both St-Laurent and Dussault (2012) and Fortin et al. (2013) suggested that fidelity may be a maladaptive trait in disturbed landscapes. By maintaining site fidelity regardless of the disturbances and habitat around them, caribou may be placing themselves in suboptimal habitat that may increase their and their calves' mortality rates (James 1999; Mosnier et al. 2008; Latham et al. 2011a; Whittington et al. 2011; Dussault et al. 2012) creating what is called an ecological trap to which they continue to return (St-Laurent and Dussault 2012). An ecologiocal trap occurrs when an animal chooses an area that it perceives to be of higher quality yet - often due to human alterations - is actually poorer while avoiding less attractive, higher quality habitats (Battin 2004). The continued use of these high risk locations may indicate that the range retractions currently occurring are the result of historic disturbances, and indicative of mortality and local extirpation rather than movement away from the disturbances.

While my results show that the habitat has little influence on site fidelity in these three populations, the predictive ability of the landscape models is equivalent to that of many published studies in ecology (Møller and Jennions 2002). While these results appear inconsistent with similar studies (Faille et al. 2010) they appear supportive of the hypothesis that maintenance of caribou site fidelity may be maladaptive in the face of growing challenges to survival (Dalerum et al. 2007; Tracz et al. 2010; Fortin et al. 2013). Interestingly the population with the weakest site fidelity in all seasons except Breeding (Table 6, Figures 10 to 12), the Kesagami population, was also the only population I studied that was listed as declining (Environment Canada 2012). While site fidelity of individuals appears little influenced by the quality of the habitat it may still be related to the status of the population.

Weak Influence of Intact Habitat on Fidelity

If site fidelity were maintained, despite being a potential maladaptive strategy, one would anticipate that the original interactions with the landscape, that of the natural intact forests and water, would remain the same. The influence of the intact natural landscape, however, appears to be just as weak as disturbances with the final models for Calving and Late Post-Calving not containing either mature upland conifer or water. The causes of this I believe are twofold: biological and methodological. Firstly, during the execution of the study, due to combining two different map types with low accuracies at fine classification scales across two markedly different landscapes, I was forced to amalgamate the habitat classes into three broad types (Chapter 2). As a result, classes such as mature lowland conifer included nearly all black spruce dominated habitats, which are plentiful and could include dense black spruce forests and open muskegs in the same category despite the fact that caribou interact differently with these habitats and use them differently during different times of the year (Ferguson and Elkie 2004; Courbin et al. 2009). This poor discriminatory ability for the landscape variables likely reduced my ability to detect the effect of forest cover. Additionally, as distances were measured from the nearest feature, no matter its size, there exists potential that the distances from each caribou location point may be measured from patches of forest or water that, while present, are small enough to likely be of no consequence to caribou. While these issues exist for the apparent lack of influence of the intact natural landscape, it doesn't hold true for disturbances.

Predictive Ability during the Winter

The Winter season proved an interesting exception. Winter, the period considered to be when caribou show little site fidelity (Schaefer et al. 2000), had the weakest site fidelity of all five seasons. It was also the only season that showed a strong relationship between the variation in site fidelity and landscape characteristics, while the influence of animal identity, strongly predictive in the other four seasons, was essentially non-existent. This period was the only season where the global model emerged as the final model. It was also the only period that showed a strong influence of population. Also, counter to expectations, it appeared that fidelity increased in closer proximity to disturbances, both natural and anthropogenic (Table 8).

Caribou are known to shift habitat according to weather conditions during the winter, selecting different habitats during heavy snow years than those that they select during winters with less snow (Brown 2005; Wittmer et al. 2006; Popp et al. 2011) and is indicative of a shift in the limiting factor from predator avoidance for much of the year to food acquisition during the winter (Wittmer et al. 2006).

Winter was the only period showing an individual's population membership as highly influential towards site fidelity strength. In addition to the vast differences in the three range sizes (Figure 1), this may be attributed to the differences in the landscapes of the three populations. The Owl-Flintstone and Atikaki-Berens populations are both composed mainly of rocky uplands interspersed with black spruce dominated bogs and lakes (Rowe 1972) whereas the Kesagami range features little variation beyond bogs, muskegs and fens, with upland forests and lakes exceedingly rare in comparison (Rowe 1972). This is in addition to the large differences in snowfall between the two provinces, with the Kesagami range receiving nearly double the amount of snowfall (Environment Canada 2013). Weather conditions and their influence on food acquisition drives habitat use especially during the Winter (Brown 2005; Wittmer et al. 2006). The increased energetic cost of digging for food in deep snow requires caribou to shift to upland sites where snow conditions are more favourable (Darby and Pruitt 1984; Darby and Duquette 1986). These upland sites are exceedingly rare in the Kesagami range and during years of increased snowfall their relative value may increase, forcing the caribou to travel further to find adequate habitat and survive the harsh winters. It has also been shown that caribou will normally increase their movement during winters with deep snow (Wittmer et al. 2006). Indeed, membership in the Kesagami population indicated much weaker site fidelity measurements during the Winter than in the Atikaki-Berens population (Table 8).

The remaining variables in the final model showed a weaker influence than population assignment while distance to roads and naturally generated young forests presented a counterintuitive relationship with site fidelity strength. The closer a caribou was to a road or naturally disturbed young forest, the stronger its fidelity. Multiple studies have shown that disturbances are detrimental to caribou survival whether it is through human activity (Calef et al. 1976; Bradshaw et al. 1997; Dyer et al. 2001; Brown 2005), predators (Huggard 1993; James 1999; Latham et al. 2011a), or food availability (Gates et al. 1986; Boertje 1990; Sveinbjörsson 1990; Proceviat et al. 2003; Dunford et al. 2006; McMullin et al. 2013). Despite this, this relationship is likely less indicative of caribou preferentially choosing habitat near disturbances and instead is likely the result of caribou in highly disturbed habitats showing smaller annual ranges (Faille et al. 2010; Beauchesne et al. 2014) as they are restricted in their habitat selection by the unsuitable habitat caused by disturbances. In the Owl-Flintstone population all location points from the Winter season fell in the north-eastern half of the range where disturbance was the lowest (Figure 20). By restricting their winter ranges to a less disturbed area, they showed stronger site fidelity than the other two populations in less disturbed habitat. In a period marked by a lack of site fidelity, those animals in more pristine environments (Atikaki-Berens and much of the Kesagami range) are free to relocate to suit the needs of that specific winter, while those in disturbed habitats such as the Owl-Flintstone may be forced to stay in a smaller range and therefore exhibit strong site fidelity. Beauchesne et al (2014) suggested that this behaviour could also be considered maladaptive; as caribou reduce the size of their range they would effectively be making themselves easier for predators to find. The reduction in site fidelity and the strong influence of the weather on habitat use during the winter may also reduce the influence of individual animal identity on site fidelity. If movement during the winter is dictated by a random and external force such as the weather, movements themselves may appear more random and less governed by individual differences. It may also be a result of the fact that caribou tend to aggregate into groups during the winter (Stuart-Smith et al. 1997; Brown 2005; Wittmer et al.

2006). These aggregations appear to move in tandem, changing their yearly wintering grounds together (Figure 21). Decisions based upon group needs or the tendency to aggregate may override individual tendencies and serve to reduce the influence of the individual.

Directional Changes in Habitat Use

Directional changes in the proximity to certain habitats appears to be inconsistent in relation to landscape conditions. In four of the five seasons there appeared to be no influence of the years since collared on the distances to each of the habitat variables under investigation. Only during the Winter season did there appear to be an effect, with caribou from the Kesagami population more likely to move towards mature upland conifer forests while those in the Owl-Flintstone range moved away (Table 10 and Figure 17). Caribou tend to shift from open and treed muskegs in light snow to mature upland forest in heavy snow (Darby and Pruitt 1984; Darby and Duquette 1986; Rettie and Messier 2000; Brown 2005). Upland conifer forests accumulate less snow allowing increased access to lichens during the winter (Darby and Pruitt 1984; Darby and Duquette 1986; Manitoba Conservation 2011). With greater snowfall compared to the two Manitoba populations (Environment Canada 2013), the relative value of upland sites in the Kesagami range may be increased, resulting in caribou showing more movement towards these rare upland sites. This may also account for the movement away from upland conifer sites in the Owl-Flintstone population as they face winters of reduced snowfall and instead select for lowland muskeg sites (Darby and Pruitt 1984; Darby and Duquette 1986; Rettie and Messier 2000; Brown 2005; Wittmer et al. 2006).

Additionally, all Kesagami animal locations fell within a three year period whereas the Manitoba populations contained points spread across more than a decade. Though speculative, there exists the potential for a particularly mild winter in the Kesagami range, followed by a harsh winter, to create this behaviour.

In all five seasons there was a significant interaction between population and the habitat variable in governing habitat use. This result is consistent with the marked structural differences between the two provinces, with Ontario's landscape being dominated by a flat and boggy terrain, upland and deciduous forests and large lakes are far rarer than in Manitoba as were large lakes (Rowe 1972). Disturbances also differed significantly between provinces. The Manitoba populations experience far more forest fires than those in Ontario and the forest fires that do occur in Ontario are much smaller, likely as a result of the wetter soils. Additionally human disturbances varied between populations, with Owl-Flintstone representing a highly disturbed population, Atikaki-Berens a relatively undisturbed population, while the Kesagami population experiences a moderate amount. A significant difference and interaction between these was expected and of little biological interest here.

Conclusions

Female woodland caribou site fidelity appears strongest during the Calving and Post-Calving seasons. This fidelity appears weakly influenced by the surrounding landscape with only the proximity to naturally generated young forests accounting for a moderate amount of variation in site fidelity during the Calving and Late Post-Calving periods. Population membership also appears to be of moderate consequence to site fidelity during the Winter with the Kesagami population showing much greater distances between years than the two Manitoba populations. This pattern of maintaining or degrading site fidelity regardless of their proximity to disturbed sites may indicate that caribou have fallen into an ecological trap. The results of this study also shows that site fidelity may be more complicated than earlier thought, with animals exhibiting what I termed “multiannual” site fidelity, showing site fidelity across multiple years, with intervening years spent in a secondary location.

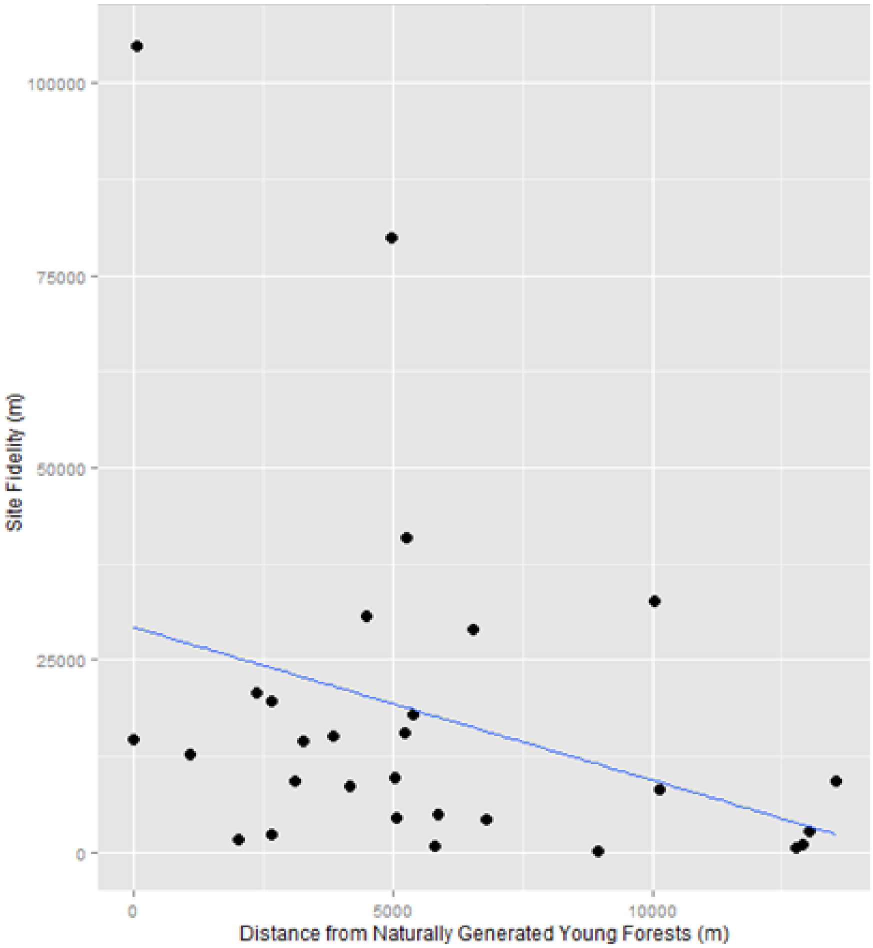

Figure 18. Calving site fidelity in the Kesagami population plotted against the distance to naturally generated young forests during year 1. The blue line represents the strong relationship within the poor predictive ability of the marginal R^2 model.

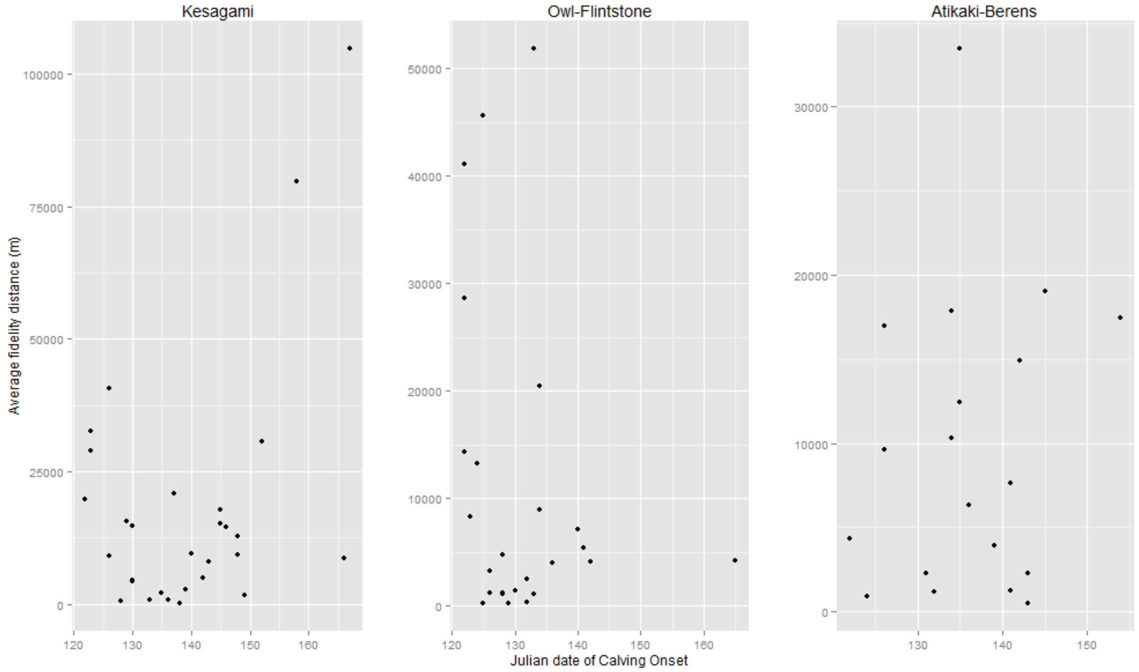

Figure 19. Calving site fidelity (m) compared against the onset of calving in the first year.

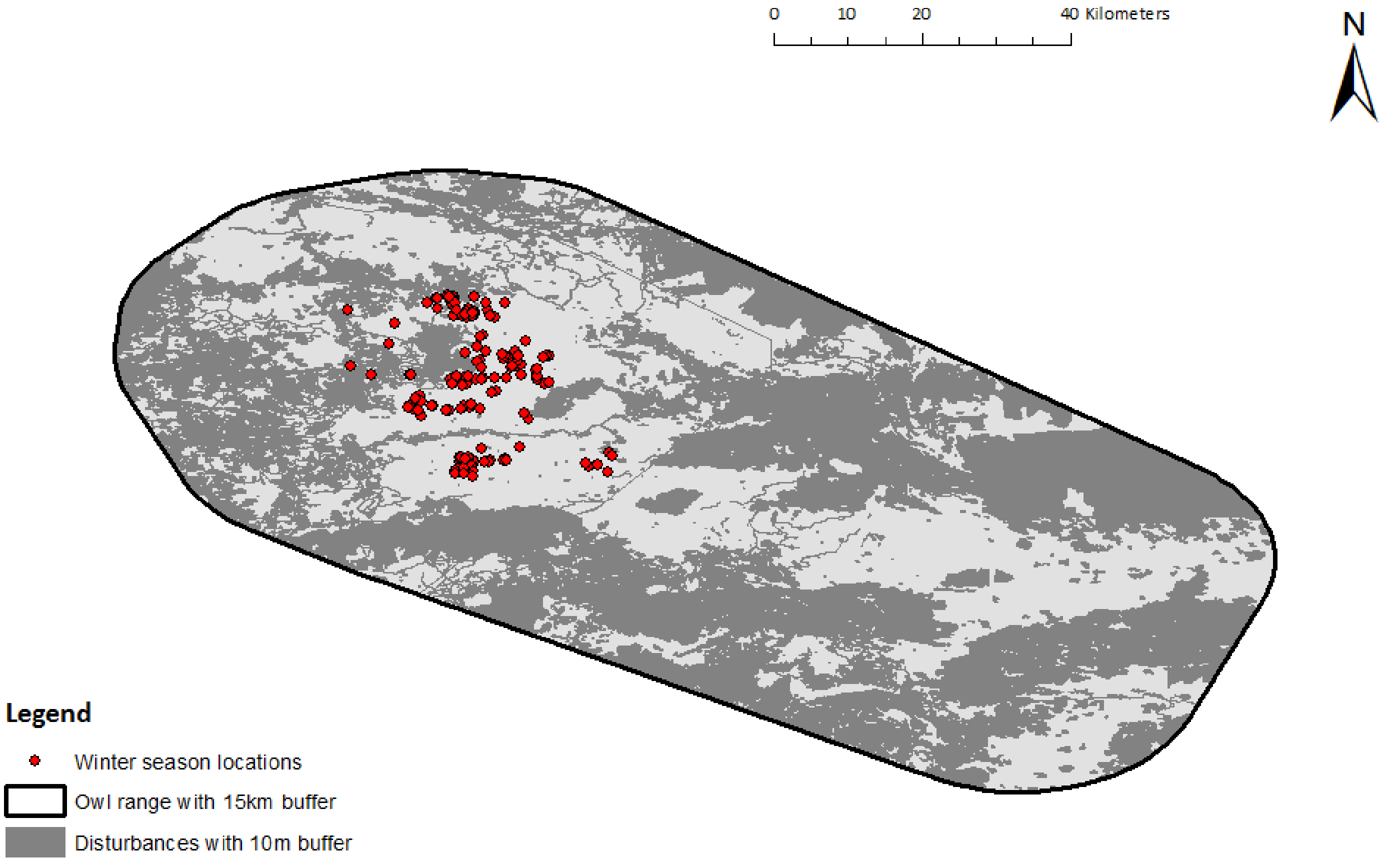

Figure 20. Winter season locations for the Owl-Flintstone population showing their consistent behaviour of remaining in the less disturbed northwest portion of their range.

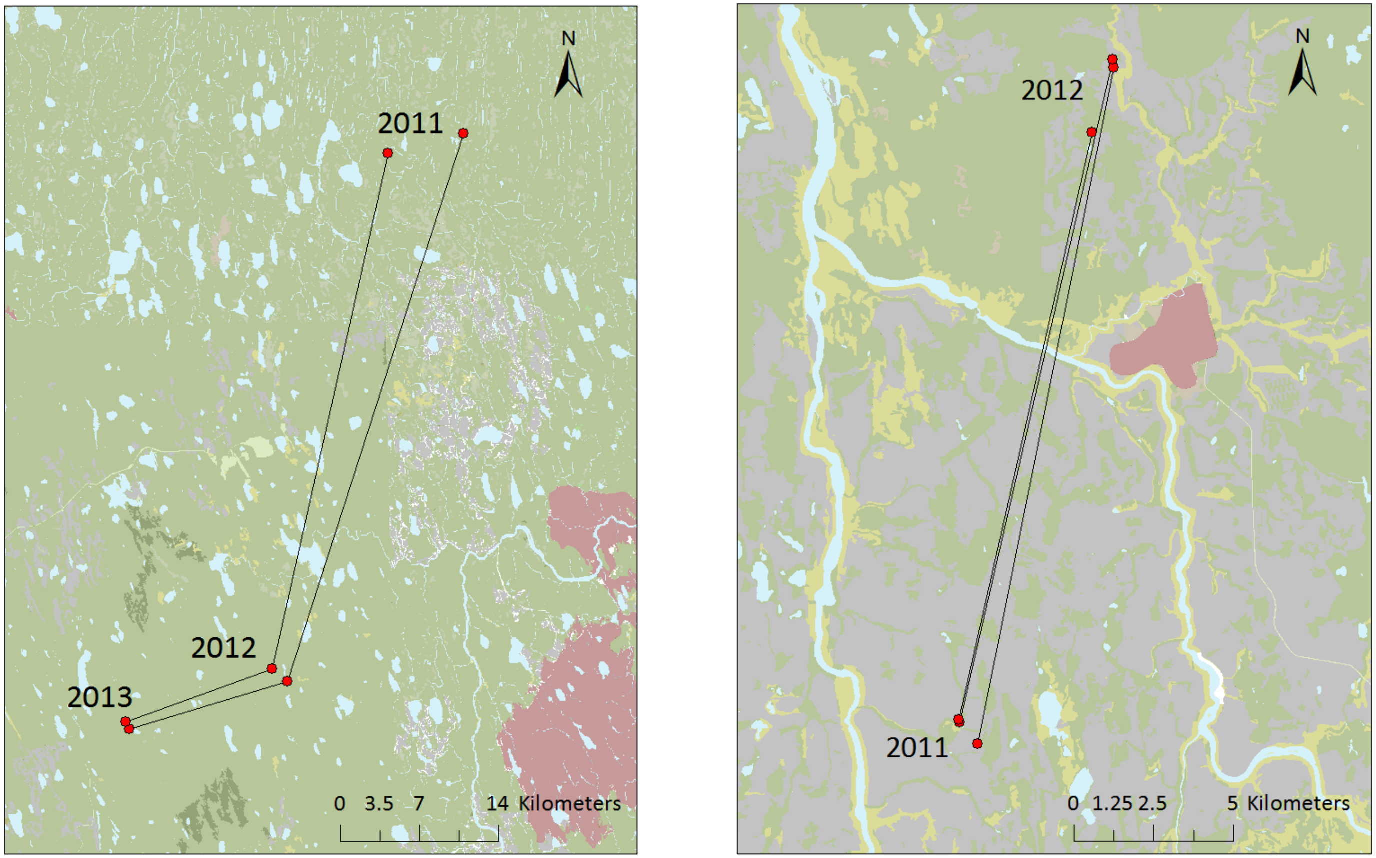

Figure 21. Aggregations of caribou (two and three individuals respectively) in the Kesagami range showing aggregations moving together across years during the Winter season. Points represent average locations for an animal for that Winter season.

Chapter 4: General Discussion

Woodland caribou are susceptible to human disturbances (Dalerum et al. 2007; McCarthy et al. 2011; Wasser et al. 2011; Dussault et al. 2012; Fortin et al. 2013), yet appear to maintain site fidelity independent of the influences of anthropogenic disturbances (Table 8 and 9). As adult and calf mortality increases nearer to disturbances (James and Stuart-Smith 2000; Courtois et al. 2007; Dussault et al. 2012), the maintenance of site fidelity regardless of the presence of anthropogenic disturbances suggests that its maintenance could be maladaptive under certain conditions. Instead site fidelity strength at the scale measured within this study appears contingent upon individual caribou habits and may represent a more complicated behaviour than previously thought, with site fidelity occurring not just in consecutive years. My research has shown that caribou may be exhibiting site fidelity on a longer, multiannual, temporal scale. Caribou appear to return to locations they had visited multiple years before, after having visited a different site in the intervening year(s) (Figure 16 and B25 to B29).

This study is limited by both the technology and resources at its disposal. Collaring caribou with GPS collars is infrequent due to costs and the GPS collars have a limited functional lifespan. We must therefore make assumptions about life-long behaviours from snapshots of time. The presence of non-consecutive-year site fidelity and its prevalence among caribou should be validated on more populations and more individuals that have been followed for longer than three years. A better understanding of the relationships among habitat, fidelity behaviour, and fitness requires more detailed information than available in this study. Longer term individual animal data may give insight into the presence of multiannual site fidelity. As site fidelity in this study was measured on the scale of kilometres, merely being near a disturbance may not result in a caribou altering their behaviour while a direct interaction with a predator may. Direct measures of reproduction and survival coupled with more detailed environmental data would shed light on the whether the patterns of site fidelity identified here are truly maladaptive.

Preservation of important habitats is a key goal in the strategy for woodland caribou conservation (Manitoba Conservation 2005; Ontario Woodland Caribou Recovery Team 2008; Environment Canada 2012), yet identifying areas such as calving and post-calving grounds is complicated by an incomplete understanding of caribou movement behaviour. Without a more complete understanding of site fidelity we may be asking the wrong questions in the wrong way. Future studies conducted solely within populations with little anthropogenic disturbance may give insight into the dynamics of how site fidelity manifested itself before humans appeared on the landscape and provide us with a baseline to compare fidelity behaviour in response to anthropogenic disturbances. It may also be of interest to study populations that are in frequent contact with humans across the entire study period, in landscapes where disturbances are constant. Both the Atikaki-Berens and the Owl-Flintstone ranges experienced a large portion of their disturbances before collaring, and experienced a signficiant decrease in logging activity in the latter part of the study. Additionally, a study investigating site fidelity before and after a large scale disturbance (e.g. the logging of a previously intact forest) would likely prove valuable in understanding the effects of disturbances on site fidelity.

Strengthening our knowledge of population and habitat health and the influence of disturbances on animal behaviour are consistently highlighted as knowledge gaps that need to be filled and will undoubtedly assist in the efforts to conserve woodland caribou (Manitoba Conservation 2005; Ontario Woodland Recovery Team 2008; Environment Canada 2012; Environment Canada 2012). The effectiveness of current conservation strategies is debatable with populations still in decline (Schneider et al. 2010; McCarthy et al. 2011; Environment Canada 2012), however, this decline may be the delayed result of historical disturbances (Vors et al. 2007). Future research needs to be aware of the potential discrepancy in our understanding of caribou fidelity and the limitations of the mapping techniques used in the boreal forest. My study highlights the importance of developing simple solutions like the creation of habitat mapping products that are tailored to the species and study area of interest, and provide consistent and continuous coverage for study populations that span multiple administrative boundaries. This would enable an increased confidence in the results of studies based on such maps.

Management Recommendations

These results indicate several important implications for population management and northern development that should be considered in future planning and development.

- Caribou likely rely on multiple calving and post-calving sites between years; conservation planning should take this into account when assessing development plans.
- Development should be limited as much as possible to areas that are less frequented by caribou particularly during the calving and post-calving

periods. Relying on caribou to relocate to a pristine area may be unfounded given the potential for these disturbed areas acting as an ecological trap.

- Monitoring of behavioural responses, in the absence of measures of reproductive success and survivorship, may not reveal important effects on caribou, since evidence elsewhere demonstrates proximity to natural disturbances has a negative effect on survival.
- In the three populations that I studied, there appears to be a connection between the amount of disturbances within the range and the size of the Winter range. This suggests that preserving areas of sufficient size, known to be used during the Winter, may help ensure population viability.
- Individual site fidelity of female woodland caribou does not appear to serve as an effective indicator of habitat quality (denoted by proximity to disturbances), at least at our current state of knowledge. Population-level analyses have not been satisfactorily investigated.

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Appendix A

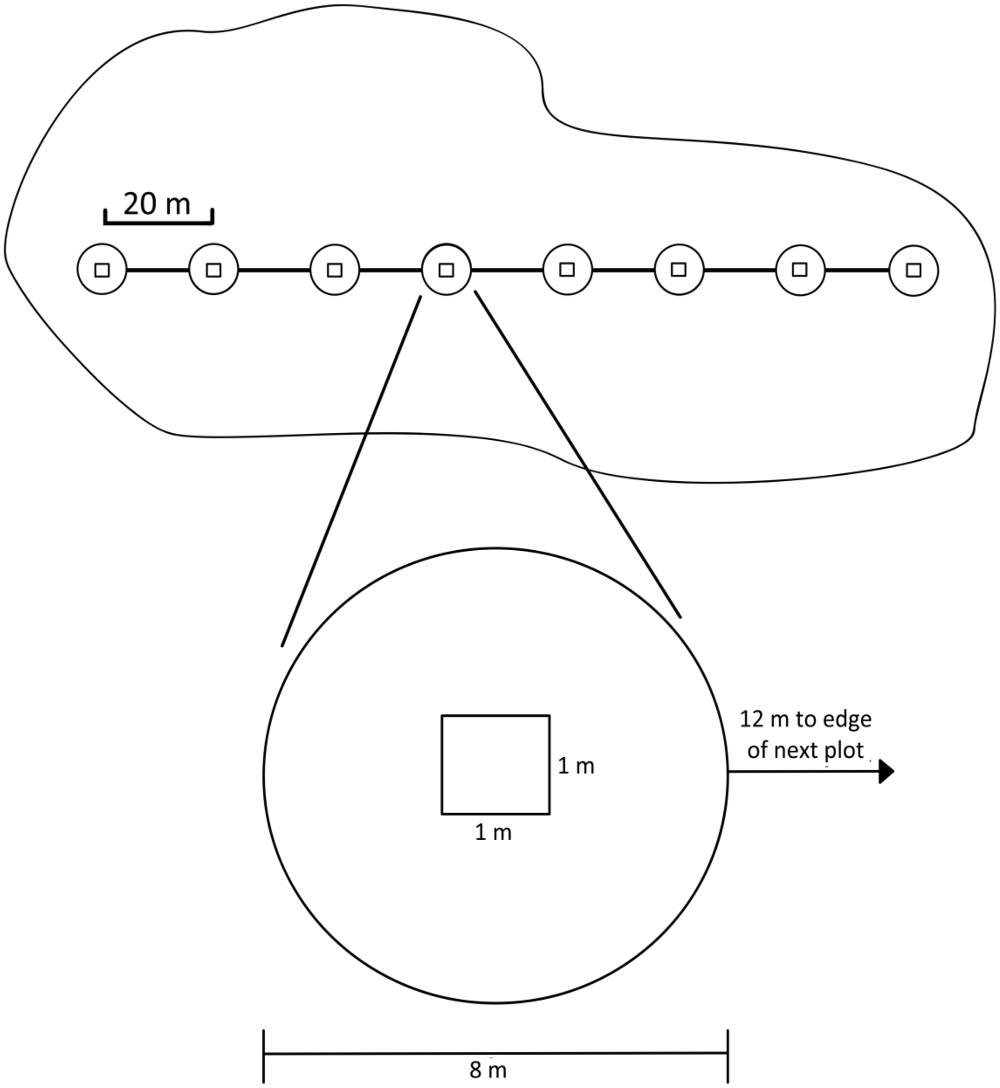


Figure A1. Schematic diagram illustrating the arrangement of sampling points along the transect for vegetation sampling.

Table A1. GPS locational fix information for each animal obtained in each population. Number of years refers to the number of calendar years that contained at least a single location fix.

