FIELD GUIDE: MAPPING CARIBOU FORAGE RESOURCES IN ONTARIO

Prepared by:
National Council for Air and Stream Improvement, Inc.

Prepared for:
Ontario's 2023-24 CCA Stewardship Program

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February 28, 2025

Acknowledgements

The development of this publication and the associated mapping products was supported through funding from Ontario's 2023-2024 Caribou Conservation Stewardship Program (CCSP) and in-kind support provided by the National Council for Air and Stream Improvement.

The collection of data forming the foundation for this report was supported through funding and in-kind support provided by the National Council for Air and Stream Improvement, Ontario Ministry of Natural Resources and Forestry, Centre for Northern Forest Ecosystem Research (Lakehead University), University of Alberta, Alberta Chapter of The Wildlife Society, and Natural Sciences and Engineering Research Council of Canada Collaborative Research and Training Experience Environmental Innovation. We acknowledge that our vegetation plots were sampled across Treaty 3 and 9 territories, which are the traditional lands of the Abitibiwinni, Anishinaabe, Cree, and Metis people. Yvonne Allen, Derrick Lancaster, Emily Cormier, Tasha Rabinowitz, Kristin Denryter, Brooke Davison, Peter Appleton, Anthony Francis, Evan Rundle, Landon Birch, and Janelle Giberson assisted with data collection in the field. Art Rodgers, Janet Lane, Jevon Hagens, and Chris Chenier provided local expertise and helped facilitate this research in Ontario.

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Cite this report as:

NCASI. 2025. *Field Guide: Mapping Caribou Forage Resources in Ontario*. Cary, NC: National Council for Air and Stream Improvement, Inc.

About this report:

In 2024, NCASI received a grant from Ontario's 23-24 Caribou Conservation Stewardship Program to predict and map nutritional resources during the summer season for 8 boreal caribou ranges in northern Ontario, extending west to east from Berens to Kesagami. Here, we present a user/field guide tailored to the study region that provides: (1) a description of the final nutrition prediction models and the approach used to develop the models, including the cross-walking steps to tailor it to the areas of model application; (2) a description of the caribou habitat use prediction model and its development; (3) a description of the approach used and assumptions of the nutritional implications map; (4) a description of the forage resources (species lists of accepted forage species) and abundance levels of key forage species that the different ecosites provide to caribou, including tables, figures, and photographs, with emphasis on the effects of forest management on caribou forage and nutrition; and (5) a discussion of practical strategies to manage the forage base on behalf caribou in light of other conservation considerations.

The views expressed in the publication are the views of the authors and do not necessarily reflect those of the Ontario Province.

About NCASI:

NCASI (National Council for Air and Stream Improvement, Inc.) is a non-profit environmental research organization that seeks to create credible scientific information required to address the environmental information needs of the forest products industry in North America. NCASI conducts surveys, performs field measurements, undertakes scientific research, and sponsors research by universities and others to document the environmental performance of industry facility operations and forest management, and to gain insight into opportunities for further improvement in meeting sustainability goals.

Field Guide: Mapping Caribou Forage Resources in Ontario

EXECUTIVE SUMMARY

Over the past two decades, a growing body of research has established that nutritional limitations, particularly during summer, are pervasive and often exert important influences on populations of large ungulates in North America. In 2009, NCASI initiated a research program aimed at improving understanding of how nutritional resources influence caribou populations and how forest management can influence these resources. The research program featured a novel approach: deploy captive, trained lactating caribou and their calves in temporary electrified pens constructed in native habitats to evaluate the relationships between fine-scale forage attributes (e.g., forage abundance, composition, and quality) and caribou foraging dynamics and nutrition (e.g., dietary preferences, the digestible energy and protein content of diets and forage and nutrient intake rate). Such data represent a substantial advancement over simply inferring nutritional value of habitat from forage quality and quantity surveys by actually measuring their nutrition in these habitats. Four years of data were collected in the mountains and boreal forests of northeastern British Columbia (2013–2016) and 2 years were in the boreal forests of Northwestern Ontario (2017–2018), both using captive caribou to evaluate nutritional resources.

In 2024, NCASI received a grant from Ontario's 23-24 Caribou Conservation Stewardship Program to use the Ontario data to predict and map forage resources during the summer season for 8 boreal caribou ranges in northern Ontario, extending west to east from Berens to Kesagami. We developed: (1) maps of nutritional resources, known as "foodscape maps," that provide a spatially-explicit depiction of the variation in nutritional resources across our region of mapping; and (2) a map of caribou habitat use primarily as a function of variation in nutritional resources and predation risk. In addition, because maps of forage and nutritional resources alone may be difficult to interpret relative to management and conservation applications, we developed 2 raster layers to aid interpretation that rate each pixel according to three criteria to evaluate nutritional implications to caribou: (1) nutritional adequacy of forage intake rate specifically in relation to nutritional requirements of lactating females and their calves in summer; (2) expected changes in current nutritional resources with the addition or in the absence of stand-replacing disturbance; and (3) probability that stand-replacing disturbance and resultant forage development in early seral communities will be beneficial for moose and thus detrimental to caribou. The mapping products are expected to (1) aid in the assessment of the current state of nutritional resources particularly to identify high-quality areas that might be earmarked for protection from development or where nutritional improvements may be of value for improving poor nutritional resources, and (2) aid in evaluating changes over time such

as forecasting changes in nutritional resources as a function of stand-replacing disturbance and human-based developments.

Here, we present a user/field guide tailored to the study region that provides: (1) a description of the final nutrition prediction models and the approach used to develop the models, including the cross-walking steps to tailor it to the areas of model application; (2) a description of the caribou habitat use prediction model and its development; (3) a description of the approach used and assumptions of the nutritional implications map; (4) a description of the forage resources (species lists of accepted forage species) and abundance levels of key forage species that the different ecosites provide to caribou, including tables, figures, and photographs, with emphasis on the effects of forest management on caribou forage and nutrition; and (5) a discussion of practical strategies to manage the forage base on behalf caribou in light of other conservation considerations. These considerations include comments about changes in predation risk as a function of roads and linear features, and concerns about improving the forage base too much, such that moose and their predators are favoured to the detriment of caribou (i.e., seeking balance between modest improvements in the forage base for caribou versus large increases in forage that support increased density of moose and their predators).

The products resulting from this project offer a landscape planning tool to help identify how forage, caribou nutrition, and habitat use patterns may change over time as a function of forest management actions or wildlife and help identify actionable forest management options to enhance caribou forage across the landscape in northern Ontario.

KEYWORDS

boreal caribou, disturbance, forage quantity, forage quality, forest management, management, northern Ontario, nutrition, nutritional ecology, nutritional resources, predation risk, *Rangifer tarandus*

INTRODUCTION

The negative influences of predation and anthropogenic disturbances on boreal caribou (*Rangifer tarandus*) populations are well recognized across Canada. This is largely because increasing anthropogenic disturbances may increase rates of predation, and this pathway is believed to be a primary cause of declines of boreal caribou (Festa-Bianchet et al. 2011, Hervieux et al. 2014, Serrouya et al. 2017, Fryxell et al. 2020). Nevertheless, various environmental factors that affect health and performance can influence long-term sustainability of caribou and can be important not only in areas where anthropogenic disturbance remains modest, but also in areas where these disturbances are common. After all, in areas where predation and other risks are elevated, the need for good health and performance is at least as important as in areas with lower risk of predation and other causes of mortality.

Foremost among these environmental factors are nutritional influences. Substantial advances have occurred in the last two decades primarily with elk (*Cervus canadensis*), moose (*Alces alces*), bighorn sheep (*Ovis canadensis*), and deer (*Ococoileus* spp.) in understanding how nutrition, and the forage attributes that affect nutrition, in turn impact health, performance, survival, and reproduction. These advances have altered long-held perceptions of how inadequate forage may influence herbivore populations. A pervasive assumption of the past was that nutritional limitations were a consequence of winter severity when most plants are dormant and often covered in snow (Wallmo et al. 1977, Mereszczak et al. 1981, Houston 1982, Leege 1984, Skogland 1990, Christianson and Creel 2007). However, accumulating evidence clearly indicates that nutritional limitations also occur in summer for a variety of large ungulate species in the northern ecosystems around the globe (Hjeljord and Histol 1999; Peek et al. 2002; Cook et al. 2013, 2018, 2021*c*; Hurley et al. 2014, Monteith et al. 2015; Proffitt et al. 2016; Rolandsen et al. 2017; Horne et al. 2019; Schrempp et al. 2019; Stephenson et al. 2020). During late spring and through the growing season, key periods of animal production occur, including the third trimester of gestation, birthing, calf-rearing, juvenile growth and development, restoring endogenous energy reserves used the previous winter and spring, and breeding in autumn. Requirements for digestible energy and protein for adult females raising juveniles to support many of these processes are two-to-three-times greater than in winter (Oftedal 1985, Cook et al. 2004, National Research Council 2007). Nutritional deficiencies stunt animal performance and include reduced (1) rates of body fat accrual during summer and autumn; (2) probability and timing of ovulation and breeding; (3) body condition at the beginning of winter; (4) neonatal vigor and size; (5) juvenile growth, development, and survival; (6) growth and development of subadults; and (7) age at first breeding (Verme and Ozoga 1980; Crête et al. 1993; Cook et al. 1996, 2004; Tollefson et al. 2010).

Evidence indicating summer nutritional limitations for caribou is also accumulating in the far north in Alaska (Post and Klein 1999, Cameron et al. 2005, Dale et al. 2008, Johnson et al. 2022) and Canada (Pachkowski et al. 2013, Schaefer and Mahoney 2013, Schaefer et al. 2016). In woodland caribou ranges in western Canada, Denryter et al. (2022) reported that quality, and particularly quantity, of acceptable forage in many habitats in boreal and montane settings in northeastern British Columbia provided levels of nutrition that routinely failed to satisfy nutritional requirements of lactating female caribou and their calves in summer. Heard and Zimmerman (2021) reported that supplementing wild caribou diets with a high-quality food in autumn boosted populations (lambda) in the mountains of central British Columbia across a range of predator densities. In addition, of 61 bones from female carcasses in western British Columbia evaluated by McLellan et al. (2012), 30% had marrow fat between 20–70% (which equates to 1 to 6% body fat; Cook et al. 2021b), indicative of animals susceptible to predation, parasites, or disease (Mech and Delgiudice 1985, Bender et al. 2008, Sand et al. 2012). Another 15% had marrow fat <20% (<1% body fat; Cook et al. 2021b), which indicates starvation (Ratcliffe 1980, Peterson et al. 1984, Mech and Delgiudice 1985, Depperschmidt et al. 1987, Sand et al. 2012).

In 2009 and 2010, we conducted surveys of body condition of caribou across Ontario (Fig. 1) by measuring body fat levels of caribou captured to deploy GPS collars in February and March using validated ultrasound and body condition scoring techniques (Cook et al. 2021*b*). Estimates of body fat provide valuable insight into the adequacy of nutrition experienced by the animal over the previous several months and insight into how well the animal is likely to perform over the upcoming months (Keech et al. 2000; Cook et al. 2004, 2013, 2021*a*; Monteith et al. 2013, 2014; Stephenson et al. 2020; Cook et al. 2021*c*). Results for Ontario caribou illustrate a 2-fold range in body condition among populations across Ontario, which span the range in variation that we have documented among a variety of populations in western Canada (Fig. 1). These results suggest that the nutritional environment varies markedly among ecological settings.

The seasonality of nutritional limitations is a question with key implications for developing strategies to evaluate and manage nutritional resources, and it has long been thought that limitations in winter, not the growing season, are of primary interest (e.g., Wallmo et al. 1977, Mereszczak et al. 1981, Houston 1982, Leege 1984, Skogland 1990, Christianson and Creel 2007). However, despite the harsh winter conditions that prevail on most caribou ranges, caribou in many populations appear to be remarkably resilient to loss in body condition across winter. Couturier et al. (2009) reported that body condition as measured by body fat levels in some populations may even increase in winter and concluded that nutrition during summer should receive greater emphasis. In the Northwest Territories, Cook et al. (2021a) found that loss of body condition in winter was comparatively minor and depended on body fat levels at the end of autumn. Caribou that exited autumn with low levels of condition tended to exit winter with no loss in condition or, as reported by Couturier et al. (2009), even gained condition over winter; caribou that exited autumn with higher levels of body fat lost relatively minor amounts of body fat over winter.

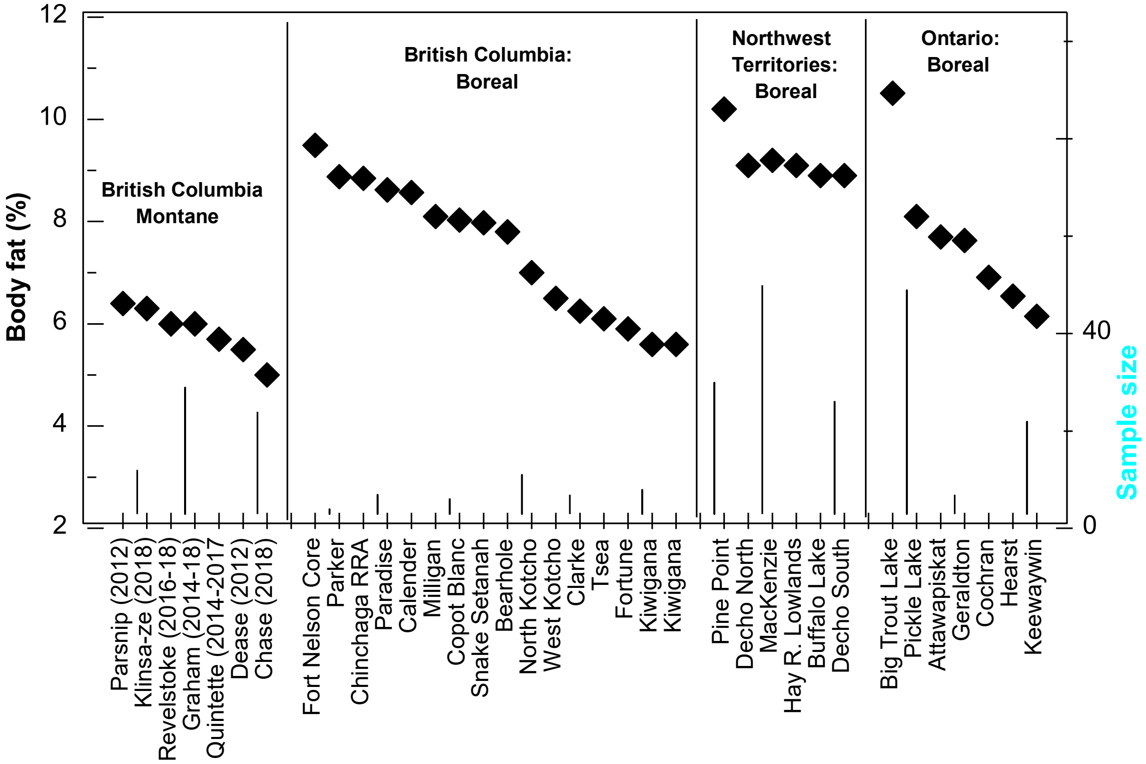


Figure 1. Estimates of body fat (%) determined using rump fat thickness and a body condition score (Cook et al. 2021b) of adult female caribou captured during February and March in western Canada and northern Ontario (2009–2018). Black diamonds indicate mean body fat levels for each population and bars indicate sample sizes.

Alternatively, animal production processes during the growing season substantially increase nutritional needs, e.g., nutritional demands of lactation and juvenile growth during summer increase nutritional requirements 2–3 fold compared to winter (Oftedal 1985, Cook et al. 2004, National Research Council 2007). Nutritional deficiencies during the summer season can reduce body fat levels and body size, breeding success, neonatal vigor and size, juvenile growth and survival, subadult development, and age at first breeding, and these deficiencies can increase susceptibility to predation, disease, parasites, and weather extremes. New research indicates that nutritional limitations during the growing season are pervasive across many ungulates ranges across the continent and recognition is increasing for the need to prioritize animal nutrition, health, and forage adequacy during this season for North American ungulates, including caribou (Dale et al. 2008, Couturier et al. 2009, Cook et al. 2021a).

Annual cycles of body condition for a number of caribou populations suggest minor losses over winter in most populations and sharp declines between early spring and early summer, with the annual minimum by early summer (Fig. 2). This minimum in body condition during summer coincides with a pattern of maximum mortality rates of adult female caribou, at least in western Canada (Wittmer et al. 2001, McLoughlin et al. 2003, Kelly 2020, Cook et al.

2021*a*). In the Northwest Territories, Kelly (2020) found that starvation was a primary cause of death (40% of mortalities) during summer, suggesting that the low levels of body condition in summer, likely a result of inadequate nutrition and other factors at that time, may substantially account for the levels of elevated mortality. Thus, the levels of body fat that they accrue by mid-autumn strongly influence the levels of body fat they carry through winter and for most of the annual cycle. Late spring through summer evidently is the season of nutritional limitations that matter, and nutrition during summer through early autumn may disproportionally affect animal productivity and the level of body reserves of energy and protein that caribou carry through winter well into the following spring.

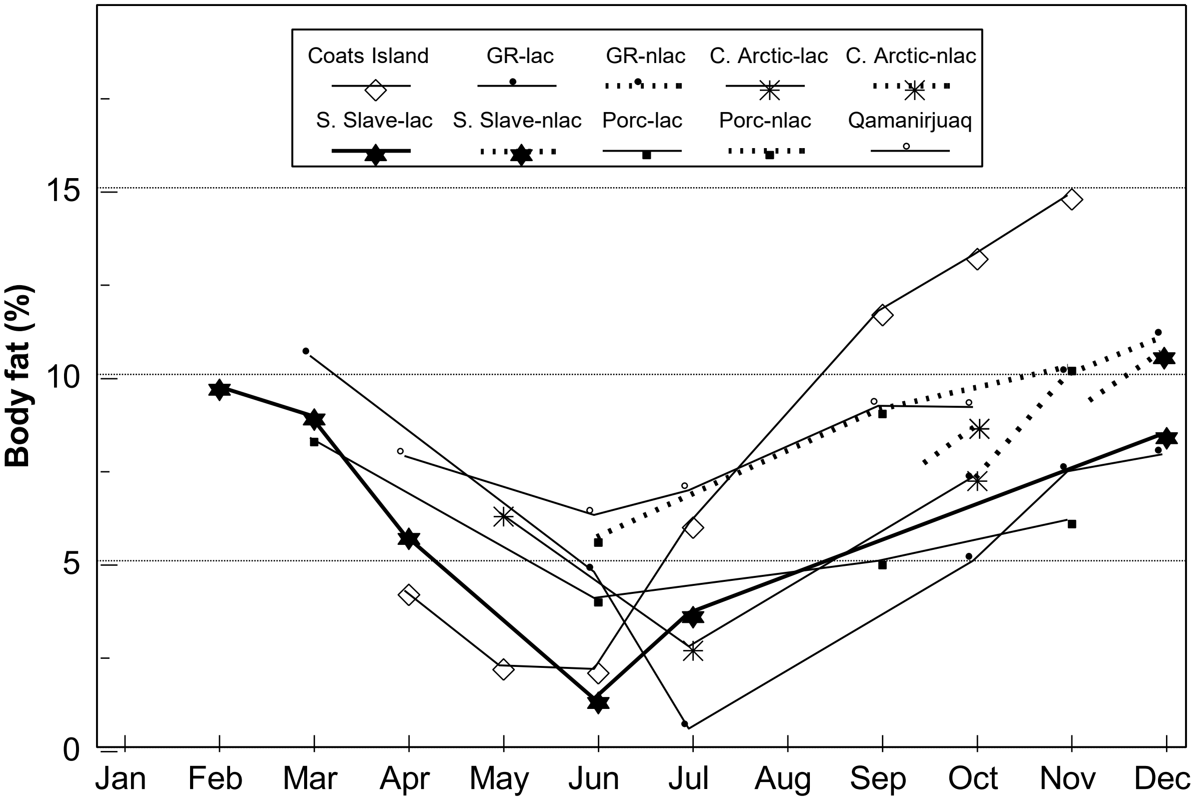


Figure 2. Percent fat of the ingesta-free body of adult females reported for boreal caribou in the in the southern Northwest Territories (S. Slave) and barren-ground caribou for the Coats Island population in northern Hudson Bay (Adamczewski et al. 1987); George River population (GR) mainly in northern Quebec (Huot, 1989; Crête and Huot 1993); the Central Arctic population (C. Arctic) in Alaska (Table 2 in Gerhart et al. 1996); the Porcupine (Porc) population in eastern Alaska and western Canada (Table 1 in Chan-McLeod et al. 1999); and the Qamanirjuag population in eastern NT and northern Manitoba to Hudson Bay (from Fig. 15 [prime age caribou] in Dauphiné 1976). Data for the southern Northwest Territories were collected as described in this study; all other data sets were from harvested caribou. Where available, data for lactating (lac) and non-lactating (nlac) females in summer and autumn within populations are presented. Reprinted from Cook et al. 2021a.

In 2009, NCASI initiated a research program aimed at improving understanding of how nutritional resources influence caribou populations and how forest management can influence these resources. The research program featured a novel approach: deploy captive, trained lactating females and their calves in temporary electrified pens constructed in native habitats to evaluate the relationships between fine-scale forage attributes (e.g., forage abundance, composition, and quality) and caribou foraging dynamics and nutrition (e.g., dietary preferences, the digestible energy and protein content of diets, and, most importantly, forage and nutrient intake rate). This approach provided new insights into the forage attributes directly influencing nutrition and provided quantitative equations for predicting caribou nutrition at fine scales. Findings from this work contributed additional evidence that nutritional limitations arising at fine scales during summer are an important consideration for caribou conservation (Denryter et al. 2022). In addition, equations constructed using this captive animal approach were used for landscape nutritional evaluations for elk in the U.S. (Cook et al. 2016; Rowland et al. 2018), for caribou in British Columbia (Denryter et al. 2022), and recently for caribou in Ontario (Walker 2023). These efforts demonstrated that metrics of forage resources, when quantified at fine scales using captive animal foraging trials in native habitats, provide an effective basis for describing forage resources across landscapes, in ways that are significantly related to habitat use (Rowland et al. 2018; Walker 2023) and animal performance (i.e., body condition levels, pregnancy rates) (Cook et al. 2018). In all these studies (in addition to two ongoing studies on elk), nutrition metrics derived from captive animal foraging trials significantly outperformed metrics of forage quantity, forage quality, or remote sensed greenness indices.

During late spring through early autumn in 2017 and 2018, as part of a collaborative effort between NCASI ungulate researchers (Drs. John Cook and Rachel Cook), the University of Alberta (Dr. Evelyn Merrill and her student at that time, Phil Walker), and with collaborative support from Dr. Art Rodgers, Ontario Ministry of Natural Resources and Forestry, field studies were undertaken to evaluate nutritional resources for caribou in Ontario that featured two aspects: (1) surveys of forage quality and quantity in Northwestern Ontario (Red Lake south to Dryden and northeast to Pickle Lake) and Northeastern Ontario (north and west of Cochrane to the Québec border), and (2) sampling of fine-scale nutrition of caribou using captive caribou placed in native habitats in the Northwestern Ontario study area. Portions of the overall data set were successfully used to evaluate habitat use and influences of nutritional resources on female caribou in summer for the Kinlock, Nipigon, and Kesagami caribou populations (Walker 2023). For this effort, landscape maps of nutritional resources and habitat use were constructed using the captive caribou foraging data and data sets of forage quality and quantity. Here, we extend this effort to map nutritional resources and habitat use for the boreal caribou range across northern Ontario.

Using the data sets of forage quality, quantity, and caribou nutritional responses from Ontario, our project aims to assess forage resources during the summer season for boreal caribou ranges across most of northern Ontario. Our primary objectives were to develop (1) spatially explicit maps of nutritional resources, known as "foodscape maps," by utilizing predictive models for nutritional resources; and (2) maps of caribou habitat use primarily as a function of nutritional resources. These maps depict areas with abundant and high-quality forage that supports high nutrition for caribou and areas with limited and poor-quality forage that supports poor nutrition and also show the probability of caribou use as a function of these forage attributes. Our approach explicitly emphasized the effects of forest disturbance, primarily stand-replacing wildfire and timber harvest, on forage abundance, composition, and caribou nutritional responses. It provides a basis to identify how forage, caribou nutrition, and habitat use patterns may change over time as a function of forest management actions and to help identify actionable forest management options to enhance caribou forage across the landscape.

Herein, we present methods used to develop foodscape maps, with emphasis on the nutritional relationships used to develop our approach. Much of the information we provide is focused on differences in nutritional resources among ecosites and changes in these resources as a function of stand-replacing disturbance and subsequent succession, reflecting the considerable differences in nutritional resources as a function of site potential (a function of biogeoclimatic influences that together affect vegetation) and its interaction with disturbance and succession. We present photographs, forage resource descriptions, and management considerations for each of the common vegetation types (hereafter referred to as ecosites) in our study areas. We then present a series of foodscape maps and provide summaries of nutritional resources for our mapping region. Our objective was to summarize our findings and insights in a “field guide” that will be useful for wildlife biologists, land managers, and the interested public. Finally, we present the details of developing the maps, with the intent that users in the future will be able to take our products, modify to reflect future changes in vegetation, and to explore how caribou nutritional resources are likely to change as a result of planned or projected changes in habitat conditions.

FUNDAMENTAL CONCEPTS OF NUTRITIONAL ECOLOGY AND HOW TO MEASURE IT

The attributes of forage that are important for affecting caribou nutrition and body condition and how to measure forage in ways that are meaningful to the animal are fundamental questions important for identifying the nutritional value of different types of plant communities for caribou. The composition of plant species (that is, the relative abundance of different plant species in plant communities), their abundance, their nutrient content, and their level of plant defensive compounds all can have important effects on the ability of caribou to acquire and satisfy their nutritional needs in the summer. These can vary greatly among ecosites and seral stages. In general, the combination of these attributes defines “nutritional resources”, and nutritional resources in any given plant community type may variously affect the level of nutrition that caribou are able to obtain from the community. The level of nutrition can range from very poor to excellent, and the term “nutritional value” refers to the level of nutrition caribou can obtain in relation to the amount of nutrients they need each day to satisfy their requirements. If nutritional value is low and remains low for extended periods of time, then performance of the animal is impacted, e.g., particularly for summer, the ability of the mother to provide milk and build body fat levels prior to winter is impacted, her ability to survive and successfully reproduce may be reduced, and the survival, growth, and development of her calf may be reduced. Nutritional deficiencies at a level that significantly impacts performance of females and their calves are common in many areas of North America. In our efforts to measure nutritional value, several key elements were important.

First, determining which plant species are palatable for caribou and which they avoid or will not eat is a crucial first step in describing forage resources in a manner relevant to foraging caribou. Much of the available plant material within reach of caribou is too fibrous, too high in plant defensive compounds, or too low in nutrients to be useful forage for caribou in summer. Based on foraging research extending back to 2000 using tame animals (first with elk, then deer, and finally with caribou after 2012), our work has demonstrated that 40% to 95% of the vegetation biomass within reach of these animals is typically composed of plant species that should not be considered food (Cook et al. 2016, Denryter et al. 2017, Ulappa et al. 2020).

We sampled in detail the diet composition, foraging efficiency, and nutrition of captive caribou and their calves that were transported to 79 electrified enclosures (pens) placed in native habitats in Northwestern Ontario during summer and early autumn in 2017 and 2018. We distinguished useful forage from non-forage by measuring the proportion of each plant species in the caribou diets and compared this to the proportion of each plant species available to the caribou in the habitats in which they foraged. This was calculated for each plant species across all enclosures with caribou, providing a measure that spanned the variety of habitats included in our overall sample. For example, if willow overall comprised 50% of their diet, but comprised only 5% of the plant species in the in the enclosures, this indicates that caribou selected for willow; if Labrador tea comprised 1% of the diet, but comprised 30% of the species available to them, then this indicates that the caribou avoided this species. The calculations require that estimates of proportion observed in animal diets and proportion available to the foraging animal are both sampled in the same location at the same time. The rigor and reliability of such data is greatly enhanced using captive animals placed in native habitats.

In practice, we separated plant species into two classes: accepted (defined as plant species that caribou either significantly selected or ate in the same proportion that was available to them in their pens) and avoided (defined as plant species which caribou did not eat or significantly avoided) (Appendix A). Over the years based on studies for elk, deer, and caribou, our data analysis indicated forage evaluations that focused on accepted species provided better correlations between nutritional response of the animal and the available vegetation than did evaluations based on total forage (i.e., that included avoided species). We focus on accepted forage throughout this report.

Second, two general attributes are important from a caribou nutrition perspective: how fast they can eat on a per minute and daily basis and the nutrient content (digestible energy and digestible protein for our work) of the forage that they consume. Regarding the former, animals in summer have 12 to 15 hours a day they devote to foraging (the rest of the time they spend resting, ruminating, caring for young, and being vigilant). The size of bites (usually measured in grams per bite) of accepted forage has a dominant effect on how fast caribou can eat (Spalinger and Hobbs 1992, Shipley 2007). During our caribou foraging trials, bite size ranged from 0.0005 grams to 3.54 grams per bite. At the biting rate (i.e., number of bites taken per minute) that is typical for caribou and assuming 15 hours of day spent foraging, simple calculations show that caribou must focus on accepted plants providing bite sizes of at least 0.28 grams (a conservative estimate) if they are to consume sufficient forage each day to satisfy their requirements (Fig. 3). Caribou normally cannot consume sufficient forage each day if their bite size is less than about 0.15-0.2 grams per bite, and at this level, important nutritional deficiencies for lactating females are likely (Denryter et al. 2022).

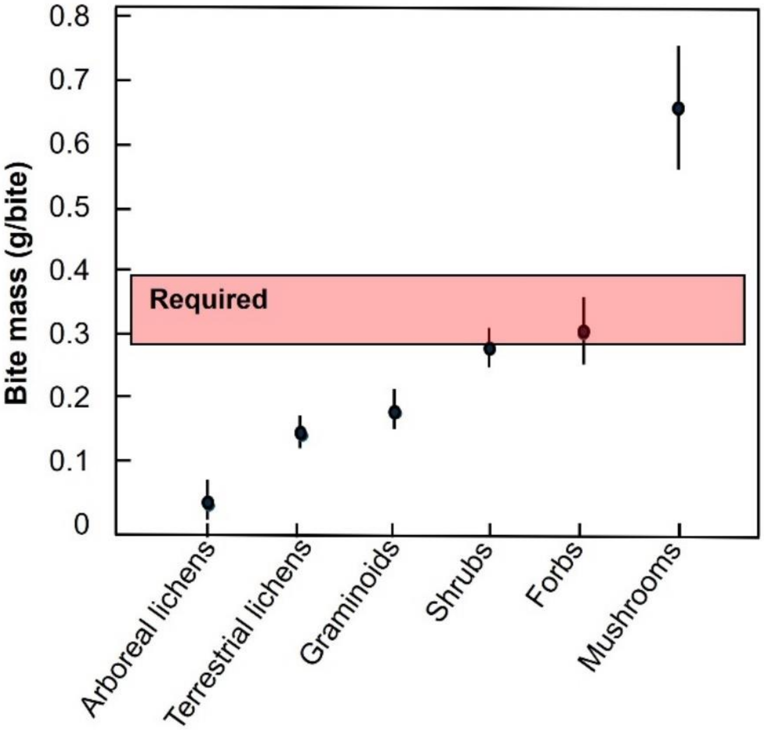


Figure 3. Bite size (mass ± 95% confidence intervals) of caribou for six forage classes based on data collected from captive caribou sampled in native habitats in Northwestern Ontario in summers of 2017 and 2018. The horizontal line indicates the approximate bite size required to satisfy daily forage intake requirements of a lactating female in summer if she forages for 15 hours. In other seasons when nutritional needs are lower, the bite size requirement would be lower.

How fast animals can eat also is a function of the abundance of accepted food. Multiple studies have demonstrated this for elk (Wickstrom et al. 1984), as have studies for caribou (Trudell and White 1981) and our own work in British Columbia and Ontario (Fig. 4).

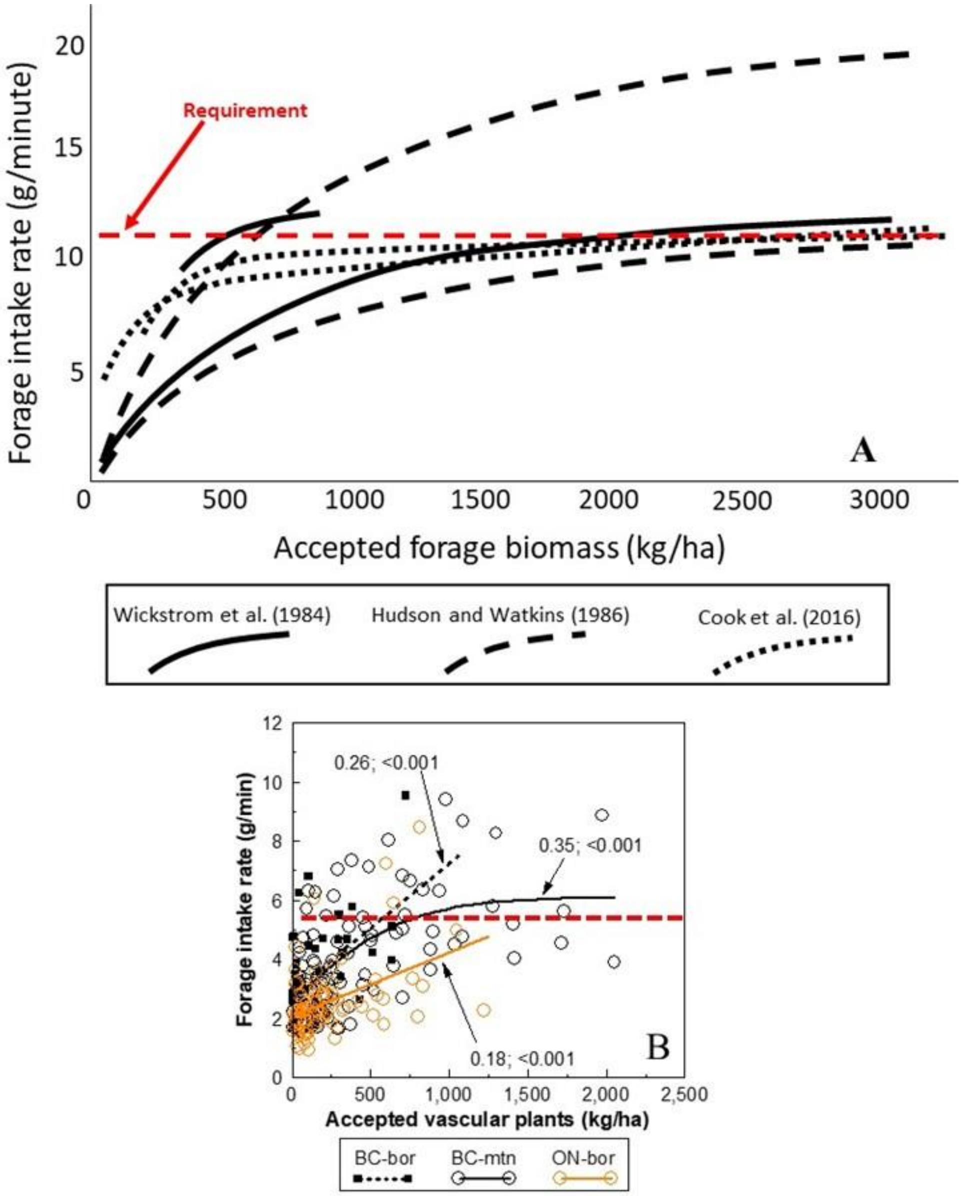


Figure 4. Forage intake rate as a function of biomass of accepted forage species for (A) elk and (B) caribou at 3 study areas: BC-bor = boreal forest in northeastern British Columbia (BC); BC-mtn = montane area in northeastern BC; and ON-bor = boreal study area in Northwestern Ontario. The red horizontal dashed line represents approximate forage intake rate required to satisfy daily forage consumption requirements for lactating females in summer assuming a 15-hour foraging day, and for caribou, digestible energy levels of 12.1 kJ/g in consumed forage (National Research Council 2007, Denryter et al. 2022).

The relationship between how fast animals can eat and biomass of forage is referred to as a functional response (Spalinger and Hobbs 1992). Accepted forage that is relatively sparse slows how fast they eat because animals may spend time inefficiently searching for and traveling to patches where this forage occurs. In practice, the relationship between forage biomass and forage intake rate can be highly variable, but based on the available information, when accepted forage biomass falls below about 250-300 kg/ha, and certainly below about 150-200 kg/ha, forage intake rates average markedly below rates that are required for the animal to satisfy daily forage intake requirements, and nutritional deficiencies are likely to occur. Thus, biomass of accepted species is a key variable relating the nutritional value of habitats to caribou in settings where biomass of accepted species commonly ranges below about 250 kg/ha.

Typically, nutritional ecologists focus on digestible energy and digestible protein when evaluating the nutritional adequacy of forage, because these nutrients most often are limiting to animal performance (Van Soest 1994). The nutrient content of forage that animals consume has two primary effects on animal nutrition. The first is a simple direct effect. The higher the concentration of digestible energy and digestible protein in the diets of herbivores, the more of these nutrients they consume per kilogram of forage they eat. But often more importantly, the higher the fiber content in forage, the lower the digestible energy and digestible protein and other nutrients in forage, the slower caribou can digest their food, and this reduces the amount of forage they eat each day (Robbins 1983, Weston and Poppi 1987, Minson and Wilson 1994, Owen-Smith 2002, Cook et al. 2004). Thus, to a large degree, abundant forage of relatively low quality does not compensate for sparse forage of high quality (Owen-Smith 2002, Cook et al. 2004, Hanley et al. 2012).

Over the five years of work with captive caribou in Canada (three years in British Columbia and two years in Ontario), we have found that forage quality levels during summer tend to be higher in the far north than in areas further south in the western U.S. (e.g., Cook et al. 2016), suggesting forage quality may be less important of a factor contributing to depressed animal performance in the far north. Instead, to a large degree, the low levels of accepted forage biomass that reduces how fast they can eat likely has a greater negative influence on caribou nutrition. Hence, our focus here for evaluating forage resources is more on biomass, particularly of accepted forage, and how rapidly caribou can eat.

The value of lichens as a key food of caribou is well recognized, but the supportive role of lichens for caribou nutrition during summer is less clear. Biomass of accepted vascular plants, particularly accepted deciduous shrubs, is well correlated to how fast caribou can eat largely because many of the plants in this forage category provide relatively large bite size that contributes to high rates of forage intake (Fig. 3). These plants also typically provide relatively high levels of digestible protein. However, the value of lichens as a summer forage for caribou may be marginal, because lichens contain virtually no digestible protein and because bite size is low, typically averaging from 0.04 to 0.2 grams per bite (Fig. 3). In general, we found very little positive correlation between lichen abundance (either arboreal or terrestrial lichens) and forage intake rates (Fig. 5). And our final equations to predict forage intake rate during summer exclude lichens, because lichen biomass failed to enter the equations during the model selection process. These results indicated that for caribou in summer, good forage resources involve substantially more than abundant lichens that are common in late seral communities. Nevertheless, lichens are high in digestible energy that is certainly of value in summer, and the high levels of digestible energy in lichens are generally unmatched in winter. Thus, we avoid diminishing the importance of this high-quality food source, particularly in autumn after vascular plants senesce until the following spring when growth of vascular plants begins.

