

## Changes in vegetative cover on Western Arctic Herd winter range from 1981 to 2005: potential effects of grazing and climate change

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**Abstract:** The population of the Western Arctic Herd, estimated at 490 000 caribou (*Rangifer tarandus granti*) in 2003, is at its highest level in 30 years. Twenty permanent range transects were established in the winter range of the Western Arctic Herd in 1981 to assess the impacts of grazing. These transects were revisited in 1995 and 1996 (1995/96). Only 18 of the transects were re-located, so an additional 7 transects were established in 1996. In 2005, all 25 remaining transects were revisited. Lichen coverage dropped by a relative 45.1% between 1981 and 1995/96 and by an additional relative 25.6% between 1995/96 and 2005. There was a significant decline in primary forage lichens between 1995/96 and 2005. Caribou use was greater in areas with high lichen abundance. Graminoid cover increased by a relative 118.4% from 1981 to 1995/96 and again by a relative 26.1% from 1995/96 to 2005. Shrub cover increased during the study whereas forb cover declined. The decline in lichen abundance on the winter range of the Western Arctic Herd over 24 years is an index of caribou habitat condition. The observed changes in vegetation cover can be attributed to caribou grazing, fire, and possibly global climate change. Continued declines in lichen cover could lead to population declines within the herd, range shifts, or both.

**Key words:** Alaska, assessment, habitat, lichens, monitoring, *Rangifer tarandus granti*, Western Arctic Herd, winter range.

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### Introduction

The Western Arctic Herd (WAH), estimated at 490 000 individuals in 2003 by photocensus (Dau, 2005), is the largest caribou (*Rangifer tarandus granti*) herd in the region and is arguably the most important subsistence resource in northwestern Alaska. Approximately 15 000 animals are harvested annually from the herd (Dau, 2003), yielding some 500 000 kg of meat (Valkenburg, 1994). There are 40 villages that utilize the herd within the WAH's annual range, which covers about a ¼ of Alaska (Fig. 1). The status of the herd is not only of great importance to subsistence hunters and rural communities, but to sport hunters, recreationists, conservationists, biologists, land managers, and reindeer (*Rangifer t. tarandus*) herders as well.

Reindeer herding has occurred on the Seward Peninsula since 1891 (Stern *et al.*, 1980; Swanson & Barker,

1992). The number of reindeer on the Seward Peninsula peaked in 1932 at around 127 000 and soon after there were signs of serious range deterioration (Stern *et al.*, 1980; Swanson & Barker, 1992). The herding industry continues on the Seward Peninsula to this day. In 1981, permanent range transects were deployed in the Buckland Valley (Fig. 2), an area that potentially could have had both caribou and reindeer (Adams & Connery, 1981). These transects were deployed in an effort to monitor the effects of grazing, potentially by both caribou and reindeer, on winter range.

Between 1970 and 1976, the WAH experienced a dramatic crash in which the population plummeted from 242 000 to 75 000 individuals (Dau, 2003). From this low point, the herd rebounded quickly, growing at a rate of 13% annually until 1990 (Dau, 2003). The WAH has continued to grow, albeit at a

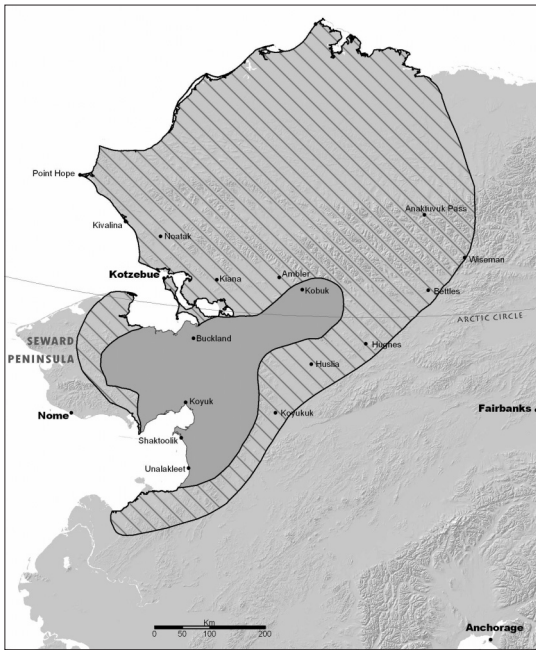


Fig. 1. Range of the Western Arctic Herd, 1981 – 2005, northwestern Alaska. Year-round distribution is hatched and core winter range is colored dark gray. Distribution is based on Alaska Department of Fish and Game data.

