

## EFFECTS OF OVERABUNDANT MOOSE ON THE NEWFOUNDLAND LANDSCAPE

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**ABSTRACT:** The long-term effects of introduced and overabundant herbivores on community development must be monitored and managed in an ecosystem-based forest management approach. This paper builds on previously published ecological descriptions and hypotheses offered on the effects of moose overabundance in Newfoundland. The island, in the absence of wolves, provides a setting for study of local irruptions in moose populations, which now affect an increasing area of the forest. Moose effects occur most often after natural disturbances and logging, involving unique forest succession patterns. We describe some of these changes, along with anticipated and realised changes in associated forest biodiversity. We offer suggestions to improve or refine monitoring of moose populations, especially at local scales, to detect cases of overabundance. Finally, we offer recommendations for the management of overabundant moose populations.

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The long-term effects of introduced and overabundant herbivores on forest community development must be monitored and managed in an ecosystem-based approach to forestry and wildlife interests. Adapting forest management to shifting baselines created by the effects of overabundant herbivores, especially in eastern North America, increasingly defeats the interest of biodiversity protection (Lindenmayer and Franklin 2003). Logging in the boreal forest and other forest management tailored to the spatial scale and frequency of wildfires or insect-related tree mortality are recommended to protect and/or restore ecological integrity (Hunter 1993, Niemelä 1999), as a

minimum way of recognizing the adaptation of diverse organisms to the forests they occupy, by considering their disturbance regimes. Disturbance is usually defined as a rapid release or reallocation of resources in a forest community (White and Jentsch 2001), thereby possibly ignoring more gradual changes to forest community development caused by irruptive population phases or overpopulations of herbivorous mammals, especially deer (Cervidae). Such changes can be considerable, unpredictable, and relatively irreversible (Davidson 1993, Côté et al. 2004).

In this paper, we offer a case study of introduced moose (*Alces alces andersoni*),

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its overabundance, and its enduring effect on biodiversity in specific areas of boreal forest on the island of Newfoundland. We show that moose are significantly influencing several aspects of some ecosystems, including forest succession and composition, soils, and other wildlife. Previously it has been shown that moose are capable of producing negative economic effects on the forests of Newfoundland (Pimlott 1963, Thompson 1988), but such studies are often limited just to a portion of a forest rotation (Thompson and Curran 1993, McLaren et al. 2000a). This paper builds on published ecological descriptions and hypotheses offered for longer-term effects of moose overabundance in Newfoundland (e.g., Bergerud and Manuel 1968, Thompson and Mallik 1989, Thompson et al. 1992). We make recommendations toward monitoring, conservation planning, and management of overabundant moose from the perspectives of past examples and anticipated future challenges in Newfoundland.

### **Moose in Newfoundland: A Background**

The island of Newfoundland, Canada, is a landmass of 112,000 km<sup>2</sup> in the north-west Atlantic, of which about two-thirds is forested and/or qualifies as excellent moose habitat. Moose were introduced to central Newfoundland in 1878 with the release of a male and female from nearby Nova Scotia (Pimlott 1953). A second release of two males and two females from New Brunswick, into western Newfoundland, followed in 1904. Moose rapidly colonized Newfoundland (Fig. 1), as they exploited new habitat and as wolves (*Canis lupus*) were extirpated (Pimlott 1959). Caribou (*Rangifer tarandus terra-novae*) are the only other ungulate in Newfoundland, primarily occupying non-forested habitat and existing prior to the arrival of Europeans. Where the two cervid species now co-exist, there has been no recorded direct competi-

tion between them.

Today, moose occupy all ecoregions on the island, at densities in primarily forested habitat in many instances exceeding 4 moose / km<sup>2</sup> (> 1,000 kg / km<sup>2</sup>). Population reconstruction and aerial survey estimates for a moose management unit normally do not show this finer-scale spatial variation in density, but overabundance is suggested by our temporal series (Fig. 1), considering that approximately 75,000 km<sup>2</sup> of habitat supported on average about 2 moose / km<sup>2</sup> during two periods, in the late 1950s and late 1980s (Mercer and McLaren 2002). This average is met by considerable variation and any densities > 2 moose / km<sup>2</sup> are considered above management targets (Newfoundland and Labrador Inland Fish and Wildlife Division, unpublished). The national parks in Newfoundland, where hunting is prohibited, form special cases of overabundance. The second and likely only successful moose introduction to Newfoundland was ca. 20 km from what is now the boundary of Gros Morne National Park, in the western part of the island. Local hunting kept the population relatively low until this activity ceased with the establishment of the park in 1974. Within the park area of 1,805 km<sup>2</sup>, of which only ca. 30% is suitable habitat, moose have increased steadily from some 1,000 animals to > 7,000, with local densities as high as 7 moose / km<sup>2</sup> by 1995 (McLaren et al. 2000b).