Population	Animal ID	Total Number of Points	Total Number of Points after Filtering	Number of Years	Used In Analysis
Atikaki-Berens	Atiko01	4549	647	4	Yes
	Atiko02	807	79	1	No
	Atiko03	4065	581	3	Yes
	Atiko04	286	55	1	No
	Atiko05	21	4	1	No
	Atiko07	11721	491	2	No
	Atiko10	969	114	1	No
	Atiko11	2705	331	2	Yes
	Atiko12	1696	210	1	No
	Atiko14	20850	1083	4	Yes
	Atiko19	21042	1082	4	Yes
	Atiko20	14770	608	2	Yes
	BEREN03	2633	310	2	No
	BEREN04	2893	334	2	Yes
	BEREN05	2638	306	2	No
	BEREN06	2552	294	2	No
	BLDVN01	4417	557	3	Yes
	BLDVN06	2951	338	2	Yes
	BLDVN07	967	110	1	No
	BLDVN08	3609	399	2	Yes
	BLDVN10	14492	724	1	Yes
	BLDVN12	2726	320	2	No
	BLDVN13	16345	676	3	Yes
	ROUND02	4591	595	3	Yes
	ROUND05	4741	628	3	Yes
	ROUND06	5250	622	3	Yes
	ROUND08	1400	159	1	No
	ROUND09	5655	663	3	Yes
Kesagami	107	2606	1027	4	Yes
	108	1988	917	4	Yes
	109	626	621	3	Yes
	111	2503	929	4	Yes
	112	2530	960	4	Yes
	120	874	155	1	No
	169	1938	370	2	No
	170	2571	1003	4	Yes
	172	1622	842	4	Yes
	173	2584	1014	4	Yes

	174	1673	294	2	No
	175	2158	600	3	Yes
	176	2577	1010	4	Yes
	177	2360	794	3	Yes
	178	628	110	2	No
	179	2579	1021	4	Yes
	180	2584	1020	4	Yes
	181	2213	676	3	Yes
	182	2221	680	3	Yes
	183	2232	673	3	Yes
	184	2528	967	3	Yes
	188	2574	1002	4	Yes
Owl-Flintstone	GPS01	2033	196	2	No
	GPS02	1164	239	2	No
	GPS03	684	127	1	No
	GPS05	2614	371	4	No
	GPS06	1575	227	3	Yes
	GPS07	1576	249	2	No
	GPS09	3850	485	4	Yes
	GPS10	965	117	1	No
	GPS11	3675	450	3	Yes
	OWL07	23088	1483	6	Yes
	OWL10	7467	753	3	Yes
	OWL11	28412	1356	5	Yes
	OWL12	2654	199	2	Yes
	OWL17	42351	1727	6	Yes
	OWL18	17781	713	3	Yes
	OWL22	33475	1364	5	Yes
	OWL23	8804	355	2	No
	OWL31	15438	675	3	Yes
	OWL32	16140	674	3	Yes
	OWL33	15253	673	3	Yes
	OWL34	15048	671	3	Yes
	OWL36	16135	674	3	Yes
Grand Total		467692	42783		

Table A2. Classification used to define field sites based on field observations for FRI maps. Modified from Elkie et al. (2009). When there were multiple values for the classification order "**OR**" represents the beginning of a new equation.

Classification	**Equation***	**Classification Order**
Clearcut	Visually identified as Clearcut on ground – presence of stumps <40 years old	1
Rock	Visually identified as Rock	2
Marsh-shrub	Shrub densiometer cover >51% OR >51% cover above 1 metre entire transect average	3
Open Muskeg	Visually identified as Muskeg AND Canopy cover <10% AND (soil dampness >4 OR sphagnum cover percentage >5)	4
Treed Muskeg	Visually identified as Muskeg AND canopy cover >10%	5
Disturbed/Young	Visually identified as young **OR** density of trees with dbh > 20cm < 10% of trees	6
Deciduous	Any deciduous >= 20%	7
Upland Conifer	RP >= 70% **OR (**(WP +RP + WS + HE) >= 40% **AND** WP > 30%) **OR** ((WP + RP) >= 40%) **OR** ((BS+WS+BF+CE+LA+WP+JP+RP+HE) >= 70% **AND** (BF+CE+WP+LA+WS+HE) >= 20%) **OR**	8
Black Spruce	(BS >= 80% **AND** (HM + YB + EL + OA + BW + RP) = 0%)	9
Jack Pine	JP > 70%	10
Lowland Conifer	((CE+LA+BS) >= 80% **AND** (HM+YB+EL+OA+BW+RP) = 0%	11
Jack Pine	JP > 70% **OR** ((JP+BS+RP)>70% **AND** (JP+BS+BF+WS+HE+WP+RP+CE+LA)>=70% **AND** (BF+WS+HE+WP+CE+LA) <=20% **AND** (JP> BS)) **OR** ((JP >= 50% AND (JP+BS+BF+WS+HE+WP+RP+CE+LA)>=70% **AND** (BF+WS+HE+WP+CE+LA)<= 20 % **AND** (JP >= SB))	12
Upland Conifer	(JP >= 30% AND (BS+WS+BF+CE+LA+WP+JP+RP+HE) >=70%)	13
Mixed Conifer	(BS+WS+BF+CE+LA+WP+JP+RP+HE) >= 70%	14

*Forest stand abbreviations - JP = Jack Pine (*Pinus banksiana*), BS = Black Spruce (*Picea mariana*), RP = Red Pine (*Pinus resinosa*), BF = Balsalm Fir (*Abies balsamifera*), WS = White Spruce (*Picea glauca*), HE = Hemlock (*Tsuga spp.*), WP = White Pine (*Pinus stroba*), CE = Eastern White Cedar (*Thuja occidentalis*), LA = Tamarack (*Larix laricina*), HM = Maples (*Acer spp.*), YB = Birches (*Betula spp.*), EL = Elms (*Ulmus spp.*), OA = Oaks (*Quercus spp.*)

Table A3. Classification used to define field sites based on field observations for Landsat TM maps. Modified OMNR (2012)

Classification	**Equation***	**Classification Order**
Marsh-Shrub	Hydrophytic shrubs over 1m > 25% **AND** canopy cover <11% **AND** (Soil dampness rating >3 **OR** Sphagnum cover > 50% **OR** Water >5%)	1
Disturbance	Visually identified as young **AND** no trees present with dbh greater than 20cm	2
Open Wetland	Canopy cover <11% **AND** (Soil dampness rating >3 **OR** Sphagnum cover > 50% **OR** Water >5%)	3
Deciduous	Deciduous canopy cover > 75%	4
Mixed	Deciduous canopy cover >10%	5
Sparse	Coniferous canopy cover <25% **AND** >10%	6
Treed Wetland	Tall shrub canopy cover < 25% **AND** BS + TA >50% of all coniferous trees **AND** Canopy cover >10% **AND** (Soil dampness rating >3 **OR** Sphagnum cover > 50% **OR** Water >5%) **AND** Ericaceous Shrub cover >35%	7
Coniferous Swamp	(Hydrophytic Shrubs below 1m > 25% **OR** Coniferous canopy cover > 25%) AND Soil dampness rating >2	8
Conifer	Deciduous canopy cover = 0 **AND** Tamarack < 25% of all trees	9

*Forest stand abbreviations **-** TA = Tamarack, BS = Black Spruce, dbh = Diameter at breast height

Table A4. Number of sites of each FRI class selected for field sampling with the resultant classification from field measurements.

FRI Site Classification	Field sampling			After field calculations		
	Total	**West**	**East**	**Total**	**West**	**East**
Black Spruce	22	10	12	30	11	19
Clearcut	14	14	0	13	13	0
Deciduous	43	18	25	27	14	13
Jack Pine	16	6	10	21	12	9
Lowland Conifer	21	13	8	10	7	3
Marsh-Shrub	24	14	10	22	14	8
Mixed Conifer	11	6	5	31	12	19
Open Muskeg	23	13	10	20	10	10
Rock	13	13	0	15	15	0
Treed Muskeg	18	6	12	26	11	15
Upland Conifer	13	5	8	3	3	0
Young Conifer	19	6	13	29	8	21
Young Deciduous	24	12	12	14	6	8

Table A5. Total number of sites for each Landsat TM class selected for field sampling with the resultant classification from field measurements.

Landsat TM Site Classification	**Number selected for field sampling**	**Totals after field calculations**
Conifer	12	13
Coniferous Swamp	36	37
Deciduous	12	14
Disturbance	12	15
Marsh-Shrub	3	2
Mixed	16	38
Open Wetland	11	15
Sparse	15	1
Treed Wetland	22	5
Water	1	0

Table A6. Confusion matrix for all FRI maps used in the study. Map Classification is the original classifications, and Field Results are the ground-truthed classifications. Bold numbers indicate correct classification. Zero entries are blank.

	Field Classification													
Map Classification	Black Spruce	Clearcut	Deciduous	Jack Pine	Lowland Conifer	Marsh-Shrub	Mixed Conifer	Open Muskeg	Rock	Treed Muskeg	Upland Conifer	Young Conifer	Young Deciduous	Total
Black Spruce	**13**		1	1		1	2			1	1	2		22
Clearcut		**10**	3										1	14
Deciduous	3		**12**	1	1	1	15		3	1	1	2	3	43
Jack Pine			1	**14**								1		16
Lowland Conifer	6		1	1	**5**		4			1		2	1	21
Marsh-Shrub	2					**13**	1	3		1		3	1	24
Mixed Conifer	1		1	1			**5**					2	1	11
Open Muskeg								**13**	1	8		1		23
Rock		1		1	1				**10**					13
Treed Muskeg	2				2	1		3		**10**				18
Upland Conifer	2		2	2	1		2			1	**1**	2		13
Young Conifer	1	1				3	1	1		2		**9**	1	19
Young Deciduous		1	6			3	1		1	1		5	**6**	24
Total	30	13	27	21	10	22	31	20	15	26	3	29	14	**261**

Table A7. Confusion matrix for only eastern region FRI maps. Map Classification is the original classifications, and Field Results are the new ground-truthed classifications. Bold numbers indicate correct classification. Zero entries are blank.

	Field Results											
Map Classification	Black Spruce	Deciduous	Jack Pine	Lowland Conifer	Marsh-Shrub	Mixed Conifer	Open Muskeg	Rock	Treed Muskeg	Upland Conifer	Young Conifer	Young Deciduous	Total
Black Spruce	**8**				1	2					1		12
Deciduous	3	**7**		1		9			1		2	2	25
Jack Pine		1	**9**										10
Lowland Conifer	4	1		**0**		2					1		8
Marsh-Shrub	1				**5**	1					2	1	10
Mixed Conifer						**2**					2	1	5
Open Muskeg							**6**		4				10
Rock								**0**					0
Treed Muskeg	1			1			3		**7**				12
Upland Conifer	2	1		1		2				**0**	2		8
Young Conifer					1		1		2		**8**	1	13
Young Deciduous		3			1	1			1		3	**3**	12
Total	19	13	9	3	8	19	10	0	15	0	21	8	**125**

Table A8. Confusion Matrix for only western region FRI maps. Map Classification is the original classifications, and Field Results are the new ground-truthed classifications. Bold numbers indicate correct classification. Zero entries are blank.

	Field Results													
Map Classification	Black Spruce	Clearcut	Deciduous	Jack Pine	Lowland Conifer	Marsh-Shrub	Mixed Conifer	Open Muskeg	Rock	Treed Muskeg	Upland Conifer	Young Conifer	Young Deciduous	Total
Black Spruce	**5**		1	1						1	1	1		10
Clearcut		**10**	3										1	14
Deciduous			**5**	1		1	6		3		1		1	18
Jack Pine				**5**								1		6
Lowland Conifer	2			1	**5**		2			1		1	1	13
Marsh-Shrub	1					**8**		3		1		1		14
Mixed Conifer	1		1	1			**3**							6
Open Muskeg								**7**	1	4		1		13
Rock		1		1	1				**10**					13
Treed Muskeg	1				1	1				**3**				6
Upland Conifer			1	2						1	**1**			5
Young Conifer	1	1				2	1					**1**		6
Young Deciduous		1	3			2			1			2	**3**	12
Total	11	13	14	12	7	14	12	10	15	11	3	8	6	**136**

Table A9. Confusion Matrix for all Landsat TM maps used in the study. Map Classification is the original classifications, and Field Results are the ground-truthed classifications. Bold numbers indicate correct classification. Zero entries are blank.

	Field Results									
Map Classification	Conifer	Coniferous Swamp	Deciduous	Young	Marsh-Shrub	Mixed	Open Wetland	Sparse	Treed Wetland	Total
Conifer	**6**	4				2				12
Coniferous Swamp	4	**17**		2		9	2		2	36
Deciduous			**8**			4				12
Young		1		**10**		1				12
Marsh-Shrub			1		**1**	1				3
Mixed			5			**11**				16
Open Wetland	2	2		1		2	**4**			11
Sparse	1	5		2		7		**0**		15
Treed Wetland		8			1	1	8	1	**3**	**22**
Water							1			1
Total	13	37	14	15	2	38	15	1	5	140

Table A10. New classification scheme after commonly misclassified stands were amalgamated to increase accuracy in FRI maps.

Original Classification	**Amalgamated Classification**
Deciduous	Deciduous
Mixed Conifer	
Upland Conifer	
Clearcut	Disturbance
Jack Pine	Jack Pine
Black Spruce	Lowland Conifer
Lowland Conifer	
Marsh-Shrub	Marsh-Shrub
Open Muskeg	Muskeg
Treed Muskeg	
Rock	Rock
Young Conifer	Young
Young Deciduous	

Table A11. Confusion Matrix for all FRI maps used in the study after commonly misclassified sites were amalgamated. Map Classification is the original classifications, and Field Results are the ground-truthed classifications. Bold numbers indicate correct classification. Zero entries are blank.

	Field Results								
Map Classification	Deciduous	Disturbance	Jack Pine	Lowland Conifer	Marsh-Shrub	Muskeg	Rock	Young	Total
Deciduous	**33**		2	6	1	1	3	8	54
Disturbance	3	**10**						1	14
Jack Pine	1		**14**					1	16
Lowland Conifer	12		4	**29**	1	3		7	56
Marsh-Shrub	1			2	**13**	4		4	24
Muskeg				4	1	**34**	1	1	41
Rock		1	1	1			**10**		13
Young	8	2		1	6	4	1	**21**	43
Total	58	13	21	43	22	46	15	43	**261**

Table A12. Confusion matrix for FRI maps from the eastern region after commonly misclassified sites were amalgamated. Map classification is the original classifications, and field results are the new ground-truthed classifications. Bold numbers indicate correct classification. Zero entries are blank.

	Field Results						
Map Classification	Deciduous	Jack Pine	Lowland Conifer	Marsh-Shrub	Muskeg	Young	Total
Deciduous	**18**		4		1	7	30
Jack Pine	1	**9**					10
Lowland Conifer	8		**15**	1		4	28
Marsh-Shrub	1		1	**5**		3	10
Muskeg			2		**20**		22
Young	4			2	4	**15**	25
Total	32	9	22	8	25	29	**125**

Table A13. Confusion matrix for FRI maps from the western region after commonly misclassified sites were amalgamated. Map classification is the original classifications, and field results are the new ground-truthed classifications. Bold numbers indicate correct classification. Zero entries are blank.

	Field Results								
Map Classification	Deciduous	Disturbance	Jack Pine	Lowland Conifer	Marsh-Shrub	Muskeg	Rock	Young	Total
Deciduous	**15**		2	2	1		3	1	24
Disturbance	3	**10**						1	14
Jack Pine			**5**					1	6
Lowland Conifer	4		4	**14**		3		3	28
Marsh-Shrub				1	**8**	4		1	14
Muskeg				2	1	**14**			17
Rock		1	1	1			**10**	1	14
Young	4	2		1	4		1	**6**	18
Total	26	13	12	21	14	21	14	14	**135**

Table A14. New classification scheme after commonly misclassified stands were amalgamated to increase accuracy in Landsat TM maps.

Original Classification	**Amalgamated Classification**
Conifer	Conifer
Coniferous Swamp	Coniferous Swamp
Deciduous	Deciduous
Marsh-Shrub	
Mixed	
Sparse	
Disturbance	Disturbance
Open Wetland	Wetland
Treed Wetland	

Table A15. Confusion Matrix for Landsat TM maps used in the study after commonly misclassified sites were amalgamated. Map Classification is the original classifications, and Field Results are the ground-truthed classifications. Bold numbers indicate correct classification. Zero entries are blank.

	Field Results						
Map Classification	Conifer	Coniferous Swamp	Deciduous	Young	Wetland	Water	Total
Conifer	**6**	4	1		2		13
Coniferous Swamp	4	**17**	5	1	10		37
Deciduous	2	9	**38**	1	5		55
Young		2	2	**10**	1		15
Wetland		4			**15**	1	20
Total	12	36	46	12	**33**	1	140

Table A16. Confusion matrix of sites used to determine the predictive ability of the common map classification scheme. Using either FRI or Landsat TM maps to predict the classification of the other. Zero entries are blank and bolded entries are where each map predicted the same classification.

	Landsat TM Classification						
FRI Classification	Deciduous	Disturbance	Lowland Conifer	Upland Conifer	Water	Young	Total
Deciduous	**1028**	14	1269	553	98	112	3074
Disturbance	10	**80**	45	17	7	27	186
Lowland Conifer	447	42	**15116**	1144	590	595	17934
Upland Coniferous/Rock	194		809	**1485**	33	332	2853
Water	11		148	79	**1801**	37	2076
Young	737	91	1168	376	68	**3734**	6174
Total	2427	227	18555	3654	2597	4837	**32297**

Table A17. Common Map Classification accuracy for FRI maps across entire study. Bold numbers indicate correct classification. Zero entries are blank.

	Field Results					
Map Classification	Clearcut	Deciduous	Lowland Conifer	Upland Coniferous/Rock	Young	Total
Clearcut	**10**	3			1	14
Deciduous		**33**	8	5	8	54
Lowland Conifer		13	**91**	5	12	121
Upland Coniferous/Rock	1	1	1	**25**	1	29
Young	2	8	11	1	**21**	43
Total	13	58	111	36	43	**261**

Table A18. Common Map Classification accuracy for eastern region FRI maps. Zero entries are blank.

	Field Results				
Map Classification	Deciduous	Lowland Conifer	Upland Coniferous /Rock	Young	Total
Deciduous	**18**	5		7	30
Lowland Conifer	9	**44**		7	60
Upland Coniferous/Rock	1		**9**		10
Young	4	6		**15**	25
Total	32	55	9	29	**125**

Table A19. Common Map Classification accuracy for western region FRI maps. Zero entries are blank.

	Field Results					
Map Classification	Clearcut	Deciduous	Lowland Conifer	Upland Coniferous /Rock	Young	Total
Clearcut	**10**	3			1	14
Deciduous		**15**	3	5	1	24
Lowland Conifer		4	**47**	5	5	61
Upland Coniferous/Rock	1		1	**16**	1	19
Young	2	4	5	1	**6**	18
Total	13	26	56	27	14	**136**

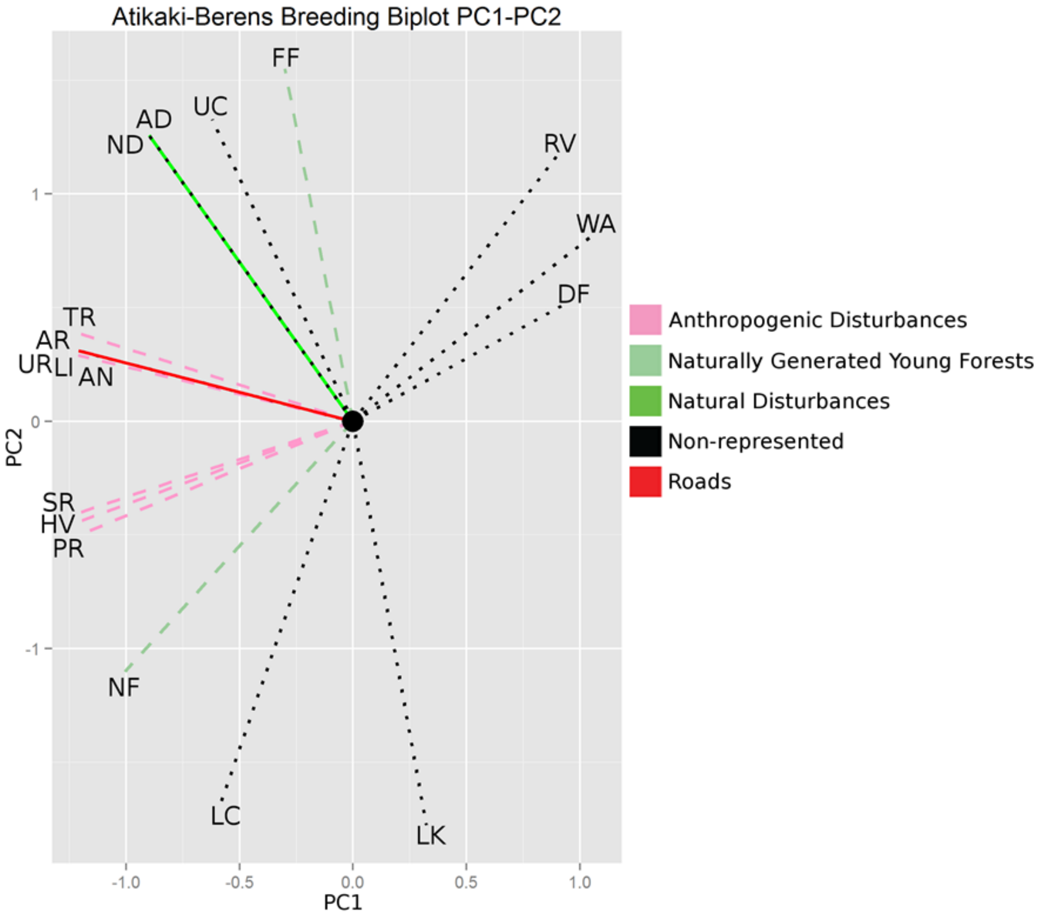
Table A20. Common Map Classification accuracy for Landsat TM maps across the entire study area. Bold numbers indicate correct classification. Zero entries are blank.

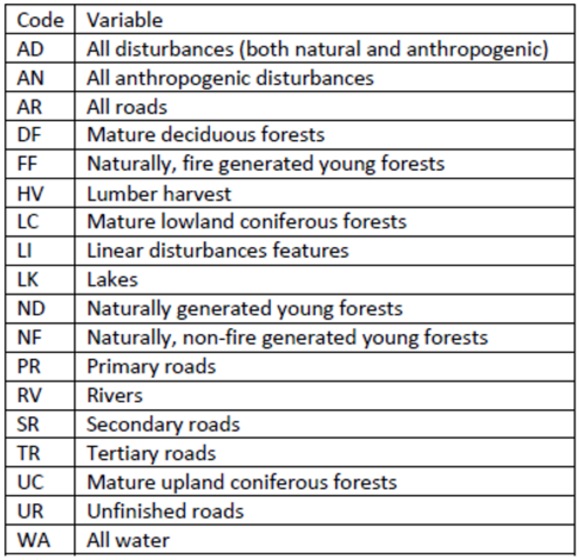
	Field Results				
Map Classification	Deciduous	Lowland Conifer	Upland Coniferous/Rock	Young	Total
Deciduous	**31**				31
Lowland Conifer	15	**62**	1	3	81
Upland Coniferous/Rock	7	6	**0**	2	15
Young	1	1		**10**	12
Water		1			1
Total	54	70	1	15	140

Table A21. FRI map accuracy when including only those stands that provided standardized definitions and removing non-merchantable stands. Zero entries are blank.

	Field Results						
Map Classification	Black Spruce	Deciduous	Jack Pine	Lowland Conifer	Mixed Conifer	Upland Conifer	Total
Black Spruce	**16**	1	1	1	1	1	21
Deciduous	4	**15**	4	1	15	1	40
Jack Pine		1	**15**			1	17
Lowland Conifer	7	2	1	**9**	4		23
Mixed Conifer	1	2	1		**8**		12
Upland Conifer	2	1	3	1	2	**0**	9
Total	30	22	25	12	30	3	**122**

Appendix B





Code	Variable
AD	All disturbances (both natural and anthropogenic)
AN	All anthropogenic disturbances
AR	All roads
DF	Mature deciduous forests
FF	Naturally, fire generated young forests
HV	Lumber harvest
LC	Mature lowland coniferous forests
LI	Linear disturbances features
LK	Lakes
ND	Naturally generated young forests
NF	Naturally, non-fire generated young forests
PR	Primary roads
RV	Rivers
SR	Secondary roads
TR	Tertiary roads
UC	Mature upland coniferous forests
UR	Unfinished roads
WA	All water

Figure B1. PCA biplot for the Breeding period in the Atikaki-Berens population. Solid lines denote variables used to represent a PC, dashed lines are variables that either are covariates or components of the representative variables. Dotted lines are unused variables. Two letter codes are provided above.

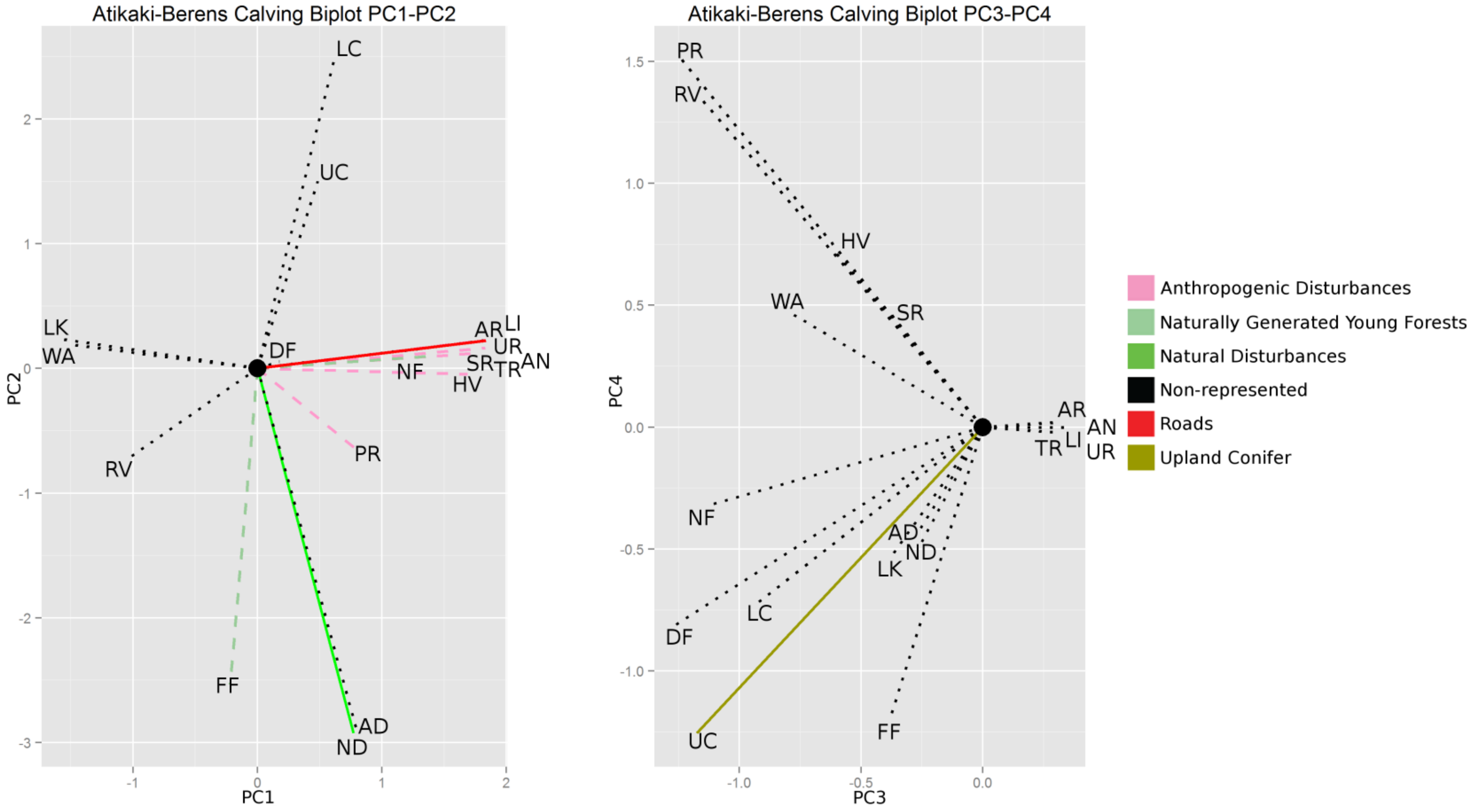


Figure B2. PCA biplot for the Calving period in the Atikaki-Berens population. Solid lines denote variables used to represent a PC, dashed lines are variables that either are covariates or components of the representative variables. Dotted lines are unused variables. Two letter codes are provided in Figure 1A.

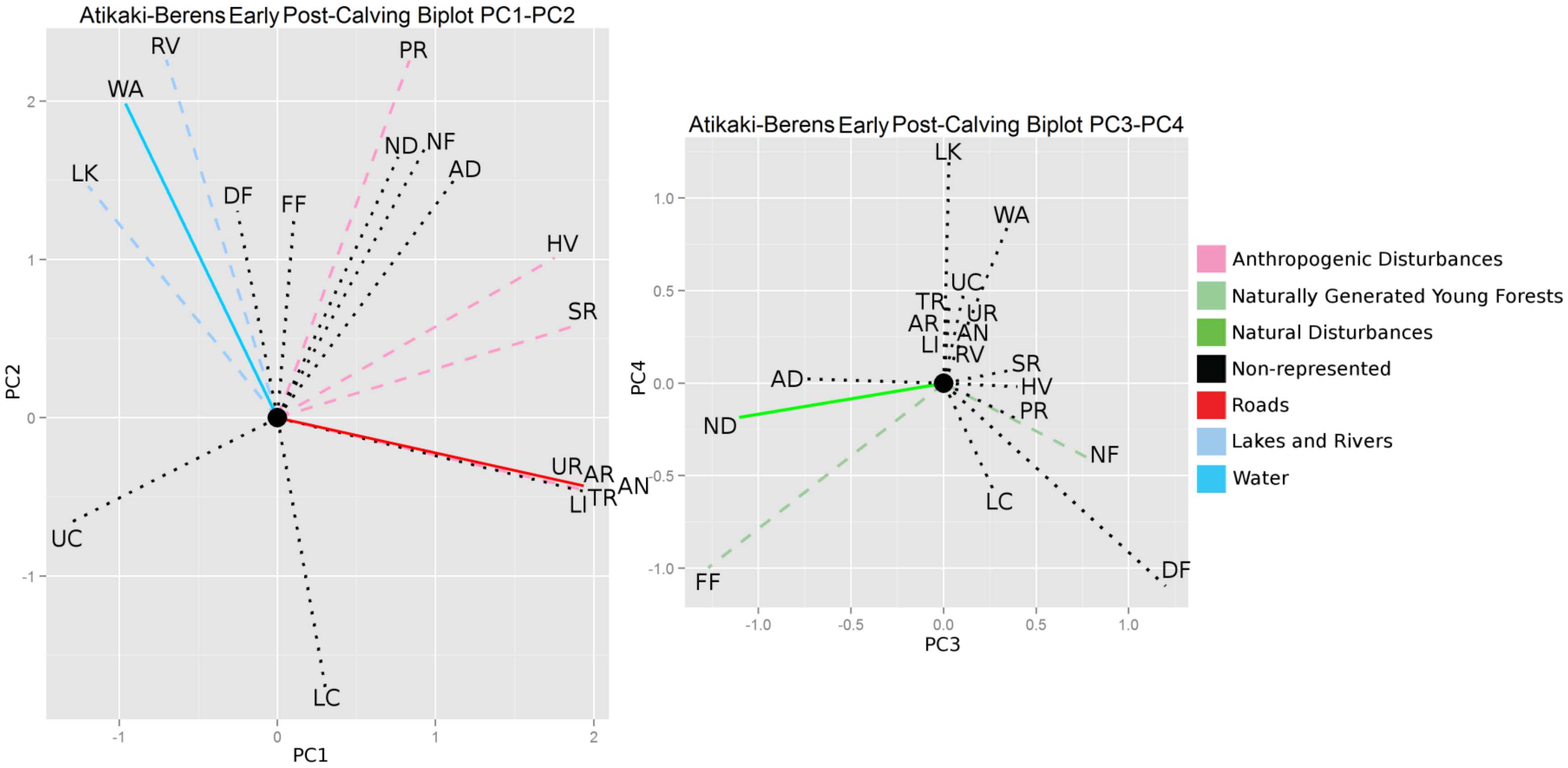


Figure B3. PCA biplot for the Early Post-Calving period in the Atikaki-Berens population. Solid lines denote variables used to represent a PC, dashed lines are variables that either are covariates or components of the representative variables. Dotted lines are unused variables. Two letter codes are provided in Figure 1A.

