

**Relationships Between Late-autumn Body Condition of Adult Female Caribou and
Nutritional Resources in the southern Northwest Territories**

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Executive summary: The plight of woodland caribou represents one of the most serious conservation issues in Canada. In some areas, anthropogenic development impacts caribou by favoring alternate prey such as moose and their predators, and this effect may increase predation on caribou, impacting caribou populations. At the same time, evidence is increasing across North America that recognizes the importance of influences of nutritional resources, with new studies showing that ungulate populations, including woodland caribou, are routinely challenged by marginal forage not only in winter but also during summer.

ECC and NCASI, in past collaborations involving monitoring body condition of adult female caribou in the southern Northwest Territories, found that caribou had relatively good body condition throughout winter, but experienced substantial declines in spring to an annual minimum by early summer. Additional studies by ECC documented that a primary peak in annual mortality of adult females in late June, July, and August coincided with this minimum and that starvation was a common cause of death at that time. These findings suggest that nutritional and bioenergetic environments in summer may have important influences on caribou populations in the southern Northwest Territories. Thus, having a better understanding of relationships between nutritional resources on these summer ranges and body condition and nutritional status of caribou may provide a more holistic and effective basis for conservation planning for caribou.

With funding from the NWT Cumulative Impacts Monitoring Program, additional studies by ECC and NCASI were initiated to better define good habitat from a nutritional perspective. This work integrated two phases of investigation. The first (see companion CIMP report [Cook et al. 2023]) involved a detailed survey of forage quality and quantity to better understand at fine scales which habitats best provide good nutrition to female caribou with calves in summer. The second, the primary focus of this report, involved evaluating linkages between body condition (as indexed by body fat levels) of collared adult female caribou in December and their habitat use over summer and early autumn to identify those environmental conditions that best supported nutrition and body condition of caribou. Objectives of this work were to identify which forage types, vegetation types, and post-wildfire successional stages on

caribou summer ranges influenced body fat levels of caribou in late autumn, and to provide landscape maps of nutritional resources that showed areas of good and poor nutritional resources that would be useful for range evaluation and landscape planning. We also evaluated relationships between body fat levels and human activities based on the distance of caribou locations from actively used gravel and paved roads. We constructed spatially-explicit, detailed (30 m x 30 m resolution) landscape maps of 18 habitat and nutrition metrics that potentially directly and indirectly reflected nutritional resources. These 18 variables included dominant ecosites of the study area, forest overstory characteristics, forage biomass, and predicted caribou nutrition (based on prediction equations developed with captive caribou trials in northeastern British Columbia). Using validated ultrasound and body condition scoring techniques, we measured December body fat levels of 29 caribou in 2019 and 25 in 2021 that had raised a calf the previous summer and autumn. We plotted the GPS relocation points (three locations per caribou per day) of each caribou in summer (from 15 July to 15 September) on our landscape maps to determine caribou use of each of our 18 habitat and nutritional resource metrics. We used regression analysis to identify significant relationships between body fat and each of our metrics, with a focus on identifying a combination of these metrics that best defined body fat relationships with their environment. We included year as an analysis metric because weather patterns in spring and summer between the two years were markedly different (roughly double the precipitation in 2021 compared to 2019).

In both years, our data indicated that body fat levels were correlated to predicted daily intake of digestible energy, in turn indicating forage conditions that best contributed to caribou body fat were those that provided high forage biomass to support high forage intake rates and forage conditions that allowed caribou to consume forage of high digestible energy content. None of our digestible protein metrics were significant in either year. However, the habitats that best provided good nutrition differed significantly between years. In 2019, body fat of caribou was positively related to their use of younger stands with higher levels of predicted daily intake of digestible energy (reflecting higher levels of vascular plants that caribou preferred) and negatively related to stands with abundant lichens (primarily older stands in the upland black spruce and, to a lesser extent, wetland bog ecosites). In 2021, body fat of caribou was

positively related to their use of areas with abundant lichens including upland black spruce and areas with higher levels of predicted daily intake of digestible energy (reflecting higher levels of lichens) and exhibited an inverse trend (not significant) with stand age. For each of the 18 metrics, we found that the nature of their relationship with body fat (indicated by the sign of the regression slope coefficient) was opposite each of the two years, suggesting that strategies of caribou to acquire good nutrition that supported body fat levels differed substantially between years. These results complicate simple conclusions about what types of habitat best contribute to nutrition and body condition of caribou, but indicate that a wide diversity of habitat conditions are important for caribou over the long term.

In combination with findings from the forage quality surveys described in the CIMP companion report (Cook et al. 2023), we conclude the following.

- (1) Much of the vegetation (roughly about half) that grows in caribou habitat is non-food, meaning that palatability, nutrient content, and/or toxic compounds render these plants of low value to caribou. Caribou exhibit strong preferences for certain plant species (Denryter et al. 2017), e.g., willows, birch, several species of forbs, hairy wildrye, reindeer lichens, arboreal hair lichens, and, perhaps most of all, mushrooms, most of which compose small portions of total vegetation biomass available to caribou.
- (2) Amount of accepted forage varies markedly among ecosites and seral stages (stand age). Biomass of accepted grasses, forbs, and deciduous shrubs are most abundant in early seral stages and declines to relatively low levels in older stands. Biomass of these plants tend to be highest in fens and upland white spruce ecosites and lowest in bogs, upland black spruce, sphagnum bog, and marshes ecosites. Accepted lichens are largely absent in early seral stages up to about 40 years old, are sparse in marshes, sphagnum bogs, wet bogs, and upland white spruce ecosites of all ages, and tend to be abundant in relatively dry bogs and some upland black spruce ecosites. We found some evidence that accepted lichens tended to decline in older stands especially where overstory canopy cover and basal area were high. These results suggest a dichotomy of forage resource types, those that provide relatively high biomass of accepted vascular plants and those that provide high abundance of accepted lichens, and that they tend to occur in different ecosites and different successional stages.

(3) Body fat in December of adult female caribou that raised a calf over summer and early autumn in December ranged similarly in 2019 and 2021, from 4-5% to 13- to 15% body fat. The wide range in body fat within years (2019 and 2021) suggests that varying forage abundance, composition, and quality available to each caribou may have important influences on their body condition, a primary hypothesis of our study.

(4) Predicted levels of caribou nutrition (primarily daily intake rates of digestible energy) at caribou GPS locations during summer was modestly, but significantly correlated to their body fat levels in December, but which habitat features and metrics of nutritional resources that were correlated with body fat varied between years, which probably reflects the substantial difference in precipitation during spring and summer during the two years. In the first year, biomass of accepted vascular plants, especially grasses, forbs, and deciduous shrubs (combined) and the habitats in which these are most abundant (young stands) was positively related to body fat whereas biomass of accepted lichens was inversely related. In the second year when precipitation levels were markedly elevated, biomass of accepted lichens was positively related to body fat.

(5) The alternative findings between 2019 and 2021 preclude simple conclusions about which types of habitat and forage best contributed to body fat of caribou. They suggest that a highly diverse combination of habitats and forage types best accommodate nutritional needs of caribou across a wide range of long-term climatic conditions. However, our results for the 2019 evaluation of body fat are supported by the resource selection studies of DeMars et al. (2020) that indicate boreal caribou in the Northwest Territories select for early seral vegetation communities during the summer; vegetation communities that result from wildfires. On the other hand, our findings do not in any way distract from the importance of lichens in winter, and the importance of lichens in our second year of data (2021) may underscore the importance of lichens during summer at least during some years depending on weather.

(6) We found no evidence that body fat of caribou was reduced for those animals that ranged near roads, indicating that under the conditions of this study, human activity had little or no influence on caribou body condition in autumn.

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INTRODUCTION

The negative influences of predation and anthropomorphic disturbances on boreal caribou (*Rangifer tarandus*) populations have received considerable emphasis for research and conservation in Canada. Bottom-up factors, those that involve the environmental (climate, habitat quality) and especially forage factors that can affect nutrition and health of caribou and, in turn, also affect caribou populations, typically receive less emphasis. This is largely because increasing anthropomorphic disturbances may increase rates of predation and these are 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). Across much of the Northwest Territories, however, anthropomorphic disturbances are modest compared to many boreal caribou ranges in Canada, and the relative importance of bottom-up environmental factors for long-term sustainability of caribou may be greater than other ranges, particularly if climate change advances. The debate about the relative importance of bottom-up and top-down (predation) factors in the control of herbivore populations has been on-going for decades and remains a “surprisingly contentious issue” (Schmitt and Holbrook 2007). Despite this, both are important and the relative influences of each undoubtedly vary depending on local circumstances that vary across space and time (Hunter and Price 1992, Gustine et al. 2006, Johnson et al. 2019, Neufeld et al. 2021). Increased understanding of their separate and interacting effects on herbivore populations may improve conservation planning for these animals.

Substantial advances have occurred in the last two decades primarily with elk (*Cervus canadensis*), moose (*Alces alces*), bighorn sheep (*Ovis canadensis*) and deer (*Odocoileus* spp.) in the understanding of nutritional influences, a key bottom-up factor. The advances have altered perspectives on the way in which 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, Mereszczyk 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, refuting an old perception that the quantity and quality of forage in summer is sufficient to fully satisfy nutritional requirements. During the growing season, key periods of animal production occur, including the third trimester of gestation (depending on latitude), birthing, calf-rearing, growth and development of juveniles, restoring endogenous energy reserves used the previous winter and spring, and breeding in autumn. Requirements for digestible energy and protein for adult females with juveniles at heel to support many of these processes are 2- to 3-times greater than in winter (Oftedal 1985, Cook et al. 2004, National Research Council 2007). Captive-animal studies show that nutritional deficiencies stunt animal performance including reduced rates of body fat accrual during summer and autumn; probability and timing of ovulation; body condition at the beginning of

winter; neonatal vigor and size; juvenile growth, development, and survival; growth and development of subadults; and age at first breeding (Verme and Ozoga 1980; Cook et al. 1996, 2004; Tollefson et al. 2010), including for caribou (Crête et al. 1993). Studies conducted for free-ranging wild herbivores corroborate findings from these captive studies for a variety of large ungulate species in the northern ecosystems around the globe (Cook et al. 2013, 2018; Hjeljord and Histol 1999; Monteith et al. 2015; Peek et al. 2002; Hurley et al. 2014, Proffitt et al. 2016; Rolandsen et al. 2017; Horne et al. 2019; Schrempp et al. 2019; Stephenson et al. 2020, Cook et al. 2001c).

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 (λ) in the mountains of central British Columbia across a range of predator densities. Cook et al. (2021a) reported that the minimum in body condition around the annual cycle occurred in late spring and early summer in many caribou populations, that the level of body condition at this time can be so low as to be life threatening (i.e., directly through starvation or indirectly through vulnerability to predation), at least in some populations, and that causal links may exist between survival and nutrition in late spring and summer. Kelly (2020), who conducted one of the largest studies of mortality of adult female caribou, documented marked differences in mortality among seasons with two primary peaks in mortality in the Northwest Territories. The first occurred in April and coincided with a period of elevated movement rates of caribou just prior to birthing, and the second was in July and August, when starvation was a primary cause of death (40% of mortalities). The peak in mortality during mid-summer is a well-documented phenomenon for woodland caribou in western Canada (Wittmer et al. 2001, McLoughlin et al. 2003, Cook et al. 2021a). Finally, caribou exhibit surprising tolerance to harsh winter settings. Based on studies that tracked changes in body condition over winter, the preponderance of data show that (1) adult female caribou generally retain body condition levels over most of winter in most populations (Couturier et al. 2009, Cook et al. 2021a), and (2) that body condition levels over winter are largely a function of their body condition in late autumn, which in turn is a function of their reproductive state and the bioenergetic and nutritional environment they occupied during the previous summer and early autumn (Cook et al. 2021a).

The ability to satisfy nutritional requirements for reproduction, growth, and survival in caribou reflects two basic foraging attributes: how fast they can eat and the nutrient content of the food that they

consume. These attributes are fundamentally set at very fine scales and scale up to affect higher order processes (i.e., reproduction and survival) (Shipley 2007, Cook et al. 2018). Abundance of forage that is nutritious and that caribou readily consume (e.g., acceptable forage) influences short-term (i.e., per minute) intake rates by influencing encounter rate with accepted forage and time spent searching for these forages (a type of functional response; Spalinger and Hobbs 1992, Cook et al. 2016, Denryter et al. 2022). Forage quality directly affects the amount of nutrients per gram of consumed food, but potentially more importantly, lower forage quality limits forage intake rate at longer time intervals (e.g., daily) by slowing rate of digestion and increasing the time spent ruminating (Shipley and Spalinger 1992, Owen-Smith 2002, Tafaj et al. 2005, White et al. 2014). In addition, as plant species composition, plant phenology and chemistry, and abundance vary across landscapes, so too does forage quality and quantity (Johnstone et al. 2002, Cook et al. 2016). Thus, nutritional value of plant communities to ungulate herbivores and the extent of nutritional limitations operating these animals varies substantially across space and time (Shipley 2007; Cook et al. 2013, 2018).

Beginning in 2013, staff of NCASI and the Department of Environment and Climate Change of the Government of the Northwest Territories initiated studies to better understand the influence of nutritional resources on the productivity of caribou populations in the southern Northwest Territories. Earlier studies focused on measuring body condition of adult female caribou using ultrasound and body condition scoring techniques during capture work to fit caribou with GPS collars. Estimates of body fat provide valuable insight regarding the adequacy of nutrition experienced by the animal over the previous several months and insight regarding how well the animal is likely to perform in the future (Keech et al. 2000; Cook et al. 2004, 2013, 2021b; Monteith et al. 2013, 2014; Stephenson et al. 2020, Cook et al. 2021c). Efforts to document body condition of caribou in the Northwest Territories began with a late-winter survey to identify the potential for nutritional limitations in part by comparing to results of similar capture efforts in multiple herds in British Columbia and northern Ontario. The effort continued in the Northwest Territories where caribou were captured in December and recaptured in March to document overwinter changes in body fat. These surveys of body condition indicated two important patterns: (1) caribou in the southern Northwest Territories were in relatively good body condition at least in late winter compared to other herds in the surveys (Fig. 1), and (2) as mentioned above, body condition of these caribou declined only slightly through March but seemed to decline rapidly to summer lows thereafter (Cook et al. 2021a). Nevertheless, surveys conducted in December indicated considerable variation in body condition of adult females that succeeded in raising a calf that year, from poor at 4-5% body fat to excellent at 13-15% body fat. Understanding what habitat factors accounted for these levels of body fat may provide valuable insight with useful management applications.

As has occurred on other caribou ranges further south in Canada, anthropomorphic development is likely to increase in the future in the Northwest Territories, and thus the urgency for protecting important habitats may increase as well. Additional collaborative studies were initiated between ECC and NCASI, funded by the NWT Cumulative Impacts Monitoring Program (NWT CIMP), to better define good habitat from a nutritional perspective for caribou in the region. This work integrated two phases of investigation. The first involved a detailed survey of forage quality and quantity to describe forage resources of our study area (described in CIMP companion report by Cook et al. [2023]). The second, the topic of this report, evaluated linkages between body condition of adult female caribou and their use of habitat and nutritional resources in the southern Northwest Territories. We measured body condition in December of 2019 and 2021 of caribou that had raised a calf over the previous summer and autumn, where body condition was indexed by levels of body fat expressed as a percent of the ingesta-free body (Cook et al. 2021b). These caribou had been captured in previous years and equipped with global positioning system (GPS) collars, and thus their locations during summer prior to their capture in December was known and provided a basis for comparing environmental conditions (e.g., habitats, forage abundance, and predicted caribou nutrition) at location points during the summer with body fat levels in late autumn. Measures of body condition in December are particularly useful because they indicate how well the nutritional resources available to the caribou during summer and early autumn satisfy their nutritional needs and indicate the levels of body stores of energy available to the animal for withstanding harsh winters of the region and the energetic demands of the following spring and summer.

We relied on previous research in British Columbia, in particular, and in northwestern Ontario with captive caribou to identify forages that caribou readily consumed and those that they avoided, and to develop equations to predict key caribou nutrition levels as a function of forage attributes available to them in native habitats (see additional details as described by Denryter et al. 2017, 2022).

Our goal was to identify those summer habitat conditions that best supported development of body condition of adult caribou that had dealt with satisfying the nutritional demands raising a calf. We evaluated relationships between each caribou's body fat level as a function of the habitats and nutritional resources they used over summer, where nutritional resources were based on forage surveys conducted during 2016 and 2019. Our analysis emphasized the effects of disturbance (primarily due to wildfire events) and subsequent forest succession on forage resources and body fat levels of the caribou, largely because wildfire disturbance and succession represent one of the most influential factors affecting abundance and quality of forage in the southern Northwest Territories. The data sets provided a unique opportunity to evaluate several key predictions. (1) Levels of body fat in December were a function of the habitat and forage resources that caribou used in summer while raising a calf. (2) Body fat levels are more strongly correlated with habitats and nutritional resources caribou used in late summer than early summer.

This is because forage quality typically declines over summer at a time when caribou greatly need good nutrition to recover body fat and body protein as well as provide milk for their calves, and thus body fat levels may be more affected by the ability of caribou to access habitat that provides good nutrition in late summer than in early summer. (3) Use of early seral communities, and the vascular plants that are typically more abundant in these stands, is better correlated to body fat levels in December than older communities even though biomass of lichens typically is lower in young stands. This prediction reflects that lichens provide virtually no digestible protein, often very low bite sizes that can impede how fast caribou can eat, and that digestible energy levels in non-lichen food may provide digestible energy levels just as high as lichens (and much higher digestible protein levels) (see CIMP companion report [Cook et al. 2023]).

METHODS

Our overall study included five main tasks.

- (1) Conduct forage surveys and relate forage characteristics to ecosites and forest succession in response to wildfires and subsequent succession in the southern Northwest Territories (SNWT). Details of methods and results from this work were presented as a separate, companion report under the NWT CIMP agreement (Cook et al. 2023).
- (2) Using this forage data, calculate caribou nutrition from forage resources primarily based on prediction equations developed using captive caribou deployed to enclosures in native habitats of northeastern British Columbia (see Denryter et al. 2017, 2021) and relate these estimates to ecosites and disturbance and succession. Detailed methods and results of this step are also presented in the CIMP companion report of Cook et al. (2023) and only briefly summarized here.
- (3) Develop spatially-explicit maps of habitat types, forage, and caribou nutrition across our study area to provide a basis for relating caribou GPS locations to the habitat conditions that they used.
- (4) Capture and evaluate body condition of wild caribou that had raised calf in the SNWT.
- (5) Evaluate relationships between key measures of habitat and nutritional resources that caribou used and their levels of body fat. These measures included prominent ecosites (i.e., vegetation communities) in the SNWT, abundance (kg/ha) of accepted forage (accepted is defined below) of vascular and lichens, and key measures of caribou nutrition including daily intake rate of digestible energy (MJ/day) and digestible protein (g/day) (if caribou spent the entire day foraging in the ecosite) and concentrations of digestible energy (kJ/g) and digestible protein (%) in caribou diets.

Study area

We sampled vegetation primarily along all open roads across much of the SNWT, from Jean Marie River in the west to the Pine Point region in the east (Fig. 1). However, about 15% of our sampling

was in remote areas far from roads accessed via helicopter. Our sampling was conducted in the Taiga Plains Mid Boreal ecoregion (Ecosystem Classification Group 2007). The boreal system consisted of complex mosaics of wetlands and upland forests. Upland forests supported stands of black spruce, Jack pine, white spruce, and conifer-hardwood mixes with mixed-shrubs dominating. Wetlands included open fens dominated by dwarf birch, willows, and other deciduous shrubs, treed fens with sparse larch and black spruce forests, open and treed bogs with sparse black spruce and ericaceous shrubs, and sedge-dominated wetlands (Ecosystem Classification Group 2009, Ducks Unlimited Canada 2014). Palatable ground lichens common in the region included reindeer lichens and other species that occurred sporadically on favorable growing sites across the study area. (Scientific names of plant species are presented in Appendix A.)

Climate of the boreal forests of the southern Northwest Territories and northeastern BC was dominated by continental arctic and continental polar air masses (DeLong et al. 2011). Summers were warm and moist, and winters were cold and snowy, with mean annual temperature of -1 C to -4.5 C and annual precipitation averaging from 400-460 mm across the region (Ecosystem Classification Group 2009). Based on weather data collected and averaged from two Hay River weather stations (Hay River A, 2202401 and Hay River Climate, 2202402)

(https://climate.weather.gc.ca/prods_servs/cdn_climate_summary_e.html) and long-term normals (1991-2020) at Hay River

(https://climate.weather.gc.ca/climate_normals/results_1991_2020_e.html?searchType=stnName_1991&xtStationName_1991=Hay+River&searchMethod=contains&txtCentralLatMin=0&txtCentralLatSec=0&txtCentralLongMin=0&txtCentralLongSec=0&stnID=407000000&dispBack=1), precipitation during spring through summer differed markedly between the two years of our study, with about twice the amount of precipitation during 2021 versus 2019. In addition, precipitation was 57% and 66% of long term normals in 2019 and 137% and 152% of normal in 2021 in spring and summer, respectively (Fig. 2). Temperature was more consistent during the spring through summer period, although temperature averaged nearly 2 C warmer in 2021 than 2019 (Fig. 2).

Vegetation sampling

Plant community type classification approach.—Fundamental to the work that we conducted is the vegetation classification system we used for evaluating caribou forage. The system we used reflects a decade of work mainly in British Columbia and Ontario and was relatively simple to apply, applicable to all areas that we worked, and, critical for our work, was reasonably effective at grouping sites based on similar undergrowth vegetation (forage) composition. We extended this system to our work in the SNWT. It borrowed from several classification systems in existence in the areas that we worked including the Ducks Unlimited system for wetlands (Ducks Unlimited Canada 2014), the biogeoclimatic potential

natural vegetation system of British Columbia (DeLong et al. 2011), and the ecosite classification system developed for western Ontario (Racey et al. 1996). The British Columbia biogeoclimatic strategy is based on a site potential, i.e., a potential natural vegetation approach that strives to classify sites based on the soils and climate characteristics and the vegetation that will occur in late seral stages. 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 bogs with black spruce overstory will eventually grow back to bogs with a black spruce overstory after stand-replacing wildfire, given sufficient time for the black spruce bog community to reestablish.

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 provides a superior basis for landscape planning than those based on overstory cover types (Haufler et al. 1994). For the SNWT, our approach recognized two upland forest types and four wetland types: upland black spruce on coarse-textured, unproductive soils, upland white spruce on fine-textured, deep, and productive soils, and, in wetlands, bogs, fens, marshes, and sphagnum bogs (Table 1).

Measuring forage quantity.—Our sampling of forage quantity and quality emphasized measuring in detail those plant species that caribou prefer or at least readily consume; we deemphasized those species that caribou typically will not eat in summer or at least caribou exhibit significant avoidance. The identification of these species was based on sampling of caribou forage selection patterns using captive caribou placed in a wide variety of native plant communities in the boreal and montane ecosystem of northeastern British Columbia during summer and early autumn of 2013 to 2016 (described in detail by Denryter et al. 2017, 2022). In that study, composition of plant species in caribou diets were compared to intensively sampled composition of plant species in the plant communities available to them, providing a rigorous basis to statistically identify which plant species the caribou ate in higher proportion than what was available, ate in proportion to composition of plant species in the communities available to them, and those species which caribou significantly avoided. For our (SNWT) study, we grouped the species into two categories, accepted species, which are those that caribou significantly selected or at least readily consumed (i.e., proportionally more common in diets than proportion available or proportionally equal in diets and available in the plant community) and avoided species, those which caribou did not eat or were proportionally less common in diets than proportionally available to them in the plant community. Based on a variety of studies (Cook et al. 2016, 2018; Denryter et al. 2022, Ulappa et al. 2020; Monzingo et al.

2023), such a distinction provides a valuable approach for evaluating nutritional resources for large ungulates. Even though the caribou dietary data were collected in northeastern British Columbia, not the SNWT, basing caribou forage preferences on detailed foraging and plant community composition data has considerable advantages due to the statistical rigor provided by detailed comparisons of consumed versus available. In Appendix A, we list the species we encountered in the SNWT and the caribou selection categories for these species determined from the work in northeastern British Columbia.

We sampled vegetation during two summers between 3 July and 31 August, 2016, and 17 June and 11 September, 2019. We measured overstory and undergrowth characteristics in 0.418 ha rectangular macroplots (Cook et al. 2016, 2023). Macroplot locations were randomly selected with randomization constrained to within 200 meters of roads. This approach tended to result in a proportional allocation of sampling effort (i.e., under sampling of rare ecosites). However, we bolstered our sampling for uncommon ecosites and seral stages with opportunistic site selection. In 2019 and for a day in 2021, we used helicopters to a limited extent to sample vegetation in areas far removed from roads. Our intent was to sample vegetation types that were infrequent along roads, and this included the Sphagnum bogs and lichen-rich communities that were typically dominated by black spruce or black spruce-Jack pine mixes (usually relatively dry bogs or upland-black spruce ecosites). Aerial imagery was used to identify locations where these ecosites were clustered, and a subset of these were randomly chosen for sampling. The helicopter landed at these locations early in the workday and returned at the end of the workday to retrieve the crew. Staff traveled from the drop point along a pre-planned route to stands pre-selected for sampling until the end of the workday.

We measured forest overstory characteristics and forage biomass along three parallel, 46-m transects in each macroplot. For overstories, we measured overstory canopy cover at ten equally spaced points along each transect with a moosehorn (a vertical sighting tube) (Cook et al. 1995). At two 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. We estimated density (stems/ha) of seedlings (< 1 cm DBH) and trees by counting each in 5x15-m belt transects (2.5x15-m in very dense forest stands), with one belt transect per sample transect. We measured mean stand height, slope, and aspect at the center of the macroplot using a clinometer and compass. 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). Stand ages could be more reliably measured in upland stands that tended to be even-aged in these fire-prone settings; tree age as a measure

of stand age may often be less accurate in wetland forest types due to less frequent stand replacing fire and slower reestablishment of trees after perturbation.

For forage quantity, we sampled undergrowth biomass by individual species of lichens and vascular plants by clipping from two meters in height to a two-cm stubble in six, two-m² circular plots per macroplot (two plots per transect). We clipped current annual growth of vascular plants. Arboreal hair lichens also were clipped up to two m in height above ground. For terrestrial lichens, dead lichen matter and other detritus (sticks, conifer needles) were removed from the lichen mat at the time of clipping. Caribou carefully avoid eating non-live matter in these mats while foraging in summer. The clippings were bagged individually by species in paper sacks and oven-dried to constant mass and weighed.

Measuring caribou nutrition and food habits.—Tame animal foraging trials used to measure nutrition and food habits of caribou in northeastern British Columbia were conducted during summer of 2013 through 2016 (methods were described in detail by Denryter et al. (2017, 2022) and to some extent in the CIMP companion report by Cook et al. [2023]). Briefly, lactating females and their calves (about 15 total pairs were used over the course of the study) were transported to enclosures (four pairs per enclosure) averaging about one hectare in size that were constructed with electric fencing in native habitats. Using bite counting methods that involved close observation (<1 meter) by trained personnel, caribou were sampled while foraging to determine species composition of their diets, intake rate of forage on a per-minute basis, digestible energy and digestible protein content of their diets, and intake rate of digestible energy and digestible protein on a per-minute and daily basis. Digestible energy and protein content of caribou diets was based on fiber fractionation techniques described by Robbins et al. (1987a,b) with adjustments as described by Cook et al. (2022) and was conducted by the Wildlife Habitat and Nutrition Lab and the Soil, Plant, and Waste Analytical Lab at Washington State University, Pullman, Washington. Foraging time per 24-hours was estimated using activity accelerators placed on caribou collars and was multiplied by per-minute intake rate of digestible energy and digestible protein to estimate daily intake of both nutritional attributes. In addition, sampling to estimate forest overstory characteristics and forage quantity by plant species was conducted in each enclosure using field sampling methods identical to those used to measure forage and overstories in the SNWT. This sampling of forage biomass and composition in each enclosure provided the basis to compare composition of plant species in caribou diets with composition of plant species available to the caribou in each enclosure, with each enclosure composing the experimental unit for analysis. Forage biomass and composition also provided the basis to develop prediction equations relating caribou nutrition to the vegetation from which they foraged.