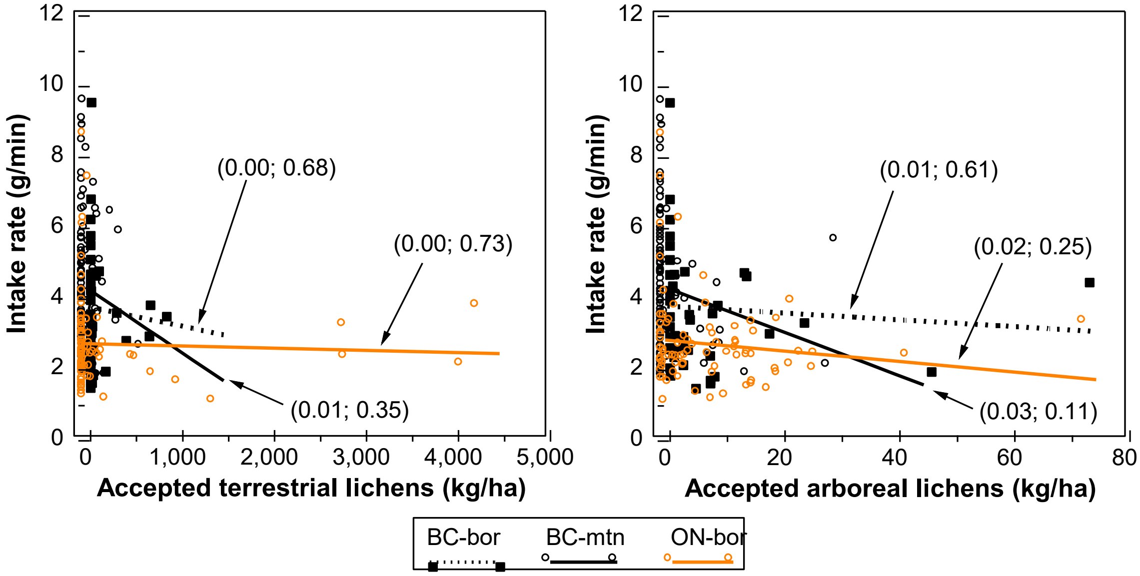


Figure 5. Relationships (linear regressions) between lichen abundance and per minute intake rate of forage by lactating female caribou in Ontario boreal forests (ONT – boreal), boreal forests in British Columbia (B.C. – boreal), and montane forests in British Columbia (B.C. – montane). The "Required" line indicates the approximate rate of forage intake required to satisfy daily forage and nutrient intake requirements for lactating female caribou in summer. Data were collected during summer using captive caribou from 2013-2016 in northeastern British Columbia and captive caribou from 2017-2018 in Northwestern Ontario. These results indicate that either no significant relationship (N.S.) or an inverse relationship exists between lichen abundance and forage intake rates.

Our mapping effort described herein includes mapping those forage attributes that were found to contribute to forage intake rate of captive caribou in our foraging trials: accepted biomass of grasses, forbs, and deciduous shrubs (combined), horsetails (in the genus *Equisetum*), and mushrooms. From these forage attributes, we then predicted per-minute forage intake rate, and used forage intake rate as a key metric to represent nutritional influences on habitat selection and our maps representing nutritional adequacy. We also mapped lichen abundance primarily to identify areas that are valuable as winter foraging areas. The equations were developed for uplands and lowlands separately and were described in detail by Walker (2023).

We also predicted and mapped forage intake rate (g/min) to provide a basis for identifying the actual level of nutrition that caribou may acquire from different plant communities given the types and abundance of forage present in the community. Simple estimates of forage biomass or other factors potentially perceived to be surrogates of caribou nutrition by themselves introduce considerable uncertainty about the actual nutritional value of plant communities to foraging caribou (Searle et al. 2007). Because many of our models to predict caribou nutrition contained date as a covariate (due to changes in DE and DP in plant species over the growing season), we set our predictions to 1 Aug and 15 Sept for the purposes of this guide to facilitate comparisons among plant community types and in response to disturbance and succession. These dates were chosen to reflect important time periods relative to the phenological stage of vegetation and relative to nutritional requirements of accruing body fat prior to winter in female caribou (Cook et al. 2021a).

STUDY AREA DESCRIPTION

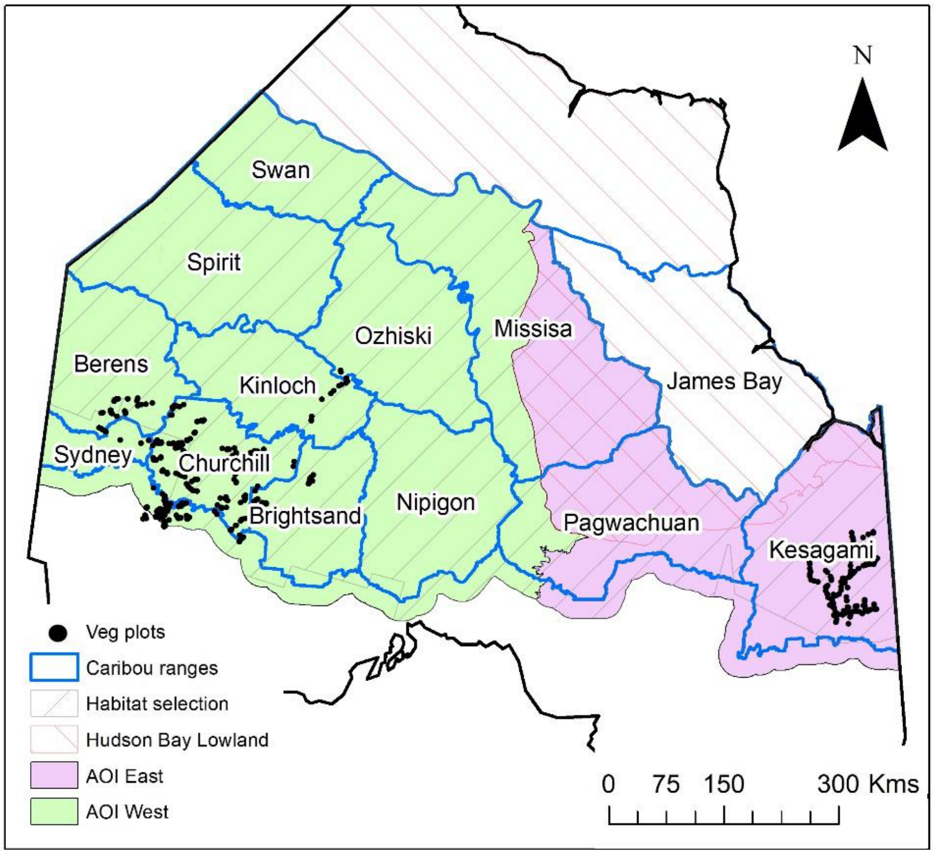


Figure 6. Area of interest (AOI) across woodland caribou ranges in northern Ontario (green and purple shaded areas). The slightly smaller area identified as 'habitat selection' on the map is the modified AOI where habitat selection will be predicted due to a more limited coverage of required GIS layers. Black dots indicate the spatial locations of the 508 field vegetation macroplots across northwestern and Northeastern Ontario on which the forage resources maps are based.

Our vegetation sampling was conducted in Dryden within the Boreal Shield of Northwestern Ontario and in Cochrane in the Northern Clay-Belt region of Northeastern Ontario (McMullin et al. 2013, Thompson et al. 2015; Figure 6). Cochrane has low topographic variation, extensive lowlands, and a relatively low fire frequency (~120-year fire-cycle; Baldwin et al. 2000; McMullin et al. 2013, Thompson 2000, Walker et al. 2021), whereas Dryden is a fire-prone (~65-year fire-cycle) area in with greater topographical variation and a diversity of forested communities (Baldwin et al. 2000; McMullin et al. 2013, Thompson 2000, Walker et al. 2021). Forested communities within each region consist of black and white spruce (*Picea mariana* and *P. glauca*), Jack pine (*Pinus banksiana*), balsam fir (*Abies balsamea*) trembling aspen (*Populus tremuloides*), and white birch (*Betula papyrifera*; Rowe 1972).

The two regions differ climatically; Cochrane is influenced by the maritime climate of Hudson and James Bay, reducing the number of growing-degree-days (Baldwin et al. 2000), with delayed phenology due to twice as much snow in winter compared to Pickle Lake (Walker et al. 2021). Based on 20-year averages (1991-2010), Cochrane had a greater total annual precipitation (824 ± 36 mm; x̄ ± SD, 20-yr average) compared to Dryden (736 ± 43 mm), whereas January daily temperatures (PL: -19.26°C ± 3.60°C; CO: -17.82°C ± 3.69°C) were slightly lower in Dryden, and July (PL: 17.66°C ± 1.49°C; CO: 17.36°C ± 1.25°C) daily temperatures were similar between areas (Environment and Climate Change Canada, https://climate.weather.gc.ca/historical_data/search_historic_data_e.html, accessed 14 June 2019).

Based on data from weather stations reporting both long-term normals and monthly data nearest our study areas, precipitation during 2017 and 2018 in Northwestern Ontario and 2018 in Northeastern Ontario (when data were collected) was generally below normal. In Northwestern Ontario, precipitation data from the Dryden station indicated very low precipitation in winter and spring of 2017 and roughly normal precipitation during summer in 2017, and very low precipitation during each of the three seasons in 2018. Drought conditions were less severe at the Kenora station, with recorded precipitation even above normal during summer in 2017 (Fig. 7). The Dryden station is more centrally located to our study area and probably provides a better measure of precipitation than the Kenora station. In Northeastern Ontario during 2018, both the Timmins and Moosonee stations indicated less precipitation than normal in all seasons, with the Moosonee station recording summer precipitation of about 25% of normal. The majority of the study area was located mid-way between the two stations, suggesting that the average of data from both stations may be equally applicable to our study area in Northeastern Ontario. Thus, in general, the data indicate drier conditions than normal on both study areas winter through summer. Temperature differences from normal recorded at both the Dryden and Kenora stations indicated a warmer winter in 2017 than 2018, and similar temperatures compared to normals in spring and summer at both weather stations. At our northeastern study area, temperatures during each season were similar to long-term normals (Fig. 7).

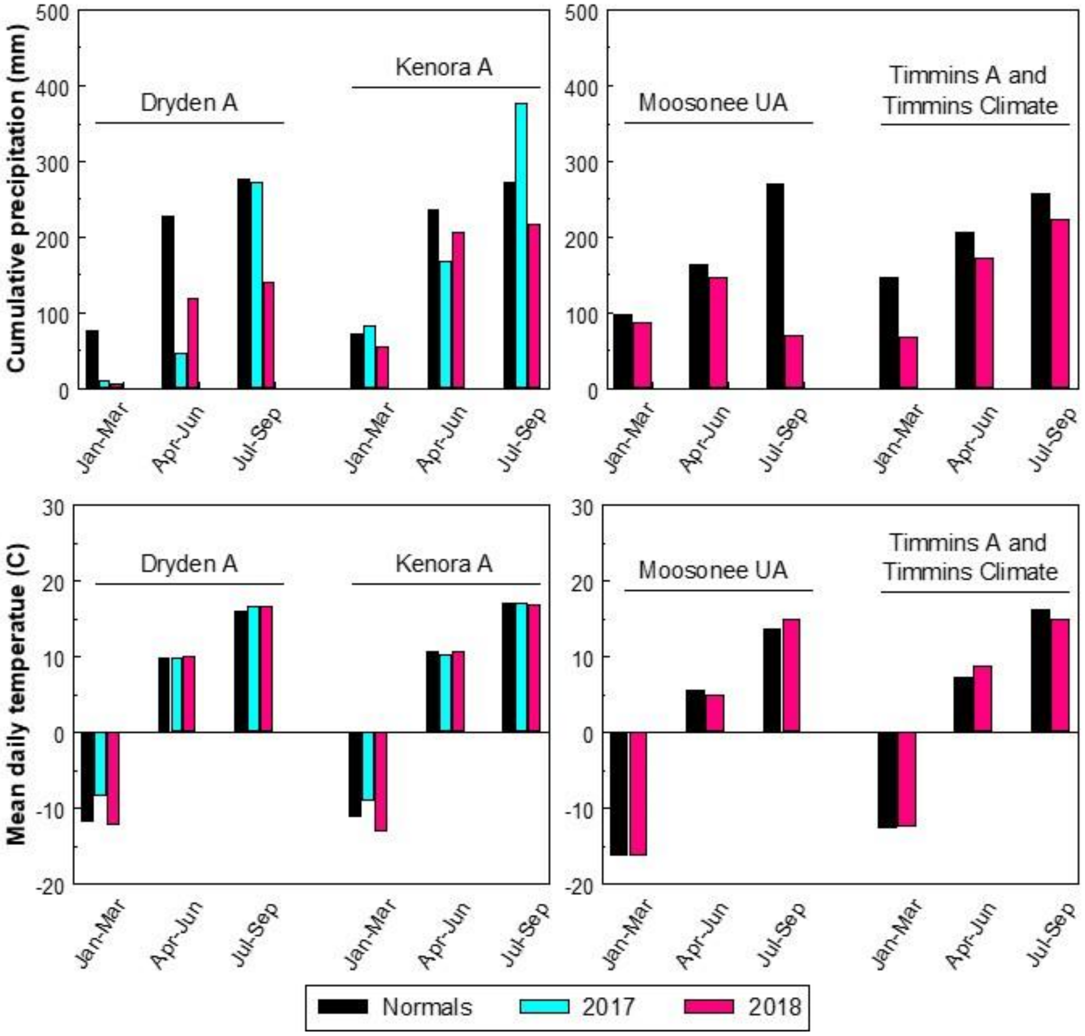


Figure 7. Long term normals (1991-2020) and seasonal cumulative precipitation and average temperature at two weather stations in Northwestern Ontario (Dryden A and Kenora A) and Northeastern Ontario (Moosonee UA and Timmins A (for normal) and Timmins Climate (for recent seasonal data). Data are presented for 2017 and 2018 in Northwestern Ontario, and 2018 only in Northwestern Ontario, matching the years data were collected.

METHODS

Plant Community Type (Ecosite) Classification Approach

Fundamental to our work is the vegetation classification system we used for evaluating caribou forage. The system we used reflects a decade of work mainly in British Columbia, the Northwest Territories, and Ontario and was relatively simple to apply and, critical for our work, was reasonably effective at grouping sites based on similar undergrowth vegetation composition. For

Ontario, we relied primarily on the ecosite classification system developed for Northwestern Ontario (Racey et al. 1996) but extended this approach to Northeastern Ontario using the ecosite descriptions of Taylor et al. (2000).

The ecosites identified by both classification systems were far too numerous to be practical for our purposes (Appendix B), from both a sampling perspective and for mapping purposes. Hence, we developed eight ecosite groups primarily based on similar undergrowth composition as described in the ecosite guides (Racey et al. 1996, Taylor et al. 2000). In addition, we strove to create groups that reflected site potential, i.e., a potential natural vegetation approach that classifies sites based on the soils and climate characteristics and the vegetation that occurs there (DeLong et al. 2011). Classification that is based on site potential recognizes that certain soils and climate will consistently support relatively specific types of plant communities at each stage of development after stand-replacing disturbance, e.g., sites that currently support old-growth bog with black spruce will eventually grow back to a bog with black spruce after stand-replacing wildfire or logging, given sufficient time. This strategy differs from vegetation categories merely representing current cover types that tend to be transitional and poorly related to the site conditions and undergrowth vegetation that might occur there at any point in time. The potential vegetation approach that reflects site potential is ecologically-based (DeLong et al. 2011), provides a better approach for identifying undergrowth vegetation communities of the site and their changes across succession after disturbance, and provides a superior basis for landscape planning than those based on overstory cover types (Haufler 1994).

For our work in Ontario, our approach recognized four upland forest types and four wetland types. This included upland black spruce (Upl-BS) on relatively coarse-textured, unproductive soils and upland black spruce-white spruce (Upl-BS-WS) on fine-textured, deep, and productive soils. We subdivided the upland black spruce type into 3 ecosite groups, one that occurred where rocky outcrops were abundant (Upl-BS-Rocky), one where sandy soils predominated (Upl-BS-Sandy), and a relatively mesic type where soils were sandy-loams (Upl-BS-Mesic). In lowlands (wetlands), we included poor swamps, bogs, fens, marshes, and thicket swamps, the latter dominated by tall shrubs with or without white cedar (Appendix B). These grouped ecosites form the basis for subsequent data analysis. However, for mapping purposes, cross-walking was required from our 8 ecosite groups to those vegetation types that were included in vegetation maps that we used for mapping (the Far North Layer, described below), because this layer did not include the ecosites described by Racey et al. and Taylor et al. (Appendix B).

Measuring Forage Quality and Quantity

We sampled vegetation in macroplots during 2 summers (2017, 2018) in Northwestern Ontario. In 2017, sampling was conducted between 15 June and 10 October. In 2018, we sampled vegetation between 15 June and 2 October. In Northeastern Ontario in 2018 (no sampling was conducted in 2017 at this study area), we sampled between 5 June and 27 August. We sampled overstory and undergrowth characteristics along 3 parallel, 46-m transects in 0.42 ha rectangular macroplots (Fig. 8) (Cook et al. 2016). We used identical techniques to sample vegetation in vegetation macroplots and enclosures where caribou were deployed (described below), except that 4 transects of varying length were sampled where caribou were used, instead of three, reflecting the larger size of the enclosures of the caribou enclosures.

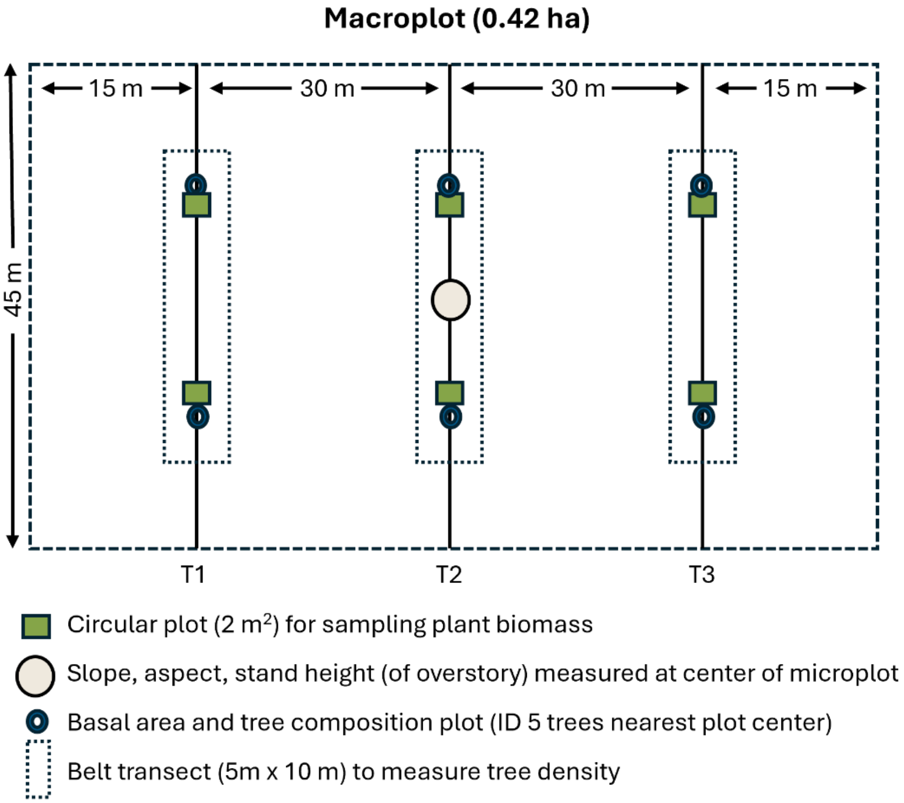


Figure 8. Sampling layout of macroplots and the vegetation attributes measured. Overstory canopy cover was measured at 10 equidistant points along each transect.

We emphasized sampling those ecosites that we assumed *a priori* to be of greater nutritional value to caribou and that were relatively common on our study area (e.g., Denryter et al. 2022), including all upland forest ecosites, bogs, swamps, and fens. We allocated reduced sampling to marshes and swamp thickets. Locations of macroplots were randomly selected *a priori* using a geographical information system in proportion to available forest types (upland conifer, mixed-wood, deciduous, bogs, and fens) and seral stages (early [<20 yrs] and mid-late [≥20 yrs], based on the Ontario eFRI) and within 300 m of a road. At each location, macroplots were located within a single, homogeneous stand and situated ≥ 50 m from stand edges. This approach tended to result in a proportional allocation among ecosites of sampling effort, and we bolstered our sampling for hard-to-find, uncommon communities (e.g., fens) to the extent we were able with opportunistic site selection.

We measured overstory canopy cover with a moosehorn (a vertical sighting tube) (Cook et al. 1995) at equal intervals, 10 per transect (Fig. 8). At two equally spaced points per transect, we measured basal area (m^2/ha) with a Cruz-All (JIM-GEM Forestry Suppliers, Inc., Jackson, MS), and identified species of the 10 trees (> 1 cm diameter at breast height [DBH] and a component of the overstory layer) nearest to the sampling point ($n = 60$ trees/macroplot) from which we calculated composition of tree species (Fig. 8). We estimated density (stems/ha) of seedlings (< 1 cm DBH) and trees by counting each in 5 x 15-m belt transects (2.5 x 15-m in very dense forest stands), with one belt transect per sample transect (Fig. 8). We measured mean stand height, slope, and aspect at the center of the macroplot using a clinometer and compass (Fig. 8). We estimated the average age of trees in each macroplot from a core sample from the largest tree that was representative of the stand (i.e., we excluded unrepresentative older trees that had survived the previous stand replacement event).

We sampled undergrowth biomass of each individual plant species by clipping all current annual growth up to 2 m in height to a 2 cm stubble in two 2-m^2 circular plots per macroplot transect (Fig. 8) (i.e., 6 total plots for vegetation macroplots and we added an additional transect for 8 total plots in the caribou macroplots). The clippings were bagged individually by species in paper sacks and oven-dried to constant mass and weighed.

We collected samples of current annual growth for nutritional analysis of the most abundant species at the macroplots (both accepted and avoided species). Immediately after collection, we sealed the samples in plastic freezer storage bags, buried these in ice, and transferred the samples to a freezer within four days after collection. We analyzed nutritional quality of plant samples at the Wildlife Habitat and Nutrition Lab and the Soil, Plant, and Waste Analytical Lab at Washington State University, Pullman, Washington. We freeze-dried the frozen plant samples and ground them to pass a 1-mm screen. We determined neutral detergent fiber (NDF), acid detergent fiber (ADF), acid detergent lignin (ADL), and acid insoluble ash composition (AIA) (%) using the filter bag method and the Ankom fiber analyzer$^{200/200®}$ (Ankom, Fairport, NY, Goering and Van Soest 1970) with sodium sulfite for shrubs and forbs. We adjusted estimates of NDF, ADF, ADL, and AIA based on correction equations presented by Cook et al. (2022). We estimated gross energy content (kJ/g of forage) using bomb calorimetry (C5000, IKA Works, Inc., Wilmington, NC, USA). We estimated nitrogen content (%) using a Carbon-Nitrogen TruSpec analyzer (LECO, St. Joseph, MI, USA) and calculated crude protein content (%) as 6.25 x the nitrogen content (Robbins 1993). We determined the protein-binding capacity of tannins using the BSA precipitation assay (mg bovine serum albumin precipitated/mg forage) (Martin and Martin 1982). We used the summative equations of Robbins et al. (1987a,b) to estimate dry matter digestibility (%) and digestible protein (DP) content (g protein/100 g forage). We then calculated digestible energy (DE) content (kJ/g) for all samples from the product of the GE content and dry matter digestibility.

Measuring Caribou Nutrition

A key component of our sampling strategy involved transporting the captive caribou and placing them in native habitats to measure their nutrition in response to the composition and abundance of forage available to them. A total of 79 enclosures were constructed of electric wire and sampled with tame caribou in 2017 and 2018 in Northwestern Ontario (the caribou were not deployed to our northeastern study area). Sampling was stratified to obtain nutrition estimates for each common ecosite and seral stage. The study area was divided into zones, and we rotated the caribou among these zones to provide interspersion of the enclosures across the study area. In addition to stratification to reflect the different ecosites and seral stages, specific placement of enclosures reflected access requirements and safety for caribou and project staff (e.g., along roads that were infrequently traveled by the public, adequate parking space for staff vehicles). Electric fences were used to move the caribou from the road well into the plant community to be sampled to exclude the ecotonal shift in vegetation from the road to the sampled stand. Caribou were deployed to the enclosures from late July through early October (excessive harassment by insects precluded sampling caribou foraging earlier than late July).

Methods of using the captive caribou in foraging trials were described in detail by Denryter et al. (2017, 2022). Briefly, highly trained staff observed caribou foraging in the enclosures at close range (within 0.5 to 2 m) during four 20-minute trials per caribou per day to estimate which plants and plant parts they ate. This approach provided roughly 390,000 observed bites in Ontario by caribou over the course of the study. We sampled forage selection patterns to identify those plant species that caribou readily ate (accepted species) and those that they rarely or never ate (avoided species) by statistically comparing the proportion of each plant species in caribou diets to the proportion available in the plant community in which they foraged. Each caribou, a lactating female with her calf at heel, was equipped with an activity monitor placed on a VHF collar to determine time spent foraging each day and night period, to provide a basis for estimating total daily intake of forage, DE, and DP. Finally, samples of the parts and species of forage consumed by caribou were combined in proportion to that in the diet and were submitted to the Wildlife Habitat Analysis Laboratory to determine dietary digestible energy and dietary digestible protein using assays as described above for individual forage samples. We were able to estimate 7 nutrition metrics with our data: forage (dry matter, DM, in g/min) intake per minute while foraging, which required estimating bite mass (g/bite) and bite rate (bites/min); digestible energy (DE) content of caribou diets (kJ/g of food); digestible protein (DP) content of caribou diets (g protein/100 grams or food, i.e., %); per minute intake rate of digestible energy (kJ/min); per minute intake rate of DP (g/min); per day (24-hr) intake of DE (kJ/day); and per day intake of DP (g/day).

DATA ANALYSIS

Forage Quality

We limited our evaluation of forage quality samples to those collected in July–September to better match the dates we used for mapping forage resources. We categorized each plant species collected according to caribou forage selection—i.e., accepted or avoided (Appendix A)—and plant group (deciduous shrub, evergreen shrub, forb, grass, graminoid, fern, arboreal lichen, terrestrial lichen, mushrooms, berries, and moss). We graphed and compared means of crude protein, digestible protein, and digestible energy across plant groups, for accepted plant species versus avoided species, and relative to nutritional requirements for a lactating female.

Predictive Equations for Forage Abundance

Our focus on analysis of forage attributes was on accepted species because we have found based on analysis for our previous studies (Cook et al. 2016; Denryter 2022), including this one (Walker 2023), that caribou foraging and nutrition responses are better related to characteristics of accepted biomass than total biomass. Our analysis primarily focused on 4 forage variables: accepted grasses, forbs, and deciduous shrubs (combined); accepted lichens (terrestrial and arboreal combined); accepted horsetails; and accepted mushrooms, all on a kg/ha basis. We also included a fifth forage variable (total accepted forage biomass) that was simply the sum of each of the 4 forage variables. Because an objective of the study was to clarify how vegetation changed in response to disturbance and succession, we first evaluated changes in overstory and these forage classes as a function of stand age, our best estimate of time since disturbance. This analysis was conducted for each ecosite, to reflect differences in site potential on forage attributes. Next, we developed prediction equations for the forage metrics using various site characteristics (e.g., canopy cover) that had associated GIS layers, also for each ecosite. The equations used to predict our 4 main forage metrics as a function of site characteristics were in turn used to construct spatially explicit landscape maps of forage biomass and as the foundation for predicting caribou nutrition (e.g., nutrient intake rate) across landscapes.

Stand age-vegetation relationships typically are non-linear in forested ecosystems (Cook et al. 2016, 2018), so we used Akaike information criterion corrected for small sample sizes (AICc) calculated by Curve-Expert (CurveExpert 2.6, D. G. Hyams, Madison, AL, USA) to determine if nonlinear regressions better accounted for variation in the dependent variable than linear regression and, if so, to select the appropriate non-linear formulation using AICc as a guide. If AICc of the non-linear regression was >4 units lower than for the linear regression, we used a non-linear formulation. If multiple non-linear versions met this criterion, we typically chose the one that (1) made the most biological sense, and (2) was the simpler of the non-linear formulations, avoiding those that included estimation of four or more regression parameters. In some cases, even if AICc failed to differ by more than four units, we chose to use non-linear formulations if the linear equations produced substantially negative estimates of the dependent variable over the period represented in our data set. Because Curve-Expert provides no statistical test of significance for non-linear models, we used PROC NLIN (model estimation method was Gauss-Newton; SAS Institute 1988) to calculate an *F*-test of model significance.

We developed the forage prediction equations using five covariates representing site characteristics: overstory canopy cover, basal area, stand height, tree density, and proportion of trees that were hardwood species (we did not use stand age as a covariate because GIS maps of stand age were not available for much of our study area at the time of this analysis). We also included date of sampling, normalized difference moisture index (NDMI), change in the enhanced vegetation index (ΔEVI), and percent clay, silt, or sand as covariates. These analyses were conducted for each ecosite separately, to account for different influences of site potential on forage abundance. However, we tailored the analysis to reflect the ecosites and seral stages included in the Far North Layer, or those ecosites to which we could crosswalk our ecosites. (For example, most of the ecosites in the Far North Layer were divided into 2 seral stages, and thus we developed a prediction equation for each of these 2 stages for that ecosite.) This helped to ensure that our prediction equations were appropriate for the ecosites and seral stages used in the Far North Layer (see Appendix B for cross-walking details).

For model selection, we first screened covariates to identify those that displayed non-linear versus those that displayed linear relationships with our forage response variables using methods described above (i.e., used Curve Expert and AICc to determine if non-linear regression was superior to linear regression, although, for this analysis, we used 2 AICc units instead of 4 to identify best models). For those that exhibited non-linear relations, we used the non-linear equation to transform covariate values to linearize the relationship of the covariate with the forage response variable (Walker 2023).

Following the methods of Walker (2023), we then developed prediction models of the 4 accepted forage metrics (Figure 1) (plus total accepted forage created by summing across the 4 forage metrics). Briefly, Walker (2023) used vegetation data collected between June and September 2017–2018 in Northwestern and 2018 in Northeastern Ontario. Using a model selection approach for each ecosite group, these predictive models were developed for each study region (Pickle Lake and Cochrane) by seral-specific (early: < 20 years; mid-late ≥20 years) ecosites as a function of Julian day and site conditions. Unless otherwise stated, all statistical analyses were conducted in R (R version 4.2.1, www.R-project.org, accessed 2 July 2022).

Predictive Equations for Caribou Nutrition

The analysis we conducted of caribou nutrition was primarily focused on developing equations to predict caribou nutrition as a function of the types and abundance of accepted forage available to them. We initially evaluated for prediction seven metrics: forage (dry matter, DM) per minute intake rate (g/min), digestible energy (DE) content of caribou diets (kJ/g of food), digestible protein (DP) content of caribou diets (grams of protein/100 grams or food, i.e., %), per minute intake rate of digestible energy (kJ/min), per minute intake rate of digestible protein (g/min), per day (24-hr) intake of digestible energy, and per day intake of digestible protein. There are two key attributes of interest about these metrics: (1) many of them are highly intercorrelated (e.g., $r > 0.95$ among per-minute forage intake and per-minute and daily intake of digestible energy and digestible protein), and (2) the higher up the scale of calculation (from forage intake rate to daily intake of DE and DP), the more types of data are needed to calculate the metric, thereby potentially introducing additional random error. For example, calculating forage DM intake rate requires estimates of bite size and bite rate; daily DE intake requires estimates of bite size, bite rate, DE content of forage consumed, and daily (24-hr) time spent foraging. In addition, digestible energy and digestible protein in caribou diets were largely insensitive to variation in availability of the different forage types and thus could not be reliably predicted for developing nutritional landscape maps. Finally, Walker (2023) reported significant correlations between how fast they eat (g of forage/min) and caribou habitat use responses in Northern Ontario. For these reasons, our metric of caribou nutrition for our mapping effort was per minute forage intake rate.

Earlier analysis of foraging data of captive caribou collected in British Columbia (Denryter 2017) and Northwestern Ontario (Walker 2023) indicated that forage intake rate was substantially influenced by the abundance of accepted species and indicated that separating the accepted species into the groups defined above (i.e., accepted grasses, forbs, and deciduous shrubs (combined); accepted lichens; accepted horsetails; and accepted mushrooms) improved predictions (J. Cook and R. Cook, unpublished data). Thus, the primary covariates for the modeling work included these 4 forage metrics, all on a kg/ha basis. We also considered overstory canopy cover and basal area for inclusion as covariates in our nutrition prediction models. Sample size (number of macroplots where caribou were deployed) for developing the forage intake prediction equations was too limited to support analysis for each ecotype. Thus, we separately developed prediction equations for uplands and lowlands ($n = 65$ for uplands and 14 for lowlands).

To develop the forage intake equations, we used a model selection strategy similar to that described above for developing forage abundance prediction equations. We first screened for non-linear relationships with Curve-Expert using an AICc of >4 to identify non-linear versus linear relationships. Non-linear equations were used to transform the relation for subsequent linear analysis (as described above). We then used AIC in PROC REG in SAS to guide the selection of the best models. We set the total number of covariates to 1 per 10 samples, and we required that each covariate was significant at $P \leq 0.10$ when included in the model (to simplify and avoid over-parameterization of the final models). We used variance inflation factor (VIF) to identify issues with multicollinearity and excluded covariates with VIF > 10 to reduce multicollinearity issues. Finally, we generated QQ normality graphs and plotted model residuals to evaluate violations of normality and homoscedasticity for each model.

Constructing Foodscape Maps

Accuracy assessment of predictor covariates.—We evaluated the accuracy of the seral-specific ecosite maps that was developed using the Ontario Far North v2 (OMNRF 2014; FN) and 2015 North America Land Change Monitoring System (Latifovic et al. 2017 [methods for Canadian extents]; NALCMS) land cover maps (methods above) based on the field-derived seral-specific ecosite classification at macroplots sampled across the Ontario AOI. The FN land cover map was developed for the northern extent of Ontario using Landsat imagery collected between 2005 to 2011, whereas the NALCMS land cover map used 2015 Landsat imagery. We extracted the proportion of each seral-specific ecosite within the spatial extent of each macroplot. When multiple land cover classes were identified within a single macroplot, we assigned the land cover class with the greatest proportion. If the proportions of two seral-specific ecosites were equal, we assigned the seral-specific ecosite that was the same or most similar to the seral-specific ecosite identified in the field.

Because of a temporal mismatch between our field sampling (2017−2018) and the year of imagery used in each land cover map (NALCMS: 2015; FN: 2005-2011), we removed temporally mismatched macroplots (i.e., macroplots that were logged or burned by forest fires between the year of imagery and when we sampled the macroplot in the field) in our evaluations. Because the year of imagery for the FN land cover ranged between 2005–2011 and was unknown for each raster cell, we selected 2011 as the imagery year as a conservative estimate for identifying temporally mismatched macroplots.

Once each macroplot was assigned a seral-specific ecosite classification and temporally mismatched macroplots were removed, we compared these classifications to those identified in the field by calculating an overall and user’s accuracy (Olofsson et al. 2014). We did not calculate a producer’s accuracy because we were not concerned about the predictive capacity of the models used to produce the land cover maps.

For continuous forest covariates included in the forage models (i.e., canopy cover and basal area), we calculated the mean value of each covariate within the spatial extent of each macroplot. We then graphed the relationships and calculated a Pearson’s correlation coefficient between the field-derived and the mean map-derived values for each covariate after temporally mismatched macroplots were removed (based on year 2015).

Building biomass and intake rate layers.—Our modeling effort was designed to provide three primary products: (1) maps of forage intake rate as a measure of nutritional resources; (2) a map of caribou habitat selection as a function of forage intake rate and predation risk; and (3) a nutritional implications map that rates the ability of habitats to provide more forage if subjected to stand-replacing disturbance or less forage as a function of time since disturbance, and flags those habitats that are currently more likely to benefit moose than caribou. Producing these maps first required constructing layers of accepted forage biomass (grasses, forbs, and deciduous shrubs (combined), horsetails, lichens, and mushrooms), and these attributes were predicted for mapping based on existing layers of site conditions as described above, e.g., ecosites and seral stages, basal area, and canopy cover. In addition, we developed a habitat selection map primarily based on intake rate and predation risk (described below) (Fig. 9).

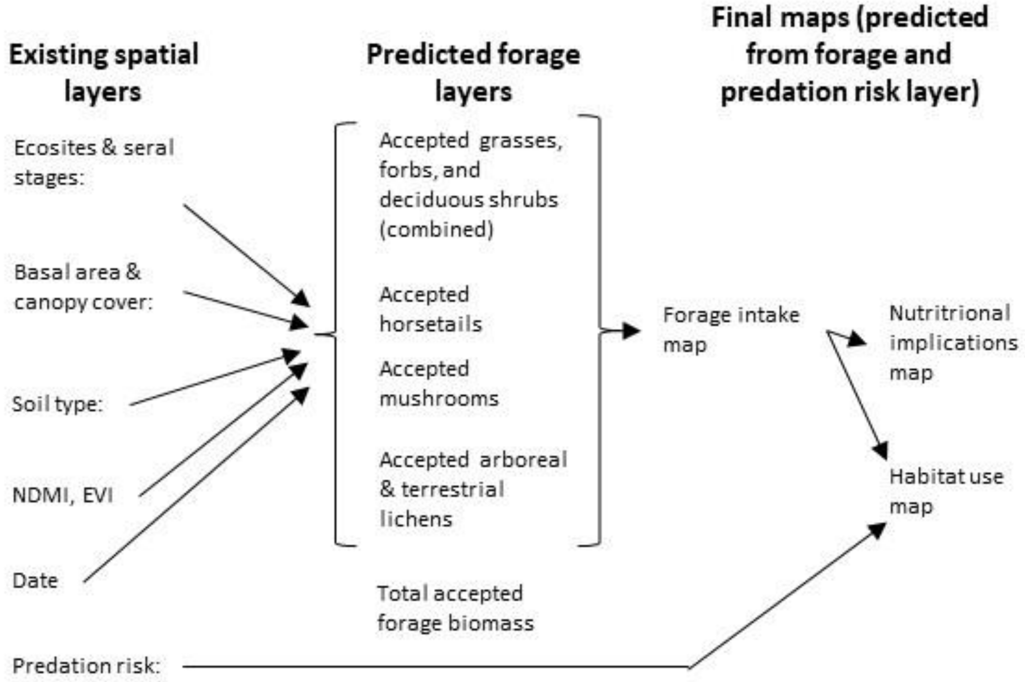


Figure 9. Illustration of two-step nutritional resource and habitat use prediction approach where forage attributes (i.e., biomass of different forage classes) are first predicted from existing layers of vegetation types, overstory conditions, and other attributes and then caribou forage intake rate is predicted from layers of forage attributes.

Following the methods of Walker (2023), we predicted 5 caribou forage metrics (kg/ha) across the Ontario AOI: biomass of grass, forbs, and deciduous shrubs (combined); biomass of lichens; biomass of mushrooms; biomass of horsetails; and total accepted forage biomass (kg/ha) using prediction equations developed as described above. Although Walker (2023) predicted these forage biomass attributes across the Kinloch (via the Pickle Lake models) and Kesagami (via the Cochrane models) caribou ranges, we extrapolated the predictions from these models across all boreal caribou ranges in Northern Ontario (with an additional 20 km buffer to the southern boundary) except the James Bay range (Fig. 6). We excluded the James Bay range because it is fully situated in the Hudson's Bay Lowlands, which is not representative of the vegetative conditions used to develop the Walker (2023) caribou forage models (i.e., boreal forest). We used the Ontario ecoregion map (Crins et al. 2009) to delineate the west and east AOI and used the Pickle Lake and Cochrane models to predict these forage metrics across each AOI, respectively. The west AOI was specified as the Big Trout Lake, Lake St. Joseph, Lake

Nipigon, and Lake Wabigoon ecoregions, and the east AOI was the Lake Abitibi and a portion of the James Bay ecoregion (Figure 6).