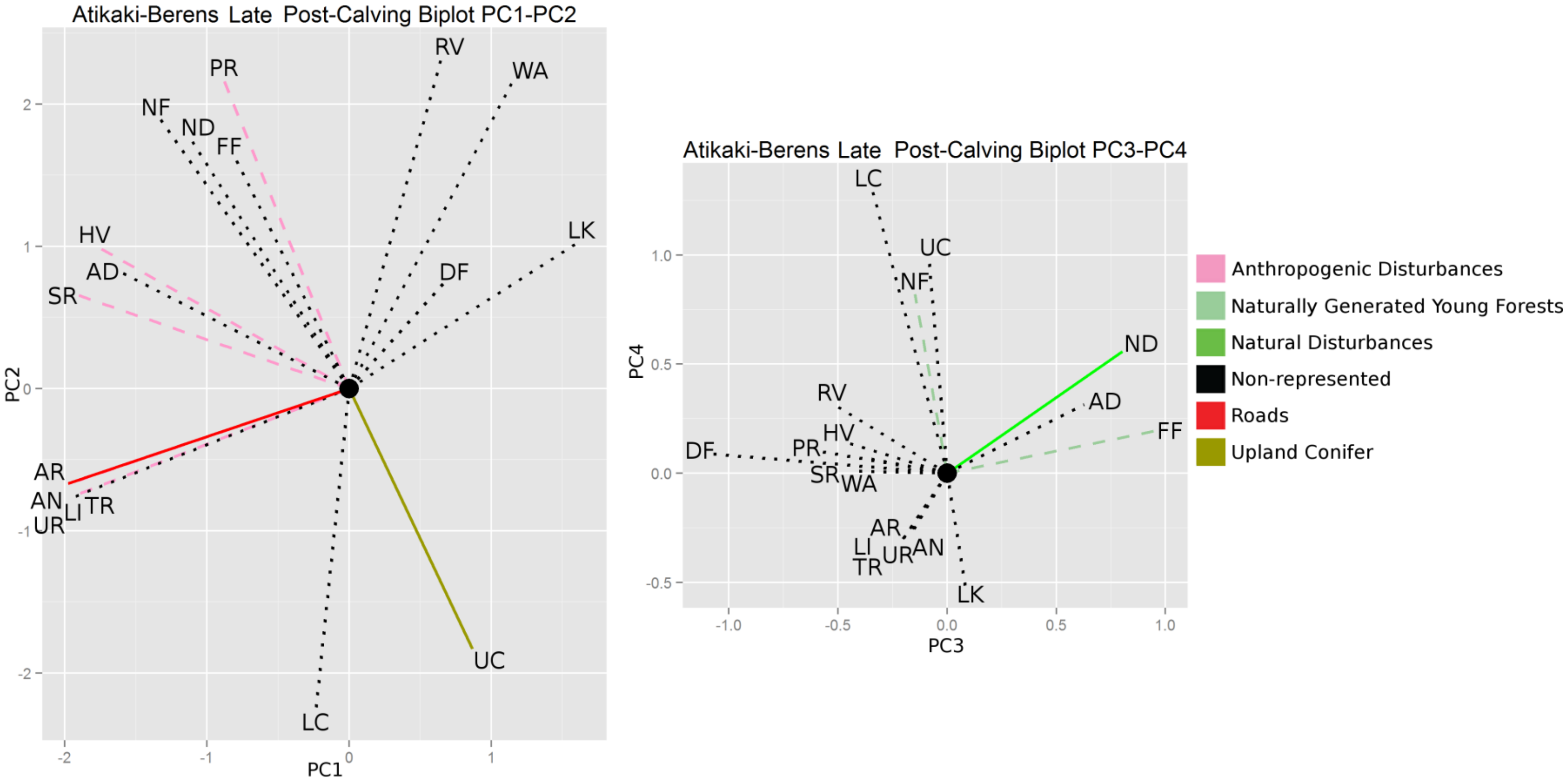


Figure B4. PCA biplot for the Late Post-Calving period in the Atikaki-Berens population. Solid lines denote variables used to represent a PC, dashed lines are variables that either are covariates or components of the representative variables. Dotted lines are unused variables. Two letter codes are provided in Figure 1A.

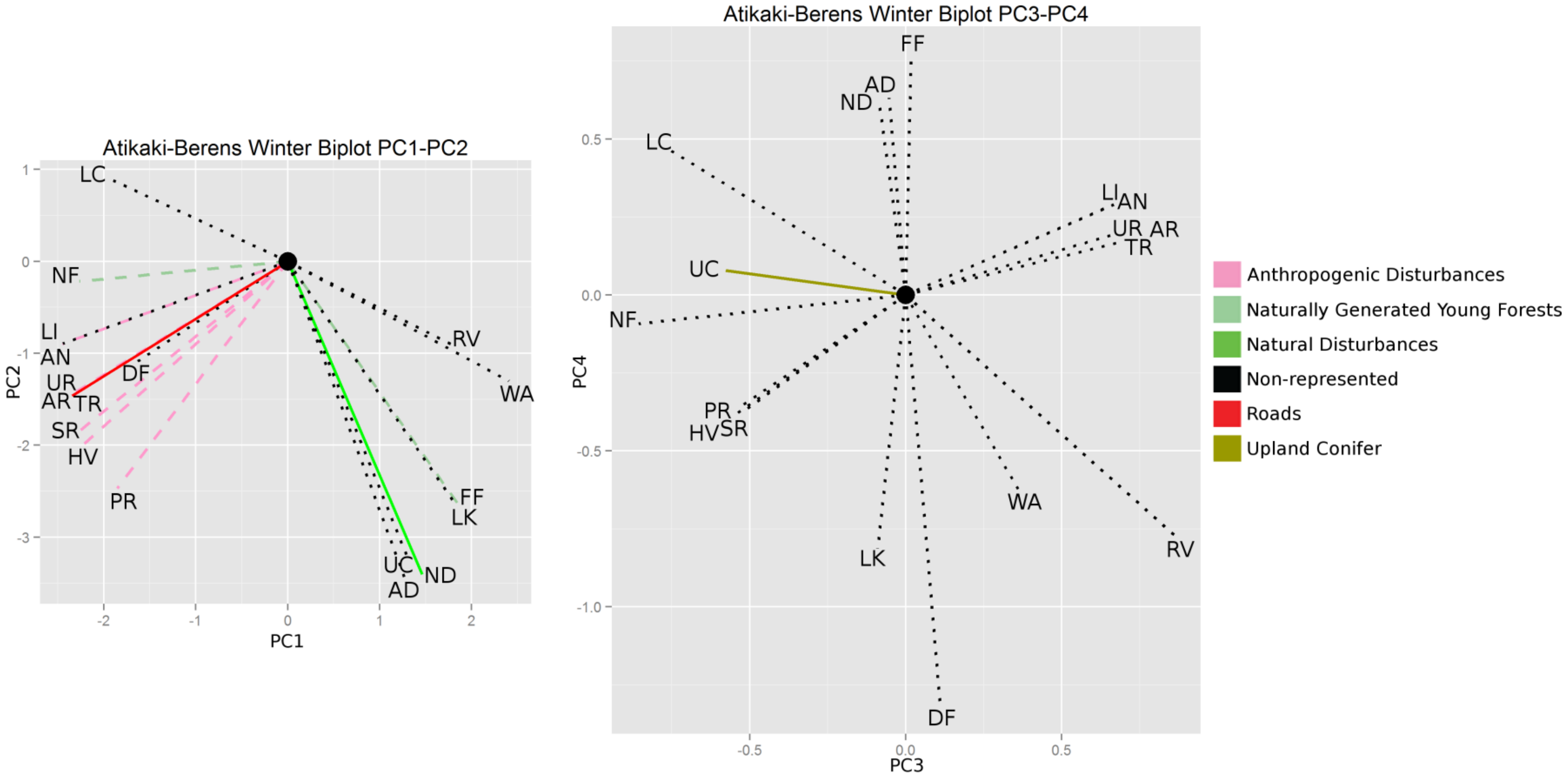


Figure B5. PCA biplot for the Winter period in the Atikaki-Berens population. Solid lines denote variables used to represent a PC, dashed lines are variables that either are covariates or components of the representative variables. Dotted lines are unused variables. Two letter codes are provided in Figure 1A.

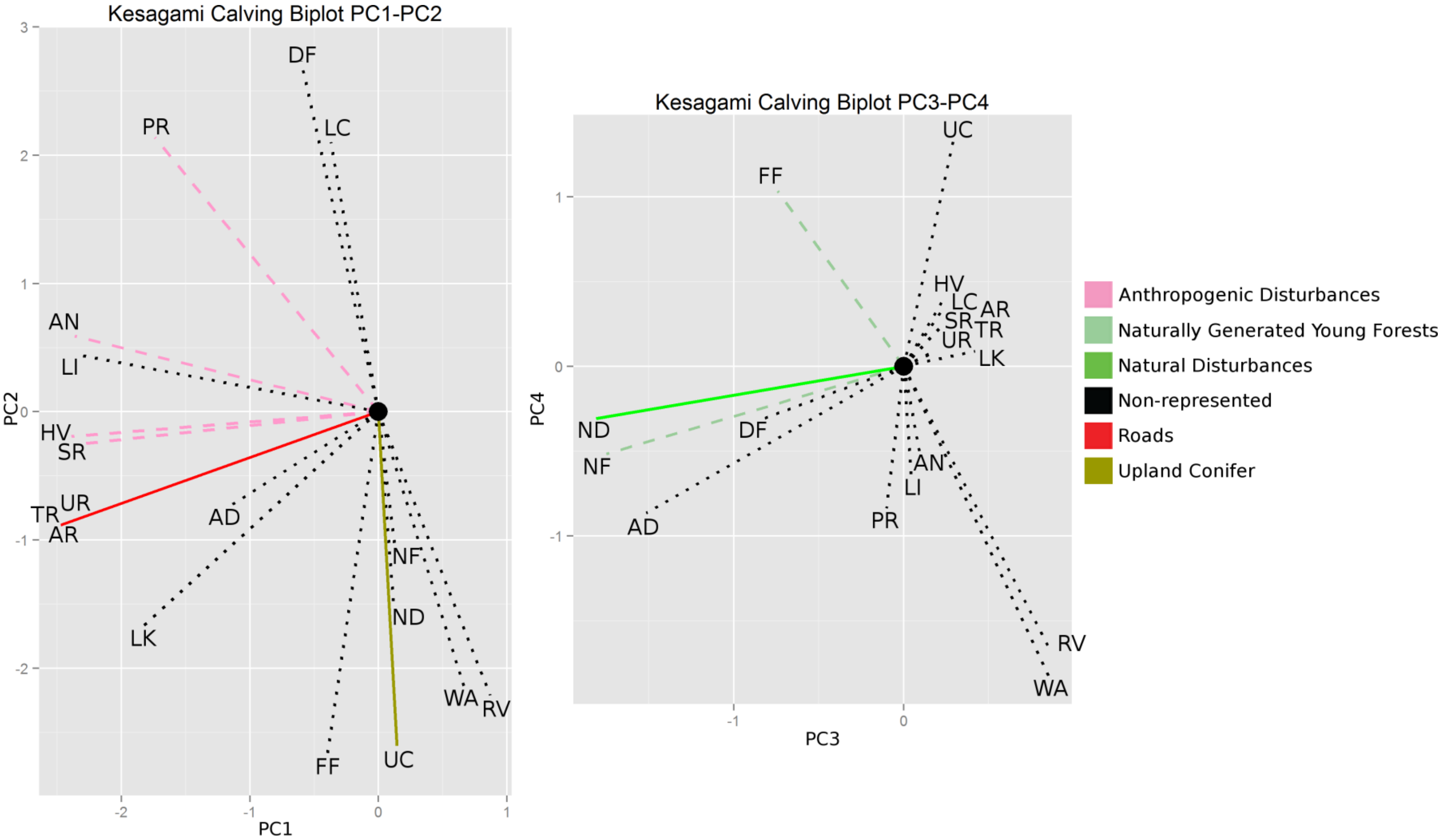


Figure B6. PCA biplot for the Calving period in the Kesagami population. Solid lines denote variables used to represent a PC, dashed lines are variables that either are covariates or components of the representative variables. Dotted lines are unused variables. Two letter codes are provided in Figure 1A.

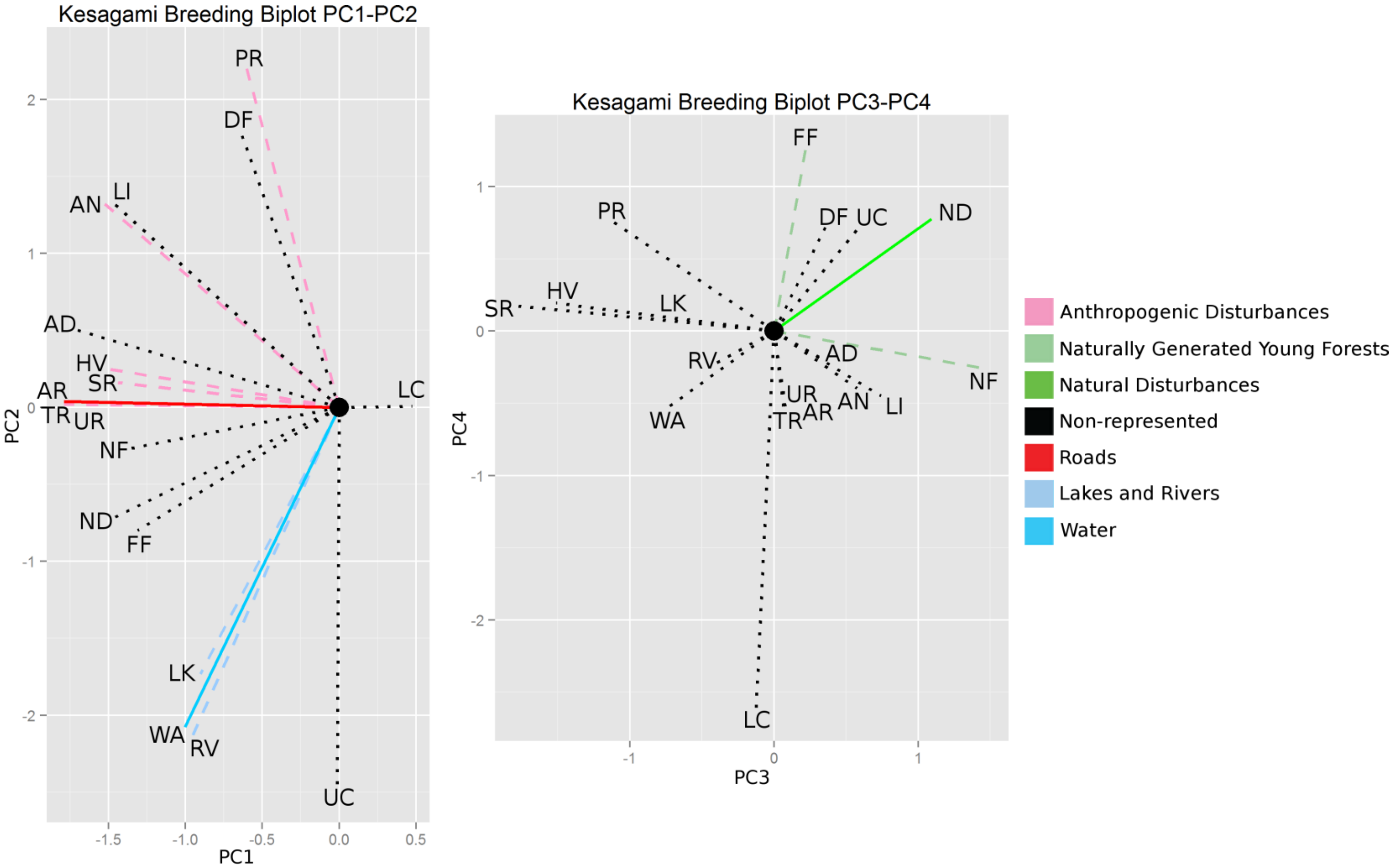


Figure B7. PCA biplot for the Breeding period in the Kesagami population. Solid lines denote variables used to represent a PC, dashed lines are variables that either are covariates or components of the representative variables. Dotted lines are unused variables. Two letter codes are provided in Figure 1A.

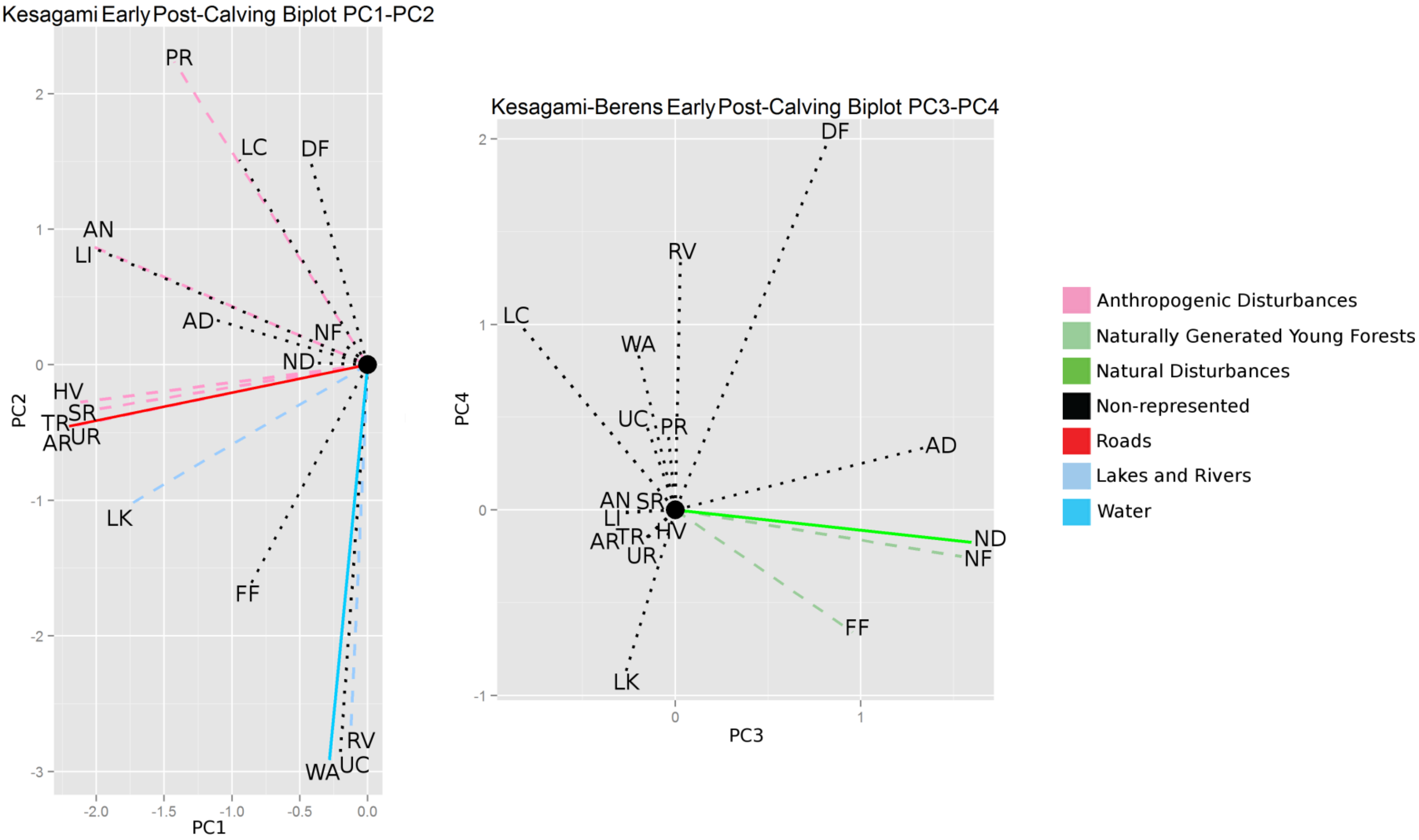


Figure B8. PCA biplot for the Early Post-Calving period in the Kesagami population. Solid lines denote variables used to represent a PC, dashed lines are variables that either are covariates or components of the representative variables. Dotted lines are unused variables. Two letter codes are provided in Figure 1A.

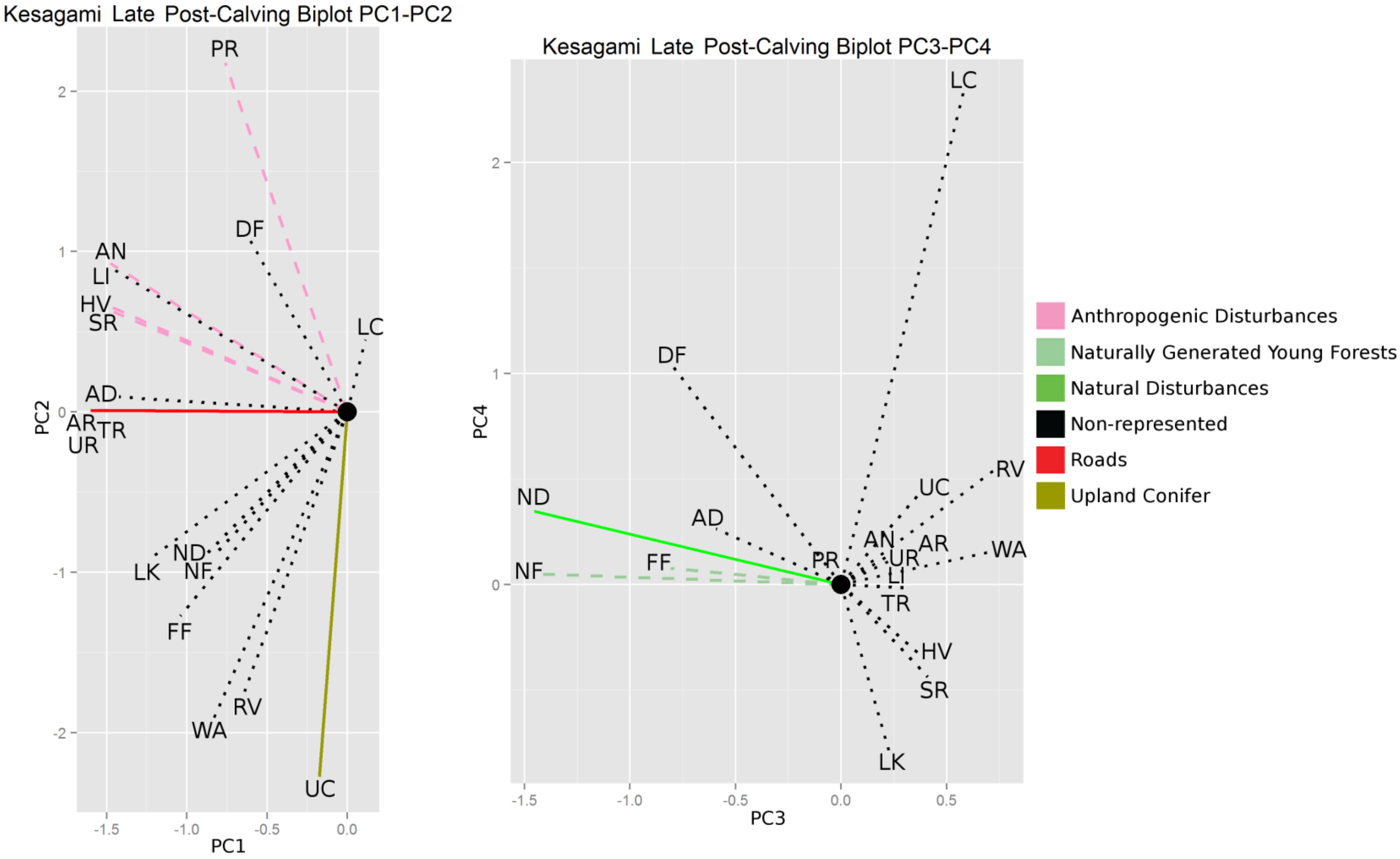


Figure B9. PCA biplot for the Late Post-Calving period in the Kesagami population. Solid lines denote variables used to represent a PC, dashed lines are variables that either are covariates or components of the representative variables. Dotted lines are unused variables. Two letter codes are provided in Figure 1A.

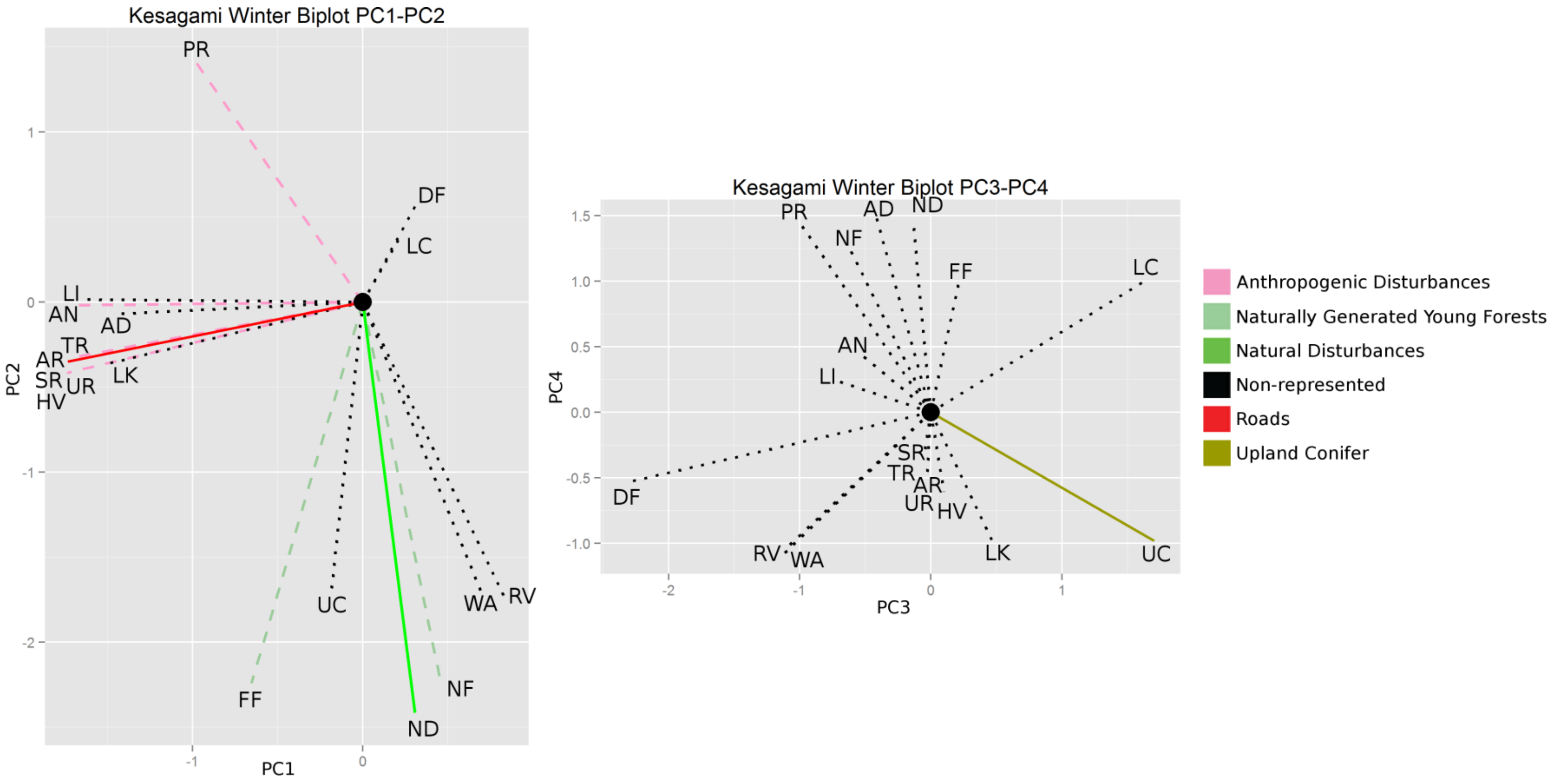


Figure B10. PCA biplot for the Winter period in the Kesagami population. Solid lines denote variables used to represent a PC, dashed lines are variables that either are covariates or components of the representative variables. Dotted lines are unused variables. Two letter codes are provided in Figure 1A.

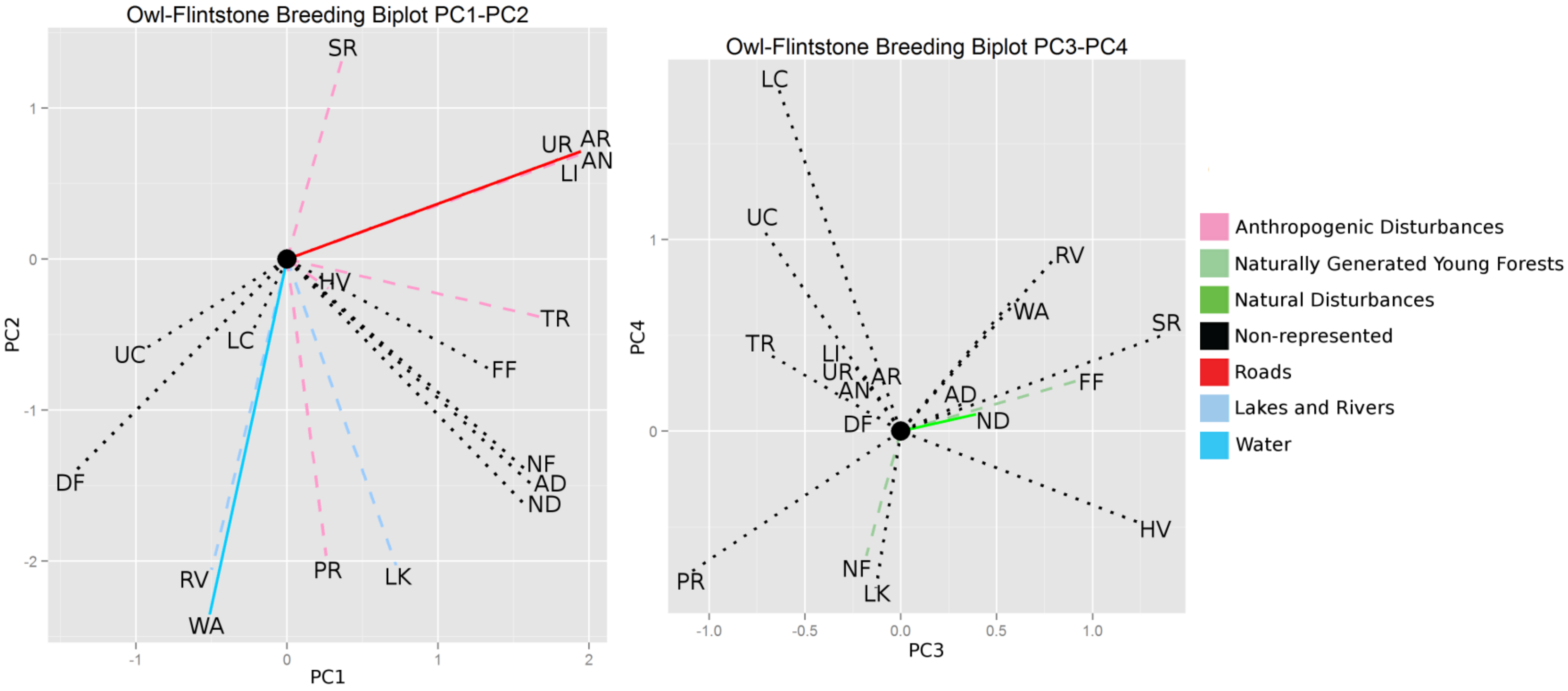


Figure B11. PCA biplot for the Breeding period in the Owl-Flintstone population. Solid lines denote variables used to represent a PC, dashed lines are variables that either are covariates or components of the representative variables. Dotted lines are unused variables. Two letter codes are provided in Figure 1A.