Estimating caribou body condition

We captured adult female caribou to evaluate body condition using a net gun fired from a helicopter (general methods of capture and sampling of body condition for the SNWT were presented by Cook et al. 2021a). Our goal was to capture 30 adult female caribou per year in December 2019 and 2021 that had previously been equipped with GPS collars and raised a calf the year of capture. We captured caribou already carrying GPS collars so that data would be available describing where the caribou spent the summer and autumn period before their capture in December ultimately to relate habitat use to body condition. We captured caribou cows that had raised a calf because of the strong effect on body condition of lactation and other activities involved with raising a calf (Cook et al. 2013, 2021a). For the latter, we conducted aerial surveys in late October/early November to identify which caribou females with GPS collars also had a calf at heel just prior to the beginning of capture in early December of each of the two years. Knowledge of which animals raised a calf that year precluded expending resources capturing caribou that had not raised a calf that year.

We estimated body fat expressed as a percent of the ingesta-free body, body mass (weight), and thickness of the loin muscle (Cook et al., 2021b), although we used only body fat data for analyses herein. Using an Ibex ultrasonograph with a 5.0 MHz, 7.0-cm probe (E.I. Medical Imaging, Loveland, Colorado), we measured maximum thickness of the rump fat layer (MAXFAT; Stephenson et al. 1998; Cook et al. 2001, 2007, 2021b) and assigned a body condition score (Cook et al., 2021b). We arithmetically combined rump fat thickness, allometrically scaled to surface area using a small ungulate scaling factor ($\text{MAXFAT}/0.150 \times \text{BM}^{0.630}$ where BM = body mass in kg), to adjust for differences in caribou size, and body condition score (i.e., scaledLIVINDEX: method 3 described in detail by Cook et al. 2010, 2021b) to calculate fat of the ingesta-free body. We measured the thickness (cm) of the loin muscle between the 12th and 13th rib using ultrasonography (Cook et al., 2021b). We used chest girth circumference to estimate body mass (kg) (Cook et al. 2003), minus the products of conception, using equations that accounted for body fat levels (Cook et al., 2021b). An experienced member of the capture crew estimated age of caribou based on incisor wear.

As a confirmation that the caribou we captured indeed raised a calf (initially evaluated during the aerial surveys the previous month), we evaluated udder characteristics (e.g., presence, color, and amount of fluid from the udder [Cook et al. 2013] and udder size). We classified as females having raised a calf if (1) her udder contained milk or (2) a calf accompanied the female during either the aerial survey or was present during helicopter pursuit, and the udder displayed characteristics of recent nursing (swollen udder and presence of fluid) (Chan-McLeod et al. 1999; Cook et al. 2021a).

Vegetation Data Analysis

We included six primary steps in the analysis of our data in support of our study objectives. These included the following.:

- (1) Quantify vegetation response (overstory characteristics and forage quantity) to wildfire disturbance and succession for each ecosite, the latter (ecosites) to account for biogeoclimate effects on vegetation in relation to disturbance and succession. This analysis was reported in the companion CIMP report (Cook et al. 2023), although we illustrate important relationships here.
- (2) Develop predictive models of forage composition and production (to be used for development of maps of nutritional resources) for each ecosite over time since fire. This analysis was also reported in the companion CIMP report (Cook et al. 2023) but we include a table of predictive models because these were used to develop the foodscape maps described below.
- (3) Develop predictive models of caribou nutrition using data from captive caribou field research in northeastern British Columbia (to be used for development of maps of nutritional resources). These were described in the companion CIMP report by Cook et al. (2023), but we include a table of final models and graphs of predictions to summarize the results.
- (4) Construct foodscape models of caribou nutritional resources.
- (5) Describe caribou body condition.
- (6) Evaluate relationships between caribou body condition and their use of habitats and nutritional resources.

Thus, we present only a brief summary of analyses methods here for items 1 through 3 but present detailed methods for items 4 through 6.

Vegetation characteristics.—Because wildfire disturbance and succession is a primary driver of forage composition and quantity, a primary step in the analysis was to evaluate forage changes in relation to time since disturbance, using stand age as our primary surrogate measure of succession after wildfire. We first plotted forest overstory characteristics including overstory canopy cover, basal area, stand height, composition of trees that were hardwoods, and composition of trees that were Jack pine with stand age for each ecosite to evaluate forest overstory re-establishment after disturbance. We then evaluated relationships between stand age and quantity of forage with emphasis on four forage groups: biomass (kg/ha) of accepted grasses, forbs, and deciduous shrubs (combined), accepted lichens (including arboreal hair lichens and terrestrial lichens), accepted horsetails (*Equisetum* spp.), and accepted mushrooms because each of these contributed to predicting our caribou nutrition variables (i.e., concentration of digestible energy and digestible protein in caribou diets and daily intake rate of digestible energy and digestible protein). The influences of stand age on overstory and understory vegetation are often non-linear. Hence, we screened our vegetation variables using Akaike information criterion corrected for small sample sizes (AICc) calculated by Curve-Expert (Hyams 2017) as a guide 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. If AICc of the non-linear regression was > 4 units lower than for the linear regression, we used a non-linear formulation.

Predicting forage biomass levels.—A primary objective of the overall study was to develop models for predicting biomass of each of our four forage classes (accepted grasses, forbs, and deciduous shrubs combined, accepted horsetails, accepted lichens, and accepted mushrooms) that we in turn used to construct foodscape maps. We developed prediction equations using five predictor covariates representing site characteristics: stand age, overstory canopy cover, basal area, stand height, tree density, and proportion of trees that were hardwood species. (We also considered and found to be ineffective three vegetation greenness indices, NDVI, NDMI, and ESI as predictor covariates of forage quantity). These analyses were conducted for each vegetation ecosite separately, to account for different influences of site potential on forage abundance.

For model selection, we again 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). 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. This linearization process involved replacing the original covariate (e.g., stand age) with the non-linear equation relating it to the dependent variable (e.g., biomass of lichens) in the multiple regression with dependent variable. This effectively transformed or linearized the original data in a way that allowed the non-linear relationship to be incorporated into the multiple regression analysis with other the other covariates.

We used multiple regression to develop or forage quantity prediction models. Selection was based on AIC in Proc Regression in SAS as a guide to select the best model. For our primary forest overstory covariates, we recognized these tended to be collinear, so we included each in all regression runs, but used a variance inflation factor (VIF) of < 10 for each covariate as an indicator to evaluate extent of multicollinearity (if included covariates resulted in $VIF > 10$, then one of the offending variables was removed, and the analysis was rerun). We limited the number of covariates included in the model to $1/10$ the sample size (McGarigal et al. 2000). In our initial analyses, we found AIC tended to include covariates that were substantially non-significant (e.g., $P > 0.20$). To avoid over-parameterized models, we excluded covariates if their P value was > 0.10 . This resulted in simpler models that in our opinion were less likely to provide spurious predictions of forage biomass when applied to the landscape. Finally, we generated QQ normality graphs and plotted model residuals to evaluate violations of normality and homoscedasticity for each model.

Predicting caribou nutrition.—Based on the foraging trials conducted using captive caribou in our northeastern British Columbia study areas, we developed models to predict key caribou nutrition metrics as a function of the abundance of accepted species. We predicted seven total 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 of 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. Of these, however, we used four for foodscape development here: digestible energy and digestible protein content of caribou diets and daily intake rate of each of these nutrients.

Analysis of BC and ON captive caribou foraging data (Cook et al., *in prep*) indicated that caribou nutrition 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. Thus, the primary covariates for predicting caribou nutrition included these four forage metrics, all on a kg/ha basis. Sample size of caribou enclosures was too small for model development for each ecosite, so we opted to develop prediction equations for uplands and lowlands separately ($n = 47-51$ for uplands and 17-18 for wetlands; the lower samples sizes reflected sites where we were unable to hold caribou for 24 hours).

To develop the nutrition prediction 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 again 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 (Statistical Analysis Systems, 1988) to guide the selection of the best models. We set total number of covariates to 1 per 10 samples, although in a few cases we allowed 1.5 covariates per 10 samples, and we required that each covariate was significant at ≤ 0.10 when included in the model (to simplify and avoid over-parameterization of the final models). We used a variance inflation factor (VIF) to identify issues with multicollinearity and excluded covariates with $VIF > 10$ as appropriate to reduce multicollinearity issues. Finally, we generated QQ normality graphs and plotted model residuals to evaluate violations of normality and homoscedasticity for each model.

In the analysis of our foraging data from British Columbia and Ontario, the role of accepted lichens was overall insignificant in predicting nutrition responses. However, we found in our development of caribou nutrition prediction equations that, in some ecosites in the British Columbia data sets, when the effects of other predictor covariates were included in the model, that biomass of accepted

lichens was significantly, positively related to forage and digestible energy intake rates. This contrasted sharply with results from our work in Ontario (Walker 2023) and some ecosites in British Columbia where accepted lichen biomass was an unimportant predictor of caribou nutrition, and thus created uncertainty about the role of accepted lichens in predicting caribou forage intake rates. We created two versions of caribou nutrition equations, those that included accepted lichens as a predictor covariate when significant, and those that excluded accepted lichens as a covariate, even if it was significant. Our intention was to generate alternative estimates of predicted caribou nutrition and use both in our analysis of their effects on caribou body fat, ultimately to use the caribou body fat data to guide our decision regarding which prediction equations (those with and those without accepted lichen biomass) best accounted for body condition of caribou in southern Northwest Territories.

We then used our equations to predict caribou nutrition (e.g., dietary DE and DP per day, etc.) for each macroplot sampled in Northwest Territories to illustrate the effects of ecosite and disturbance and succession. Because many models to predict caribou nutrition contained date as a covariate, we set our predictions to 15 Aug to facilitate comparisons among ecosites in response to disturbance and succession. We used Curve-expert to fit appropriate regression models (non-linear versus linear) for each predicted caribou nutrition variable with stand age on the x-axis and presented scatterplots of caribou nutrition by stand age for each forested ecosite. We presented simple means and standard errors of predicted caribou nutrition in marshes and the moss-dominated portions of sphagnum bogs (no stand age data were collected).

Developing foodscape maps

Our foodscape maps provided a high resolution (30 m x 30 m pixels), spatially-explicit depiction of nutritional resources in digital format that may have management applications for showing where areas of good forage and poor forage exist. We used the maps to describe nutritional value of habitats where the caribou occurred during summer before their capture and estimation of body condition in December. This strategy provided a basis to evaluate relationships between nutritional resources that caribou used with their body condition levels. We developed multiple foodscape maps, one for each forage and caribou nutrition metric that we considered potentially useful for predicting body condition of caribou in December. These metrics included biomass of accepted grasses, forbs, and deciduous shrubs (combined), biomass of accepted lichens, biomass of accepted horsetails, biomass of mushrooms, and our four main metrics of caribou nutrition (daily intake of digestible energy and digestible protein and concentrations of digestible energy and protein in caribou diets).

Developing foodscape maps involved two primary steps. The first involved assembling digital layers of landscape attributes that provided spatially explicit estimates of the covariates needed for our prediction models. To predict accepted GFS and accepted lichens, these covariates included stand age,

overstory canopy cover, percent of trees that were hardwood species, stand height, basal area, and ecosites.

Of these, predicting ecosites was most challenging and we undertook several analysis steps to improve the prediction accuracy based on comparison our field-based macroplots. First, we evaluated how well the Canadian Wetland Inventory Map (CWIM3; Mahdianpari et al. 2021) differentiated between lowlands and uplands. Because we observed a misclassification rate of lowlands of 46%, we used a combination of the CWIM3 and threshold values of normalized difference vegetation index (NDVI) to better differentiate between the two drainage classes. We resampled the CWIM3 map (10 m) to have the same resolution as the NWT Earth Observation for Sustainable Development (EOSD) map (30m), which was used to map uplands ecosites (see below). Using NDVI threshold values determined from receiver operating characteristic (ROC) curves (Fig. 3) increased the accuracy of identifying lowlands from 54% to 84% at macroplots, when compared to only using the CWIM3 classification of lowlands. NDVI values for each 30-m pixel were from 2019 Landsat 8 where each pixel was assigned the maximum NDVI value derived between 1 June – 30 September. Next, although CWIM3 mapped bogs, fens, marshes and other lowland types, we found that accuracy based on comparisons to our field macroplot data were poor (32% correct classification accuracy). Therefore, we used NDVI threshold values to differentiate between the lowland ecosites, which when compared to the macroplot ecosite classification, had an overall accuracy of 70% (Fig. 4).

For areas we identified as uplands, we used the NWT EOSD layer to separate upland black spruce from upland white spruce ecosites. We combined EOSD upland shrub and conifer types to represent Upland black spruce, and mixedwood and deciduous communities to represent upland white spruce. However, classification of upland white spruce was poor at 45% correct classification. We increased the accuracy to identify upland white spruce by 28 percentage points by incorporating thresholds based on a map of probability of white spruce (*Picea glauca*) (derived from the tree species probability map developed for the boreal extent of Canada by Hermosilla et al. 2022). We used a separate threshold value for the probability map of white spruce occurrence to differentiate between upland black spruce and upland white spruce in cases when CWIM3 with NDVI thresholds classified the pixel as an upland, but EOSD did not classify the pixel as shrub, conifer, mixedwood, or deciduous (e.g., classified as treed wetland). User's correct classification of upland black spruce and upland white spruce was 73% based on this approach.

We used Matasci et al. (2018; herein referred to as NRCAN layer) which provided estimates of basal area and overstory canopy cover that were well correlated to our field-derived estimates at macroplots (basal area: $r = 0.72$; canopy cover: $r = 0.75$) but relatively poorly correlated with stand height ($r = 0.28$) (Fig. 5). The spatial layer of canopy cover overestimated field-values by approximately 50%

when compared to our ground estimates of canopy cover (Fig. 5 A) and therefore, we applied a correction factor to the layer of canopy cover ($CC_{cor} = CC_{NRCAN} \times 0.57 + 5.2$, where CC_{cor} = corrected canopy cover (%) and CC_{NRCAN} = canopy cover from the NRCAN layer (%)).

A stand age layer for the year 2019 was developed by Maltman et al. (2023), which we updated for subsequent years (when caribou body condition data were collected (i.e., 2019 and 2021) to account for annual forest succession and forest fires (via the Government of NWT fire history layer: <https://www.geomatics.gov.nt.ca/en/fire-history>). To estimate proportion of trees that were hardwoods in upland ecosites, we used our field data for upland black and upland white spruce and stand age. For proportion hardwood for upland black spruce ecosites, we used a constant value of 3.27 for stands < 20 years and 30.4 for stands ≥ 20 years (mean values from macroplots in each of these stand age categories). At upland white spruce sites, we assigned a proportion hardwood value of 68.0 at stands < 20 years, whereas at stands ≥ 20 years, we used the probability of aspen (*Populus tremuloides*) layer developed by Hermosilla et al. (2022) (Fig. 6). We applied a correction factor to the probability of aspen (*Populus tremuloides*: $POTR$) occurrence to represent total percent hardwoods based on estimates derived at macroplots ($HRD_{cor} = POTR_{NRCAN} \times 1.57 + 13.4$, where HRD_{cor} = corrected hardwood proportion of trees and HRD_{NRCAN} = probability of aspen occurrence from the NRCAN layer).

With estimates of covariates derived for each pixel across our study area, we then used our forage prediction equations to predict abundance of accepted grasses, forbs, and deciduous shrubs (combined), lichens, horsetails, and mushrooms. From these, we predicted caribou nutrition (daily intake rate of digestible energy and protein) for each pixel and concentration of both in caribou diets using equations developed with captive caribou in northeastern British Columbia.

Caribou body condition

We developed histograms to display the range in body fat of the caribou we captured in 2019 and 2021 for this study along with histograms of December-caught females that had raised a calf that summer based on previous studies in December 2016 and 2017 (Cook et al. 2021a) to provide an initial indication of the consistency of body condition from 2016-2021. The 2019 and 2021 data were then compared to individual habitat use estimates as described below.

Evaluating body condition and habitat use

We considered 18 habitat and nutritional resource variables (collectively referred to as environmental covariates) to be potentially useful for defining relationships between the environment used by each caribou and its body fat level in December. These can be grouped into four categories: (1) those that may indirectly affect forage resources and included stand age, forest overstory cover, basal area, and tree height; (2) the ecosites used by caribou; (3) quantity of key accepted forage (biomass of accepted grasses, forbs, and deciduous shrubs (combined) and accepted lichens); and (4) predicted

caribou nutrition, which included quality of their diets (digestible energy and digestible protein content) and daily intake of digestible energy and digestible protein. Because data were collected in two separate years and thus response of body fat to environmental variables might vary by year, we also included year as a covariate in our analysis.

Caribou carried Telonics GPS collars that were programed to provide three locations per 24 hours. We plotted each GPS location of each caribou from 15 July to 15 September on our derived layers representing each environmental covariate. At each location, we assigned levels of each environmental covariate based on these layers. This process provided an estimate of each environmental covariate used by caribou at each caribou location point. We also calculated the distance of each relocation point for each caribou to the nearest open road (paved and gravel), to evaluate the potential effects of anthropogenic activity on caribou body fat.

Our approach provided two types of data for environmental covariates. For categorical covariates (e.g., ecosites), we calculated proportion of relocations of each caribou in each category. Thus, our variable for upland black spruce ecosites was proportion of relocations in that ecosite per time period. For continuous covariates (e.g., accepted lichens, daily intake of digestible energy, distance to roads), we simply calculated the mean across all relocations for the period. We considered three primary time periods for analysis, the entire summer period from 15 July to 15 September, early summer from 15 July to 14 August, and late summer from 15 August to 15 September. Sample size of relocation points for each caribou was about 180 relations from 15 July to 15 September (and about 90 in early and late summer each). From these time periods, we were able to evaluate which period was best related to body fat of caribou in December.

Our analysis of relationships between body fat and environmental covariates involved several steps. We first screened each covariate and body fat for non-linear relationships again using regression analysis via Curve-Expert with an AICc of >4 to identify non-linear versus linear relationships as described above. This analysis was conducted separately by year and by summer periods (all summer, early summer, late summer) to identify changes that might occur in relationships between year and summer period.

We then conducted a series of multiple regression analyses that evaluated contributions of combinations of environmental covariates to account for variation in caribou body fat. We first conducted multiple regression analysis on each group of environmental covariates primarily to identify which covariate in each group best accounted for body fat, and then conducted multiple regression using the best covariates in each group in a final analysis. For example, if stand age was the best of the forest overstory covariates, fens the best of the ecosites, accepted lichens of the forage biomass group, and daily intake of digestible energy from the predicted caribou nutrition group, then our final multiple regression analysis

would include each of these “best” covariates to provide a final model for predicting body fat levels. In addition, if our initial analysis indicated meaningful differences in relationships among years, then year would be formatted as a dummy variable and would be included as a potential covariate in all multiple regression analysis. For the summer periods, however, our approach was to identify which of the three summer periods best overall accounted for body fat in December and limit our multiple regression analysis to this time period (otherwise, the number of potential covariates and their interactions would greatly exceed what our sample size of caribou would support).

We again selected environmental covariates based on minimizing AIC using Proc Regression in SAS as a guide to select the best model in our multiple regression analysis. We used a variance inflation factor (VIF) of <10 for each covariate as an indicator to evaluate extent of multicollinearity (if included covariates resulted in $VIF > 10$, then one of the offending variables was removed, and the analysis was rerun). We limited the number of covariates included in the model to $1/10$ the sample size (McGarigal et al. 2000), i.e., up to five covariates per model. To avoid over-parameterized models, we excluded covariates if their P value was >0.15 . Finally, we generated QQ normality graphs and plotted model residuals to evaluate violations of normality and homoscedasticity for each model.

RESULTS

Vegetation Responses

Forage biomass.—We sampled vegetation in a total of 244 macroplots, of which 45 were sampled in bogs, 43 in fens, 84 in upland black spruce, 57 in upland white spruce, eight in marshes, six in the moss-dominated areas of Sphagnum bogs, and one in which ecosite could not be reliably determined. Stand ages varied from 1 to 250 years old in forest ecosites (stand age was not estimated for ecosites (marshes and sphagnum bogs) that did not support forests.

We observed prominent patterns of overstory development after stand replacing disturbance, of which rate of development and end-point levels varied markedly among ecosites (Fig. 7). Rate of development was more rapid in the upland black spruce and upland white spruce ecosites than wetland sites, requiring about 50 years to achieve peak overstory canopy cover. Overstory canopy cover averaged 50% to 60% in upland white spruce stands of 75-100 years old, and averaged 30% to 40% in upland black spruce stands at 50 to 75 years old and then declined slowly in older stands. In fens and bogs, 50 to 125 years were required to achieve peak overstory canopy cover with, again, some evidence of decline thereafter. Development of basal area exhibited similar patterns, although basal area continued to increase throughout the period of analysis in the upland white spruce type (Fig. 7). Percentage of the overstory composed of hardwood trees (mostly aspen) declined from 65% to 15% from stand ages of 1 to 200 years in the Upland white spruce ecosite. Otherwise, hardwoods were an important component

(>20% of trees) in forested stands only in very young stands less than about 25 years old in the other forest ecosites (Fig. 8). Jack pine also exhibited marked changes over succession in some ecosites. In the upland black spruce ecosite, it composed on average up to 65% to 70% of trees in stands up to 50 years old and declined thereafter. In other forested ecosites, Jack pine composed from 10% up to 30% in stands less than 50 years old and then declined subsequently thereafter (Fig. 8).

Changes in biomass of accepted forage also varied with disturbance and succession, particularly that of accepted grasses, forbs, and deciduous shrubs (combined) and accepted lichens. Biomass of accepted grasses, forbs, and deciduous shrubs exhibited prominent, usually non-linear, and variable patterns among ecosites in relation to disturbance and succession, with highest biomass levels occurring in early seral communities (Fig. 9). This pattern was most evident in the upland white spruce ecosite, where biomass averaged 600-800 kg/ha in stands 3 to 20 years old, 3 to 4 times greater than in stands of > 75 years old. In upland black spruce sites, the flush of early seral vegetation averaged 250 to 350 kg/ha in stands up to 20 years old, 2 to 3 times greater than in stands > 40- to 50 years old. The pattern for bogs was similar to that for upland black spruce, with 250 – 400 kg/ha in very young stands, declining to 100 kg/ha and less by stand age of 40 years, on average. In contrast, we observed a slow, linear decline in biomass of accepted grasses, forbs, and deciduous shrubs (combined) in fens. Through the first 100 years after disturbance, fens provided the highest biomass levels of this forage class, averaging from 300 to 400 kg/ha during this time and averaging roughly 200 kg/ha even in stands >150 years old (Fig. 9).

Biomass of accepted horsetails exhibited little or no consistent pattern with stand age (Fig. 9). Although the data were highly variable, we observed a trend of higher horsetail biomass in wetland than upland types, about 25 kg/ha versus about 2 to 5 kg/ha, respectively.

Biomass of mushrooms on average was quite low, generally averaging less than 5 kg/ha, and exhibited few patterns with stand age and ecosites. However, biomass of mushrooms showed a seasonal trend, with peaks in late August and early September (Fig. 10). In this late summer period, our data suggest the highest levels of mushrooms was in upland black spruce ecosites.

Perhaps the greatest variation in biomass among our 4 forage categories was for lichens, with biomass levels ranging from zero to just over 4,000 kg/ha (Fig. 11). Biomass of accepted arboreal lichens averaged less than 10 kg/ha in all nearly all ecosites and stand ages, and thus was typically a small component of accepted lichen biomass in any ecosite. In general, accepted lichen biomass averaged <50 kg/ha in fens and upland white spruce ecosites but was often abundant in bogs and upland black spruce ecosites >40 years old. In bogs, our data suggested a peak in lichen biomass in stands of about 125 years old, with a modest decline in older stands. In upland black spruce types, our data suggested a gradual increase in lichen biomass that extended into stands >150 years old. No matter the ecosite, substantial

increases in accepted lichen biomass occurred only in stands >40 years old, best seen by evaluating the data points rather than the fitted curves (Fig. 11).

Predicting forage biomass.-- We developed equations to predict biomass of forage for each of our four forage classes by ecosite to reflect effects of disturbance and succession and ecosite on forage biomass (a total of 24 individual regressions) (details of modeling results are presented in the CIMP companion report (Cook et al. [2023])). Our QQ plots to evaluate assumptions of normality suggested for nearly all analysis minimal violations of the assumption of normality, and residual plots indicated only modest violations of homoscedasticity. We considered neither to be overly serious concerns and opted to use standard regression analyses to develop the prediction equations.

Of our four forage classes, biomass of accepted grasses, forbs, and deciduous shrubs (combined) exhibited strong response (declines) with stand age, but variably so among ecosites (Appendix B). The strongest relationships were in upland forests ($R^2 = 0.57$ in both upland black spruce and upland white spruce), then bogs ($R^2 = 0.37$) and fens ($R^2 = 0.08$) (we had no metrics reflecting disturbance and succession in marshes and sphagnum bogs). In each of these forested ecosites, transformed stand age (reflecting the highly non-linear relation of stand age with forage biomass) was the primary covariate included in the analysis, but percent hardwood composition (a positive, linear effect) and non-linear formulation of overstory canopy cover for upland white spruce ecosites were also included in the models.

In general, biomass of accepted horsetails was invariant to our predictor covariates (Appendix B) except in fens for which our model included two covariates with positive, linear effects: stand age and overstory canopy cover ($R^2 = 0.19$). For all other ecosites, constants by ecotype provided the best estimates of abundance.

No mushrooms were detected in our macroplots in marsh and peat plateau ecosites. In the other ecosites, a non-linear formulation of date was the primary covariate in all equations, reflecting the marked increase in mushrooms by September. In bogs, we also included overstory canopy cover and the canopy cover x stand height interaction which together accounted for 50% of the variation in mushroom biomass. In upland black spruce, a non-linear formulation of stand age and the stand height x percent hardwood interaction was included with date that together accounted for 37% of the variation in mushroom biomass. For fens and upland white spruce ecosites, non-linear formulations of date were the only covariates included in our final models, both of which explained 20% of the variation in mushroom biomass (Appendix B).

Biomass of accepted lichens varied among ecosites, with very low levels in the upland white spruce, fens, and marshes (and to some extent, sphagnum bogs) ecosites. For these ecosites, we used mean lichen biomass as the best predicted value of lichen biomass (Appendix B). The development of prediction equations for bogs and upland black spruce ecosites was complex. For bogs, a single data

point of 4,400 kg/ha of lichen biomass exceeded the overall lichen biomass mean for this ecosite by ten-fold, and we excluded this data point from analysis. In addition, two alternative patterns emerged based on scatter plots of lichen biomass with our measures of forest development: a gradually increasing linear relation between biomass and stand age versus a humped-shaped pattern with low levels of lichen biomass in young stands, a peak in biomass as stands developed, and a decline in biomass as overstory development stabilized (compare Fig. 11 with Appendix C, Fig. C1 and C2). For our primary forest overstory metrics (canopy cover, basal area, stand height, and tree density), AICc for non-linear equations to predict accepted lichen biomass was >4 units lower than for linear versions for each of these metrics in both bogs and upland black spruce ecosites, strongly suggesting that the humped-shaped pattern exhibited with forest overstory metrics was biologically more realistic than the linear pattern (Appendix C, Fig. C1 and C2). For bogs, our best final prediction model included non-linear versions of basal area and tree density and explained 47% of the variation in lichen biomass. An alternative model using available site characteristic data was a much simpler non-linear model with only overstory canopy cover that explained 23% of the variation in lichen biomass (Appendix B). For development of our foodscape maps (described below), we used the equation based on basal area (Appendix B) for predicting lichen abundance in bog ecosites. For upland black spruce ecosites, our final model included stand age, a non-linear version of stand height, and the interaction of stand height and overstory canopy cover and explained 30% of the variation in lichen biomass (Appendix B).