Prediction of the caribou forage metrics used as covariates data reflecting site conditions derived from existing layers of ecosites, forest overstory and soil characteristics, greenness indices, and date. Following Walker (2023), we used the Far North Layer v. 1.4 (OMNRF 2014) and Ontario Land Cover Data Base second edition (2000; hereafter Ontario 2000); for ecosites and seral stages, layers of tree basal area and canopy cover developed by Matasci et al. (2018); the SoilGRid250m dataset for percent clay, sand, and silt (250 m; Hengl et al. 2017); and NDMI and ΔEVI calculated from 2018 Landsat 8 and 2018 MODIS imagery (250 m), respectively, to predict each forage metrics across the Ontario AOI. We used the original Far North Layer v. 1.4 and Ontario 2000 layers, instead of the Far North v2.0 (which combined these products and the Southern Ontario Land Resource Information System v1.2 to develop an Ontario wall-to-wall land cover compilation), because it combined conifer, deciduous, and thicket swamp into a single "swamp" landcover type. This combined swamp landcover type inhibited cross-walking the Far North v2.0 to our ecosite classifications. Classification of seral stages from the Far North v1.4 and Ontario 2000 layers was early (< 20 years) and mid-late (≥20 years). However, these layers do not indicate the land cover type for early-seral. Therefore, Walker (2023) used the North America Land Change Monitoring System land cover layer (Latifovic et al. 2017) to identify the ecosite type within these early-seral pixels. Additionally, productive (Upland-Black Spruce-White Spruce) and unproductive (Upland-Black Spruce) upland conifer sites were not distinguished in either the Far North v1.4/ Ontario 2000 or the North America Land Change, and therefore Walker (2023) used threshold values of NDVI (30 m; Landsat 8, June– September 2010) calculated at vegetation macroplots to differentiate between these two ecosites with > 75% accuracy (based on field-derived classification of ecosites). We assumed no forage biomass in areas of silvicultural harvest and forest fires that were less than 1 years old (i.e., occurred in 2018) based on layers developed by Hermosilla et al. (2016). To avoid extreme extrapolation errors, we capped predicted biomass to the maximum values we measured in our macroplots (grass, forb, shrub combined cap = 1567.78 kg/ha; lichen cap = 4816.38 kg/ha; horsetail cap = 185.42 kg/ha; mushroom cap = 11.60 kg/ha). Also, when predicted biomass was less than zero, we set those values to 0.1 kg/ha. Most of the prediction models included Julian date and thus were temporally dynamic. However, we developed two static maps for each of the forage metrics on 1 August and 15 September. Biomass of accepted forage was predicted for each 30 m pixel in our AOI. For further details on developing an ecosite map for northern Ontario, see Walker (2023).

Predicting caribou forage intake rate across our Ontario AOI was based on predictions of biomass of accepted grasses, forbs, deciduous shrubs (combined), horsetails, and mushrooms, used as separate covariates in the intake rate prediction equations (biomass of accepted lichens did not enter as a useful predictor covariate [described below], nor did we use total biomass of all accepted forage as a covariate in the equations). Though Julian date was not included in our intake rate prediction equations, by default it impacted intake rate through effects on some of the biomass covariates. Forage intake rate was also predicted for each 30 m pixel.

Constructing Nutritional Implications Map

Though important, maps of nutritional resources (biomass and intake rate) alone may be difficult to interpret relative to management and conservation applications if users have little understanding of nutritional ecology. Therefore, we also produced 2 raster layers to aid with interpretation (referred to as the "nutritional implications" map) that rates each pixel according to three criteria: (1) nutritional adequacy of forage intake rate specifically in relation to nutritional requirements of lactating females and their calves in summer; (2) changes in nutritional resources to stand-replacing disturbance; and (3) potential that stand-replacing disturbance and resultant forage development in early seral communities will be highly beneficial for moose and thus detrimental to caribou.

For the forage intake layer, we created 4 categories of nutritional value including good to excellent, marginal, poor, and inadequate based on forage intake rate in relation to expected performance of lactating females and their calves. These categories are based on studies of nutrition effects on performance of lactating elk and their calves, the largest such study conducted for native herbivores during summer (Cook et al. 2004). Good to excellent is a nutrition level that allows lactating females to acquire at least 10% body fat levels by mid-autumn and enter the breeding season with a high probability of becoming pregnant and their calves to grow rapidly; marginal is a nutrition level in which lactating females approximately hold constant body fat levels over summer through early autumn but with some reductions in probability of pregnancy (e.g., ~80%) and calves with growth rates below which they are capable; poor is a nutrition level in which lactating females are likely to lose body fat over summer and have highly reduced probability of pregnancy (e.g., <50%); and inadequate is a level which will not support lactating females and their calves. We used the different daily digestible energy levels of 3 groups of elk subjected to different levels of nutrition and adjusted these levels from elk to caribou based on metabolic mass (MM = kg of body mass$^{0.75}$). We divided the daily DE intake by digestible energy content of 2.87 kcal/g of ingested forage (the mean dietary digestible energy from all foraging trials conducted with the captive caribou). The result of this calculation was daily forage intake required to satisfy the daily digestible energy requirement, and daily forage intake was divided by the maximum number of hours caribou are likely to feed each day (15 hours; Denryter 2017, J. Cook and R. Cook, unpublished data) to estimate per minute intake rate necessary to satisfy required daily digestible energy levels (Table 1).

Table 1. Calculations to develop forage intake – performance relationships for caribou.

Mapped categories	Performance level (from elk studies)	Minimum daily intake of DE to achieve mapped category (kcal/kg of MM)	Minimum daily intake of DE for caribou to achieve category (kcal/day)[a]	Minimum daily forage intake to satisfy minimum DE levels[b] (g/day)	Forage intake (g of forage/min) to satisfy daily DE intake categories (assuming 15 hours/day foraging)
1. Good to excellent	High	360	12,960	4,516	>5.0
2. Marginal	Medium	290	10,440	3,637	>4.0–5.0
3. Poor	Low	220	7,920	2,759	>3.0–4.0
4. Inadequate					≤3.0

[a] Converted from elk to caribou assuming a mean body mass of caribou of 120 kg (or 36 kg of metabolic mass (MM) [kg body mass$^{0.75}$]).

[b] Assumes mean DE content of caribou diets is 2.87 kcal/g of consumed forage.

To address effects of management on current forage conditions, we identified 5 categories for mapping (Table 2) reflecting stable environments unlikely to be affected by stand-replacing disturbance, those that are likely to improve with disturbance, and those that are likely to decline without disturbance, with flags for stands that currently or will likely provide forage that benefits moose to a greater extent than caribou. We used our data on stand age-forage production relationships to identify for each ecosite the period during which forage is elevated after stand-replacing disturbance (Fig. 10). To a large degree, these data indicate substantial improvements in accepted forage production to last 20 to 30 years after disturbance. However, because the Far North Layer categorizes early seral stands as <20 years after disturbance, we used this criterion to identify those areas where forage is likely to be elevated. To identify areas with abundant forage likely to benefit moose more than caribou, we assumed that stands with >400 kg/ha of accepted forage of grasses, forbs, deciduous shrubs, horsetails, and mushrooms disproportionally benefit moose compared to caribou (i.e., total biomass of accepted vascular plants—lichens were excluded because we assume that consumption of lichens by moose in summer is rare). We based this 400 kg/ha threshold on studies of elk, where the relationships between accepted forage biomass and per-minute forage intake rate have frequently been documented (Fig. 4). We assumed that elk serve as a reasonable proxy for moose; to our knowledge, there have been no studies that provided these kinds of data during summer for moose.

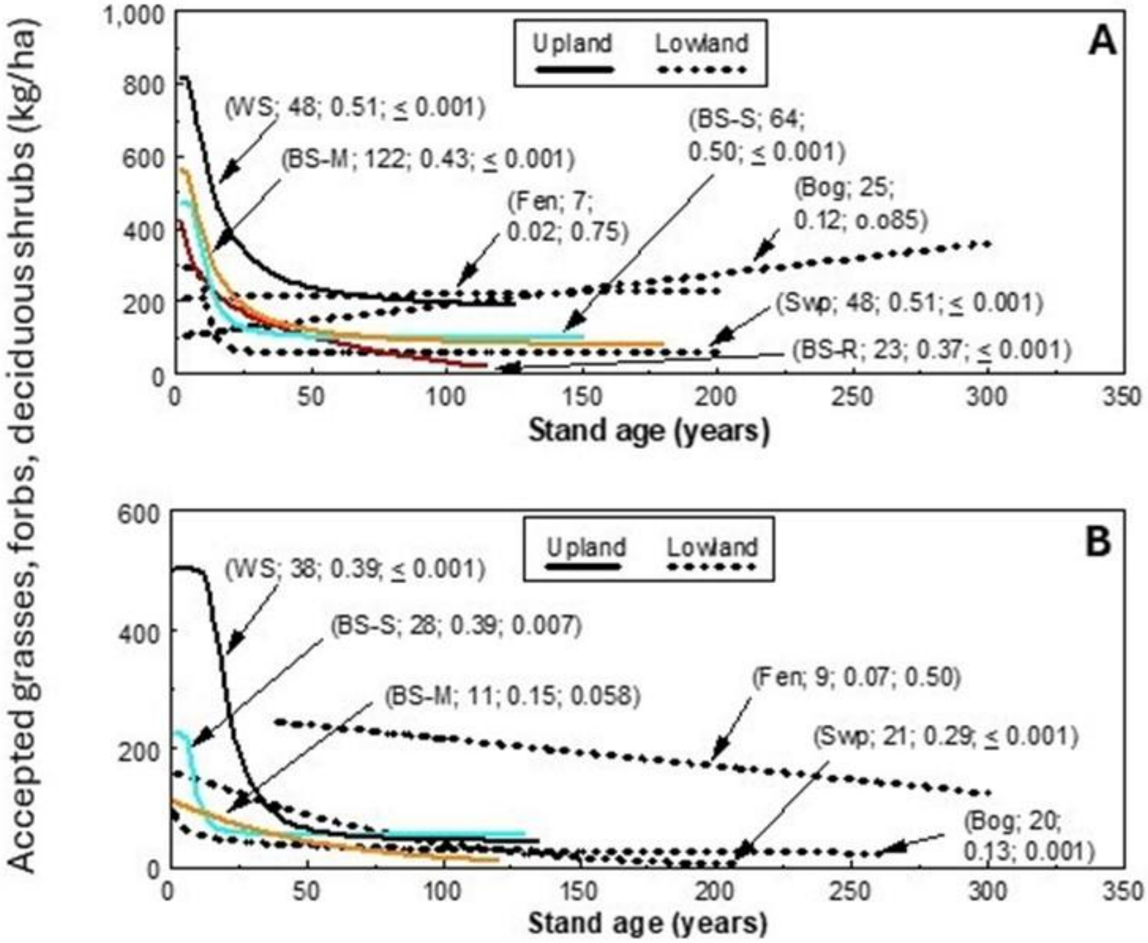


Figure 10. Forage biomass as a function of stand age after stand-replacing disturbance in Northwestern Ontario (A) and Northeastern Ontario (B). In parentheses, the first number is the sample size of field plots, the second is the r^2*, and the last is the* P*-value of the regression.*

Table 2. Mapped categories reflecting changes in forage resources over time as a function of stand-replacing disturbance (codes are: Lwl = lowland; Upl = upland; BS = black spruce; WS = white spruce).

Mapped categories	Plant community conditions
1. Stable over time regardless of disturbance	Lwl-fen Lwl-marsh (no early seral data, but assume these wet types very rarely burn) Lwl-swamp thicket (we have no early seral data, assume rarely burn)
2. Current conditions where abundant forage may favor moose over caribou	Any stand currently having > 400 kg/ha of accepted grasses, forbs, deciduous shrubs (combined), horsetails, and mushrooms. (Any stand with this condition automatically also falls under category 5; any stand with this condition was not re-classified under condition 5)
3. Biomass may increase with disturbance	Upl-BS > 20 years old Upl-BS-Rocky > 20 years old Lwl-Bog > about 20 years old (forage biomass in swamps respond better than bogs after disturbance, but had to combine bogs and swamps into Lwl-Bog ecosite for mapping)
4. Biomass may increase with disturbance to levels that better support moose than caribou	Upl-BS-WS > 20 years old
5. Biomass may decline without disturbance	Upl-BS-WS < 20 years old Upl-BS < 20 years old Upl-BS-Rocky < 20 years old Lwl-Bog < 20 years old

Constructing Caribou Selection Map

Walker (2023) fit integrated step selection analysis (iSSA; Avgar et al. 2016) models for three seasons (calving [1 May to 14 June], early summer [15 June to 31 July], and late summer [1 Aug to 15 Sept]) and three reproductive states (barren, calf lost < 5-weeks postpartum, and calf alive >5-weeks postpartum) using GPS telemetry data from female caribou across Pickle Lake, Nakina, and Cochrane. Using these iSSAs, we developed a probability of selection map across the Ontario AOI for female caribou with calf survived (>5-weeks postpartum during late summer (1 Aug to 15 Sept). We selected this reproductive state and season because these caribou raised calves throughout the greatest nutritional requirements of peak lactation and late summer has been identified as a critical period for ungulate to accrue body fat before fall reproduction and winter. The top iSSA model for caribou with calf alive >5-weeks postpartum during late summer included a linear and quadratic term for intake rate (g/min; reflecting selection for intermediate intake rates), summer predation risk (Avgar et al. 2015), an interaction between intake rate and summer predation risk (representing a trade-off between forage and risk), and step length and turn angles as fixed effects. We used the selection coefficients from this exponential iSSA model to predict caribou probability of selection across the Ontario AOI (west and east combined; Table 3). We used the mean logged step length (5.25) and cosine-transformed turn angle (0.07) for caribou during the late summer when predicting the probability of selection. The iSSA developed Walker (2023) included daily predictions of intake rates; however, we used intake rates predicted for the mid-point of the late summer period (23 August) to develop the static probability of selection map.

Table 3. Selection coefficients used to predict probability of selection for caribou with calf survived (>5-weeks postpartum) during late summer (1 August to 15 September) across the Ontario AOI.

Covariate	Selection coefficient
Intake rate	1.16
Intake rate^2	-0.06
Predation risk	1.88
Intake rate* Predation risk	-1.50
Step length	0.03
Turn angle	-0.17

We followed the methods of Avgar et al. (2015) to predict summer predation risk across the Ontario AOI for the year 2018, which was based on GPS telemetry of 52 wolves across 34 packs within Nakina (37 individuals from 23 packs) and Pickle Lake (15 individuals from 11 packs) from May to October in 2010–2012. Avgar et al. (2015) followed methods outlined in Kittle et al. (2015) and used pack-specific, 95% Brownian bridge utilization kernels weighted by pack size to estimate predation risk values from each wolf pack across study regions, which were then summated across the landscape to derive a population-level utilization distribution. The values of the population-level utilization distribution were then modeled as a function of landscape covariates (normalized difference vegetation index [NDVI] for 2018, Far North Land Cover types, and distance to primary and secondary roads, waste management sites [i.e., dumps], human settlements, and rivers and large lakes [>500 m in diameter]) and predicted at the scale of

30-m using a generalized least squares regression model. Predictions at the 30-m scale were then scaled to a 220 m^2 hexagonal grid by taking the average value within each hexagon, resulting in a static map of summer predation risk across the Ontario AOI for 2018. For further detail, see Avgar et al. (2015). Finally, we normalized the predicted probability of selection to range between 0 and 1 after truncating the maximum probability of selection to not exceed the 99th quantile of all pixels across the Ontario AOI.

RESULTS

Ecosite Classification

For each ecosite, we (1) provide a description of the ecosite; (2) present details of relationships between stand age and overstory and undergrowth (i.e., vegetation changes after disturbance); (3) offer management implications based on these relationships; (4) provide tables of mean biomass of the most abundant plant species by stand age category; and (5) provide representative photos in Appendix D. In general, the grouping of Ontario's ecosites to simplify nutritional resource evaluation and mapping greatly simplified evaluating and mapping nutritional resources and reflected important differences in forage resources for caribou often with important management implications (Table 4).

Table 4. General ranking of ecosite for caribou nutrition evaluation (nd = no data) (see Appendix D for specific details about each ecosite group). (Codes are: Upl = upland; BS = Black Spruce; WS = White Spruce; Lwl = lowland).

Ecosite	Biomass of accepted lichens	Intake rate & biomass of accepted vascular forage in early seral stages	Value of early seral forage to moose	Intake rate & biomass of accepted vascular forage in mid- to late seral stages	Potential to improve nutrition for caribou
Upl-BS-Rocky	Moderate to high	Moderate	Poor to moderate	Low	Good
Upl-BS-Sandy	Moderate to high	Moderate	Poor to moderate	Low	Good
Upl-BS-Mesic	Low	Moderate to high	Moderate	Low to moderate	Moderate to good
Upl-BS-WS	Low	Very high	High	Moderate	Good[a]
Lwl-bog	Low to moderate	Low to moderate	Poor	Very low	Moderate
Lwl-swamp (poor)	Low to moderate	Low to moderate	Poor to moderate	Very low	Moderate
Lwl-fen	Low	nd-projected to be moderate to high	nd-probably moderate	Moderate	nd-probably moderate to good
Lwl-swamp thicket	Low	nd	Unsure	Moderate to high	Unsure
Lwl-marsh	Low	nd	Unsure	Moderate to high	Unsure

[a] Although habitat treatments that bolster forage production in this ecosite may improve nutrition to caribou, the benefits to moose are also likely high in this ecosite, and thus the net benefits to caribou may be low.

Forage Quality

Forage quality averaged from July–September ranged widely among forage types and selection categories based on a total of 467 samples. We found consistently higher levels in crude protein, digestible protein, and digestible energy in accepted plant species and avoided species clearly suggesting that caribou endeavor to consume vegetation of higher quality (Fig.

11). Both protein metrics averaged 2–5 percentage points and digestible energy averaged 0.2–0.4 kcal greater in accepted versus avoided species. At first glance, these differences may seem small, but small differences in quality levels translate to large differences in animal nutrition and performance (i.e., the multiplier effect; White 1983, Cook et al. 2004). For example, most means for accepted species approached or exceeded protein requirements, as did many of the means for digestible energy, whereas means of each group of avoided species were nearly all below requirements for lactating caribou in summer. Among accepted species of life form groups, protein content of all life form groups approached or exceeded requirements, except for arboreal and terrestrial lichens, berries, and moss. Digestible protein content of terrestrial lichens and berries was even negative, indicating it costs caribou more endogenous protein to digest these forages than they can obtain from the protein in these forages. Digestible energy content of accepted deciduous shrubs, forbs, lichens mushrooms, and berries satisfied requirements, whereas grasses, graminoids (such as sedges), horsetails, and rushes, and mosses did not. Overall, mushrooms provided the best nutrition and moss the worst.

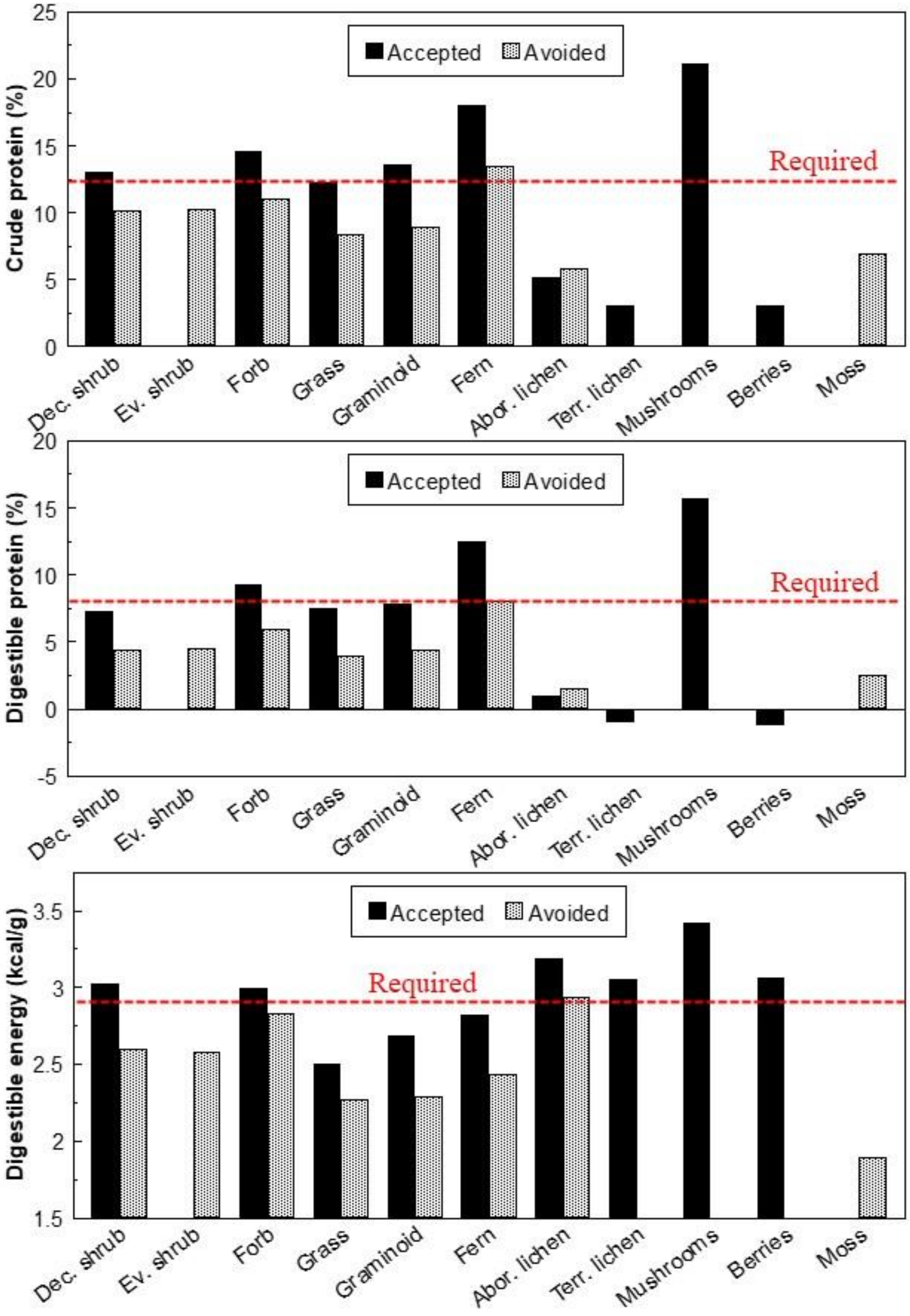


Figure 11. Forage quality averages during summer (July–September) 2017–2018 in Northwestern Ontario and 2018 in Northeastern Ontario for accepted and avoided plant species by caribou forage selection categories (accepted and avoided; Appendix A). The red line represents approximate requirements of a lactating female caribou during summer. Standard errors of the mean averaged 0.27 and 0.26 for crude and digestible protein and 0.023 for digestible energy of the common, accepted life form groups (i.e., too small to depict on the graphs). Abbreviations are: Dec. = deciduous; Ev. = evergreen; Abor. = arboreal; Terr. = terrestrial.

Forage Prediction Equations

Biomass of accepted forage was typically well correlated to stand age, although this varied by ecosite (see Appendix D for graphs of these relationships). The site conditions included across all top models were basal area (m^2/ha); canopy cover (%); normalized difference moisture index (NDMI); change in the enhanced vegetation index (ΔEVI); and percent clay, silt, or sand (Appendix C). Equations to predict forage biomass for five forage classes for mapping purposes provided moderate coefficients of determination (r^2) ranging from 0.11–0.52 (Appendix C). These were typically lower than the r^2 values for relationships between stand age and forage biomass (Appendix D). The lower r^2 for our prediction equations largely resulted from separating regressions between early seral versus mid-to-late seral stages to accommodate the classification scheme of the Far North vegetation layer. When predicted levels of forage biomass were correlated to observed levels of forage biomass across all individual macroplots, r^2 values were typically higher than those for the individual equations (Table 5; Figure C1).

Table 5. Number of macroplots (*n*) and variation explained (r^2) in accepted forage biomass using forage prediction equations and observed forage biomass at macroplots at our Northwestern and Northeastern Ontario study areas, 2017–2018.

Forage metric	Northwestern		Northeastern	
	n	r^2	*n*	r^2
Total accepted biomass	341	0.34	126	0.38
Grasses, forbs, and deciduous shrubs (combined)	342	0.57	128	0.63
Lichens	343	0.29	127	0.35
Horsetails	344	0.23	128	0.30
Mushrooms	344	0.17	127	0.16

Caribou Nutrition Prediction Equations

Across all macroplots with caribou data, forage intake rate (g/min) was moderately correlated to abundance primarily of accepted grasses, forbs, and deciduous shrubs (combined), showed some correlation with accepted biomass of horsetails, but exhibited little or no significant correlation with biomass of accepted terrestrial and arboreal lichens or mushrooms (we included results from foraging trials conducted in BC with those of Ontario for comparison) (Fig. 12). Separating lowlands from uplands and using multiple regression to simultaneously evaluate the contribution of each forage type provided a more complete expression of the importance of each forage type. Biomass of accepted grasses, forbs, and deciduous shrubs (combined), horsetails, and mushrooms all contributed positively to forage intake rate, whereas biomass of accepted lichens did not (Table 6). These were the equations used to predict forage intake rate across our AOI.

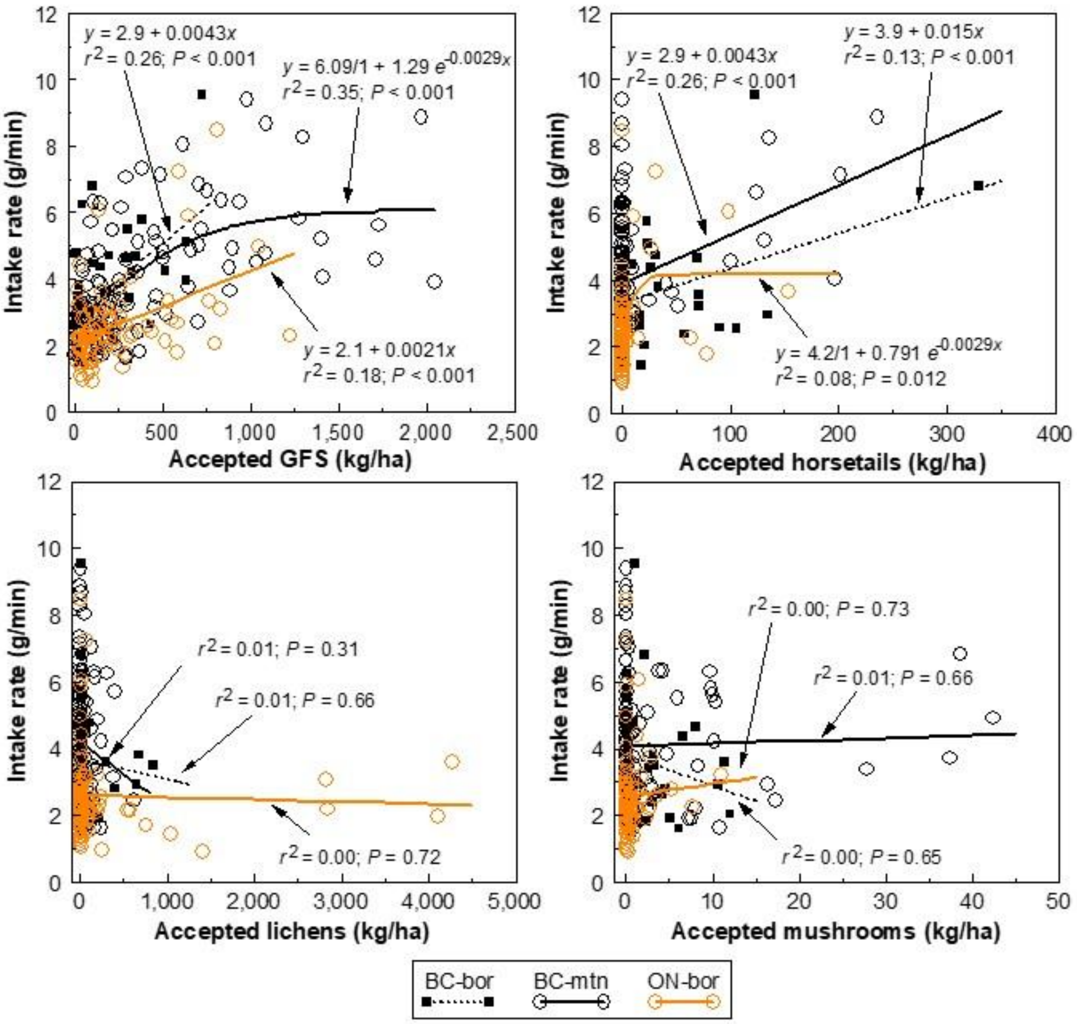


Figure 12. Per-minute intake rate of forage during summer and early autumn on a dry matter basis at three Canadian study areas: the boreal forests (BC-bor; 2013-2016) and montane (BC-mtn; 2013-2016) forests of northeastern British Columbia and the boreal forests of Northwestern Ontario (ON-bor; 2017-2018). (GFS = grasses, forbs, and deciduous shrubs).

Table 6. Multiple regression equations relating forage intake (*y*: g/min) with biomass of forage groups. We capped predicted intake rate estimates at both the low and high end to prevent unduly large or small estimates of caribou nutrition that arise from extrapolation of our data. We set this cap based on the highest (9.56 g/min) and lowest (0.92 g/min) values of intake observed during our entire study (ON and BC combined).

Equation[a]	R^2	*n*
Lowlands		
$y = -0.887 + 1.119 \times T\_AccGFS + 0.010 \times AccHors + 0.702 \times Mush$	0.64	14
Uplands		
$y = 0.585 + 0.685 \times T\_AccGFS + 0.082 \times AccHors + 0.103 \times Mush$	0.30	65

[a] Covariate codes are: *T_AccGFS* = biomass of accepted grasses, forbs, and deciduous shrubs (combined) (kg/ha) transformed from curvilinear form using the $T\_AccGFS = 4.383/(1+1.4328exp^{-0.02183*x})$; *AccHors* = biomass of accepted horsetails (genus *Equisetum*) (kg/ha); *Mush* = biomass of mushrooms (kg/ha).

Maps of Nutritional Resources for Caribou

Accuracy assessment of predictor covariates.—The correlation between field-derived estimates of canopy cover and basal area and the corresponding maps developed by Matasci et al. (2018) was 0.77 and 0.69, respectively, with little evidence of significant bias (Figure 13).

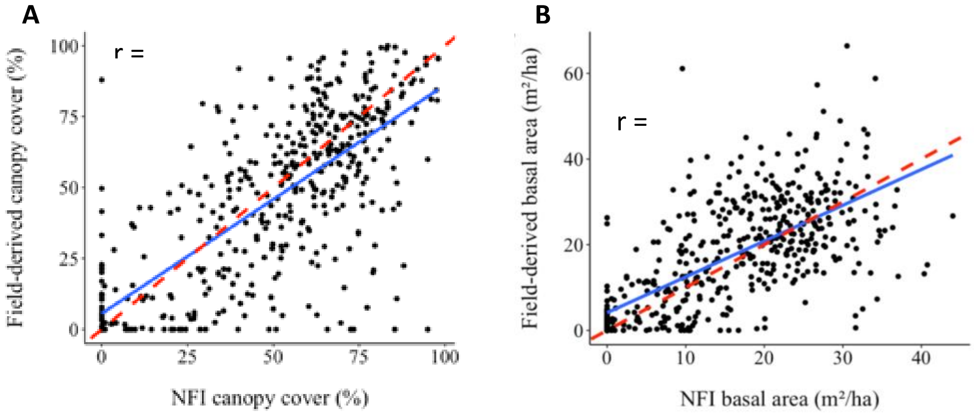


Figure 13. Relationship between field-derived and map-derived estimates of A) canopy cover (%), and B) basal area (m^2/ha) via Matasci et al. (2018; NFI). Red-dashed line indicates the 1:1 line (i.e., no bias).

The overall classification accuracy of the ecosite map developed for Northern Ontario was 0.41 (Table 7). Ecosites with the highest user's accuracy were early-seral, upland-black spruce (Upl-BS) (81%), and Upl-BS-white spruce (WS) (73%), followed by mid-late seral Upl-BS (44%) and Upl-BS-WS (46%). The ecosites with the lowest user's accuracy were mid-late Upl-BS-Rocky (0%), Lwl (lowland)-Swamp Thicket (0%), Lwl-Marsh (0%), and Lwl-Fen (6%); however, these ecosites generally had fewer macroplots ($n \leq 17$).

Table 7. User’s classification accuracy for the Ontario map of ecosites (Upl = upland; BS = black spruce; WS = white spruce; Lwl = lowland) and seral stages (early <20 years old; mid-late ≥ 20 years old) developed by Walker (2023) based on field-derived ecosite classification at 326 macroplots sampled during 2017-2018 across Northern Ontario, Canada.

Field-derived classification	Field-derived classification										User's accuracy		
	Upl-BS-Rocky-Mid-late	Upl-BS-Early	Upl-BS-Mid-late	Upl-BS-WS-Early	Upl-BS-WS-Mid-late	Lwl-Bog-Early	Lwl-Bog-Mid-late	Lwl-Swamp	Lwl -Fen	Lwl-Marsh	*n*	Total	Accuracy
Upl-BS-Rocky-Mid-late		4	10		3						0	17	0.00
Upl-BS-Early		26		3	2	1					26	32	0.81
Upl-BS-Mid-late	1	34	49	7	8		12				49	111	0.44
Upl-BS-WS-Early	1	2		8							8	11	0.73
Upl-BS-WS-Mid-late		3	6	13	24		5	1			24	52	0.46
Lwl-Bog-Early		3		2		3	3				3	11	0.27
Lwl-Bog-Mid-late	1	14	23	2		3	24		4		24	71	0.34
Lwl-Swamp thicket							3				0	3	0.00
Lwl-Fen			2	2			11		1		1	16	0.06
Lwl-Marsh							1		1		0	2	0.00
Total	3	86	90	37	37	7	59	1	6	0	135	326	0.41

Map of seral stage.— Early seral forest communities (< 20 years old) compose less than 20% of the land area across our overall mapping area. The proportion of early seral was higher across the west AOI (0.18) vs. the east AOI (0.14) (Fig. 14).

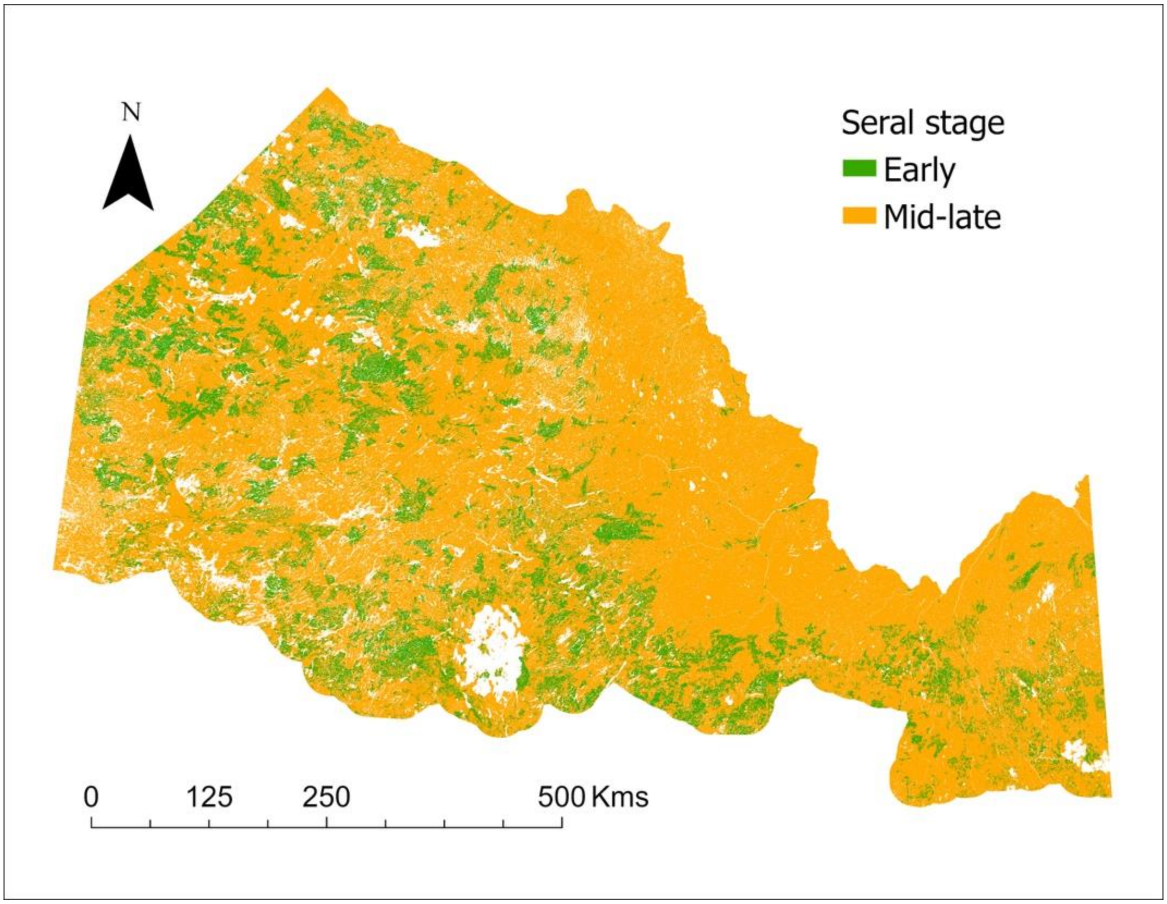
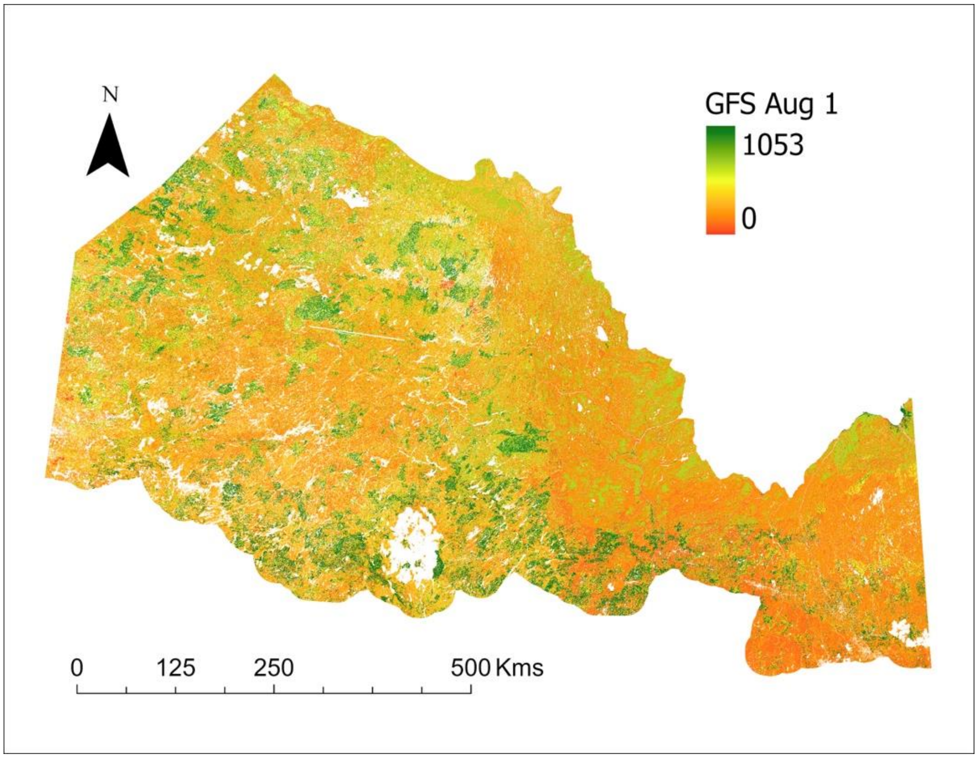


Figure 14. Distribution of early (<20 years) and mid-late (≥20 years) seral stands across the Ontario AOI.

Maps of accepted forage.— Below, we present maps depicting predictions of biomass (kg/ha) across the AOI for grass, forbs, and deciduous shrubs combined (Fig. 15), horsetails (Fig. 16), mushrooms (Fig. 17), and total accepted biomass (Fig. 18) for both 1 Aug and 15 Sept. In addition, we present a map depicting predictions of lichen biomass (Fig.19). The equations for predicting lichen biomass did not include date, and therefore, only one map was developed.