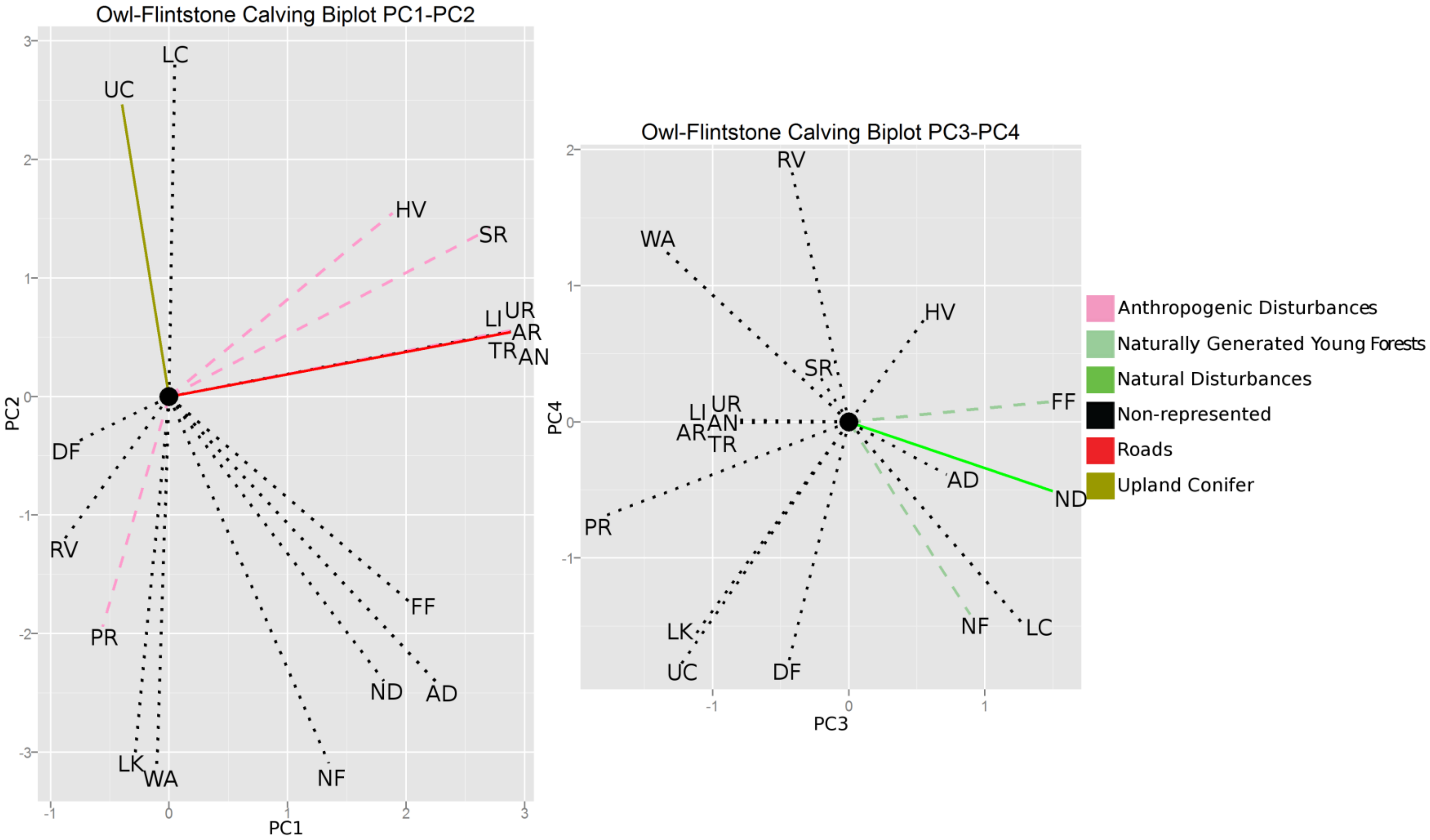


Figure B12. PCA biplot for the Calving period in the Owl-Flintstone population. Solid lines denote variables used to represent a PC, dashed lines are variables that either are covariates or components of the representative variables. Dotted lines are unused variables. Two letter codes are provided in Figure 1A.

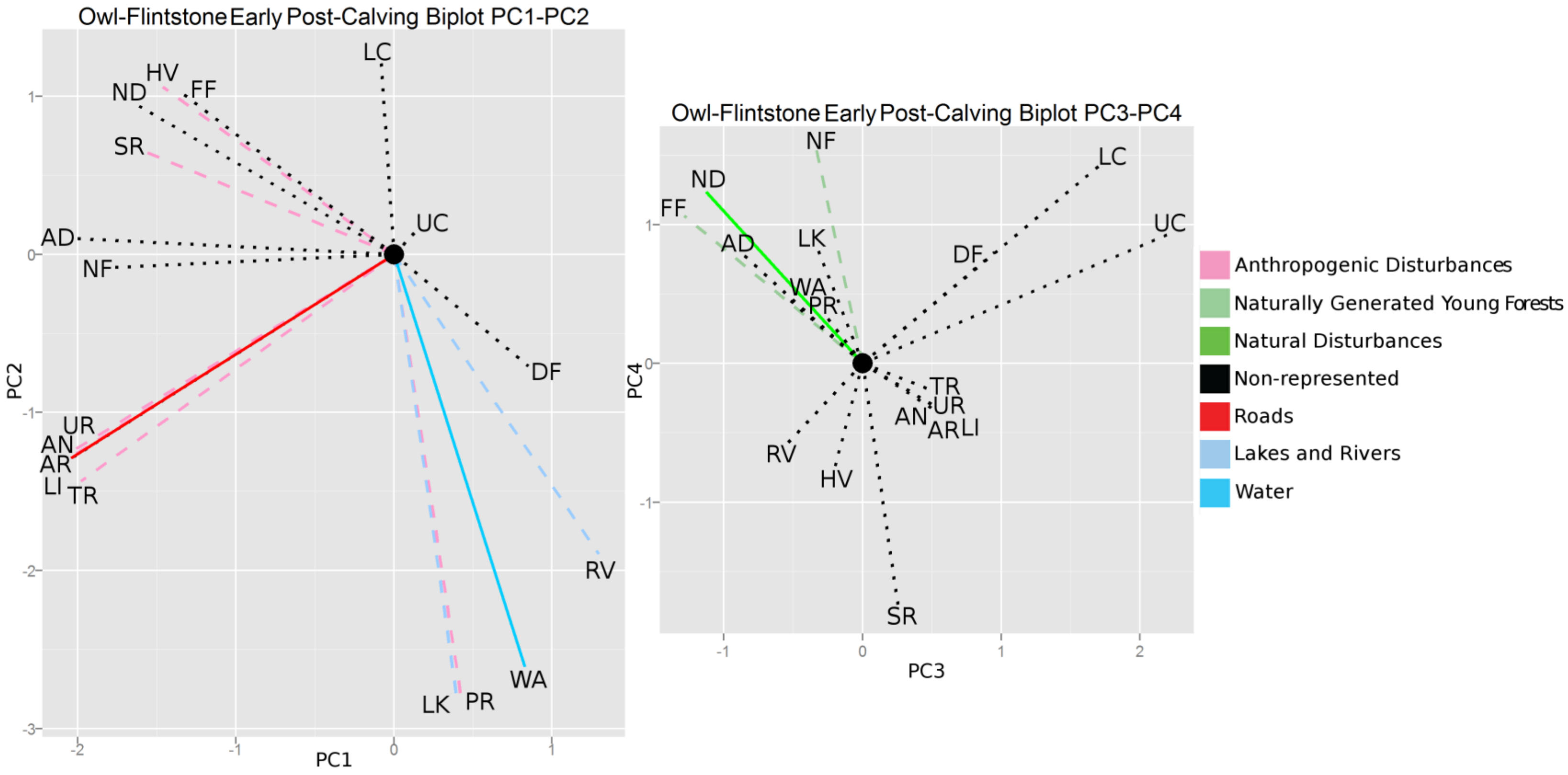


Figure B13. PCA biplot for the Early Post-Calving period in the Owl-Flintstone population. Solid lines denote variables used to represent a PC, dashed lines are variables that either are covariates or components of the representative variables. Dotted lines are unused variables. Two letter codes are provided in Figure 1A.

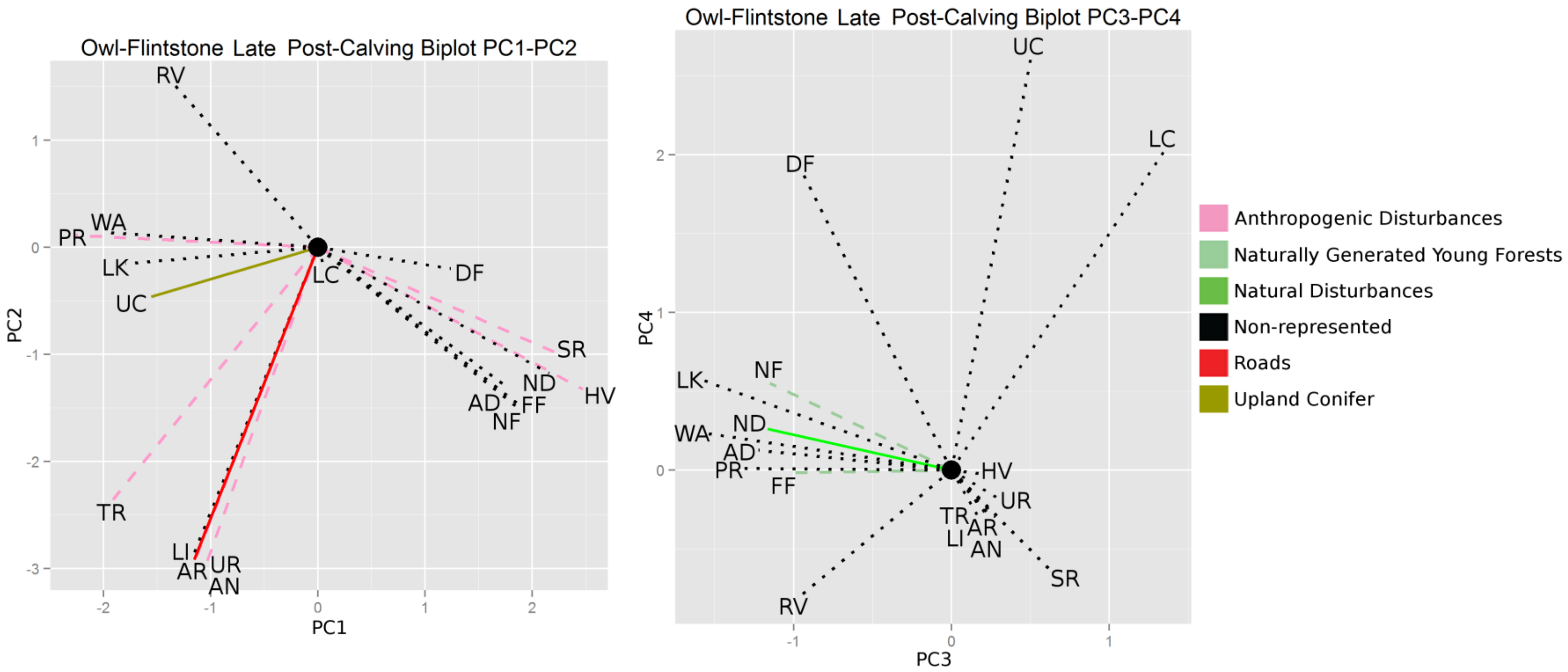


Figure B14. PCA biplot for the Late Post-Calving period in the Owl-Flintstone population. Solid lines denote variables used to represent a PC, dashed lines are variables that either are covariates or components of the representative variables. Dotted lines are unused variables. Two letter codes are provided in Figure 1A.

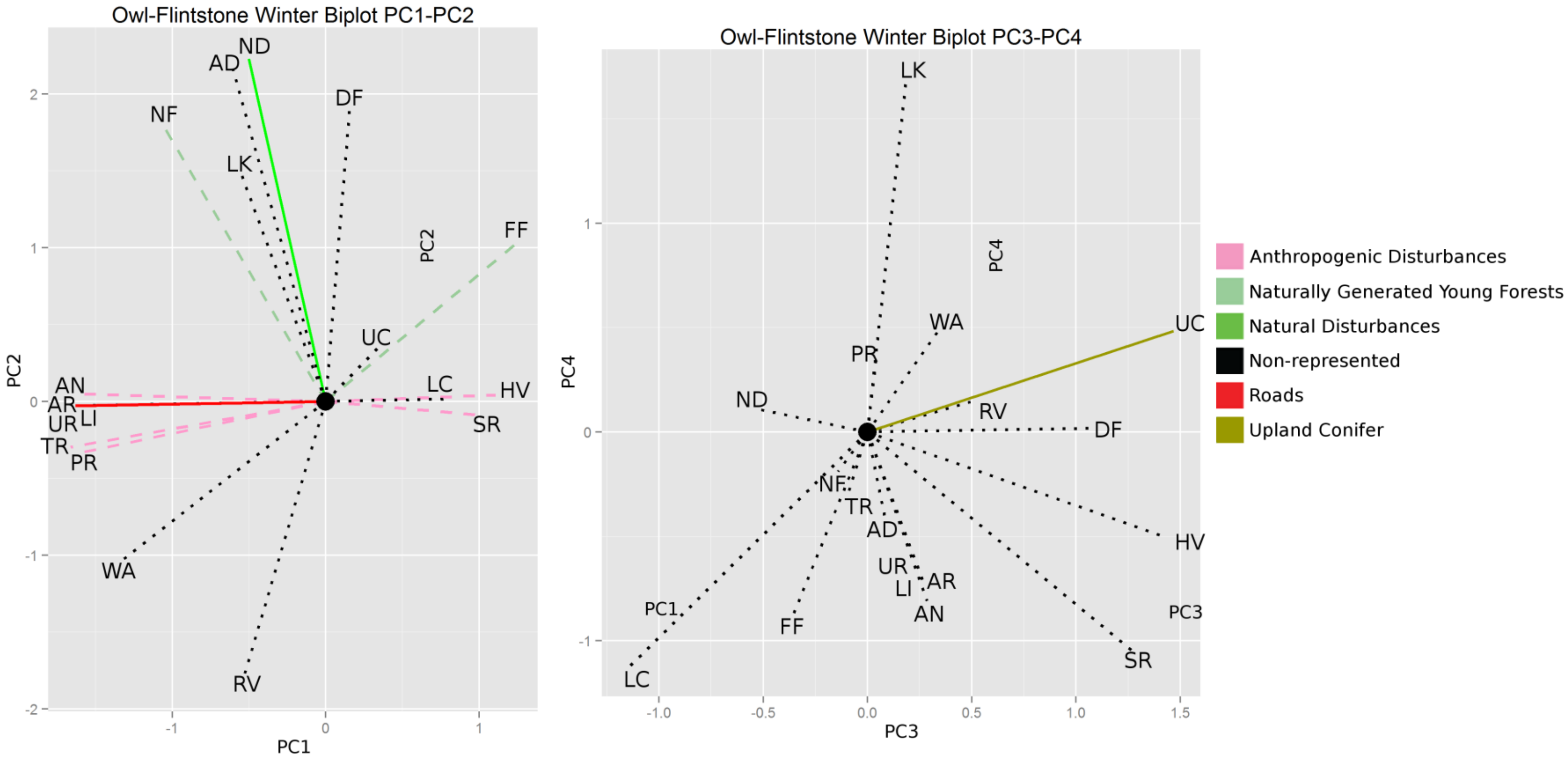


Figure B15. PCA biplot for the Winter period in the Owl-Flintstone population. Solid lines denote variables used to represent a PC, dashed lines are variables that either are covariates or components of the representative variables. Dotted lines are unused variables. Two letter codes are provided in Figure 1A.

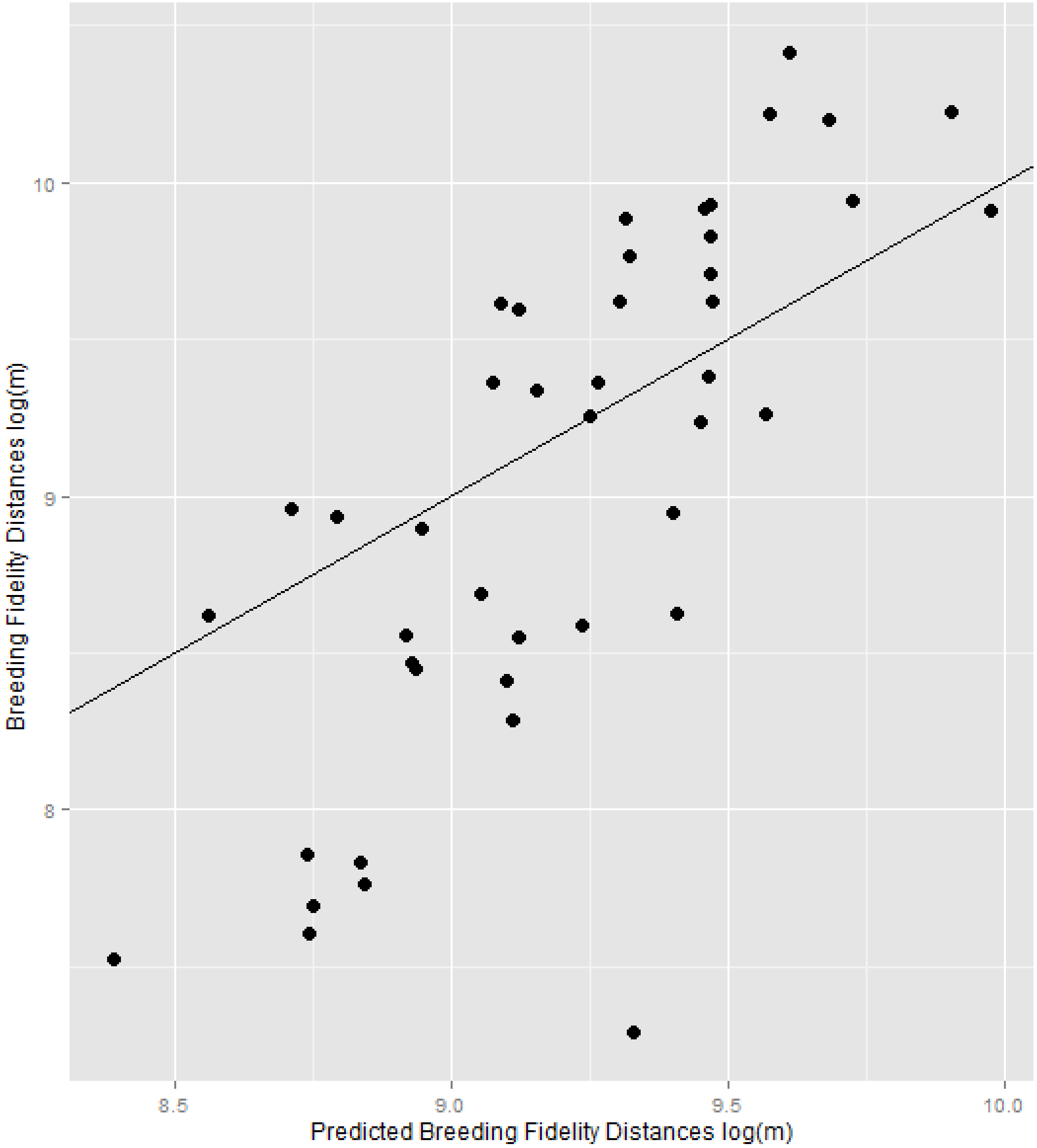


Figure B16. Predicted Breeding site fidelity distances (log[m]) from the final models compared against logged actual Breeding site fidelity scores (log[m]). The solid black line represents a perfect relationship.

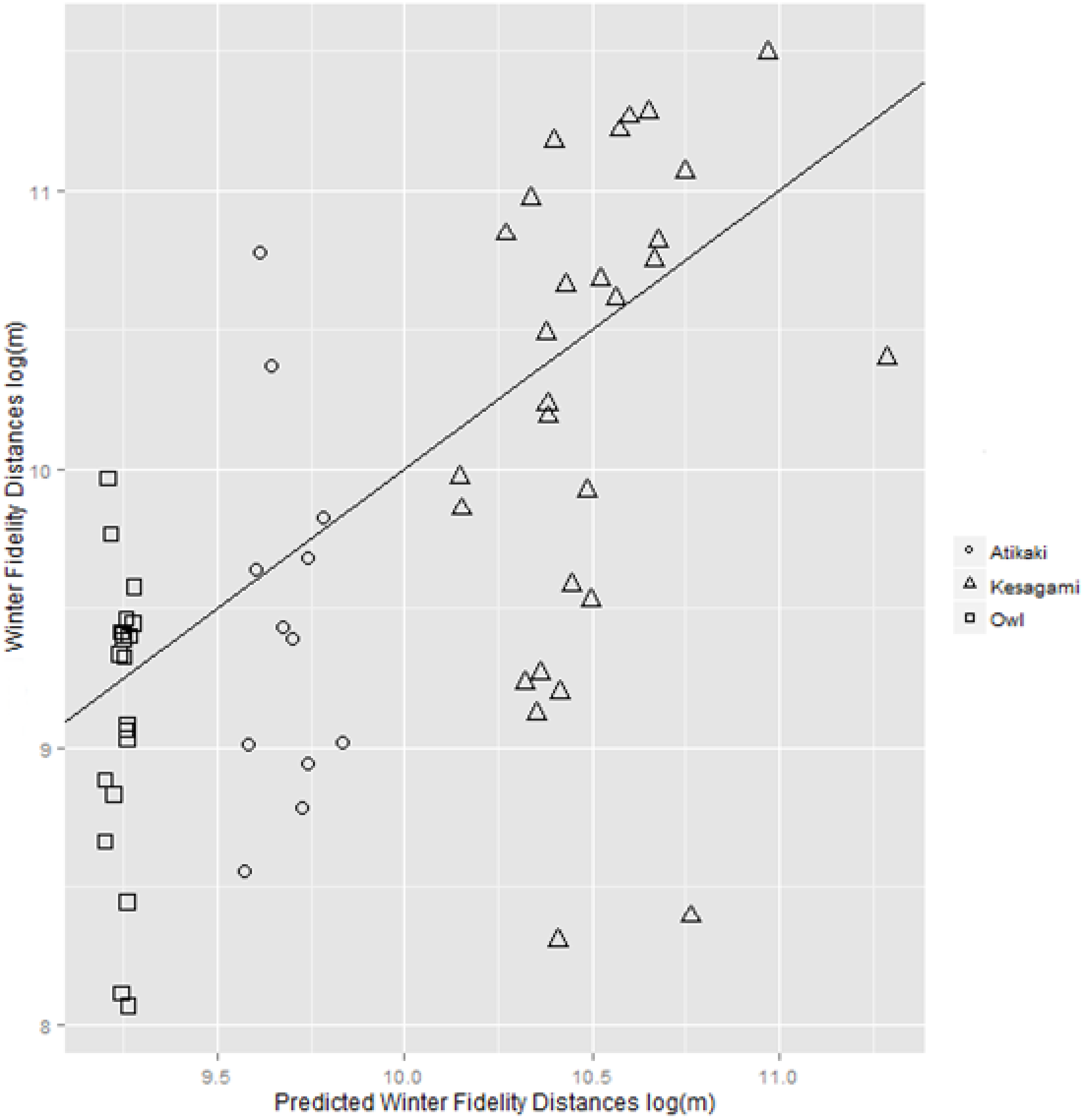


Figure B17. Predicted Winter site fidelity distances (log[m]) from the final models compared against logged actual Winter site fidelity scores (log[m]). The solid black line represents a perfect relationship.

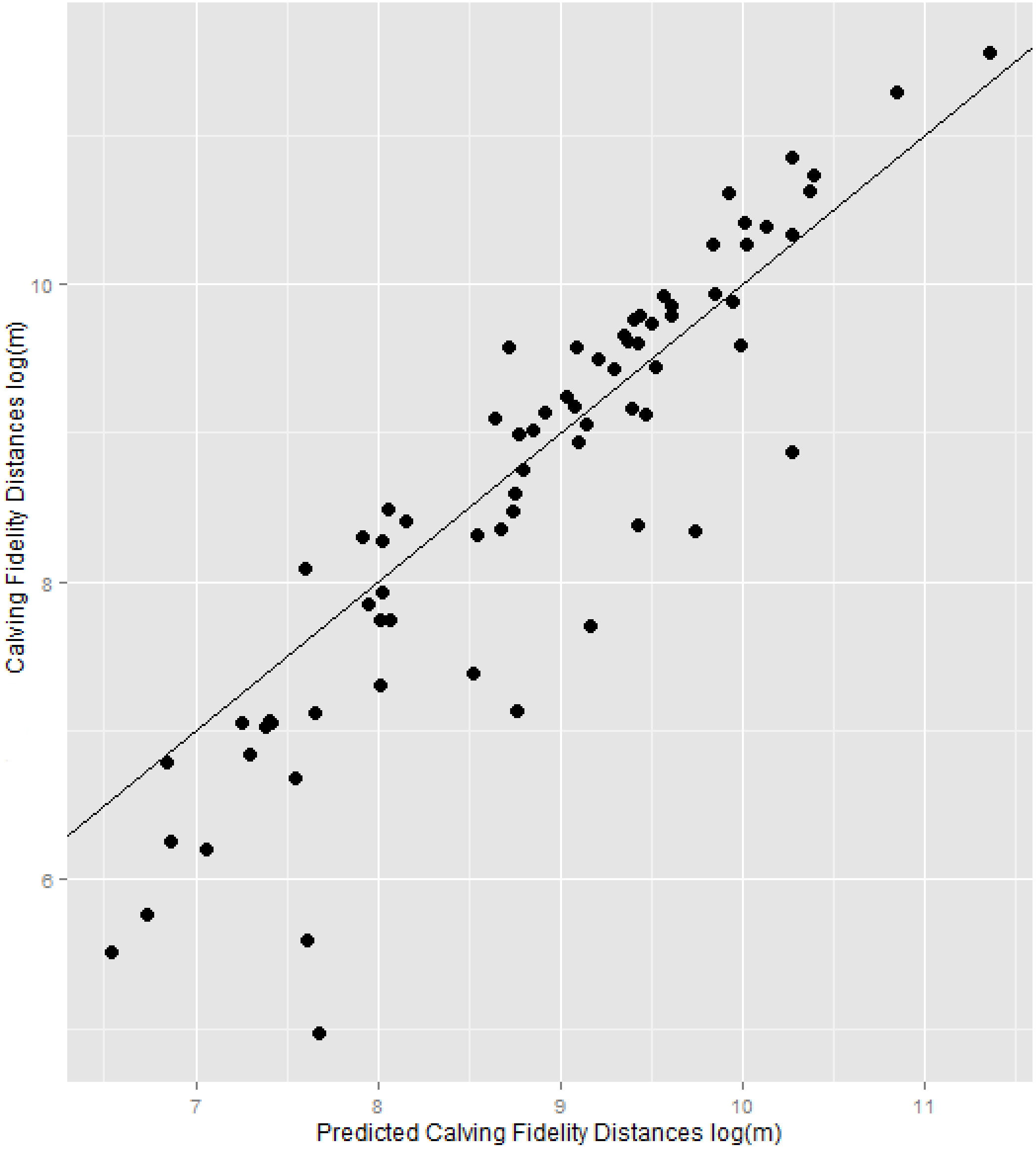


Figure B18. Predicted Calving site fidelity distances (log[m]) from the final models compared against logged actual Calving site fidelity scores (log[m]). The solid black line represents a perfect relationship.

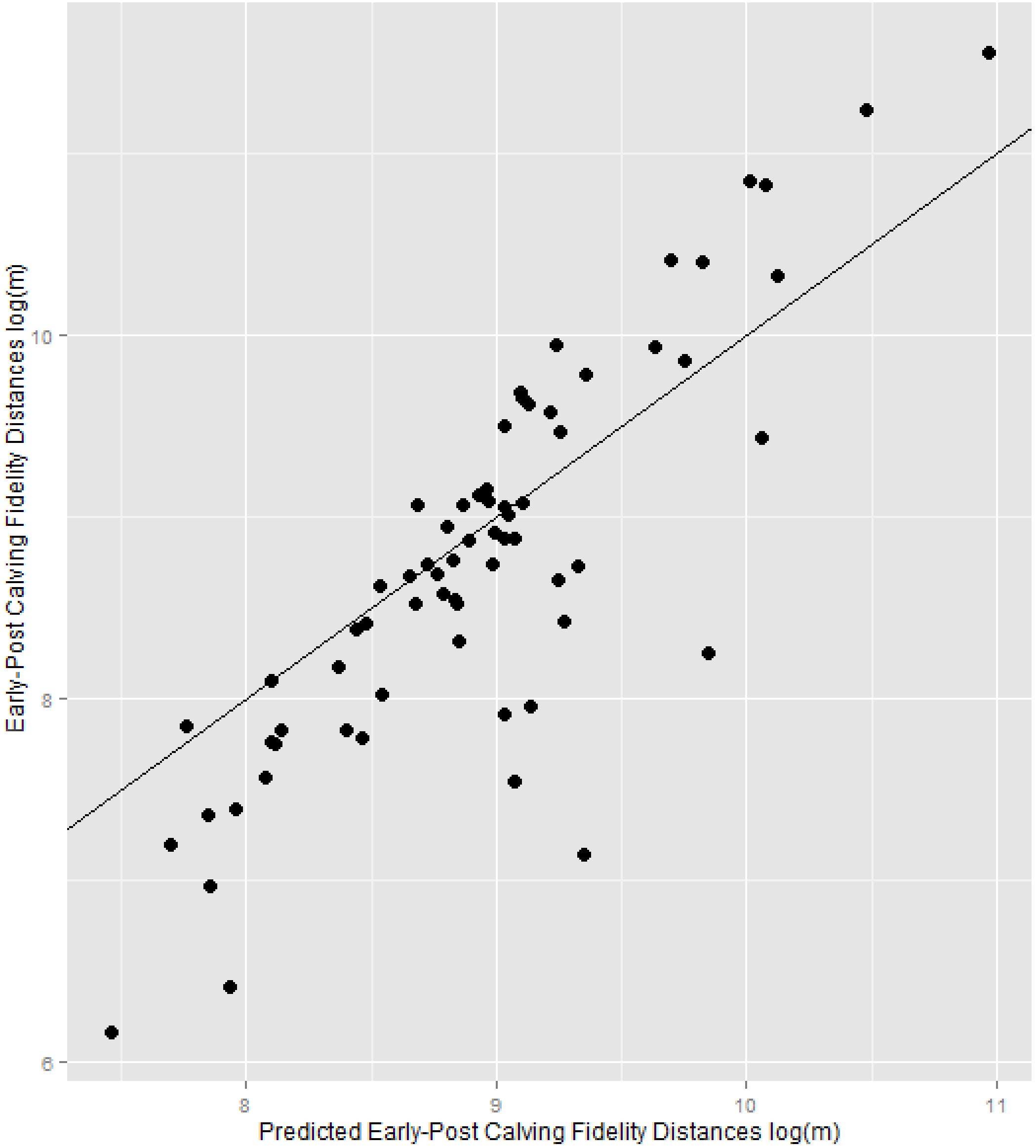


Figure B19. Predicted Early Post-Calving site fidelity distances (log[m]) from the final models compared against logged actual Early Post-Calving site fidelity scores (log[m]). The solid black line represents a perfect relationship.