Caribou Nutrition Responses

The equations to predict caribou nutrition were derived from foraging trials conducted with captive caribou in northeastern British Columbia from 2013-2016 and used to predict digestible energy and digestible protein content of caribou diets, per minute and daily intake rate of forage digestible energy and digestible protein (a total of seven nutrition metrics). The covariates in the equations were based primarily on biomass levels of each of our four forage groups (accepted grasses, forbs, and deciduous shrubs combined; accepted horsetails, accepted lichens, and mushrooms), date, and, in a few cases, included covariates representing overstory canopy cover (Appendix D).

The prediction equations of caribou nutrition provided reasonably good fit for most metrics. For example, adjusted coefficients of multiple determination (R^2) ranged from 0.60 to 0.76 and from 0.55 to 0.66 for nutrition metrics in uplands and wetlands, respectively, with two exceptions, 0.41 and 0.46 for dietary digestible energy in uplands and wetlands (Appendix D). Accepted grasses, forbs, and deciduous shrubs (combined) were included as a significant influence in all equations (except dietary DE in uplands). The relationship in all cases was positive, i.e., nutrition improved as abundance of this forage class increased. Accepted lichens were included as a positive effect in many dietary DE equations but were not included, as expected, in any of the dietary digestible protein equations. Biomass of horsetails

was a consistently included as a covariate positively related to caribou nutrition in upland equations but was inconsistently included in the wetland equations. Although extremely nutritious, mushrooms were excluded from most equations, largely owing to their rarity in the caribou enclosures in British Columbia where the equations were developed. Finally, date was included in almost all equations as a negative, linear effect on caribou nutrition, and overstory attributes were included as a main and interactive effect in only the prediction equation for dietary digestible energy in wetlands (Appendix D).

Displaying predictions of four caribou nutrition metrics with stand age by ecosite in combination with estimates of caribou requirements provided a basis to evaluate how well forage resources satisfied nutritional needs in relation to forest succession and biogeoclimate (indexed by ecosite). Because most of our equations to predict caribou nutrition contained date as a covariate, we set date for predictions to 15 Aug, the mid-point of our analysis period, to facilitate comparisons in response to disturbance and succession. Our data indicated higher content of digestible energy and protein in caribou diets in young stands, particularly in wetland ecosites (Fig. 12). Higher levels of dietary digestible protein were evident in early seral stands in lowland, but this pattern was weaker in uplands. The uptick in dietary quality in young stands was primarily due to the positive effect of accepted grasses, forbs, and deciduous shrubs (combined) as a covariate in the prediction equations—the equation to predict digestible energy in upland forests, where we showed no uptick in dietary digestible energy, did not contain accepted grasses, forbs, and deciduous shrubs (combined) as a covariate. Our predictions indicate deficiencies of digestible energy in caribou diets in bogs, except in young stands, slight deficiencies in fens and no deficiencies in upland forests. For dietary digestible protein, we found greater probability of deficiencies in mid- and late seral stages in wetland forests and in all stages in upland forests (Fig. 12).

For our analysis, predicted intake of nutrients may be a more relevant metric of caribou nutrition than concentration of nutrients in caribou diets, because the former accounts for both how fast the caribou can eat and the nutrient content of their food. Predicted daily intake rates of dietary digestible energy and protein again indicated a prominent influence of stand age (Fig. 13), with higher intake rates in stands less than about 25 years old (depending on ecosite). The exception to this was for daily intake of digestible energy in upland black spruce and lowland bog communities. Using a prediction equation with lichen as a covariate, our data indicated elevated daily intake of digestible energy in mid-seral communities, but using a prediction equation based on accepted grasses, forbs, and shrubs, we see a more typical response (high in young stands, declining thereafter) for this variable (Fig. 13). (This points to the importance of fully understanding how lichens contribute to caribou diets in summer.) In contrast to our findings of digestible energy and digestible protein content in caribou diets, our predictions for intake rates suggest roughly similar deficiencies of daily intake of digestible energy and digestible protein relative to requirements for most all ecosites, where the ability of caribou to eat fast enough to satisfy daily

requirements is greatest in early seral communities and substantially lower in stands greater than about 25 years old. Again, inferences pertaining to daily intake of digestible energy in upland ecosites depend on if equations include lichen covariates or covariates of accepted grasses, forbs, and deciduous shrubs (combined) (Fig. 13).

In marshes and sphagnum bogs where stand ages were unknown, our data indicate substantially depressed levels of daily intake of digestible energy and protein and digestible protein content of diets far below required levels. Only DE content of diets approached, on average, required DE levels (Fig. 14).

Foodscape maps

Predicting ecosites, a fundamentally key aspect of constructing foodscape maps, was based primarily on Canadian Wetland Inventory Map (CWIM3) and the NWT's Earth Observation Sustainable Development map (EOSD) with adjustments incorporating NDVI thresholds that provided an overall accuracy of 65% (Table 2). Accuracy was lowest for bogs (41%) and greatest for sphagnum bogs (83%) and intermediate for all other types (57% to 71%) (Table 2). Final correlation coefficients for overstory canopy cover, basal area, stand height, and stand age were 0.75, 0.72, 0.27, 0.57, respectively. In Fig. 15, we present example foodscape maps for our study area for biomass of accepted grasses, forbs, and deciduous shrubs (combined), accepted lichens. In Fig. 16, we present a map of predicted daily digestible energy intake based on the equation that uses accepted lichen biomass as a predictor variable and a map that presents daily intake of digestible energy predicted based on the equation that excludes lichen biomass as a predictor variable. These maps depict a substantially different range of daily intake of digestible energy between the two nutrition mapping equations, with the equation excluding accepted lichen biomass portraying a bleaker nutritional landscape than the map constructed using the equation that includes lichen biomass (Fig. 16).

Caribou Body Fat

We captured 29 adult female caribou in December 2019 and 25 in December 2021 in the South Slave region of the SNWT, each of which had raised a calf that year and for which we measured body fat. For comparison, we also present body fat data for 18 and 13 adult caribou captured in December 2016 and December 2017, all of which also had raised a calf just before they were captured in December (but were not wearing a GPS collar at time of capture to provide location data from the preceding months). Body fat ranged from 5.4% to 14.9% ($\bar{x}$ = 9.4%) in 2019 and 4.7% and 13.3% ($\bar{x}$ = 8.9%) body fat in 2021. The range in our sample was more restricted in the earlier years, ranging from 4.7% to 11.7% ($\bar{x}$ = 8.5%) in 2016 and 6.0% to 9.8% ($\bar{x}$ = 7.9%) in 2017 (Fig. 17), albeit yearly sample size of females that raised a calf was about half that in the early years compared to 2019 and 2021, likely contributing to the more restricted range in body fat. Body fat of all adult female caribou (whether they raised a calf or not) in late winter was 10.0 (*S.E.* = 0.62; *n* = 27), 10.4 (*S.E.* = 0.83; *n* = 15), 9.0 (*S.E.* = 0.30; *n* = 44), 8.8 (*S.E.*

= 0.37; $n = 26$), 8.2 ($S.E. = 0.41$; $n = 26$), and 9.1 ($S.E. = 0.65$; $n = 17$) for 2015, 2016, 2017, 2018, 2019, and 2023, respectively. These values suggest relatively consistent results over the years, especially because the late winter estimates include caribou that raised a calf and those that did not (i.e., variation in the proportion of those that did and those that did not would contribute to variation among years).

Environmental Influences on Caribou Body Fat

Body fat of two anomalous caribou, both captured in 2021, were excluded from analysis. The first had the lowest body fat of all caribou sampled and was estimated to be the oldest (over 12 years old). The second was captured on Big Island and had a home range that was 10 times larger than the mean for all other caribou; her body fat levels were also among the lowest of any caribou captured. Thus, sample size for analysis was 29 caribou in 2019 and 23 in 2021. Locations of caribou plotted with foodscape maps are presented in Appendix E.

Our total pool of environmental covariates for analysis was 18 variables representing ecosites, forest overstory characteristics, forage abundance, and predicted caribou nutrition, plus distance to roads as a measure of caribou proximity to human activities (Table 3). Our initial analysis using simple regression was intended to identify non-linear relationships, differences among years, and provide insights regarding if relationships with body fat were better based on caribou use of habitat and nutritional resources over the entire summer, early summer (15 July to 14 August), or late summer (15 August to 15 September).

Several general patterns emerged from these regression analyses (Table 3). First, correlations between covariates were modest, with all coefficients of determination (r^2) values < 0.20 . Second, we found no evidence of non-linear relationships between body fat and environmental covariates for any covariate during either year or either summer period. Third, we found a marked pattern of opposite relationships between the two years for virtually each covariate based on the slope coefficient for each covariate, with the exception of distance from roads (Table 3). Thus, for example, if our data indicated a negative relation between body fat and biomass of accepted lichens at caribou relocation points in the first year, then the relationship was positive the second year. This opposite pattern held for nearly every covariate, whether the relationships were statistically significant or not, suggesting that factors that affected caribou body fat were substantially different between the two years. In addition, the covariates that were statistically significant ($P < 0.10$) usually differed between the two years (Table 3). Third, the relationships (sign of the slope coefficient) between body fat and each covariate were the same based on analysis conducted for the entire summer and for late summer (and thus, by default, for early summer). However, the coefficients of determination were usually higher (and never lower) for relationships between body fat and the covariates in late summer than across all of summer (Table 3), suggesting that habitat and nutritional resources used in late summer were modestly better correlated to body fat in

December than habitat and nutritional resources used in early summer or across all of summer. Thus, the focus of our subsequent analyses emphasized relationships between body fat and habitat and nutritional resource use from 15 August to 15 September.

Body fat in 2019 was unrelated to any of our ecosite variables (i.e., percent of relocations in each ecosite), including lowlands (and by default uplands), fens, bogs, upland black spruce and upland white spruce ecosites. However, in 2021, body fat was significantly, negatively related to use of bogs but positively related to use of upland black spruce ecosites (Table 3, Fig. 18). Combining the ecosites into uplands and lowlands suggested little or no influence of either (Table 3). Caribou habitat use of forest overstory covariates was unrelated to December body fat with one exception. In 2019, body fat was negatively related to stand age and was the most correlated of all the covariates with body fat in the 2019 data set (Table 3, Fig. 19).

We observed mixed body fat responses in relation to forage abundance. The most prominent pattern was a negative relation between body fat and quantity of accepted lichens in 2019 but a positive relation with quantity of accepted lichens in 2021 (Table 3, Fig. 20). Although not significant, the sign of the slope coefficients suggested that caribou body fat trended higher in areas with higher accepted grasses, forbs, and deciduous shrubs (combined) in 2019 but trended lower in areas with lower accepted grasses, forbs, and deciduous shrubs in 2021 (Table 3).

Among our predicted caribou nutrition metrics, we observed a significant positive relationship between body fat and daily intake of digestible energy in 2021 and a trend of a negative relationship in 2019 when this nutrition metric was calculated using our prediction equation that accounted for accepted lichen's effect on digestible energy intake rate (Table 3, Fig. 21 A, B). With daily intake of digestible energy calculated using the equation that lacked lichen's effect, we found the opposite pattern, i.e., positive relationship between body fat and daily intake of digestible energy in 2019 and a trend of a negative relationship with body in 2021 (Table 3, Fig. 21 C, D). (Note, the digestible energy intake estimates with the equation that includes the lichen covariate will indicate that intake is highest in those ecosites and stand ages where lichens are abundant; the digestible energy intake estimates with the equation without the lichen covariate will estimate intake to be highest in those ecosites and stand ages where accepted grasses, forbs, and deciduous shrubs (combined) are most abundant, primarily in early seral forested communities.) Body fat was significantly, positively correlated to predicted digestible energy content of diets in 2021 when calculated using the prediction equation that accounted for lichen's effect, but body fat trended negatively to predicted digestible energy content of diets using this equation in 2019 (Table 3, Fig. 21 E, F). Body fat was unrelated to digestible energy content of diets when predicted digestible energy was calculated using the equation that lacked lichen's effect, and we found no significant relationship between body fat and any of our digestible protein metrics (Table 3).

Finally, body fat was similarly, inversely related ($r^2 = 0.13$ to 0.16) to distance from roads in each summer period and in both years. Mean distance of locations of each caribou from roads ranged from 2.7 to 65 km in late summer (virtually identical to that across all of summer) with means of 16.8 km and 25.9 km in 2019 and 2021, respectively.

In general, the nature of results from the simple regression analyses foreshadowed issues that would and did arise in a more comprehensive multiple regression analysis. The opposite effects between years of covariates on body fat required that multiple regressions include a dummy variable to reflect year effects, and that its interaction with each covariate in regressions on body fat be included in the analysis. In addition, multiple regressions typically were unenlightening that included those covariates that were important (significant) in one year and not the other. For example, a regression that included stand age, the dummy variable for year, and their interaction term resulted in an overall regression equation far from statistically significant and an R^2 that was roughly the average of the r^2 values for stand age in each simple regression for the two years (i.e., 0.17 in 2019 and 0.04 in 2021, Table 4, resulting in an overall R^2 of 0.10 and a P -value of 0.30). This pattern held for every covariate that was significant in one year but not the other (i.e., six covariates, with only biomass of accepted lichens being significant in both years, Table 4). The other effect was potentially more important but more complicated. Because dummy variables take on values of 0 and 1 (as required for regression analysis), any interaction term that included the dummy variable for year also had to include main effects of all terms included in the interaction term. This had three practical concerns for our multiple regression analysis (using again for an example, body fat = year, stand age, and the year x stand age interaction). (1) Without the main effect of stand age included, the regression equation would, in essence, only model for the data where year is coded as one, because the interaction term would = zero for data where the interaction term was set to zero. In practice, we found that the P -level and coefficient associated with the main effect typically was highly insignificant with a very small coefficient, which provided a poor model with little value. (2) In the interest of including multiple covariates in the same regression run, because each covariate had to be brought in with its interaction term (i.e., with the year dummy variable) and its main effect, the number of terms in the regression model that included multiple covariates was excessive, and multi-collinearity (variance inflation factor, VIF) was far too high for reliable analysis based on our sample size of about 55 caribou. (3) In light of the strong year effects and potentially a strong stand age effect, the evaluation of ecosite's effect, in particular on caribou body fat, would theoretically require a three-way interaction term (e.g., year x stand age x ecosite, one three-way interaction term for each ecosite). To correctly include such three-way interactions further increased the number of required terms and multicollinearity in the multiple regression runs to fully evaluate factors that affect body fat of caribou.

Therefore, we considered the simple regression analysis that we conducted to provide the most useful insights on the influences of our environmental covariates on caribou body fat levels. Two themes, one each year, seem evident. In 2019, body fat was typically higher in caribou that used areas in relatively young stands with little regard to ecosites, where accepted lichens were less abundant, and predicted daily intake of digestible energy was higher based on the non-lichen daily digestible energy intake (which predicts highest daily intake to occur in early seral communities), and although not significant, in areas with providing higher daily intake of digestible protein and higher concentrations of dietary digestible protein, lower levels of dietary energy based on the lichen equation. These all point to positive linkages between body fat in early seral forest communities that provide abundant accepted grasses, forbs, and deciduous shrubs (combined) and, in turn, suggest that older stands that provide abundant accepted lichens are negative linked to caribou body fat, or at least have little positive influence on body fat. The opposite seems to be true for 2021—body fat was positively linked to areas where accepted lichens were more abundant, predicted daily digestible energy intake rate based on the lichen prediction equation and concentration of digestible energy in caribou diets also based on the lichen prediction equation were greater, especially in upland black spruce ecosites, and although not significant, was negatively linked to younger stands, areas with higher accepted grasses, forbs, and deciduous shrubs (combined) and concentration of digestible protein in caribou diets averaged lower, in turn suggesting that older stands in upland forests with relatively abundant lichens best supported caribou body fat.

DISCUSSION

Relating the spatial distribution of nutrition resources to individual animal performance (i.e. pregnancy, survival, body condition among others) of ungulate herbivores is a relatively recent strategy in the realm of nutritional ecology for which there are few examples. This is especially true for studies which strive to relate performance of individual animals to the resources that these animals use based on satellite telemetry. At the core of such an effort is the need to measure and map nutritional resources in ways that have relevance to animal nutrition, meaning that animals perceive and functionally respond to the metrics that are intended to represent nutritional resources (Searle et al. 2007). This is a challenge. Decades of livestock research has demonstrated that animal performance is fundamentally driven by the nutrition that animals acquire particularly when high nutritional requirements are high such as during late gestation, lactation, and periods of rapid juvenile growth and development. Although not as well studied, sufficient studies clearly illustrated that the same is true for wild ungulates—animal performance is fundamentally a function of the nutrition that animals can acquire (Verme and Ozoga 1980; Crête et al. 1993; Cook et al. 1996, 2004; Tollefson et al. 2010). For free-ranging wild ungulates that exist across broad landscapes of spatially and temporally dynamic resources, however, measuring their actual

nutrition is virtually impossible, and so ecologists are faced with using a variety of measures that are assumed or hypothesized to represent nutritional resources in ways that in fact relate to the nutrition that herbivores can obtain.

We used a variety of metrics that we hypothesized to either affect forage quality and quantity indirectly, are direct measures of forage abundance, and others that in fact are known to strongly influence animal performance. The former (indirect habitat metrics) may be less useful overall but estimates of which are relatively easy to acquire from various remotely sensed sources. Forage composition and abundance metrics are difficult to measure and must be predicted at high resolution across vast landscapes, a process that is complicated and substantially prone to error. The latter (caribou nutrition) should be the superior metrics, but these were based on captive caribou nutritional trials conducted in northeastern British Columbia and were extrapolated to our study area via a complicated prediction process. Thus, to deal with the uncertainty associated with these metrics, our approach with the 18 metrics was to identify consistent and common themes among the covariates that together would identify habitat and nutritional resource factors that accounted for variation in caribou body fat in December.

Our evaluation of these covariates indicated patterns that account for body fat of caribou, but the patterns were markedly different between the two years of our study, greatly complicating analyses and inferences. In 2019, body fat was most related to three statistically significant covariates at caribou relocation points: (1) inversely related to stand age; (2) positively related to predicted daily intake of digestible energy, where digestible energy intake was calculated using the prediction equation that excluded lichen biomass and thus daily intake of digestible energy was a function of accepted forage biomass of vascular plants (primarily grasses, forbs, and deciduous shrubs); and (3) inversely related to biomass of accepted terrestrial and arboreal hair lichens. These results suggest that body fat responded to younger stands which typically provided relatively high abundance of accepted forage from vascular plants and are less likely to contain abundant biomass of accepted lichens. The trend (non-significant) of body fat to be positively related to use of areas with higher digestible protein also supports this contention, because lichens contain very little or no digestible protein (see CIMP companion report Cook et al. 2023). That is, digestible protein is likely to be higher the less lichens are consumed.

In 2021, findings were the opposite. Body fat of caribou was significantly correlated to five covariates at their relocation points: (1) positively related to their use of upland black spruce communities but negatively related to their use of bogs; (2) positively related to the biomass levels of accepted lichens at the relocation points; and (3) positively related to predicted daily intake of digestible energy and predicted concentration of digestible energy when these were predicted from equations that included accepted lichen biomass as predictor covariates. These results suggest that body fat responded to accepted lichens and the vegetation types where these were most common, i.e., in upland black spruce

and in stands sufficiently old for lichens to have recovered from the last wildfire. The non-significant trend of lower body fat in areas used by caribou that provided higher levels of digestible energy using the equation that excludes lichens for prediction is consistent with this finding (again because digestible protein is lower in lichen-dominated diets). The inverse relationship between body fat and use of bogs perhaps may be a contradiction because bogs in some cases contain abundant lichens (primarily in drier extents of this ecosite) (and where flooding was an issue, bogs may have been inundated for significant portions of the summer).

The opposite findings of contributors to body fat levels between the two years substantially confounds one of the objectives of the study, to evaluate which of the two forage classes (accepted lichens or accepted vascular plants) better accounts for body fat levels, because their abundance varies substantially among habitats and seral stages. The implications of the opposite findings must consider the prevailing weather patterns of the two years of assessment. The first year was notably dry, averaging about 60% of normal in spring and summer combined (and a moderately drier winter) (Fig. 2). But spring and summer in 2021 were unusually wet, averaging about 45% above normal and nearly double that in 2021. Our (A. Kelly) anecdotal observations from the ground and during aerial wildlife surveys were that the high water table entering the 2021 season as well as the greater precipitation in spring and summer greatly affected the landscape, with flooding of water bodies widespread and many areas that were normally dry in summer inundated with water, including sites that we classified as uplands and normally dry during most summers. We can only speculate on how this might affect caribou in summer, but we hypothesize that many low-lying areas that might compose suitable habitat in late spring and summer in most years was avoided by caribou, perhaps causing a shift to drier areas. Caribou may have been forced to use nutritional resources differently than they would in a more typical year. The substantially greater moisture in 2021 in combination with modestly warmer summer temperatures may have favored abundance of biting insects, which may in turn have affected habitat use and local distribution of caribou in parts of summer. In addition, these differing weather patterns likely affected nutritional resources in a variety of ways that would be difficult to predict. In this regard, we especially wonder about the abundance of mushrooms, the single-most nutritious forage source available to caribou on this landscape (see CIMP companion report Cook et al. [2023]). If greater precipitation increased mushroom abundance in habitats that remain above water in 2021, this seemingly subtle difference might significantly change the nutritional landscape, and our caribou nutrition equations would not reflect this. The relatively strong correlation between body fat and caribou use of upland black spruce communities could be due to elevated mushroom abundance in this type, which typically occurs on coarse-textured soils in upland areas that on average may be least likely to be inundated in very wet years and, in which, abundance of mushrooms in late summer evidently was greatest (Fig. 10).

The value of lichens in winter is well-recognized largely because they contain among the highest levels of digestible energy of any common forage on northern caribou winter ranges. But as a summer forage, its value to lactating caribou and their calves is less certain largely due to four aspects: 1) its digestible protein levels are the lowest of any forage type; 2) its digestible energy content is not necessarily higher than that of many vascular plants particularly in late spring through mid-summer; 3) it typically provides very low bite mass (weight of individual bites by plant species), the latter of which can reduce per minute and daily forage intake rate if lichens compose a large portion of caribou diets, and 4) often handling time associated with separating lichen bites from pine needles, moss, and detritus can be substantial (based on observations of foraging by captive caribou). The first two attributes are reasonably well recognized (see CIMP companion report Cook et al. [2023]) but understanding of the latter two has required detailed studies with researchers closely observing captive caribou foraging and measuring the weight of their bites in native habitats (e.g., Denryter et al. 2017, 2021). Bite mass is a primary determinant of how rapidly herbivores can consume food, and this forage attribute can influence daily foraging dynamics sufficiently to scale up and influence higher order processes (i.e., body condition, reproduction, survival) (Spalinger and Hobbs 1992, Shipley 2007, Cook et al. 2016). Lichens provide among the lowest bite mass of any plant life form group, ranging as low as 1/100 of that of many vascular plants (especially deciduous shrubs) (Fig. 22). Our accumulated data sets from foraging trials with captive caribou in Ontario and the mountains and boreal forests of British Columbia together show that biomass of accepted lichens is not correlated, and can be inversely correlated, with forage intake rates (Fig. 23). However, bite mass of lichens can vary, and under certain conditions, bite mass of lichens may approach that of vascular plants. These conditions include dense mats growing on sites where moss, sticks, sand and gravel, pine needles, and other detritus is largely absent from lichen mats. Under conditions where lichens are dense and “clean”, our observations are that bite mass and intake rate of lichens average higher than that in stands where pine needles, in particular, accumulate and lichen mats are sparse (such as under relatively dense canopies of Jack pine). Whatever the case, our collective data sets from captive caribou foraging trials show the considerably greater influence of accepted vascular plants on forage intake rates (Fig. 23).

Because biomass of accepted lichen is low and accepted vascular plants averages higher in early seral communities the first ~25 years after wildfire, we originally hypothesized that nutrition of caribou and body fat would be higher for caribou that used relatively young forests, despite the reduced levels of accepted lichens in young stands. This pattern was not apparent in the wetter year of 2021, but was in 2019 during the drier year, substantially complicating our ability to derive simple, definitive conclusions about which habitats and food resources typically are best for caribou in our study area. However, our findings for the drier year of 2019 of the nutritional value of young forests is substantially supported by

detailed habitat selection analysis reported by DeMars et al. (2020) for caribou across the Northwest Territories. In general, caribou exhibited selection, often strong selection, for burned areas during late spring into autumn and a propensity to avoid burned areas in winter, especially late winter. Selection for burned areas varied by stand age classes (i.e., 1-10, 11-20, 21-30, 31-40, and 41-60) which may reflect differences after wildfire in slash buildup from fallen dead trees, density of tree regeneration (which can be very dense and may discourage use by caribou), and perhaps other factors in addition to changes in forage quality and quantity. Whatever the case, however, across most of these stand ages (up to stand age of about 40 years), our data indicated that accepted lichens were strongly reduced (Fig. 11), suggesting that the selection that caribou exhibited for these younger stands was *not* due to their desire to forage in areas with abundant lichen. Our data indicated markedly increased abundance of accepted grasses, forbs, and deciduous shrubs (combined) in young stands that can translate to improved caribou nutrition, but persistence of these varied by ecosite. For example, persistence of elevated forage from accepted vascular plants persisted ≤ 10 years in bogs, ≤ 20 years in upland black spruce, at least 50 years in fens, and at least 50 years in upland white spruce ecosites (Fig. 9). For upland white spruce ecosites, however, our observations that conifer and hardwood regeneration established within about five years after fire, also creating a very dense vegetation sward that caribou may be reluctant to enter. This also can occur in the more productive upland black spruce ecosites, where dense, dog-hair stands of Jack pine may rapidly establish (see CIMP companion report by Cook et al. [2023]). Thus, DeMars' et al. (2020) findings about caribou habitat selection combined with our data of abundance of accepted vascular plants support the hypothesis that early seral communities occurring after wildfire are sought after and nutritionally important for caribou in late spring into autumn. Nevertheless, our results for 2021 suggest caution—strongly varying weather conditions may alter the spatial distribution of the better foraging conditions and caribou response to these conditions.

Whatever the case, our findings support our initial prediction that caribou body fat in autumn was related, one way or another, to the nutritional value (i.e., daily intake of digestible energy) of areas in which they foraged for both 2019 and 2021. However, we acknowledge that amount of variation in body fat accounted for by our metrics of habitats and nutritional resources was low (with all r^2 values less than 0.20). To some degree, this is not surprising, and we attribute much of the low correlations to only modest accuracy of readily available habitat maps available for our study region. In our initial steps to developing our foodscape maps, we found that the wetland layer (CWIM3) and the EOSD layer that we used poorly to moderately matched the habitats identified our field macroplots, especially for separating ecosites within wetland and upland categories. Seemingly, our use of NDVI to improve our estimation of ecosites above that provided by the original CWIM3 and EOSD layers improved the prediction of ecosites compared to the original layers, but an additional round of field sampling (which was beyond the

scope of this study) to provide a fully independent set of field plots to evaluate the accuracy of our final ecosite predictions is needed to reliably identify our final mapping accuracy. Thus, the overall accuracy (65%) described in Table 2 is probably overestimated to some extent, and we suspect that actual overall accuracy levels may have been no better than about 50%. Because our prediction of accepted forage biomass and caribou nutrition levels were derived in part from our ecosite map, accuracy of about 50% represents a substantial source of error that undoubtedly impacted our ability to identify relationships between habitat and nutritional resource metrics and contribute to the relatively low r^2 values that we report (Table 3).