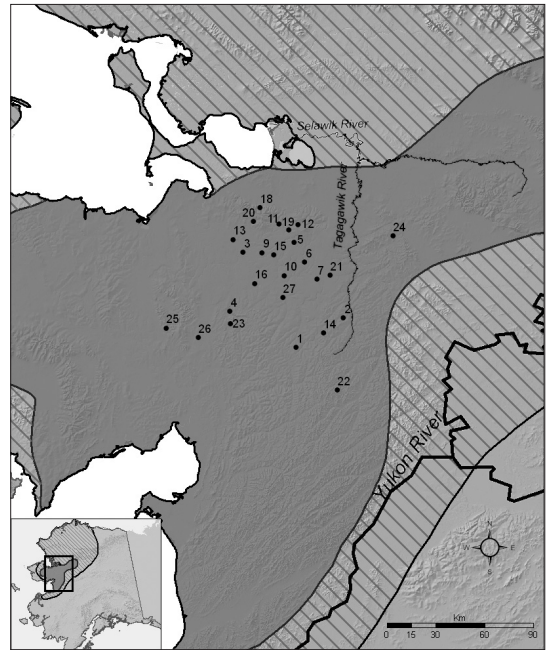


Fig. 2. Locations of the permanent range transects in the Buckland Valley, Seward Peninsula, Alaska. Transects 1 – 20 were deployed in 1981 and 21 – 27 in 1996. Transects #8 and #17 were not relocatable after 1981.

much more measured pace, until 2003 (Dau, 2005). As the WAH grew and expanded its range, the free-ranging reindeer would intermingle and depart their range with caribou as they migrated out in the spring, crippling the reindeer industry (Bader & Finstad, 2001). Since the range transects were deployed, only caribou have utilized the Buckland Valley, which they have done regularly over the years (Davis & Valkenburg, 1978; Davis *et al.*, 1982)

As the herd has grown, its winter range has concurrently expanded into new areas as well (Dau, 2005). The Buckland River valley, however, continues to be within the core winter range of the herd (Dau, 2003). The large size of the WAH has precipitated speculation about possible overgrazing of its range and when the inevitable decline of the herd will take place. The density ( $1.35/\text{km}^2$ ) of caribou on the WAH's range in 2003 is 25% greater than the density ( $1.08/\text{km}^2$ ) Messier (1988) thought to be excessive for the George River Herd in northern Quebec. These concerns have been magnified by recent reports of winter die-offs that have been linked to poor nutritional condition associated with severe winter weather (Dau, 2005) and the rapid decline of the Mulchatna Caribou Herd in southwestern Alaska from 1996 to 2004 (Hinkes *et al.*, 2005; Woolington, 2005). The contribution of range conditions to population

dynamics has frequently been a subject of controversy, but actual studies of range conditions are very limited.

Our goals for the range monitoring study were 3-fold. First, we wanted to periodically assess range conditions and utilization. Second, we sought to identify changes in winter range over the 3 different decades for which we have quantitative range data. Finally, we hoped to assess changes in range condition in terms of overuse/community type (see van der Wal, 2006) and determine if these changes may have implications for the population status of the WAH.