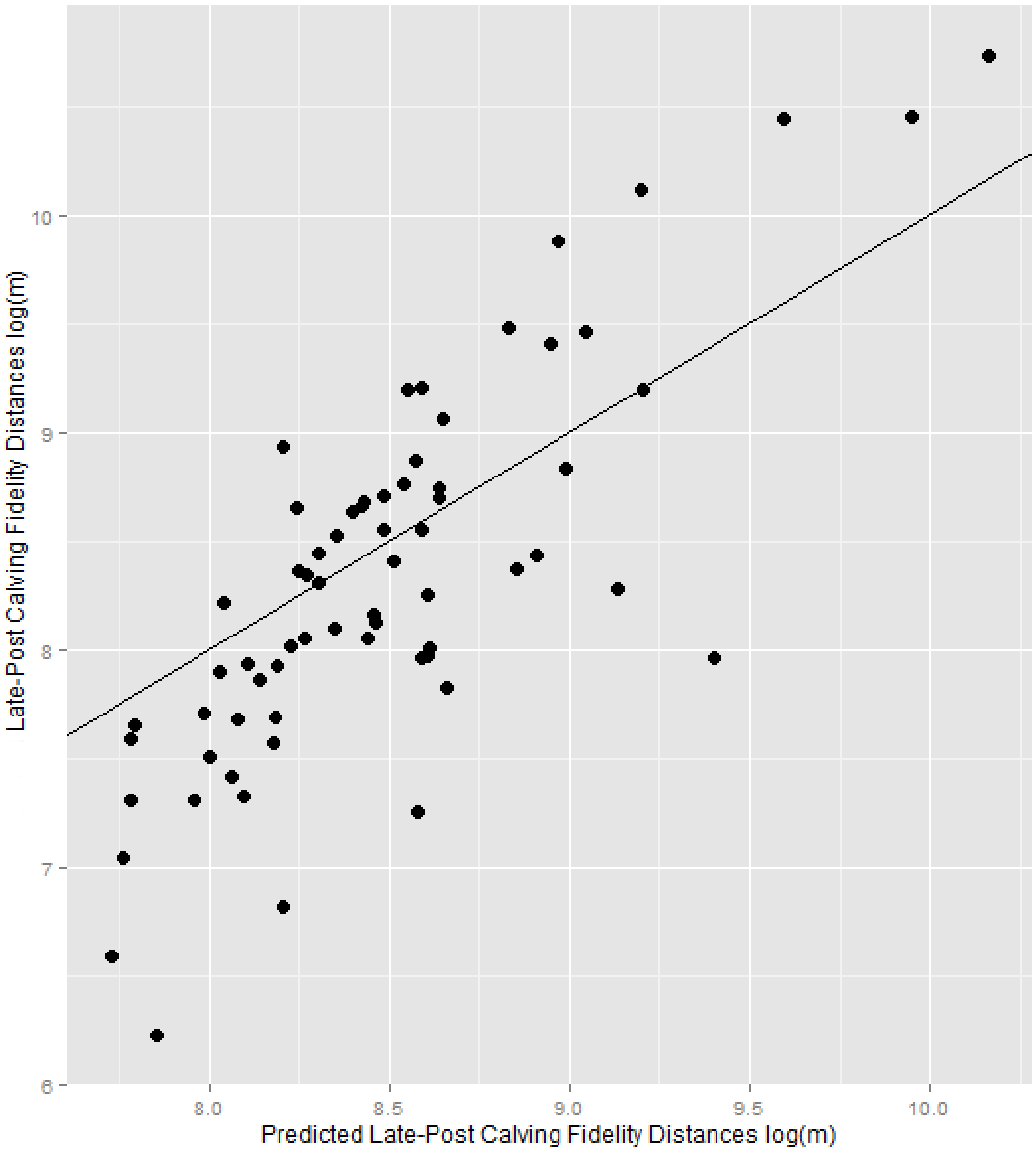


Figure B20. Predicted Late Post-Calving site fidelity distances (log[m]) from the final models compared against logged actual Late Post-Calving site fidelity scores (log[m]). The solid black line represents a perfect relationship.

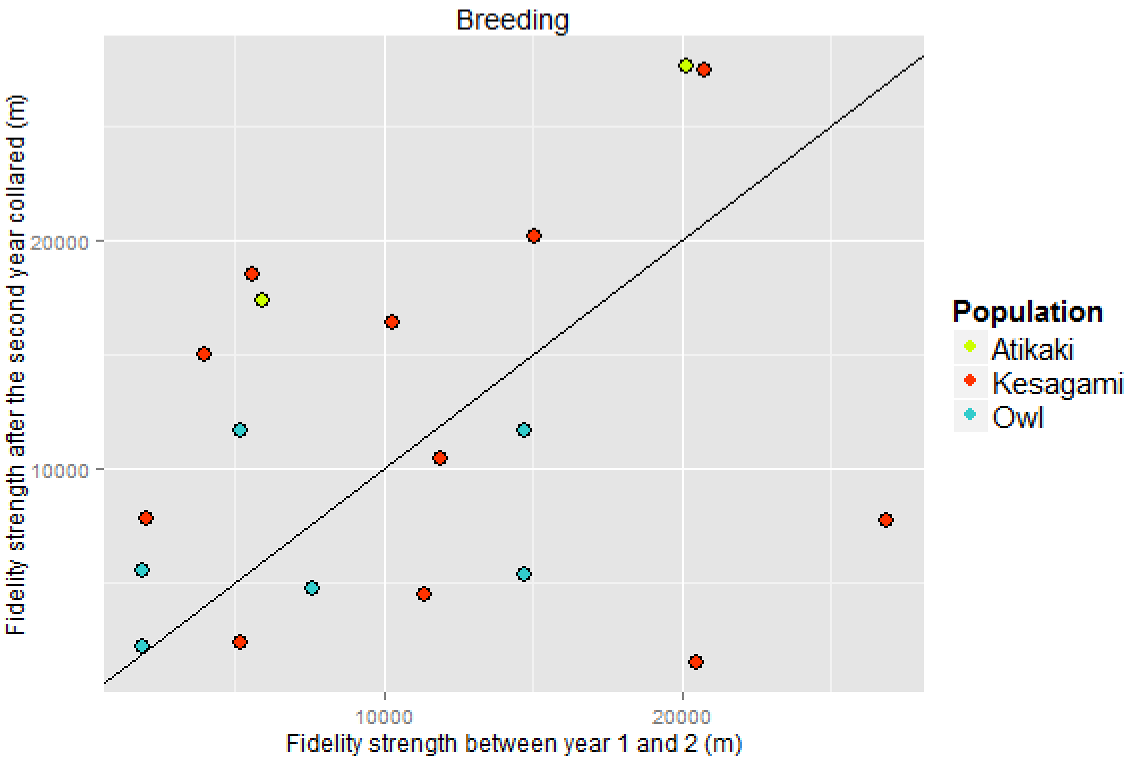


Figure B21. Breeding fidelity strength (m) in the first year collared against fidelity strength in all other years.

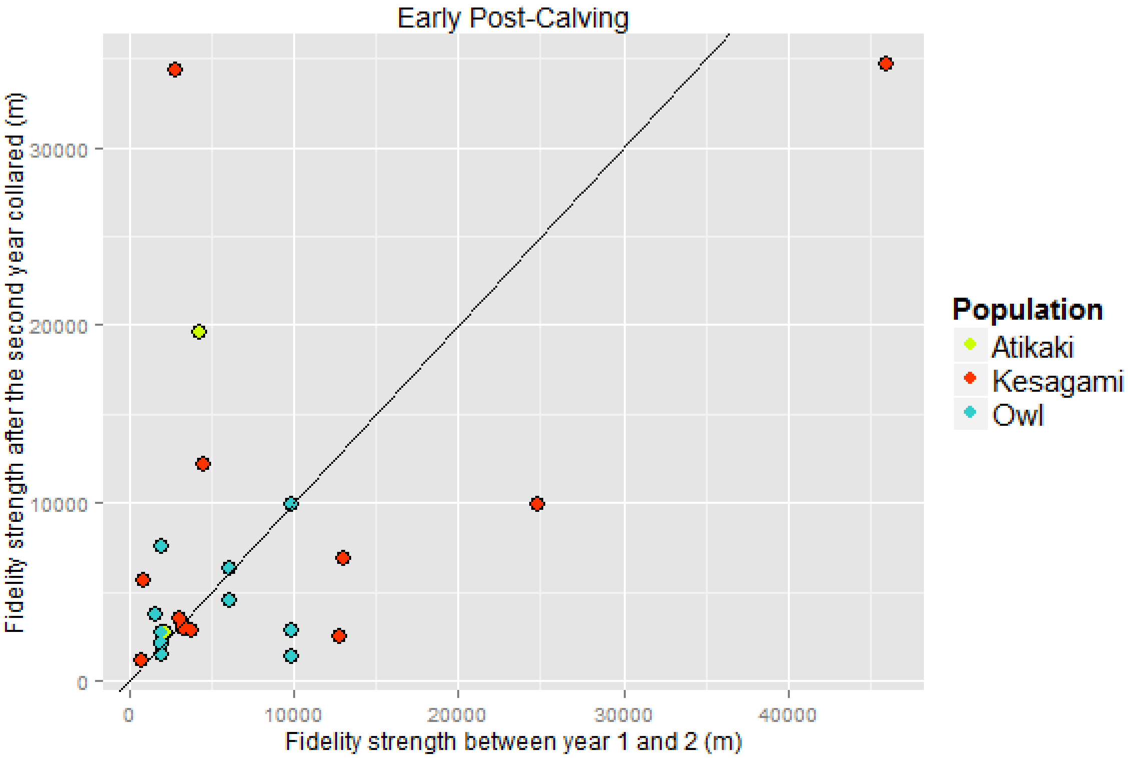


Figure B22. Late Post-Calving fidelity strength (m) in the first year collared against fidelity strength in all other years.

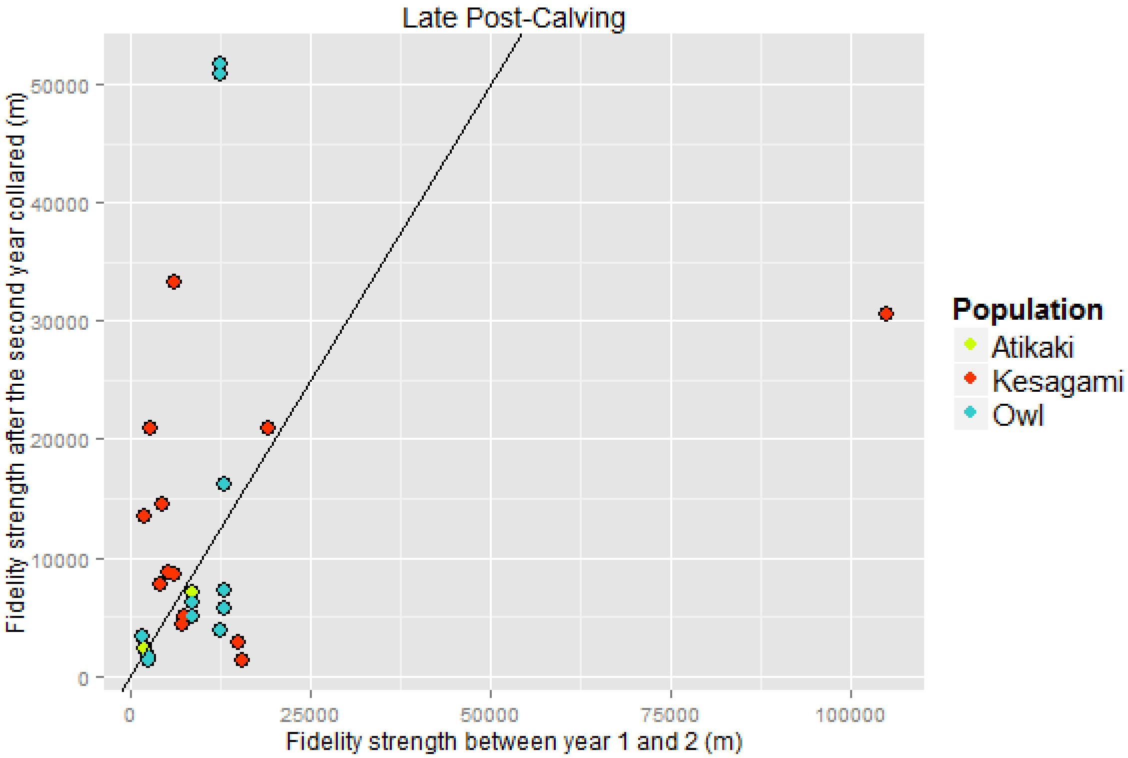


Figure B23. Early Post-Calving fidelity strength (m) in the first year collared against fidelity strength in all other years.

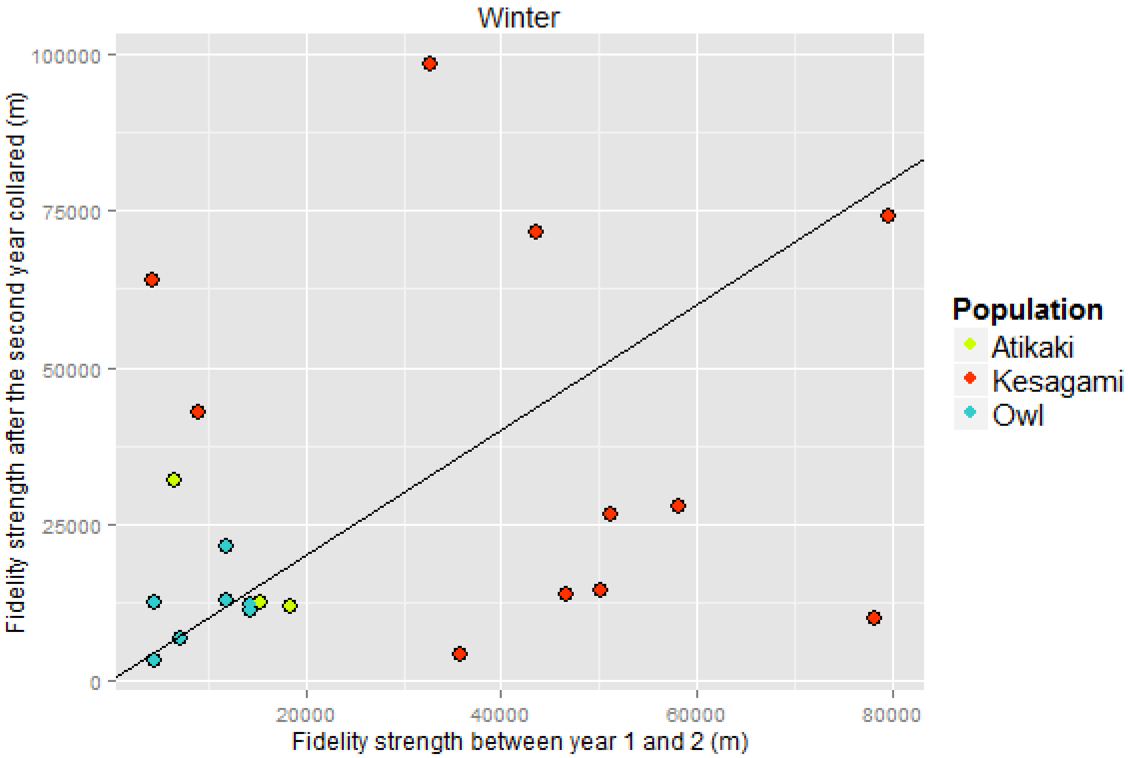


Figure B24. Winter fidelity strength (m) in the first year collared against fidelity strength in all other years.

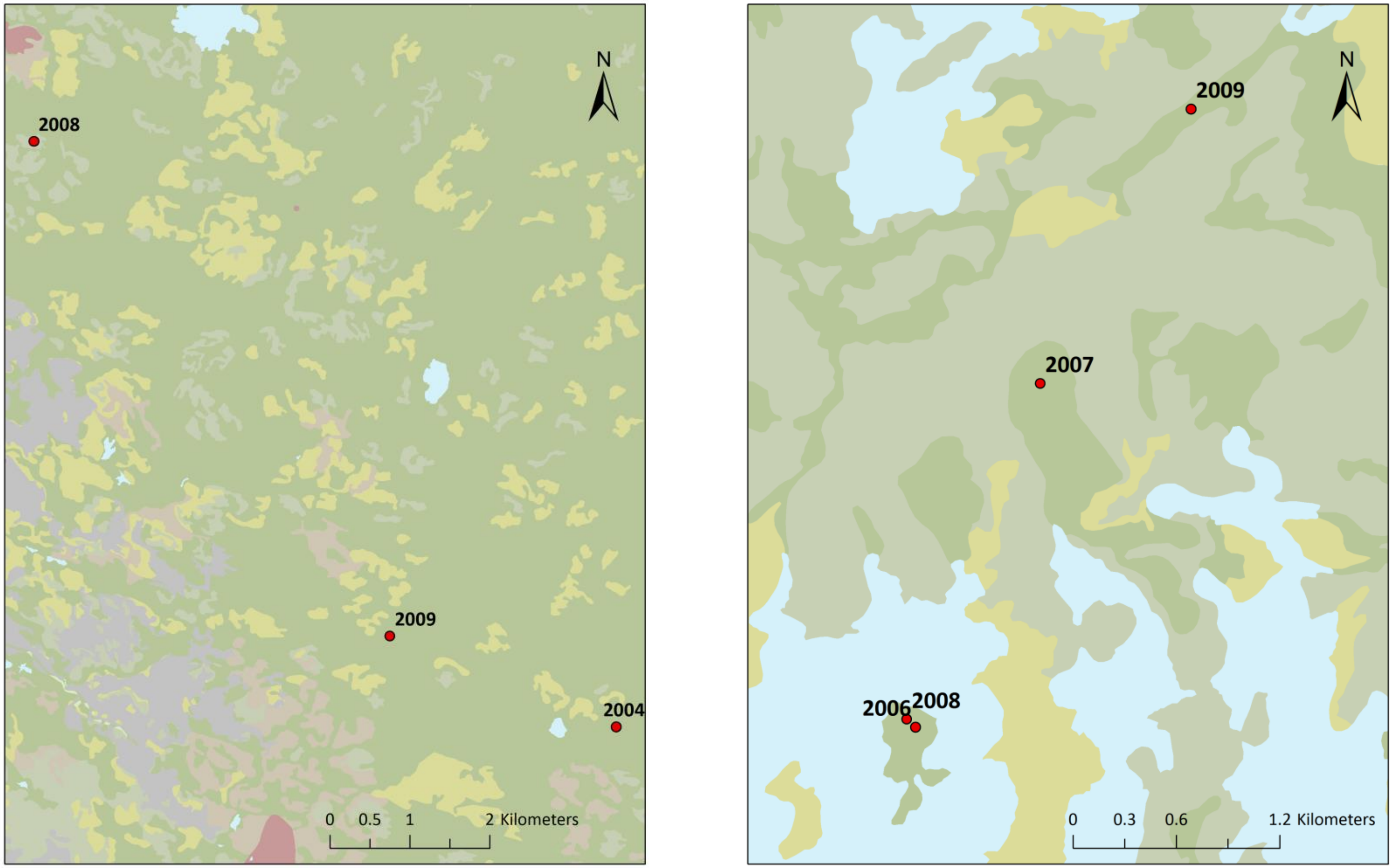


Figure B25. Animals from the Atikaki-Berens population showing non-consecutive calving site fidelity. Red points are averaged locations for the Calving season.

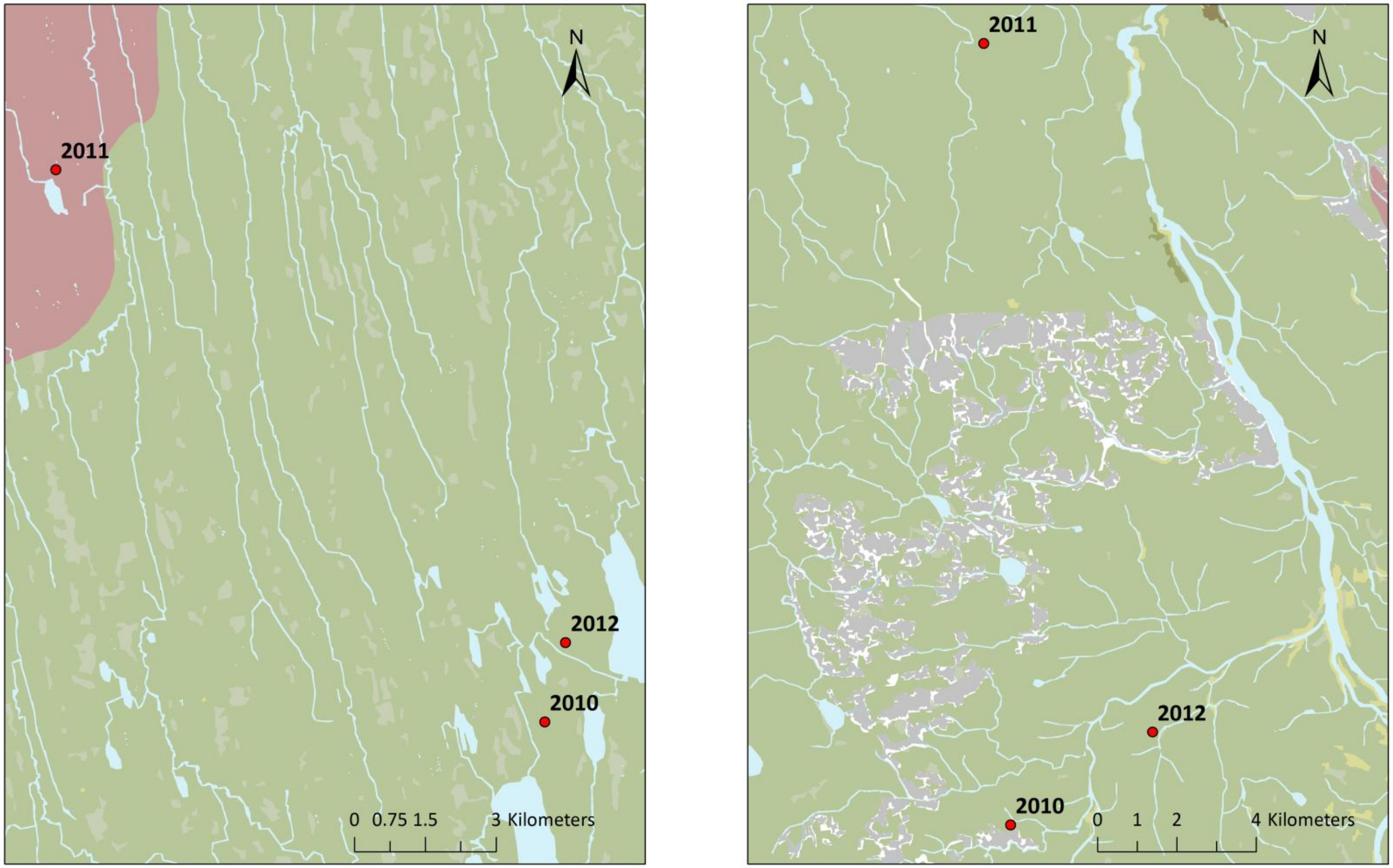


Figure B26. Animals from the Kesagami population showing non-consecutive calving site fidelity. Red points are averaged locations for the Calving season.

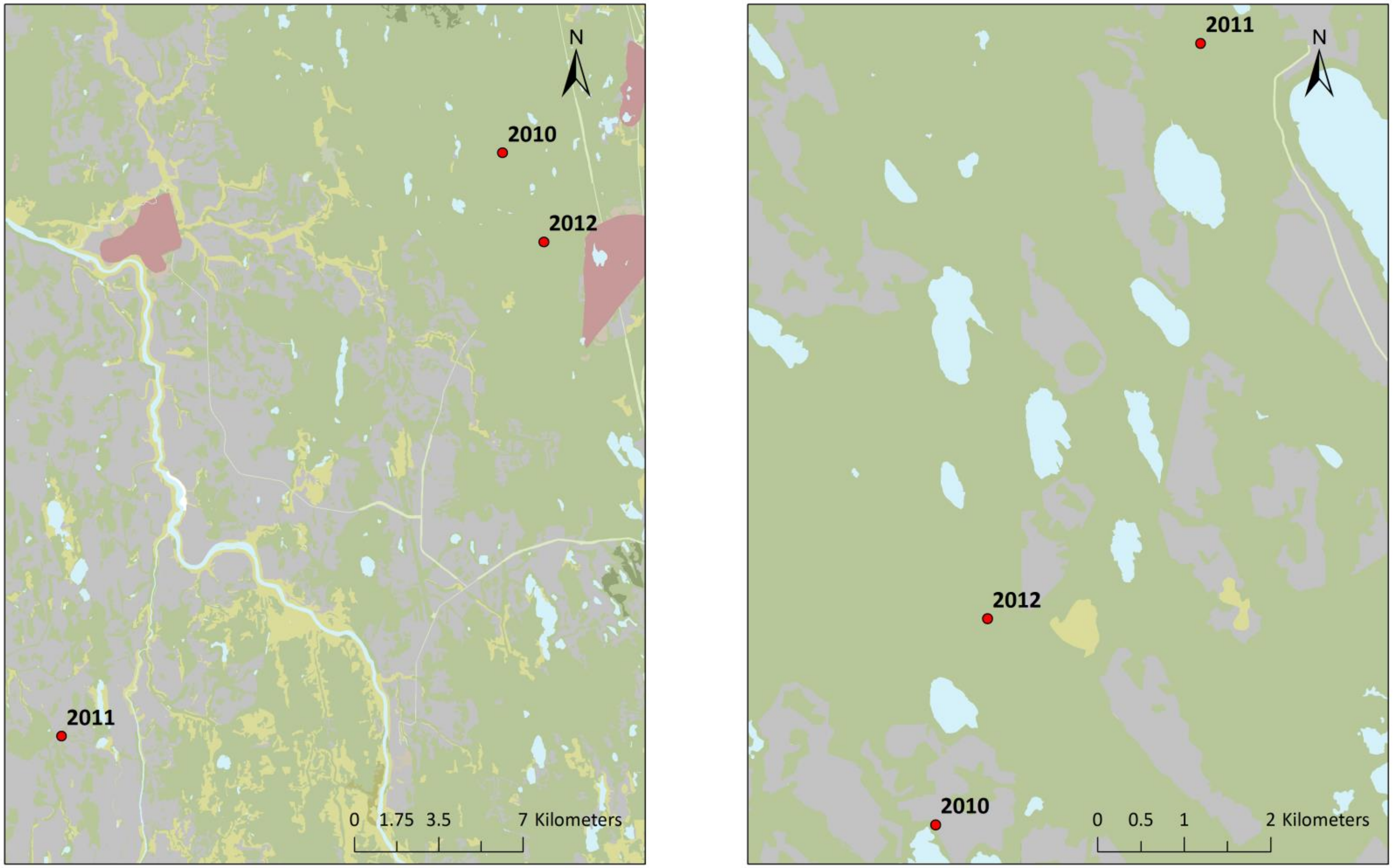


Figure B27. Animals from the Kesagami population showing non-consecutive calving site fidelity. Red points are averaged locations for the Calving season.

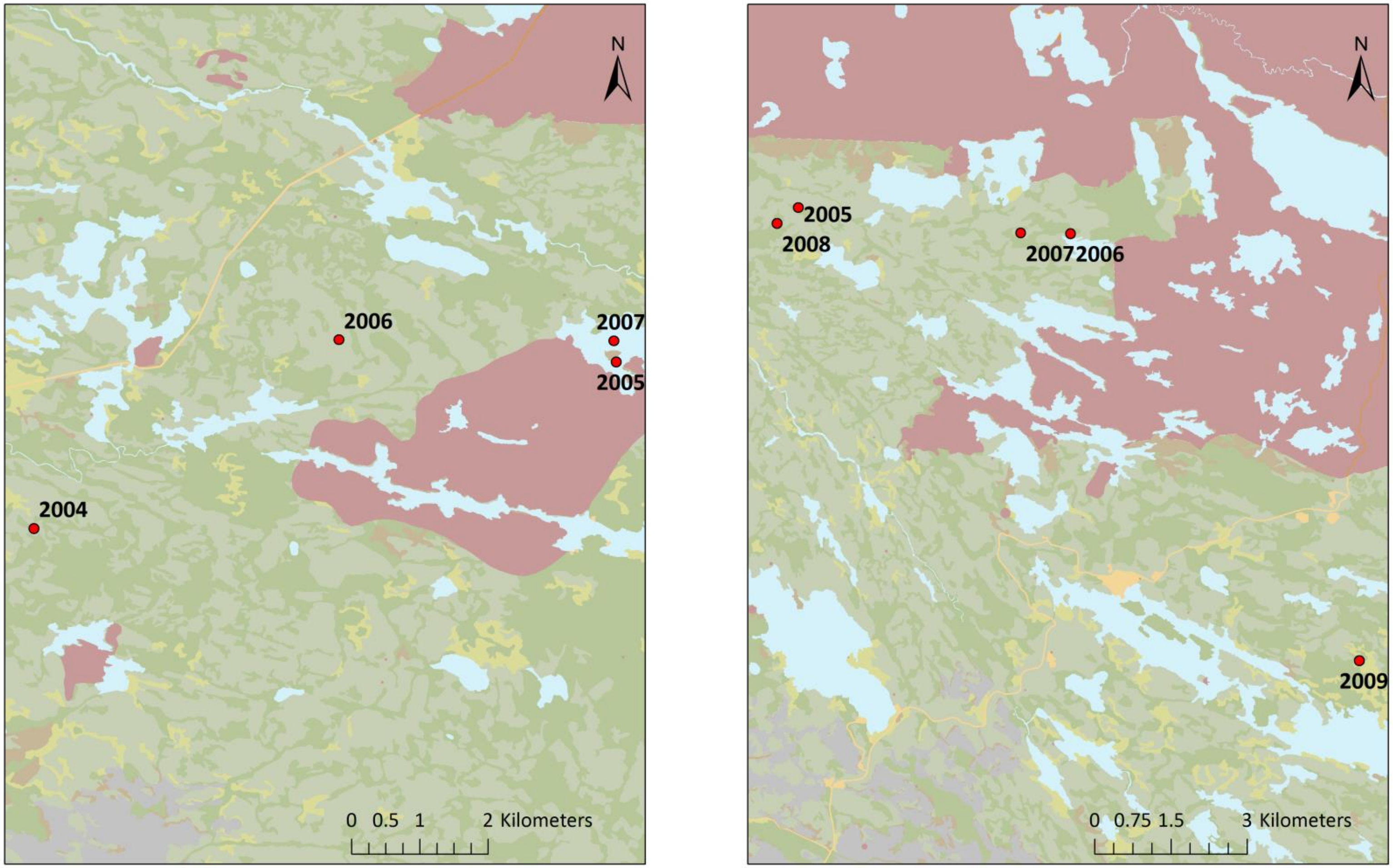


Figure B28. Animals from the Owl-Flintstone population showing non-consecutive calving site fidelity. Red points are averaged locations for the Calving season.