An additional source of uncertainty and concern involves the prediction of biomass of accepted lichens and the prediction of caribou nutritional responses to the abundance of this forage. First, no maps existed at the time of this study that provided lichen estimates for all habitats of our study area that were useful for our purposes. What makes lichen prediction difficult in our situation is that soil characteristics (soil texture, soil moisture in particular) predominantly determine where lichens will grow (Cichowski et al. 2022), and detailed information on these soil characteristics at a high resolution for our study area was unavailable. Our ecosite classification helped in this regard; biomass of accepted lichens was too low to be an important forage resource in marshes, fens, sphagnum bogs, and upland white spruce ecosites. Lichens of significant abundance were present on upland black spruce and bogs, but only in some of them. Our observation was that in bogs, lichens were abundant only in the drier extents and absent in wetter extents of this ecosite. In upland black spruce, lichens seemed abundant in the less productive sites on coarse textured soils and generally sparse on more productive sites (Fig. 11). Thus, for bogs and upland black spruce ecosites, many sampled sites had abundant lichens whereas others did not (<100 kg/ha). Stand development after disturbance played a role to some extent; virtually no young stands up to 40-50 years old in both ecosites contained abundant lichens. For locations where our ecosite mapping was accurate, our ability to identify areas where lichens were sparse should have been reasonably accurate (i.e., marshes, bogs, fens, and upland black spruce, and young stands less than 40 years old), but otherwise, predicting where accepted lichens were relatively abundant was challenging. We also emphasize that our data suggests a decline in lichen abundance in old stands depending on overstory conditions (Appendix C). Results from lichen studies in other regions support this finding (Coxson 2015, Sulyma and Coxson 2001, Coxson and Marsh 2001), although the extent that this pattern holds evidently reflects conditions at growing sites (Cichowski et al. 2022). Our data is suggestive that lichen persistence in old stands reflects extent of overstory development, i.e., that and may reflect observations that relatively dense overstory canopies both hinder development of lichen and the decline in lichens in older stands (Sulyma and Coxson 2001, Coxson and Marsh 2001).

Second, the prediction of daily intake of digestible energy and dietary digestible energy levels depended on how lichen biomass was treated in the prediction equations. In the overall analysis of the relationship between digestible energy and lichen biomass from British Columbia and Ontario using the captive caribou data, biomass of accepted lichens bore no significant relationship with daily intake of digestible energy. This also was true in multiple regression analysis for the Ontario data, where other forage variables, primarily biomass of accepted grasses, forbs, and deciduous shrubs (combined) and horsetails were important, and accepted lichen biomass was not. However, in British Columbia, our multiple regression analysis did include lichen biomass as a significant, positive predictor variable of daily intake of digestible energy. This inconsistency led us to develop two daily digestible energy prediction equations, one that included lichens and another that did not. Our intent was to use both for our evaluations for the southern Northwest Territories, anticipating that separate analysis would help to clarify which of the two equations best related to body fat levels. The results from 2019 indicated that the equation lacking lichens as a predictor variable was better but results from 2021 indicated the opposite: including lichens to predict daily intake of digestible energy was better. The potential implications of this are important. First, these two equations resulted in substantially different depictions of daily intake on the landscape of our study area, with one suggesting considerably more areas providing high predicted daily intake of digestible energy (based on the equation with lichen biomass), and the other indicating substantially lower daily intake of digestible energy (based on the equation without lichen biomass) (Fig. 16). Secondly, the variation in lichen's influence on how fast caribou eat contributes to uncertainty about which ecosites and seral stages are important for caribou in summer.

The concern about extrapolation errors of caribou nutrition predicted from British Columbia may be most relevant in the context of predicting lichen effects. Nearly all sites in British Columbia used to develop the daily intake prediction equations supported less than 200 kg/ha of accepted lichens (Fig. 23), but many sites we sampled in the Northwest Territories supported accepted lichen levels well in excess of this level, from 200 to 2,000 kg/ha (with one supporting 4,400 kg/ha) (Fig. 11). Thus, the lichen effect on caribou nutrition was developed under substantially different circumstances than in the Northwest Territories where the equations were used. There is no way to check the accuracy of predictions of caribou nutrition in the Northwest Territories, but we assume that they at least provide a *relative* ranking of the nutritional value that has value for understanding effects of nutritional resources on body fat levels of caribou, and it may be unrealistic for these to predict with high accuracy the *absolute* levels of nutrition on our study area. This distinction should be considered in comparing predicted daily digestible energy intake maps in Fig. 16—we suggest caution in making conclusions that the substantially higher estimates of daily intake of digestible energy depicted on the map that was based on lichen influences really is more accurate in an absolute sense than the map derived from equations that excluded lichens. Whatever the

case, we emphasize that daily intake of digestible energy emerged as among the most important of our habitat and nutritional resources metrics in both years, supporting their value for predicting caribou body fat.

Developing foodscapes and relating spatial distribution to herbivore responses is a new area of study, but some recent examples have emerged that point to the value of this type of research. Walker (2023) found that forage intake rate was the primary driver of habitat variables evaluated on habitat selection of boreal caribou in Ontario, and that lactating caribou selected for areas that provide relatively high forage intake rates but, at the same time, avoided areas of high predation risk during mid- to late-summer. Forage intake rate was determined using captive caribou foraging trials over two years in Ontario (i.e., the study we alluded to above), and methods used by Walker (2023) to develop foodscapes reflecting caribou nutrition were similar to those we used in our study area (prediction equations of forage intake rate excluded accepted lichens as a predictor covariate because of its lack of importance). Denryter et al. (2022) modeled caribou selection of nutritional resources depicted in a spatially-explicit foodscape to evaluate the effect of resource selection patterns on nutritional outcomes reflecting the intensity of caribou selection for areas of high predicted nutrition, and clearly showed that caribou must focus their foraging on areas of good nutrition to avoid highly inadequate levels of nutrition that would undoubtedly negatively impact their performance. The foodscapes for their work were developed based on captive caribou foraging trials in northeastern British Columbia using the same data sets we used to predict caribou nutrition on our southern Northwest Territories study area. In Ontario, Avgar et al. (2015) reported that caribou movements reflected food availability (biomass of digestible forage), wolf density, and preferred moose habitat and that “forage selection coefficients were most consistent of the ecological variables considered here, both within and between individuals”, in turn suggesting “that bottom-up effects may be of central importance to landscape use by woodland caribou”. Although DeMars et al. (2020) did not use mapped foodscapes to evaluate habitat selection of caribou in the Northwest Territories, they nevertheless concluded that the general pattern of selection for young (1-10 years old) and older burns (>30 years old) by females during the snow-free seasons “may be associated with greater forage abundance or quality in these areas and the higher nutritional demands experienced by lactating females during this period”. This is notable because the scope of the effort in terms of sample size and temporal and geographic extent was second to none and because the finding that shows the importance of early seral communities after wildfire is often unrecognized (DeMars et al. 2020).

Studies with other ungulate species also support the value of coupling foodscapes with animal responses in the context of improving understanding habitat use and animal performance. For mule deer in northeastern Oregon, Merems et al. (2020) found that variation in foodscape use significantly predicted early-winter levels of body fat in adult females, and that this overall pattern was most strongly related to

use in relation to accepted forage biomass during summer ($r^2 = 0.45$) rather than spring. Their general approach was similar to that we used in our study, in that they evaluated based on GPS telemetry locations the relationship between biomass of accepted forage (of vascular plants in this case) at use locations with body fat levels of individual animals within a single study area. An alternative approach was described by Cook et al. (2018) for elk. In their study, autumn body fat levels and pregnancy rate of nine populations of elk in the forests of western Oregon and Washington were related to average predicted levels of dietary digestible energy available to elk on each of the nine population ranges. Spatially-explicit dietary digestible energy estimates were derived from sampling foraging and nutrition dynamics of captive elk placed in native habitats of the region over a three-year period. Equations to predict nutritional metrics were generated and used to build foodscape models extending from the Canadian border to nearly the California border (using methods similar to those we used in our study area). Relationships were strong with predicted dietary digestible energy at each study area accounting for at least 78% of the variation in pregnancy rates and body fat levels of elk in the nine populations. As part of this work, investigators also evaluated habitat selection of elk in relation to spatially explicit foodscape of dietary digestible energy and found that this nutrition metric was the strongest and most consistent variable affecting habitat selection in the study (Rowland et al. 2018).

Together, these studies are establishing the importance of nutritional resources for large ungulate populations across a remarkably wide geographic region in North America. These studies indicated that herbivores seek and use areas of superior nutritional resources and the extent which they succeed in focusing their foraging on areas that provide abundant, high-quality food that satisfies their nutritional needs substantially impacts their performance. In addition, these studies, all of which evaluated nutritional resources at least during summer (year-round in some cases) establish that nutritional resources on summer ranges are fundamentally important in the context of animal performance.

Finally, our analysis indicated that body fat in both years was inversely related to distance from roads of caribou locations, i.e., higher levels of body fat near roads, an unexpected finding. However, there are few roads in our study area and many caribou never encountered a road during the study period. It is possible that our analysis was confounded by the spatial distribution of habitats with good nutrition in relation to existing roads. Under the conditions of our study area, we were able to find no evidence that close proximity during summer to roads reduced the body fat levels of caribou with calves in December.

Despite the challenges encountered in our study, particularly the huge swings in precipitation between the two years, our study adds additional evidence of the importance of nutrition in summer for caribou in the southern Northwest Territories. In combination with findings from the forage quality surveys described in the CIMP companion report (Cook et al. 2023), our findings indicate the following.

- (1) Much of the vegetation (roughly about half) that grows in caribou habitat is non-food, meaning that palatability, nutrient content, and/or toxic compounds render these plants of low value to caribou. Caribou exhibit strong preferences for certain plant species (Denryter et al. 2017), e.g., willows, birch, several species of forbs, hairy wildrye, reindeer lichens, arboreal hair lichens, and, perhaps most of all, mushrooms.
- (2) Amount of accepted forage varies markedly among ecosites and seral stages (stand age). Biomass of accepted grasses, forbs, and deciduous shrubs are most abundant in early seral stages and decline to relatively low levels in older stands. They tend to be highest in fens and upland white spruce ecosites and lowest in bogs, upland black spruce, sphagnum bog, and marshes dominated by graminoids. Biomass of accepted horsetails are largely invariant to stand age, are abundant in some wetland ecosites, and sparse in upland forested sites. Accepted lichens are largely absent in early seral stages up to about 40 years old, are sparse in marshes, sphagnum bogs, wet bogs, and upland white spruce of all ages, and tend to be abundant in relatively dry bogs and some upland black spruce stages. We found some evidence that accepted lichens tended to decline in older stands especially where overstory canopy cover and basal area were high. These results suggest a dichotomy of forage resource types, those that provide relatively high biomass of vascular plants and those that provide high abundance of lichens, and that both tend to occur in different ecosites and particularly different successional stages.
- (3) Body fat in December of adult female caribou that raised a calf over summer and early autumn in December ranged similarly in 2019 and 2021, from 4-5% to 13- to 15% body fat. Long term body fat levels dating back to 2015 are generally consistent across all years. The wide range in body fat within years (2019 and 2021) is suggestive that varying forage abundance, composition, and quality available to each caribou may have important influences on their body condition, a primary hypothesis that we evaluated in our study.
- (4) Predicted levels of caribou nutrition (primarily daily intake rates of digestible energy) at caribou GPS locations during summer was modestly, but significantly correlated to their body fat levels in December, but which habitat features and metrics of nutritional resources that were correlated with body fat varied between years. We attribute the differences to strikingly different precipitation levels between the two years. In the first year, biomass of accepted vascular plants, especially grasses, forbs, and deciduous shrubs (combined) and the habitats in which these are most abundant (young stands) was positively related to body fat whereas biomass of accepted lichens were not. In the second year when precipitation levels were markedly elevated, biomass of accepted lichens was positively related to body fat.

(5) The alternative findings preclude simple and definitive conclusions about which types of habitats and forage best contributed to body fat of caribou. These findings suggest that a highly diverse combination of habitats and forage types best accommodate nutritional needs of caribou across a wide range of long-term climatic conditions. However, our results for the first year of body fat in combination with the resource selection studies by DeMars et al. (2020) indicate the value of early seral vegetation communities for boreal caribou in the southern Northwest Territories, a finding that expands on common perceptions that biomass of lichens and old growth forests best provide for their nutritional needs. On the other hand, our findings do not in any way distract from the importance of lichens in winter, and our second year of data collected during the wet summer of 2021 suggests that lichens may be important for caribou during summer as well.

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Table 1. Description of ecosites used for vegetation classification in the southern Northwest Territories, 2016 and 2019 (scientific names of plants are listed in Appendix A).

Ecosite name (code)	Description
Upland black spruce (Upl-BS)	An unproductive forest ecosite dominated by Jack pine in early and mid-seral stands and black spruce in old stands (white spruce is rare or absent). Undergrowth is sparse, with bearberry, juniper, other scattered shrubs, and, often, abundant terrestrial reindeer lichens on drier phases, and rose, willow, wildrye, and occasionally paper birch, with terrestrial lichens rare, on moister phases. After disturbance, aspen, fireweed, willow, rose, and wildrye dominate the ecosite.
Upland white spruce (Upl-WS)	A productive forest ecosite usually with a mix of conifer and hardwoods in early and mid-seral stands and with white spruce present usually as a codominant with aspen, balsam poplar, Jack pine, or black spruce. Hardwoods decline as stands age. Undergrowth is typically very productive, especially in early seral communities. Rose, alder, soapberry, and highbush cranberry usually dominate the undergrowth layer. Soon after disturbance, aspen, rose, willow, and other deciduous shrubs, fireweed, and wildrye often dominate. Terrestrial reindeer lichens are absent to sparse.
Bogs (Bog)	Bogs are peatlands with or without trees. The overstory, if present, is normally <10 m and is dominated by black spruce in all stand ages, although aspen or Jack pine can be present in early seral stages. The underbrush is dominated by ericaceous shrubs, particularly Labrador tea and leatherleaf (indicators of this type), and deciduous shrubs are sparse, with abundant terrestrial reindeer lichens in drier phases. After wildfire, either ericaceous shrubs or a more diverse community of deciduous shrubs and forbs dominate, probably depending on soil type and dominant vegetation prior to fire. We included swamps in this category, which typically have undergrowth vegetation as described for bogs, but trees are typically denser and taller (often > 10 m tall) than typically identified for bogs (many types of swamps exist as described elsewhere ([Ducks Unlimited 2014; Racey et al. 1996]) that were typically not encountered in our study.
Fens (Fen)	Fens are wetlands with or without trees. The overstory if present is typically <10 m and is dominated by black spruce and larch. The undergrowth layer is dominated by deciduous shrubs, particularly dwarf birch and shrubby cinquefoil, with willow and paper birch often common. Graminoids, mainly sedges, vary from sparse to abundant, but shrubs dominant in this type (as we defined it). Terrestrial reindeer lichens are absent to sparse. After disturbance, deciduous shrubs, fireweed, and sedges rapidly reestablish, usually resulting in a diverse and productive community.

Marshes (Marsh)	Marshes are wetlands lacking trees. Vegetation is graminoid-dominated, usually by various sedge species, and shrubs are sparse or absent as we defined the type. Sites are typically very wet with standing water much of the year. Accepted lichens are virtually absent.
Sphagnum bogs (SpBg)	Sphagnum bogs are an extreme of bogs relatively common on the study area where very little vegetation other than moss is present. As we defined this type, sphagnum bogs support a deep peat layer and very sparse vascular plant life forms and lack trees. Vascular vegetation is usually limited to evergreen ericaceous shrubs with virtually no deciduous shrubs, forbs, or grasses present. Terrestrial lichens also are usually sparse and range from absent to moderately abundant.

Table 2. User's accuracy of CWIM3 and NWT EOSD with NDVI and white pine thresholds to identify ecosites types compared to field-derived classification of ecosites at 223 macroplots in the Northwest Territories, Canada.

Field-derived	CWIM3 & EOSD						n	Total	User's accuracy
	Bog	Fen	Marsh	PtPl	Upl-BS	Upl-WS			
Bog	16	12			10	1	16	39	41%
Fen	4	30	3	1	2	2	30	42	71%
Marsh		3	4				4	7	57%
PtPl			1	5			5	6	83%
Upl-BS	1				51	23	51	75	68%
Upl-WS	2				14	38	38	54	70%
Total	23	45	8	6	77	64	144	223	65%

Table 3. Coefficient of determination (r^2) and sign of the slope coefficient (indicating either a positive or negative relationship) for simple regression relationships between caribou body fat in December and environmental covariates by year and summer period (early and late summer = 15 July to 15 September; late summer = 15 August to 15 September). Acronyms are: DE = digestible energy and DP = digestible protein.

Covariate	Early and late summer				Late summer only			
	2019		2021		2019		2021	
	r^2	Sign ^a	r^2	Sign ^a	r^2	Sign ^a	r^2	Sign ^a
Distance to roads	0.15	-*	0.15	-*	0.13	-*	0.16	-*
Ecosite (proportion of locations)								
Proportion lowlands	0.01	-	0.00	na	0.03	-	0.03	-
Proportion fens	0.05	-	0.01	+	0.08	-	0.00	na
Proportion bogs	0.06	+	0.12	-*	0.02	+	0.13	-*
Proportion Upl-BS	0.00	na	0.12	+	0.00	na	0.19	+
Proportion Upl-WS	0.01	+	0.07	-	0.02	+	0.07	-
Forest overstory								
Stand age (years)	0.12	-*	0.06	+	0.17	-**	0.04	+
Overstory canopy cover (%)	0.02	+	0.00	na	0.06	+	0.01	-
Overstory basal area (m ² /ha)	0.02	+	0.00	na	0.04	+	0.00	na
Overstory height (m)	0.01	+	0.00	na	0.03	+	0.00	na
Forage quantity								
Accepted grass, forb, shrub (kg/ha)	0.04	+	0.08	-	0.03	+	0.08	-
Accepted lichens (kg/ha)	0.08	-	0.09	+	0.11	-*	0.14	+
Total accepted forage (kg/ha)	0.02	-	0.05	+	0.05	-	0.08	+
Predicted caribou nutrition								
DE intake/day (lichen eq.) (MJ) ^b	0.01	-	0.11	+	0.04	-	0.17	+
DE intake/day (non-lichen eq.) (MJ) ^b	0.03	+	0.04	-	0.16	+	0.08	-
DP intake/day (g)	0.08	+	0.03	-	0.08	+	0.07	-
Dietary DE (lichen eq.) (kJ) ^b	0.01	-	0.08	+	0.08	-	0.17	+
Dietary DE (non-lichen eq.) (kJ) ^b	0.01	+	0.00	na	0.01	-	0.02	+
Dietary DP (%)	0.03	+	0.04	-	0.05	+	0.06	-

^a Sign of the slope coefficient. Codes are * = significant at $P \leq 0.10$; ** = significant at $P \leq 0.05$; na = not applicable (i.e., $r^2 = 0.00$).

^b Estimates based on either a lichen-based equation (eq.) or one that omits lichen as a predictor variable.

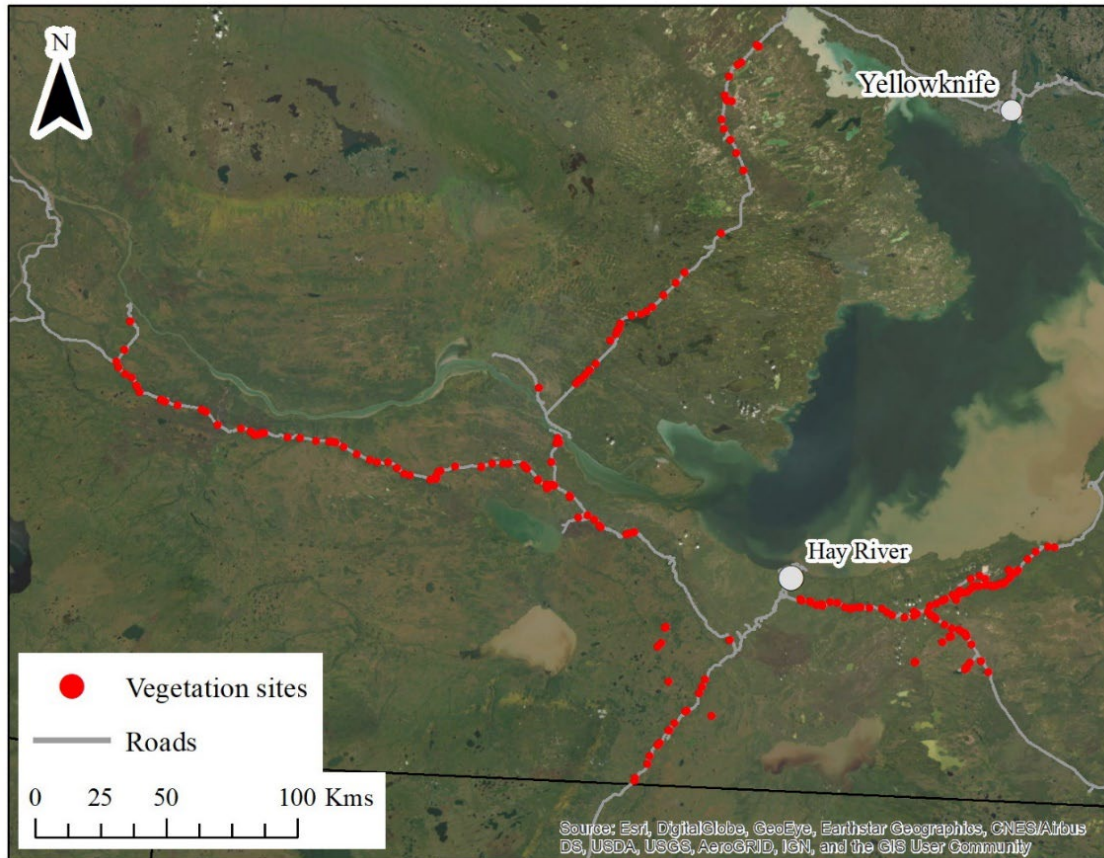


Figure 1. Location of study area in the southern Northwest Territories. Symbols indicate locations of field plots sampled to estimate forage quality and quantity and other vegetative and topographic attributes.

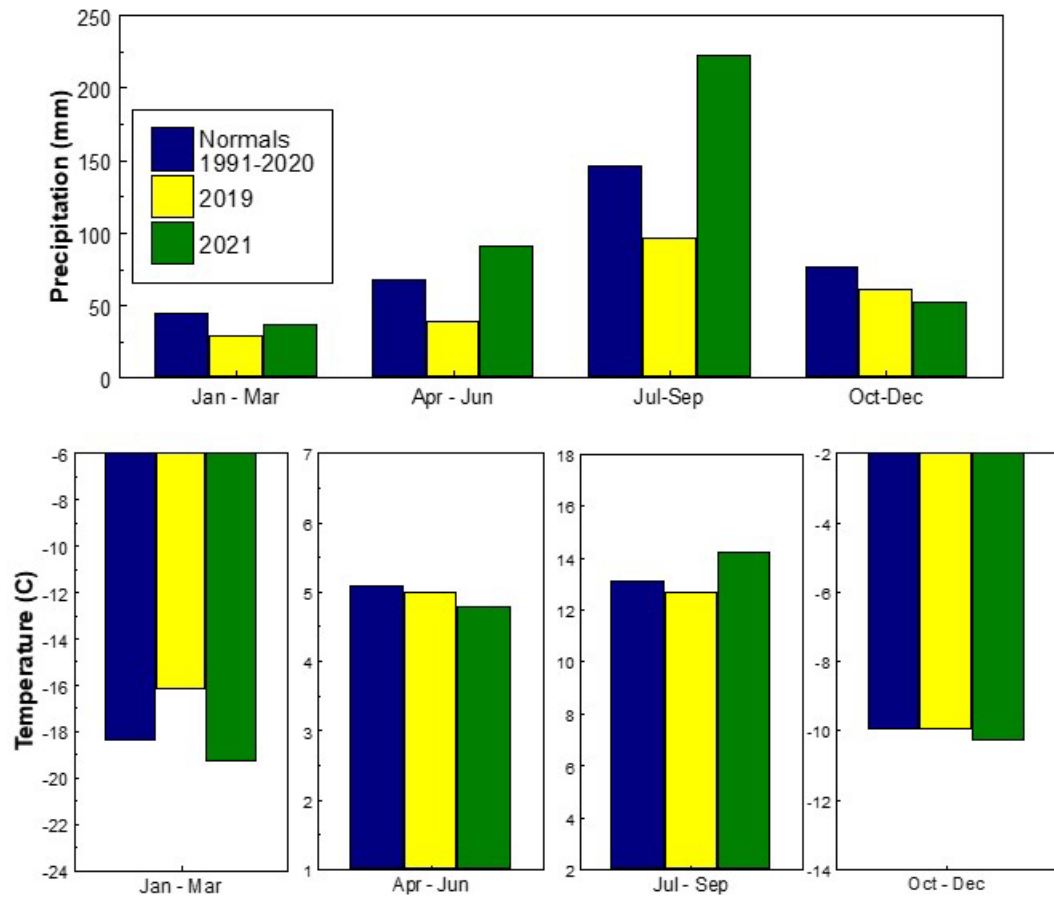


Figure 2. Cumulative precipitation and average ambient temperature by season at Hay River, Northwest Territories, during 2019 and 2021 compared to long term (1991-2020) means (normals).

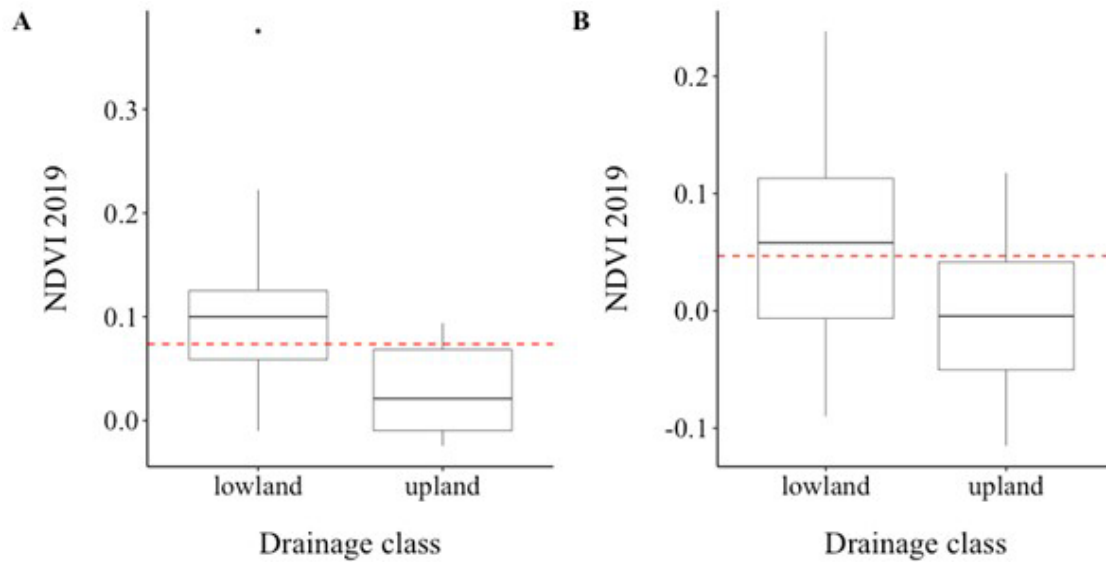


Figure 3. Greenness index (NDVI) threshold values (red-dashed line) derived from receiver operating characteristic curve analysis (ROC) to assist CWIM3's classification of A) lowlands and B) uplands. If CWIM3 classified a pixel as lowlands and the NDVI values was greater than 0.07 we classified it as lowlands and less than 0.07 as uplands. Additionally, if CWIM3 classified a pixel as uplands and the NDVI value was greater than 0.05, we classified it as lowlands and less than 0.05 as uplands.

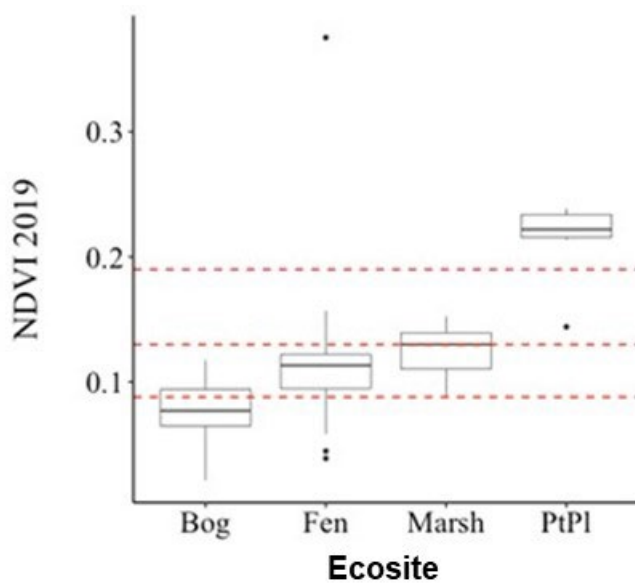


Figure 4. Greenness index (NDVI) threshold values to differentiate between lowland ecosites. If NDVI value was less than 0.09 we classified the pixel as a bog, between 0.09 and 0.13 was classified as a fen, between 0.13 and 0.15 was classified as a marsh, and greater than 0.15 it was classified as a sphagnum bog (labeled on graph as PtPl).