## Material and methods

### Study area

Annually, the WAH ranged over  $363\,000\text{ km}^2$  of northwestern Alaska ( $63^\circ$  to  $71^\circ\text{N}$  and from  $148^\circ$  to  $166^\circ\text{W}$ ; Fig. 1; Davis *et al.*, 1982; Dau, 2003). Although individual members of the WAH can be found across a broad swath of northwest Alaska, the Buckland River valley has been and continues to be in the core winter range (Davis & Valkenburg, 1978; Dau, 2003). The study area encompassed the entire Buckland River drainage, but also extended into surrounding uplands to the north, south, and southeast (Selawik Hills, Granite Mountain, and Nulato Hills, respectively).

The study area was dominated by treeless tussock tundra (primarily *Eriophorum vaginatum*), but contained rolling hills up to 900 m and large riparian corridors. Fruticose lichens (*Cladina* spp.), preferred caribou winter forage, mosses (primarily *Sphagnum* spp.) and shrubs (*Betula nana*, *Empetrum nigrum*, *Ledum palustre*, *Vaccinium uliginosum* and *V. vitis-idaea*) were important components of the tundra tussock community. Alpine communities were supported in the higher elevations. The riparian corridors were lined with willows (*Salix* spp.), alder (*Alnus crispa*) and white spruce (*Picea glauca*). Black spruce (*P. mariana*) stands were more common along the southern and eastern edges of the study area. Mean annual precipitation was about 30–40 cm. Snow cover, typically persisting from November through May, can be hard and crusted in wind scoured areas. Temperatures can drop to -45 °C during the winter months. However, mean temperatures have risen over the study period in this region, especially during the winter months (Stafford *et al.*, 2000).

#### Data collection and analysis

Twenty permanent vegetation transects were created in 1981 throughout the Buckland River valley. Canopy cover was ocularly estimated (Daubenmire, 1959) using a 20 cm x 50 cm quadrat placed every 2 m along a 50 m transect. Utilization (evidence of grazing such as signs of cratering and cropped or dropped lichens) was noted in each quadrat and reported as frequency for the transect. The transects were revisited and reread during 1995 and 1996 (henceforth 1995/96); however only 18 of the original 20 transects were located. A burn may have hidden 1 of these 2 transects from our survey team. Seven additional transects were created in 1996 (Fig. 2). These transects were selected because they fell within the core winter range and contained enough initial lichen cover so that changes could be detected. The methodology employed in 1981 was repeated during 1995/96, but all 25 transects were also reread using a more objective point intercept method (Floyd & Anderson, 1987). A 1.0 m x 0.5 m sampling frame was strung every 10 cm along both axes which created 50 intercepts. The frame was laid out every 4 m along the identical 50 m transect, for a total of 12 frames per transect (see Jandt *et al.*, 2003 for more details). The first species observed under the intercept was recorded. Non-vegetative observations (*e.g.*, rock, bare ground or water) were also recorded. In 2005, we only used the point intercept method. We employed transects as our sample unit for statistical analyses.

We assigned a category to each record; lichen, graminoid, shrub, forb, moss, or non-vegetated. *Andromeda polifolia* and *Oxycoccus microcarpus* were classified as forbs in 1981 (Adams & Connery, 1981).

Though they are better categorized as shrubs (Viereck & Little, 1972), we adhered to the 1981 convention. Cover values for both of these species were extremely low (< 0.5%) and unlikely to influence analyses. *Rubus chamaemorus* was also classified as a forb. Other members of this genus are correctly categorized as shrubs but this species is not (Viereck & Little, 1972).

We further subdivided lichens into 3 categories, which were primary (most preferred forage species), secondary (other forage species), and non-forage lichens. *Cladina mitis*, *C. rangiferina*, *C. stellaris*, and *Cladonia uncialis* were assigned to the primary forage category based on published literature (Ahti, 1959; Scotter, 1967; Pegau, 1968; Holleman & Luick, 1977; Thomas & Hervieux, 1986; Thomas & Kiliaan, 1998) and our experience conducting range work in northwestern Alaska. We similarly assigned *Cetraria cucullata*, *C. ericetorum*, *C. islandica*, *C. nivalis*, *Cladonia amaurocraea*, and *C. gracilis* to the secondary forage lichen category. All other lichens were labeled as non-forage species. We tallied the number of different species to determine species richness (an index of diversity) for each transect for the 1995/96 and 2005 datasets. Utilization was calculated in the same manner as before.