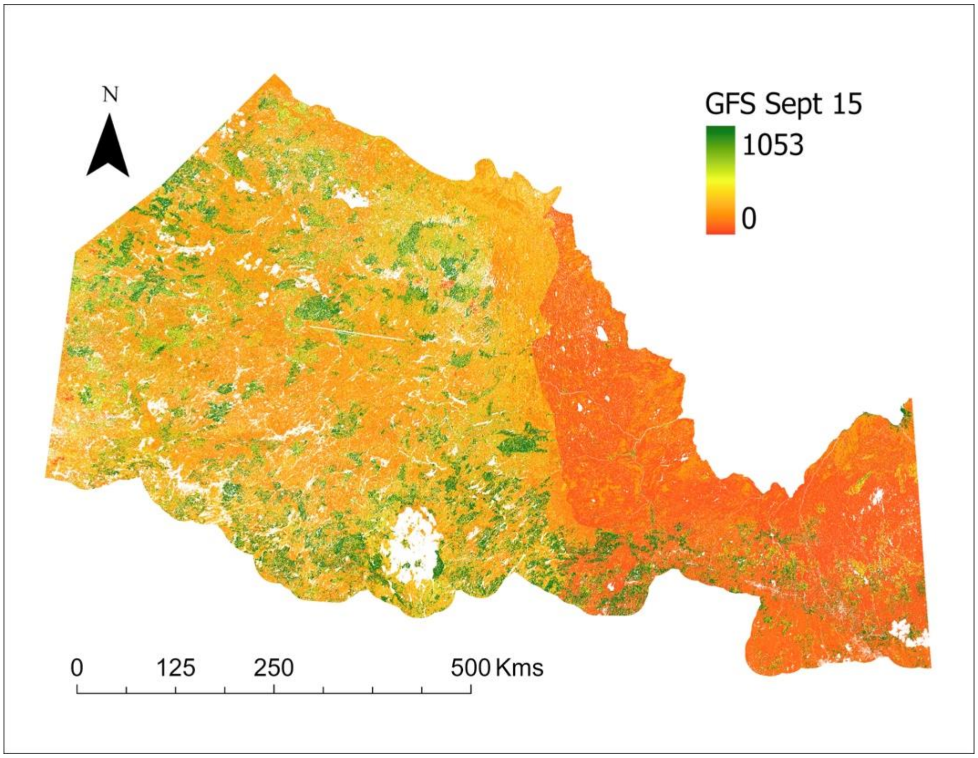
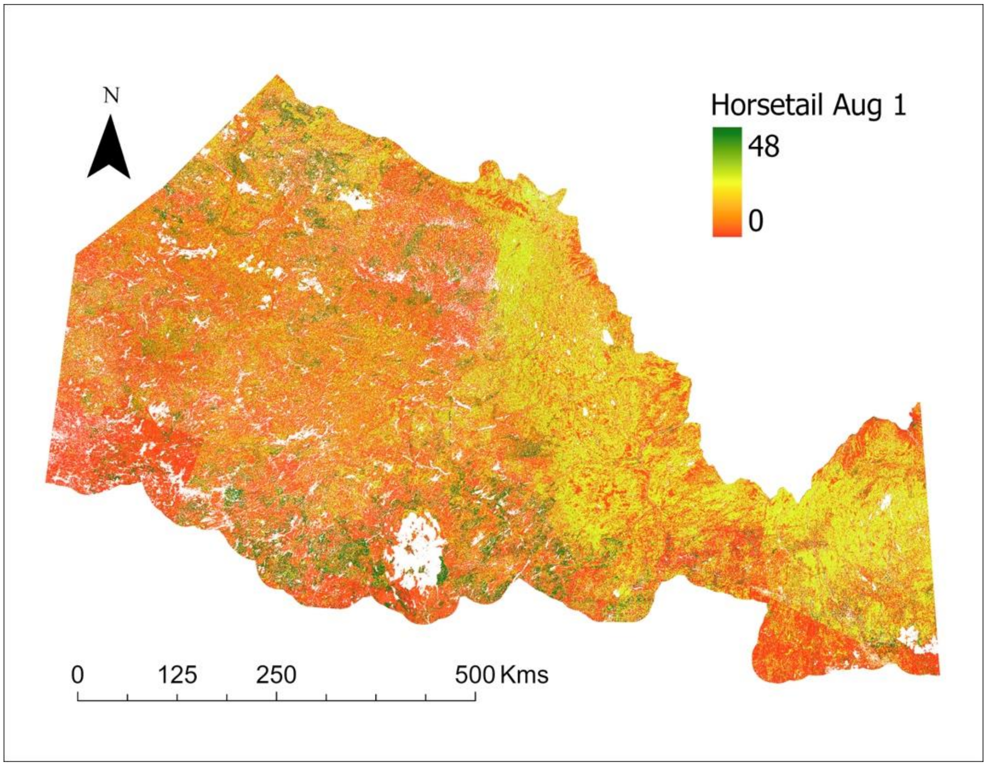


Figure 15. Biomass (kg/ha) of grasses, forbs, and deciduous shrubs combined (GFS) predicted for 1 August (top map) and 15 September (bottom map) across the Ontario AOI.



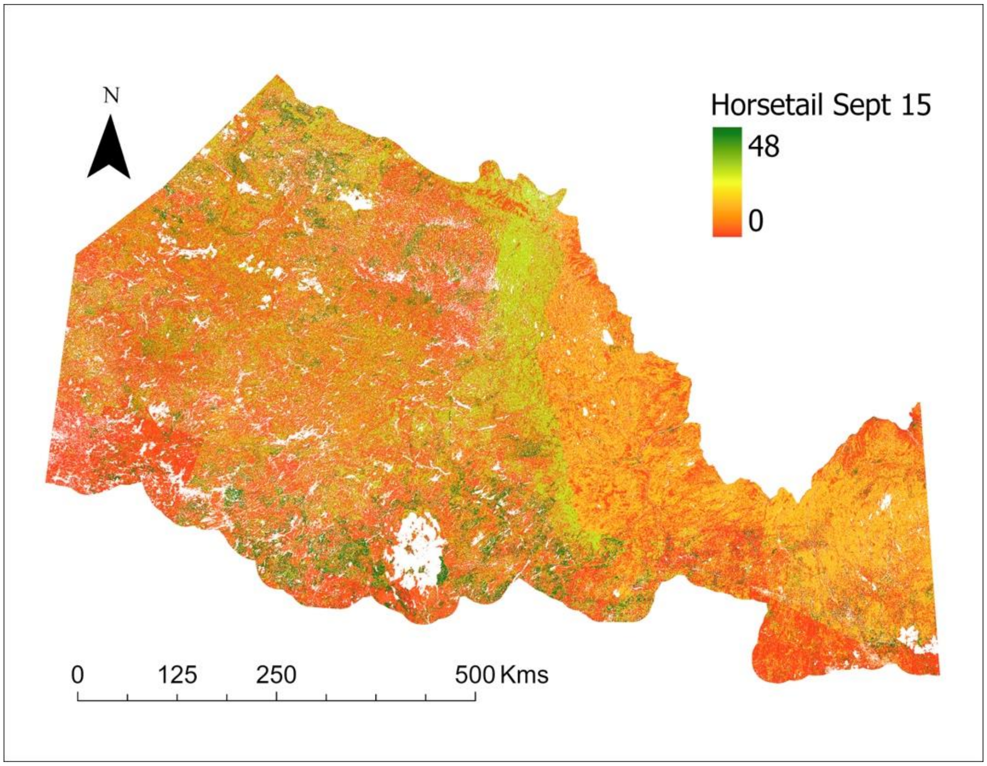


Figure 16. Horsetail biomass (kg/ha) predicted for 1 August (top map) and 15 September (bottom map) across the Ontario AOI.

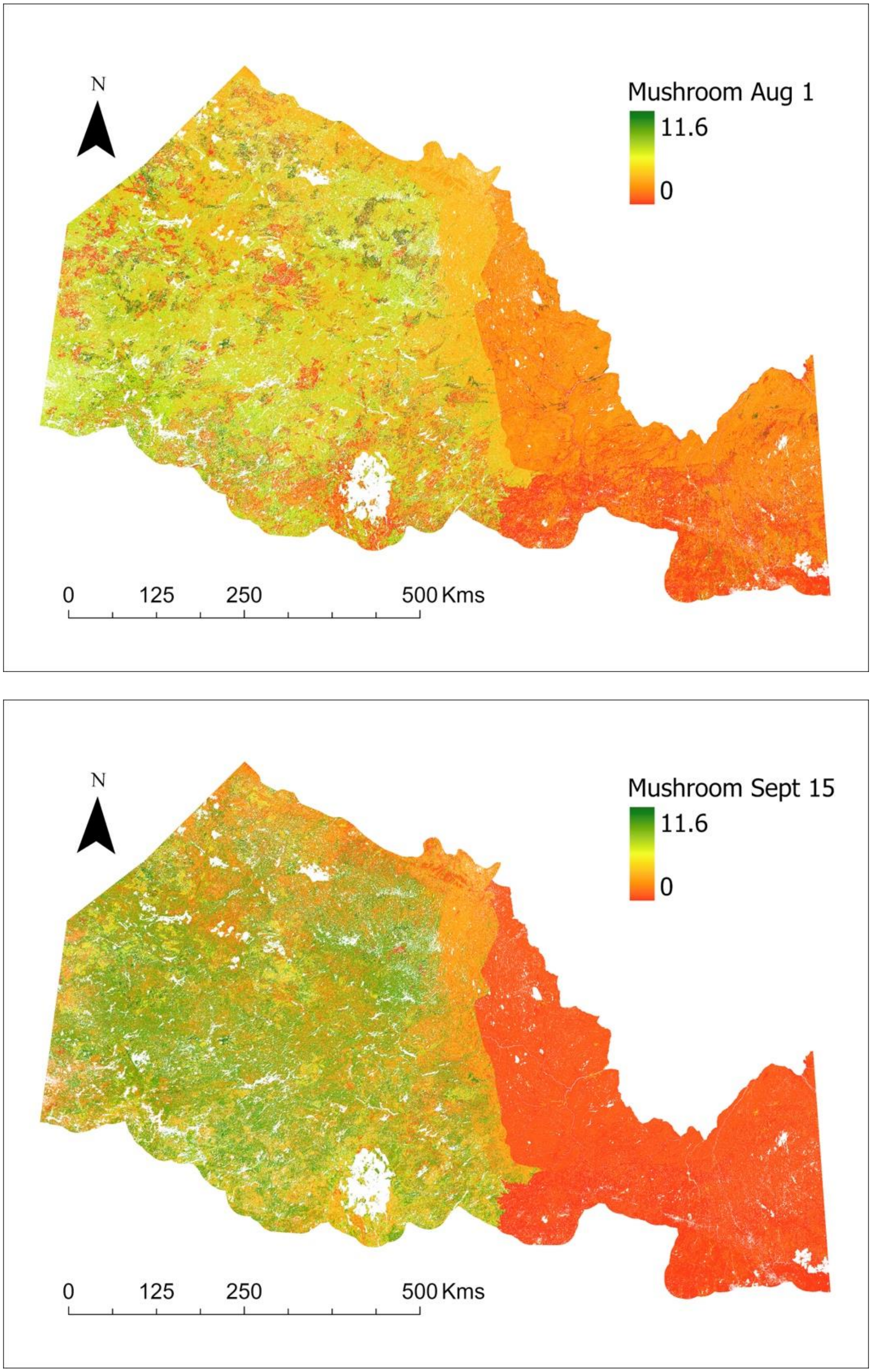
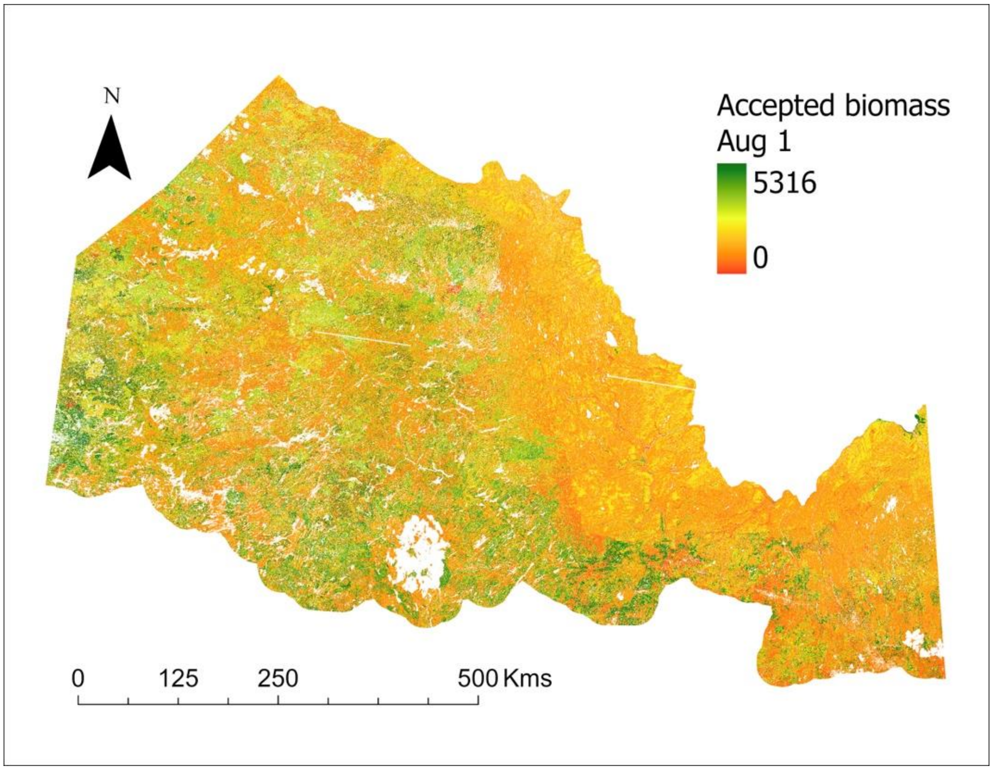


Figure 17. Mushroom biomass (kg/ha) predicted for 1 August (top map) and 15 September (bottom map) across the Ontario AOI.



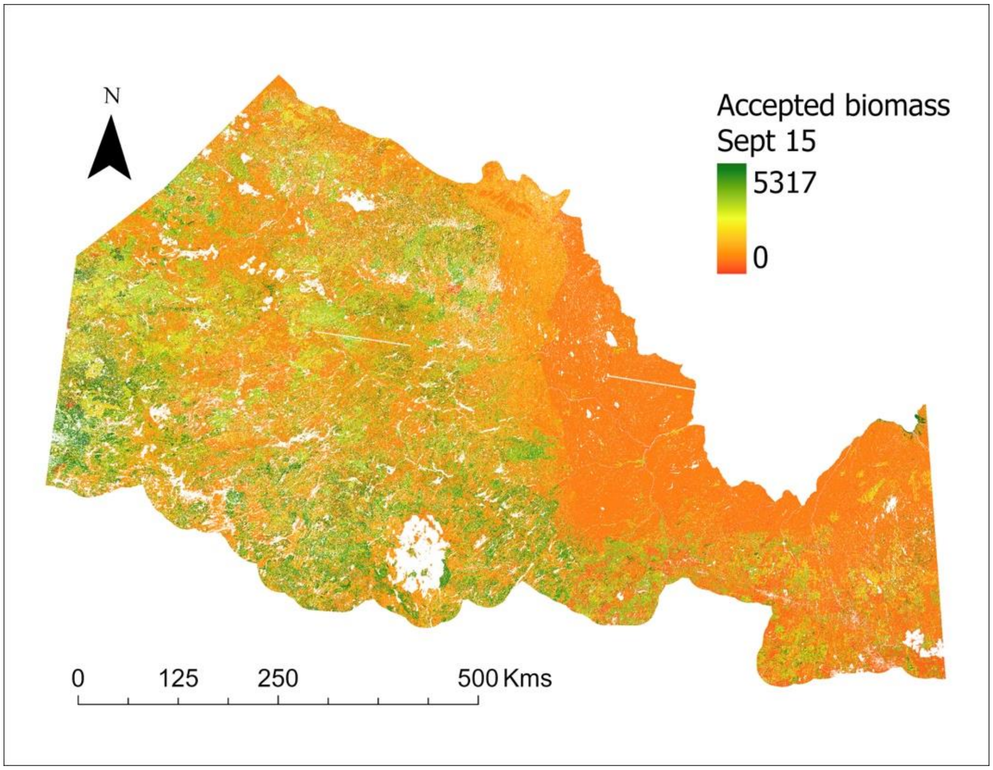


Figure 18. Total accepted biomass (kg/ha) predicted for 1 August (top map) and 15 September (bottom map) across the Ontario AOI.

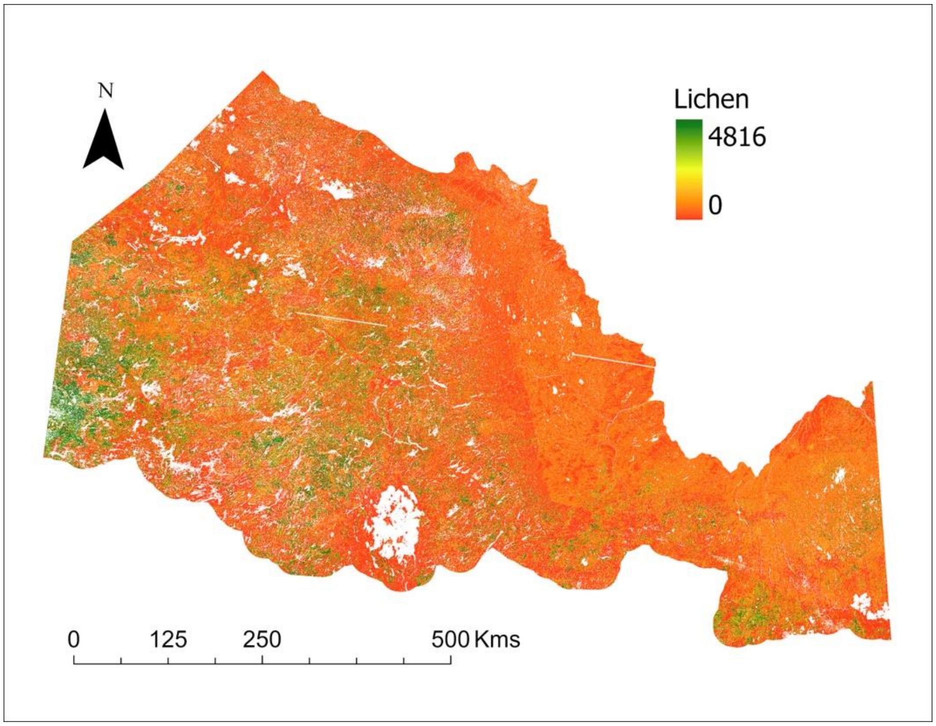


Figure 19. Biomass (kg/ha) of accepted terrestrial and arboreal lichens predicted across the Ontario AOI.

Maps of intake rate.— Below, we present maps depicting predictions of intake rate (g/min) across the AOI for both 1 Aug and 15 Sept.

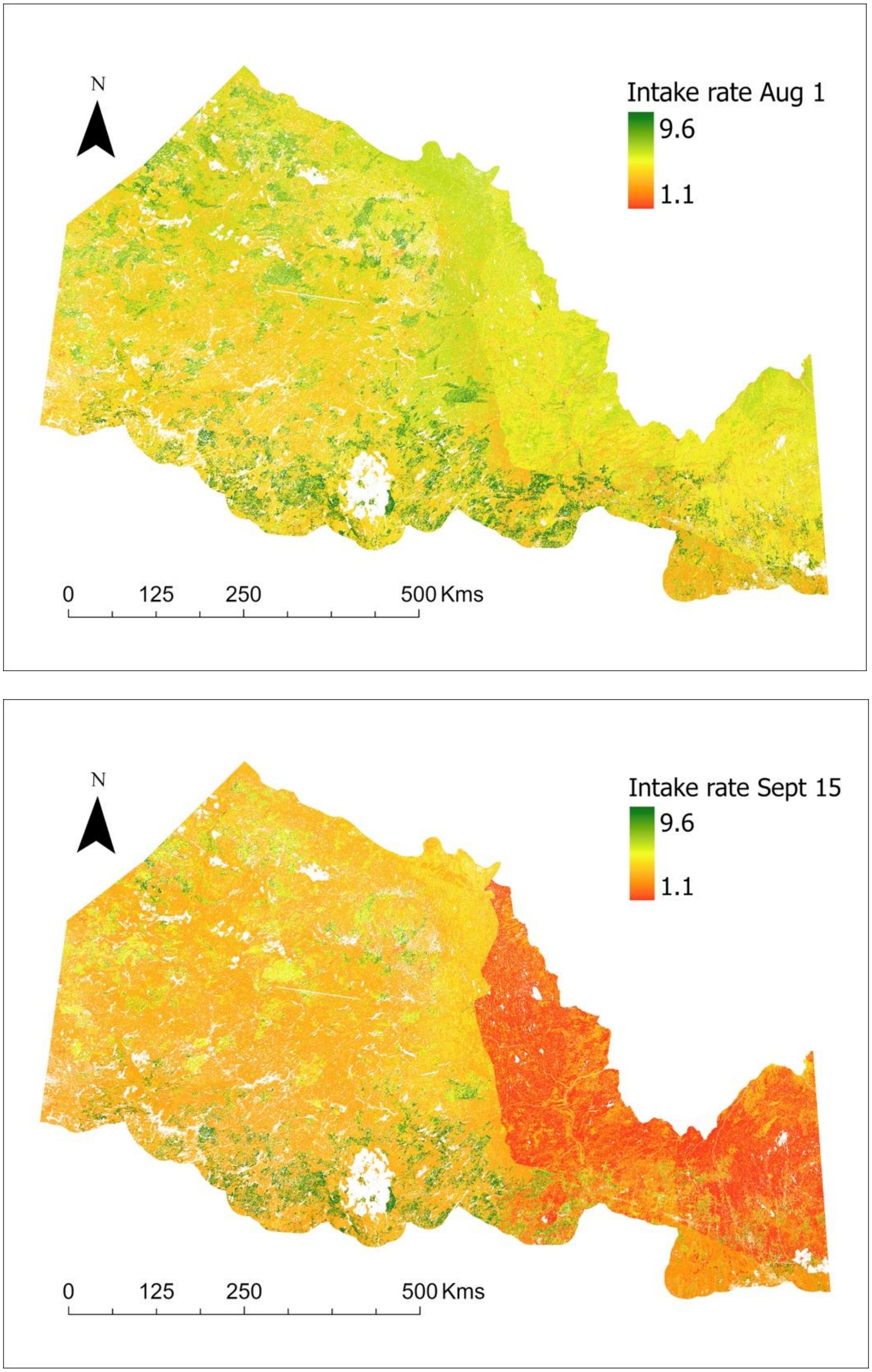


Figure 20. Intake rate (g/min) predicted for 1 August (top map) and 15 September (bottom map) across the Ontario AOI.

Nutritional Implication Maps

Maps of accepted vascular plants.— Below, we present maps depicting predictions of biomass across the AOI for accepted grass, forbs, deciduous shrubs, horsetails, and mushrooms combined (kg/ha) (Fig. 21) for both 1 August and 15 September. These data were used to identify areas where high forage abundance is more likely to benefit moose than caribou (i.e., >400 kg/ha of accepted forage). On both 1 August and 15 September, the proportion of area that we ranked as more beneficial to moose was 11% and 6% in the West and East AOI, respectively (Table 8).

Maps of nutritional adequacy.— The map of nutritional adequacy consists of two separate spatial maps derived from raster layers: (1) four categories (Table 1) of intake rate relative to nutritional requirements, and (2) five categories (Table 2) of forage levels that benefit moose and forage stability in relation to stand-replacing disturbance. The former raster layer is depicted in Fig. 22, and the second raster layer is presented as Fig. 23. Proportion of high (i.e., ≥5.0 g/min) intake rates (g/min) was the same between study regions on 1 August and 0.2 higher in the west AOI compared to the east AOI of Ontario (Table 8). Proportion of very low (i.e., ≤ 3.0 kg/ha) intake rate (g/min) was 0.05 and 0.65 higher in the east AOI compared to the west AOI of Ontario on 1 August and 15 September, respectively (Table 8).

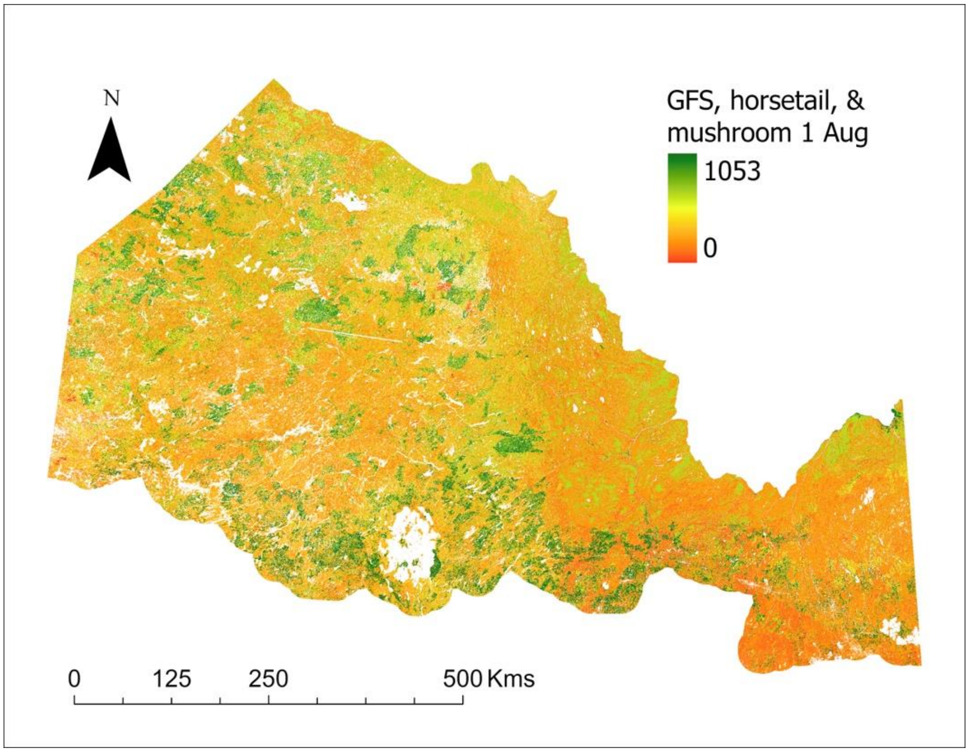
Both the west AOI (65%) and the east AOI (67%) of Ontario had a high proportion of area where forage is likely to increase with disturbance (category 3; Table 8; Figure 23). The east AOI of Ontario had a higher proportion of stable forage regardless of disturbance events (18%) compared to the west AOI of Ontario (9%) (category 1; Table 8; Figure 23).

We present proportion of area among caribou ranges relative to categories of accepted vascular plant biomass (grass, forbs, shrubs, horsetails, and mushrooms), lichen biomass, intake rate, and forage stability (Appendix E). Those herd ranges with the highest proportion of area with biomass of accepted vascular plants >400 kg/ha were Brightsand (17%), Nipigon (17%), and Spirit (14%). Those herd ranges with the highest proportion of area with biomass of accepted vascular plants ≤150 kg/ha were Kesagami (78% on 1 Aug and 88% on 1 Sept) and Pagwacuan (66% on 1 Aug and 86% on 1 Sept). Only one herd range had >10% of its area with lichen biomass >800 kg/ha (Syndey with 13%) while Kinloch, Spirit, Ozhiski, Missisa, Swan, Churchill, Kesagami, Nipigon, and Pagwachuan were dominated (e.g., ≥75%) by areas with lichen biomass ≤150 kg/ha.

Only Brightsand and Njpigon had >10% of their area in the highest intake category (>5.0 g/min) but only on 1 Aug while Kinloch, Churchill, Berens, Kesagmai, Brightsand, Sydney and Pagwachuan all had >10% of their area in the lowest intake category (≤3.0 g/min). By 15 Sept, Missisa (31%), Kesagami (72%), Sydney (19%) and Pagwashuan (58%) had a much higher proportion of the area in the lowest intake category. Across all herd ranges and for both dates, areas supporting 3.0 – 4.0 g/min were the most abundant. All herd ranges had more than half (up to 79%) of the area in forage stability category 3, suggesting biomass of accepted species and thus caribou nutrition could be improved with disturbance, potentially without promoting a moose response.

Table 8. Proportion of area in 4 categories of intake rate (g/min based on 4 categories) and 5 categories of forage stability in response to disturbance within each study region (west AOI and east AOI) across the Ontario AOI for 1 August and 15 September.

Metric	Value	Category	West AOI		East AOI	
			01-Aug	15-Sep	01-Aug	15-Sep
Intake rate (g/min)	>5.0 g/min	Good to Excellent	0.05	0.06	0.05	0.04
	>4.0 – 5.0 g/min	Marginal	0.25	0.18	0.23	0.03
	>3.0 – 4.0 g/min	Poor	0.60	0.73	0.57	0.24
	≤3.0 g/min	Inadequate	0.11	0.04	0.16	0.69
Forage stability	Stable	1	0.09	0.09	0.18	0.18
	Support Moose	2	0.11	0.11	0.06	0.06
	Biomass increase	3	0.65	0.65	0.67	0.67
	Biomass increase and support moose	4	0.07	0.07	0.05	0.05
	Biomass decrease	5	0.08	0.08	0.04	0.04



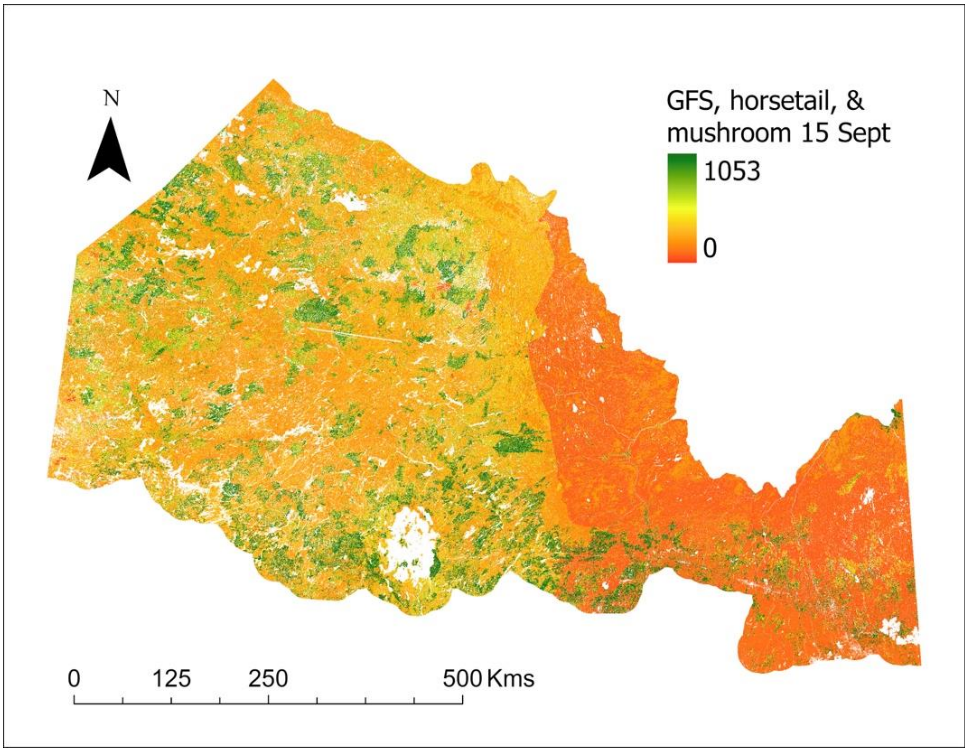
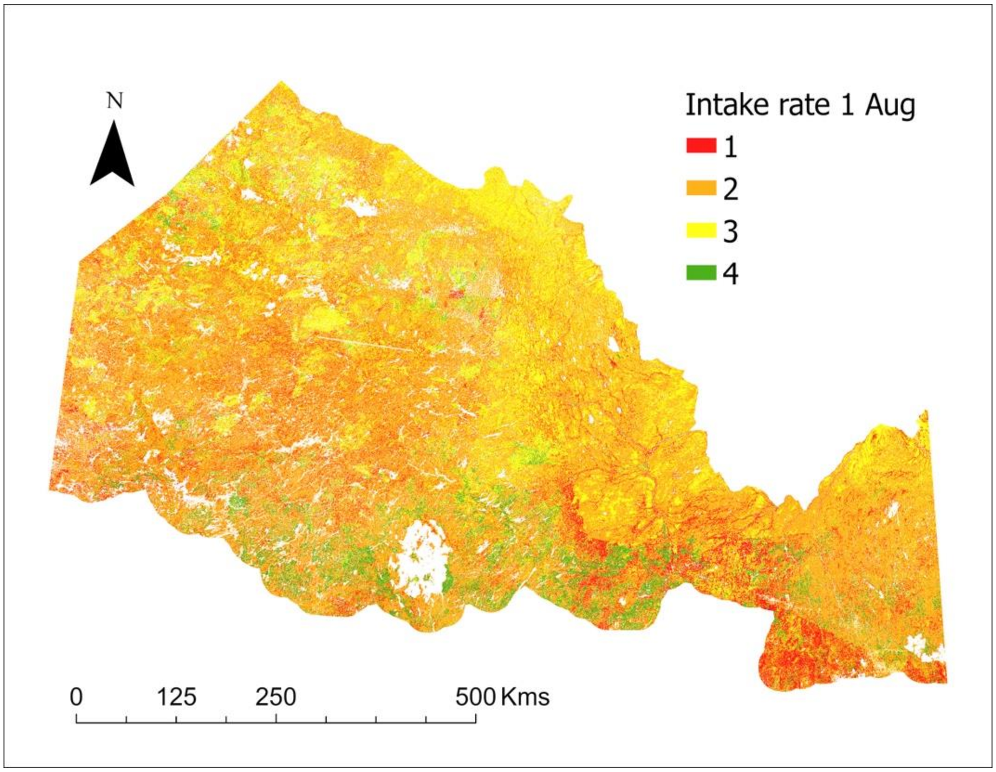


Figure 21. Combined biomass (kg/ha) of grass, forbs, and deciduous shrubs (GFS), horsetails, and mushrooms predicted for 1 August (top map) and 15 September (bottom map) across the Ontario AOI. These data were used to identify areas with very high accepted forage biomass that may benefit moose more than caribou (i.e., >400 kg/ha of forage).



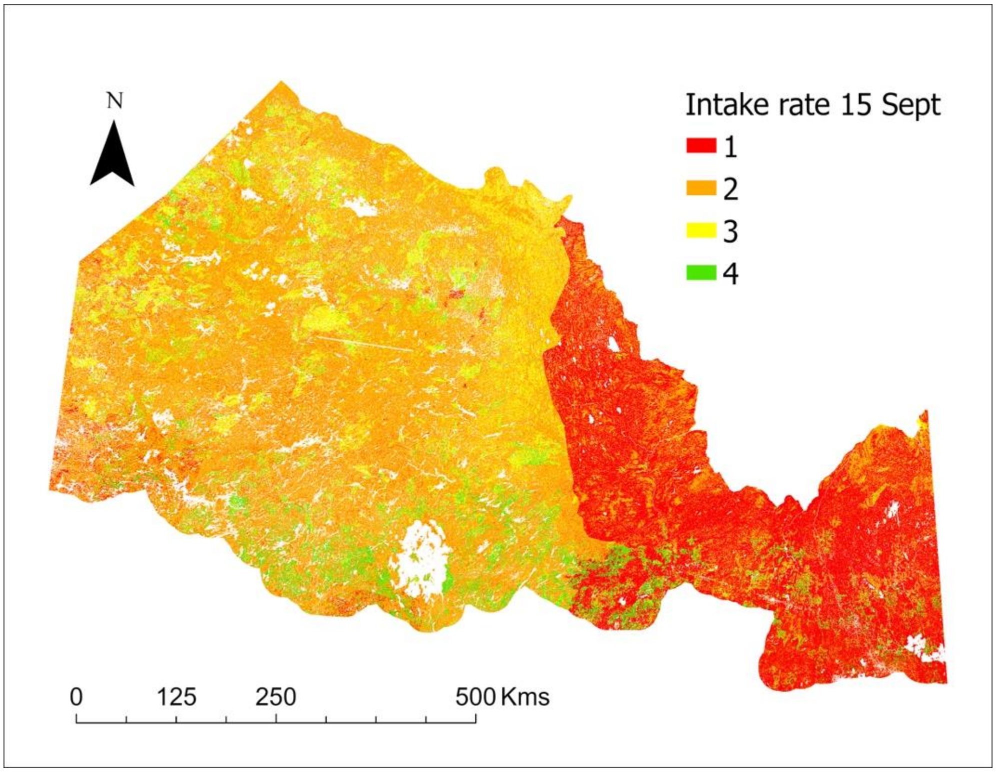


Figure 22. Intake rate (g/min) ranging from good to excellent (4) to inadequate (1) (see Table 1 for category descriptions) predicted for 1 August (top map) and 15 September (bottom map) across the Ontario AOI.

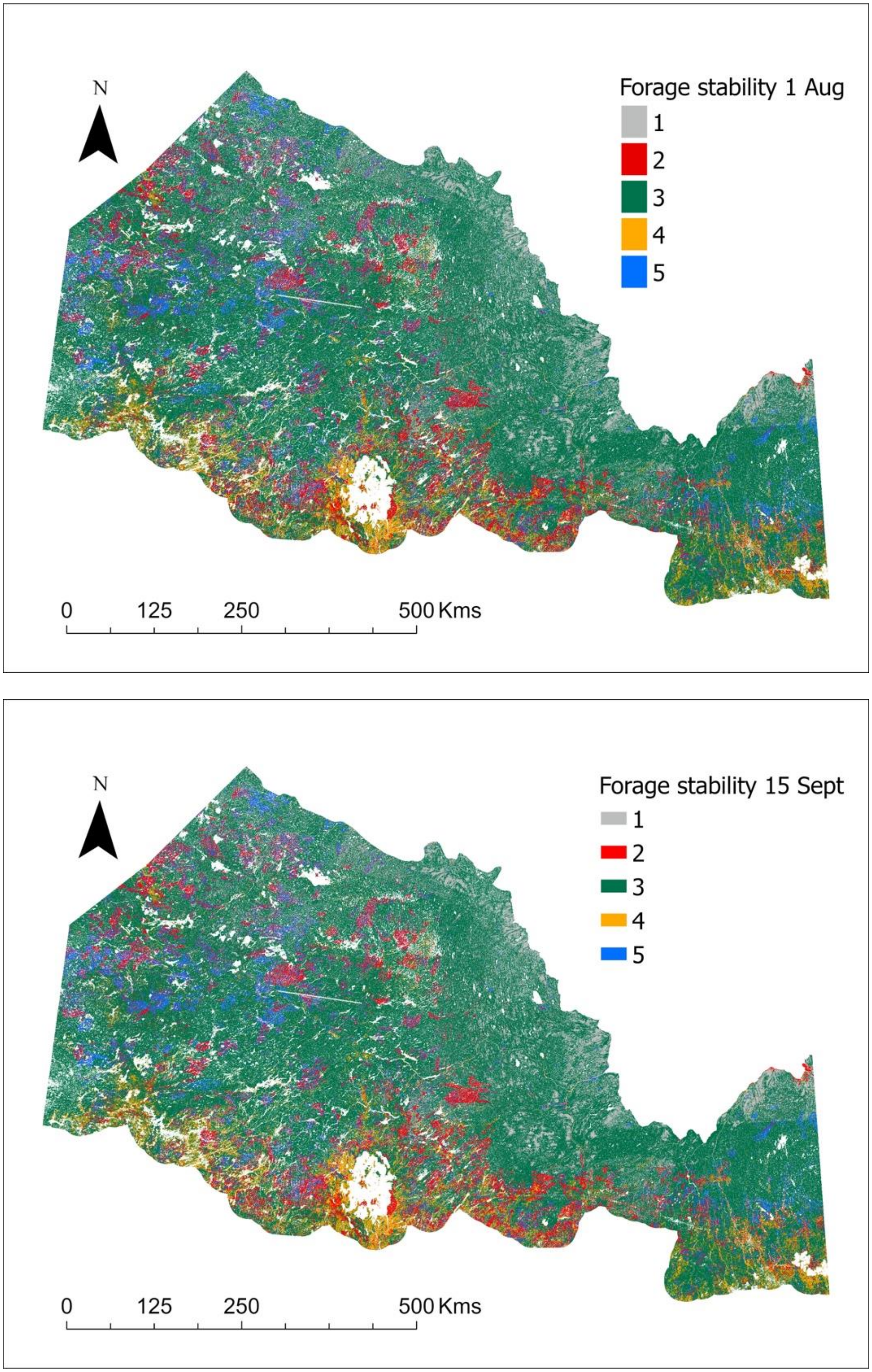


Figure 23. Five categories of forage stability in respond to disturbance (see Table 2 for category descriptions) predicted for 1 August (top map) and 15 September (bottom map) across the Ontario AOI.