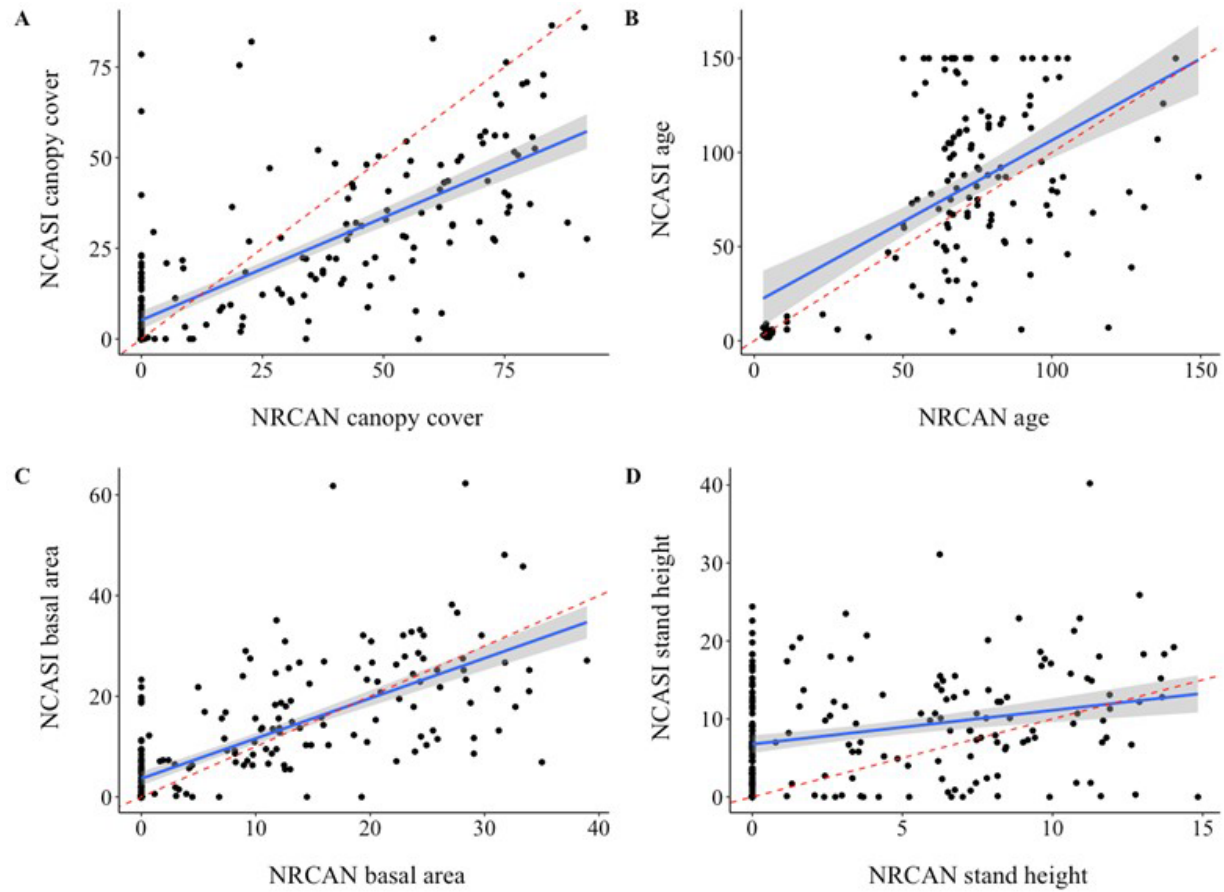


Figure 5. Relationship between field-derived (NCASI) estimates of (A) canopy cover, (B) stand age (years), (C) basal area (m²/ha), and (D) stand height (m), and NRCAN spatial layers developed by Matasci et al. (2018; canopy cover, basal area, and stand height) and Maltman et al. (2023; stand age) with linear regression trend line (blue line) and 95% CI (grey), and 1:1 line (red-dashed line).

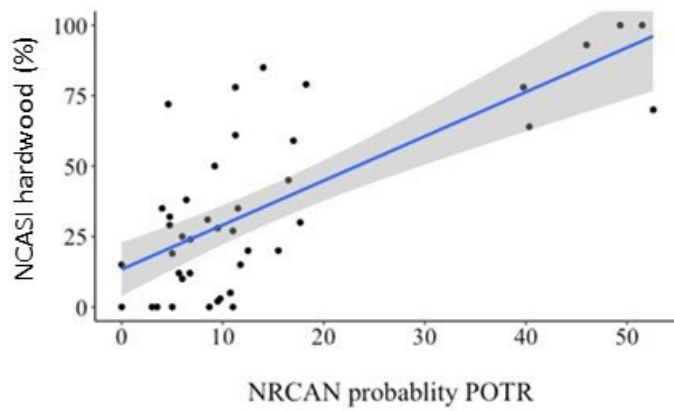


Figure 6. Relationship between field-derived (NCASI) estimates of percent hardwoods and NRCAN percent probability of aspen occurrence developed by Hermosilla et al. (2022) with linear regression trend line (blue line) and 95% CI (grey).

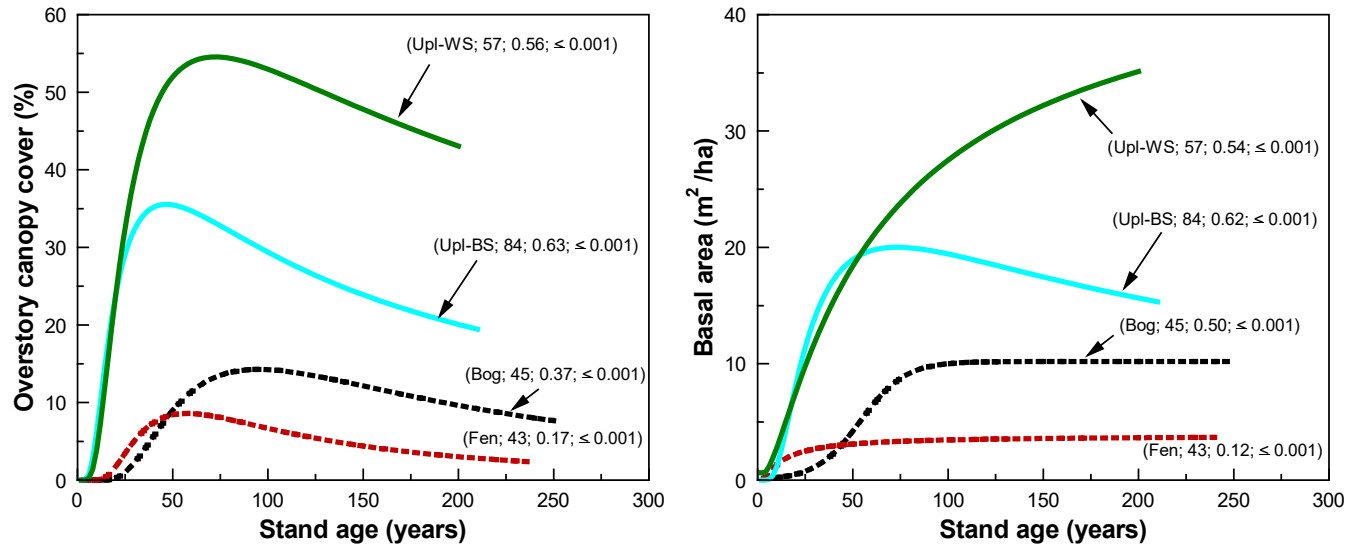


Figure 7. Successional development of overstory canopy cover and basal area of forests after stand-replacing disturbance in the southern Northwest Territories based on data collected in summer, 2016 and 2019. Information in parenthesis is ecosite code, sample size, coefficient of determination (r^2), and probability value of F-test for the overall regression equation. Ecosite codes are: Upl-WS = upland white spruce, Upl-BS = upland black spruce, Bog = bogs, and Fen = fens.

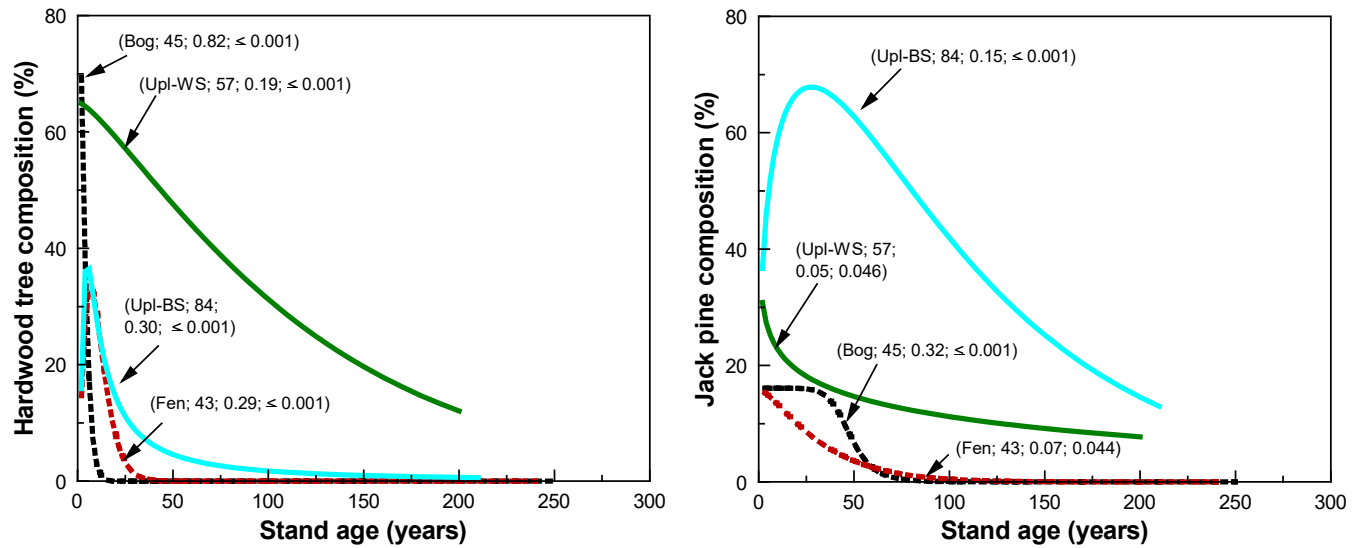


Figure 8. Changes across succession of overstory hardwood tree composition and Jack pine composition in the southern Northwest Territories, 2016 and 2019. Parenthetical information includes ecosite, sample size, coefficient of determination (r^2), and P-value of F-test for overall regression result. Ecosite codes are: Upl-WS = upland white spruce, Upl-BS = upland black spruce, Bog = bogs, and Fen = fens.

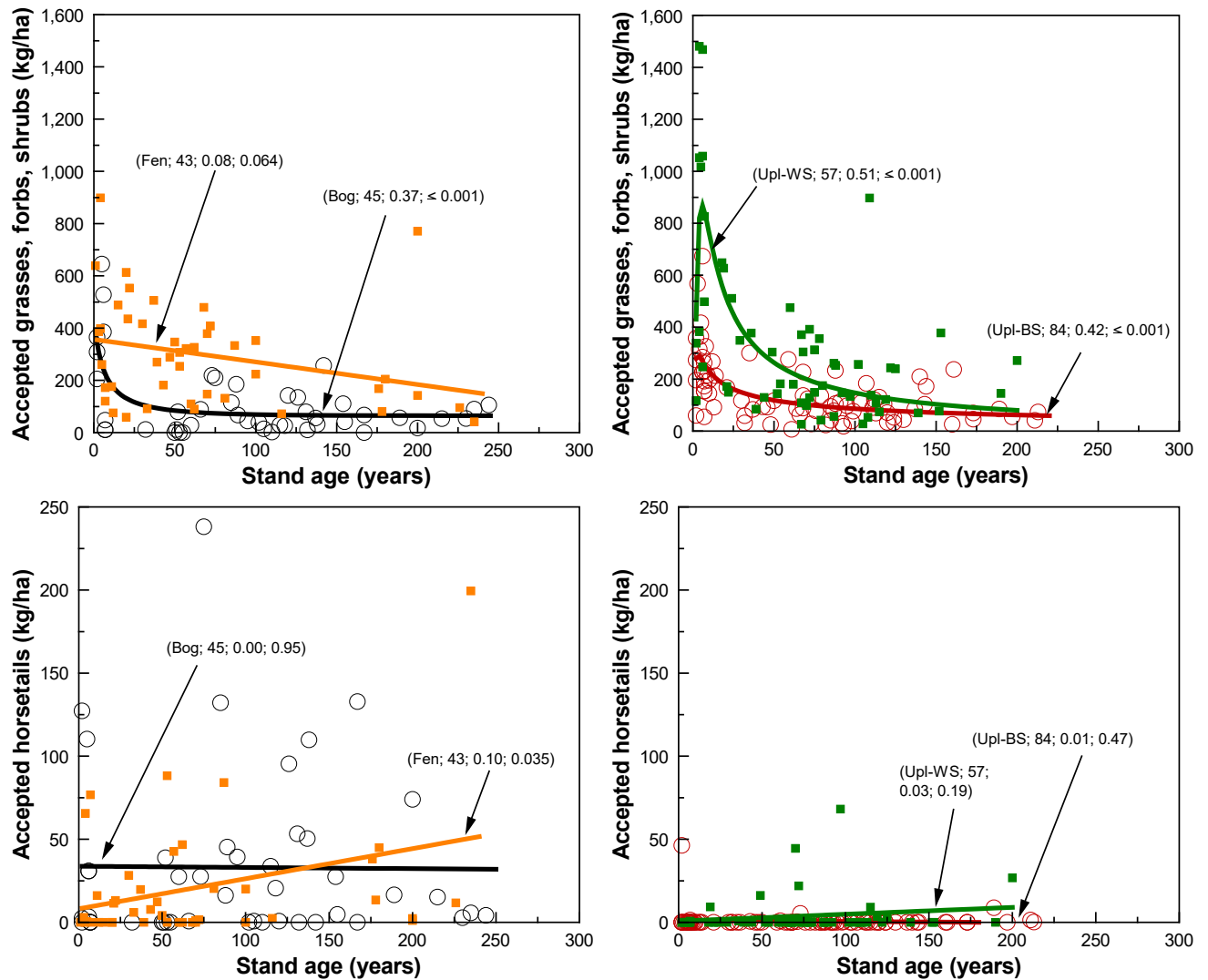


Figure 9. Successional development of accepted grasses, forbs, and deciduous shrubs and accepted horsetails in the southern Northwest Territories during summer, 2016 and 2019. Information in parenthesis is ecosite code, sample size, coefficient of determination (r^2), and probability value of F-test for the overall regression equation. Ecosite codes are: Upl-WS = upland white spruce, Upl-BS = upland black spruce, Bog = bogs, and Fen = fens.

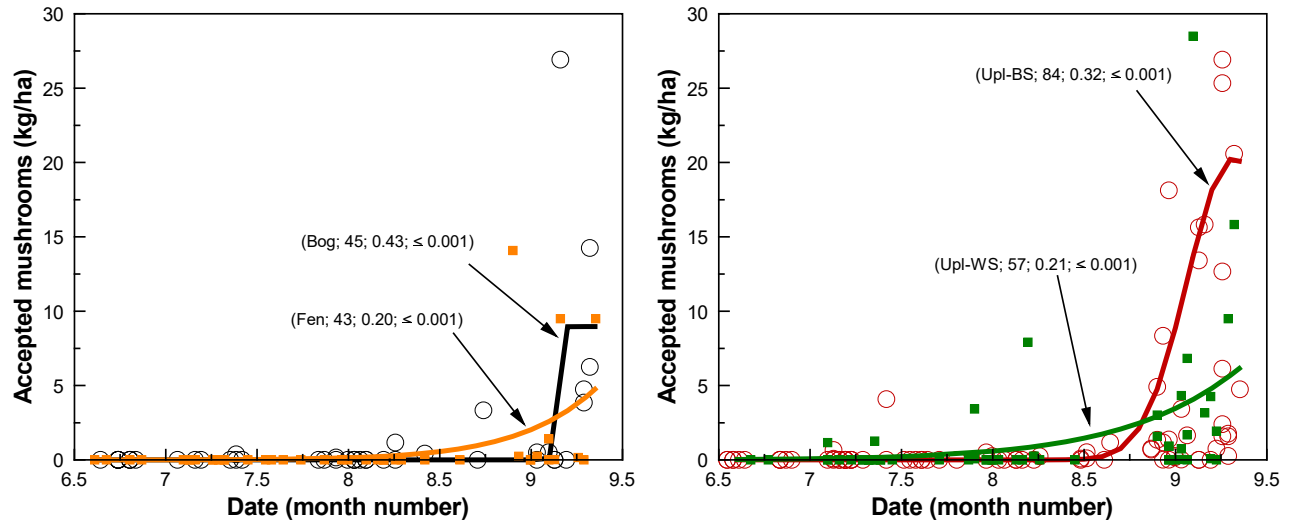


Figure 10. Biomass of accepted mushrooms in the southern Northwest Territories as a function of date sampled during summer, 2016 and 2019. Information in parenthesis is ecosite code, sample size, coefficient of determination (r^2), and probability value of F-test for the overall regression equation. Ecosite codes are: Upl-WS = upland white spruce, Upl-BS = upland black spruce, Bog = bogs, and Fen = fens.

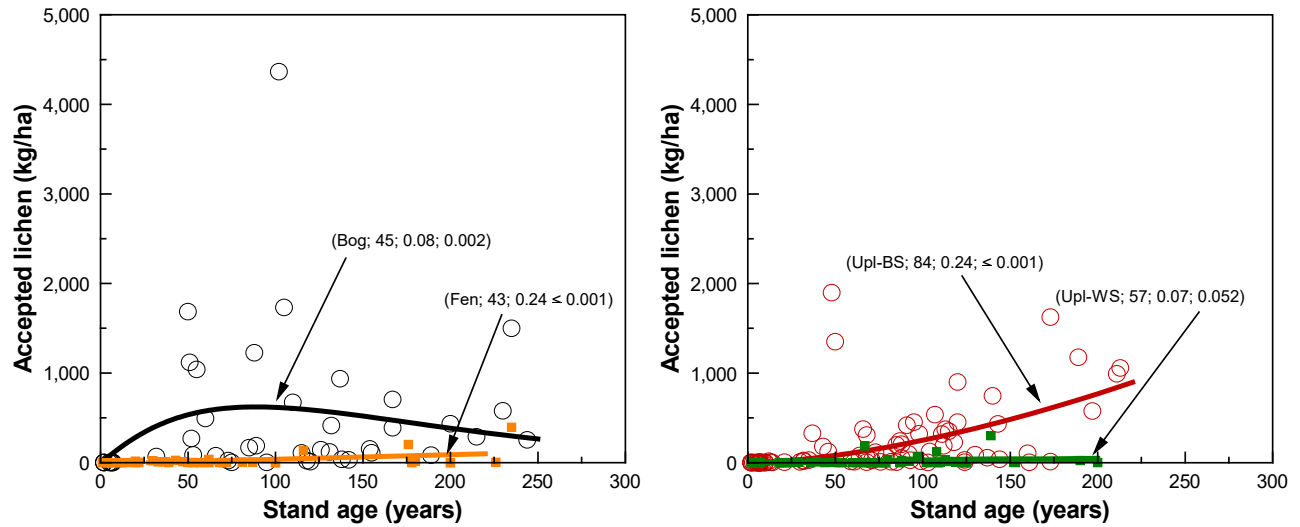


Figure 11. Successional development of accepted arboreal hair and accepted terrestrial lichens in the southern Northwest Territories during summer, 2016 and 2019. Information in parenthesis is ecosite code, sample size, coefficient of determination (r^2), and probability value of F-test for the overall regression equation. Ecosite codes are: Upl-WS = upland white spruce, Upl-BS = upland black spruce, Bog = bogs, and Fen = fens.

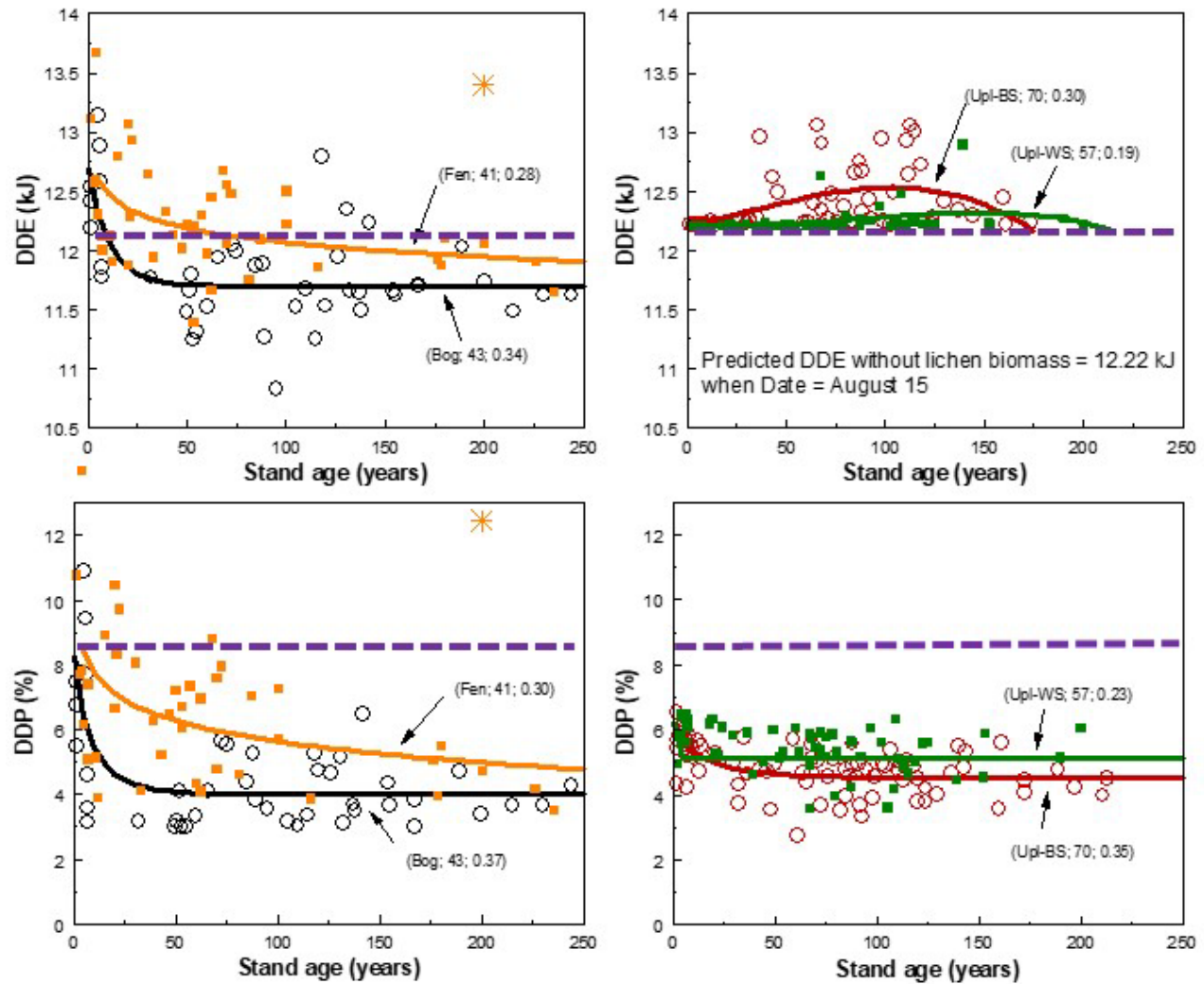


Figure 12. Content of digestible energy (DDE) and digestible protein (DDP) in caribou diets predicted for mid-August based on forage biomass data collected in the southern Northwest Territories in summer 2016 and 2019 and prediction equations developed using captive caribou in northeastern BC in summer 2013-2016. Horizontal dashed lines indicate approximate nutritional requirements during summer of lactating caribou. Ecosite code, sample size, and r^2 are presented parenthetically. Ecosites are: Upl-BS = upland black spruce, Upl-WS = upland white spruce, Bog = bog, Fen = fen. The starred data values were treated as outliers for calculating the regression curves. Using an alternative equation to predict DDE in upland black spruce ecotypes that did not contain lichens as a covariate provided modestly lower estimates of DDE (top right graph).

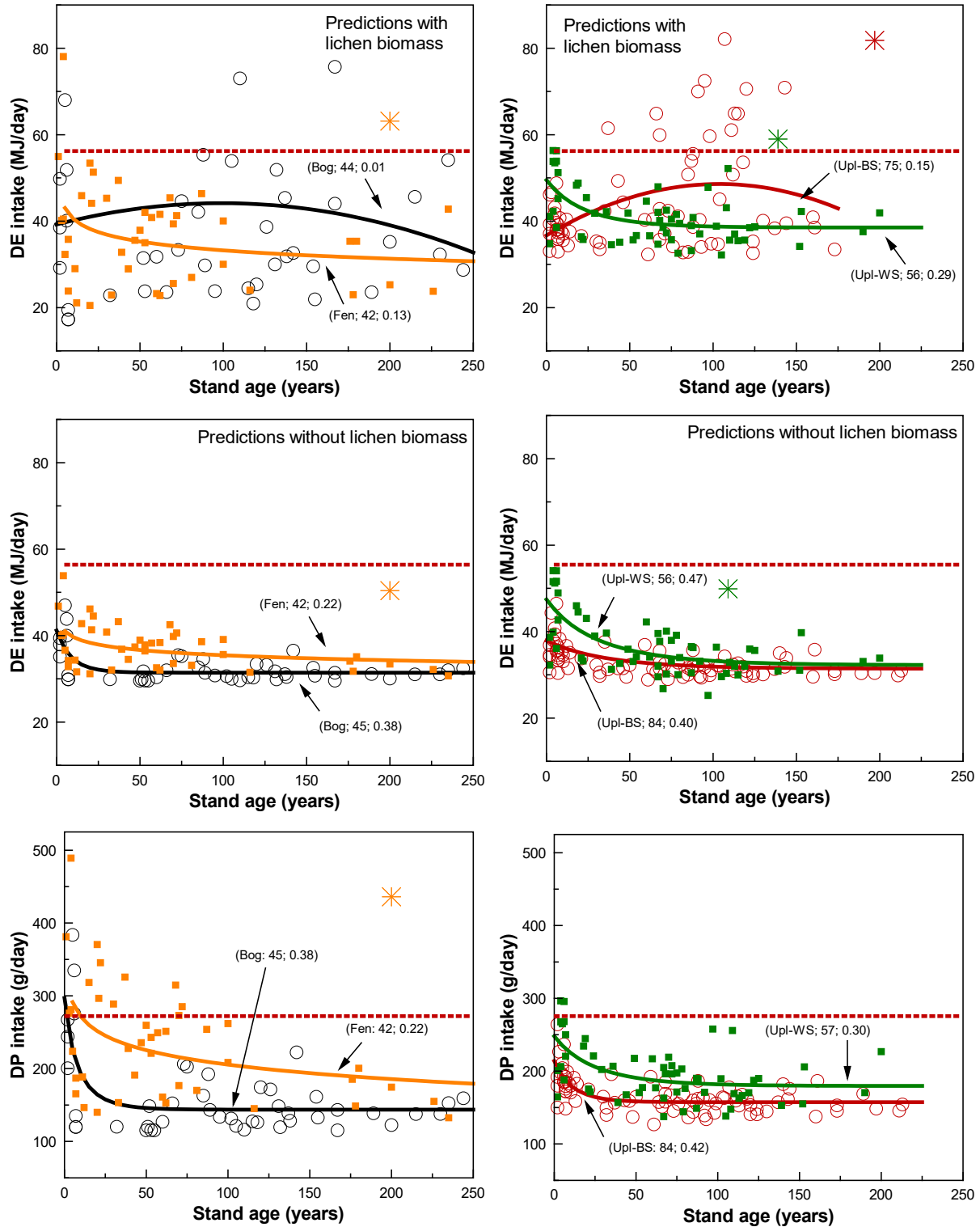


Figure 13. Daily intake rate of digestible energy (DE) and digestible protein (DP) by caribou predicted for mid-August based on forage biomass data collected in the southern Northwest Territories in summer 2016 and 2019 and prediction equations developed using captive caribou in northeastern BC in summer 2013-2016. Horizontal dashed lines indicate

approximate nutritional requirements during summer of lactating caribou. The starred data values were treated as outliers for calculating the regression curves. Prediction of daily intake of DE reflect two alternative equations, , one that included lichen biomass as a covariate and the other without lichen biomass as a covariate to illustrate the effect of including lichens in the prediction equation in this ecosite. Ecosite code, sample size, and r^2 are presented parenthetically. Ecosites are: Upl-BS = upland black spruce, Upl-WS = upland white spruce, Bog = bog, Fen = fen.