We made 2 sets of comparisons. First, we compared ocular estimates from 1981 with ocular estimates from 1995/96 on the 18 relocated transects. Second, we compared point intercept estimates from 1995/96 with point intercept estimates from 2005 ( $n = 25$ ). We employed paired  $t$ -tests to identify significant changes for both sets of comparisons. We utilized analysis of variance (ANOVA) to make other comparisons between the 1995/96 and 2005 point intercept data and between the original (1981) and newer transects (those added in 1996). We used linear regression techniques to test for association among elevation, utilization, species diversity, and change in lichen cover.

## Results

The percent cover of the various categories (lichen, graminoid, shrub, forb, or moss) are displayed in Fig. 3. Based on the 1995/96 data, we found that both total lichen and primary forage lichen coverage were greater in the 7 transects added in 1996 than the original 18 transects deployed in 1981 ( $F_{1,24} = 8.53$ ,  $P = 0.008$ ,  $F_{1,24} = 7.70$ ,  $P = 0.011$ , respectively). Average rate of utilization was not significantly ( $F_{1,49} = 0.25$ ,  $P = 0.619$ ) different between 1995/96 (38.7%) and 2005 (35.0%). Utilization was able to loosely predict lichen coverage ( $R^2 = 0.072$ ,  $F = 3.72$ ,  $df = 49$ ,  $P = 0.060$ ; Fig. 4a). Caribou use of areas with < 5% lichen cover was negligible (Fig. 4a). Loss of lichen

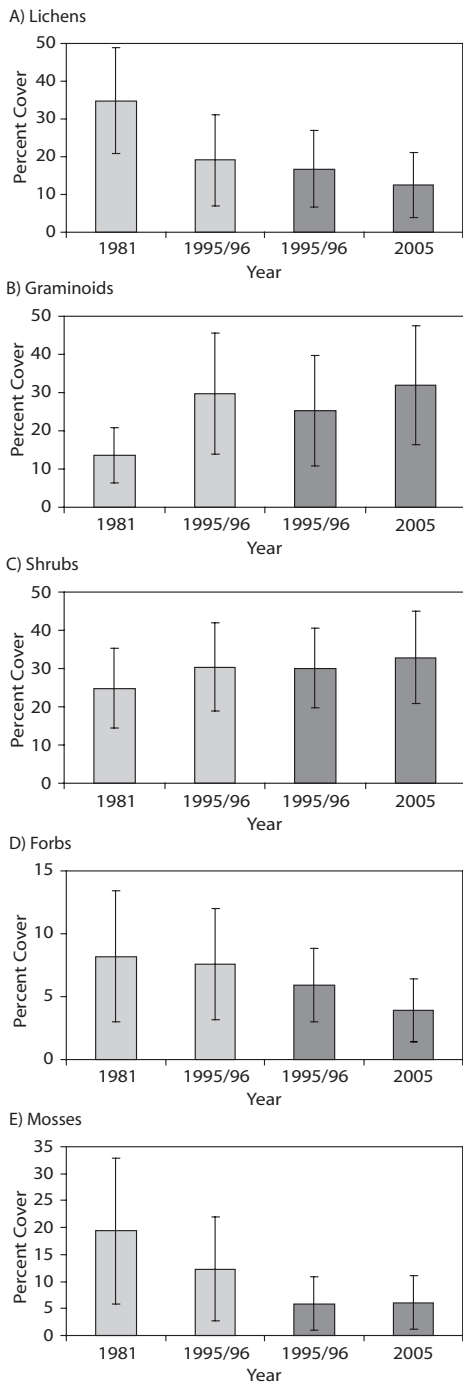


Fig. 3. Changes in the vegetative cover on winter range transects in the Buckland Valley, Seward Peninsula, Alaska from 1981 – 2005 for various categories. Ocular estimation techniques were employed during 1981 and 1995/96 (light gray bars), but a point intercept method was used in 1995/96 and 2005 (dark gray bars). Figure depicts mean  $\pm$  SD.