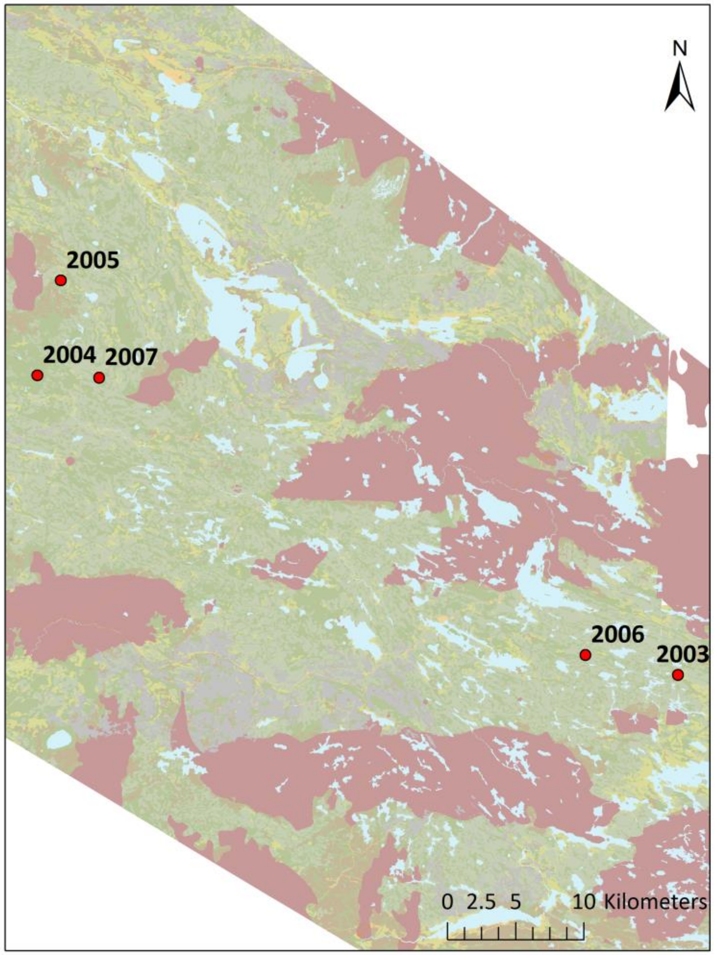


Figure B29. Animal from the Owl-Flintstone population showing non-consecutive calving site fidelity. Red points are averaged locations for the Calving season.

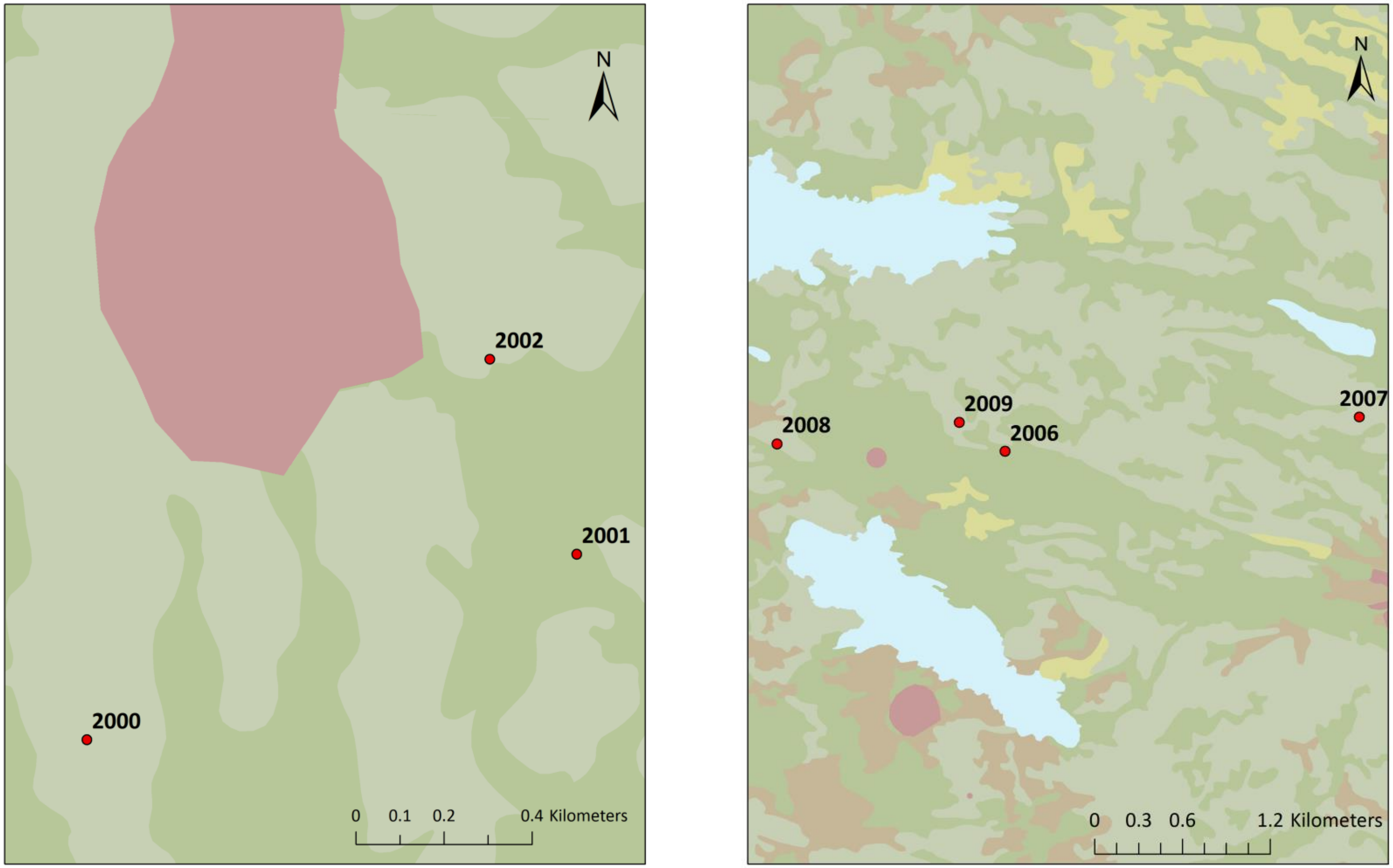


Figure B30. Animals from the Owl-Flintstone population showing strong calving site fidelity. Red points are averaged locations for the Calving season.

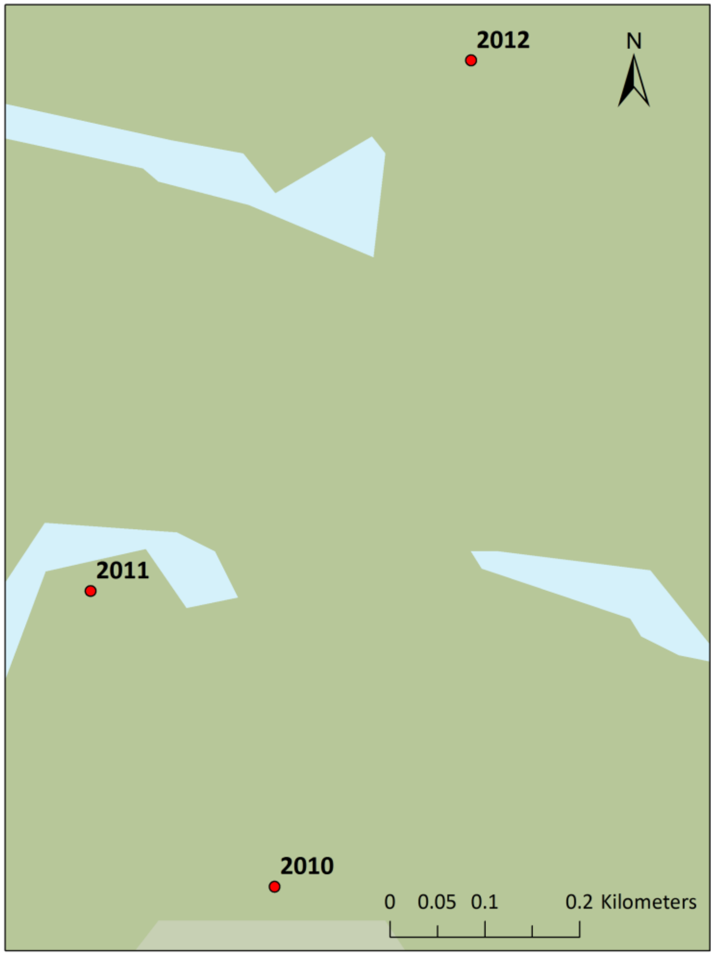


Figure B31. Animal from the Kesagami population showing strong calving site fidelity. Red points are averaged locations for the Calving season.

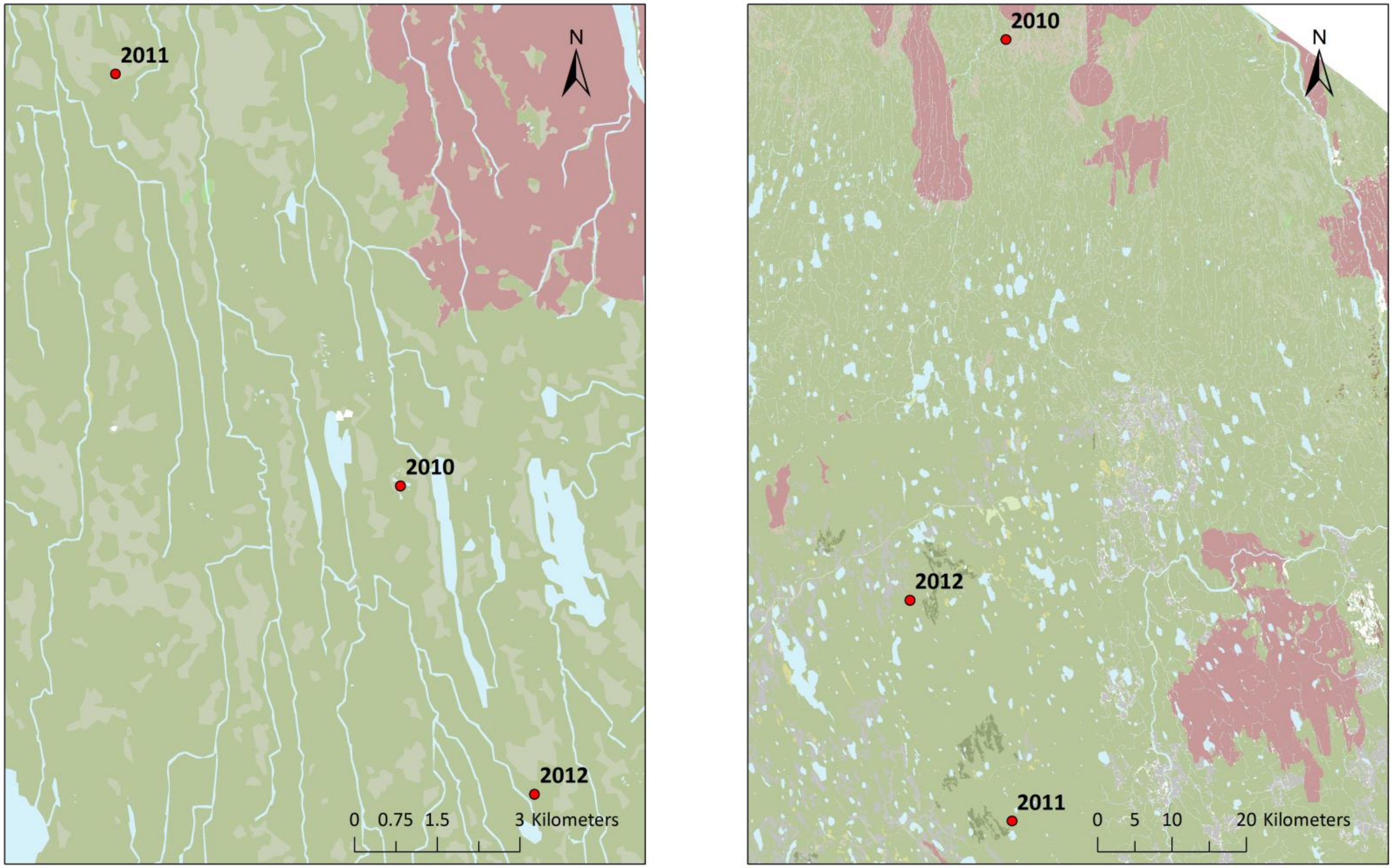


Figure B32. Animals from the Kesagami population showing weak calving site fidelity. Red points are averaged locations for the Calving season.

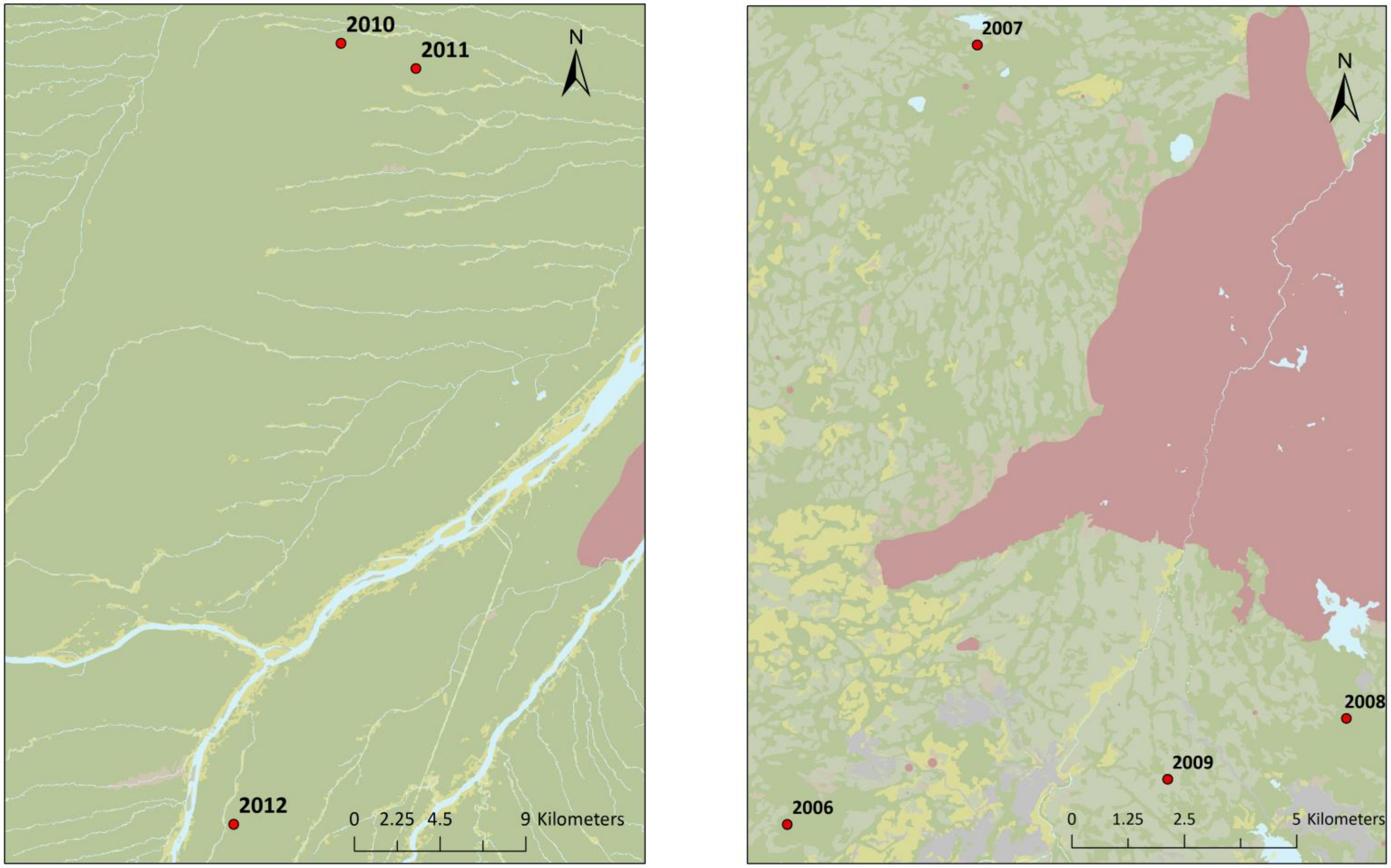


Figure B33. Animals from the Kesagami (left) and Atikaki-Berens (right) populations showing mixed strength calving site fidelity. Red points are averaged locations for the Calving season.

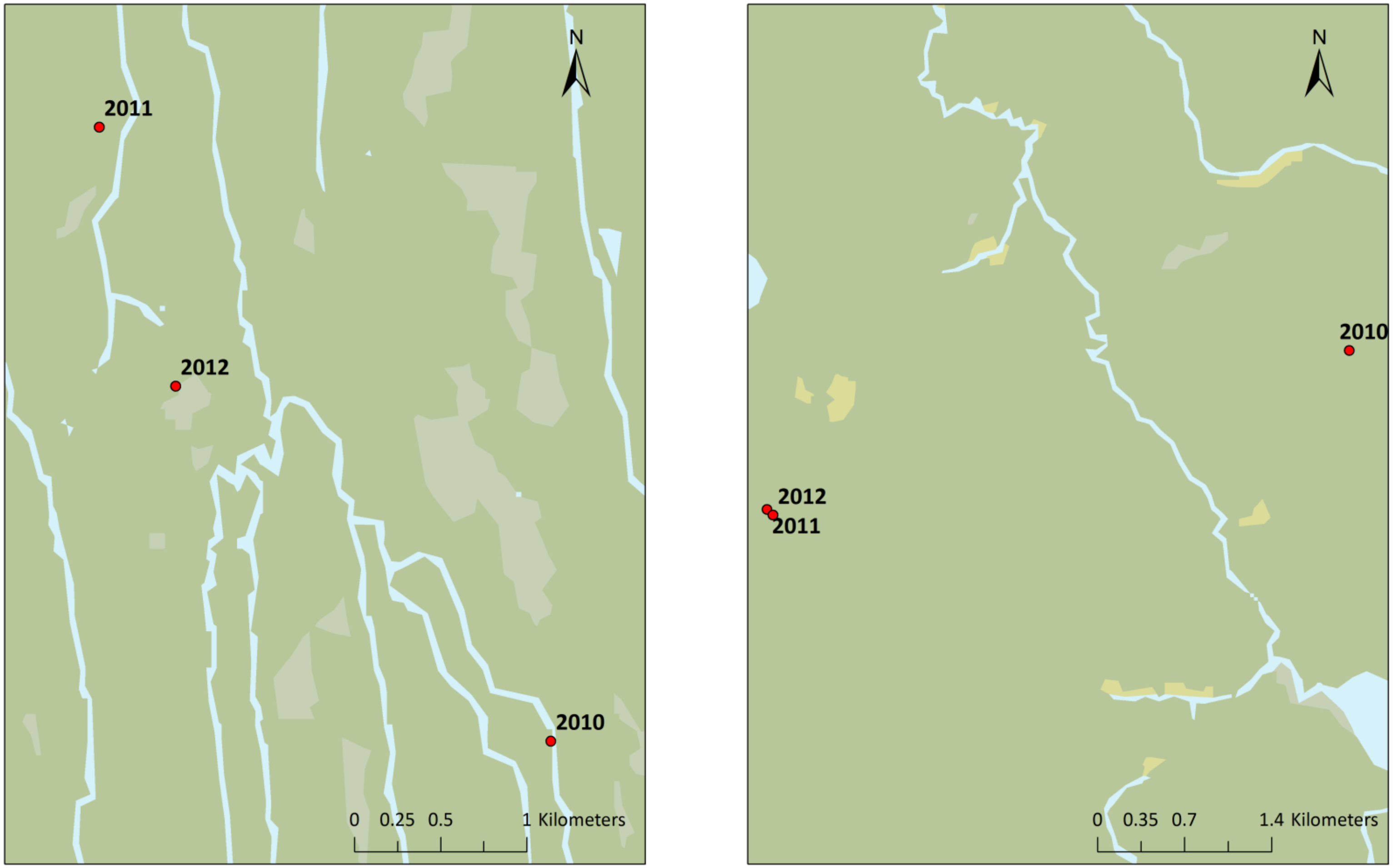


Figure B34. Animals from the Kesagami population showing mixed strength calving site fidelity. Red points are averaged locations for the Calving season.

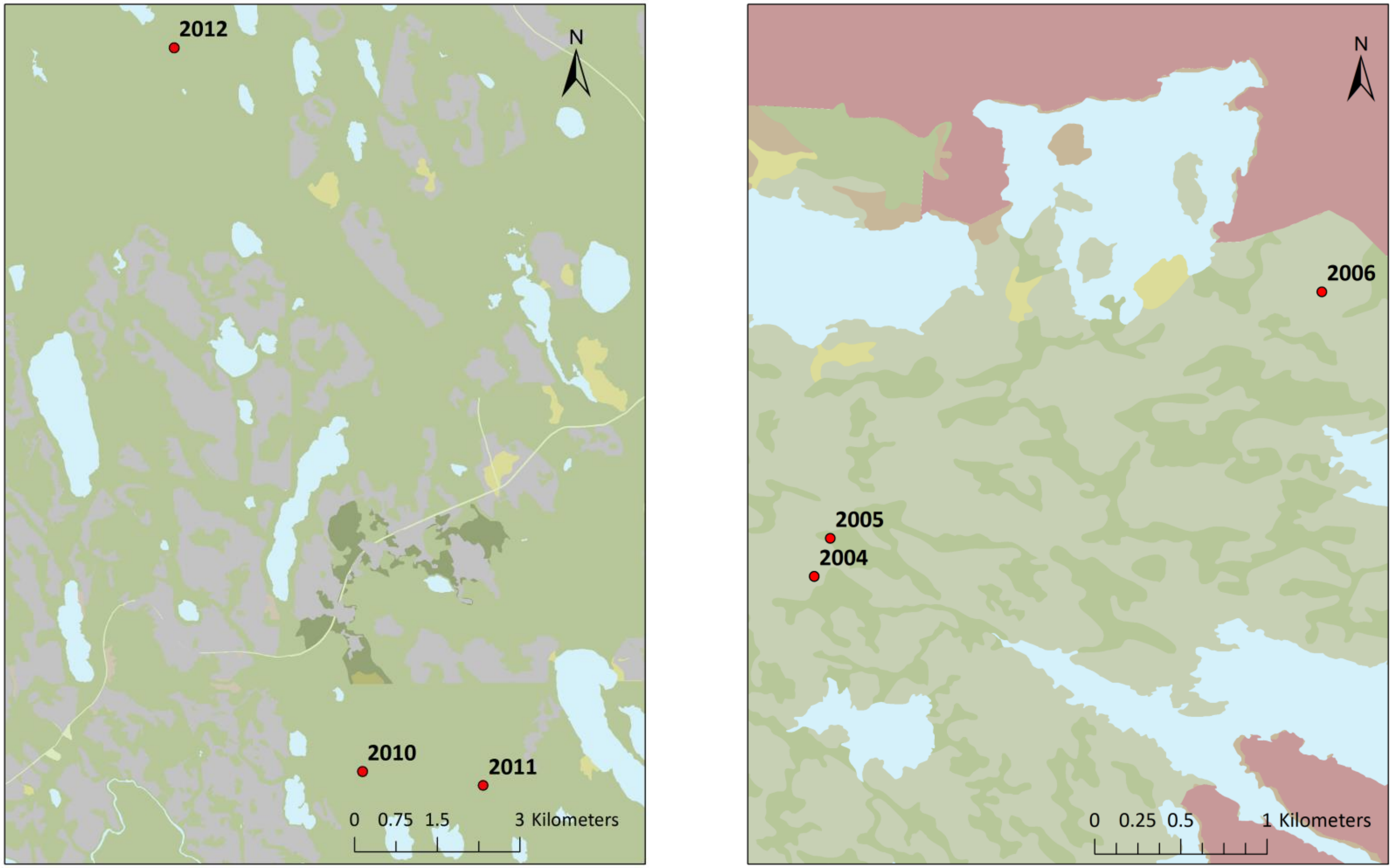


Figure B35. Animals from the Kesagami (left) and Owl-Flintstone (right) populations showing mixed strength calving site fidelity. Red points are averaged locations for the Calving season.

Table B1. Disturbance classification schemes for each province.

	Province-specific equations		
Disturbance	**Manitoba**	**Ontario**	**Quebec**
Railroad	Linear feature defined as railroad	Linear feature defined as railroad	Linear feature defined as railroad
Power-lines	Linear feature defined as power-line	Linear feature defined as power-line	Linear feature defined as power-line
Roads – Primary	Labelled as Highway	Labelled as Primary, Arterial, Expressway or paved	Labelled as Acces ressources or Nationale road or paved
Roads – Secondary	Labelled as A1, A2, A3A, A3B, ACom, AMin, AMun, APark, AProv	Labelled as Secondary, Local/Street, Ramp,	Labelled as *Chemin carrosable non-pavée, Route d'access aux resources non pavé, Rue non-pavee*, Non-MTQ road
Roads – Tertiary	Labelled as A4, A4M, AT, D2, D3A, D3B, D4, DMin, R3A, R3B, R4, RMin, RT	All roads remaining not classified as Primary or Secondary	Labelled as Chemin non carrosable, Voie de communication abandonee, RFA, RFNA, AUCUN
Communities	Labelled as Northern Community, Compact Rural Community or Community	Labelled as Disperse Rural Community, Town, Dispersed Rural Community, or Compact Rural Community.	Labelled as Disperse Rural Community, Town, Dispersed Rural Community, or Compact Rural Community.
Mines	Labelled as Operating	Labelled as Past Producing Mine with Reserves	Labelled as Mine active or Mise en valeur
Forest harvest	Area of forest harvest in last 40 years. Forest harvest data obtained directly from government agency.	Forest with age < 40 year unattributed to fire or natural disturbance	All stands with a listed origin < 40 years that are not contained in the Forest fire or natural disturbance category.
Forest fire	Area of forest fires in last 40 years. Forest fire data obtained directly from government agency.	Area of forest fires in last 40 years. Forest fire data obtained directly from government agency.	Area of forest fires in last 40 years. Forest fire data obtained directly from government agency, additionally any stand whose origin was attributed as Brûlis total
Natural disturbance	Forest with age < 40 years unattributed to fire or harvest	Forest with age < 40 years attributed to natural disturbances other than fire.	Forest with age < 40 years with origin attributed to Chablis total and Épidémie grave

Table B2. Combined disturbance types used within the Principal Component Analysis and their component variables.

Combined Disturbance Features	**Original Disturbance Features**
Unfinished roads	Secondary Roads Tertiary Roads
All roads	Primary Roads Unfinished roads
Linear disturbance features	Railroad Power-lines All Roads
All natural disturbances	Natural, non-fire disturbances Natural, fire disturbances
All anthropogenic disturbances	Community Mines Linear disturbance features Harvest
All disturbances	All anthropogenic disturbances All natural disturbances

Appendix C

Figure C1. Biotic understory composition for each habitat classification for the original field classes for the FRI maps. The # column represents the number of transects that contain that species and the Area column represents the average number of 10cm x 10cm squares it represented (1% of 1 plot) in plots where it was present. Bold values identify the 10 most common species across all classes.

	Black Spruce		Young Deciduous		Young Conifer		Upland Conifer		Treed Muskeg		Rock		Open Muskeg		Mixed Conifer		Marsh-Shrub		Lowland Conifer		Jack Pine		Deciduous		Clearcut	
	#	Area	#	Area	#	Area	#	Area	#	Area	#	Area	#	Area	#	Area	#	Area	#	Area	#	Area	#	Area	#	Area
Abies balsamifera	8	36	10	50	11	83	3	55	2	1	1	27	0	0	28	58	10	32	5	76	6	19	21	37	4	14
Acer spicatum	0	0	4	7	4	6	0	0	0	0	0	0	0	0	10	42	4	38	2	10	0	0	10	51	0	0
Achillea millefolium	0	0	1	4	0	0	0	0	0	0	1	2	0	0	0	0	0	0	0	0	0	0	1	7	1	1
Actaea rubra	0	0	0	0	0	0	0	0	1	1	0	0	0	0	1	2	0	0	0	0	0	0	2	4	0	0
Alnus spp. (shrub)	18	38	7	41	14	25	1	5	12	23	2	12	7	10	15	26	19	62	8	43	7	10	15	22	5	24
Amelanchier alnifolia	3	1	3	3	6	3	0	0	1	26	3	8	0	0	8	4	0	0	0	0	6	3	8	6	1	10
Andromeda polifolia	2	7	0	0	2	9	0	0	12	18	1	53	13	17	0	0	1	35	6	11	0	0	0	0	0	0
Anemone spp.	1	1	1	1	0	0	0	0	1	1	0	0	0	0	1	1	1	1	1	2	0	0	4	1	0	0
Apocynum androsaemifolium	0	0	0	0	1	8	0	0	0	0	0	0	0	0	1	5	0	0	0	0	1	16	0	0	0	0
Aralia nudicaulis	3	5	10	11	8	17	2	1	0	0	4	14	0	0	20	12	5	8	0	0	4	13	16	33	5	22
Arctostaphylos uva-ursi	0	0	3	5	2	6	0	0	0	0	5	23	0	0	1	35	1	1	0	0	2	29	3	7	8	26
Aster spp.	4	5	11	48	8	8	0	0	2	8	2	1	3	1	7	33	9	23	0	0	4	6	12	29	1	1
Betula papyrifera	0	0	5	9	6	3	1	3	1	1	3	7	2	9	8	5	2	6	1	1	2	6	10	3	0	0
Betula spp. (shrub)	1	24	0	0	5	30	0	0	5	16	1	10	9	14	0	0	6	21	6	16	1	7	0	0	0	0
Calla palustris	0	0	1	2	0	0	0	0	1	8	0	0	2	7	0	0	1	4	0	0	0	0	1	3	0	0
Caltha palustris	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	4	5	26	1	17	0	0	1	1	0	0
Chamaedaphne calyculata	20	31	2	44	12	60	0	0	23	123	3	266	16	159	2	23	7	46	6	40	1	1	0	0	1	5
Chimaphila umbellata	0	0	0	0	3	1	2	1	0	0	1	2	0	0	4	1	1	1	0	0	2	1	1	1	6	2

Cirsium arvense	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	10	0	0	0	0	0	0	1	8
Cladina mitis	10	17	7	3	15	7	2	20	10	15	14	84	3	25	5	26	0	0	1	2	14	35	7	15	9	31
Cladina rangiferina	18	36	6	4	19	31	2	6	15	15	12	39	3	34	7	7	5	5	1	6	16	35	8	6	10	37
Cladina stellaris	3	18	0	0	4	1	0	0	1	6	3	72	0	0	1	2	0	0	0	0	8	15	0	0	2	7
Cladonia	12	1	4	3	15	2	3	18	6	2	6	9	3	2	12	1	4	1	2	1	5	2	9	4	1	1
Cladonia borealis	1	2	1	1	7	1	0	0	3	2	0	0	1	1	2	1	1	2	0	0	1	3	1	1	0	0
Cladonia spp.	13	1	4	1	6	2	0	0	4	2	0	0	2	1	13	1	4	1	2	2	5	1	6	4	0	0
Climacium dendroides	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	6	3	6	0	0	0	0	1	30	0	0
Coptis trifolia	8	6	4	3	13	5	1	2	6	4	0	0	1	4	20	6	9	6	4	6	1	1	10	4	1	3
Cornus canadensis	20	20	13	20	20	26	2	13	4	10	2	3	1	10	28	18	12	10	6	10	12	9	25	19	11	16
Cornus stolonifera	1	3	3	11	3	5	0	0	0	0	1	1	0	0	0	0	7	28	0	0	0	0	4	15	1	1
Corydalis sempervirens	0	0	0	0	1	1	0	0	0	0	5	1	0	0	0	0	0	0	0	0	0	0	1	3	1	14
Corylus cornuta	1	1	5	27	2	29	1	12	0	0	0	0	0	0	3	7	3	34	0	0	0	0	8	28	0	0
Crust Lichen	0	0	3	40	2	10	1	4	4	4	11	51	0	0	2	1	0	0	0	0	4	57	3	10	5	11
Diervilla lonicera	1	10	7	24	7	46	2	24	0	0	4	13	0	0	8	22	4	16	0	0	6	17	14	49	4	57
Drosera rotundifolia	0	0	0	0	0	0	0	0	4	1	0	0	3	1	0	0	0	0	1	1	0	0	0	0	0	0
Dryopteris austriaca	0	0	0	0	0	0	0	0	0	0	0	0	0	0	3	3	0	0	0	0	0	0	1	2	0	0
Epilobium angustifolium	6	3	7	10	8	11	0	0	3	4	3	7	0	0	2	15	11	11	1	1	1	1	11	7	6	12
Equisetaceae	19	15	9	4	20	15	1	1	8	9	2	1	4	10	19	7	12	9	4	2	4	4	15	9	3	6
Feather Moss (Hypnales)	29	273	13	84	29	186	3	131	21	118	13	170	6	15	30	250	18	39	7	103	21	417	25	55	13	175
Feather Soil Moss	26	14	10	24	21	14	3	31	11	6	12	15	3	11	29	19	17	16	4	19	19	52	21	25	12	22
Fern	3	4	7	13	6	18	1	5	0	0	3	1	0	0	16	14	11	20	2	9	1	5	13	30	3	11
Fragaria virginiana	4	7	6	8	5	1	2	5	2	1	2	9	1	7	3	1	10	7	2	2	3	2	13	10	10	4
Galium spp.	0	0	8	2	4	1	0	0	1	10	0	0	1	1	8	2	10	4	1	3	0	0	14	3	2	1
Gaultheria hispidula	23	39	3	3	18	20	0	0	12	9	0	0	3	1	15	21	9	11	4	16	10	22	0	0	2	15