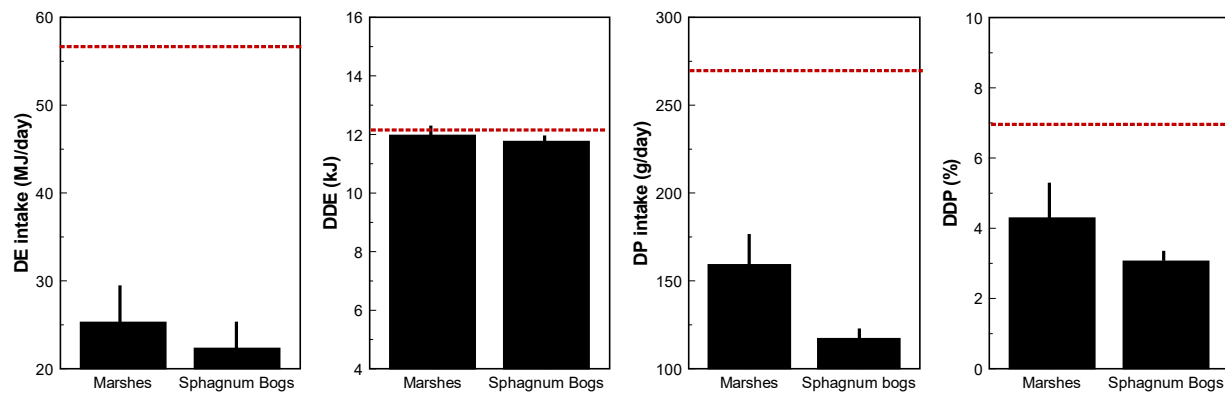


Figure 14. Daily intake rate of digestible energy (DE) and dietary digestible protein (DP) and concentration of DE and DP in caribou diets (DDE and DDP) predicted for mid-August based on forage biomass data collected in the southern Northwest Territories in summer 2016 and 2019 and prediction equations developed using captive caribou in northeastern British Columbia in summer 2013-2016. Horizontal dashed lines indicate approximate nutritional requirements during summer of lactating caribou. Vertical lines indicate standard error of the mean. All estimates are projected to 15 August.

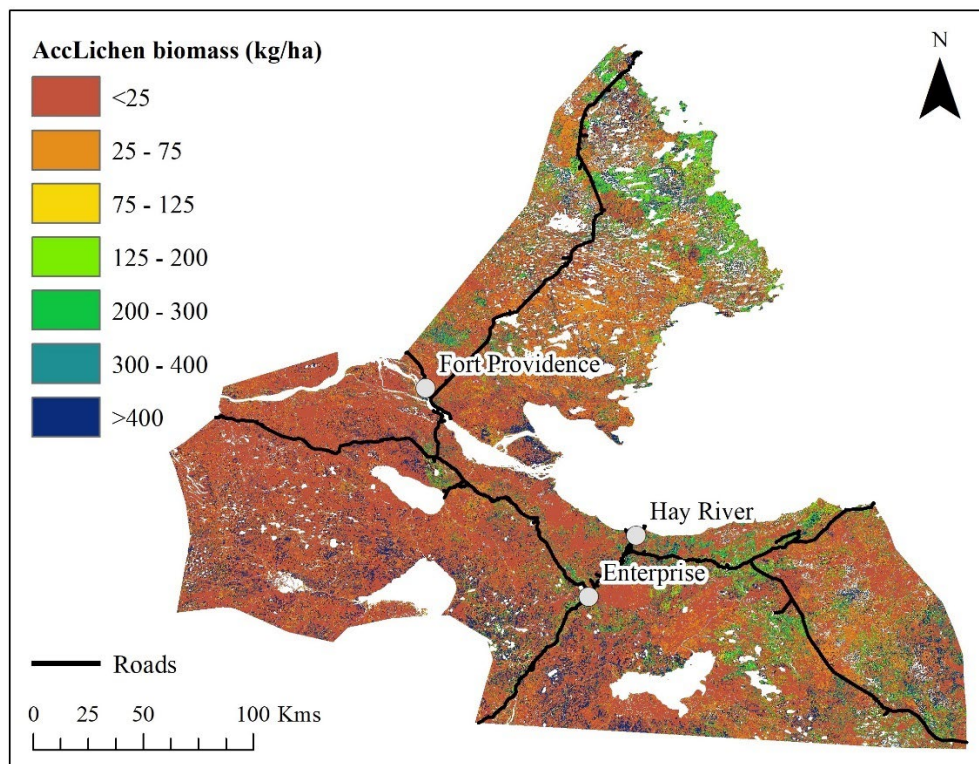
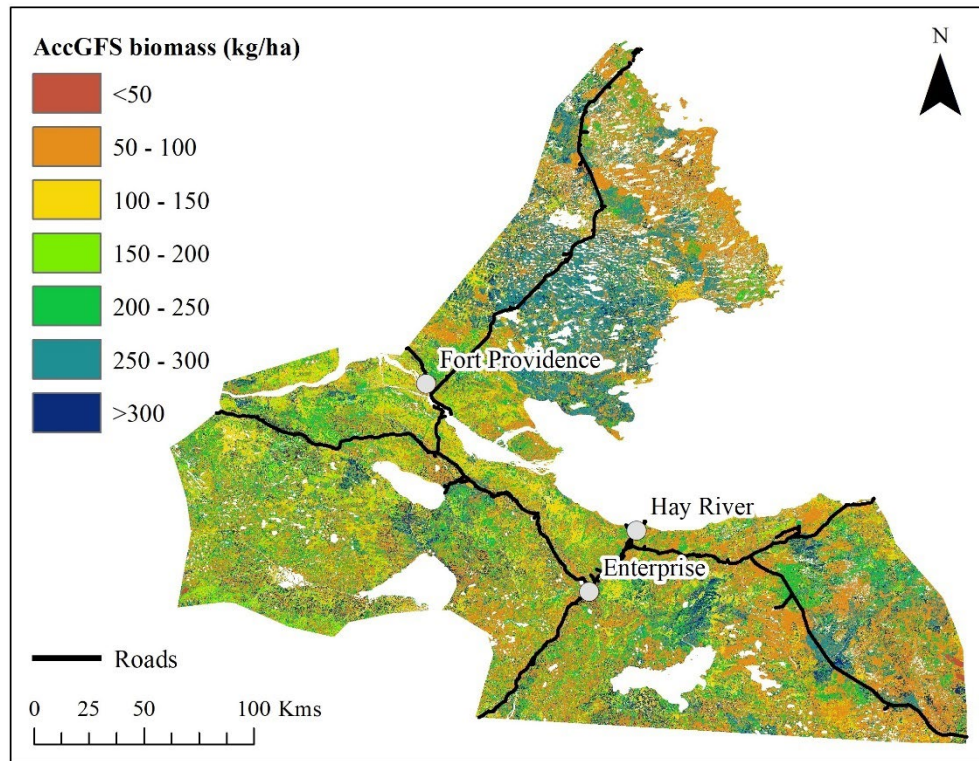


Figure 15. Maps of biomass of accepted grasses, forbs, and deciduous shrubs (top panel) and accepted lichens (bottom panel) in the southern Northwest Territories based on vegetation sampling in summer 2016 and 2019.

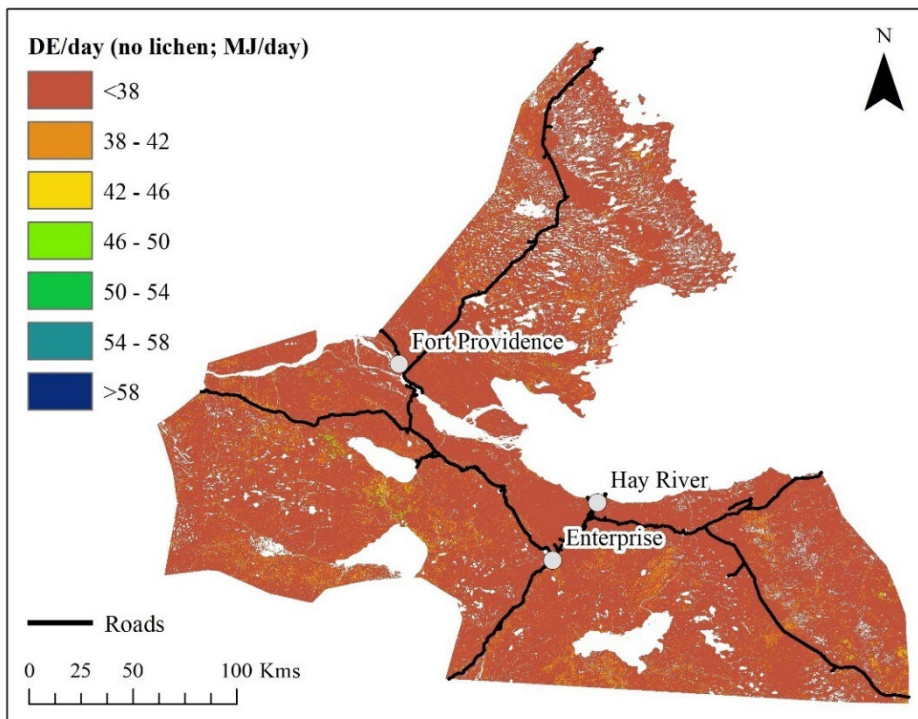
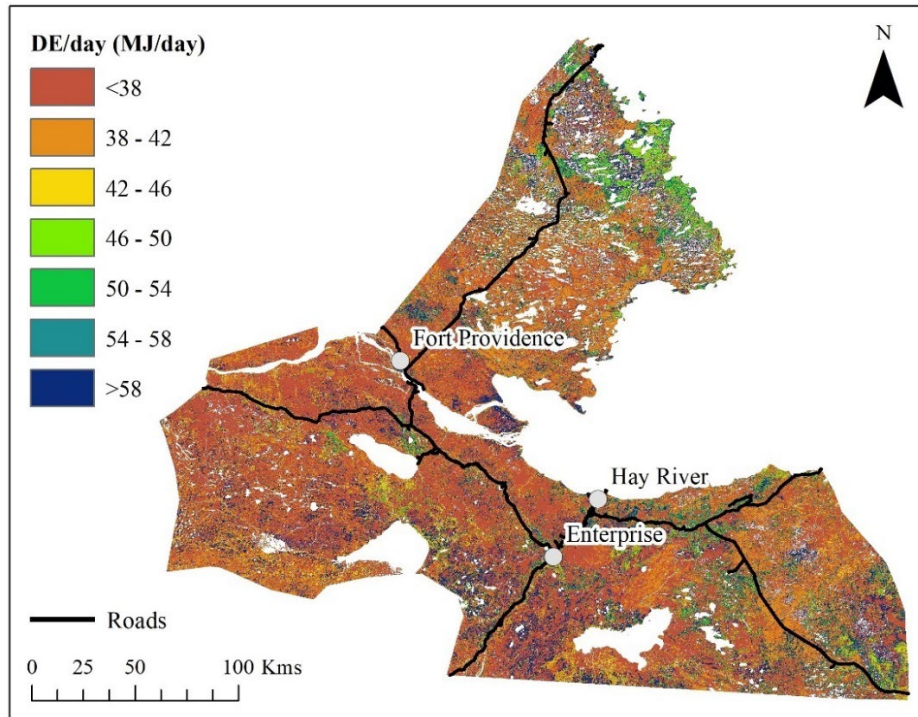


Figure 16. Maps of predicted caribou nutrition (daily intake of digestible energy [MJ/day]) based on prediction equations that included lichen biomass (top panel) and prediction equations that excluded lichen biomass (bottom panel) as predictor covariates, calculated for 15 August. Roughly 60 MJ of digestible energy is required each day for lactating female caribou.

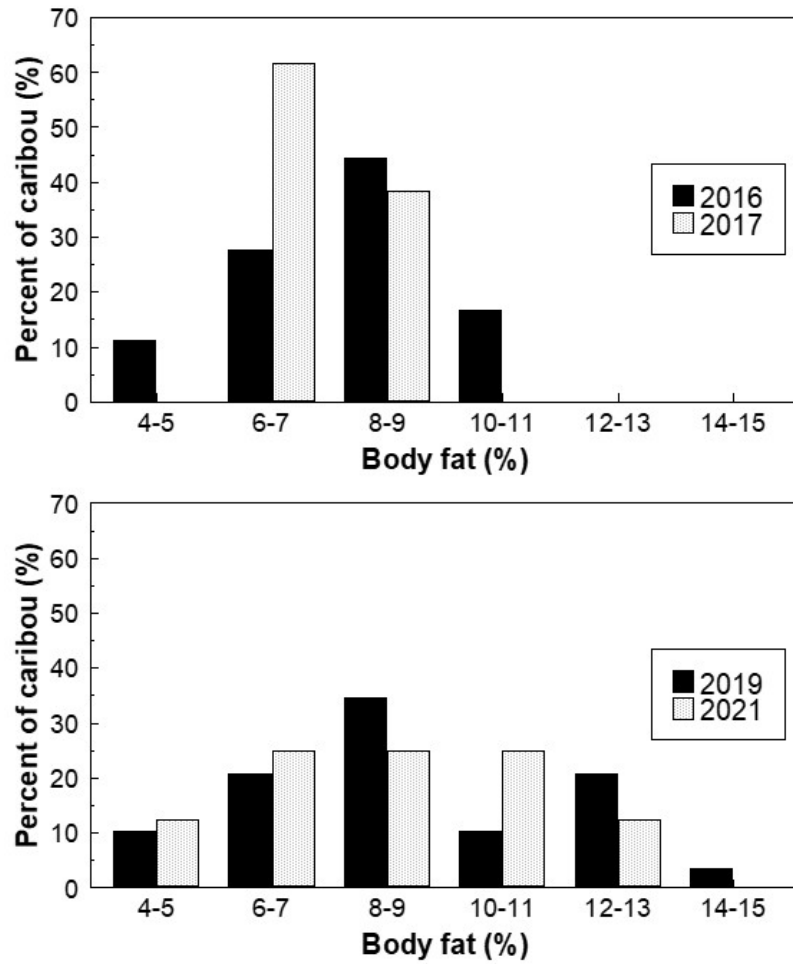


Figure 17. Histogram of body fat for adult female caribou in December that raised a calf the previous summer and autumn.

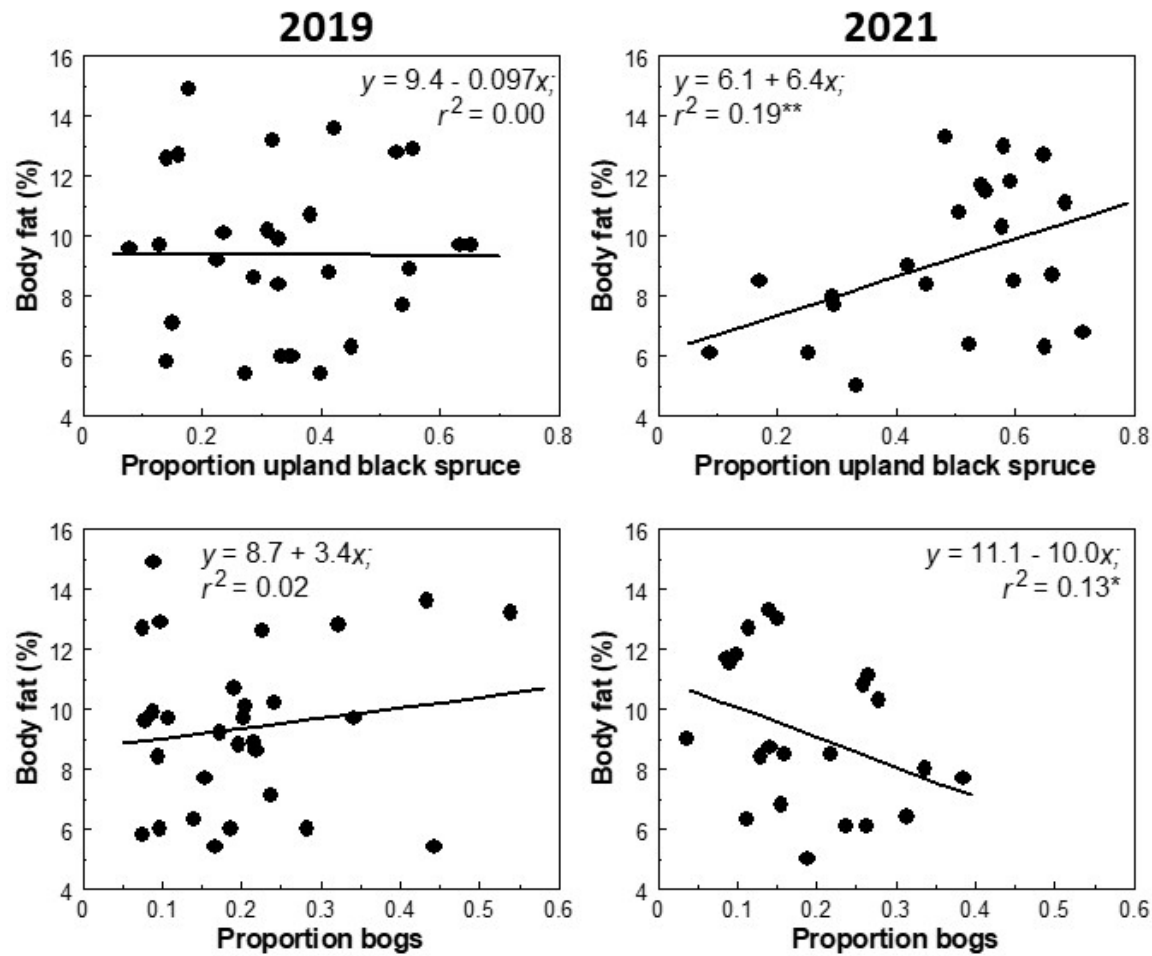


Figure 18. Relationships between body fat and proportion of caribou telemetry locations in the upland black spruce ecosite (top row of panels) and in the bog ecosite (bottom row of panels) during late summer (15 August to 15 September), 2019 and 2021, in the southern Northwest Territories (* = significant at $P \leq 0.10$; ** = significant at $P \leq 0.05$).

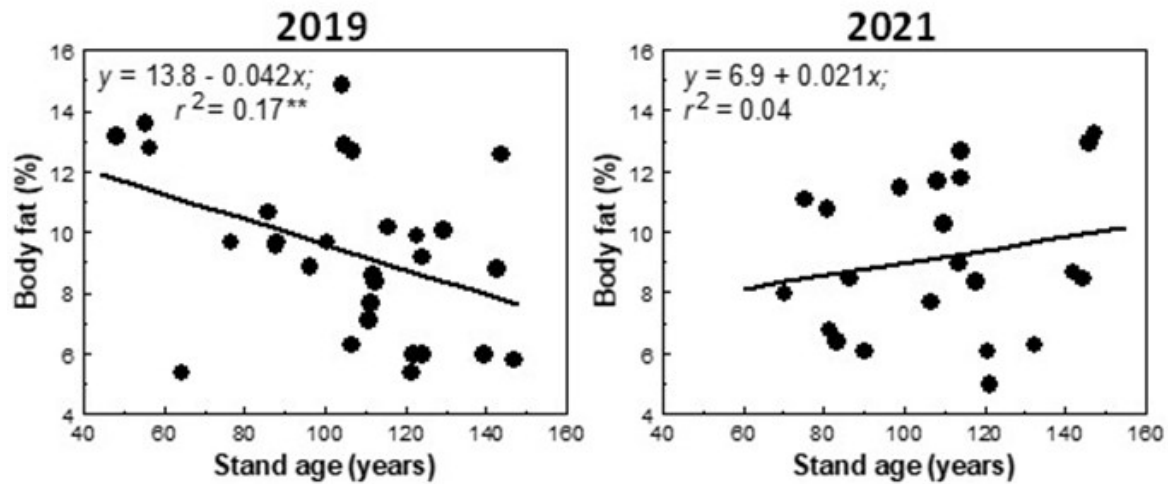


Figure 19. Relationships between body fat and stand age of caribou telemetry locations during late summer (15 August to 15 September) in 2019 and 2021 in the southern Northwest Territories (* = significant at $P \leq 0.10$; ** = significant at $P \leq 0.05$).

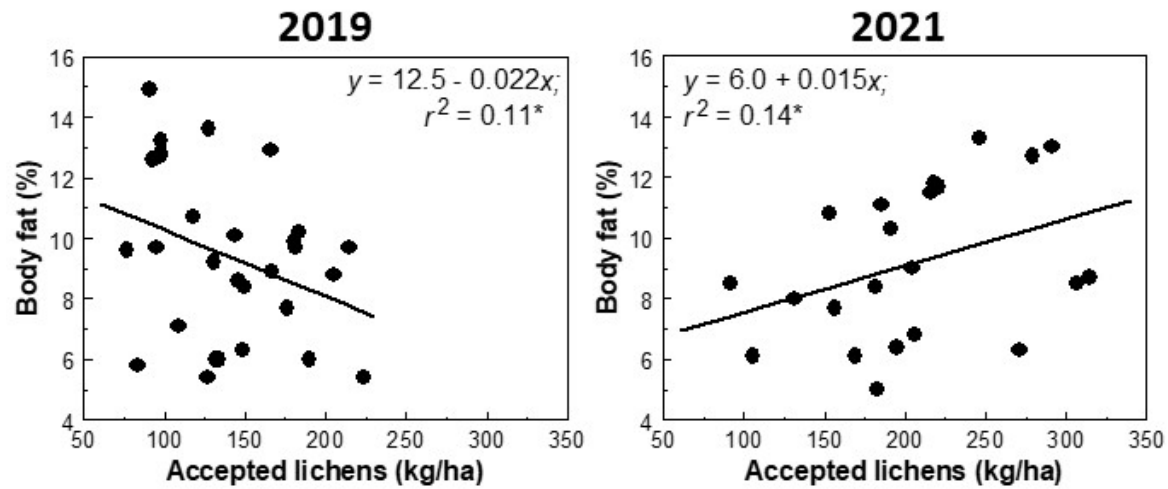


Figure 20. Relationships between body fat and biomass of accepted arboreal and terrestrial lichens at caribou telemetry locations in during late summer (15 August to 15 September) 2019 and 2021 in the southern Northwest Territories (* = significant at $P \leq 0.10$; ** = significant at $P \leq 0.05$).

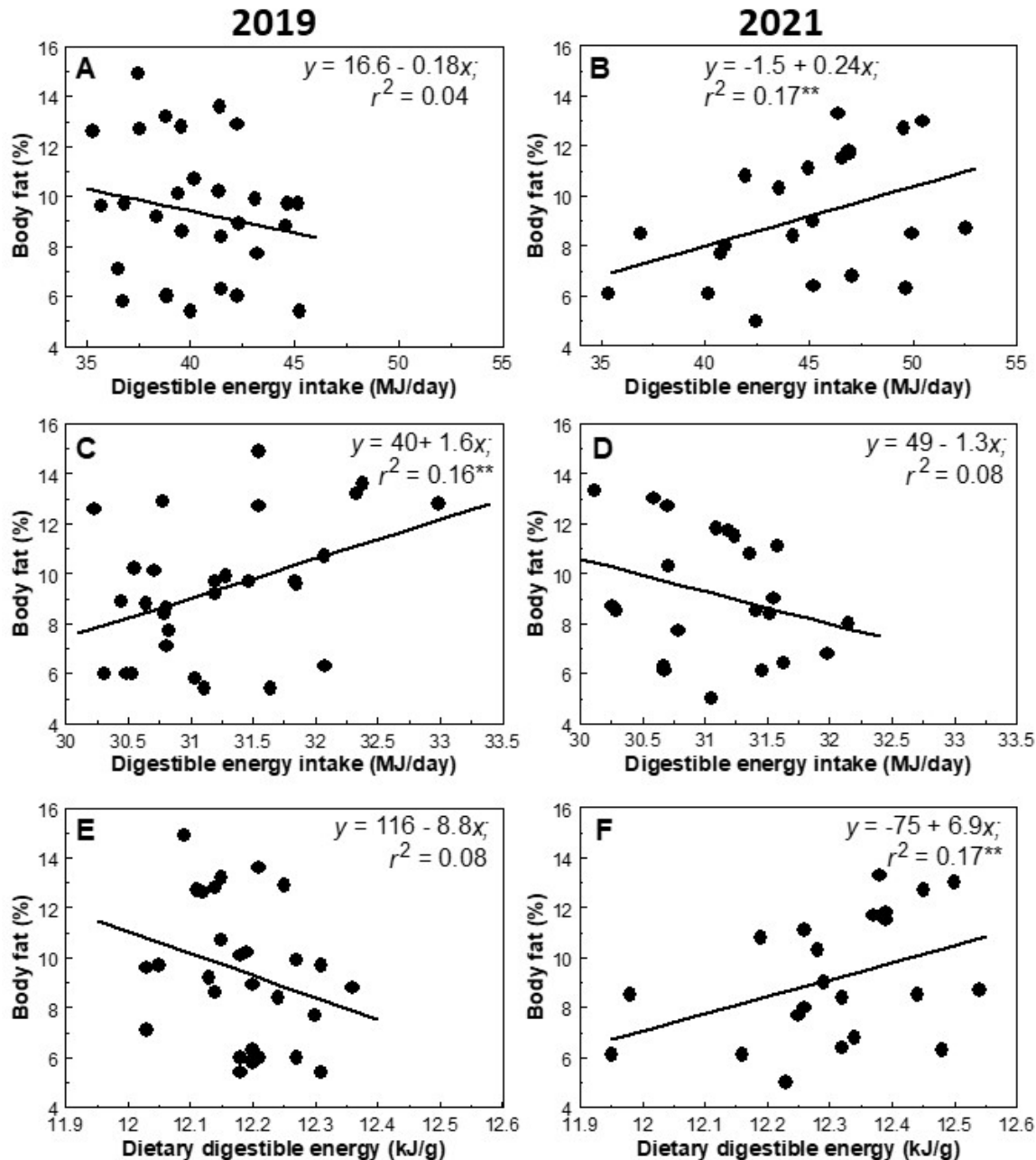


Figure 21. Relationships between body fat and daily intake of digestible energy (MJ/day = mega Joules of digestible energy consumed per day) (panels A – D) and concentration of digestible energy (kJ = kilo Joules per gram of consumed forage) (panels E and F) at caribou telemetry locations during late summer (15 August to 15 September), 2019 and 2021, in the southern Northwest Territories (* = significant at $P \leq 0.10$; ** = significant at $P \leq 0.05$). Daily intake of digestible energy for panels A and B were calculated based on a prediction equation that included accepted lichen biomass as a predictor covariate; daily intake of digestible energy for panels C and D were calculated based on a prediction equation that excluded lichen biomass as a predictor covariate.