cover from 1995/96 to 2005 was a good predictor of initial (1995/96) lichen cover ( $R^2 = 0.309$ ,  $F = 10.27$ ,  $df = 24$ ,  $P = 0.004$ ; Fig. 4b). Vegetative species richness was positively associated with elevation in 1995/96 ( $R^2 = 0.134$ ,  $F = 3.55$ ,  $df = 24$ ,  $P = 0.072$ ) and 2005 ( $R^2 = 0.174$ ,  $F = 4.85$ ,  $df = 24$ ,  $P = 0.038$ ). Percent lichen cover was not significantly associated with elevation in 1995/96 ( $R^2 = 0.009$ ,  $F = 0.21$ ,  $df = 24$ ,  $P = 0.652$ ) nor in 2005 ( $R^2 = 0.002$ ,  $F = 0.05$ ,  $df = 24$ ,  $P = 0.825$ ). Species richness was not significantly different between 1995/96 and 2005 ( $F_{1, 49} = 0.05$ ,  $P = 0.816$ ).

Lichen coverage declined significantly, from 34.8% to 19.1%, between 1981 and 1995/96 ( $t = 5.69$ ,  $df = 17$ ,  $P < 0.01$ ). Both *Cladina rangiferina*, a primary forage lichen, and *Cetraria cucullata*, a secondary forage lichen, significantly declined in coverage between 1981 and 1995/96 ( $t = 2.92$ ,  $df = 17$ ,  $P < 0.01$ ;  $t = 4.05$ ,  $df = 17$ ,  $P < 0.01$ , respectively). Moss also significantly decreased over this time period, from 19.4% to 12.3% ( $t = 3.74$ ,  $df = 17$ ,  $P < 0.01$ ). Graminoid and shrub cover significantly increased from 13.6% to 29.7% ( $t = 5.63$ ,  $df = 17$ ,  $P < 0.01$ ) and 24.8% to 30.4% ( $t = 3.87$ ,  $df = 17$ ,  $P < 0.01$ ) between 1981 and 1995/96, respectively.

An analysis of the 1995/96 data revealed that differences in cover resulting from the differences between the ocular and point intercept methodologies were minor (see Fig. 3). Cover estimates were similar for most species, with an overall mean difference of just 0.7% (Jandt *et al.*, 2003). The two methods were in relatively close agreement even when comparing rare or inconspicuous species (Jandt *et al.*, 2003).

Our analysis of the point intercept data revealed that lichen cover declined significantly ( $t = 3.21$ ,  $df = 24$ ,  $P < 0.01$ ) from 16.8% to 12.5% during the 1995/96 to 2005 time period. The decline in primary forage lichens, from 7.8% to 4.6%, was also significant ( $t = 3.62$ ,  $df = 24$ ,  $P < 0.01$ ). *Cetraria cucullata*, a secondary forage lichen present on every transect, declined by a relative 17.1% from 1995/96 to 2005. However, changes in both secondary and non-forage lichens were not significant ( $P > 0.05$ ) between 1995/96 and 2005. The decline in overall and primary forage lichen cover over the study period coincided with the rapid expansion of the number of individuals in the WAH (Fig. 5).

We found that the amount of lichen loss between 1995/96 and 2005 was a good predictor of percent cover of lichen in 1995/96 ( $R^2 = 0.309$ ,  $F = 10.27$ ,  $df = 24$ ,  $P = 0.004$ ). Only one transect (# 24) with high initial lichen cover (31.3%) did not show a decline in 2005. Lichen cover on this transect, which had no sign of utilization in 2005, increased to 37.3% by 2005. We were unable to detect an association



between lichen loss and elevation ( $R^2 = 0.043$ ,  $F = 1.02$ ,  $df = 24$ ,  $P = 0.323$ ).