Gaultheria procumbens	1	9	0	0	1	14	0	0	1	13	1	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Geocaulon lividum	5	2	0	0	1	1	0	0	7	1	0	0	0	0	1	1	0	0	1	5	1	1	3	2	2	12
Geum spp.	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	1	10	0	0
Impatiens capensis	1	1	2	1	0	0	0	0	0	0	0	0	0	0	2	7	6	14	0	0	1	1	2	13	1	1
Juniperus communis	0	0	1	7	0	0	0	0	1	50	10	75	1	14	0	0	1	3	1	24	1	90	3	12	5	42
Kalmia angustifolia	15	28	1	15	14	23	0	0	13	23	0	0	6	11	3	12	1	1	0	0	8	118	2	9	0	0
Kalmia polifolia	10	3	0	0	7	3	0	0	15	13	0	0	8	17	1	1	2	1	0	0	1	2	0	0	1	1
Larix laricina	1	3	1	21	5	7	0	0	7	22	0	0	4	13	1	15	6	8	9	7	0	0	1	4	0	0
Lathyrus spp.	1	2	3	4	1	6	1	6	0	0	1	1	0	0	0	0	1	33	0	0	1	2	5	10	4	11
Leaf Lichens	2	3	3	5	2	3	2	7	1	3	10	27	0	0	2	18	2	3	0	0	2	9	5	3	4	4
Ledum groenlandicum	28	131	3	16	21	84	2	14	25	105	6	63	9	45	12	48	12	54	10	99	12	39	3	15	5	55
Liliaceae spp.	25	18	11	26	24	15	3	7	21	23	10	15	8	29	24	24	20	25	8	13	14	9	23	35	11	15
Linnaea borealis	14	7	7	6	17	8	2	3	3	4	2	1	2	4	24	6	9	5	5	10	13	8	17	5	7	9
Lonicera oblongifolia	0	0	1	12	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lonicera villosa	6	3	0	0	1	1	0	0	1	11	0	0	1	2	1	2	4	11	2	1	0	0	1	16	0	0
Lycopodium complanatum	0	0	0	0	2	8	1	3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	21
Lycopodium obscurum	2	21	4	3	5	9	2	9	0	0	3	1	0	0	6	17	1	5	0	0	1	1	11	10	1	12
Lycopodium spp	13	33	7	19	9	19	3	7	3	78	2	49	0	0	21	52	3	14	2	19	1	2	12	61	5	18
Matted Rock Moss	0	0	1	1	3	3	1	1	1	1	8	5	0	0	0	0	0	0	1	1	3	12	2	5	4	6
Matted Tree Moss	9	11	6	35	11	7	2	14	1	18	4	38	3	3	18	8	9	11	2	4	8	6	19	8	8	4
Mentha arvensis	0	0	0	0	0	0	0	0	1	11	0	0	1	5	0	0	2	7	0	0	0	0	0	0	1	3
Menyanthes trifoliata	0	0	0	0	0	0	0	0	1	31	0	0	2	29	0	0	0	0	0	0	0	0	0	0	0	0
Mitella nuda	5	2	6	11	9	6	0	0	1	6	0	0	0	0	14	9	13	7	5	7	2	4	14	14	2	3
Myrica gale	0	0	0	0	2	22	0	0	3	57	2	23	8	58	1	5	1	127	0	0	0	0	0	0	0	0
Oxalis montana	0	0	1	3	0	0	0	0	0	0	0	0	0	0	2	17	0	0	0	0	0	0	2	20	0	0

Persicaria amphibia	0	0	0	0	0	0	0	0	0	0	0	0	1	7	0	0	0	0	0	0	0	0	0	0	0	0
Petasites palmatus	8	7	6	3	8	3	1	2	2	2	2	2	0	0	10	3	6	6	0	0	2	7	7	4	2	9
Petasites sagittatus	1	1	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	2	14	0	0
Picea glauca	0	0	4	8	3	22	0	0	0	0	0	0	0	0	2	2	2	11	0	0	0	0	6	11	0	0
Picea mariana	28	119	7	34	25	79	1	1	23	127	11	56	8	136	21	46	12	43	6	64	18	63	11	22	11	61
Pinus banksiana	0	0	0	0	9	51	0	0	1	8	5	24	0	0	1	120	0	0	0	0	10	11	1	7	7	33
Polytrichum juniperinum	0	0	0	0	1	6	0	0	0	0	1	6	0	0	0	0	0	0	0	0	0	0	0	0	1	4
Polytrichum spp.	3	4	3	5	10	28	1	2	10	5	10	27	4	13	8	12	4	5	1	1	2	1	6	7	6	15
Populus spp.	1	1	7	9	4	5	1	5	2	2	0	0	0	0	5	3	7	4	0	0	1	5	17	7	6	5
Potentilla fruticosa	0	0	0	0	0	0	0	0	0	0	1	1	1	60	0	0	0	0	0	0	0	0	0	0	0	0
Potentilla palustris	1	2	2	5	1	1	0	0	1	1	2	7	6	9	1	1	3	4	0	0	1	1	0	0	0	0
Potentilla tridentata	0	0	0	0	2	1	0	0	0	0	2	7	0	0	0	0	0	0	0	0	4	2	1	9	4	13
Prunus pensylvanica	0	0	1	10	2	3	1	9	0	0	2	4	0	0	0	0	1	6	1	3	0	0	3	9	3	1
Prunus virginiana	0	0	1	4	0	0	0	0	0	0	0	0	0	0	2	8	0	0	0	0	1	13	3	11	0	0
Pyrolla spp.	2	4	6	6	4	3	1	1	0	0	0	0	0	0	11	7	3	2	1	3	2	1	11	5	1	4
Ranunculus lapponicus	0	0	0	0	1	1	0	0	0	0	0	0	0	0	1	8	0	0	1	3	0	0	0	0	0	0
Rhamnus alnifolia	0	0	1	11	0	0	0	0	0	0	0	0	1	50	2	45	2	41	0	0	0	0	0	0	1	7
Ribes americanum	1	4	1	6	1	4	0	0	1	1	1	1	0	0	2	2	1	2	0	0	0	0	2	24	0	0
Ribes glandulosum	1	3	7	11	2	2	0	0	0	0	1	3	0	0	8	10	9	6	4	3	2	7	16	18	3	16
Ribes lacustre	0	0	2	2	1	15	0	0	0	0	0	0	0	0	4	8	0	0	0	0	0	0	2	3	0	0
Ribes oxyacanthoides	0	0	3	1	2	2	0	0	1	8	0	0	0	0	2	1	5	5	1	4	0	0	7	10	0	0
Ribes triste	1	3	3	2	2	4	1	3	0	0	0	0	0	0	3	4	2	29	1	1	0	0	4	8	0	0
Rosa acicularis	7	7	7	7	8	4	2	6	2	3	1	1	1	8	10	4	6	7	1	1	7	4	16	7	6	3
Rubus chamaemorus	17	13	0	0	10	5	0	0	11	20	0	0	5	8	2	3	1	4	2	13	3	8	0	0	0	0
Rubus idaeus	6	7	9	24	14	14	1	18	2	3	2	1	1	1	12	20	13	27	2	9	2	2	12	38	7	11

Rubus pubescens	11	9	12	34	9	13	1	2	3	10	2	2	0	0	19	10	17	33	4	9	3	9	18	15	3	29
Salix spp. (shrub)	4	17	9	16	13	9	1	1	9	26	7	14	12	14	3	2	15	23	5	22	5	8	6	6	5	4
Sambucus spp.	0	0	1	20	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	7	0	0
Sanicula marilandica	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	1	4	0	0	0	0	2	1	0	0
Sarracenia purpurea	0	0	0	0	1	1	0	0	3	3	1	3	3	5	0	0	0	0	0	0	0	0	0	0	0	0
Senecio eremophilus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	43	0	0
Senecio spp	0	0	0	0	0	0	0	0	0	0	1	18	0	0	0	0	0	0	0	0	0	0	1	2	3	3
Soil Moss	12	11	2	64	7	13	0	0	3	14	0	0	0	0	8	11	5	16	2	9	6	6	8	24	2	6
Solidago canadensis	1	2	1	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Sorbus spp.	0	0	0	0	1	1	0	0	0	0	0	0	0	0	8	5	2	9	1	5	0	0	4	6	0	0
Sphagnum	23	307	6	30	19	283	1	60	26	519	8	249	14	433	15	127	15	184	10	276	8	62	8	50	6	27
Sterocaulon tomentosum	0	0	0	0	1	4	0	0	0	0	5	10	0	0	0	0	0	0	0	0	1	2	0	0	1	1
Taraxacum officinale	1	1	6	4	4	2	0	0	0	0	0	0	0	0	1	1	2	18	0	0	0	0	3	1	0	0
Taxus canadensis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	9	0	0	0	0	0	0	0	0	0	0
Tetraphis pellucida	0	0	2	18	3	6	0	0	1	6	0	0	0	0	6	15	3	10	1	3	0	0	10	2	3	1
Thuja occidentalis	1	3	1	10	0	0	0	0	0	0	0	0	0	0	0	0	1	98	0	0	0	0	0	0	0	0
Trientalis borealis	7	77	7	1	5	1	1	1	0	0	3	2	2	1	18	2	12	2	3	2	3	2	12	4	3	1
Trifolium spp.	0	0	3	12	1	4	0	0	0	0	0	0	0	0	0	0	1	3	0	0	0	0	0	0	1	5
Typha latifolia	0	0	0	0	1	5	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Unknown A	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	10	0	0	0	0	0	0	1	10
Unknown B	0	0	1	10	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Unknown C	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	7	0	0
Unknown D	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	27	0	0	0	0	0	0	0	0
Unknown E	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	20	0	0	0	0	1	3	0	0
Unknown F	0	0	1	8	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0

Unknown G	1	3	0	0	0	0	0	0	0	0	0	0	0	0	1	6	1	3	0	0	0	0	0	0	0	0
Unknown H	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	4	0	0	1	7	0	0	0	0	0	0
Unknown I	0	0	0	0	1	6	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Unknown J	0	0	1	10	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Unknown K	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	10	0	0	0	0	0	0	0	0
Unknown L	0	0	0	0	1	10	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Unknown M	1	3	0	0	0	0	0	0	0	0	0	0	0	0	1	10	0	0	0	0	0	0	0	0	0	0
Unknown Moss A	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	7	0	0	0	0	0	0	0	0	0	0
Unknown Moss B	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	6	0	0
Unknown N	0	0	0	0	2	3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	6	0	0
Unknown O	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	5	0	0	0	0	0	0	0	0	0	0
Unknown P	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	8	0	0
Unknown Q	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	1	6	0	0
Unknown R	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	11	0	0
Unknown Shrub	1	1	0	0	0	0	0	0	0	0	0	0	1	6	0	0	1	2	0	0	0	0	1	2	1	11
Vaccinium angustifolium	15	22	3	25	17	17	0	0	10	26	1	9	2	5	7	5	1	3	1	10	10	42	7	11	0	0
Vaccinium myrtilloides	25	25	4	26	22	33	3	9	13	14	13	29	4	23	21	8	6	18	2	2	16	33	13	20	10	30
Vaccinium oxycoccos	18	4	1	1	8	8	0	0	22	9	2	13	11	8	3	1	6	10	7	19	2	3	0	0	1	1
Vaccinium vitis-idaea	8	18	2	33	4	2	2	2	6	11	6	9	0	0	1	2	0	0	1	5	5	11	1	1	5	7
Viburnum edule	2	10	4	14	3	8	0	0	1	1	0	0	0	0	5	3	1	16	1	5	3	9	8	20	0	0
Vicia spp.	0	0	2	5	0	0	1	11	0	0	1	1	0	0	0	0	0	0	0	0	0	0	1	1	1	1
Viola spp.	8	2	10	3	7	3	0	0	3	1	3	1	1	1	14	4	16	7	6	5	1	1	17	9	5	1

Figure C2. Shrub layer over 1 m for each classification for the original classes for the FRI maps. The # column represents the number of transects that contain that species and the Area column represents the average number of 1m x 1m squares it represented in plots where it was present.

	Black Spruce		Young Deciduous		Young Conifer		Upland Conifer		Treed Muskeg		Rock		Open Muskeg		Mixed Conifer		Marsh-Shrub		Lowland Conifer		Jack Pine		Deciduous		Clearcut	
	#	Area	#	Area	#	Area	#	Area	#	Area	#	Area	#	Area	#	Area	#	Area	#	Area	#	Area	#	Area	#	Area
Alnus spp.	21	75	11	105	19	63	2	15	13	68	5	51	10	12	24	67	21	314	7	107	10	45	18	88	7	96
Ledum groenlandicum	21	3	1	1	9	4	1	2	8	11	6	2	0	0	3	1	4	2	4	5	7	1	2	3	2	4
Kalmia angustifolia	6	2	0	0	3	1	0	0	2	1	0	0	0	0	2	1	0	0	0	0	7	5	0	0	0	0
Salix spp.	15	27	12	37	20	21	2	6	11	12	8	24	12	25	9	19	19	139	7	18	13	8	16	28	9	30
Sorbus spp.	2	1	2	11	5	10	0	0	2	1	0	0	0	0	13	11	4	23	0	0	0	0	9	30	0	0
Prunus pensylvanica	0	0	7	27	7	9	1	2	1	10	8	9	0	0	4	11	2	143	1	31	3	1	11	30	7	4
Prunus virginiana	0	0	2	28	1	5	0	0	0	0	0	0	0	0	2	1	1	25	1	6	3	17	5	4	0	0
Betula spp.	3	11	1	3	7	53	0	0	9	55	1	3	12	25	0	0	8	167	6	94	1	5	0	0	1	1
Rubus ideae	1	1	4	2	1	6	1	1	0	0	0	0	0	0	4	9	2	19	0	0	1	1	4	24	0	0
Chamaedaphne calyculata	11	1	0	0	0	0	0	0	8	13	3	1	8	8	0	0	4	9	2	1	2	1	0	0	0	0
Amelanchier alnifolia	4	1	8	7	5	7	2	1	4	1	5	3	0	0	10	1	4	1	1	1	7	3	13	4	3	3
Acer spicatum	1	30	4	48	6	12	0	0	1	60	0	0	0	0	12	99	6	118	0	0	0	0	12	104	1	1
Rosa acicularis	4	1	3	11	0	0	1	1	0	0	1	1	0	0	6	1	3	4	0	0	2	2	6	2	1	1
Juniperus communis	0	0	1	1	0	0	0	0	1	1	10	9	0	0	0	0	1	5	0	0	3	4	0	0	1	1
Corylus cornuta	1	30	5	85	4	23	2	11	0	0	2	8	0	0	5	8	4	67	0	0	1	21	12	62	0	0
Myrica gale	0	0	0	0	1	4	1	1	2	11	1	4	8	38	1	5	1	15	0	0	0	0	0	0	0	0
Cornus stolonifera	0	0	2	1	2	10	0	0	0	0	0	0	0	0	1	1	6	18	0	0	0	0	5	2	1	1

Figure C3. Abiotic understory composition for each habitat classification for the original field classes for the FRI maps. The # column represents the number of transects that contain that species and the Area column represents the average number of 10cm x 10cm squares it represented (1% of 1 plot) in plots where it was present.

	Black Spruce		Young Deciduous		Young Conifer		Upland Conifer		Treed Muskeg		Rock		Open Muskeg		Mixed Conifer		Marsh-Shrub		Lowland Conifer		Jack Pine		Deciduous		Clearcut	
	#	Area	#	Area	#	Area	#	Area	#	Area	#	Area	#	Area	#	Area	#	Area	#	Area	#	Area	#	Area	#	Area
Rock	0	0	4	1	3	12	1	2	0	0	13	20	0	0	3	1	0	0	0	0	7	4	4	7	8	4
Water	4	1	3	9	3	14	0	0	5	11	3	9	14	37	3	0	14	22	8	4	2	9	2	13	3	14
Bare soil	1	0	3	5	3	12	1	0	1	0	2	1	2	55	2	0	4	1	2	1	1	0	3	1	2	2
Grass	26	4	13	8	20	10	0	0	23	9	11	6	20	36	15	5	21	17	9	18	9	1	21	5	11	8
Leaf Litter	30	23	14	75	28	43	3	52	26	20	15	32	19	33	31	53	22	49	10	32	21	29	28	77	13	45
CWD	30	249	14	130	29	292	3	28	25	113	14	67	15	36	31	307	22	178	10	52	21	181	28	328	13	162

Figure C4. Structural composition for the original field classes for the FRI maps. Values in the Arboreal lichen and Plot CWD (coarse woody debris) rows represent the number of plots where the quantity was represented on a scale of A to E (low to high). The dominant and secondary tree species were identified for each transect but was not limited to the plots.

	Arboreal Lichen					Plot CWD (1-5)					Dominant Tree	Secondary Tree	Dominant Shrub	Secondary Shrub
	A	B	C	D	E	A	B	C	D	E				
Black Spruce	12	8	6	2	2	2	15	8	1	4	Picea mariana	Abies balsamifera	Ledum groenlandicum	Alnus spp.
Young Deciduous	11	2	0	0	1	5	6	3	0	3	Populus spp.	Pinus banksiana	Alnus spp.	Salix spp.
Young Conifer	20	4	1	3	1	3	13	4	5	4	Picea mariana	Picea mariana	Ledum groenlandicum	Alnus spp.
Upland Conifer	2	1	0	0	0	0	3	0	0	0		Populus spp.	None	None
Treed Muskeg	11	6	3	1	5	16	7	2	1	0		Larix laricnia	Chamaedaphne calyculata	Ledum groenlandicum
Rock	8	4	0	3	0	6	7	1	1	0	Pinus banksiana	Picea mariana	Juniperus communis	Prunus pensylvanica
Open Muskeg	13	3	1	2	1	11	8	1	0	0	Picea mariana	Larix laricnia	Chamaedaphne calyculata	Myrica gale
Mixed Conifer	13	5	5	2	6	1	6	10	9	5		Abies balsamifera	Alnus spp.	Ledum groenlandicum
Marsh-Shrub	20	2	0	0	0	7	6	7	1	1		Picea mariana		Alnus spp.
Lowland Conifer	5	2	0	2	1	4	2	0	2	2	Larix laricnia		Betula spp.	
Jack Pine	14	5	0	0	2	4	10	5	0	2	Pinus banksiana		Alnus spp.	Ledum groenlandicum
Deciduous	23	4	0	1	0	2	12	6	5	3	Populus spp.	Abies balsamifera		Corylus cornuta
Clearcut	13	0	0	0	0	1	2	3	3	4	Pinus banksiana	Picea mariana		Ledum groenlandicum

Figure C5. Biotic understory composition for each habitat classification for the original field classes for the Landsat TM maps. The # column represents the number of transects that contain that species and the Area column represents the average number of 10cm x 10cm squares it represented (1% of 1 plot) in plots where it was present. Bold values identify the 10 most common species across all classes.

	Treed Wetland		Sparse		Open Wetland		Mixed		Marsh-Shrub		Disturbance		Deciduous		Coniferous Swamp		Conifer	
	#	Area	#	Area	#	Area	#	Area	#	Area	#	Area	#	Area	#	Area	#	Area
Abies balsamifera	1	50	0	0	0	0	34	53	0	0	6	37	12	48	11	61	2	112
Acer spicatum	0	0	0	0	0	0	14	56	0	0	0	0	6	46	0	0	0	0
Achillea millefolium	0	0	0	0	0	0	0	0	0	0	1	4	0	0	0	0	0	0
Actaea rubra	0	0	0	0	0	0	2	3	0	0	0	0	1	4	1	1	0	0
Alnus spp. (shrub)	3	12	0	0	5	21	18	53	1	192	8	24	10	31	28	53	3	11
Amelanchier alnifolia	0	0	0	0	0	0	11	5	0	0	2	2	1	4	2	14	4	2
Andromeda polifolia	2	4	0	0	10	20	0	0	0	0	1	1	0	0	5	31	1	1
Anemone spp.	0	0	0	0	0	0	2	1	0	0	0	0	4	1	3	1	0	0
Apocynum androsaemifolium	0	0	0	0	0	0	1	5	0	0	1	8	0	0	0	0	1	16
Aralia nudicaulis	0	0	0	0	0	0	25	19	0	0	4	3	10	37	2	4	2	10
Arctostaphylos uva-ursi	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0
Aster spp.	0	0	0	0	3	2	12	29	0	0	6	16	9	83	5	10	2	11
Betula papyrifera	1	1	0	0	2	9	15	4	1	40	1	1	7	4	2	4	3	4
Betula spp. (shrub)	2	3	0	0	6	19	0	0	1	43	3	21	0	0	6	19	0	0
Calla palustris	0	0	0	0	2	7	0	0	1	2	0	0	0	0	1	8	0	0
Caltha palustris	0	0	0	0	0	0	1	20	0	0	0	0	0	0	0	0	0	0
Chamaedaphne calyculata	5	104	1	112	13	136	6	41	2	53	6	71	0	0	27	44	3	79
Chimaphila umbellata	0	0	0	0	0	0	3	1	0	0	0	0	0	0	0	0	0	0
Cirsium arvense	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cladina mitis	1	2	0	0	5	16	11	6	1	3	8	3	0	0	12	23	6	3

Cladina rangiferina	3	39	1	26	4	28	15	3	1	2	12	40	1	5	23	30	11	32
Cladina stellaris	0	0	0	0	0	0	1	2	0	0	2	2	0	0	3	14	3	27
Cladonia	2	1	1	6	3	2	17	3	1	5	8	2	4	1	22	2	5	1
Cladonia borealis	0	0	1	6	1	1	3	1	0	0	4	1	1	2	6	1	0	0
Cladonia spp.	1	2	1	2	1	1	19	2	0	0	4	1	2	2	16	1	4	1
Climacium dendroides	0	0	0	0	0	0	1	6	0	0	0	0	1	30	0	0	0	0
Coptis trifolia	2	1	0	0	2	7	24	6	0	0	7	4	9	4	20	5	2	5
Cornus canadensis	1	9	0	0	2	13	29	25	0	0	10	32	14	22	28	17	6	14
Cornus stolonifera	0	0	0	0	0	0	4	11	1	142	3	10	2	8	2	3	0	0
Corydalis sempervirens	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Corylus cornuta	0	0	0	0	0	0	6	24	0	0	1	11	3	17	0	0	0	0
Crust Lichen	0	0	1	7	0	0	4	5	0	0	3	40	0	0	4	3	1	1
Diervilla lonicera	0	0	0	0	0	0	12	20	0	0	0	0	9	61	0	0	2	22
Drosera rotundifolia	0	0	0	0	4	1	0	0	0	0	0	0	0	0	3	1	1	1
Dryopteris austriaca	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Epilobium angustifolium	0	0	0	0	0	0	7	8	1	1	8	17	7	2	10	4	1	2
Equisetaceae	2	15	0	0	5	8	20	9	0	0	12	13	13	4	28	17	1	13
Feather Moss (Hypnales)	3	121	1	59	9	16	38	196	0	0	15	150	14	31	36	262	13	493
Feather Soil Moss	2	4	1	16	2	7	37	22	1	3	12	13	14	22	27	9	9	16
Fern	0	0	0	0	0	0	21	18	0	0	4	2	12	37	7	7	0	0
Fragaria virginiana	0	0	0	0	0	0	6	2	0	0	2	5	3	1	5	7	0	0
Galium spp.	0	0	0	0	0	0	13	2	1	1	3	1	11	3	3	5	0	0
Gaultheria hispidula	3	21	0	0	5	4	14	12	0	0	10	11	1	18	35	37	10	22
Gaultheria procumbens	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Geocaulon lividum	1	1	0	0	1	3	1	1	0	0	0	0	0	0	10	2	3	1

Geum spp.	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0
Impatiens capensis	0	0	0	0	0	0	4	4	0	0	0	0	1	1	0	0	0	0
Juniperus communis	0	0	0	0	1	14	0	0	0	0	0	0	0	0	1	24	0	0
Kalmia angustifolia	4	19	1	33	9	10	6	14	0	0	9	29	2	2	21	23	11	98
Kalmia polifolia	3	12	1	12	11	15	2	3	0	0	5	1	0	0	16	7	2	22
Larix laricina	1	1	0	0	4	12	1	7	0	0	3	9	1	8	7	20	0	0
Lathyrus spp.	0	0	0	0	0	0	2	4	0	0	0	0	0	0	0	0	0	0
Leaf Lichens	0	0	0	0	0	0	1	7	0	0	0	0	0	0	0	0	0	0
Ledum groenlandicum	5	201	1	68	10	77	11	38	0	0	12	80	3	41	36	108	9	55
Liliaceae spp.	4	22	1	19	9	44	30	32	1	9	12	14	13	47	37	15	9	16
Linnaea borealis	0	0	0	0	3	3	26	6	0	0	8	13	7	7	23	7	6	11
Lonicera oblongifolia	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lonicera villosa	0	0	0	0	1	2	2	8	0	0	1	1	1	5	9	8	0	0
Lycopodium complanatum	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0
Lycopodium obscurum	0	0	0	0	0	0	12	14	0	0	3	7	7	8	0	0	0	0
Lycopodium spp	0	0	0	0	0	0	24	34	0	0	4	21	9	39	10	15	2	7
Matted Rock Moss	0	0	0	0	0	0	2	1	0	0	0	0	0	0	0	0	0	0
Matted Tree Moss	0	0	0	0	0	0	20	16	0	0	2	4	5	4	11	3	2	3
Mentha arvensis	0	0	0	0	1	5	1	5	0	0	0	0	0	0	1	11	0	0
Menyanthes trifoliata	0	0	0	0	1	3	0	0	0	0	0	0	0	0	1	31	0	0
Mitella nuda	0	0	0	0	0	0	17	14	0	0	4	5	10	10	13	5	1	4
Myrica gale	0	0	0	0	2	46	1	5	1	127	0	0	0	0	3	62	0	0
Oxalis montana	0	0	0	0	0	0	3	17	0	0	0	0	2	13	0	0	0	0
Persicaria amphibia	0	0	0	0	1	7	0	0	0	0	0	0	0	0	0	0	0	0
Petasites palmatus	0	0	0	0	0	0	10	3	0	0	3	6	4	3	10	5	1	3

Petasites sagittatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Picea glauca	0	0	0	0	0	0	5	12	0	0	3	22	3	6	0	0	0	0
Picea mariana	5	105	1	210	10	152	27	41	0	0	13	101	8	22	37	137	13	125
Pinus banksiana	1	8	0	0	0	0	3	48	0	0	4	98	0	0	0	0	6	9
Polytrichum juniperinum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Polytrichum spp.	3	9	1	1	1	24	9	3	0	0	2	46	2	8	7	2	1	1
Populus spp.	0	0	0	0	0	0	11	6	0	0	4	8	10	6	1	2	0	0
Potentilla fruticosa	0	0	0	0	1	60	0	0	0	0	0	0	0	0	0	0	0	0
Potentilla palustris	0	0	0	0	3	13	0	0	2	4	0	0	0	0	2	1	0	0
Potentilla tridentata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Prunus pensylvanica	1	3	0	0	0	0	3	5	0	0	2	8	2	13	0	0	0	0
Prunus virginiana	0	0	0	0	0	0	3	12	0	0	0	0	1	6	0	0	0	0
Pyrolla spp.	0	0	0	0	0	0	15	2	0	0	1	1	8	9	4	3	0	0
Ranunculus lapponicus	0	0	0	0	0	0	0	0	0	0	1	1	0	0	1	8	0	0
Rhamnus alnifolia	0	0	0	0	1	50	3	34	1	17	0	0	0	0	0	0	0	0
Ribes americanum	0	0	0	0	0	0	1	45	0	0	2	5	0	0	1	4	0	0
Ribes glandulosum	0	0	0	0	0	0	8	11	0	0	4	16	9	23	3	1	0	0
Ribes lacustre	0	0	0	0	0	0	6	8	0	0	0	0	3	2	0	0	0	0
Ribes oxyacanthoides	0	0	0	0	0	0	3	1	0	0	0	0	1	1	1	8	0	0
Ribes triste	0	0	0	0	0	0	6	4	0	0	0	0	6	6	4	15	0	0
Rosa acicularis	0	0	0	0	1	8	19	5	0	0	3	8	7	3	6	6	4	4
Rubus chamaemorus	3	20	1	11	8	17	2	4	0	0	6	6	0	0	23	12	2	7
Rubus idaeus	0	0	0	0	1	3	19	16	0	0	8	32	12	53	8	16	0	0
Rubus pubescens	0	0	0	0	0	0	26	14	1	17	6	38	12	28	16	11	2	3
Salix spp. (shrub)	1	37	0	0	7	11	8	15	2	8	10	19	3	5	11	23	3	4

Sambucus spp.	0	0	0	0	0	0	1	7	0	0	0	0	1	20	0	0	0	0
Sanicula marilandica	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Sarracenia purpurea	0	0	0	0	2	1	0	0	0	0	0	0	0	0	0	0	1	3
Senecio eremophilus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Senecio spp	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Soil Moss	2	9	0	0	0	0	18	16	0	0	1	11	4	42	17	11	5	7
Solidago canadensis	0	0	0	0	0	0	0	0	0	0	1	4	0	0	1	2	0	0
Sorbus spp.	0	0	0	0	0	0	12	5	0	0	0	0	2	7	2	5	0	0
Sphagnum	5	582	1	570	12	601	16	127	2	73	11	296	3	47	35	314	4	388
Sterocaulon tomentosum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Taraxacum officinale	0	0	0	0	0	0	3	3	0	0	6	4	3	1	0	0	0	0
Taxus canadensis	0	0	0	0	0	0	1	9	0	0	0	0	0	0	0	0	0	0
Tetraphis pellucida	0	0	0	0	0	0	12	14	0	0	0	0	7	2	4	6	0	0
Thuja occidentalis	0	0	0	0	0	0	1	10	0	0	0	0	0	0	2	51	0	0
Trientalis borealis	0	0	0	0	2	1	23	3	0	0	4	1	9	3	8	2	1	1
Trifolium spp.	0	0	0	0	0	0	1	15	0	0	2	10	0	0	0	0	0	0
Typha latifolia	0	0	0	0	0	0	0	0	0	0	1	5	0	0	0	0	0	0
Unknown A	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Unknown B	0	0	0	0	0	0	0	0	0	0	1	10	0	0	0	0	0	0
Unknown C	0	0	0	0	0	0	0	0	0	0	0	0	1	7	0	0	0	0
Unknown D	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	27	0	0
Unknown E	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Unknown F	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Unknown G	0	0	0	0	0	0	3	4	0	0	0	0	0	0	0	0	0	0
Unknown H	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Unknown I	0	0	0	0	0	0	0	0	0	0	1	6	0	0	0	0	0	0
Unknown J	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Unknown K	0	0	0	0	0	0	0	0	1	10	0	0	0	0	0	0	0	0
Unknown L	0	0	0	0	0	0	0	0	0	0	1	10	0	0	0	0	0	0
Unknown M	0	0	0	0	0	0	1	3	0	0	0	0	0	0	0	0	0	0
Unknown Moss A	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Unknown Moss B	0	0	0	0	0	0	1	6	0	0	0	0	0	0	0	0	0	0
Unknown N	0	0	0	0	0	0	0	0	0	0	2	3	1	6	0	0	0	0
Unknown O	0	0	0	0	0	0	1	5	0	0	0	0	0	0	0	0	0	0
Unknown P	0	0	0	0	0	0	0	0	0	0	0	0	1	8	0	0	0	0
Unknown Q	0	0	0	0	0	0	2	3	0	0	0	0	0	0	0	0	0	0
Unknown R	0	0	0	0	0	0	1	11	0	0	0	0	0	0	0	0	0	0
Unknown Shrub	0	0	0	0	1	6	1	2	0	0	0	0	0	0	0	0	0	0
Vaccinium angustifolium	3	20	0	0	4	9	10	10	0	0	11	25	5	8	25	20	10	45
Vaccinium myrtilloides	4	6	1	1	6	24	21	17	0	0	13	43	7	12	31	27	12	27
Vaccinium oxycoccos	4	9	1	4	11	7	5	6	0	0	5	8	0	0	25	6	3	5
Vaccinium vitis-idaea	0	0	0	0	0	0	1	61	0	0	0	0	0	0	0	0	0	0
Viburnum edule	0	0	0	0	0	0	9	11	0	0	4	7	4	30	4	6	0	0
Vicia spp.	0	0	0	0	0	0	1	11	0	0	1	9	0	0	0	0	0	0
Viola spp.	0	0	0	0	1	1	22	6	2	1	3	4	13	10	12	4	0	0

Figure C6. Shrub layer over 1 m for each habitat classification for the original field classes for the Landsat TM maps. The # column represents the number of transects that contain that species and the Area column represents the average number of 1m x 1m squares it represented (approximately 2% of 1 plot) in plots where it was present.