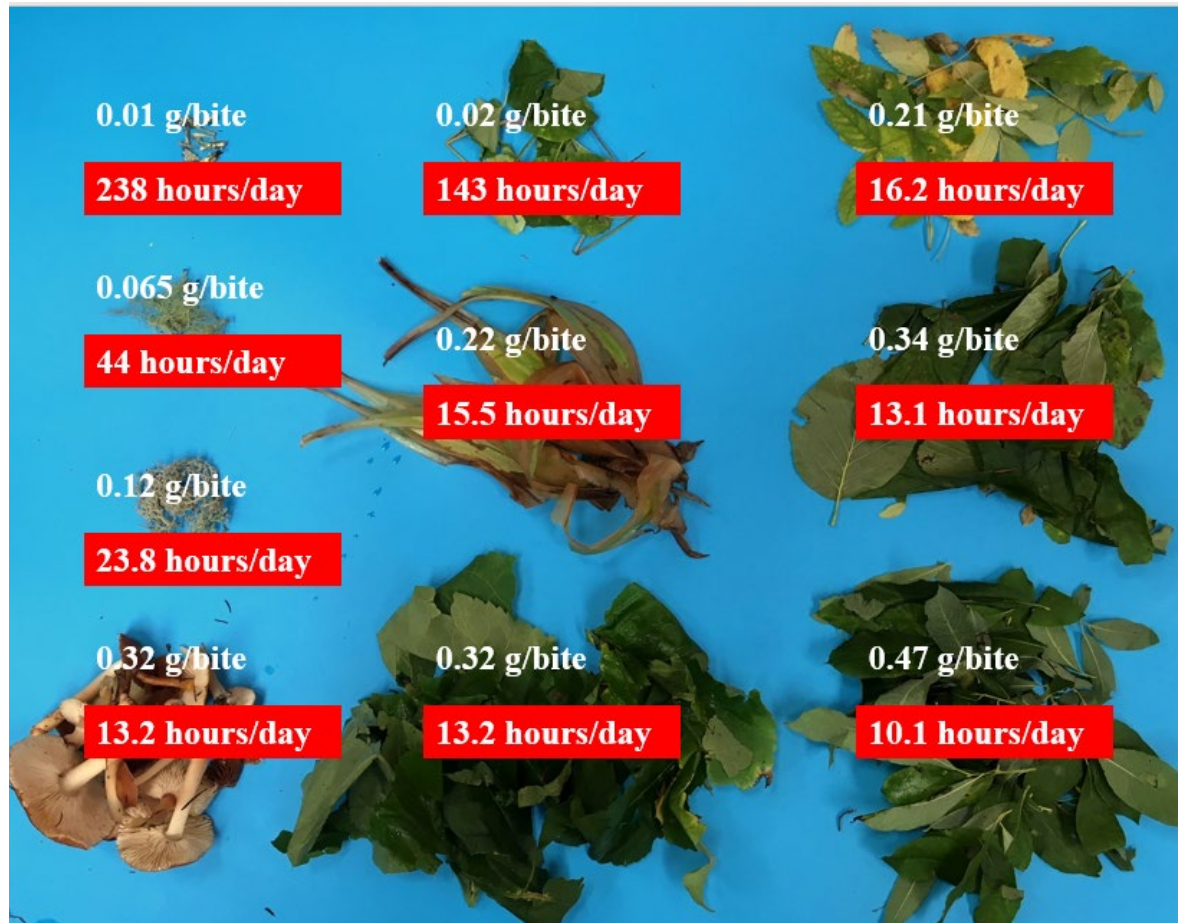


Figure 22. Illustration of bite mass effects on time required for caribou to satisfy daily forage intake requirements. Each pile of vegetation represents 10 bites of a single species ranging from pixie cup lichens (*Cladina* spp) in the top left pile to deciduous shrubs such as willows (*Salix* spp.) in the bottom right pile. The hour estimates are the number of hours caribou would have to spend each day eating only that food at their average biting rate to satisfy daily nutritional requirements. For species such as lichens with very small bite mass (the top 3 piles in the left column), caribou would have little or no chance of satisfying daily forage requirements in a 15-hour foraging day (the approximate upper limit of time spent feeding each day in summer—Cook et al. 2016, Denryter et al. 2022). In addition, at the level of digestible protein in lichens, caribou would be unable to satisfy daily protein requirements no matter the bite mass of lichens.

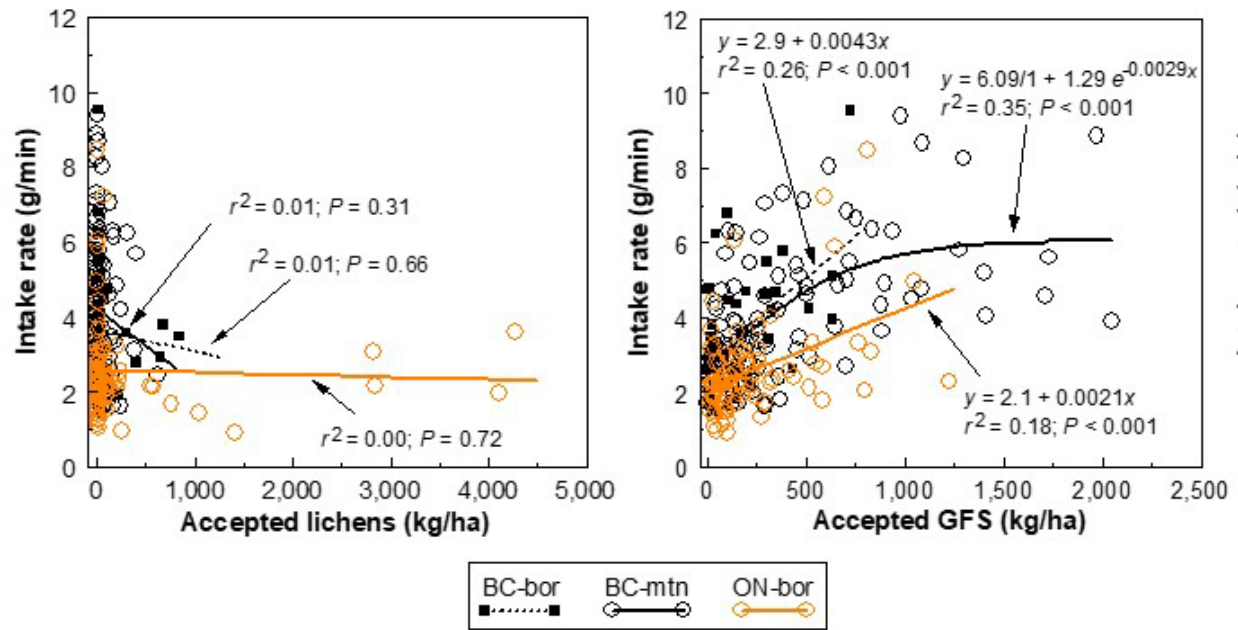


Figure 23. Forage intake rate of caribou as a function of biomass of accepted terrestrial and arboreal hair lichens (left panel) and biomass of accepted grasses, forbs, and deciduous shrubs (combined) at 3 study areas: BC-bor = boreal forest in northeastern British Columbia (BC); BC-mtn = montane area in northeastern BC; and ON-bor = boreal study area in northwestern Ontario. The approximate forage intake rate required to satisfy daily forage consumption requirements is about 5.5 grams per minute for lactating females in summer assuming a 15-hour foraging day and digestible energy levels of 12.1 kJ/g in consumed forage (National Research Council 2007, Denryter et al. 2022).

Appendix A. Forage class (life form group) and caribou selection category of plants sampled in the southern Northwest Territories during summer sampling in 2016, 2019, and 2021.

Scientific Name	Common Name	Forage Class ^a	Diet Selection ^b	<i>n</i> ^c
<i>Alectoria</i> spp.	Witch's hair lichen	Arb. Lichen	Sel	122
<i>Alnus crispa</i>	Green alder	Dec. Shrub	Sel	23
<i>Amelanchier alnifolia</i>	Saskatoon serviceberry	Dec. Shrub	Sel	37
<i>Aster ciliolatum</i>	Fringed blue aster	Per. Forb	Sel	126
<i>Aster</i> spp.	Aster	Per. Forb	Sel	35
<i>Betula papyrifera</i>	Paper birch	Dec. Shrub	Sel	21
<i>Betula pumila</i> ^d	Bog birch	Dec. Shrub	Sel	119
<i>Bryoria</i> spp.	Horsehair lichen	Arb. Lichen	Sel	47
<i>Cladina mitis</i>	Green reindeer lichen	Terr. Lichen	Sel	113
<i>Cladina rangiferina</i>	Grey reindeer lichen	Terr. Lichen	Sel	69
<i>Cladonia</i> spp.	Cup lichen	Terr. Lichen	Sel	139
<i>Cladonia uncialis</i>	Thorn lichen	Terr. Lichen	Sel	9
<i>Elymus innovatus</i>	Hairy wildrye	Grass	Sel	101
<i>Flavocetraria cucullata</i>	Curled snow lichen	Terr. Lichen	Sel	26
<i>Flavocetraria nivalis</i>	Crinkled snow lichen	Terr. Lichen	Sel	8
<i>Lathyrus ochroleucus</i>	Pale vetchling	Per. Forb	Sel	27
Mushroom (various spp.)	Mushroom	Fungus	Sel	107
<i>Populus balsamifera</i>	Cottonwood	Dec. Shrub	Sel	23
<i>Populus tremuloides</i>	Quaking aspen	Dec. Shrub	Sel	88
<i>Rosa acicularis</i>	Prickly wild rose	Dec. Shrub	Sel	175
<i>Salix</i> spp. ^e	Willow	Dec. Shrub	Sel	183
<i>Symphyotrichum boreale</i>	Northern bog aster	Per. Forb	Sel [¥]	7
<i>Taraxacum</i> spp.	Dandelion	Per. Forb	Sel	39
<i>Vaccinium myrtilloides</i>	Velvet-leaved blueberry	Dec. Shrub	Sel	16
<i>Vaccinium uliginosum</i>	Bog blueberry	Dec. Shrub	Sel	11
<i>Achillea millefolium</i>	Common yarrow	Per. Forb	Neu	44
<i>Achillea sibirica</i>	Alpine Yarrow	Per. Forb	Neu [¥]	2
<i>Agropyron</i> spp.	Wheatgrass spp.	Grass	Neu	1
<i>Agropyron trachycaulus</i>	Slender wheatgrass	Grass	Neu [¥]	21
<i>Apocynum androsaemifolium</i>	Fly-trap dogbane	Per. Forb	Neu [¥]	2
<i>Arctostaphylos rubra</i>	Red fruit bearberry	Dec. Shrub	Neu	157
<i>Arctostaphylos uva-ursi</i>	Bearberry	Ev. Shrub	Neu	115
<i>Aster conspicuous</i>	Showy aster	Per. Forb	Neu	1
<i>Bromus richardsonii</i>	Fringed brome	Grass	Neu	9
<i>Castilleja</i> spp.	Paintbrush spp.	Per. Forb	Neu	1
<i>Cetraria islandica</i>	Iceland moss	Terr. Lichen	Neu	42

<i>Cladina stellaris</i>	Reindeer lichen	Terr. Lichen	Neu	48
<i>Cornus stolonifera</i>	Red osier dogwood	Dec. Shrub	Neu	15
<i>Cypripedium calceolus</i>	Lady's slipper orchid	Per. Forb	Neu [¥]	6
<i>Cypripedium passerinum</i>	Sparrow's egg lady's slipper	Per. Forb	Neu [¥]	1
<i>Dryas integrifolia</i>	Entireleaf mountain avens	Per. Forb	Neu	1
<i>Epilobium angustifolium</i>	Fireweed	Per. Forb	Neu	120
<i>Epilobium palustri</i>	Marsh willow-herb	Per. Forb	Neu [¥]	1
<i>Epilobium watsonii</i> , <i>E. ciliatum</i>	Willow herb	Per. Forb	Neu	16
<i>Equisetum arvense</i> , <i>E. pratense</i>	Common/meadow horsetail	Graminoid	Neu	99
<i>Equisetum fluviatile</i>	River horsetail	Graminoid	Neu [†]	30
<i>Equisetum sylvaticum</i>	Wood horsetail	Graminoid	Neu	5
<i>Equisetum variegatum</i>	Variegated horsetail	Graminoid	Neu [†]	3
<i>Eriophorum</i> spp.	Cotton grass	Graminoid	Neu	1
<i>Eriophorum vaginatum</i>	Tussock cottongrass	Graminoid	Neu [†]	9
<i>Galium boreale</i>	Northern bedstraw	Per. Forb	Neu	159
<i>Geum macrophyllum</i>	Large-leaved avens	Per. Forb	Neu	1
<i>Hedysarum boreale</i>	Utah sweetvetch	Per. Forb	Neu [†]	40
<i>Hieracium</i> spp.	Hawkweed	Per. Forb	Neu	10
<i>Juncus</i> spp.	Bulrush	Graminoid	Neu	56
<i>Lonicera dioica</i>	Limber honeysuckle	Dec. Shrub	Neu	22
<i>Mentha arvensis</i>	Wild mint	Per. Forb	Neu [¥]	1
<i>Menyanthes tripholiata</i>	Buckbean	Per. Forb	Neu [¥]	6
<i>Mertensia paniculata</i>	Tall bluebells	Per. Forb	Neu	6
<i>Petasites frigidis</i>	Arctic sweet coltsfoot	Per. Forb	Neu [†]	2
<i>Petasites palmatus</i>	Palmate coltsfoot	Per. Forb	Neu	25
<i>Petasites sagittatus</i>	Sweet coltsfoot	Per. Forb	Neu [†]	38
<i>Poa pratensis</i>	Kentucky bluegrass	Grass	Neu	2
<i>Poa</i> spp.	Meadowgrass	Graminoid	Neu	2
<i>Pyrola uniflora</i>	One-flowered pyrola	Per. Forb	Neu	4
<i>Pyrola virens</i>	Green-flowered wintergreen	Per. Forb	Neu	41
<i>Ribes glandulosum</i>	Skunk currant	Dec. Shrub	Neu	1
<i>Ribes hudsonianum</i>	Northern black currant	Dec. Shrub	Neu	12
<i>Ribes oxycanthoides</i>	Northern gooseberry	Dec. Shrub	Neu	3
<i>Rubus idaeus</i>	Red raspberry	Dec. Shrub	Neu	13
<i>Salix myrtillofolia</i>	Blueberry willow	Dec. Shrub	Neu [†]	52
<i>Senecio</i> spp.	Groundsel	Per. Forb	Neu [†]	4
<i>Shepherdia canadensis</i>	Soapberry	Dec. Shrub	Neu	150
<i>Smilacina trifolia</i>	Three-leaf Solomon's-seal	Per. Forb	Neu	56
<i>Solidago canadensis</i>	Canada goldenrod	Per. Forb	Neu [†]	2
<i>Solidago spathulata</i>	Coast goldenrod	Per. Forb	Neu [¥]	44
<i>Solidago</i> spp.	Goldenrod	Per. Forb	Neu [¥]	5
<i>Usnea</i> spp.	Old man's beard	Arb. Lichen	Neu	117
<i>Vaccinium</i> spp.	Blueberry	Dec. Shrub	Neu	3

<i>Viburnum edule</i>	Highbush cranberry	Dec. Shrub	Neu	60
<i>Vicia americana</i>	American vetch	Per. Forb	Neu	20
<i>Agrostis spp.</i>	Bentgrass	Grass	Avd	4
<i>Andromeda polifolia</i>	Bog rosemary	Ev. Shrub	Avd [†]	35
<i>Anemone canadensis</i>	Canada a anemone	Per. Forb	Avd [¥]	1
<i>Anemone multifida</i>	Pacific anemone	Per. Forb	Avd [¥]	19
<i>Anemone parviflora</i>	Small-flowered anemone	Per. Forb	Avd [¥]	37
<i>Anemone quinquefolia</i>	Wood anemone	Per. Forb	Avd [†]	1
<i>Antennaria microphylla, A. alpina</i>	Pussytoes	Per. Forb	Avd	3
<i>Antennaria pulcherrima</i>	Showy pussytoes	Per. Forb	Avd	14
<i>Arabis spp.</i>	Rockcress	Per. Forb	Avd [¥]	1
<i>Arnica angustifolia</i>	Narrowleaf arnica	Per. Forb	Avd [¥]	1
<i>Arnica spp.</i>	Arnica	Per. Forb	Avd [¥]	1
<i>Blitum capitatum</i>	Strawberry blite	Per. Forb	Avd [¥]	1
<i>Calamagrostis canadensis</i>	Canada bluejoint	Grass	Avd	33
<i>Calamagrostis inexpansa</i>	Northern reedgrass	Grass	Avd [¥]	1
<i>Calamagrostis spp.</i>	Reedgrass	Grass	Avd [†]	6
<i>Campanula rotundifolia</i>	Bluebell bellflower	Per. Forb	Avd [†]	34
<i>Campanula uniflora</i>	Arctic harebell	Per. Forb	Avd [†]	1
<i>Carex rostrata/aquaticilis</i>	Beaked or Water Sedge	Graminoid	Avd	26
<i>Carex spp.</i>	Sedge	Graminoid	Avd	196
<i>Chamaedaphne calyculata</i>	Leatherleaf	Ev. Shrub	Avd	16
<i>Corallorhiza trifida</i>	Early coralroot	Per. Forb	Avd [†]	2
<i>Cornus canadensis</i>	Bunchberry	Per. Forb	Avd	118
<i>Corydalis aurea</i>	Scrambled eggs	Per. Forb	Avd [¥]	5
<i>Danthonia intermedia</i>	Timber oatgrass	Grass	Avd [¥]	2
<i>Deschampsia cespitosa</i>	Tufted hair grass	Grass	Avd [†]	2
<i>Dracocephalum parviflorum</i>	American dragonhead	Per. Forb	Avd [¥]	9
<i>Drosera rotundifolia</i>	Round-leaved sundew	Per. Forb	Avd [†]	1
<i>Empetrum nigrum^f</i>	Black crowberry	Ev. Shrub	Avd	31
<i>Equisetum scirpoides</i>	Dward scouring rush	Graminoid	Avd	126
<i>Erigeron purpuratus</i>	Purple fleabane	Per. Forb	Avd [¥]	1
<i>Fragaria virginiana</i>	Wild strawberry	Per. Forb	Avd	71
<i>Galearis rotundifolia</i>	Roundleaf orchis	Per. Forb	Avd [¥]	6
<i>Galium trifidum</i>	Three-petaled bedstraw	Per. Forb	Avd [†]	7
<i>Galium triflorum</i>	Fragrant bedstraw	Per. Forb	Avd	1
<i>Gentiana spp.</i>	Gentian	Per. Forb	Avd	2
<i>Gentianella propinqua</i>	Four-parted gentian	Per. Forb	Avd [†]	2
<i>Geocaulon lividum</i>	Falso toadflax	Per. Forb	Avd [†]	106
<i>Geranium bicknellii</i>	Bicknell's cranesbill	Per. Forb	Avd [¥]	9
<i>Goodyeara repens</i>	Dwarf rattlesnake-plantain	Per. Forb	Avd [†]	10
<i>Hordeum jubatum</i>	Squirreltail barley	Grass	Avd	2
<i>Juniperus communis</i>	Common juniper	Ev. Shrub	Avd [†]	63

<i>Juniperus horizontalis</i>	Creeping juniper	Ev. Shrub	Avd [‡]	66
<i>Kalmia polifolia</i>	Bog laurel	Ev. Shrub	Avd	1
<i>Larix laricina</i>	Tamarack	Conifer	Avd	71
<i>Ledum groenlandicum</i>	Bog Labrador tea	Ev. Shrub	Avd	79
<i>Linnaea borealis</i>	Twinflower	Ev. Shrub	Avd	161
<i>Listera cordata</i>	Heart-leaved twayblade	Per. Forb	Avd	1
<i>Lobelia kalmii</i>	Kalm's lobelia	Per. Forb	Avd [‡]	1
<i>Lycopodium complanatum</i>	Northern ground-cedar	Moss	Avd	2
<i>Mimulus guttatus</i>	Seep monkey flower	Per. Forb	Avd [‡]	1
<i>Mitella nuda</i>	Naked miterwort	Per. Forb	Avd	51
<i>Myrica gale</i>	Sweetgale	Dec. Shrub	Avd [‡]	38
<i>Oryzopsis asperfolia</i>	Rough-leaved rice grass	Grass	Avd [‡]	14
<i>Parnassia fimbrata</i>	Fringed grass	Per. Forb	Avd	1
<i>Parnassia palustris</i>	Marsh grass of Parnassus	Per. Forb	Avd [‡]	25
<i>Pedicularis spp.</i>	Lousewort	Per. Forb	Avd	9
<i>Peltigera spp.</i>	Peltigera	Peltigera	Avd	113
<i>Picea glauca</i>	White spruce	Conifer	Avd	46
<i>Picea mariana</i>	Black spruce	Conifer	Avd	161
<i>Pinguicula vulgaris</i>	Common butterwort	Per. Forb	Avd [‡]	2
<i>Pinus banksiana</i>	Jack pine	Conifer	Avd [‡]	59
<i>Piptatherum pungens</i>	Mountain rice grass	Grass	Avd [‡]	22
<i>Platanthera spp.</i>	Orchid	Per. Forb	Avd	47
<i>Potentilla fruticosa</i>	Shrubby cinquefoil	Dec. Shrub	Avd [‡]	131
<i>Potentilla palustris</i>	Marsh cinquefoil	Per. Forb	Avd [‡]	17
<i>Pyrola asarifolia, P. grandiflora</i>	Pink/Arctic wintergreen	Per. Forb	Avd	82
<i>Pyrola secunda</i>	Sidebells wintergreen	Per. Forb	Avd	92
<i>Ribes lacustre</i>	Black swamp gooseberry	Dec. Shrub	Avd	6
<i>Rubus arcticus</i>	Artic raspberry	Per. Forb	Avd [‡]	89
<i>Rubus chamaemorus</i>	Cloudberry	Per. Forb	Avd	16
<i>Rubus pubescens</i>	Trailing raspberry	Dec. Shrub	Avd	30
<i>Scutellaria galericulata</i>	Common skullcap	Per. Forb	Avd [‡]	2
<i>Senecio pauperculus</i>	Balsam groundsel	Per. Forb	Avd [‡]	16
<i>Sium suave</i>	Water parsnip	Per. Forb	Avd [‡]	1
<i>Sonchus arvensis</i>	Field milk thistle	Per. Forb	Avd [‡]	1
<i>Stellaria spp.</i>	Stitchwort	Per. Forb	Avd [‡]	7
<i>Stereocaulon paschale</i>	Foam lichen	Terr. Lichen	Avd	10
<i>Thalictrum occidentale</i>	Western meadow-rue	Per. Forb	Avd	1
<i>Tofieldia glutinosa</i>	Western false asphodel	Per. Forb	Avd [‡]	11
<i>Tofieldia pusilla</i>	Small false asphodel	Per. Forb	Avd [‡]	1
<i>Triglochin maritima</i>	Seaside arrowgrass	Graminoid	Avd [‡]	14
<i>Vaccinium caespitosum</i>	Dwarf blueberry	Dec. Shrub	Avd	1
<i>Vaccinium oxycoccos</i>	Bog cranberry	Ev. Shrub	Avd	24
<i>Vaccinium vitis-idaea</i>	Lingonberry	Ev. Shrub	Avd	123

<i>Viola spp.</i>	Violets	Per. Forb	Avd	90
<i>Zygadensus elegans</i>	Mountain death camas	Per. Forb	Avd [‡]	54

^a Abor. Lichen = arboreal hair lichens; Dec. Shrub = deciduous shrubs; Ev. Shrub = evergreen shrubs; Per. Forb = perennial forbs; Terr. Lichen = terrestrial lichens.

^b Diet selection category as determined from forage selection studies over 3 years using captive, trained caribou in alpine, mountain, and boreal ecosystems of northeastern British Columbia (see Denryter et al. 2017). 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.

^c Number of macroplots in which the plant species occurred (out of 238 macroplots sampled in 2016, 2019, excluding plots sampled in 2021).

^d In 2016, we failed to separate *Betula pumila* and *B. glandulosa* but did in 2019. Both are present in our study area but *B. pumila* was by far more common (*B. glandulosa* occurred in only 2 macroplots in 2019). We grouped both together, and refer to these as *B. pumila*, recognizing that in a few cases, *B. glandulosa* was also present in our sample.

^e Not all species of willow were selected by caribou. Catkins were usually not available for identification in the field, but based on leaf and twig characteristics, we identified *Salix arbusculoides*, *S. bebbiana*, *S. drummondiana*, *S. scouleriana*, *S. alaxensis*, *S. maccalliana* as species accepted by caribou, whereas *S. reticulata* and *S. barclayii* were avoided.

^f Caribou consumed the berries of *Empetrum nigrum* in late summer, but almost never consumed the foliage of these plants. Because the biomass of this species is largely foliage, we treated biomass of *E. nigrum* as "avoided" despite a neutral electivity score driven by the presence of berries in some diets.

[‡] 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, and from selection patterns among similar plants (e.g., same genus).

[‡] New species encountered in the Northwest Territories; thus, we based the selection category on genus or family rather than on caribou foraging trials.

Appendix B. Forage prediction models by vegetation type for the southern Northwest Territories based on field data collected in summer 2016 and 2019.

Variable ^a	Vegetation ^b type	Equation ^c	<i>n</i>	<i>S_{y-x}</i> ^d	<i>R</i> ² _{adj} ^e	<i>P</i> ^f
GFS	Bog	$y = 64.8 / (1 - 0.856 \times \exp(-0.0260 \times \text{Age}))$	45	115.7	0.37	≤ 0.001
GFS	Fen	$y = 357 - 0.863 \times \text{Age}$	43	193.7	0.08	0.064
GFS	Upl-BS	$y = 151 - 16.4 \times D + 0.691 \times T_Age^g + 2.25 \times Hrd$	84	79.4	0.56	≤ 0.001
GFS	Upl-WS	$y = 638 - 87.0 \times D + 0.501 \times T_Age^i + 0.461 \times T_CC^j + 2.12 \times Hrd$	57	220.3	0.57	≤ 0.001
GFS	Marsh	$y = 105.9$	8	38.6	na	na
GFS	Sphagnum bog	$y = 4.82$	6	3.39	na	na
Horsetails	Bog	$y = 33.1$	45	7.58	na	na
Horsetails	Fen	$y = 1.22 + 0.172 \times \text{Age} + 1.71 \times CC$	43	33.2	0.19	0.005
Horsetails	Upl-BS	$y = 0.80$	84	0.563	na	na
Horsetails	Upl-WS	$y = 3.57$	57	1.54	na	na
Horsetails	Marsh	$y = 21.9$	8	18.1	na	na
Horsetails	Sphagnum bog	$y = 0.0$	6	0.0	na	na
Lichens	Bog ^k	$y = -59.5 + 0.784 \times T_BA^l + 0.410 \times T_TD^m$	44	351.3	0.47	≤ 0.001
Lichens	Bog ⁿ	$y = (74.7 + 9.63 \times BA) / (1 - 0.365 \times BA + 0.0379 \times BA^2)$	44	368.4	0.46	≤ 0.001
Lichens	Bog ^o	$y = 140 \times 0.865^{CC} \times CC^{1.25}$	44	434.7	0.23	≤ 0.001
Lichens	Fen	$y = 20.9$	43	10.5	na	na
Lichens	Upl-BS	$y = -35.5 + 3.43 \times \text{Age} - 0.622 \times T_Ht^p - 0.526 \times CCxHt$	84	316.0	0.30	≤ 0.001

Lichens	Upl-WS	$y = 15.2$	57	6.5	na	na
Lichens	Marsh	$y = 0.00$	8	0.001	na	na
Lichens	Sphagnum bog	$y = 64.7$	6	38.8	na	na
Mushrooms	Bog	$y = -0.97 + 1.01 \times T_D^q + 0.295 \times CC - 0.0199 \times CCxHt$	45	3.25	0.50	≤ 0.001
Mushrooms	Fen	$y = 0.000000244 \times D^{(0.803 \times D)}$	43	2.62	0.20	0.002
Mushrooms	Upl-BS	$y = -0.494 + 0.899 \times T_D^r + 0.630 \times T_Age^s - 0.0082 \times HrdxHt$	84	4.51	0.37	≤ 0.001
Mushrooms	Upl-WS	$y = 2.656e-15 \times (D + 0.306)^{15.6}$	56	4.20	0.20	≤ 0.001
Mushrooms	Marsh	$y = 0.00$	8	0.0	na	na
Mushrooms	Sphagnum bog	$y = 0.00$	6	0.0	na	na

^a Units for response variables are kg/ha and codes are: GFS = accepted grasses, forbs, and deciduous shrubs; horsetails = accepted horsetail species in the genus *Equisetum*; lichens = accepted arboreal and terrestrial lichens; and mushrooms = accepted mushrooms.