In order to visualize both maps simultaneously in a GIS system, we recommend that the forage stability raster and the intake rate raster layers are converted into a polygon shapefiles, which have more flexible display options (e.g., cross-hatching or stippling; Figure 24). However, given the large extent of the Ontario AOI, a single polygon shapefile could not be produced (i.e., the polygon would exceed the 2 GB size limitation of a shapefile). We therefore recommend that users clip these raster layers to their area of interest and then convert both into a shapefile.

In addition, given the number of categories, we found that effective visualization of the map of nutritional adequacy depends on the spatial variability of the forage stability classes. For example, in the southern extent of the Ontario AOI, which has greater spatial variation across forage stability classes, users may need to visualize this map at a smaller window size (e.g., ~2.5 km; Fig. 24). However, in regions with less spatial variability in forage stability classes (e.g., in the northern extent of the Ontario AOI) a user may be able to visualize this map with a larger window size (e.g., > 2.5 km). We therefore recommend that users clip the intake and forage stability raster layers to their specific area of interest and then convert them into a shapefile for visualization purposes.

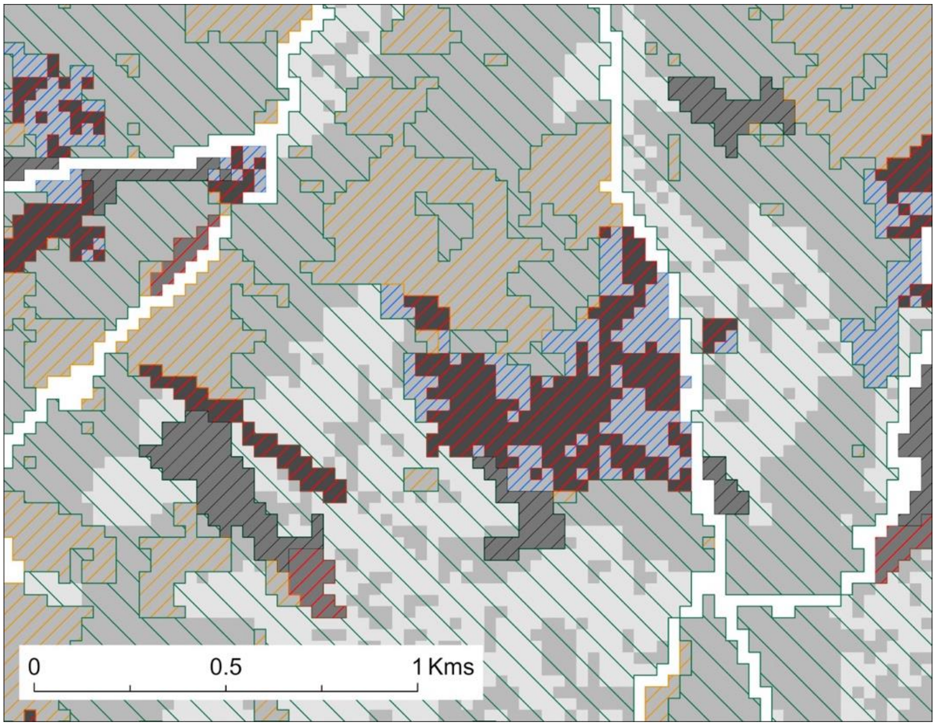


Figure 24. Example map of nutritional adequacy based on 4 categories of caribou intake rate (g/min; four colors) (Table 1) and 5 categories of forage stability (different colors of cross-hatching) (Table 2).

Caribou Selection

Below, we present maps depicting probability of selection for adult female caribou (with calf that survived >5-weeks postpartum) predicted for late summer (23 August) across the Ontario AOI (Fig. 25). For visualization purposes, we also present the same information using four classes of the probability of selection based on quantiles calculated from range of probability of selection values across the Ontario AOI. Value of 4 reflects high probability of selection and 1 is low probability of selection (Fig. 25).

The proportion of high caribou probability of selection was 0.06 higher in the east AOI compared to the west AOI of Ontario, whereas the proportion of low caribou probability of selection was 0.06 higher in the west AOI compared to the east AOI of Ontario (Table 9, Appendix F).

However, we note that caution is needed when interpreting the map of caribou resource selection. The resource selection model was derived using caribou locations from only three of the 12 caribou ranges across the Ontario AOI and caribou selection could change as a function of availability (i.e., functional response). The map of caribou probability of selection assumes the same selection patterns for caribou across all ranges and no functional response in resource selection.

Table 9. Proportion of western and eastern Ontario within 4 categories of probability of selection for adult female caribou (with calf that survived >5-weeks postpartum) predicted for late summer (22 August) based on quantiles calculated across the entire extent.

Metric	Value	Category	West AOI	East AOI
Caribou probability of selection	> 0.66	High	0.23	0.29
	> 0.60 – 0.66	Medium	0.25	0.25
	>0.05 – 0.60	Low	0.25	0.25
	≤ 0.05	Very low	0.27	0.21

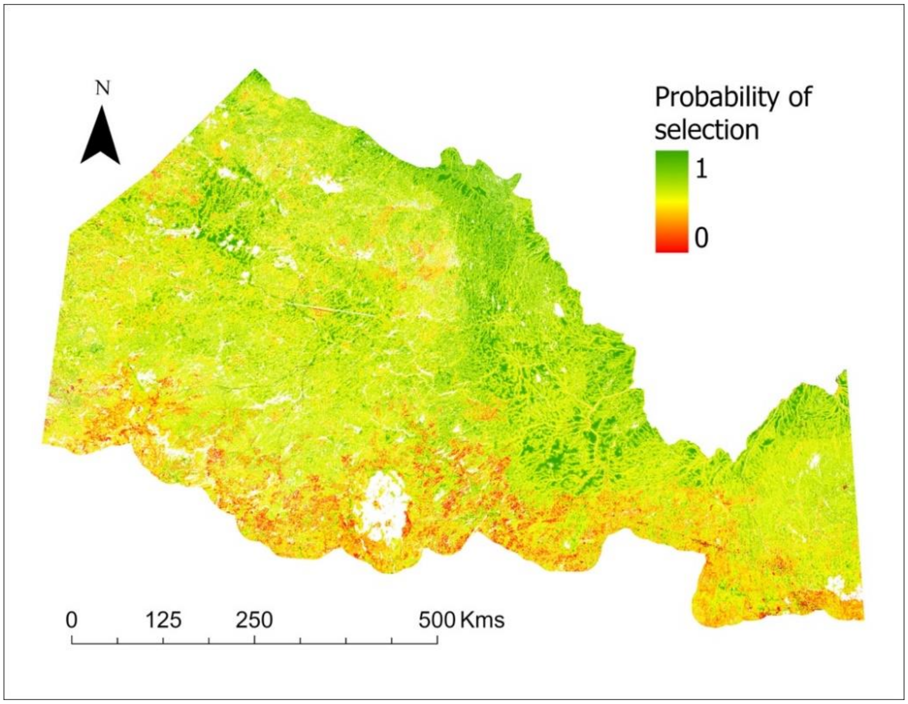


Figure 25. Probability of selection for adult female caribou (with calf that survived >5-weeks postpartum) predicted for late summer (23 August) across the Ontario AOI.

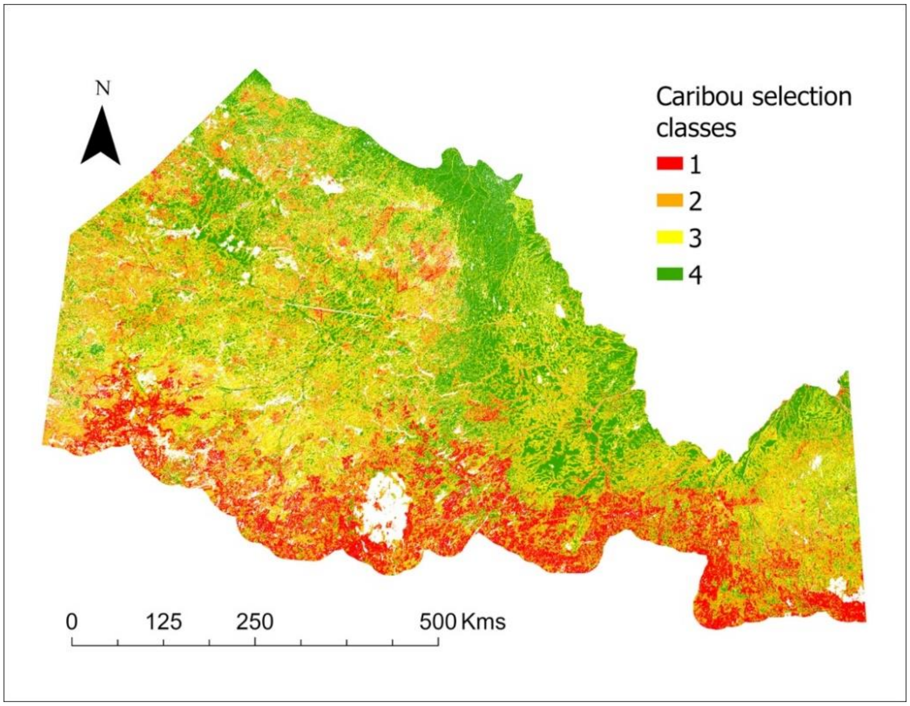


Figure 26. Four classes of probability of selection for adult female caribou (with calf that survived >5-weeks postpartum) predicted for late summer (23 August) based on quantiles calculated from range of probability of selection values across the Ontario AOI. Value of 4 reflects high probability of selection and 1 is low probability of selection.

DISCUSSION

Developing highly detailed, spatially-explicit maps of nutritional resources for research and management is a relatively new area of endeavor that is rapidly becoming more common. Maps of nutritional resources can be very powerful tools for a variety of applications. To date, most applications of foodscape mapping have been for research purposes, primarily to test hypotheses about the spatial distribution of nutritional resources and movements of herbivores (Avgar et al. 2015, Rowland et al. 2018, Johnson et al. 2021) or, less frequently, the state of nutritional resources and herbivore performance (Cook et al. 2018, Merems et al. 2020). The use of these maps for practical management applications has lagged behind the use for research applications, despite the considerable potential for management applications (e.g., Monzingo et al. 2023). In this regard, these maps can provide a basis for identifying key areas that offer superior nutritional resources to be earmarked for protection or areas of poor nutritional resources that could be targeted for treatments to benefit herbivores. For landscape planning, they provide a basis for forecasting nutritional resources and associated impacts on animal performance over time from changes in land use, vegetation disturbance and succession, and climate change. Here, we discuss strategies and assumptions associated with developing the maps, consider current nutritional conditions across our mapping area, and review management implications in the context of nutritional resources for caribou in Ontario.

General Strategies and Assumptions

As with most new science, how best to develop foodscape maps is uncertain and debated. A primary issue is what metric of nutritional resources is most useful—i.e., relevant— for ungulate herbivores. Possible nutrition metrics range from simple estimates of total vegetation biomass, forage quality, bite rates or step rates of herbivores, and actual herbivore nutrition (e.g., intake rate or diet quality). For example, Searle et al. (2007) argued that foodscape maps are most useful if they reflect herbivore behaviour responses to forage (e.g., bite rates or step rates and so forth) rather than forage attributes themselves. This recommendation has rarely been applied in developing foodscape maps because of the challenge of collecting this data on wild herbivores. Instead, foodscape maps of nutritional resources typically are based on surveys of forage quantity, often of a limited number of species that are considered key forage species (Silva et al. 2019), on surveys of forage quality such as protein or energy (Walker 2023), or some combination of the two (Avgar et al. 2015).

Our approach relied on captive caribou placed in native habitats to measure actual caribou nutrition as a function of the forage types and abundance available to them. This allowed: (1) rigorous statistical evaluation of diet selection (e.g., what plants caribou select, what plants they eat in proportion to availability, and what plants they avoid; (2) the nutritional content of diets that they consume; (3) the rate of intake of forage and nutrients such as digestible energy and protein; and (4) the ability of caribou to satisfy nutritional requirements among plant community types. These key nutrition metrics cannot be evaluated, only inferred (poorly in many cases), from data collected during standard surveys of forage quality and quantity, nor can they be reliably estimated from sampling of wild, free-ranging caribou. In addition, sampling with captive caribou across a variety of native vegetation community types supports identifying which nutrition metrics are most limiting and most predictable across ecosites and seral stages. Certainly, use of captive animals introduces concerns about how well they represent their wild counterparts. Yet, comparative studies show that diet selection and foraging dynamics of tame cervids are indistinguishable from wild cervids (Bergerud and Nolan 1970, Wallmo and Neff 1970, Bergerud 1972, Olson-Rutz and Urness 1987, Spalinger et al. 1997).

Our approach for developing spatially explicit maps of nutritional resources largely depended on characterizing the overstory and undergrowth characteristics, caribou nutrition, and nutritional implications for each ecosite. For this to be effective, we strove to develop ecosite groups for mapping that reflected substantial differences in site potential (largely a function of soil type and microclimate) that in turn caused biologically significant differences in forage composition, production, and phenology, and provided a basis for predicting changes in forage resources following stand-replacing disturbance and across subsequent succession. Vegetation classification systems explicitly based on site potential include the biogeoclimatic system of British Columbia (Pojar et al. 1987) and the potential natural vegetation systems of the northwestern United States (Cooper et al. 1991). The ecosites of Racey et al. (1996) and Taylor et al. (2000) for Ontario reflect both cover types and site potential, and the very large number of ecosites even within regions of Ontario were far too numerous to be practical for describing nutritional resources for caribou or mapping these resources. Our grouping of ecosites was based on differences in undergrowth composition as described by Racey et al. (1996) and Taylor et al. (2000) but also reflected broad differences in site potential. We acknowledge that our amalgamation was a subjective effort, but the groupings greatly simplified evaluating and mapping nutritional resources and reflected important differences in forage resources for caribou, often with important management implications (Table 4).

Within the framework provided by the ecosite groups, the nutritional value of habitat was largely based on the per-minute intake rate of forage. This deviates notably from work we have conducted elsewhere, where nutritional value of habitats was based on the digestible energy content of diets (e.g., for elk in northwest United States; Cook et al. 2018). Even though it may seem that a quality-based metric (i.e., energy, protein) might provide a better nutrition metric than the simpler forage intake rate, we found in Ontario that variation in dietary quality varied little among ecosites and seral stages and so too did our ability to predict it (Walker 2023). We believe this is largely because quality of forage tends to be higher in Canada than in more southerly latitudes (Klein 1970), and particularly because caribou typically ate accepted forage species, and thus the quality of their diets typically satisfied or exceeded requirements (Fig. 11). In contrast, our data indicated that how fast caribou consumed forage was often far below requirements and varied greatly among ecosite groups and seral stages and was a reasonably predictable function of forage abundance. How rapidly ruminant herbivores can ingest forage often has a primary effect on the nutritional well-being of the animal (Minson and Wilson 1994). Thus, unlike for our previous work, biomass of accepted forage (due to its influence on forage intake rate) rather than forage quality or quality of diets, was the more important nutritional metric for our purposes in Ontario.

Our efforts to flag habitats that might be valuable for moose to meet their nutritional requirements and as a result, potentially detrimental to caribou also was based on biomass of vascular forage and its expected effect on per-minute forage intake responses of moose. A fundamental challenge is that increasing biomass of accepted vascular forage generally corresponds to increasing forage intake rate of both species. Our efforts for flagging areas of high value for moose were based on the assumption that moderate biomass of accepted forage of vascular plants (e.g., 200–300 kg/ha) will benefit the smaller-bodied caribou to a greater extent than moose, whereas very high levels of accepted forage of vascular plants (e.g., >400 kg/ha) will benefit moose to a much greater extent than caribou. This represents a working hypothesis that has not been evaluated for landscape applications. In addition, several other assumptions underpin this approach: (1) the biomass of accepted vascular plants we estimated for caribou provides a reasonable estimate of this biomass for moose; and (2) relationships between accepted forage biomass and per-minute intake rates of elk serve as an adequate model for moose (no such data for lactating moose in summer exist to our knowledge). Moreover, the *amount of area* that is created after stand-replacing disturbance also should greatly influence the benefits and detriments to moose and caribou. A low density of patches of very abundant forage may benefit caribou and provide little value to the larger-bodied moose, and vice versa. What remains far from certain is how much forage improvement over how much area can provide a benefit to caribou without supporting significantly higher moose numbers. Our mapping does not address this issue; instead, we simply flagged seral stages and ecosites that are likely to overly benefit moose based on abundance of accepted vascular forage.

Patterns of habitat selection by caribou in Northern Ontario seem to support fundamental concepts incorporated into our mapping products. Walker (2023) showed that of the variety of nutrition metrics that he initially evaluated (forage intake rate, biomass of accepted vascular plants, biomass of accepted lichens, digestible energy and digestible protein content of accepted species), habitat selection by caribou in the Pickle Lake and Cochrane area was best correlated to predicted forage intake rate (positive correlation between foraging intake rate and caribou use). He also showed that for lactating females with a calf at heel in mid-to-late summer, selection for areas of moderate forage intake rate was stronger than selection for areas with low predation risk. However, selection tended to decline in areas of very high predicted forage intake rate. These areas normally have very high levels of biomass of accepted vascular plants, suggesting in turn that selection by caribou tended to decline in areas that might be particularly beneficial to moose. It is difficult to know if this decline in selection was to avoid moose or some other attribute of habitat that caribou avoided (such as very dense vegetation that might impede their movements or ability to detect predators), but the finding provides support for the hypothesis integrated into our nutritional implications map: areas that provide very high forage biomass may not be particularly beneficial to caribou.

Normally, lichens are considered *the* primary food for caribou. We agree that in winter, lichens are an especially valuable forage; their digestible energy content in winter is higher, far higher in most cases, than other available forage. Even during summer, our data indicate that accepted species of arboreal and terrestrial lichens are well eaten, as do other studies (Thompson et al. 2015, Denryter et al. 2017). But for a summer forage, both types of lichens have three important shortcomings: virtually no digestible protein; increased handling times to ensure caribou are eating only lichen and not substantial amounts of moss, pine needles, or detritus; and low bite mass that substantially impedes forage intake rates. In regard to the latter, bite mass of lichens are so low that not only is per-minute intake rate impacted if they eat lichen, but lactating caribou may be unable to satisfy total daily forage intake requirements if they consume lichen-dominated diets, because there are not enough hours in the 24-hour period for caribou to eat to make up for the low per-minute intake rate. Accepted lichens did not significantly contribute to forage intake rate when other predictor covariates were included during development of forage intake prediction models, and thus this forage class does not contribute to our maps of caribou nutrition. We nevertheless mapped lichen biomass because of its importance for winter nutrition, and below we discuss the need for sustaining lichens on ranges occupied by caribou during winter.

The accuracy of these types of mapping efforts is always a key issue, especially so in far north areas where funding is often limited to develop detailed maps of habitat conditions. For our purposes, two primary issues are of concern: the accuracy of online sources of habitat conditions such as overstory canopy cover, soil type, and ecosites, and our predictive equations of forage abundance and caribou nutrition. We conducted an analysis of accuracy based on observed values at field macroplots compared to values for these sites indicated in the online sources of habitat characteristics, and by comparing predicted values to observed values for our equations of forage attributes and caribou nutrition. For ecosites (Table 7), which underpins nearly all predictions associated with our maps, overall user classification accuracy was 41%, and ranged from 34% to 81% for ecosites well represented in our sample (i.e., >30 macroplots per ecosite). At first glance, this may be encouraging, but the Far North Layer could not reliably distinguish among most lowland ecosites (Table 7). Plots of overstory canopy cover and basal area indicated moderate levels of association between observed at field sites and online sources for these sites, with r^2 values of 0.48 to 0.59, and evidence for bias (over and underestimating either attribute) was minimal (Fig. 13). However, these plots indicated frequent, large errors in estimates from the online sources. Finally, our comparisons of predicted versus observed for each category of forage biomass and forage intake rate generally ranged from modest (r^2 = 0.30) to good (r^2 = 0.64) (Tables 5, 6; Appendix C). We note that prediction of biomass of accepted grasses, forbs, and shrubs, the forage class that with the greatest influence on our final maps, was relatively high (r^2 = 0.57–0.63).

Given these results, managers should be aware that the accuracy of our maps is modest at best. If the maps are used to predict habitat attributes and nutritional resources at very small scales (i.e., individual stands), then managers should expect frequent errors. Selecting areas for habitat treatments, for example, should ultimately depend on "in-the-field" site selection, not on the predictions from the map. On the other hand, the larger the area of application, the more the errors should "average out" and provide a reasonably reliable depiction of nutritional resources. Thus, the maps and mapping strategy are probably better suited for large-scale planning and summarizing habitat conditions than site specific activities.

Current Status of Nutritional Resources in Ontario

A fundamental question that arises when depicting forage resources across a landscape pertains to the extent and magnitude of nutritional deficiencies, and whether these deficiencies may reduce performance, reproduction, and survival to a biologically significant degree in Ontario. If the extent and magnitude is low, then the need for the information and insights our maps provide is also low. There have been no detailed, province-wide evaluations of nutritional limitations on caribou populations in Ontario of which we are aware. Walker (2023) noted that lower levels of body mass, body fat, and pregnancy rate in combination with delayed parturition in the Cochrane region versus around Pickle Lake corresponded to lower biomass levels of accepted vascular plants and lichens and lower predicted forage intake rate at Cochrane versus Pickle Lake. However, his finding only suggests at causality. The data in Fig. 1 show substantial variation in body fat levels of female caribou during winter among populations across Ontario. Body fat levels of caribou in mid-to-late winter are usually a result of body fat levels in autumn, and thus a function of nutritional resources available to caribou the previous summer and early autumn (Couturier et al. 2009, Cook et al. 2021a). As such, the variation in body fat levels in female caribou (Fig. 1) is very likely due to differences in the quality of nutritional resources available to these animals during summer and early autumn (Couturier et al. 2009; Cook et al. 2013, 2021a). These data suggest some populations exist on marginal nutritional resources and others exist on substantially better nutritional resources during the growing season. Certainly, additional studies are needed to confirm this perception, but body condition studies across other areas of North America indicate similar variation (Fig. 1; Cook et al. 2013). Whatever the case, as ecological conditions that affect forage quality and quantity vary across ecoregions, so too does the adequacy of nutritional resources for supporting ungulate populations across the continent.

By calculating percent of area relative to our intake rate and forest stability categories for each caribou range in our Ontario AOI, we found a few broad patterns potentially of interest (Appendix E). For 15 Aug, 28% to 45% (mean = 38%) of area in the northern tier of herd ranges (Missisa, Oghiski, Spirit, and Swan) provided >4.0 g/min (marginal to excellent levels) of predicted forage intake rates. The area for the remaining ranges composing the southern tier averaged about half of this (19.7%), yet also demonstrated substantial variation in nutritional resources (range 8% to 32%). Flipping the perspective and looking at percent of area in the lowest intake rate categories, these data also indicated that from about 50% to 90% of area, depending on the caribou range, provided forage intake levels that are likely to be far below that needed by lactation females in mid-August, suggesting important nutritional limitations may occur on some of the caribou ranges. The difference between the northern and southern tiers is primarily due to differences in percent of area providing at least 275 kg/ha of accepted vascular plants (Appendix E), perhaps due to lower tree density and thus potentially greater forage production. How this translates to differences in caribou performance such as body condition, reproduction, and survival is unknown, although we note that by far the highest body fat levels we measured in Ontario in 2009-2010 were from Big Trout Lake (Fig. 1), located at the extreme northern extent of our mapping area (all other body fat data were collected in the southern tier of caribou ranges). In addition, our data indicate a modest to marked decline in nutritional resources based on forage intake rate as the season progressed from 15 August to 15 September. By 15 September, 70% to 93% of area provided forage intake rates likely to be far below that required by lactating females in summer. This finding suggests a period of increasing nutritional limitations in late summer that likely challenge lactating females to recover body fat and calves to grow in preparation for winter (Cook et al. 2021a). Much more research is needed to understand how variation in nutritional resources translates to performance of caribou.

Management Approaches

To a large degree, evaluating summer habitat and improving or changing habitat in the context of nutritional resources boils down to evaluating and managing the biomass of accepted vascular plants. Generally, for ungulates, the simple rule is the more acceptable forage (i.e., vascular plants) available, the better the nutritional resources leading to a more productive herd (e.g., Cook et al. 2018). But for boreal caribou, some additional caveats may apply. In particular, in some settings, higher amounts of accepted vascular plants could invoke increases in alternative prey such as moose or white-tailed deer and, importantly, their predators that could be detrimental to caribou. In these situations, areas providing moderate biomass levels of accepted vascular plants in floristically diverse plant communities may be the most important nutritional habitat for caribou (Walker 2023), not areas providing very high levels of forage biomass

Our data clearly show that stand-replacing disturbance plays a key role in providing abundant forage. Thus, wildfire and regeneration harvest may play a crucial role in sustaining good nutritional resources. The extent to which wildfire versus regeneration harvest differentially influence the forage base is poorly studied in North America (we strove to include recently burned areas in our sampling to evaluate this, but these areas were very rare at the time of our work), and we assumed that their effect on forage quality and quantity is reasonably similar. However, with timber harvest comes road development and other human activities, and these have great potential to negate any benefits arising from the potentially positive effect on forage resources from disturbance. Discussions contained herein assume that roads needed for timber extraction are minimized, public use is disallowed, roads that are created are decommissioned and natural vegetation is quickly restored to the extent possible, and structural impediments are used to eliminate their value as travel corridors for large predators. In the following, we provide guidance on the use of our mapping products to evaluate, manage, and plan for nutritional resources on behalf of caribou (Appendix D provides specific recommendations for individual ecosites). We assume that use of the maps will normally involve a reasonably large area for nutritional evaluations (ANE) (e.g., >100 km^2); using the maps for evaluations at substantially smaller scales may introduce mapping error as described above.

In general, the maps (i.e., raster layers) depicting intake rate categorized relative to nutritional adequacy, forest stability classification, and caribou habitat selection (or the predation risk raster layer that feeds into the selection layer; Appendix G) can be used for objectives such as evaluating current nutritional resources or for evaluating changes over time either for retrospective (in the past) or forecasting into the future to evaluate likely changes in nutritional resources given natural or human-caused changes in disturbance regimes.

Evaluating current nutritional resources in a single ANE may be an objective for two primary purposes: to identify areas for protection, i.e., to maintain and provide caribou access to habitats with adequate levels of nutritional resources, and to identify areas where habitat management could be used to improve nutritional resources. For the former, we recommend locating areas that have a high probability of selection, i.e., areas of good nutrition (moderate to high forage intake rate; >3.0 g/min) but moderate to low predation risk. These areas should be a priority for maintaining into the future. However, if these areas are in ecosites where biomass is negatively affected by canopy closure, they may only contain adequate nutritional resources for decade or two as succession progresses. We thus envision multiple categories of habitat in areas with a high selection probability, including: (1) good to high nutrition of *short-term* duration with low predation risk (probably due to stand-replacing disturbance in areas without human activities); (2) good to high nutrition that is *stable* over time if undisturbed and with low predation risk; and (3) good to high nutrition that will convert to moose nutrition areas if subjected to stand replacing disturbance. Stand replacing disturbance may be required to maintain the nutritional value over time if the areas are in category 1, whereas stand replacing disturbance is probably of little value for areas in either category 2 or 3.

Similarly, these maps can be used to identify areas where habitat management in the form of stand-replacing disturbance may be most useful to improving nutritional resources for caribou. For example, there may be little point in trying to improve nutritional value of areas where predation risk is high (i.e., causes low probability of selection) unless predation risk can simultaneously be lowered (e.g., close and reclaim roads). Alternatively, greater potential exists to effectively improve nutritional resources on behalf of caribou in areas where predation risk and nutritional resources are both low. Our maps of forage stability provide some initial insights of the probability that stand-replacing disturbance will significantly improve forage resources for caribou and if such disturbance will produce habitat that best supports moose. For example, stand replacing disturbance targeted to the relatively unproductive upland forest ecosites (upland - black spruce-rocky and sandy) and lowland ecosites (lowland bogs, poor swamps, fens) may be beneficial by providing a moderate boost in production of accepted vascular plants. In contrast, the productive upland-black spruce-white spruce ecosite and productive swamp thickets (if burned) may unduly benefit moose. Stand replacing disturbance in these productive sites might be avoided, or perhaps herbicides could be used to reduce undergrowth production to some degree (e.g., <400 kg/ha) to make them less valuable for moose.

The maps also may be used to compare current nutritional resources among areas, much like we did to compare among Ontario caribou ranges (Appendices E, F). The objectives may vary, but this approach might be used to better understand the range in predicted caribou nutrition and other habitat values addressed by our models across a subset of ranges or areas of interest. For example, managers may be interested in knowing biomass levels of forage, predicted caribou nutrition, and predation risk across a wide swath of habitat just north of active forestry operations to compare to those in areas where forestry has been ongoing.

Another primary value of the maps may be to evaluate changes in nutritional resources over time as a function of changes in land use patterns and natural disturbance patterns. Although more of research interest, using the maps to evaluate changes in nutritional resources over past decades may provide insights that short-term investigations, typical in wildlife research, do not. Such retrospective studies may offer insight into how changes made to the landscape historically may be impacting current condition and caribou populations.

However, change-over-time evaluations may be of greatest interest for forecasting future states of caribou nutrition, predation risk, and habitat use in response to planned forest management projects or in response to simulated wildfire events, the latter perhaps to evaluate climate change effects on nutritional resources and habitat use for caribou. We note that doing either will require developing alternative maps than those we have submitted to reflect changes in habitat conditions over time. Forecasting maps that apply changes in habitat conditions (primarily overstory canopy cover, basal area, seral stage categories as used in the Far North vegetation layer, and changes in roads and other anthropogenic features to incorporate changes in predation risk) would have to be developed that reflect the changes for which planners may wish to forecast (Appendix H).

Management actions described here focus on summer ranges, yet management of habitat should also aim to sustain lichen resources for winter use by caribou. Our map of accepted lichen biomass is the only mapping product we provide that pertains to this key winter forage. It can be used to flag areas that may currently have abundant terrestrial lichens. Using the lichen map, our analysis indicated a high proportion of area in each caribou range with very low accepted lichen biomass (Appendix E), although some geographic variation is apparent. For example, habitats with low biomass (<150 kg/ha) of accepted lichen comprised >90% of land area in caribou ranges in the eastern portion of our mapping area (Pagwachuan, Kesagami, and Missisa) but comprised 44% to 84% of land area in most of the central and western herd ranges (Appendix E). Proportion of area with relatively high lichen biomass (>400 kg/ha) ranged from 3% to nearly 40% among these ranges, evidently indicating a remarkable range in lichen abundance across our mapping area. It is beyond the scope of this document to review management effects on lichens, but we emphasize that forest management is usually much less detrimental to terrestrial lichens than wildfire. In addition, stand-replacing disturbance may reduce lichens in the short-term but may be required to maintain lichens over the long-term, and effects of habitat management on terrestrial lichens is highly dependent on soil type, microclimate, and overstory conditions (Sulyma and Coxson 2001, Coxson and Marsh 2001, Chichowski et al. 2022). Coxson (2015) provided detailed recommendations regarding best practices for clearcutting or partial harvesting to reduce the damage to lichens and enhance their recovery.

Our data were collected from existing stands in our study region, most of which were created by regeneration harvest and wildfire, and thus our evaluation of approaches to manage nutritional resources for caribou was based only on stand-replacing disturbance. We have little or no data to compare the effects of various commercial timber harvest strategies, silvicultural prescriptions, and differences between logging and wildfire on nutritional resources. The work that does exist generally suggests that the greater the removal of the overstory, the greater the response of undergrowth, particularly shade-intolerant plants that, in general, are more palatable than shade-tolerant species (Irwin and Peek 1979, Cook et al. 2016, Hull et al. 2020). Thus, in general, we would recommend stand-replacing disturbance for less productive ecosites (upland-black spruce-rocky and sandy; most lowland forest types) and perhaps some form of partial harvest or patch selection on highly productive sites (upland-black spruce-mesic; upland-black spruce-white spruce) to reduce response of the undergrowth. In addition, strategies that extend the early seral window, particularly on less productive upland ecosites, may help extend the period of elevated forage beyond the first 20–25 years after disturbance. In this regard, we have observed that high density planting of Jack pine on upland sites that clearly support abundant terrestrial lichens (evident where holes in the forest canopy allow light penetration) suppress these lichens. In fact, some of the greatest production of lichens sampled during this project was on a 24-year-old logged site that had been planted but pre-commercially thinned (Fig. 27). Although perhaps counter to current reforestation regulations, sparse and slow regeneration of trees on sites that can support abundant terrestrial lichens may be required to allow lichens to thrive.

Figure 27. Early seral stand (24 years old) in Jack pine forest on deep sandy soils showing abundant terrestrial lichens (4,500 kg/ha) during September 2017 near Sioux Lookout, Ontario. The stand had been clearcut logged, planted to Jack pine, and pre-commercially thinned and, in 2017, provided among the greatest biomass of accepted terrestrial lichens sampled during this study.

Forest disturbance has the potential to both improve habitat for caribou and worsen it, and the compendium of evidence of past practices suggests that the net effect is more often negative (Stevenson et al. 2024). However, much remains to be learned about how to use various forest management practices and management of roads and human activities to better support the positive benefits of forest disturbance, such as improved forage production, while at the same time, reducing the negative impacts on behalf of caribou.

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Appendix A. Forage class and caribou selection of plants in Ontario.

Table A1. Forage class (life form group) and caribou selection category of plants sampled in Northwestern Ontario during summer in 2017 and 2018.

Scientific Name	Common Name	Forage Class[a]	Diet Selection[b]	*n*[c]
Acer spicatum	Mountain maple	Dec. Shrub	Sel	12
Alnus crispa	Green alder	Dec. Shrub	Sel	56
Alnus rugosa	Speckled alder	Dec. Shrub	Sel	21
Amelanchier sanguinea	Roundleaf serviceberry	Dec. Shrub	Sel	46
Amelanchier spp.	Serviceberry	Dec. Shrub	Sel	36
Aster macrophyllus	Big leaf aster	Per. Forb	Sel	19
Betula papyrifera	Paper birch	Dec. Shrub	Sel	58
Cladina mitis	Green reindeer lichen	Terr. Lichen	Sel	71
Cladina rangiferina	Grey reindeer lichen	Terr. Lichen	Sel	77
Cladina stellaris	Reindeer lichen	Terr. Lichen	Sel	42
Cladonia spp.	Cladonia lichen	Terr. Lichen	Sel	79
Clintonia borealis	Blue-bead lily	Per. Forb	Sel	53
Evernia mesomorpha	Boreal oakmoss lichen	Arb. Lichen	Sel	76
Hieracium spp.	Hawkweed	Per. Forb	Sel	29
Mushroom (various spp.)		Fungus	Sel	78
Parmeliaceae family[d]	Lichen-forming fungi	Fungus	Sel	na
Populus tremuloides	Trembling aspen	Dec. Shrub	Sel	55
Prunus pensylvanica	Fire cherry	Dec. Shrub	Sel	42
Rosa acicularis	Wild rose	Dec. Shrub	Sel	52
Salix bebbiana	Bebb’s willow	Dec. Shrub	Sel	19
Salix discolor	Pussy willow	Dec. Shrub	Sel	21
Salix humilis	Prairie willow	Dec. Shrub	Sel	55
Sorbus decora	Showy mountain ash	Dec. Shrub	Sel	39

Umbilicaria hyperborea[d]	Rock tripe	Fol. Lichen	Sel	na
Usnea spp.	Old man’s beard	Arb. Lichen	Sel	78
Achillea millefolium	Yarrow	Per. Forb	Neu[e]	6
Actaea rubra	Red baneberry	Per. Forb	Neu[e]	5
Apocynum androsaemifolium	Spreading dogbane	Per. Forb	Neu	28
Aralia hispida	Bristly sarsaparilla	Dec. Shrub	Neu	17
Asarum canadense	Canadian wild ginger	Per. Forb	Neu[e]	2
Aster ciliolatum	Lindley’s aster	Per. Forb	Neu	20
Aster puniceus	Purple-stemmed aster	Per. Forb	Neu[e]	2
Aster spp.	Aster	Per. Forb	Neu[e]	7
Athyrium filix-femina	Lady fern	Fern	Neu	11
Betula pumila	Bog birch	Dec. Shrub	Neu[e]	2
Bromus spp. mainly *B. richardsonii, B. ciliatus*	Brome grass	Grass	Neu[e]	4
Calamagrostis canadensis	Bluejoint	Grass	Neu	24
Chrysanthemum leucanthemum	Oxeye daisy	Per. Forb	Neu[e]	2
Cornus stolonifera	Red osier dogwood	Dec. Shrub	Neu	10
Corylus cornuta	Beaked hazelnut	Dec. Shrub	Neu	9
Cypripedium acaule	Pink lady’s slipper	Per. Forb	Neu	35
Danthonia spicata	Poverty oat grass	Grass	Neu[e]	5
Danthonia spp.	Oat grass	Grass	Neu[f]	0
Diervilla lonicera	Northern bush honeysuckle	Dec. Shrub	Neu	50
Epilobium angustifolium	Fireweed	Per. Forb	Neu	53
Equisetum arvense	Field horsetail	Graminoid	Neu[e]	7
Equisetum pratense	Meadow horsetail	Graminoid	Neu[e]	3
Equisetum sylvaticum	Horsetail	Graminoid	Neu	33
Eriophorum spp.	Wooly sunflower	Graminoid	Neu[e]	2
Geum macrophyllum	Large leaf avens	Per. Forb	Neu[e]	2
Impatiens capensis	Jewelweed	Per. Forb	Neu[e]	2

Lathyrus ochroleucus	Cream pea	Per. Forb	Neu[e]	7
Lonicera canadensis	American fly honeysuckle	Dec. Shrub	Neu[e]	2
Lonicera spp.	Honeysuckle	Dec. Shrub	Neu[f]	0
Luzula acuminata	Hairy woodrush	Graminoid	Neu[e]	6
Menyanthes trifoliata	Bogbean	Per. Forb	Neu[f]	0
Mertensia paniculata	Tall bluebells	Per. Forb	Neu[e]	5
Monotropa uniflora	Ghost plant	Per. Forb	Neu[e]	4
Osmunda claytoniana	Interrupted fern	Fern	Neu[e]	7
Peltigera spp.	Peltigera	Peltigera	Neu	15
Petasites palmatus	Palmate coltsfoot	Per. Forb	Neu	21
Polygonum cilinode	Fringed black bindweed	Per. Forb	Neu	16
Potentilla tridentata	Three-leaved cinquefoil	Per. Forb	Neu[e]	4
Prunus virginiana	Chokecherry	Dec. Shrub	Neu[e]	1
Pyrola secunda	Wintergreen	Per. Forb	Neu[e]	6
Ribes spp. mainly *R. glandulosum, R. hudsonianum, R. laxiflorum*	Currant	Dec. Shrub	Neu	15
Ribes triste	Swamp red currant	Dec. Shrub	Neu	11
Rubus chamaemorus	Cloudberry	Per. Forb	Neu	9
Rubus pubescens	Dwarf raspberry	Dec. Shrub	Neu	18
Salix pedicellaris	Bog willow	Dec. Shrub	Neu[e]	2
Salix planifolia	Diamond-leaf willow	Dec. Shrub	Neu[e]	6
Salix pseudomonticola	False mountain willow	Dec. Shrub	Neu	9
Salix pyrifolia	Balsam willow	Dec. Shrub	Neu[e]	0
Salix serissima	Autumn willow	Dec. Shrub	Neu[e]	3
Salix spp.	Willow	Dec. Shrub	Neu[e]	5
Smilacina trifolia	Three-leaved Solomon's seal	Per. Forb	Neu	14
Solidago hispida	Hairy goldenrod	Per. Forb	Neu	14
Solidago spp.	Goldenrod	Per. Forb	Neu[e]	4