Our analysis revealed that graminoid cover significantly ( $t = 4.39$ ,  $df = 24$ ,  $P < 0.01$ ) increased from 25.3% to 31.9% between 1995/96 and 2005. Increased cover in *Eriophorum* spp. (3.4%) and *Carex* spp. (1.5%) accounted for most of this change. The wetland species *Carex aquatilis* was the only member of the genus that did not demonstrate an increase (-0.1% cover) during this period. Similarly, shrub cover significantly ( $t = 2.12$ ,  $df = 24$ ,  $P = 0.045$ ) increased during this time period from 30.1% to 32.8%. The largest increase in cover was seen in *V. uliginosum* (1.5%), but *Dryas* spp., *V. vitis-idaea*, *Empetrum nigrum*, and *Arctostaphylos alpina* also increased. *Salix* spp. cover did not significantly change during this time period. *Ledum palustre* had the biggest decline in cover (1.0%) for a shrub species. We found that forb cover declined significantly ( $t = 3.86$ ,  $df = 24$ ,  $P < 0.01$ ) between 1995/96 and 2005 from (5.9% to 3.9%), primarily due to a decline in *Rubus chamaemorus*. *A. polifolia* and *O. microcarpus* both slightly increased, which would only enhance (however slightly) the observed increase of shrubs and decline of forbs if categorized as shrubs rather than forbs. Moss cover was not significantly different between 1995/96 and 2005 ( $t = 0.43$ ,  $df = 24$ ,  $P = 0.673$ ).

## Discussion

The WAH has undergone a 30-year period of continuous growth, beginning in 1976. The pace of this growth has decelerated as the herd has reached 490 000 individuals (Dau, 2003; 2005). Portions of the herd have recently expanded into winter range outside the historic core range and there have been some indicators of poor nutrition associated with severe winter conditions (Dau, 2005). All of these factors have contributed to speculation as to when the herd will inevitably decline and if that decline will be precipitous. Our permanent range transects, deployed in 1981 and 1996, provide insight into 1 factor that plays into the complex calculus that shapes the herd's trajectory.

Terricolous lichens constitute the majority of diet for barren-ground caribou herds during winter (Thompson & McCourt, 1981; Boertje, 1984; Thomas, 1998) and the WAH is no exception (Saperstein, 1996; Jandt *et al.*, 2003). Evidence that lichens are not requisite for caribou come from low density, high Arctic, island populations (Thomas & Edmonds, 1983). Lichens appear to be a critical component of the diet of large migratory herds in North America (Klein, 1991). However, it has been argued that a transition from lichen-dominated tundra to sedge-dominated tundra

due to *Rangifer* grazing is predictable and the new system could be highly productive (van der Wal, 2006). It appears that the WAH may be able to answer this question in the future.

We found that lichen cover declined significantly (by a relative 3.0% annually) from 1981 to 1995, during which time the WAH grew at a consistently very high rate. Lichen cover continued to decline

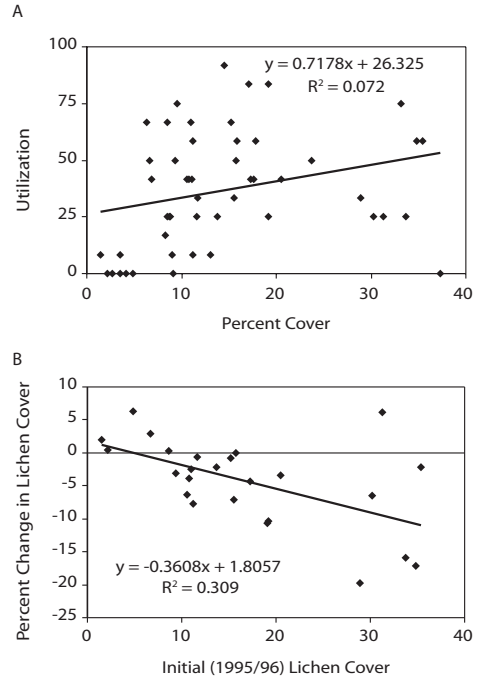


Fig. 4. Use of lichens by caribou in the Buckland Valley, Seward Peninsula, Alaska. A) Utilization versus amount of lichen cover (1995/96 and 2005 data) B) Percent change in lichen cover from 1995/96 to 2005 versus initial (1995/96) lichen cover.