^b Codes for vegetation types are Bog, Fen, Marsh, Sphagnum bog, Upl-BS (upland black spruce), and Upl-WS (upland white spruce).

^c Codes for regression covariates are: D = date (month + (dy / 31.1)); Age = stand age in years; CC = overstory canopy cover (%); BA = basal area (m²/ha); Ht = mean stand height (m); TD = tree (stem) density (number/ha); Hrd = hardwoods (percent of total stems that are hardwood species); $AgexHt$ = interaction between stand age and stand height; $CCxHt$ = interaction between overstory canopy cover and stand height; and $HrdxHt$ = interaction between hardwood composition and stand height.

^d Standard error of the estimate. Standard error of the means is presented when only the means are presented in place of simple or multiple regressions.

^e Adjusted coefficient of determination. Unadjusted coefficients of determination are presented for simple regressions.

^f Probability of F test for overall regression equation.

^g $T_Age = \exp(6.71 - (1.477 / Age) - 0.486 \times \log(Age))$ ($r^2 = 0.42$, $F_{3,81} = 90.16$; $P \leq 0.001$), a non-linear equation used to linearize the relation between stand age and biomass of accepted grasses, forbs, and deciduous shrubs, where Age = stand age in years.

^h $T_TD = 259 / (1 + (TD / 2448)^{0.6513})$ ($r^2 = 0.22$, $F_{3,81} = 60.32$; $P \leq 0.001$), a non-linear equation used to linearize the relation between tree density and biomass of accepted grasses, forbs, and deciduous shrubs, where TD = density of tree stems (no/ha).

ⁱ $T_Age = \exp(9.25 - (5.12 / Age) - 0.915 \times \log(Age))$ ($r^2 = 0.51$, $F_{3,54} = 56.15$; $P \leq 0.001$), a non-linear equation used to linearize the relation between stand age and biomass of accepted grasses, forbs, and deciduous shrubs, where Age = stand age in years.

^j $T_CC = 208 / (1 - 0.743 \times \exp(-0.0745 \times CC))$ ($r^2 = 0.47$, $F_{3,54} = 50.3$; $P \leq 0.001$), a non-linear equation used to linearize the relation between overstory canopy cover and biomass of accepted grasses, forbs, and deciduous shrubs, where CC = percent overstory canopy cover.

^k Best model if data for both covariates are available.

^l $T_BA = (74.7 + 9.63 \times BA) / (1 - 0.365 \times BA + 0.0379 \times BA^2)$ ($r^2 = 0.46$, $F_{4,40} = 17.99$; $P \leq 0.001$), a non-linear equation used to linearize the relation between basal area and biomass of accepted arboreal and terrestrial lichens, where BA = basal area in m² per ha.

^m $T_TD = (-21.8 + 0.76 \times TD) / (1 - 0.000769 \times TD + 0.000000661 \times TD^2)$ ($r^2 = 0.31$, $F_{4,40} = 12.14$; $P \leq 0.001$), a non-linear equation used to linearize the relation between tree density and biomass of accepted arboreal and terrestrial lichens, where TD = density of tree stems (no/ha).

ⁿ Second best model if data for BA is available.

^o Third best model if data for neither BA nor TD is available.

^p $T_Ht = 3.83 \times 0.620^{Ht} \times Ht^{4.06}$ ($r^2 = 0.14$, $F_{3,79} = 13.46$; $P \leq 0.001$), a non-linear equation used to linearize the relation between stand height and biomass of accepted arboreal and terrestrial lichens, where Ht = stand height in m.

^q $T_D = 9.04 / (1 + (D / 9.14)^{-1137})$ ($r^2 = 0.44$, $F_{3,43} = 14.41$; $P \leq 0.001$), a non-linear equation used to linearize the relationship between sampling date and biomass of accepted mushrooms, where D = date (month + (day/31.1)).

^r $T_D = 10.3 / (1 + (D / 9.06)^{-71.3})$ ($r^2 = 0.32$, $F_{3,81} = 19.66$; $P \leq 0.001$), a non-linear equation used to linearize the relationship between sampling date and biomass of accepted mushrooms, where D = date (month + (day/31.1)).

^s $T_Age = 0.0338 \times 0.940^{Age} \times Age^{2.12}$ ($r^2 = 0.11$, $F_{3,81} = 8.87$; $P \leq 0.001$), a non-linear equation used to linearize the relation between stand age and biomass of accepted mushrooms, where Age = stand age in years.

Appendix C. Accepted lichen biomass in relation to forest overstory development.

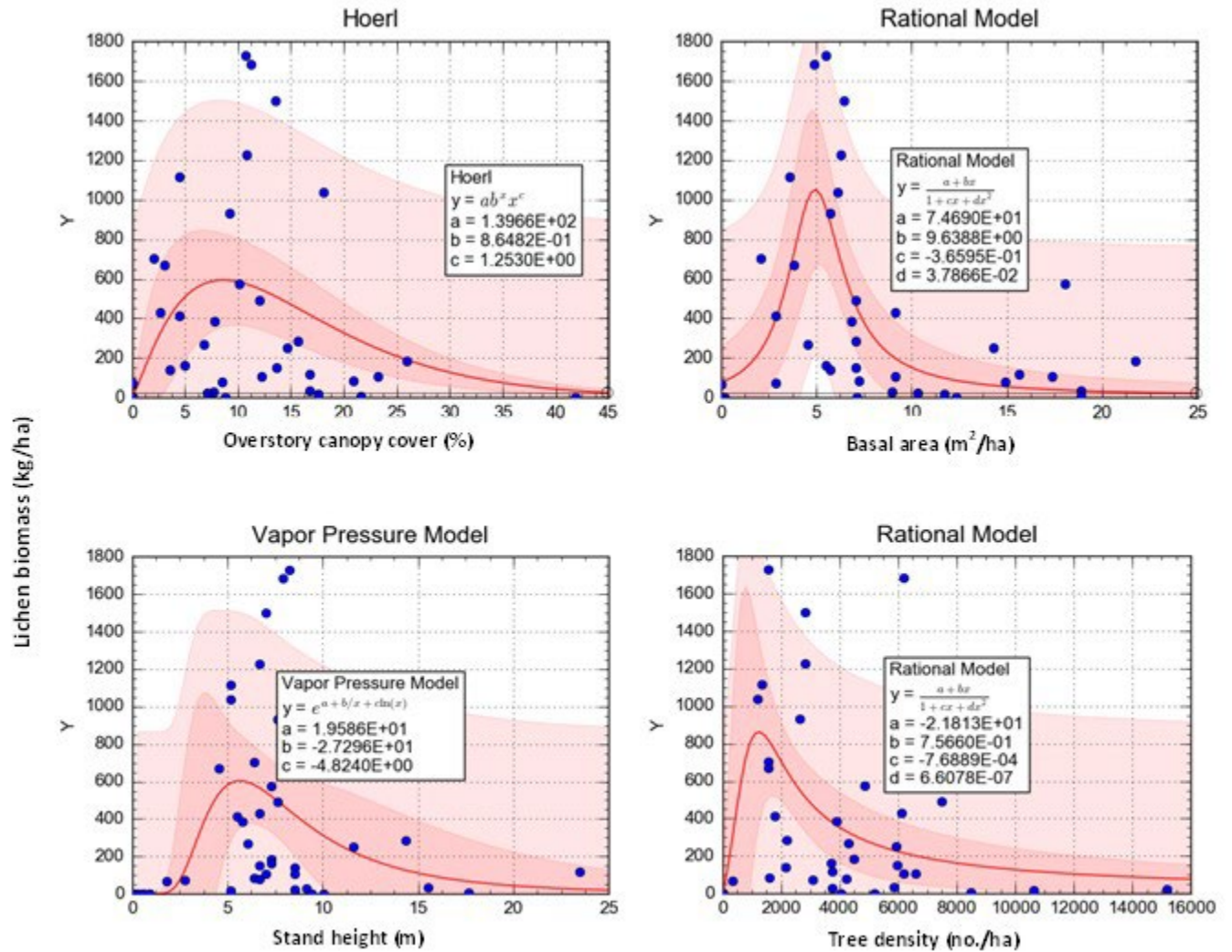


Figure C1. Non-linear relationships between four forest overstory characteristics and biomass of accepted arboreal and terrestrial lichens in bog ecosystems in the southern Northwest Territories, summer 2016 and 2019.

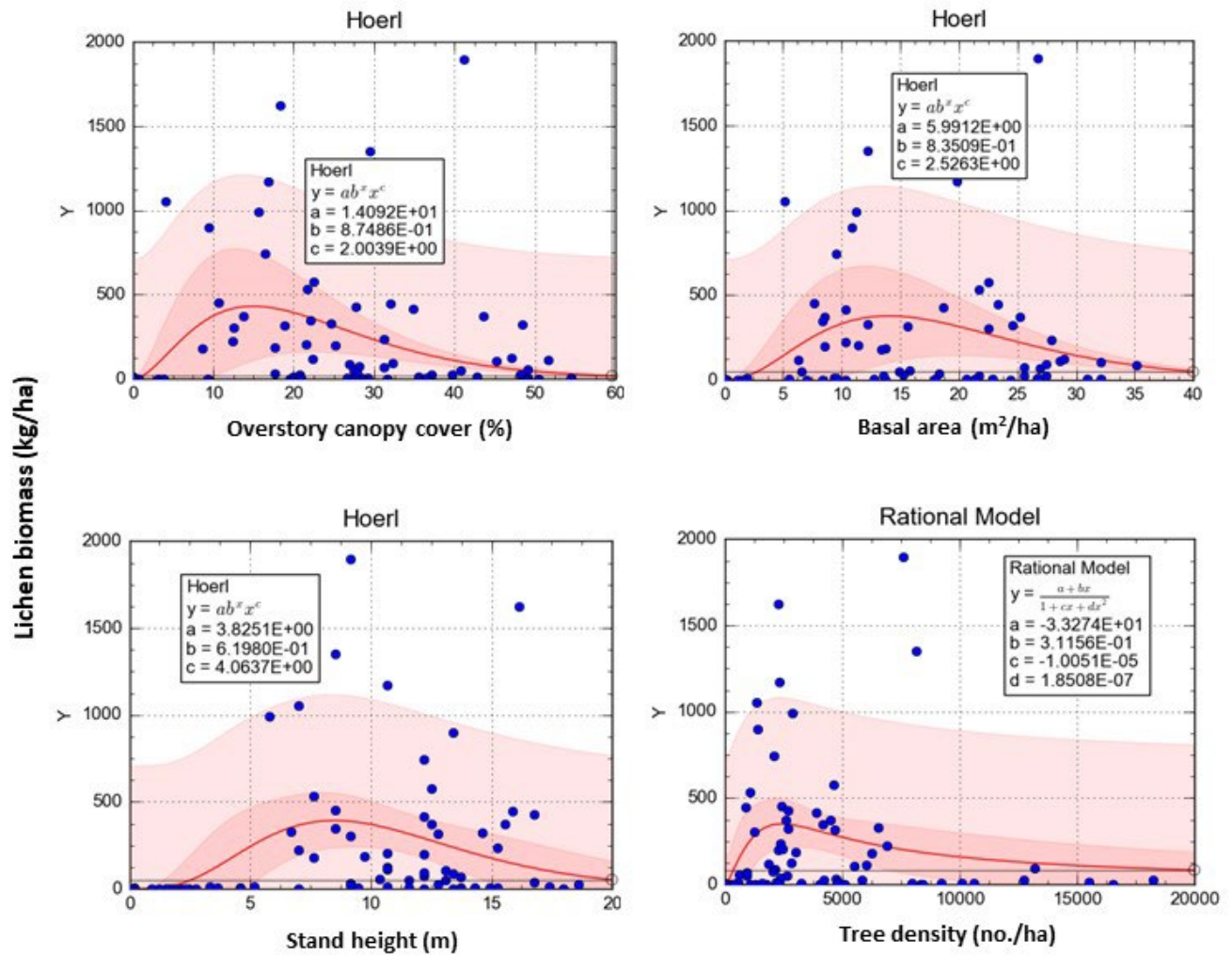


Figure C2. Non-linear relationships between four forest overstory characteristics and biomass of accepted arboreal and terrestrial lichens in upland black spruce ecosites in the southern Northwest Territories, summer 2016-2019.

Appendix D. Models to predict seven caribou nutrition variables based on forage and overstory conditions from captive caribou studies during summer in northeastern British Columbia, 2013-2016 (e.g., Denryter et al. 2017, 2022). Of these, four equations (bolded text) were used for foodscape development and include digestible energy intake per minute (kJ of DE/min), digestible protein intake (g of DP/min), concentration of digestible energy in caribou diets (kJ of DE/g of consumed forage), and digestible protein in caribou diets (g of DP/100 g of consumed forage, i.e., %) and are bolded below. For digestible energy intake and concentration of digestible energy in upland sites and digestible energy intake on wetland sites, we developed and used for prediction alternative equations, one with lichens included as a predictor covariate and one without lichens as a predictor covariate. Equations were developed to predict for upland and wetland ecosites separately by Cook et al. (*in prep*).

Variable ^a	Vegetation ^b type	Equation ^c	<i>n</i>	<i>S_{y-x}</i> ^d	<i>R</i> ² _{adj} ^e	<i>P</i> ^f
DM/min (g)	Upland	$y = 6.27 - 0.621 \times D + 0.751 \times T_GFS^g + 0.009 \times Lichens + 0.010 \times Horsetails$	51	1.1	0.63	≤ 0.001
DM/min (g)	Upland	$y = 0.77 + 0.736 \times T_GFS^g + 0.012 \times Horsetails$	51	1.3	0.49	≤ 0.001
DE/min (kJ)	Upland	$y = 92.1 - 9.12 \times D + 0.707 \times T_GFS^h + 0.116 \times Lichens + 0.145 \times Horsetails$	51	13.4	0.66	≤ 0.001
DE/min (kJ)	Upland	$y = 9.87 + 0.707 \times T_GFS^h + 0.173 \times Horsetails$	51	16.1	0.50	≤ 0.001
DE/day (MJ)	Upland	$y = 59.6 - 5.73 \times D + 0.713 \times T_GFS^i + 0.085 \times Lichens + 0.097 \times Horsetails$	47	11.0	0.60	≤ 0.001
DE/day (MJ)	Upland	$y = 8.10 + 0.722 \times T_GFS^i + 0.110 \times Horsetails$	47	12.8	0.46	≤ 0.001
DDE (kJ/g)	Upland	$y = 17.2 - 0.585 \times D + 0.0023 \times Lichens$	51	0.4	0.41	≤ 0.001
DDE (kJ/g)	Upland	$y = 16.5 - 0.507 \times D$	51	0.5	0.34	≤ 0.001
DP/min (g)	Upland	$y = 0.85 - 0.084 \times D + 0.468 \times T_GFS^j + 0.0018 \times Horsetails$	51	0.1	0.76	≤ 0.001
DP/day (g)	Upland	$y = 669 - 65.4 \times D + 0.435 \times T_GFS^k + 1.316 \times Horsetails$	47	83.0	0.73	≤ 0.001
DDP (%)	Upland	$y = 13.5 - 1.33 \times D + 0.655 \times T_GFS^l + 0.015 \times Horsetails$	51	1.8	0.61	≤ 0.001

DM/min (g)	Wetland	$y = 1.711 + 0.0068 \times GFS + 0.0022 \times Lichens + 0.0090 \times Horsetails$	17	0.6	0.55	0.004
DM/min (g)	Wetland	$y = 10.4 - 0.871 \times D + 0.0036 \times GFS$	17	0.7	0.49	0.004
DE/min (kJ)	Wetland	$y = 24.1 + 0.079 \times GFS + 0.021 \times Lichens$	17	9.1	0.46	0.005
DE/min (kJ)	Wetland	$y = 137.3 - 11.9 \times D + 0.053 \times GFS$	17	8.3	0.56	≤ 0.001
DE/day (MJ)	Wetland	$y = 16.6 + 0.060 \times GFS + 0.021 \times Lichens + 0.115 \times Horsetails$	17	5.4	0.59	0.002
DE/day (MJ)	Wetland	$y = 95.8 - 7.79 \times D + 0.027 \times GFS$	17	6.5	0.40	0.011
DDE (kJ/g)	Wetland	$y = 11.7 + 0.0022 \times GFS - 0.024 \times CC + 0.013 \times (CC \times Mushrooms)$	18	0.1	0.46	0.008
DP/min (g)	Wetland	$y = 1.253 - 0.131 \times D + 0.00065 \times GFS$	18	0.1	0.66	≤ 0.001
DP/day (g)	Wetland	$y = 932 - 96.1 \times D + 0.416 \times GFS$	18	63.3	0.57	≤ 0.001
DP (%)	Wetland	$y = 20.2 - 2.028 \times D + 0.012 \times GFS + 0.311 \times Mushrooms$	18	2.1	0.55	0.003

^a Codes are: DM = dry matter of forage (i.e., forage intake rate); DE = dietary digestible energy; DP = dietary digestible protein.

^b Upland = mountain and boreal upland black-spruce and upland black-spruce/white-spruce ecosites; Wetland = Bog, Fen, Marsh ecosites (see Appendix D for ecosite descriptions).

^c Codes for regression covariates are: D = date, calculated as $D = \text{month} + (\text{day}/31.1)$, e.g., 15 July = 7.5; GFS = accepted grasses, forbs, and deciduous shrubs (kg/ha); horsetails = accepted horsetail species in the genus *Equisetum* (kg/ha); lichens = accepted arboreal and terrestrial lichens (kg/ha); mushrooms = accepted mushrooms (kg/ha); CC = overstory canopy cover (%); $CC \times Mushrooms$ = interaction between overstory canopy cover and accepted mushrooms.

^d Standard error of the estimate. Standard error of the means are presented when only the means are presented in place of simple or multiple regressions.

^e Adjusted coefficient of multiple determination.

^f Probability of F -test for overall regression equation.

^g $T_GFS = 7.113 / (1 + 1.335 \exp^{-0.002023 \times GFS})$ ($r^2 = 0.41$, $P \leq 0.001$), a non-linear equation used to linearize the relation between DM intake rate (g/min) and GFS (biomass of accepted grasses, forbs, and deciduous shrubs).

^h $T_GFS = 83.082175 / (1 + 1.367769 \exp^{-0.002354 * GFS})$ ($r^2 = 0.40$, $P \leq 0.001$), a non-linear equation used to linearize the relation between DE/minute (k) and *GFS* (biomass of accepted grasses, forbs, and deciduous shrubs).

ⁱ $T_GFS = 65.694747 / (1 + 1.292931 \exp^{-0.002523 * GFS})$ ($r^2 = 0.40$, $P \leq 0.001$), a non-linear equation used to linearize the relation between DE/day (MJ) and *GFS* (biomass of accepted grasses, forbs, and deciduous shrubs).

^j $T_GFS = 0.0125 * GFS^{0.507792}$ ($r^2 = 0.49$, $P \leq 0.001$), a non-linear equation used to linearize the relation between DP/minute and *GFS* (biomass of accepted grasses, forbs, and deciduous shrubs).

^k $T_GFS = 10.950484 * GFS^{0.499975}$ ($r^2 = 0.51$, $P \leq 0.001$), a non-linear equation used to linearize the relation between DP/day and *GFS* (biomass of accepted grasses, forbs, and deciduous shrubs).

^l $T_GFS = 7.194745 / (1 + (GFS/75.117102)^{-0.8263})$ ($r^2 = 0.44$, $P \leq 0.001$), a non-linear equation used to linearize the relation between DP and *GFS* (biomass of accepted grasses, forbs, and deciduous shrubs).

Appendix E. Maps of caribou locations.

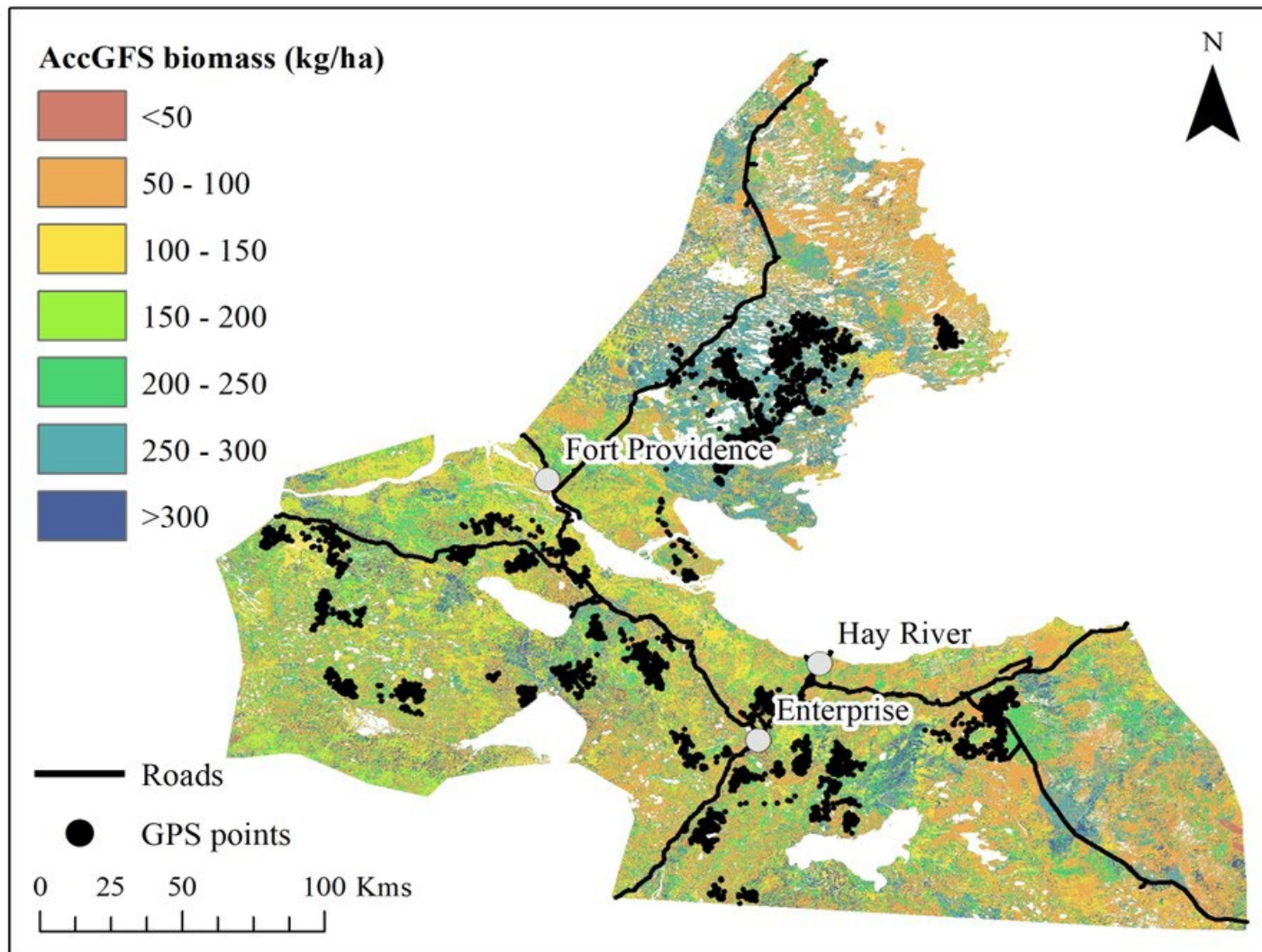


Figure E1. Caribou locations based on global positioning system telemetry plotted with predicted biomass of accepted grasses, forbs, and deciduous shrubs during summer in the southern Northwest Territories.

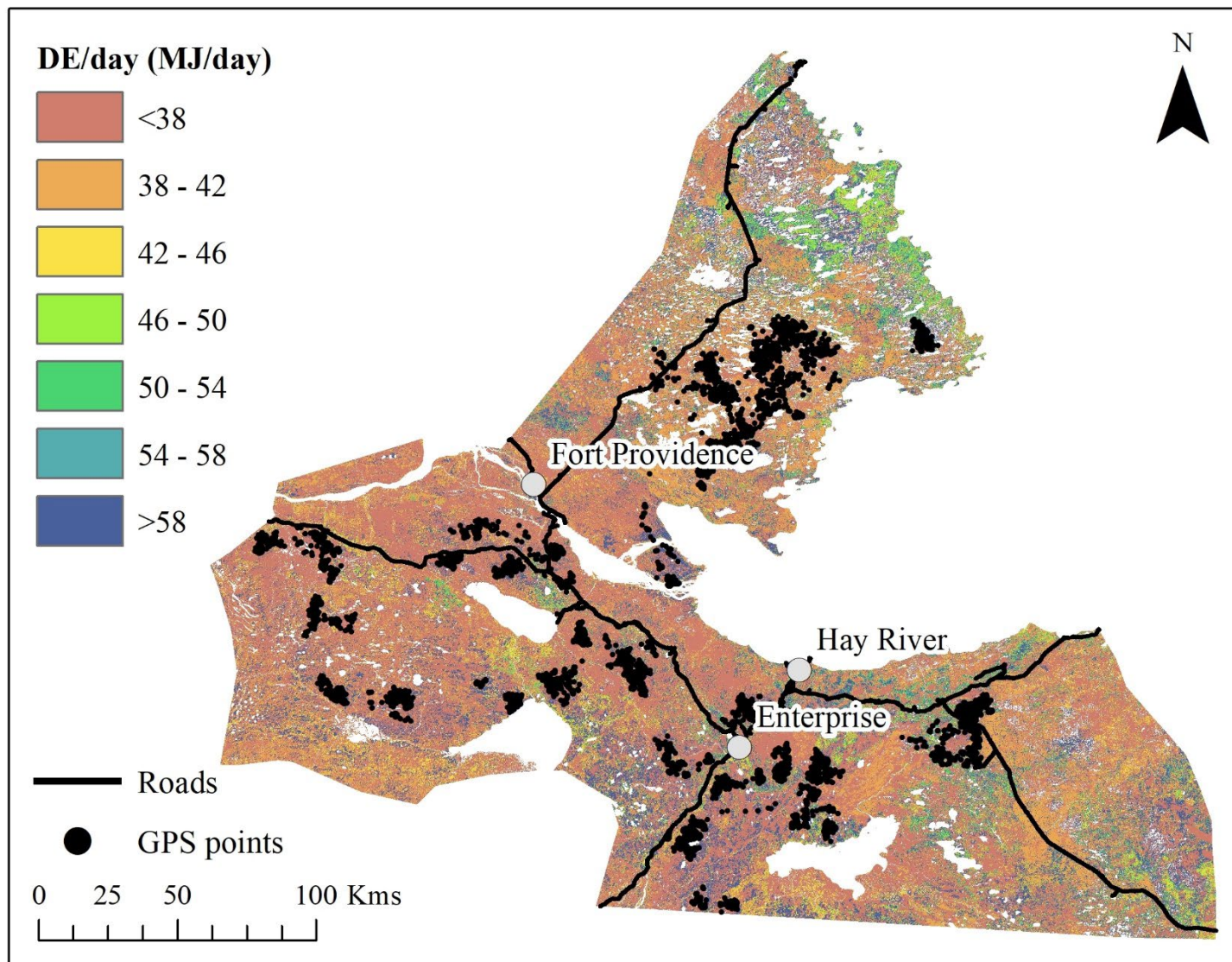


Figure E1. Caribou locations based on global positioning system telemetry plotted with predicted daily intake of digestible energy for adult female caribou with a calf during summer (15 August) in the southern Northwest Territories.