Sorbus americana	American mountain ash	Dec. Shrub	Neu[e]	6
Streptopus roseus	Rose twisted-stalk	Per. Forb	Neu	9
Taraxacum spp.	Dandelion	Per. Forb	Neu[e]	3
Typha latifolia	Broadleaf cattail	Graminoid	Neu[e]	2
Viburnum edule	Squashberry	Dec. Shrub	Neu	8
Viburnum spp.	Squashberry	Dec. Shrub	Neu[e]	1
Vicia americana	American vetch	Per. Forb	Neu[e]	2
Viola spp.	Violets	Per. Forb	Neu	28
Vaccinium angustifolium	Lowbush blueberry	Dec. Shrub	Neu	63
Vaccinium myrtilloides	Velvet-leaved blueberry	Dec. Shrub	Neu	75
Abies balsamea	Balsam fir	Conifer	Avd	17
Agrostis spp.	Bentgrass	Grass	Avd[e]	7
Anaphalis margaritacea	Pearly everlasting	Per. Forb	Avd	9
Andromeda polifolia	Bog rosemary	Ev. Shrub	Avd[e]	4
Anemone quinquefolia	Wood anemone	Per. Forb	Avd[e]	7
Aralia nudicaulis	Wild sarsaparilla	Per. Forb	Avd	29
Arctostaphylos uva-ursi	Bearberry	Ev. Shrub	Avd	19
Bryoria spp.	Horsehair lichen	Arb. Lichen	Avd	68
Caltha palustris	Marsh marigold	Per. Forb	Avd[f]	0
Carex rostrata/aquatilis	Beaked or Water Sedge	Graminoid	Avd[e]	6
Carex spp.	Sedge	Graminoid	Avd	48
Chamaedaphne calyculata	Leatherleaf	Ev. Shrub	Avd	9
Chimaphila umbellata	Prince's pine	Ev. Shrub	Avd	12
Cinna latifolia	Drooping woodreed	Grass	Avd[e]	6
Circaea alpina	Alpine enchanter's-nightshade	Per. Forb	Avd[e]	3
Cirsium spp.	Thistle	Per. Forb	Avd[e]	3
Coptis trifolia	Threelead goldthread	Per. Forb	Avd	31
Cornus canadensis	Bunchberry	Per. Forb	Avd	72

Dryopteris spp.	Wood fern	Fern	Avd	18
Epigaea repens	Trailing arbutus	Ev. Shrub	Avd	19
Epilobium spp.	Willowherbs	Per. Forb	Avd[e]	2
Equisetum fluviatile	Water horsetail	Graminoid	Avd[e]	3
Fragaria virginiana	Wild strawberry	Per. Forb	Avd	19
Galium triflorum	Fragrant bedstraw	Per. Forb	Avd	11
Gaultheria hispidula	Creeping snowberry	Ev. Shrub	Avd	37
Gaultheria procumbens	American wintergreen	Ev. Shrub	Avd	9
Geocaulon lividum	Falso toadflax	Per. Forb	Avd[e]	5
Glyceria spp.	Mannagrass	Grass	Avd[f]	0
Goodyeara repens	Dwarf rattlesnake-plantain	Per. Forb	Avd	27
Gymnocarpium dryopteris	Northern oak fern	Fern	Avd[e]	3
Juncus spp.	Bulrush	Graminoid	Avd[e]	6
Juniperus communis	Common juniper	Ev. Shrub	Avd[f]	0
Kalmia angustifolia	Sheep laurel	Ev. Shrub	Avd[f]	0
Kalmia polifolia	Bog laurel	Ev. Shrub	Avd	11
Lactuca spp.	Wild lettuce	Per. Forb	Avd[e]	6
Larix laricina	Tamarack	Conifer	Avd[e]	3
Ledum groenlandicum	Bog Labrador tea	Ev. Shrub	Avd	47
Linnaea borealis	Twinflower	Ev. Shrub	Avd	55
Lycopodium annotinum	Stiff clubmoss	Moss	Avd	20
Lycopodium clavatum	Stag's-horn clubmoss	Moss	Avd	21
Lycopodium complanatum	Northern ground-cedar	Moss	Avd	11
Lycopodium dendroideum	Tree clubmoss	Moss	Avd	16
Lycopodium digitatum	Groundcedar	Moss	Avd[f]	0
Lycopus uniflorus	Nothern bungleweed	Per. Forb	Avd[e]	1
Maianthemum canadense	Canada mayflower	Per. Forb	Avd	67
Melampyrum lineare	Narrowleaf cow wheat	Per. Forb	Avd	35

Mitella nuda	Naked miterwort	Per. Forb	Avd	12
Myrica gale	Sweetgale	Dec. Shrub	Avd[f]	0
Oryzopsis asperifolia	Rough-leafed rice grass	Grass	Avd	21
Oryzopsis pungens	Mountain rice grass	Grass	Avd	23
Panicum spp.	Panicum	Grass	Avd[e]	2
Picea glauca	White spruce	Conifer	Avd[e]	3
Picea mariana	Black spruce	Conifer	Avd	55
Pinus banksiana	Jack pine	Conifer	Avd	19
Pinus resinosa	Red pine	Conifer	Avd[e]	2
Platanthera spp.	Bog orchid	Per. Forb	Avd[f]	0
Poa spp.	Meadowgrass	Grass	Avd[e]	2
Populus balsamifera	Cottonwood	Dec. Shrub	Avd	17
Potentilla norvegica	Rough cinquefoil	Per. Forb	Avd[e]	1
Potentilla palustris	Marsh cinquefoil	Per. Forb	Avd[f]	0
Pteridium aquilinum	Bracken fern	Fern	Avd[e]	6
Pyrola spp.	Wintergreen	Per. Forb	Avd	13
Ranunculus spp.	Buttercup	Per. Forb	Avd[e]	1
Ribes lacustre	Black swamp gooseberry	Dec. Shrub	Avd[f]	0
Rubus idaeus	Red raspberry	Dec. Shrub	Avd	29
Sambucus spp.	Eldderberry	Dec. Shrub	Avd[e]	3
Scirpus cyperinus	Woolgrass	Graminoid	Avd	11
Stereocaulon paschale	Foam lichen	Terr. Lichen	Avd	12
Thuja occidentalis	Eastern white-cedar	Conifer	Avd[f]	0
Trientalis borealis	Starflower	Per. Forb	Avd	30
Vaccinium oxycoccos	Bog cranberry	Ev. Shrub	Avd	11
Vaccinium vitis–idaea	Lingonberry	Ev. Shrub	Avd	22

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[a] Abor. Lichen = arboreal lichens; Dec. Shrub = deciduous shrubs; Ev. Shrub = evergreen shrubs; Fol. Lichen = foliose lichen; Per. Forb = perennial forbs; Terr. Lichen = terrestrial lichens.

[b] Diet selection category as determined from forage selection studies over 2 years using captive, trained caribou in boreal ecosystems of Northwestern Ontario. From this work, each plant species was assigned a selection category based on Ivlev's Electivity Index (Ivlev 1961), a metric that ranges from -1 to +1, where "selected" (Sel) = positive index with confidence intervals (C.I.) that excluded zero, "avoided" (Avd) = negative index with C.I. that excluded zero, or "neutral" (Neu) = C.I. that overlapped zero (see Denryter et al. 2017 for more details).

[c] Number of caribou pen sites in which the plant species occurred and were used to calculate Ivlev's Electivity Index (Ivlev 1961) (out of 79 pen sites sampled in Northwestern Ontario during 2017 and 2018). If sample size = 0, that plant species was not found in a caribou pen site but was found in other vegetation sites sampled in northeastern or Northwestern Ontario.

[d] Lichen or lichen-forming fungi species that due to their growth structure, we were unable to clip for biomass. Caribou readily ate these species when they encountered them and thus, we included them in the table as selected species.

[e] When we did not have sample sizes > 7 from caribou grazing trials to determine a confidence interval for selection categories, we inferred one from the calculated Ivlev score, our observations of tame caribou when exposed to these plants in Ontario and in British Columbia (Denryter et al. 2017), and from selection patterns among similar plants (e.g., same genus).

[¥] Species encountered in vegetation plots but not in caribou pens; thus, we based the selection category on genus or family rather than on caribou foraging trials.

Appendix B. Cross-walking diagrams linking ecosites of Northwestern Ontario from Racey et al. (1996) and ecosites of Northeastern Ontario (i.e., Cochrane) from Taylor et al. (2000) and cross-walking for mapping purposes.

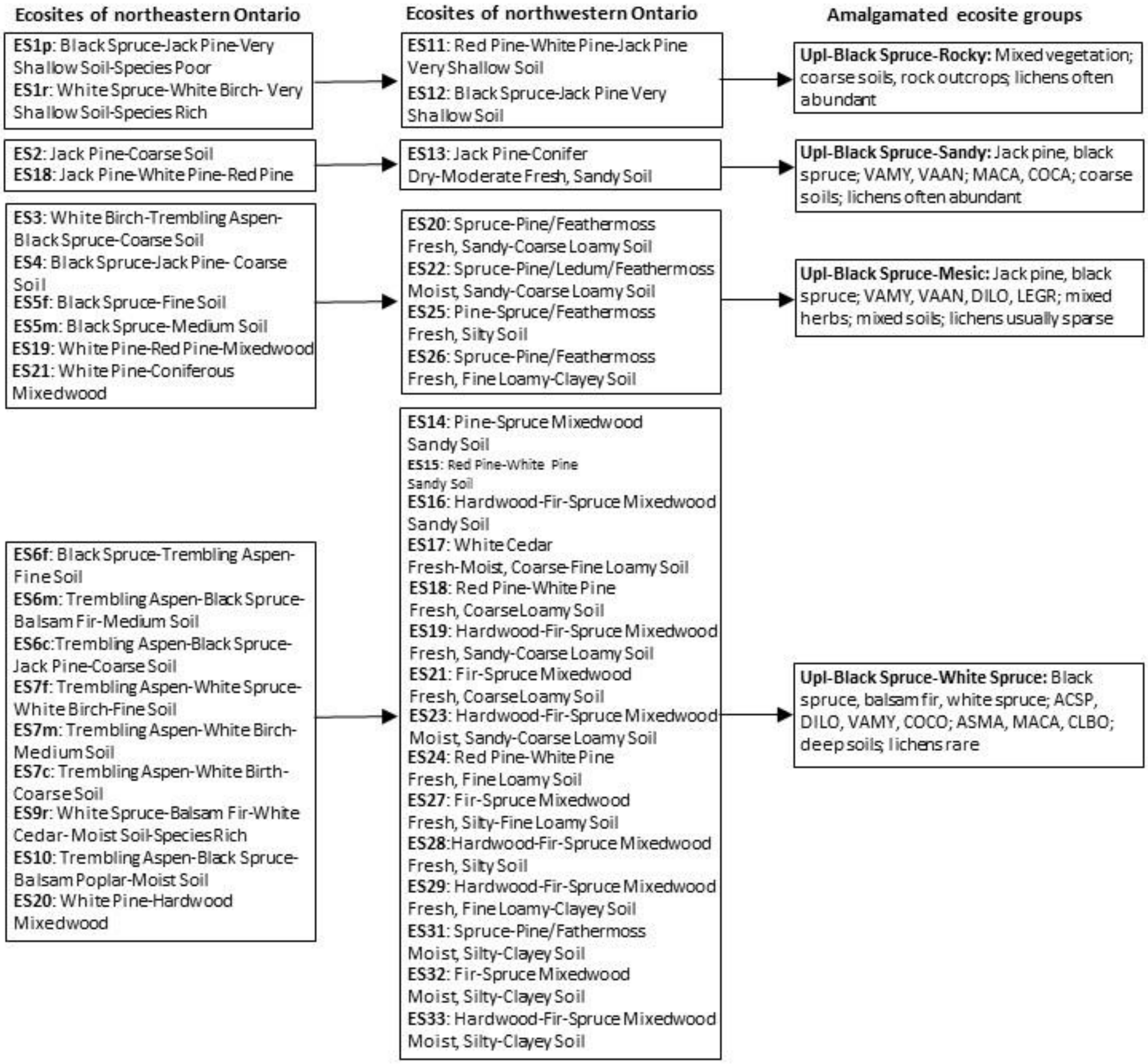

Figure B1. Cross-linkages between ecosites of northwestern and Northeastern Ontario with grouped ecosites on upland (Upl) sites used in this study. Undergrowth composition was generally similar within boxes and among boxes connected by arrows. Plant species codes are: VAMY = Vaccinium myrtilloides; VAAN = Vaccinium angustifolium; MACA = Maianthemum canadensis; COCA = Cornus canadensis; DILO = Diervilla lonicera; LEGR = Ledum groenlandicum; ACSP = Acer spicatum; COCO = Corylus cornuta; ASMA = Aster macrophyllus; CLBO = Clintonia borealis.

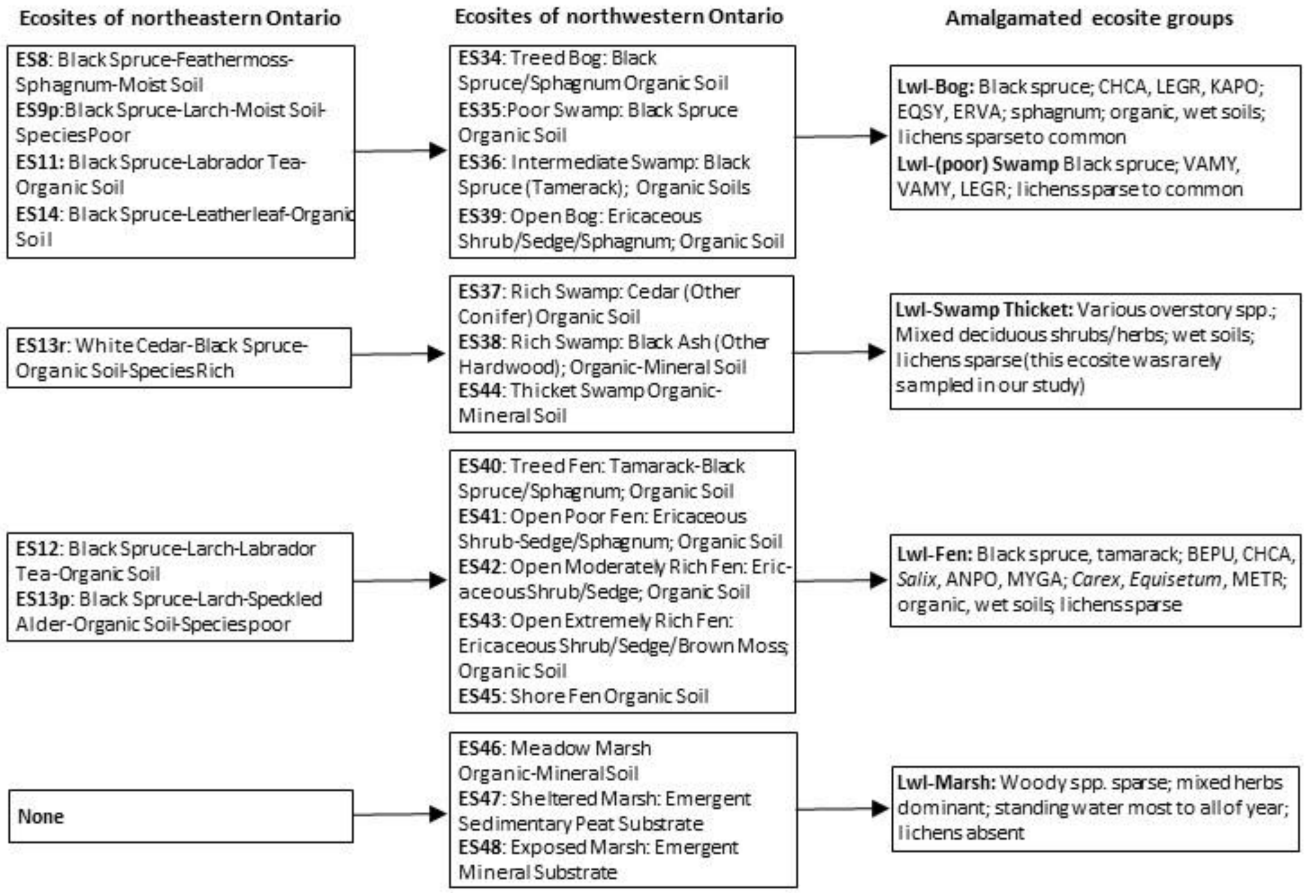

Figure B2. Cross-linkages between ecosites of northwestern and Northeastern Ontario with grouped ecosites on Lowland (wetland) (Lwl) sites used in this study. Undergrowth composition was generally similar within boxes and among boxes connected by arrows. Plant species codes are: CHCA = Chamaedaphne calyculata; LEGR = Ledum groenlandicum; KAPO = Kalmia polifolia; EQSY = Equisetum sylvaticum; ERVA = Eriophorum vaginatum; VAMY = Vaccinium myrtilloides; VAAN = Vaccinium angustifolium; BEPU = Betula pumila; ANPO = Andromeda polifolia; MYGA = Myrica gale; METR = Menyanthes trifoliata;

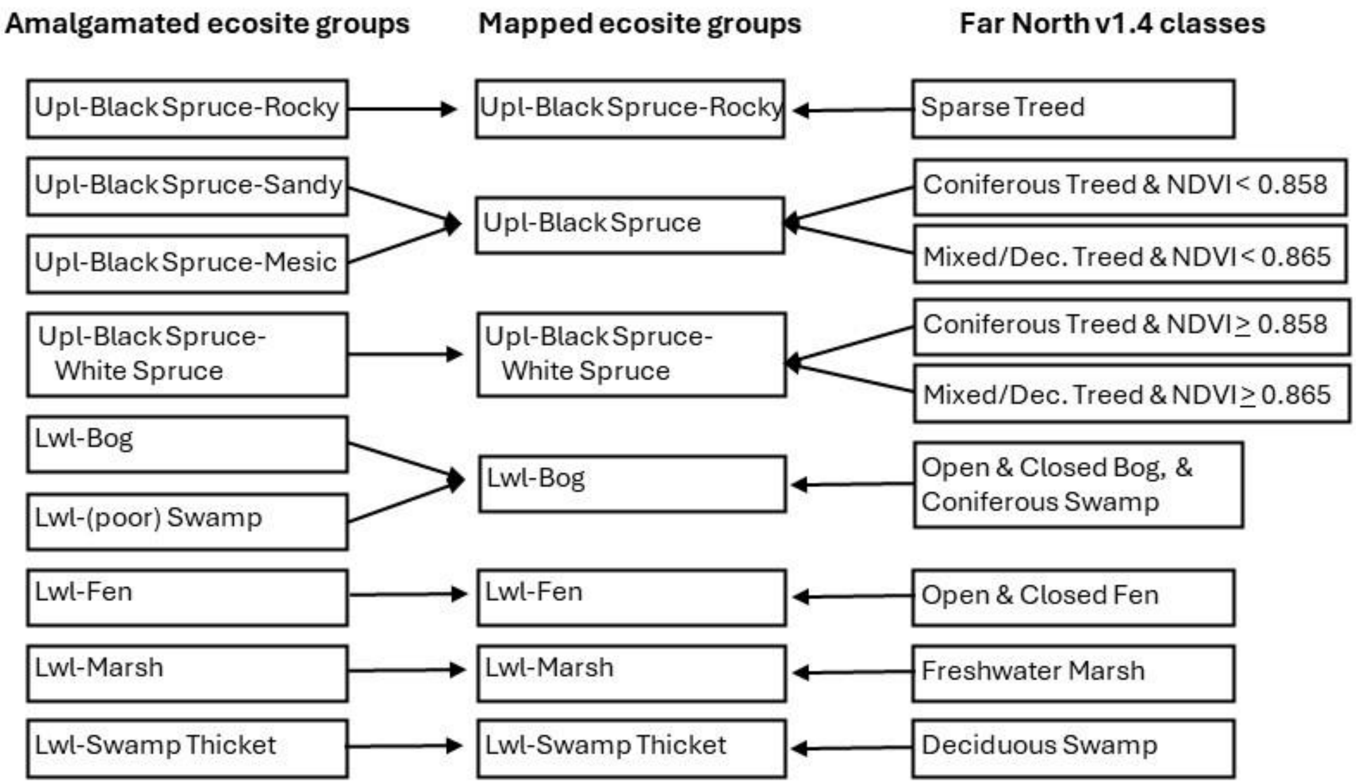

Figure B3. Cross-linkages between our amalgamated ecosite groups showing which were combined for mapping purposes (mapped ecosite groups) and how these crosswalked to the Far North vegetation layer for stands >20 years old.

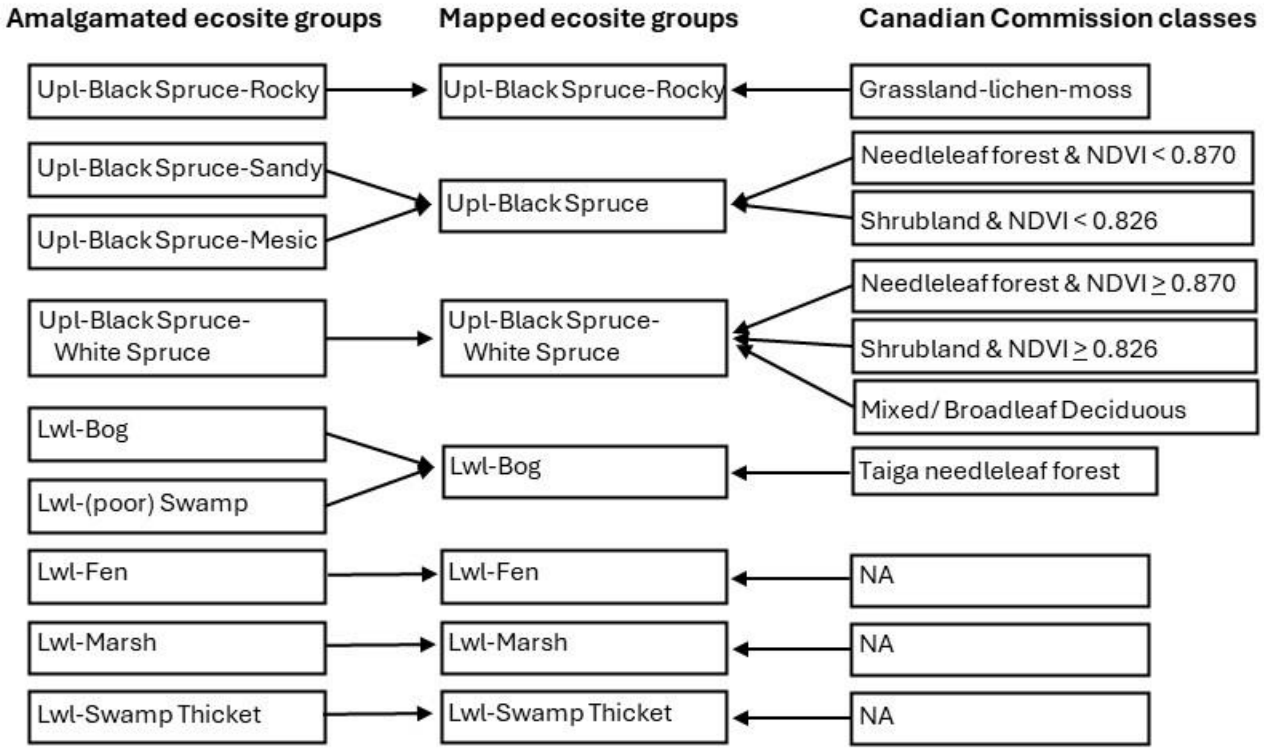

Figure B4. Cross-linkages between our amalgamated ecosite groups showing which were combined for mapping purposes (mapped ecosite groups) and how these crosswalked to the Far North vegetation layer for stands >20 years old.

Appendix C. Forage prediction equations.

Table C1. Top predictive equations of biomass of grasses, forbs, and deciduous shrub (combined) for each seral-specific (early: <20 yrs; mid–late: $\geq$ 20 yrs) ecosite and region (west AOI, east AOI, or both AOI combined) as a function of sampling date (i.e., Julian day; JD), basal area (m^2/ha; BA), and change in enhanced vegetative index (ΔEVI) and number of macroplots used to develop each model. BS = Black Spruce, WS = White Spruce.

Region	Ecosite	Seral	*n*	Equation	r^2
West AOI	Upl-BS-Rocky	Early	3	314.61	---
		Mid-late	20	0.87*(340.89*exp(-0.08*BA) + 17.19	0.36
	Upl-BS	Early	50	1.06*(413.84/(1.00+(BA/11.08)^1.31)) + 54.64	0.11
		Mid-late	136	1.04*(1.29e-51 *0.88^JD*JD^27.74) + 0.63*(413.84/(1+(BA/11.08)^1.31)) - 80.45	0.19
	Upl- BS-WS	Early	13	667.67	---
		Mid-late	34	0.98*(672.31/(1.00+(BA/8.99)^1.37)) + 56.14	0.38
	Lwl-Bog	Early	20	258.74	---
		Mid-late	52	0.62*(219.26/(1.00+(BA/4.44)^0.86)) + 33.64	0.16
	Marsh	NA	1	1053.39	---
East AOI	Upl-BS	Early	17	0.38*(1.52e-87*0.81^JD*JD^46.59) + 57.58	0.33
		Mid-late	22	49.08	---
	Upl-BS-WS	Early	10	1.53*(1.52e-87*0.81^JD*JD^46.59) -19.28	0.52
		Mid-late	28	0.88*(2.03e-96*0.79^JD*JD^51.48) + 9.30	0.22
	Lwl-Bog	Early	7	109.3	---
		Mid-late	34	1.12*(2.03e-121*0.74^JD*JD^64.68) - 9.32	0.41
Both	Lwl-Fen	Mid-late	16	1.00*(4.11e-88*0.79^JD*JD^47.82) + 0.81	0.51
	Lwl-Swamp thicket	Mid-late	5	538.74	---

Table C2. Top predictive equations of lichen biomass for each seral-specific (early: <20 yrs; mid–late: ≥ 20 yrs) ecosite and region (west AOI, east AOI, or both AOI combined) as a function of sampling date (i.e., Julian day; JD), basal area (m^2/ha; BA), canopy cover (%), percent sand, change in enhanced vegetative index (ΔEVI), and normalized difference moisture index (NDMI) and number of macroplots used to develop each model. BS = Black Spruce, WS = White Spruce.

Region	Ecosite	Seral	*n*	Equation	r^2
West AOI	Upl-BS-Rocky	Early	3	806.52	---
		Mid-late	20	-14.42*CC + 1196.62	0.30
	Upl- BS-WS	Early	14	12.03	---
		Mid-late	34	7.68	---
	Lwl-Bog	Early	20	16.53	---
		Mid-late	52	-365.38*NDMI + 166.33	0.11
	Lwl- Marsh	Mid-late	1	58.57	---
East AOI	Upl-BS-WS	Early	10	0.17	---
		Mid-late	28	-0.003*ΔEVI + 14.62	0.17
	Lwl-Bog	Early	7	176.21	---
		Mid-late	34	72.02	---
Both	Upl-BS	Early	67	328.97*NDMI - 4.94	0.06
		Mid-late	158	1.02*(11.24*0.79^BA*BA^2.70) + 44.42*Sand - 3101.93*NDMI - 1345.87	0.22
	Lwl-Fen	Mid-late	15	11.69	---
	Lwl- Swamp thicket	Mid-late	5	13.24	---

Table C3. Top predictive equations of horsetail biomass for each seral-specific (early: <20 yrs; mid-late: ≥ 20 yrs) ecosite and region (west AOI, east AOI, or both AOI combined) as a function of sampling date (i.e., Julian day; JD) and change in enhanced vegetative index (ΔEVI), and number of macroplots used to develop each model. BS = Black Spruce, WS = White Spruce.

Region	Ecosite	Seral	*n*	Equation	r^2
West AOI	Upl-BS-Rocky	Early	3	0.00	---
		Mid-late	20	0.00	---
	Upl-BS	Early	50	4.47	---
		Mid-late	136	0.86	---
	Upl- BS-WS	Early	13	34.11	---
		Mid-late	34	1.58	---
	Lwl-Bog	Early	20	42.46	---
		Mid-late	52	15.02	---
	Lwl-Marsh	Mid-late	1	0.00	---
East AOI	Upl-BS	Early	17	0.00	---
		Mid-late	22	0.10	---
	Upl-BS-WS	Early	10	0.01*ΔEVI - 46.79	0.51
		Mid-late	28	0.47	---
	Lwl-Bog	Early	7	23.16	---
		Mid-late	34	0.83*(3.30e-239*0.58^JD*JD^124.71) + 2.64	0.20
Both	Lwl-Fen	Mid-late	16	0.94	---
	Lwl-Swamp thicket	Mid-late	5	28.42	---

Table C4. Top predictive equations of mushroom biomass for each seral-specific (early: <20 yrs; mid-late: ≥ 20 yrs) ecosite and region (west AOI, east AOI, or both AOI combined) as a function of sampling date (i.e., Julian day; JD), precent clay, sand, or silt, change in enhanced vegetative index (ΔEVI), and normalized difference moisture index (NDMI), and number of macroplots used to develop each model. BS = Black Spruce, WS = White Spruce.

Region	Ecosite	Seral	*n*	Equation	r^2
West AOI	Upl-BS-Rocky	Early	3	0.007	---
		Mid-late	20	0.26	---
	Upl-BS	Early	50	1.09*(6.65e-22*JD^8.80) + 0.13*Silt - 4.02	0.25
		Mid-late	136	0.96*(1.47e-22*JD^9.09) + 0.06	0.13
	Upl- BS-WS	Early	13	0.49*(6.65-22*JD^8.80) - 0.04	0.36
		Mid-late	34	0.80*(6.65e-22*JD^8.80) - 0.0004*ΔEVI + 1.91	0.28
	Lwl-Bog	Early	20	2.21*(5.74e-15*JD^5.74) + 2.16*NDMI - 0.70	0.49
		Mid-late	52	0.96*(4.12e-15*JD^5.80) + 0.01	0.14
	Lwl-Marsh	Mid-late	1	0.00	---
East AOI	Upl-BS	Early	17	0.001	---
		Mid-late	22	-0.003*Clay + 0.00001*ΔEVI - 0.007	0.51
	Upl-BS-WS	Early	10	0.003	---
		Mid-late	28	0.04*Sand - 2.41	0.17
	Lwl-Bog	Early	7	0.66	---
		Mid-late	34	0.07	---
Both	Lwl-Fen	Mid-late	16	0.10	---
	Lwl-Swamp thicket	Mid-late	5	0.05	---

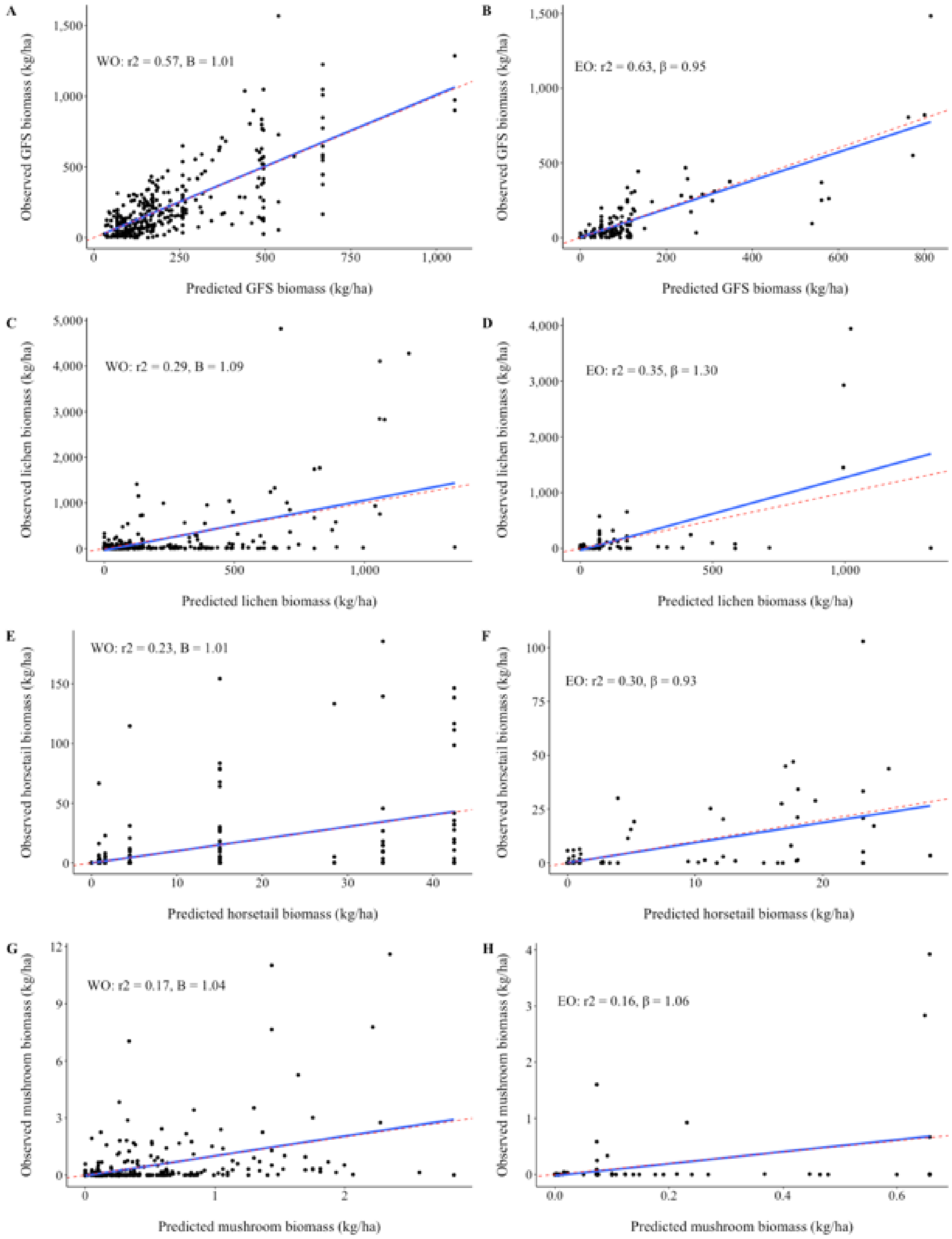

Figure C1. Relationship between predicted biomass vs. observed biomass of GFS (A,B), lichen (C, D), horsetail (E,F), and mushroom (G,H) in western (left panels) and eastern (right panels) Ontario with variation explained (r^2) and the regression coefficient estimate (B). Red-dashed line indicates the 1:1 line.

Appendix D. Forage, succession, and management implications by ecosite

Upland Black Spruce-White Spruce

Description.—Upland white spruce is a productive ecosite usually with a mix of conifer and hardwoods in early and mid-seral stands and with white spruce as a codominant with aspen, balsam poplar, Jack pine, or black spruce. Hardwood trees, primarily aspen and occasionally balsam popular are usually well-represented in this ecosite, particularly in relatively young stands. The presence of white spruce, either in the overstory layer or regeneration layer, is indicative of this ecosite. Undergrowth is typically very productive, especially in early seral communities. Bush honeysuckle, clubmoss, bunchberry, blue aster, and trailing raspberry usually dominate the undergrowth layer. Soon after disturbance, red raspberry, fireweed, mountain maple, and quaking aspen, and hazelnut often dominate (Table D1; Photos 1– 2). Upland white spruce comprised 12% of AOI west and 10% of AOI east (Fig. 6), excluding water bodies.

Vegetation changes after disturbance.--Establishment and development of the forest overstory after stand-replacing disturbance (regeneration logging or wildfire) was among the most rapid of our ecosites. Canopy cover peaked at >75% at 40–50 years on average after fire, with modest declines over the next 100 years to about 60% canopy cover (Fig. D1A). Basal area of the overstory layer also developed rapidly during the first 50 years after wildfire and appeared to stabilize thereafter. Stand age accounted for >80% of the variation in overstory canopy cover and at about 65% of the variation in basal area (Fig. D1A). Hardwoods, primarily aspen and balsam popular, were an important component of young stands, but gradually declined from 45% to about 25% of trees over the first 150 years after fire. Although stand age accounted for minor levels of the variation in Jack pine composition, it was typically a sparse component of the overstory (Fig. D1B). General patterns were similar between our two study areas, although greater overstory development and lower composition of Jack pine and hardwoods was evident at the northeastern study area.

Development of accepted grasses, forbs, and deciduous shrubs (combined) peaked at 500–800 kg/ha in young stands less than 20 years old, the highest level of this forage class of all our ecosites. Elevated biomass of this forage type persisted up to about 30–40 years after disturbance and then stabilized in stands 50 years and older (Fig. D2A). The period of sharpest decline in this forage coincided with the peak in overstory canopy cover, suggesting that re-establishment of overstory trees contributed to declines in forage. Biomass of accepted horsetails and lichens averaged < 20 kg/ha across all age classes (Fig. D2A, B). Our data suggests notably higher levels of accepted grasses, forbs, and shrubs (combined) at the northwestern study area versus that at the northeastern study area particularly in stands greater than about 30 years old but generally similar levels of horsetails and lichens in this ecosite.

Management implications.—This highly productive ecosite has potential to provide the highest levels of accepted forage mainly from accepted grasses, forbs, and deciduous shrubs (combined) compared to any of our other ecosites. Persistence of elevated biomass is less than 40 years, although biomass remained markedly higher in this ecosite at our northwestern study area than at the northeastern study area. Decline in biomass of accepted grasses, forbs, and deciduous was likely due to the relatively rapid rate of development of overstory trees after disturbance This pattern—highly elevated biomass of forage soon after stand-replacing disturbance that only persists for a relatively short period of time because of rapid re-establishment of overstory trees—is reasonably well-documented for productive upland forests sites in other ecoregions (Cook et al. 2016, Monzingo et al. 2023). The abundance of accepted horsetails and lichens is probably too low to be of much benefit to caribou in any season. The early-seral pulse in forage likely potentially has substantial value to caribou in summer. However, we note the following. (1) The decline in accepted forage as stands age is not compensated for by increased lichens in older stands as it may be in other ecosites. (2) The vegetation soon after fire often is very dense within even five years after burning, and the substantial density of this vegetation may impede access to caribou if they are reluctant to enter these kinds of stands. (3) The relatively greater tree biomass this ecosite supports before fire may result in prohibitive accumulation of dead woody material that also may impede caribou access when these trees start to fall after fire. (4) The pulse in forage from hardwood trees and mixed shrubs may be particularly beneficial to moose, perhaps increasing predation risk to caribou if enough area of this ecosite is burned. Use of herbicides to reduce the early seral development of forage may reduce the benefits to moose.

Table D1. Mean biomass (kg/ha; ±SE) of the most abundant plant species by stand age category in the upland black spruce-white spruce ecosite during summer 2017-2018 in northwestern and summer 2018 in Northeastern Ontario.