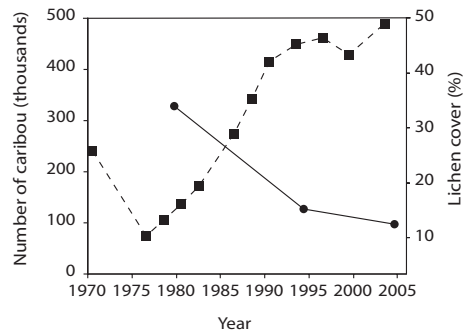


Fig. 5. The decline of lichen cover (solid line with circles) on the winter range of the WAH coincided with the rapid expansion of the number of individuals in the herd (dashed line with squares), northwest Alaska.

between 1996 and 2005. The rate of decline was slightly less (a relative 2.6 % annually) during this time period which was concurrent with the growth of the herd slowing and its expansion into new winter range (Dau, 2003). Importantly, primary forage lichens significantly declined during this time period. Utilization and the amount of decline in lichen cover were significantly associated with initial lichen cover, implying that caribou selected for areas with high lichen abundance. Although the decline in lichen cover cannot be directly correlated with the eruption of the WAH and increased grazing pressure, it does provide a compelling, simple, and logical inference (see also Moser *et al.*, 1979; Arseneault *et al.*, 1997; van der Wal, 2006). This is further supported by the fact that transect # 24 (the northeastern most transect, Fig. 2) was closest to the edge of the core winter range, contained no sign of utilization in 2005, and was the only transect with high lichen cover not to reveal a loss of lichen cover over the last decade (Fig. 4). Transects read in 1997 outside (at that time) the winter range of the herd on the southern Seward Peninsula had high lichen cover (BLM, unpubl. data). The effects of trampling may exacerbate the deterioration of lichen cover (Pegau, 1969; Manseau *et al.*, 1996). Reindeer have been absent from the study area since its inception.

The consequences of global climate change (particularly warming and drying) have also been implicated as factors that may reduce lichen abundance in the tundra ecosystem (Chapin *et al.*, 1995; Cornelissen *et al.*, 2001; Epstein *et al.*, 2004; Walker *et al.*, 2006) and may have contributed to declining lichen cover that we observed. Lichen cover declined in recently burned (< 35 years old) areas with low initial cover (< 5%) that probably would not have been utilized by caribou (Arseneault *et al.*, 1997; BLM, unpublished data).

Wildland fire, an environmental factor that can dramatically reduce lichen abundance (Vioreck & Schandelmeier, 1980; Klein, 1982; Swanson & Barker, 1992; Thomas & Kiliaan, 1998; Rupp *et al.*, 2006), is infrequent in the tundra ecosystem (Wein, 1976). Caribou foraging during winter avoid areas that have been burned by wildland fires (Schaefer & Pruitt, 1991; Thomas *et al.*, 1998; Joly *et al.*, 2003). The Seward Peninsula, including our study area, appears to have a relatively high fire frequency rate for tundra ecosystems (Racine *et al.*, 1987) and it has been predicted that fire frequency and extent will continue to increase (Wein, 1976; Rupp *et al.*, 2000; McCoy & Burn, 2005). Disturbance from fire was uncommon on our transects, though it did occur. One 1981 transect was presumably "lost" to a wildfire and a second transect burned between 1995 and 2005.

Our data did not reveal a reduction in species richness between 1995/96 and 2005, which is in contrast to the findings of other researchers investigating the consequences of global warming (Chapin *et al.*, 1995; Walker *et al.*, 2006). Species richness was positively associated with elevation, but even our highest transect was under 610 m. We found that graminoid cover doubled between 1981 and 1995/96 and increased again between 1995/96 and 2005 by a relative 26.0 %. These results are consistent with research suggesting that global warming, drier conditions, and mammalian grazing and trampling may lead to a conversion of the tundra ecosystem to a grassland steppe community (Zimov *et al.*, 1995; Rupp *et al.*, 2000, van der Wal, 2006).