	Treed Wetland		Sparse		Open Wetland		Mixed		Marsh-Shrub		Disturbance		Deciduous		Coniferous Swamp		Conifer	
	#	Area	#	Area	#	Area	#	Area	#	Area	#	Area	#	Area	#	Area	#	Area
Alnus spp.	2	131	0	0	5	14	25	105	2	188	9	66	14	117	30	102	4	44
Ledum groenlandicum	2	7	0	0	0	0	3	2	0	0	4	1	1	1	17	2	4	2
Kalmia angustifolia	1	1	0	0	0	0	1	1	0	0	2	1	0	0	9	1	7	5
Salix spp.	3	2	0	0	6	4	15	54	2	83	14	25	10	52	24	28	9	8
Sorbus spp.	0	0	0	0	1	1	18	18	0	0	3	4	7	31	7	4	1	1
Prunus pensylvanica	1	31	0	0	0	0	11	27	0	0	5	17	7	71	1	10	0	0
Prunus virginiana	1	6	0	0	0	0	3	1	0	0	0	0	3	11	0	0	0	0
Betula spp.	2	14	0	0	7	24	1	1	2	136	4	3	0	0	10	42	0	0
Rubus ideae	0	0	0	0	0	0	4	8	0	0	3	4	1	43	0	0	0	0
Chamaedaphne calyculata	0	0	0	0	3	3	1	22	0	0	0	0	0	0	6	1	0	0
Amelanchier alnifolia	0	0	0	0	1	1	12	1	0	0	3	1	5	4	8	1	4	1
Acer spicatum	0	0	0	0	1	60	20	99	0	0	1	13	6	102	0	0	0	0
Rosa acicularis	0	0	0	0	0	0	7	1	0	0	0	0	0	0	3	1	0	0
Juniperus communis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Corylus cornuta	0	0	0	0	0	0	7	41	0	0	1	5	4	29	0	0	0	0
Myrica gale	0	0	0	0	2	10	2	3	1	15	0	0	0	0	1	10	0	0
Cornus stolonifera	0	0	0	0	0	0	6	2	1	95	3	7	1	1	0	0	0	0

Figure C7. Abiotic understory composition for each habitat classification for the original field classes for the FRI maps. The # column represents the number of transects that contain that species and the Area column represents the average number of 10cm x 10cm squares it represented (1% of 1 plot) in plots where it was present.

	Treed Wetland		Sparse		Open Wetland		Mixed		Marsh-Shrub		Disturbance		Deciduous		Coniferous Swamp		Conifer	
	#	Area	#	Area	#	Area	#	Area	#	Area	#	Area	#	Area	#	Area	#	Area
Rock	0	0	0	0	0	0	5	2	0	0	0	0	0	0	0	0	0	0
Water	1	3	0	0	6	52	4	3	1	10	1	0	0	0	6	1	0	0
Bare soil	0	0	0	0	0	0	5	0	0	0	3	16	0	0	1	0	0	0
Grass	4	4	1	7	13	23	18	4	2	57	11	6	11	2	32	6	5	6
Leaf Litter	5	20	1	25	13	22	38	63	2	56	15	48	14	84	37	27	13	26
CWD	5	6	1	2	13	3	38	12	2	9	15	9	14	13	37	7	13	9

Figure C8. Structural composition for the original field classes for the Landsat TM maps. Values in the Arboreal lichen and Plot CWD (coarse woody debris) rows represent the number of plots where the quantity was represented on a scale of A to E (low to high). The dominant and secondary tree species were identified for each transect but was not limited to the plots.

		Treed Wetland	Sparse	Open Wetland	Mixed	Marsh-Shrub	Disturbance	Deciduous	Coniferous Swamp	Conifer
Arboreal Lichen	A	3	0	5	21	1	12	12	14	8
	B	0	0	3	6	1	2	1	8	1
	C	1	0	1	6	0	1	0	5	0
	D	0	0	2	1	0	0	1	6	1
	E	1	1	3	4	0	0	0	4	3
Plot (CWD)	A	2	0	6	2	1	4	0	11	4
	B	1	1	8	15	0	5	8	13	5
	C	1	0	0	11	0	2	3	11	2
	D	1	0	0	6	0	2	2	2	1
	E	0	0	0	4	1	2	1	0	1
Dominant Tree		Picea mariana	Picea mariana	Picea mariana	Picea mariana	None	Picea mariana	Populus spp.	Picea mariana	Pinus banksiana
Secondary Tree		Larix laricina	Larix laricina	Larix laricina	Betula papyrifera	None	Larix laricina	Picea mariana	Abies balsamifera	Picea mariana
Dominant Shrub		Ledum groenlandicum	Chamaedaphne calyculata	Chamaedaphne calyculata	Alnus spp.	None	Ledum groenlandicum	Alnus spp.	Ledum groenlandicum	Kalmia angustifolia
Secondary Shrub		None	Ledum groenlandicum	Ledum groenlandicum	Alnus spp.	None	Chamaedaphne calyculata	Prunus pensylvanica	Alnus spp.	Ledum groenlandicum

Example Datasheet

PLOT ID : ______________________________ TIME (ET) (24hr-clock) :
PLOT DATE : (dmy) ______________________ ____________________

Actual Easting Start (UTM): Actual Easting End (UTM):

Actual Northing Start (UTM): Actual Northing End (UTM):

GPS error +/- ____________ m Bearing:

Habitat class from field : DC MC OC MF BL TW OW SHR Herb BarL Agri Regen(~age ___) Water

Plot 1 – **0 m** Easting ____________ Northing ____________ Soil___________________________

Con ______ Dec_______	**Prism**				**4 m radius**			
Species	# alive in	# dead in	# alive border	# dead border	DBH 0-10	DBH 10-20	DBH 20-30	DBH 30 +

Plot 2 – **20 m** Easting ____________ Northing ____________ Soil___________________________

Con ______ Dec_______	**Prism**				**4 m radius**			
Species	# alive in	# dead in	# alive border	# dead border	DBH 0-10	DBH 10-20	DBH 20-30	DBH 30 +

Plot 3 – **40 m** Easting ____________ Northing ____________ Soil___________________________

Con ______ Dec_______	**Prism**				**4 m radius**			
Species	# alive in	# dead in	# alive border	# dead border	DBH 0-10	DBH 10-20	DBH 20-30	DBH 30 +

Plot 4 – **60 m** Easting ____________ Northing ____________ Soil___________________________

Con ______ Dec_______	**Prism**				**4 m radius**			
Species	# alive in	# dead in	# alive border	# dead border	DBH 0-10	DBH 10-20	DBH 20-30	DBH 30 +

Plot 5 – **80 m** Easting ____________ Northing ___________ Soil___________________________

Con ______ Dec_______	**Prism**				**4 m radius**			
Species	# alive in	# dead in	# alive border	# dead border	DBH 0-10	DBH 10-20	DBH 20-30	DBH 30 +

Plot 6 – **100 m** Easting ____________ Northing ___________ Soil___________________________

Con ______ Dec_______	**Prism**				**4 m radius**			
Species	# alive in	# dead in	# alive border	# dead border	DBH 0-10	DBH 10-20	DBH 20-30	DBH 30 +

Plot 7 – **120 m** Easting ____________ Northing ___________ Soil___________________________

Con ______ Dec_______	**Prism**				**4 m radius**			
Species	# alive in	# dead in	# alive border	# dead border	DBH 0-10	DBH 10-20	DBH 20-30	DBH 30 +

Plot 8 – **140 m** Easting ____________ Northing ___________ Soil___________________________

Con ______ Dec_______	**Prism**				**4 m radius**			
Species	# alive in	# dead in	# alive border	# dead border	DBH 0-10	DBH 10-20	DBH 20-30	DBH 30 +

% COVER below waist height	Cover (%)									Height Plot (m)
	0m	20m	40m	60m	80m	100 m	120 m	140 m		
Acer spicatum										
Acteae rubra										
Alnus sp.										
Apocynum spp										
Andromeda polifolia										
Aralia nudicaulis										
Arctostaphylos										
Aster										
Chamaedaphne calyculata										
Cladina Mitis										
Cladina Rangiferana										
Cladina stellaris										
Clintonia borealis										
Cornus canadensis										
Corylus cornuta										
Cyperaceae										
Diervilla lonicera										
Epilobium angustifolium										
Equisetaceae										
Fragaria virginiana										
Galium sp										
Gaultheria spp										
Gaultheria hispidula										
Kalmia polifolia										
Lathyrus ochroleucus										
Ledum groenlandicum										
Linnaea borealis										
Lycopodium complanatum										
Maianthemum canadense										
Melampyrum lineare										
Mertensia paniculata										
Mitella nuda										
Monotropa uniflora										
Petasites palmatus										
Poaceae										
Potentilla palustris										
Prunus pensylvanica										
Pyrola										
Ribes glandulosum										
Ribes sp.										
Rosa acicularis										
Rubus Chamaemorus										
Rubus sp										
Smilacina trifolia										
Streptopus roseus										
Trientalis borealis										
Vaccinium angustifolium										
Vaccinium myrtilloides										
Vaccinium oxycoccos										

Vaccinium vitis-idaea										
Vibrunum edule										
Viola renifolia										
Lichen and Mosses										
Club Lichen										
Foliose Lichen										
Feather Moss										
Sphagnum										
Hair Cap Moss										
Crust Lichen										
Arboreal Lichen (A = min, B = some, C = fair, D = good, E = lots)										

OTHER SPECIES	**Cover (%)**										**Height Plot (m)**
	0m	**20m**	**40m**	**60m**	**80m**	**100m**	**120m**	**140m**			
ABIOTIC											
Rock											
Water											
Bare Soil											
Grass											

Habitat Compositions

Field data was combined for each habitat classification to determine common features throughout each class. The following is a description of each habitat type for both map types and the most common features that could be used to define the classes.

For the entire landscape there were several extremely common plants. *Picea mariana* was the most common tree, representing 42 percent of those found with *Pinus banksiana*, *Abies balsamifera*, and members of the *Populus* genus representing 17, 15, and 13 percent respectively. Shrubs above 1 metre were dominated by the genus *Alnus* which represented just under 50 percent of all shrubs. Shrubs from the genera *Salix*, *Betula*, and the species *Acer spicatum* and *Corylus cornuta* were the only shrub types representing greater than 5 percent: 17, 10, 8 and 5 percent respectively. The understory and herbaceous layer was dominated by short shrubs, mosses, and saplings. *Sphagnum spp.* and Feather mosses (*Hypnales spp.*) each represented 19 percent of all plots. In total over 150 species were measured with the vast majority representing less than 0.5 percent of the total coverage. Complete classification compositions for FRI stands are included in table C1 and for Landsat TM stands within table C5. A verbal description of each classification and their most common features follows.

Forest Resource Inventory Habitat Classifications

Black Spruce

Black Spruce stands tended to be mossy damp sites in a patchy shrub layer mainly consisting of *Alnus spp*. with some *Salix spp.* growing above 1 metre tall and *Ledum groenlandicum* growing extensively below 1 metre complemented by *Chamaedaphne calyculata* and *Kalmia angustifolia*. *Sphagnum spp*. and fan like mosses alone made up nearly 50 percent of all Black Spruce stands. Other common features included *Gaultheria hispidula*, *Vaccinium myrtilloides*, *Trientalis borealis*, members of the *Liliaceae genus* and *Cornus canadensis*. As the stands name suggests *Picea mariana* represented well over 90% of the trees present in the habitat. Little lichen was found in Black Spruce stands with the exception of *Cladina rangiferina* which represented approximately 2 percent of the ground cover in the habitat.

Clearcut

Clearcut sites varied depending on the date of cutting. The more recent sites were drier while older sites were damper. The most common plants in these heavily altered sites were feather mosses, representing nearly 30% of the forest floor with young *Picea mariana* below 1 metre representing nearly 10 percent. Short shrubs such as *Vaccinium myrtilloides*, *Ledum groenlandicum*, *Diervilla lonicera*, *Juniperus communis*, and *Arctostaphylos uva-ursi* were common with few succulent plants besides *Cornus canadensis* and *Liliaceae spp.* occurring with any frequency. *Alnus spp*. and *Salix spp*. above 1 metre were common throughout the sites, with *Prunus pensylvanica* representing a small portion in the drier sites. Often these sites would have a combination of young *Pinus banksiana* and *Picea mariana* with *Populus spp.* commonly found alongside. These often appeared to be planted however this was not always the case. Dead plant material, both herbaceous and woody, was found commonly with large cut wood and naturally fallen wood being extremely abundant throughout. *Cladina rangiferina* and *Cladina mitis* were common lichens throughout with Crust Lichens relatively frequent.

Deciduous

Deciduous sites were represented by habitat stands that had greater than 20 percent or more deciduous trees found in them. These sites were usually the most diverse sites I examined, with a wide range of undergrowth and shrubs. Usually these forests consisted of a canopy of *Populus spp*. and *Betula papyrifera* with a relatively thick understory of *Abies balsamifera*. *Picea mariana* and full grown *Abies balsamifera* were also quite common as were tall shrubs. *Alnus spp*. and *Acer spicatum* represented the majority of the tall shrubs found in these sites however *Corylus cornuta*, *Salix spp.*, *Prunus pensylvanica*, and *Sorbus spp.* were also all common. Beneath these the forest floor was extremely diverse in comparison to other sites. While not a mossy habitat, some of the most common plants were Feather mosses and *Lycopodium spp*. However these both collectively represented not even 15 percent of the species. Other common species include smaller shrubs such as *Diervilla lonicera*, *Rubus idaeus*, *Ribes glandulosum*, and herbaceous plants such as *Liliaceae spp*., *Aralia nudicaulis, Cornus canadensis*, and various Fern species. As would be expected with a canopy of deciduous trees leaf litter was the most common groundcover. Lichen was nearly absent from deciduous sites, with *Cladina mitis* the most common but representing less than half of a percent of the understory.

Jack Pine

The tallest and often driest of the forested sites, Jack Pine forests were often found on high ground and were usually composed of equal parts *Pinus banksiana* and *Picea mariana*. As a result of their drier soils lichens were common in these forests. *Cladina rangiferina, Cladina mitis*, Crust Lichen, and *Cladina stellaris* were all common species found in Jack Pine forests. The sites were often quite mossy, with feather mosses accounting for nearly half of the groundcover and feather soil mosses and *Sphagnum spp*. both occurring in the 10 most common species. Short shrubs were common, including *Kalmia angustifolia*, both *Vaccinium angustifolium* and *Vaccinium myrtilloides*, *Ledum groenlandicum*, and *Gaultheria hispidula*. No herbaceous species were found with any great occurrence. *Cornus canadensis* was the most common however it only represented approximately 0.5 percent of the species found.

Lowland Conifer

The wettest sites with full grown trees, Lowland Conifer sites were similar to the Black Spruce habitats in composition however they were defined by the higher occurrence of *Larix laricina*. Few other full grown trees were present in large numbers, however an understory of

Abies balsamifera was common. *Alnus spp*. was still the most common tall shrub species, however, as was common with many of the wet sites *Betula spp*. were also quite common. *Salix spp*., *Prunus pensylvanica*, and tall *Ledum groenlandicum* were also present. The understory of Lowland conifer was dominated by mosses such as *Sphagnum spp*. and feather mosses, and low lying shrubs including the very common *Ledum groenlandicum*, *Alnus spp*., *Chamaedaphne calyculata, Vaccinium oxycoccos, Salix spp*., and *Betula spp*. Low lying tree branches were often common with *Abies balsamifera, Picea mariana*, and *Larix laricina* representing a larger proportion of the 1 metre plots than nearly all herbaceous plants, the most common of which were *Liliaceae spp*., representing just over 1 percent. Grasses were common throughout this habitat. Small portions of standing water were common however they did not represent a significant proportion of the transects. Ground growing lichen was extremely uncommon.

Shrub Marsh

Shrub Marsh sites were determined to be wet sites dominated by tall shrubs rather than by trees. These sites were often extremely thickly vegetated with *Alnus spp*. and *Salix spp*. These two shrubs alone represented 50% of the habitat above 1 metre. *Betula spp.*, *Acer spicatum*, *Prunus pensylvanica* and *Corylus cornuta* were also common, with *Betula spp*. occurring more often in the wetter sites and the remainder occurring in the drier sites. Groundcover was diverse in these sites, with Sphagnum app. and feather mosses representing approximately a quarter of all species *Ledum groenlandicum*, *Rubus idaeus*, and *Chamaedaphne calyculata* were also common below 1 metre. The most common herbaceous plants were *Liliaceae spp.*, grasses, and ferns. Feather soil mosses were also present, but in lesser quantities than the other mosses. Lichens were not common in this habitat, however *Cladina rangiferina* was occasionally found. Thick wet leaf litter often covered much of the ground. Most trees found were *Picea mariana*, however *Abies balsamifera*, *Populus spp*., and *Larix laricina* were also all found in small, approximately equal quantities.

Mixed Conifer

Mixed Conifer was a sort of catch all class with a canopy of predominantly *Abies balsamifera* and *Picea mariana*, but also included high numbers of *Populus spp*. and *Betula papyrifera* and consequently was often misclassified as Deciduous. These sites were often less diverse than Deciduous forests, with a thick mossy floor. Feather mosses, *Sphagnum spp*., and *Lycopodium spp*. represented nearly half of all plots with feather soil mosses rounding out the total to an even 50 percent. *Picea mariana* and *Abies balsamifera* saplings were common, as was short *Ledum groenlandicum, Gaultheria hispidula*, and *Rubus idaeus*. Herbaceous plants included *Liliaceae spp*., *Cornus canadensis*, and *Aralia nudicaulis*. Large wooden debris was nearly as abundant in Mixed Conifer stands as it was in Clearcut stands. Lichens were rare with *Cladina mitis* representing approximately 0.5 percent and the next most common *Cladina rangiferina* well below that.

Open Muskeg

Open muskeg stands were often extremely wet sites with open standing water and thick peat dominated soils very common, with little to no trees. *Picea mariana* was the most common tree in this habitat, but with only 4 trees on average per transect the totals were extremely low. Tall shrubs were also rare with *Salix spp*., *Betula spp*., and *Myrica gale* all representing just under 30 percent of all tall shrubs present. Open muskeg was also the only habitat where feather mosses did not represent a large portion of the understory, however *Sphagnum spp*. alone represents over 40 percent. The remainder was composed of short shrubs (*Chamaedaphne calyculata*, *Myrica gale, Ledum groenlandicum, Andromeda polifolia*, and two Vaccinium species (*V. myrtilloides* and *V. oxycoccos*)) and simple herbaceous plants such as Grasses and *Liliaceae spp*., *Cladina rangiferina* and *Cladina mitis* represented some of the most common species in this habitat, however as it was dominated by mosses and shrubs their representative percentages are both below 1 percent each. As there were few trees in these plots coarse woody debris was also very uncommon.

Rock

The Rock habitat class was restricted to Manitoba as rock represented well less than 1% of the landscape in Ontario and there were no appropriately sized habitat patches in reach from any road or lake. Despite its name exposed rock only represented slightly more than 2 percent of the groundcover in this class. Instead the majority was rock covered with mosses such as Feather moss and *Sphagnum spp.* and to a lesser extent *Polytrichum spp*., short shrubs such as *Chamaedaphne calyculata*, *Juniperus communis*, *Ledum groenlandicum*, and *Vaccinium myrtilloides*, and Lichens. Lichens appear to be most commonly found on Rock sites, with *Cladina mitis*, Crust lichens, *Cladina rangiferina*, Leaf lichens, and *Cladina stellaris* all being found in the top 15 most common species. *Cladonia spp*. and *Sterocaulon tomentosum* were also found with some regularity. Both trees and shrubs were much less common than in most other sites. Most sites consisted of small *Pinus banksiana* and *Picea mariana*. Shrubs usually occurred in much less dense aggregations than they did in other habitats. These included *Alnus spp*., *Salix spp*., *Juniperus communis*, and *Prunus pensylvanica*. Coarse woody debris was uncommon.

Treed Muskeg

Similar in tree composition to stunted Lowland Conifer and understory composition to Open Muskeg, Treed Muskeg was a wet moss dominated landscape. *Sphagnum spp*.represents nearly 50 percent of the understory cover with Fan Like moss representing nearly 10 percent itself. Short shrubs, such as *Chamaedaphne calyculata*, *Ledum groenlandicum*, *Kalmia polifolia*, *Vaccinium angustifolium*, and *Andromeda polifolia* were all common, as were *Liliaceae spp.* and *Rubus chamaemorus*. *Lycopodium spp.* were often present, but at a much smaller percentage than the previously mentioned mosses. *Alnus spp.* and *Betula spp.* were well represented above 1 metre and *Picea mariana*and *Larix laricina* represented nearly all the trees. *Cladina rangiferina* was the only common lichen, but due to the dominance of *Sphagnum spp*., it still does not account for 1 percent of the total. As most trees were stunted there was little coarse woody debris.

Upland Conifer

Upland Conifer was strongly dominated by *Picea mariana* trees with *Abies balsamifera*, *Populus spp.*, and *Pinus banksiana* each accounting for between 15-20 percent of the total. These sites were without shrubs much of the time, with *Alnus spp.*, *Hazel spp.*, and *Salix spp.* occurring most often. Small *Abies balsamifera* saplings were common under 1 metre, often complimented by a community dominated by Feather Moss (*Hypnales*), *Sphagnum spp*., *Cladonia spp.*, and matted tree mosses and a number of different Lichens. The most common were *Cladonia spp.*, *Cladina mitis*, Leaf lichens, *Cladina rangiferina*, and Rock lichens. Short shrubs were also common here, usually restricted to shrubs that are capable of growing within the Upland Conifer's drier environment, including *Diervilla lonicera* and *Vaccinium myrtilloides*. *Cornus canadensis* was the only common herbaceous plant found with any regularity in comparison to the mosses and shrubs. Fallen trees were uncommon but present.

Young Conifer

Young Conifer sites were a diverse site, with species being found that had not been found in any other site before. Common species of tree were *Picea mariana* and *Pinus banksiana*, often with a much younger *Abies balsamifera* component that was often not picked up using the prism. Tall shrubs were common, with *Alnus spp.*, *Salix spp.*, and *Betula spp*. the predominant species found. As with nearly all sites *Hypnales spp*. and *Sphagnum spp*. were the most common each comprising nearly 21 percent of the species total. Young conifer was particularly diverse in the number of short shrubs found, including *Ledum groenlandicum*, *Vaccinium myrtilloides, Chamaedaphne calyculata, Gaultheria hispidula, Diervilla lonicera*, and *Kalmia angustifolia*. Only *Cornus canadensis* was found with a comparable rate of occurrence. Deadfall was common throughout as was *Cladina rangiferina*, however all other lichens were poorly represented.

Young Deciduous

While predominantly *Populus spp*. saplings, the tree composition of Young Deciduous was diverse, with *Abies balsamifera, Picea mariana*, and *Pinus banksiana* all representing a decent proportion of the total. It should also be noted that Young Deciduous contained the highest percentage of *Thuja occidentalis* trees out of all habitat classifications, nearly equalling one tree per transect. Tall shrubs were also well represented here, with *Alnus spp*. representing nearly an entire plot per transect and both *Salix spp*. and *Corylus cornuta* both common. Less common shrubs were also found here in larger quantities than any of the other classifications, including *Rosa acicularis* and *Amelanchier alnifolia*. *Hypnales spp*. was the most common understory species but only represented 15 percent of the total. *Aster spp*., so far unmentioned, were the third most common understory species with *Rubus pubescens*, *Cornus canadensis* and *Liliaceae spp*. rounding out the herbaceous representatives. Short shrubs such as *Rubus idaeus* and *Diervilla lonicera* were also commonly found. Coarse woody debris was common as was Lichen covered rocks.

Note

There are several common species that were often found throughout the landscape that were not represented in the above classifications, either due to their diffuse nature or their small size. These included *Equisetaceae spp.* (0.46%), *Linnaea borealis* (0.39%), *Mitella nuda* (0.27%), *Epilobium angustifolium* (0.26%), *Viola spp.* (0.2%), and *Coptis trifolia* (0.17%).

Landsat TM Classification

Conifer

Conifer forest stands are generally drier coniferous forests than the Coniferous Swamp classification. Conifer stands are dominated by *Pinus banksiana* and *Picea mariana* trees with sparse tall *Alnus spp., Salix spp.,* and *Kalmia angustifolia*. The forest floor was dominated by *Hypnales spp.* with *Sphagnum spp*.and occurring frequently. A thick understory of *Picea mariana* and *Abies balsamifera* mixed with short *Kalmia angustifolia, Ledum groenlandicum, Vacciniums* (*angustifolium* and *myrtilloides*), *Chamaedaphne calyculata*, and *Gaultheria hispidula* was common. Few herbaceous plants were common with the exception of *Liliaceae spp*. and *Cornus canadensis*. Lichens were also present, with *Cladina rangiferina* being one of the most frequently occurring species. *Cladina stellaris* was also present although not as common as *Cladina rangiferina*.

Coniferous Swamp

Coniferous Swamp represents a vast percentage of the landscape, dominated by *Picea mariana* trees with predominantly damp peat dominated soils its understory was not unlike the conifer forest stands. *Alnus spp*. was found in greater quantities but dominated the shrub layer with *Salix spp*. and *Betula spp*. both occurring with regularity. The herbaceous layer was predominantly moss, with both *Hypnales spp*. and *Sphagnum spp*. representing approximately 25 percent each and short shrubs such as *Ledum groenlandicum, Gaultheria hispidula, Chamaedaphne calyculata, Vaccinium myrtilloides* and *angustifolium*, and *Kalmia angustifolia* occurring frequently. *Cladina rangiferina* was common as was, to a lesser extent, *Cladina mitis*. *Liliaceae spp*. and *Cornus canadensis* again were the most common herbaceous plants. This forest type had little downed trees but sizeable quantities of arboreal lichens were common.

Deciduous

The Landsat TM Classification of Deciduous was similar to the FRI classification in its diversity. The most common understory plant, *Rubus idaeus*, represented only 7.5 percent of the total diversity. Herbaceous plants were extremely common with *Liliaceae spp.*, *Aster spp*., Ferns, *Aralia nudicaulis*, and *Rubus pubescens* all occurring with regularity. Common tall shrubs included *Alnus spp*. at 44 percent but followed closely by *Acer spicatum*, *Salix spp*., and *Prunus pensylvanica* with *Sorbus spp*. and *Corylus cornuta* present as was *Diervilla lonicera* below 1 metre. *Sphagnum spp*. was not common, however *Hypnales spp*. represented approximately 5 percent of the variety with *Lycopodium spp*. common as well. Deciduous forests were dominated by *Populus spp*. trees but *Abies balsamifera*, *Picea mariana*, and *Betula papyrifera* trees were also common. Lichens were virtually non-existent in Deciduous stands, with *Cladina rangiferina* the most common but representing approximately a third of a percentage point of the forest floor.

Disturbance

Sites disturbed in the past 20 years were classified as Disturbed and were dominated by *Picea mariana* with *Pinus banksiana* and *Abies balsamifera*. Tall shrubs appeared with regularity but not in great numbers and were predominantly *Alnus spp.* and *Salix spp.* with some *Prunus pensylvanica* and *Acer spicatum*. Moss was well represented in the groundcover with *Sphagnum spp*. accounting for a quarter of the cover and *Hypnales spp*. approximately a sixth. *Ledum groenlandicum*, *Chamaedaphne calyculata*, *Vaccinium myrtilloides* and *angustifolium*, *Kalmia angustifolia*, and *Rubus idaeus* were all common short shrubs coinciding with *Cornus canadensis* and *Liliaceae spp.* Low growing and low lying branches of *Picea mariana* and *Abies balsamifera* were also common under 1 metre. *Cladina rangiferina* represented 3 percent of the variety with no other lichens occurring with any great frequency.

Mixed

Mixed stands were combinations of Deciduous and Coniferous rather than Mixed Coniferous like the FRI classification but were the only habitat classification dominated by *Abies balsamifera* representing nearly 40 percent, followed by *Picea mariana*, *Betula papyrifera*, and *Populus spp*. Shrubs above 1 metre was relatively common with *Alnus spp*. and *Acer spicatum* dominant with *Salix spp., Sorbus spp.,* and *Prunus pensylvanica* following. Mosses were common, with *Hypnales spp*. representing a quarter of the groundcover and *Sphagnum spp*. nearly 10 percent. *Lycopodium spp*. was also common. Low lying trees such as *Abies balsamifera* and *Picea mariana* and tall shrubs such as *Alnus spp.* and *Acer spicatum* were common. A variety of herbaceous plants were more common in Mixed stands than in most Conifer dominated stands. These included *Liliaceae spp., Cornus canadensis, Aralia nudicaulis,* Ferns and *Rubus pubescens*. Short shrubs were less common with only *Ledum groenlandicum* present in any quantity. Lichens, as with the Deciduous category, were not common.

Open Wetland

Open Wetland consisted of grassy and mossy sites with extremely damp soils or vegetated water. The only trees found in this classification were *Picea mariana* and *Larix laricina*, however neither was found in large quantities. Similarly tall shrubs, represented by *Betula spp*. and *Alnus spp*., were not common. The groundcover was over half *Sphagnum spp*. with small shrubs such as *Chamaedaphne calyculata* and *Ledum groenlandicum* were common with occasional *Kalmia polifolia, Vaccinium myrtilloides*, *Andromeda polifolia*, and *Kalmia angustifolia* occurring. Herbaceous plants such as *Liliaceae spp*. and *Rubus chamaemorus* were also present. *Cladina rangiferina* and *Cladina mitis*, although no common, were consistently present at low levels.

Treed Wetland

Heavily dominated by *Picea mariana* this habitat had few tall shrubs and a mossy wet substrate. *Sphagnum spp*. represented approximately 45 percent of the variety with *Hypnales spp*. also common. Short hydrophilic shurbs such as *Ledum groenlandicum, Chamaedaphne calyculata, Kalmia angustifolia, Gaultheria hispidula, Vaccinium angustifolium*, and *Kalmia polifolia* were common. *Liliaceae spp*. and *Rubus chamaemorus* were also occasionally present. *Cladina rangiferina* was well represented, however no other lichens were common.