Scientific Name	Common Name	Forage[a] Class	Selection[b]	$\bar{x}$ biomass ≤20 years old	$\bar{x}$ biomass >20 years old
Northwestern Ontario				***n* = 15**	***n* = 33**
Rubus idaeus	Red raspberry	D. shrub	Avd	138.2 ± 43.7	4.6 ± 1.6
Epilobium angustifolium	Fireweed	P. forb	Neu	108.0 ± 37.4	1.1 ± 0.8
Acer spicatum	Mountain maple	D. shrub	Sel	94.7 ± 43.0	26.5 ± 7.6
Populus tremuloides	Quaking aspen	D. shrub	Sel	87.7 ± 70.3	0.4 ± 0.2
Corylus cornuta	Beaked hazelnut	D. shrub	Neu	64.7 ± 68.4	13.9 ± 5.9
Calamagrostis canadensis	Bluejoint	Grass	Neu	57.3 ± 20.8	5.5 ± 5.2
Cirsium spp.	Thistle	P. forb	Avd	55.7 ± 27.8	0 ± 0
Rubus pubescens	Trailing raspberry	D. shrub	Neu	47.3 ± 14.4	15.2 ± 6.7
Fragaria virginiana	Wild strawberry	P. forb	Avd	42.4 ± 15.6	5.2 ± 3.0
Aster ciliolatus	Fringed blue aster	P. forb	Neu	41.8 ± 22.3	1.2 ± 0.9
Aster macrophyllus	Big leaf aster	P. forb	Sel	40.8 ± 19.2	28.8 ± 10.0
Cornus canadensis	Bunchberry	P. forb	Avd	36.6 ± 12.7	37.8 ± 7.4
Poa spp.	Meadowgrass	Grass	Avd	35.5 ± 20.2	9.3 ± 7.0
Diervilla lonicera	Bush honeysuckle	D. shrub	Neu	12.2 ± 4.9	53.8 ± 8.8
Lycopodium annotinum	Stiff clubmoss	C. moss	Avd	7.0 ± 4.0	70.9 ± 15.0
Northeastern Ontario				***n* = 10**	***n* = 28**
Rubus idaeus	Red raspberry	D. shrub	Avd	126.1 ± 52.8	18.9 ± 8.9
Calamagrostis canadensis	Bluejoint	Grass	Neu	115.9 ± 67.5	5.6 ± 4.9
Alnus rugosa	Speckled alder	D. shrub	Sel	86.9 ± 47.6	10.1 ± 5.7
Populus tremuloides	Quaking aspen	D. shrub	Sel	42.0 ± 16.8	0.3 ± 0.2
Diervilla lonicera	Bush honeysuckle	D. shrub	Neu	34.3 ± 33.6	8.3 ± 4.2
Lycopodium annotinum	Stiff clubmoss	C. moss	Avd	33.2 ± 13.8	28.1± 7.7
Rubus pubescens	Trailing raspberry	D. shrub	Neu	28.2 ± 14.2	8.0 ± 1.7
Linneae borealis	Twinflower	E. shrub	Avd	28.2 ± 14.8	2.0 ± 1.0
Cornus stolonifera	Red osier dogwood	D. shrub	Neu	26.9 ± 30.1	0.3 ± 0.2
Solidago spp.	Goldenrod	P. forb	Neu	25.0 ± 23.0	0.1 ± 0.1
Cornus canadensis	Bunchberry	P. forb	Avd	20.8 ± 10.2	8.1 ± 2.2
Betula papyrifera	Paper birch	D. shrub	Sel	15.4 ± 9.3	0.6 ± 0.4
Epilobium angustifolium	Fireweed	P. forb	Neu	15.1 ± 5.8	1.8 ± 1.4
Aster macrophyllus	Big leaf aster	P. forb	Sel	14.7 ± 10.9	5.3 ± 3.1
Ribes lacustre	Black gooseberry	D. shrub	Avd	13.5 ± 7.2	3.7 ± 1.8

[a] Forage class: D. shrub = deciduous shrub; E. shrub = evergreen shrub; P. forb = perennial forb; T. lichen = terrestrial lichen; C. moss = club moss.

[b] Caribou forage selection: Selected (Sel) = use > available; Neutral (Neu) = use equals available; Avoided (Avd) = use < available.

Photo 1. Early seral (13-years-old) community in an upland black spruce-white spruce ecosite. Showing abundant willows in the background and a productive herbaceous layer in the foreground.

Figure 2. Upland black spruce-white spruce mid-seral (75 years old) with an undergrowth layer dominated by sarsaparilla.

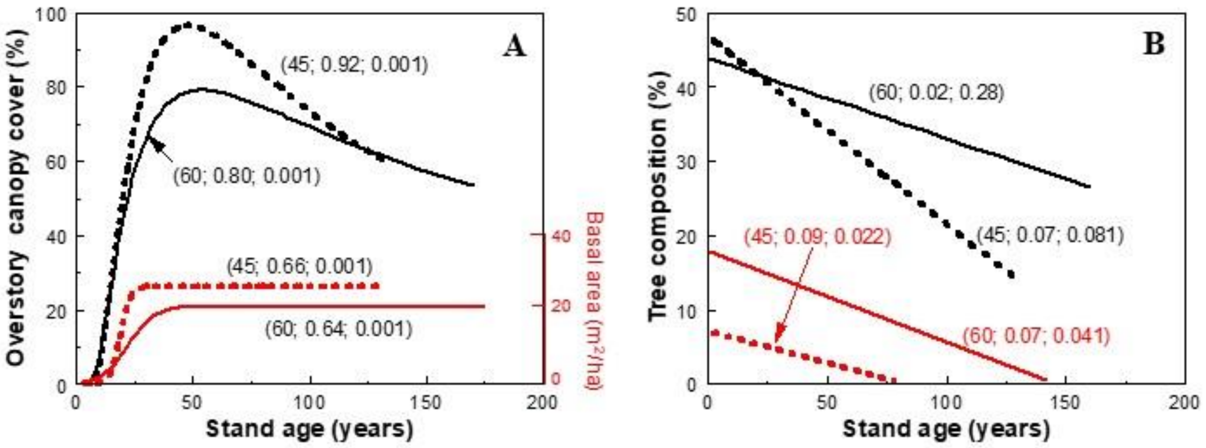

Figure D1. Overstory canopy cover and basal area in relation to stand age (graph A), and percent of trees that were hardwoods (black) and Jack pine (red) (graph B) in the upland black spruce-white spruce ecosite during summer 2017 and 2018 (solid lines = northwestern study area; dotted lines = northeastern study area). In parentheses are number of macroplots sampled, coefficient of determination (r^2), and significance level of the overall F-test.

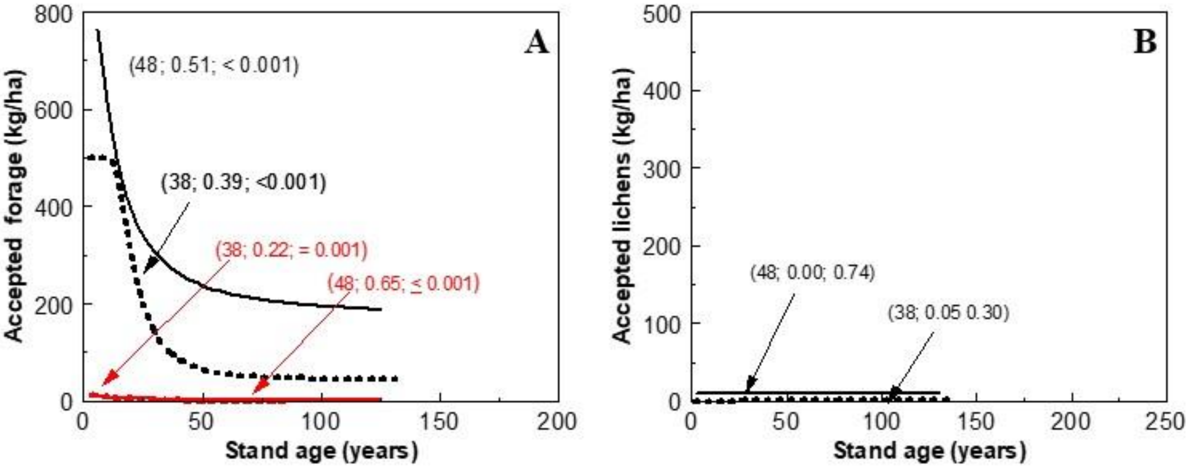

Figure D2. Development of accepted grasses, forbs, and deciduous shrubs (combined) (black lines) and accepted horsetails (red lines) (graph A) and accepted lichens (graph B) in relation to stand age in the upland black spruce-white spruce ecosite in northwestern (solid lines) and northeastern (dotted lines) Ontario study areas during summer of 2017 and 2018. In parentheses, we present number of macroplots sampled, coefficient of determination (r^2), and significance level of the overall F-test.

Upland Black Spruce

Description. —Upland Black Spruce (Upl-BS) is a relatively unproductive forest ecosite, especially drier phases on coarse textured soils, and is dominated by Jack pine in early and mid-seral stands and black spruce in old stands (white spruce is absent from overstory and regenerating layers). The upland black spruce ecosite comprised 49% (Upl-BS-Rocky accounted for 6%) of AOI west and 20% of AOI east (0% in Upl-BS-Rocky) (Fig. 6), excluding water bodies. For the analyses below, we combined the three upland black spruce ecosites (Upl-BS-Rocky, Upl-BS-Sandy, and Upl-BS-Mesic) (sample sizes were small for the Upl-BS-Rocky ecosite (not present at the northeastern study area), and in light of the approach of the Far North vegetation layer, we had to combine the Upl-BS-Sandy and Upl-BS-Mesic for mapping purposes). The Upl-BS-Rocky ecosite is typified by domination of rock outcrops with a variety of vascular plant species in the overstory and undergrowth layers. The Upl-BS-Sandy usually supported a sparse undergrowth, whereas the Upl-BS-Mesic ecosites was a relatively productive site. A key difference was the biomass of accepted lichens, with an average of 472 (n = 23), 446 (n = 64), and 66 (n = 122) kg/ha, respectively. Blueberry, Labrador tea, bush honeysuckle, and terrestrial lichens typically dominated this ecosite (Table D2; Photos 3-6).

Vegetation changes after disturbance.—In this relatively unproductive ecosite, we observed comparatively rapid initial establishment and development of the forest overstory after stand-replacing disturbance, similar to, but modestly slower and ultimately to a lower level than that in the upland black spruce-white spruce ecosite. Canopy cover peaked at about 65% at 50 years on average at the northwestern study area and 45% at the northeastern study area. (Fig. D3A). Thus, old forests in this ecosite were comparatively open compared to those in white spruce stands. Stand age accounted for 57–84% of the variation in both overstory canopy cover and basal area. Jack pine was the most common tree in stands up to 50–60 years, but comprised 30% or less of trees in older stands. Hardwoods averaged between 20–35% of trees in stands up to 20 years old but rapidly declined thereafter and averaged less than 10% of trees in old stands (Fig. D3B). Thus, black spruce typically dominated in older stands.

Biomass of accepted grasses, forbs, and deciduous shrubs (combined) were a third to a half of that versus in the upland-black spruce-white spruce ecosite. In addition, development of accepted forage diverged between the northwestern and northeastern study areas. Biomass of accepted grasses, forbs, and deciduous shrubs (combined) and accepted lichens were at least double that at the northwestern versus the northeastern study area across all successional stages (Fig. D4A). Elevated biomass of this forage type (200 to 500 kg/ha) persisted up to 20-30 years after disturbance and declined to 50–100 kg/ha in stands >50 years. Sharp declines in this forage class coincided with the peak in overstory canopy cover, but the lower levels of overstory canopy cover in older stands, compared to that in the white spruce ecosite, did not result in higher levels of accepted grasses, forbs, and deciduous shrubs (combined) in older stands (Fig. 14A). Accepted horsetails were an unimportant component in this ecosite. Development patterns of accepted lichens diverged between the two areas, with a sharp peak at about 400 kg/ha in relatively young stands (30–40 years old) in the northwestern study area, whereas no peak in lichen biomass was evident in the northeastern study area (Fig. 14B). This was apparent based on analysis of our 3 Upl-BS ecosites at northwestern study area: mean biomass of accepted lichens was 472 (n = 23), 446 (n = 64), and 122 (n = 122) kg/ha at the Upl-BS-Rocky, Upl-BS-Sandy, and Upl-BS-Mesic ecosites, respectively. Nevertheless, two types of stands were apparent in the upland black spruce ecosite in terms of lichen abundance, those with lichens too sparse to be of much benefit to caribou and those with abundant lichens. We attribute the difference to differences in soil characteristics and probably overstory characteristics, with accepted lichens most abundant in the drier sites with coarse-textured soils in the black spruce ecosites that we sampled (or on rock outcrops). Such differences driven by soil characteristics have been previously reported in western Canada (Cichowski et al. 2022). Although an anecdotal observation, we noticed at a number of sites that lichens were far more abundant near roads, and their abundance diminished with distance (30–50 meters) from the edge of the road, evidently due to higher penetration of sunlight. Thus, dense overstories apparently hindered development of lichens even on suitable soils (Cichowski et al. 2022).

Management implications.—This generally unproductive ecosite typically may have relatively marginal potential to support caribou nutrition that satisfies their nutritional requirements during summer in mid- and late seral stages, especially on the drier phases of this ecosite. The elevated levels of accepted grasses, forbs, and deciduous shrubs (combined) in very young stands provide good foraging opportunities, and the modest level of accepted biomass in these stands may be unattractive to moose. The propensity to provide relatively high levels of accepted lichen biomass may make this ecosite of particular value during winter, and managing these stands to promote lichens should be a consideration, given that drier extents of the ecosite provide some of the highest lichen biomass of any ecosite. Foresters may consider sparse spacing of planted trees after regeneration logging. One of the most productive sites we sampled at either study area was a Upl-BS-Sandy ecosite, 25 years old, that had been pre-commercially thinned. Lichen biomass (about 4,000 kg/ha of accepted lichens) on this site contrasted sharply with stands on very sandy soils that had been planted with a relatively high density of Jack pine, where lichens were only abundant adjacent to roads. Unfortunately, we were unable to separate among the 3 Upl-BS ecosites for mapping purposes, because separating the two types via remote sensing and detecting areas of unusually abundant terrestrial lichens was not possible.

We also suspect that lichen responses after logging versus wildfire may substantially differ. Wildfires largely eliminate lichens, whereas logging, especially on snow, typically only modestly reduces lichens and may even have no negative impact on lichen biomass (Webb 1996). Thus, logging followed by silvicultural practices that discourage rapid and dense regeneration of conifers may provide markedly greater biomass of lichens in the first several decades after stand replacing disturbance than will typically occur after wildfire. In addition, in well-stocked stands prior to wildfire, falling dead trees after treatment may reduce access of the site to caribou for 10-40 years.

Table D2. Mean biomass (kg/ha; ±SE) of the most abundant plant species by stand age category in the upland black spruce ecosite during summer 2017-2018 in western Ontario (first 15 rows) and during summer 2018 in the eastern Ontario (second 15 rows).

Scientific Name	Common Name	Forage Class	Selection	x̄ biomass ≤20 years old	x̄ biomass >20 years old
Northwestern Ontario				***n* = 58**	***n* = 151**
Vaccinium myrtilloides	Velvet-leaved blueberry	D. shrub	Neu	67.1 ± 13.1	45.5 ± 4.3
Ledum groenlandicum	Labrador tea	E. shrub	Avd	78.8 ± 16.9	69.8 ± 11.1
Cornus canadensis	Bunchberry	P. forb	Avd	76.5 ± 15.4	50.5 ± 4.7
Cladonia spp.	Cladonia lichen	T. lichen	Sel	49.5 ± 19.1	18.1 ± 6.7
Vaccinium angustifolium	Lowbush blueberry	D. shrub	Neu	46.7 ± 10.7	20.5 ± 2.8
Cladina rangiferina	Grey reindeer lichen	T. lichen	Sel	39.0 ± 12.6	128.2 ± 30.8
Diervilla lonicera	Bush honeysuckle	D. shrub	Neu	38.9 ± 15.8	13.8 ± 2.8
Cladina mitis	Green reindeer lichen	T. lichen	Sel	35.1 ± 13	54.6 ± 18.3
Rubus idaeus	Red raspberry	D. shrub	Avd	23.4 ± 7.3	0.7 ± 0.4
Alnus crispa	Mountain alder	D. shrub	Sel	22.8 ± 6.3	13.6 ± 3.4
Populus tremuloides	Quaking aspen	D. shrub	Sel	21.3 ± 9.1	0.5 ± 0.2
Epilobium angustifolium	Fireweed	P. forb	Neu	20.2 ± 4.5	1.0 ± 0.3
Lycopodium annotinum	Stiff clubmoss	C. moss	Avd	16.0 ± 11.4	19.8 ± 6.1
Gaultheria hispidula	Creeping snowberry	E. shrub	Avd	10.2 ± 2.8	30.7 ± 5.3
Cladina stellaris	Reindeer lichen	T. lichen	Sel	4.2 ± 2.5	58.4 ± 20.5
Northeastern Ontario				***n* = 17**	***n* = 22**
Vaccinium angustifolium	Lowbush blueberry	D. shrub	Neu	81.3 ± 25.3	21.0 ± 6.6
Kalmia angustifolia	Sheep laurel	E. shrub	Avd	78.5 ± 33.0	57.8 ± 17.4
Cladina rangiferina	Grey reindeer lichen	T. lichen	Sel	38.5 ± 17.3	146.5 ± 75.2
Vaccinium myrtilloides	Velvet-leaved blueberry	D. shrub	Neu	36.9± 12.0	14.8 ± 4.4
Ledum groenlandicum	Labrador tea	E. shrub	Avd	31.1 ± 30.4	25.0 ± 13.6
Cornus canadensis	Bunchberry	P. forb	Avd	15.9 ± 12.2	4.9 ± 2.2
Rubus idaeus	Red raspberry	D. shrub	Avd	14.8 ± 15.5	0 ± 0
Betula papyrifera	Paper birch	D. shrub	Sel	9.3 ± 5.8	0.5 ± 0.3
Cladonia spp.	Cladonia lichen	T. lichen	Sel	8.0 ± 3.8	0.7 ± 0.4
Cladina stellaris	Reindeer lichen	T. lichen	Sel	6.9 ± 3.2	254 ± 156
Gaultheria hispidula	Creeping snowberry	E. shrub	Avd	6.2 ± 4.6	25.4 ± 9.4
Prunus pensylvanica	Fire cherry	D. shrub	Sel	4.2 ± 3.9	0 ± 0
Cladina mitis	Green reindeer lichen	T. lichen	Sel	2.3 ± 1.0	6.1 ± 5.0
Epigaea repens	Trailing arbutus	E. shrub	Avd	2.0 ± 1.0	10.0 ± 5.3
Gaultheria procumbens	Boxberry	E. shrub	Avd	0.3 ± 0.3	10.0 ± 7.8

[a] Forage class: D. shrub = deciduous shrub; E. shrub = evergreen shrub; P. forb = perennial forb; T. lichen = terrestrial lichen; C. moss = club moss.

[b] Caribou forage selection: Selected (Sel) = use > available; Neutral (Neu) = use equals available; Avoided (Avd) = use < available.

Photo 3. Upland black spruce rocky ecosite showing abundant lichens growing on rock outcrops..

Photo 4. Moister phase of a young (4-years-old) upland black spruce ecosite showing a herb-dominated site with virtually no lichen.

Photo 5. Upland black spruce mid-seral (27 years old) ecosite showing abundant lichens in the foreground.

Photo 6. Upland black spruce late seral (105-years-old) showing sparse undergrowth layer.

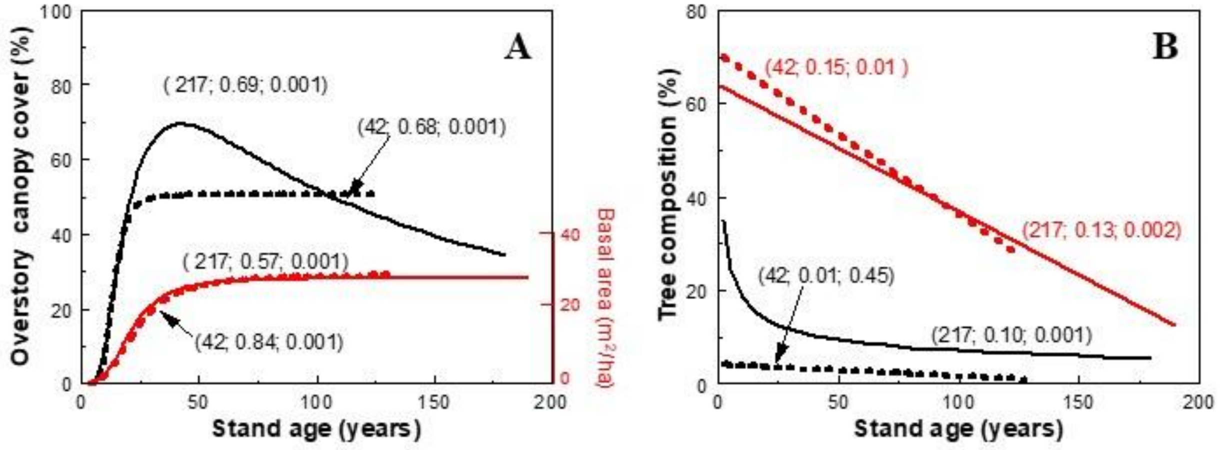

Figure. 3. Overstory canopy cover and basal area in relation to stand age (graph A), and percent of trees that were hardwoods (black) and Jack pine (red) (graph B) in the upland black spruce ecosite during summer 2017 and 2018 (solid lines = northwestern study area; dotted lines = northeastern study area). In parentheses are number of macroplots sampled, coefficient of determination (r^2), and significance level of the overall F-test. The upland black spruce-rocky, upland black spruce-sandy, and upland black spruce-mesic ecosites were combined for these analyses into an overall upland black spruce ecosite.

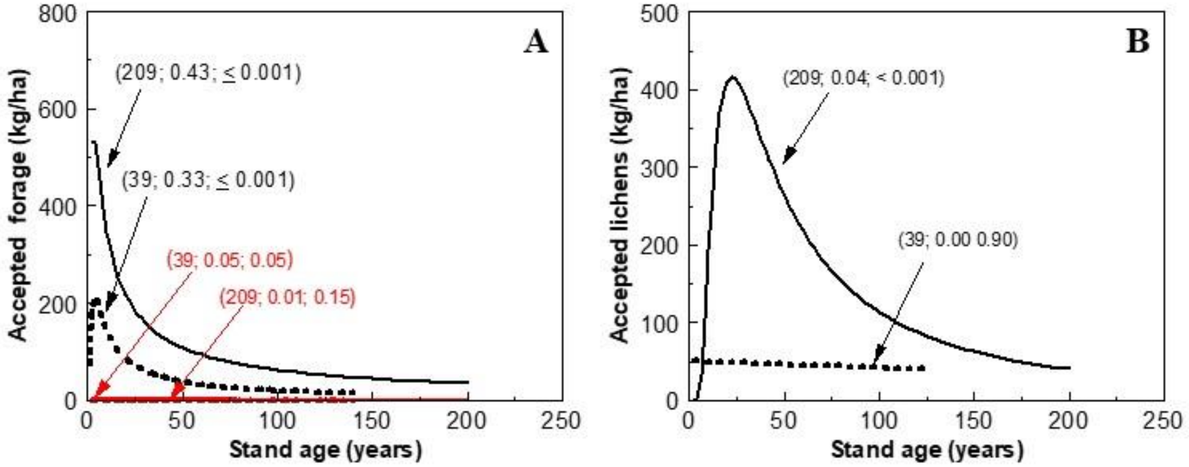

Figure 4. Development of accepted grasses, forbs, and deciduous shrubs (combined) (black lines) and accepted horsetails (red lines) (graph A) and accepted lichens (graph B) in relation to stand age in the upland black spruce ecosite in northwestern (solid lines) and northeastern (dotted lines) Ontario study areas during summer of 2017 and 2018. In parentheses, we present number of macroplots sampled, coefficient of determination (r^2), and significance level of the overall F-test. The upland black spruce-rocky, upland black spruce-sandy, and upland black spruce-mesic ecosites were combined for these analyses into an overall upland black spruce ecosite.

Lowland Fen

Description. —Fens are wetlands with or without trees. The overstory, if present, was usually <10 m tall and was dominated by black spruce and tamarack (larch). The undergrowth layer was dominated by deciduous shrubs, particularly bog birch and shrubby cinquefoil, with willow and paper birch often common. Graminoids, mainly sedges, varied from sparse to abundant, but shrubs dominated this ecosite (Table D3; Photos 7 and 8). Accepted lichens were rarely abundant. The youngest stand we sampled was 36 years old, and we have no data to evaluate forage production in early seral stages, but based on our data sets from the Northwest Territories, deciduous shrubs, fireweed, and sedges rapidly reestablish after wildfire, usually resulting in a diverse and productive community. The fen ecosite comprised 9% of AOI west and 18% of AOI east (Fig. 6), excluding water bodies.

Vegetation changes after disturbance.—We have no data for fens in stands less than 36 years old and our analysis of changes in overstory cover and basal area indicated no significant trends with stand age. Overstory cover averaged about 35% in our northwestern study area and about 5% in our northeastern study area, a pattern that was similar for basal area (Fig. D5). Jack pine and hardwood trees were virtually absent from this ecosite where we sampled. Thus, most fens sampled were open stands of black spruce and tamarack (larch).

Our data for mid- and late seral stands indicated no significant changes at either study area in relation to stand age (Fig. D6A). At both, biomass of accepted grasses, forbs, and deciduous shrubs averaged about 200 kg/ha, markedly higher than that in mid- and late seral stages in the upland black spruce ecotypes. We attribute this to low overstory canopy cover and basal area, such that dense overstories had minor impacts of forage development. Accepted horsetails were largely absent from fens. Accepted lichens also were sparse in fens; the increase in lichens depicted in Fig. D6B was due to a single elevated data point and small sample size.

Management implications.—A defining feature of many of the fen stands that we sampled in Ontario and other areas in which we have worked was the relatively high biomass of bog birch and willow, both of which are readily eaten by caribou, and both of which remain relatively abundant as stands age. The second defining feature is the sparse tree cover in most stands that likely accounts for the higher levels of accepted grasses, forbs, and deciduous shrubs in older stands compared to that in our other ecosites. Thus, stands of >50 years old tend to provide more of this forage class than the other ecosites. Our data therefore suggest that fens provide comparatively good foraging areas at times when caribou prefer to forage on vascular plants. The sparse lichen component, however, may limit foraging options for caribou during those times when caribou would benefit by feeding on lichens. In light of the sparse overstory tree cover and low basal area in older fens, impediments to access to caribou caused by accumulation of dead woody material and dense tree regeneration after wildfire may be comparatively modest in this ecosite.

Table D3. Mean biomass (kg/ha; ±SE) of the most abundant plant species by stand age category in the lowland fen ecosite during summer 2017-2018 in western Ontario (first 15 rows) and during summer 2018 in eastern Ontario (second 15 rows).

Scientific Name	Common Name	Forage[a] class	Selection[b]	x̄ biomass ≤20 years old	x̄ biomass >20 years old
Northwestern Ontario				***n* = 0**	***n* = 7**
Chamaedaphne calyculata	Leatherleaf	E. shrub	Avd	n/a	727 ± 245
Carex spp.	Sedge	Gram	Avd	n/a	150.3 ± 47.3
Ledum groenlandicum	Labrador tea	E. shrub	Avd	n/a	112.7 ± 50.5
Alnus rugosa	Speckled alder	D. shrub	Sel	n/a	89.2 ± 46.5
Cladina rangiferina	Grey reindeer lichen	T. lichen	Sel	n/a	64.9 ± 72.0
Vaccinium oxycoccos	Bog cranberry	E. shrub	Avd	n/a	58.6 ± 14.9
Andromeda polifolia	Bog rosemary	E. shrub	Avd	n/a	48.9 ± 41.5
Kalmia polifolia	Bog laurel	E. shrub	Avd	n/a	36.6 ± 17.9
Cladina mitis	Green reindeer lichen	T. lichen	Sel	n/a	33.1 ± 32.7
Smilacina trifolia	Three-leaved false lily of the valley	P. forb	Neu	n/a	32.7 ± 13.7
Betula pumila/glandulosa[c]	Bog birch	D. shrub	Neu	n/a	32.6 ± 11.6
Salix pseudomonticola	False mountain willow	D. shrub	Neu	n/a	31.8 ± 22.6
Carex rostrata/aquatilis	Beaked/water sedge	Gram	Avd	n/a	20.7 ± 21.2
Potentilla palustris	Marsh cinquefoil	P. forb	Avd	n/a	19.5 ± 13.9
Equisetum fluviatile	Water horsetail	Gram	Avd	n/a	15.9 ± 9.6
Northeastern Ontario				***n* = 0**	***n* = 9**
Ledum groenlandicum	Labrador tea	E. shrub	Avd	n/a	120.7 ± 28.8
Alnus rugosa	Speckled alder	D. shrub	Sel	n/a	94.3 ± 34.5
Gaultheria hispidula	Creeping snowberry	E. shrub	Avd	n/a	36.9 ± 15.3
Betula pumulis	Bog birch	D. shrub	Neu	n/a	31.3 ± 13.1
Chamaedaphne calyculata	Leatherleaf	E. shrub	Avd	n/a	30.7 ± 18.7
Rubus pubescens	Trailing raspberry	D. shrub	Neu	n/a	15.7 ± 8.7
Carex spp.	Sedge	Gram	Avd	n/a	15.3 ± 7.4
Lycopodium annotinum	Stiff clubmoss	C. moss	Avd	n/a	15.1 ± 11.4
Cornus canadensis	Bunchberry	P. forb	Avd	n/a	13.2 ± 9.9
Cladina rangiferina	Grey reindeer lichen	T. lichen	Sel	n/a	9.8 ± 8.6
Andromeda polifolia	Bog rosemary	E. shrub	Avd	n/a	9.6 ± 6.3
Rubus chamaemorus	Cloudberry	P. forb	Neu	n/a	9.2 ± 9.3
Salix pyrifolia	Balsam willow	D. shrub	Neu	n/a	7.8 ± 8.6
Vaccinium oxycoccos	Bog cranberry	E. shrub	Avd	n/a	6.8 ± 2.9
Linneae borealis	Twinflower	E. shrub	Avd	n/a	6.6 ± 5.8

[a] Forage class: D. shrub = deciduous shrub; E. shrub = evergreen shrub; P. forb = perennial forb; T. lichen = terrestrial lichen; C. moss = club moss; Gram = graminoids (excludes grasses).

[b] Caribou forage selection: Selected (Sel) = use > available; Neutral (Neu) = use equals available; Avoided (Avd) = use < available.

[c] We did not separate *Betula pumila* from *B. glandulosa*. The former provides larger leaves and is probably more palatable than *B. glandulosa*.

Photo 7. Lowland fen ecosite estimated to be 40 years old showing bog birch (red shrubs in background).

Photo 8. Mid-seral (about 68 years old) in the lowland fen ecosite showing abundant sedges interspersed with bog birch.

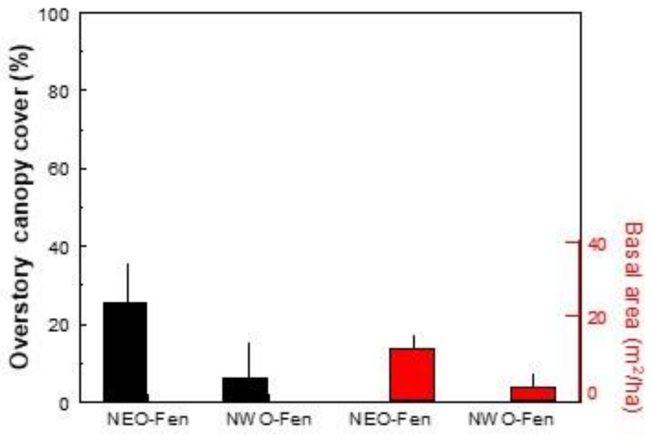

Figure D5. Overstory canopy cover and basal area in the lowland fen ecosite in northwestern (NWO) (black bars, n = 7) *and northeastern (NEO) (red bars, n* = 9) *Ontario study areas during summers of 2017 and 2018. Vertical bars indicate 1 standard error. Hardwood trees and Jack pine were virtually absent from our macroplot samples at both study areas.*

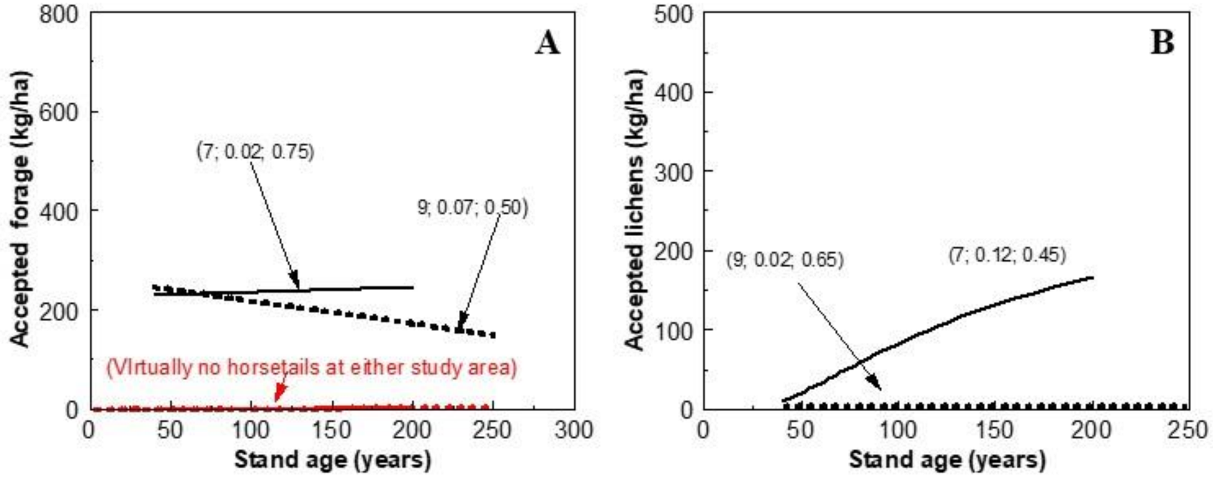

Figure D6. Development of accepted grasses, forbs, and deciduous shrubs (combined) (black lines) and accepted horsetails (red lines) (graph A) and accepted lichens (graph B) in relation to stand age in the lowland fen ecosite in northwestern (solid lines) and northeastern (dotted lines) Ontario study areas during summer of 2017 and 2018. In parentheses, we present number of macroplots sampled, coefficient of determination (r^2), and significance level of the overall F-test.

Lowland Bog

Description.—Bogs are peatlands with or without trees. The overstory, if present, was normally <10 m and was dominated by black spruce in all stand ages, although aspen or Jack pine often were present in early seral stages. The underbrush was dominated by ericaceous shrubs, particularly Labrador tea and leatherleaf (indicators of this type), and deciduous shrubs were sparse. In early seral stands, ericaceous shrubs and more palatable deciduous shrubs such as willows and bog birch often were present (Table D4, Photos 9 and 10). Swamps and bogs were very similar, and generally were separated by more blueberries and taller/denser trees in swamps than in bogs (Racey et al. 1996). For mapping, the bog and poor swamp ecosites were combined (the Far North Layer did not distinguish between the two). Together, these ecosites comprised 30% of AOI west and 51% of AOI east (Fig. 6), excluding water.

Vegetation changes after disturbance.—In the bog ecosite, trees were short and sparse—overstory canopy cover and basal area increased and then stabilized 50–75 years after stand-replacing disturbance to levels similar to that in fens and a third to half of that in upland stands (Fig. D7A). Stand age accounted for 20–45% of the variation in overstory canopy cover and basal area. Hardwood trees were a component of fens only in stands less than about 25 years old and were absent from our macroplots at the northeastern study area. Jack pine was absent from bogs at both study areas (Fig. D7B). After the first decade following disturbance, black spruce was typically the only abundant tree in the bog stands.

Biomass of accepted grasses, forbs, and deciduous shrubs (combined) exhibited a peak in young stands less than about 20 years old, although biomass levels averaged about a third at the northeastern study area compared to those at the northwestern study area across the entirety of succession. Accepted horsetails and lichens were sparse in bogs and were generally uncorrelated with stand age (Fig. D8). These findings suggest little value of this ecosite especially in summer, and the low levels of lichens suggest marginal value during winter.

Management implications.— We observed a modest boost in biomass of accepted grasses, forbs, and deciduous shrubs just after wildfire that persisted < 10 years and thereafter provided some of the lowest biomass of accepted forage. Bogs typically occurred on soils of unusually low soil nutrients (Racey et al. 1996) and many of the vascular plants that grow there over most of succession are evergreen shrubs (Labrador tea, leatherleaf), which in our experience are almost never eaten by caribou in summer. In addition, our observations were that both of these species resprout vigorously after fire and thus are often dominant even in very young stands. Thus, the nutritional value of this ecosite is typically poor. As with fens, the sparse overstory in older bogs may contribute to relatively low amounts of dead and down woody material that accumulates in the first decades after fire. Thus, impediments to access to caribou caused by accumulation of dead woody material and dense tree regeneration after wildfire may be comparatively modest in bogs.

Table D4. Mean biomass (kg/ha; ±SE) of the most abundant plant species by stand age category in the lowland bog ecosite during summer 2017-2018 in Northwestern Ontario (first 15 rows) and during summer 2018 in Northeastern Ontario (second 15 rows).

Scientific Name	Common Name	Forage Class	Selection	x̅ biomass ≤20 years old	x̅ biomass >20 years old
Northwestern Ontario				***n* = 6**	***n* = 19**
Chamaedaphne calyculata	Leatherleaf	E. shrub	Avd	354 ± 129	478 ± 104
Ledum groenlandicum	Labrador tea	E. shrub	Avd	345 ± 127	294 ± 61.2
Carex spp.	Sedge	Gram	Avd	102.3 ± 37.9	38.9 ± 10.6
Calamagrostis canadensis	Bluejoint	Grass	Neu	68.4 ± 51.9	29.6 ± 31.2
Lycopodium annotinum	Stiff clubmoss	C. Moss	Avd	68.2 ± 83.5	3.0 ± 1.9
Rubus idaeus	Red raspberry	D. shrub	Avd	65.4 ± 66.8	0 ± 0
Scirpus cyperinus	Woolgrass	Gram	Avd	63.7 ± 47.2	0.3 ± 0.3
Vaccinium myrtilloides	Velvet-leaved blueberry	D. shrub	Neu	59.7 ± 56.2	3.6 ± 1.9
Vaccinium vitis-idaea	Lingonberry	E. shrub	Avd	56.9 ± 38.1	13.1 ± 4.2
Vaccinium oxycoccos	Bog cranberry	E. shrub	Avd	36.0 ± 22.8	34.7 ± 6.9
Kalmia polifolia	Bog laurel	E. shrub	Avd	35.8 ± 22.8	42.0 ± 11.7
Cornus canadensis	Bunchberry	P. Forb	Avd	31.4 ± 26.3	0 ± 0
Equisetum sylvaticum	Wood horsetail	Gram	Neu	25.7 ± 18.9	3.6 ± 2.3
Cladonia spp.	Cladonia lichen	T. Lichen	Sel	23.7 ± 27.2	2.0 ± 1.2
Smilacina trifolia	False lily of the valley	P. Forb	Neu	19.5 ± 14.6	95.8 ± 28.0
Northeastern Ontario				***n* = 4**	***n* = 16**
Ledum groenlandicum	Labrador tea	E. shrub	Avd	315 ± 144	200.0 ± 31.5
Chamaedaphne calyculata	Leatherleaf	E. shrub	Avd	199.9 ± 56.6	89.0 ± 28.9
Cladina rangiferina	Grey reindeer lichen	T. Lichen	Sel	54.4 ± 73.2	71.9 ± 27.4
Carex spp.	Sedge	Gram	Avd	47.2 ± 23.4	12.9 ± 8.9
Gaultheria hispidula	Creeping snowberry	E. shrub	Avd	39.0 ± 44.7	16.9 ± 6.7
Kalmia angustifolia	Sheep laurel	E. shrub	Avd	38.7 ± 20.3	31.7 ± 9.0
Kalmia polifolia	Bog laurel	E. shrub	Avd	16.8 ± 9.9	25.5 ± 13.1
Calamagrostis canadensis	Bluejoint	Grass	Neu	14.4 ± 18.6	0 ± 0
Equisetum sylvaticum	Wood horsetail	Gram	Neu	13.9 ± 11.4	5.5 ± 3.5
Vaccinium oxycoccos	Bog cranberry	E. shrub	Avd	13.5 ± 11.7	7.3 ± 2.2
Eriophorum vaginatum	Tussock cottongrass	Gram	Neu	11.7 ± 9.5	4.2 ± 3.4
Smilacina trifolia	False lily of the valley	P. Forb	Neu	11.2 ± 7.8	8.0 ± 3.8
Rubus chamaemorus	Cloudberry	P. Forb	Neu	7.8 ± 8.6	7.0 ± 3.5
Vaccinium myrtilloides	Velvet-leaved blueberry	D. shrub	Neu	6.9 ± 6.3	6.4 ± 2.2
Vaccinium angustifolium	Lowbush blueberry	D. shrub	Neu	3.8 ± 2.0	10.6 ± 4.7

[a] Forage class: D. shrub = deciduous shrub; P. forb = perennial forb; E. shrub = evergreen shrub; T. Lichen = terrestrial lichen; Gram = graminoids (excludes grasses).

[b] Caribou forage selection: Selected (Sel) = use > available; Neutral (Neu) = use equals available; Avoided (Avd) = use < available.

Photo 9. Lowland bog early seral (about 8 years old) showing dominance by Labrador tea in the foreground with some deciduous shrubs present in the background.

Photo 10 Lowland bog, late seral (180 years old) community showing monotypic dominance by Labrador tea with sparse woodland horsetail in the undergrowth layer.