Shrub cover also increased during this time period, though not as dramatically. A primary finding for research documenting the response of the tundra ecosystem to global warming is an increased abundance of shrubs (Chapin *et al.*, 1995; Epstein *et al.*, 2004; Tape *et al.*, 2006; Walker *et al.*, 2006). Increased shrub cover has been implicated as another factor negatively effecting lichen abundance (Pegau, 1970; Cornelissen *et al.*, 2001). Shading and increased litter cover caused by these vascular plants may retard lichen growth, as well as alter snow melt patterns (Sturm *et al.*, 2005). Vascular species also recover from grazing more quickly than lichens (Henry & Gunn, 1991).

We detected a significant decline in the cover of forbs during the last decade of the study; however this was not mirrored in Epstein *et al.*'s (2004) findings. *R. chamaemorus*, the species driving the decline in forbs, is typically found in wet, boggy areas (Vioreck & Little, 1972). The significant decline of moss cover we found since the beginning of the study period is in accordance with previous findings on the effects of global warming (Chapin *et al.*, 1995; Epstein *et al.*, 2004; Walker *et al.*, 2006), but possibly also grazing (van der Wal & Brooker, 2004; van der Wal, 2006). The reduction in moss cover could feed into a positive feedback loop and lead to increased drying of the tundra, a northward and westward shift of treeline and even more wild fires (Zimov *et al.*, 1995; Rupp *et al.*, 2000; Rupp *et al.*, 2002).

Caribou populations naturally fluctuate and these cycles are dependant on the complex relationships among climate, the caribou, their range and predation (Messier, 1991; Gunn, 2003). Grazing is an important ecosystem driver (Hobbs, 1996; Augustine & McNaughton, 1998; Mysterud, 2006; Wisdom *et al.*, 2006). Our results are in agreement with the assertion that grazing by caribou can be an important factor in the depletion of lichen of large areas of continental ranges (Moser *et al.*, 1979; Messier *et al.*, 1988, Arseneault *et al.*, 1997) and the rapid transition

from lichen-dominated to graminoid-dominated tundra communities (van der Wal, 2006). All indications show that the Arctic is already warming (Hansen *et al.*, 1999; Barber *et al.*, 2000; Oechel *et al.*, 2000; Serreze *et al.*, 2000; Goetz *et al.*, 2005). The response to this warming will likely include changes in vegetative communities (Chapin *et al.*, 1995; Epstein *et al.*, 2004; Tape *et al.*, 2006; Walker *et al.*, 2006; this study) and increased wildland fire (Rupp *et al.*, 2000; Goetz *et al.*, 2005; McCoy & Burn, 2005). Changes in the vegetative communities may accelerate the rise in fire frequency (Starfield & Chapin, 1996; Rupp *et al.*, 2000). Our study was not designed to tease apart the relative contributions of grazing and global warming to the rapid and radical changes in the tundra ecosystem that we documented, though both appear to be important ecosystem drivers.

The WAH is currently at a known population high, has shown signs of poor nutrition associated with severe winter weather and has expanded its range (Dau, 2003; 2005). These signs were also present prior to the recent crash of the Mulchatna Caribou Herd (Hinkes *et al.*, 2005). The decline of this herd may displace additional sport hunting pressure on to the WAH. We have gone further, in the case of the WAH, by showing that lichen cover has decreased over the last 24 years and that the decline in primary caribou forage lichens was significant. Conditions on the WAH's core winter range, in terms of lichen cover, are deteriorating. All of these factors are interrelated and may or may not increase the rate of change. Although no one can accurately predict how the complex interaction of these factors will affect the WAH, the status of the indices we do have indicate that conditions are suitable for a decline in the herd.

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