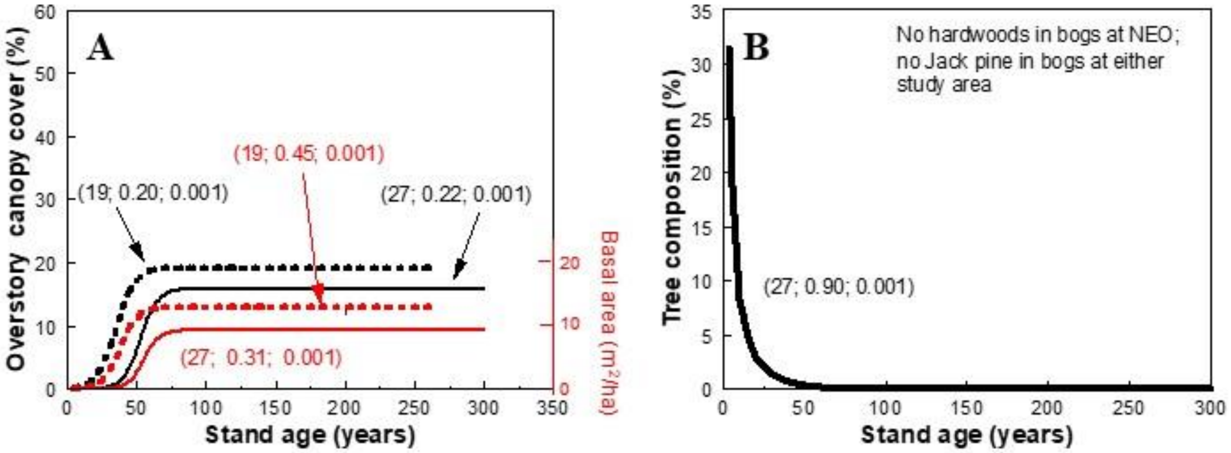

Figure D7. Overstory canopy cover and basal area (graph A), and percent of trees that were hardwoods and Jack pine (graph B) in relation to stand age in the lowland bog ecosite in the northwestern (solid lines) and northeastern (NEO) (dotted lines) study areas during summer 2017 and 2018. In parentheses, we present number of macroplots sampled, coefficient of determination (r^2), and significance level of the overall F-test.

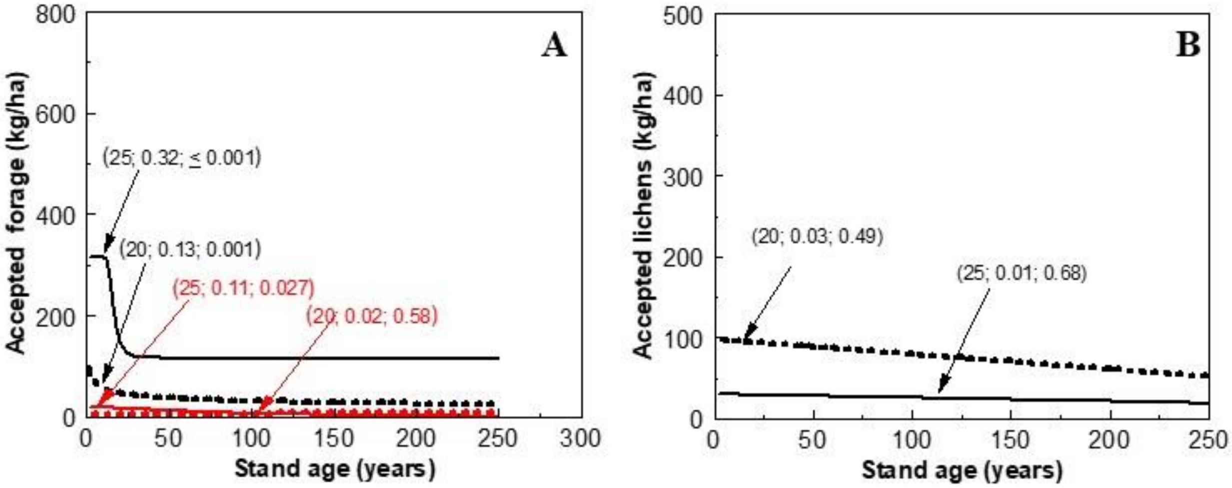

Figure D8. Development of accepted grasses, forbs, and deciduous shrubs (combined) (black lines) and accepted horsetails (red lines) (graph A) and accepted lichens (graph B) in relation to stand age in the lowland bog ecosite in northwestern (solid lines) and northeastern (dotted lines) Ontario study areas during summer of 2017 and 2018. In parentheses, we present number of macroplots sampled, coefficient of determination (r^2), and significance level of the overall F-test.

Lowland (poor) swamp

Description.—Poor swamps typically have undergrowth composition much like that of bogs, with Labrador tea a common dominant but with blueberry also common and leatherleaf uncommon to absent. The overstory was markedly more developed than in bogs, with greater canopy cover and taller trees in mid-to-late stages, and was greatly dominated by black spruce, In young stands less than about 20 years old, a variety of deciduous shrubs typically was common, including blueberry, horsetail, fireweed, and red raspberry, and at the northeastern study area, reindeer lichens were the second most common species after Labrador tea (Table D5, Photos 11 and 12). For mapping, the bog and poor swamp ecosites were combined. Together, these ecosites comprised 30% of AOI west and 51% of AOI east (Fig. 6), excluding water bodies.

Vegetation changes after disturbance.—Overstory canopy cover and basal area gradually increased and then stabilized 75–100 years after stand-replacing disturbance to about 60% cover and 30 m^2/ha basal area, although at our northwestern study area, both cover and basal area gradually declined in stands >100 years old (Fig. D9A). Stand age accounted for 70–87% of the variation in overstory canopy cover and basal area. Hardwood trees were a component of fens only in stands less than about 25 years old and were absent from our macroplots at the northeastern study area. Jack pine was a rare component of the overstory at both study areas (Fig. 9B). After the first decade following disturbance, black spruce was typically the only abundant tree in the swamp stands.

Biomass of accepted grasses, forbs, and deciduous shrubs (combined) exhibited a peak in young stands less than about 10-20 years old. The peak was lower (175 kg/ha at the northeastern study area versus 300 kg/ha at the northwestern study area) and the rate of decline with stand age was more gradual at the northeastern study area. Accepted horsetails showed an inverse relationship with stand age but were generally sparse. At the northeastern study area, our data indicate biomass of accepted lichens was elevated in early and mid-seral communities, although the small sample size suggests caution is warranted (Fig. D10).

Management implications.— Combining bogs and poor swamps for mapping reflects the overall similarity of the two ecosites, even though poor swamps were often floristically more diverse than bog. As with bogs, we observed a modest boost in biomass of accepted grasses, forbs, and deciduous shrubs just after wildfire that persisted 10–20 years and thereafter provided some of the lowest biomass of accepted forage of any ecosite. We suspect that the greater development of the overstory in swamps contributed to the decline, whereas the very low productivity of soils in bogs (Racey et al. 1996) may be responsible. Whatever the case, poor swamps in mid- and late seral stages offer poor foraging options for caribou. The modest boost in forage in early seral stages may be less beneficial to moose than caribou. In light of the greater development of forests in this ecosite, there may be more opportunities to manage by logging than in bogs to bolster accepted biomass. The elevated biomass of accepted lichens in early seral communities was surprising, but this result suggests another benefit from maintaining relatively low overstory density in this ecosite. The substantially greater development of the overstory in swamps poses questions about the effect of wildfire on access 10–30 years due to falling dead trees. Impediments to access to caribou caused by accumulation of dead woody material after wildfire may be elevated compared to bogs and fens.

Table D5. Mean biomass (kg/ha; ±SE) of the most abundant plant species by stand age category in the lowland poor swamp ecosite during summer 2017-2018 in western Ontario (first 15 rows) and during summer 2018 in the eastern Ontario (second 15 rows).

Scientific Name	Common Name	Forage Class	Selection	x̄ biomass ≤20 years old	x̄ biomass >20 years old
Northwestern Ontario				***n* = 14**	***n* = 34**
Ledum groenlandicum	Labrador tea	E. shrub	Avd	763 ± 178	393.1 ± 63.5
Gaultheria hispidula	Creeping snowberry	E. shrub	Avd	80.5 ± 25.2	67.5 ± 9.9
Scirpus cyperinus	Woolgrass	Gram	Avd	77.0 ± 52.5	1.3 ± 1.3
Vaccinium myrtilloides	Velvet-leaved blueberry	D. shrub	Neu	59.1 ± 13.1	15.8 ± 4.3
Vaccinium angustifolium	Lowbush blueberry	D. shrub	Neu	50.6 ± 33.1	8.6 ± 4.5
Equisetum sylvaticum	Wood horsetail	Gram	Neu	49.3 ± 15.6	19.6 ± 6.0
Carex spp.	Sedge	Gram	Avd	47.5 ± 10.4	12.8 ± 3.6
Epilobium angustifolium	Fireweed	P. forb	Neu	34.2 ± 10.1	0.1 ± 0.2
Vaccinium oxycoccos	Bog cranberry	E. shrub	Avd	27.6 ± 8.9	4.1 ± 1.1
Vaccinium vitis-idaea	Lingonberry	E. shrub	Avd	25.2 ± 10.0	17.9 ± 3.5
Rubus idaeus	Red raspberry	D. shrub	Avd	23.7 ± 13.3	0 ± 0
Carex rostrata/aquatilis	Beaked/water sedge	Gram	Avd	19.5 ± 15.6	0 ± 0
Cornus canadensis	Bunchberry	P. Forb	Avd	17.7 ± 8.4	7.8 ± 1.7
Alnus rugosa	Speckled alder	D. shrub	Sel	2.8 ± 3.0	12.2 ± 4.6
Cladina rangiferina	Grey reindeer lichen	T. Lichen	Sel	1.6 ± 1.0	16.7 ± 11.7
Northeastern Ontario				***n* = 5**	***n* = 16**
Ledum groenlandicum	Labrador tea	E. shrub	Avd	199.6 ± 57.3	168.4 ± 51.2
Cladina rangiferina	Grey reindeer lichen	T. Lichen	Sel	175 ± 122	149 ± 104
Alnus rugosa	Speckled alder	D. shrub	Sel	58.2 ± 75.2	22.4 ± 10.9
Kalmia angustifolia	Sheep laurel	E. shrub	Avd	51.1 ± 42.6	36.1 ± 9.5
Vaccinium angustifolium	Lowbush blueberry	D. shrub	Neu	50.3 ± 49.0	19.4 ± 7.1
Gaultheria hispidula	Creeping snowberry	E. shrub	Avd	35.6 ± 20.9	47.0 ± 10.5
Equisetum sylvaticum	Wood horsetail	Gram	Neu	30.9 ± 24.0	6.7 ± 3.3
Cladina stellaris	Reindeer lichen	T. Lichen	Sel	30.0 ± 37.9	58.4 ± 57.4
Carex spp.	Sedge	Gram	Avd	28.3 ± 20.8	3.2 ± 1.4
Cornus canadensis	Bunchberry	P. Forb	Avd	13.1 ± 9.7	4.2 ± 1.6
Vaccinium myrtilloides	Velvet-leaved blueberry	D. shrub	Neu	11.5 ± 7.6	18.5 ± 8.8
Calamagrostis canadensis	Bluejoint	Grass	Neu	11.3 ± 14.6	0.4 ± 0.4
Rubus pubescens	Trailing raspberry	D. shrub	Neu	10.9 ± 14.1	0.5 ± 0.2
Smilacina trifolia	False lily of the valley	P. Forb	Neu	7.4 ± 5.5	0.6 ± 0.3
Lycopodium annotinum	Stiff clubmoss	C. Moss	Avd	0 ± 0	8.0 ± 8.5

[a] Forage class: D. shrub = deciduous shrub; P. forb = perennial forb; E. shrub = evergreen shrub; T. Lichen = terrestrial lichen; Gram = graminoids (excludes grasses).

[b] Caribou forage selection: Selected (Sel) = use > available; Neutral (Neu) = use equals available; Avoided (Avd) = use < available.

Photo 11. Early seral (6 years old) lowland poor swamp ecosite with fireweed and deciduous shrubs interspersed among Labrador tea.

Photo 12. Mid-seral (80 years old) lowland swamp with a mix of Labrador tea and deciduous shrubs.

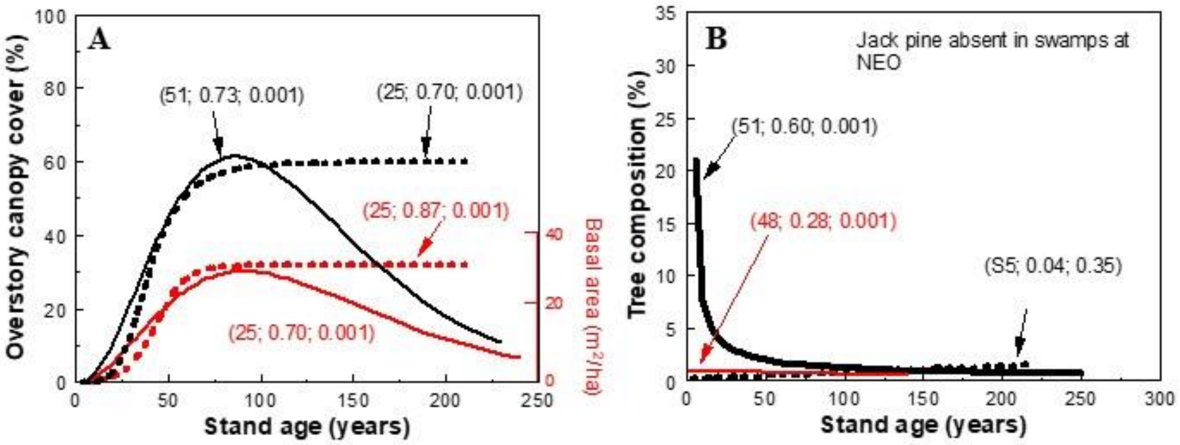

Figure D9. Overstory canopy cover and basal area (graph A), and percent of trees that were hardwoods and Jack pine (graph B) in relation to stand age in the poor swamp ecosite in the northwestern (solid lines) and northeastern (dotted lines) study areas during summer 2016 and 2019. In parentheses, we present number of macroplots sampled, coefficient of determination (r^2), and significance level of the overall F-test.

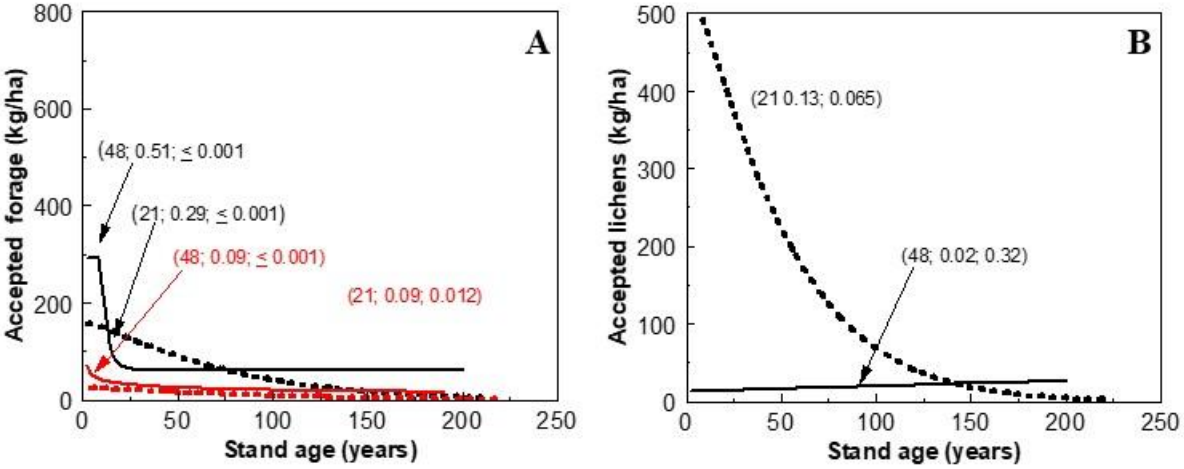

Figure D10. Development of accepted grasses, forbs, and deciduous shrubs (combined) (black lines) and accepted horsetails (red lines) (graph A) and accepted lichens (graph B) in relation to stand age in the lowland poor swamp ecosite in northwestern (solid lines) and northeastern (dotted lines) Ontario study areas during summer of 2017 and 2018. In parentheses, we present number of macroplots sampled, coefficient of determination (r^2), and significance level of the overall F-test.

Marsh

Description.--Marshes are wetlands lacking trees, vegetation is graminoid-dominated, usually by bluejoint and sedges, and shrubs are sparse or absent (as we defined the type). Sites were typically wet with standing water much of the year. Accepted lichens were virtually absent (Table D6; Photos 13 and 14). The marsh ecosite 0.1% of AOI west and 0.1% of AOI east (Fig. 6) excluding water bodies. This ecosite was a low priority for sampling and was not encountered in our northeastern study area, and we found no early seral marshes to sample at either study area.

Forage production.—Trees were absent from marshes and thus we did not estimate stand age. Biomass of accepted forage was high (Fig. D9), largely due to the considerable abundance of bluejoint. Otherwise, marshes primarily supported tall, coarse sedges and other similar graminoids that caribou typically avoided in summer. Accepted lichens averaged only 58 kg/ha based on a sample size of 3 sample macroplots.

Management implications.—By and large, the high biomass of accepted forage was due to the considerable abundance of bluejoint (Table D6). This species was eaten enough to rank as a neutral species (and by default, an accepted species), but our observations were that this species alone may not provide good forage that supports nutrition of lactating caribou—sites that provide greater abundance of a mix of accepted species are probably more beneficial to caribou. Our data set for this infrequent ecosite was too small and limited to provide reliable management insight.

Table D6. Mean biomass (kg/ha; ±SE) of the most abundant plant species by stand age category in the lowland marsh ecosite during summer 2017-2018 in Northwestern Ontario.

Scientific Name	Common Name	Forage[a] Class	Selection[b]	$\bar{x}$ biomass ≤20 years old	$\bar{x}$ biomass >20 years old
Northwestern Ontario				***n* = 0**	***n* = 3**
Calamagrostis canadensis	Bluejoint	Grass	Neu	n/a	888 ± 188
Carex spp.	Sedge	Gram	Avd	n/a	613 ± 568
Scirpus cyperinus	Woolgrass	Gram	Avd	n/a	120 ± 140
Typha latifolia	Broadleaf cattail	Gram	Neu	n/a	103.6 ± 94.1
Chamaedaphne calyculata	Leatherleaf	E. shrub	Avd	n/a	100 ± 173
Maianthemum canadense	Canada mayflower	P. forb	Avd	n/a	73.1 ± 126.7
Cladina rangiferina	Grey reindeer lichen	T. lichen	Sel	n/a	58.6 ± 101.4
Glyceria spp.	Mannagrass	Grass	Avd	n/a	57.8 ± 100.1
Caltha palustris	Marsh marigold	P. forb	Avd	n/a	39.2 ± 67.8
Alnus rugosa	Speckled alder	D. shrub	Sel	n/a	33.8 ± 58.5
Salix pseudomonticola	False mountain willow	D. shrub	Neu	n/a	22.7 ± 39.4
Dryopteris spp.	Wood fern	Fern	Avd	n/a	14.2 ± 24.6
Salix serissima	Autumn willow	D. shrub	Neu	n/a	9..3 ± 16.0
Aster spp.	Aster	P. forb	Neu	n/a	8.6 ± 7.6
Epilobium spp.	Willowherb	P. forb	Avd	n/a	3.9 ± 4.8

[a] Forage class: D. shrub = deciduous shrub; E. shrub = evergreen shrub; P. forb = perennial forb; E. shrub = evergreen shrub; T. Lichen = terrestrial lichen; Gram = graminoids (excludes grasses).

[b] Caribou forage selection: Selected (Sel) = use > available; Neutral (Neu) = use equals available; Avoided (Avd) = use < available.

Photo 13. Marsh ecosite showing mix of herbaceous species with sparse shrubs in the foreground.

Photo 14. Marsh ecosite showing nearly complete dominance by graminoids.

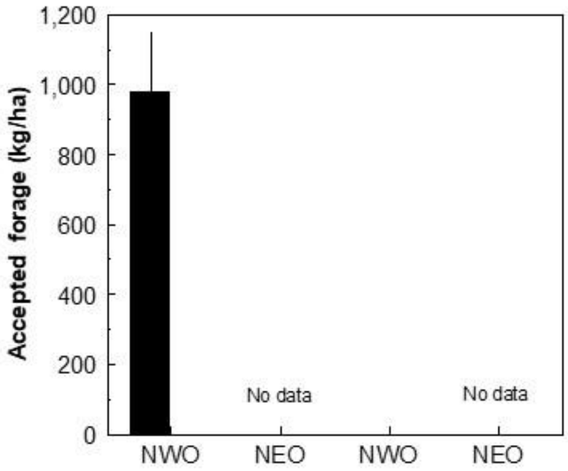

Figure D9. Biomass of accepted grasses, forbs, and deciduous shrubs (combined) (black bar) and accepted horsetails in the lowland marsh ecosite in northwestern (NWO) and northeastern (NEO) Ontario study areas during summer of 2017 and 2018. Marshes were not sampled in NEO, and biomass of horsetails was 0.0 at NWO. Biomass of accepted lichens was 58 kg/ha (n = 3) in marshes at NWO. Vertical bar indicates 1 standard error of the mean.

Lowland Swamp Thickets

Description.—Swamp thickets are wetlands that may or may not support trees, and were largely of two types, one that supported tall shrubs (>2 m tall) and lacked trees on mineral soils and the other that supported sparse stands of cedar commonly with peat and abundant shrubs. Bluejoint was the most common species in the undergrowth but a variety of shrubs were common. Sites were wet often with standing water much of the year. The swamp thicket comprised 1% of AOI west and 1% of AOI east (Fig. 6), excluding water bodies. As with marshes, this ecosite was a low priority for sampling and was virtually never encountered in our northeastern study area (n = 1).

Overstory and undergrowth.—The small sample size precluded an analysis of changes in overstory and undergrowth as a function of stand age (as with marshes, we sampled no early seral communities in swamp thickets). Overstory canopy cover and basal area averaged 19.2% and 5.0 m^2/ha at the northeastern study area and 37% and 8 m^2/ha at the northwestern study area. Biomass of accepted grasses, forbs, and deciduous shrubs (combined) was high at 650 kg/ha (Fig. D10), and, as in marshes, was mostly due to bluejoint (Table D7). To a greater extent than in marshes, however, a variety of other accepted species were present in this type. Biomass of accepted horsetails was virtually absent at the northeastern study area but averaged about 25 kg/ha at the northwestern study area. Accepted lichens averaged 64 kg/ha at the northeastern study area and only 0.6 kg/ha at the northwestern study area.

Management implications.—Although bluejoint was the dominant accepted species, the ecosite supported a diversity of accepted vascular plants. From a nutritional perspective, this ecosite may be valuable as a source of summer forage for caribou. Our concern, however, is that vegetation in many of the stands we sampled was very dense, perhaps precluding access by caribou of forage in this stand. Sparse timber in this ecosite probably precludes commercial logging. However, wildfire may greatly improve the accessibility of this ecosite and my increase the quality and quantity of forage and reduce the height of the forage, potentially providing a valuable type for foraging. Early seral stages also may be unusually valuable for moose, reducing its value to caribou. Unfortunately, we were unable to map early seral stands of this ecosite.

Table D7. Mean biomass (kg/ha; ±SE) of the most abundant plant species by stand age category in the lowland swamp thicket ecosite during summer 2017-2018 in Northwestern Ontario and 2018 in Northeastern Ontario.

Scientific Name	Common Name	Forage Class	Selection	$\bar{x}$ biomass ≤20 years old	$\bar{x}$ biomass >20 years old
Northwestern Ontario				***n* = 0**	***n* = 4**
Calamagrostis canadensis	Bluejoint	Grass	Neu	n/a	459 ± 409
Carex spp.	Sedge	Gram	Avd	n/a	83.9 ± 62.3
Alnus rugosa	Speckled alder	D. shrub	Sel	n/a	74.3 ± 45.1
Cornus stolonifera	Red osier dogwood	D. shrub	Neu	n/a	35.6 ± 46.8
Equisetum pratense	Meadow horsetail	Grass	Neu	n/a	32.9 ± 46.6
Betula papyrifera	Paper birch	D. shrub	Sel	n/a	27.0 ± 38.1
Lycopodium annotinum	Stiff clubmoss	C. Moss	Avd	n/a	26.3 ± 35.1
Carex rostrata/aquatilis	Beaked/water sedge	Gram	Avd	n/a	21.3 ± 30.1
Scirpus cyperinus	Woolgrass	Gram	Avd	n/a	12.9 ± 18.3
Ledum groenlandicum	Labrador tea	E. shrub	Avd	n/a	10.2 ± 11.6
Salix discolor	Pussy willow	D. shrub	Sel	n/a	9.9 ± 14.0
Rubus pubescens	Trailing raspberry	D. shrub	Neu	n/a	8.2 ± 5.6
Dryopteris spp.	Wood fern	Fern	Avd	n/a	8.1 ± 11.4
Aster spp.	Aster	P. Forb	Neu	n/a	6.5 ± 8.0
Lonicera spp.	Honeysuckle	D. shrub	Neu	n/a	5.5 ± 7.8
Northeastern Ontario				***n* = 0**	***n* = 1**
Cladina rangiferina	Grey reindeer lichen	T. Lichen	Sel	n/a	60.0
Alnus rugosa	Speckled alder	D. shrub	Sel	n/a	50.9
Ledum groenlandicum	Labrador tea	E. shrub	Avd	n/a	57.3
Vaccinium angustifolium	Lowbush blueberry	D. shrub	Neu	n/a	14.2
Smilacina trifolia	False lily of the valley	P. Forb	Neu	n/a	13.8
Gaultheria hispidula	Creeping snowberry	E. shrub	Avd	n/a	12.7
Rubus chamaemorus	Picking cloudberry	P. Forb	Neu	n/a	12.3
Salix pyrifolia	Balsam willow	D. shrub	Neu	n/a	10.4
Chamaedaphne calyculata	Leatherleaf	E. shrub	Avd	n/a	12.0
Linnaea borealis	Twinflower	E. shrub	Avd	n/a	5.7
Equisetum sylvaticum	Wood horsetail	Gram	Neu	n/a	3.4
Salix pedicellaris	Bog willow	D. shrub	Neu	n/a	2.6
Cladonia spp.	Cladonia lichen	T. Lichen	Sel	n/a	2.5
Vaccinium oxycoccos	Bog cranberry	E. shrub	Avd	n/a	2.3
Kalmia angustifolia	Sheep laurel	E. shrub	Avd	n/a	2.3

Photo 15. Swamp thicket of undetermined age dominated by tall deciduous shrubs and no conifer overstory.

Photo 16. Late seral stage (177 years old) in swamp thicket with a mixed overstory of cedar and tamarack.

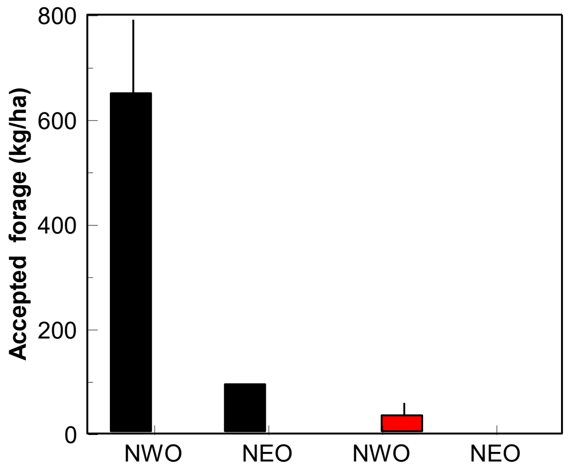

Figure D10. Biomass of accepted grasses, forbs, and deciduous shrubs (combined) (black bars) and accepted horsetails (red bars) (graph A) and accepted lichens (graph B) in the lowland bog ecosite in northwestern (NWO) and northeastern (NEO) Ontario study areas during summer of 2017 and 2018. Vertical lines indicate 1 standard error of the mean (where sample sizes permitted). Biomass of accepted lichens was 0.6 kg/ha at NWO (n = 3) and 64 kg/ha at NEO (n = 1) (not shown on graph).

Appendix E. Foodscape metrics by caribou range

Table E1. Proportion of grass, forbs, and deciduous shrubs biomass, horsetail biomass, and mushroom biomass combined based on 4-categories, lichen biomass based on 4-categories, intake rate (g/min based on 4-categories), and 5 categories of forage stability in response to disturbance within each Ontario caribou range on 1 August.

Metric	Value	Kinloch	Spirit	Ozhiski	Missisa	Swan	Churchill	Berens	Kesagami	Brightsand	Nipigon	Sydney	Pagwachuan
GFS, horsetail, and mushroom biomass (kg/ha)	>400 kg/ha	0.06	0.14	0.11	0.03	0.10	0.10	0.10	0.06	0.17	0.17	0.04	0.09
	>275 – 400 kg/ha	0.10	0.22	0.20	0.30	0.31	0.07	0.13	0.11	0.08	0.17	0.10	0.19
	>150 – 275 kg/ha	0.40	0.37	0.43	0.21	0.26	0.42	0.47	0.06	0.41	0.36	0.47	0.06
	≤150 kg/ha	0.44	0.27	0.26	0.47	0.33	0.40	0.30	0.78	0.35	0.31	0.40	0.66
Lichen biomass (kg/ha)	>800 kg/ha	0.01	0.03	0.03	0.02	0.03	0.03	0.07	0.01	0.04	0.02	0.13	0.01
	>400 – 800 kg/ha	0.09	0.06	0.08	0.01	0.02	0.09	0.18	0.03	0.13	0.07	0.26	0.03
	>150 – 400 kg/ha	0.14	0.08	0.11	0.02	0.02	0.13	0.14	0.05	0.12	0.08	0.18	0.05
	≤150 kg/ha	0.76	0.84	0.78	0.95	0.93	0.75	0.61	0.91	0.71	0.83	0.44	0.91
Intake rates (g/min)	>5.0 g/min	0.01	0.02	0.02	0.01	0.02	0.06	0.02	0.04	0.13	0.10	0.03	0.07
	>4.0 – 5.0 g/min	0.14	0.33	0.26	0.42	0.43	0.07	0.17	0.18	0.07	0.22	0.05	0.22
	>3.0 – 4.0 g/min	0.68	0.54	0.63	0.52	0.48	0.72	0.70	0.64	0.68	0.60	0.72	0.53
	≤3.0 g/min	0.17	0.10	0.08	0.05	0.06	0.15	0.12	0.14	0.12	0.09	0.20	0.18
Forage stability	Stable (1)	0.04	0.11	0.07	0.26	0.17	0.01	0.02	0.11	0.01	0.08	0.02	0.18
	Support Moose (2)	0.06	0.14	0.11	0.03	0.10	0.10	0.10	0.06	0.17	0.17	0.04	0.09
	Biomass increase (3)	0.79	0.61	0.73	0.69	0.61	0.69	0.70	0.70	0.63	0.54	0.76	0.67
	Biomass increase and support moose (4)	0.02	0.02	0.03	0.00	0.01	0.13	0.03	0.06	0.10	0.14	0.14	0.03
	Biomass decrease (5)	0.10	0.12	0.06	0.02	0.11	0.07	0.15	0.07	0.09	0.07	0.04	0.03

Table E2. Proportion of grass, forbs, and deciduous shrubs biomass, horsetail biomass, and mushroom biomass combined based on 4-categories, lichen biomass based on 4-categories, intake rate (g/min based on 4-categories), and 5 categories of forage stability in response to disturbance within each Ontario caribou range on 15 September.

Metric	Value	Kinloch	Spirit	Ozhiski	Missisa	Swan	Churchill	Berens	Kesagami	Brightsand	Nipigon	Sydney	Pagwachuan
GFS, horsetail, and mushroom biomass (kg/ha)	>400 kg/ha	0.06	0.14	0.11	0.03	0.10	0.10	0.10	0.06	0.17	0.17	0.04	0.09
	>275 – 400 kg/ha	0.05	0.10	0.11	0.03	0.14	0.06	0.11	0.00	0.06	0.07	0.07	0.01
	>150 – 275 kg/ha	0.13	0.20	0.25	0.18	0.19	0.19	0.22	0.06	0.21	0.25	0.30	0.05
	≤150 kg/ha	0.76	0.57	0.54	0.76	0.58	0.64	0.58	0.88	0.57	0.51	0.60	0.86
Lichen biomass (kg/ha)	>800 kg/ha	0.01	0.03	0.03	0.02	0.03	0.03	0.07	0.01	0.04	0.02	0.13	0.01
	>400 – 800 kg/ha	0.09	0.06	0.08	0.01	0.02	0.09	0.18	0.03	0.13	0.07	0.26	0.03
	>150 – 400 kg/ha	0.14	0.08	0.11	0.02	0.02	0.13	0.14	0.05	0.12	0.08	0.18	0.05
	≤150 kg/ha	0.76	0.84	0.78	0.95	0.93	0.75	0.61	0.91	0.71	0.83	0.44	0.91
Intake rates (g/min)	>5.0 g/min	0.01	0.04	0.03	0.01	0.04	0.06	0.03	0.03	0.13	0.10	0.04	0.06
	>4.0 – 5.0 g/min	0.11	0.23	0.20	0.18	0.27	0.07	0.16	0.03	0.07	0.15	0.03	0.04
	>3.0 – 4.0 g/min	0.83	0.70	0.74	0.50	0.68	0.79	0.75	0.21	0.76	0.72	0.74	0.32
	≤3.0 g/min	0.05	0.03	0.02	0.31	0.02	0.07	0.06	0.72	0.04	0.03	0.19	0.58
Forage stability	Stable (1)	0.04	0.11	0.07	0.26	0.17	0.01	0.02	0.11	0.01	0.08	0.02	0.18
	Support Moose (2)	0.06	0.14	0.11	0.03	0.10	0.10	0.10	0.06	0.17	0.17	0.04	0.09
	Biomass increase (3)	0.79	0.61	0.73	0.69	0.61	0.69	0.70	0.70	0.63	0.54	0.76	0.67
	Biomass increase and support moose (4)	0.02	0.02	0.03	0.00	0.01	0.13	0.03	0.06	0.10	0.14	0.14	0.03
	Biomass decrease (5)	0.10	0.11	0.06	0.02	0.11	0.07	0.15	0.07	0.09	0.07	0.04	0.03

Appendix F: Selection by caribou range

Table F1. Proportion of probability of selection for adult female caribou (with calf that survived >5-weeks postpartum) predicted for late summer (23 August) within each caribou range based on 4 categories of selection (high, medium, low, and very low) defined from quantiles calculated across the entire extent. Caution is needed when interpreting these values, because the resource selection model was derived using caribou locations from only three of the 12 caribou ranges and caribou selection could change as a function of availability (i.e., functional response).

Caribou Range	High	Medium	Low	Very low
Kinloch	0.21	0.39	0.26	0.14
Spirit	0.29	0.28	0.26	0.17
Ozhiski	0.25	0.30	0.25	0.19
Missisa	0.53	0.25	0.12	0.10
Swan	0.33	0.25	0.20	0.22
Churchill	0.06	0.26	0.30	0.38
Berens	0.15	0.31	0.32	0.23
Kesagami	0.23	0.29	0.31	0.17
Brightsand	0.07	0.25	0.30	0.37
Nipigon	0.15	0.19	0.26	0.41
Sydney	0.05	0.17	0.33	0.45
Pagwachuan	0.31	0.22	0.26	0.21

Appendix G. Summer predation risk

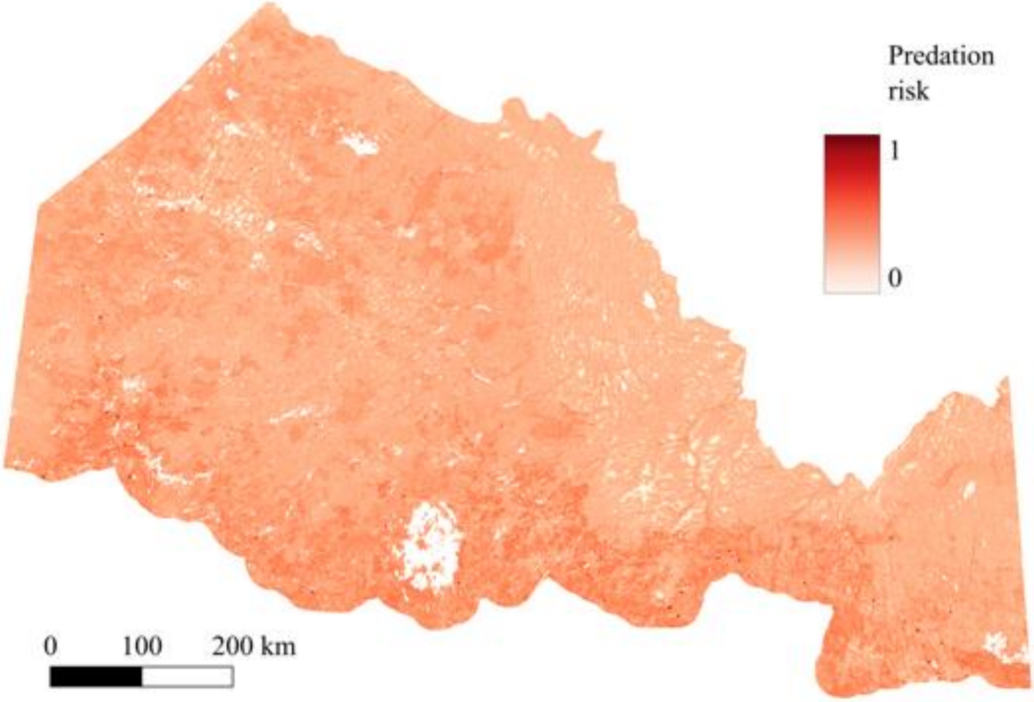

Figure H1. Summer predation risk predicted across the Ontario AOI.

Appendix H. Updating Foodscapes

For all applications, we note that our maps reflect recent conditions (i.e., 2018), and that these conditions will change reasonably rapidly in the dynamic forest systems of our mapping area. Thus, to be useful in the future, our maps will eventually need updating to reflect wildfire events, forest management and other human activities and developments, and ongoing forest succession.

In order to re-predict these foodscapes to reflect new conditions (e.g., new forest fires or silvicultural harvest), there are 5 input covariates that must be updated (Table G1). Ecosite classification does not need to be updated because they reflect soils and long-term microclimate that remains stable over time; however, the seral classification may need to be updated given stand replacing disturbances (mid-late seral resets to early-seral) or forest succession where early-seral stands (<20 years) mature into mid-late seral stands (> 20 years). Because we provided the ecosite layer, the seral classification of these ecosites can be updated. If new spatial layers of basal area or canopy cover become available these layers can be incorporated in the R code. Additionally, if stand-replacing disturbances occur, these areas can be changed to a value of zero in the forest characteristic layers. The normalized difference moisture index (NDMI) and change in enhanced vegetative index (Δ EVI) were calculated for the year 2018, but these can be recalculated for future years. We calculated NDMI for each pixel as the maximum value derived between 1 June and 30 September from Landsat imagery. We calculated Δ EVI for each pixel using MODIS data as the mean EVI value derived across 1 July - 1 August minus the mean value of EVI derived across 1 September - 1 October. Once these input layers are updated, they can be incorporated in the provided R code to re-predict the foodscapes and caribou resource selection. It is critical to evaluate the distribution of values from these re-predicted foodscapes to ensure these foodscapes are re-predicted correctly.

Table H1. Input covariates used to predict caribou foodscapes and whether these covariates need to be updated when re-predicting these foodscapes to reflect new conditions.

Input covariate	Source	Need to update
Seral-specific ecosites	Far North 1.4 land cover, Ontario 2000 land cover, and Canadian commission land cover	Ecosite classification does not need to be updated, but seral classification may need to be updated.
Basal area	Matasci et al. 2018	Yes
Canopy cover	Matasci et al. 2018	Yes
NDMI	Landsat	Yes
deltaEVI	MODIS	Yes
Sand	SoilGrid250m, Hengl et al. 2017	No
Silt	SoilGrid250m, Hengl et al. 2017	No
Clay	SoilGrid250m, Hengl et al. 2017	No