Management of moose hunting in Newfoundland, which began with the first season in 1945 (Mercer 1995), has achieved a legal kill of about half a million, the majority of which is by resident hunting (McLaren 2004). Annual license issue since 1990 has been between 20,000 and 25,000 resident and non-resident tags combined, with a roughly equal number of either-sex and male-only licences issued (Mercer and McLaren 2002). Annual kill estimates, incorporating poaching, crippling, and high-

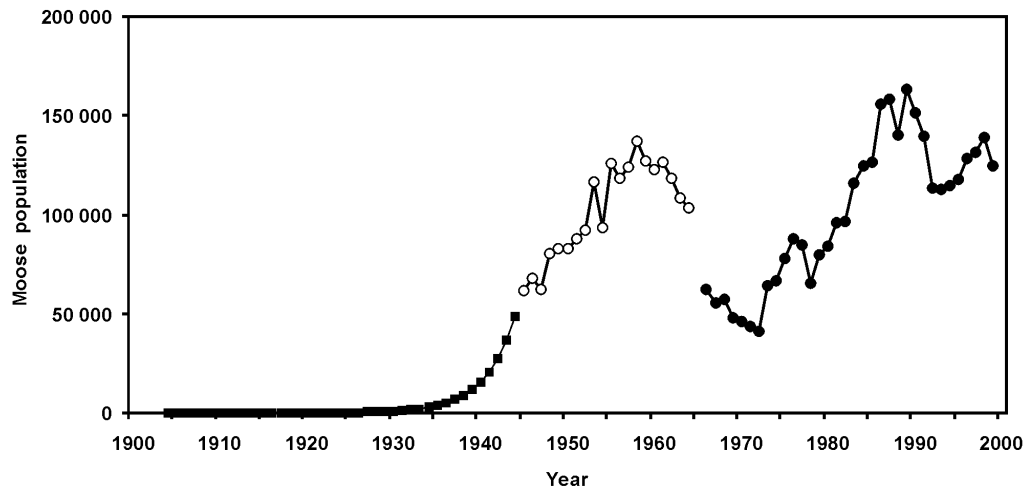


Fig. 1. Moose population trends in insular Newfoundland since introduction, using McLaren's (2004) assessment of the successful point of introduction, Pimlott's (1959) estimate of increase rate during 1904-44 (solid squares), hunter success during 1945-65 (open circles), and hunter reports of moose seen during 1966-99 (solid circles). These estimates, scaled to consolidated aerial survey estimates in the 1980s and in the 1990s (Newfoundland and Labrador Inland Fish and Wildlife Division, unpublished), occur in a wide range of habitats and densities.

way losses, have been 20,000 - 30,000 moose for several years. First Nations are currently included in the resident licensing system in the island portion of Newfoundland and Labrador. Dating at least to 1934, the provincial government has recognized that sport hunting is a major attraction for visitors and the success of the moose introduction to the island has often been applauded as a source of tourist revenue. Non-residents obtain about 10% of moose tags issued, and owing to the higher success offered by outfitters and guides, are responsible for > 10% of the annual moose kill.

Newfoundland is also home to a disproportionately large part of the North American moose population. The island population, at 125,000 moose, represents > 10% of the total continental number of moose (1.05 million), while the total island area, including areas unsuited to moose, is < 2% of the estimated continental moose range (6.44 million km<sup>2</sup>). Throughout North America, density of moose and other deer species varies according to four main factors: the

availability of habitat, the availability of alternate foods created by agriculture, management of hunting, and the presence of limiting factors like aridity or natural predators (Crête and Daigle 1999). Overpopulation usually occurs following introduction into unexploited habitat and persists in situations without natural predators (McShea et al. 1997), as is the case in Newfoundland. In these situations, moose appear to be limited primarily by the productivity of the boreal forest, as described for Québec by Crête and Courtois (1997). As productivity varies, so does the effect of moose, as expected in an unregulated trophic system. Thus, we have two arguments for moose overpopulation in Newfoundland: (1) as presented by Crête and Daigle (1999), Newfoundland hosts an anomalous deer biomass compared with the rest of the continent, presumably because of the absence of wolves; and (2) as described by Mercer and McLaren (2002), a stable equilibrium between the population and food resources does not appear to have occurred for New-

foundland, particularly problematic where moose densities are higher, in more productive areas, and/or in areas less accessible to hunting. Newfoundland thus allows us to illustrate specific cases where ecosystem management experiences new challenges as a result of moose overabundance.

### Effects of Moose on the Forest Ecosystem

#### Forest succession and composition

— Most forests in Newfoundland consist of a combination of balsam fir (*Abies balsamea*) and spruce (*Picea* spp.), with some pioneer and shade tolerant hardwoods. Fir is dominant in older forests, while insect outbreaks, fires, and logging have been frequent forest disturbances creating a generally young-forest landscape. Since the arrival of moose, their consumption of balsam fir and hardwoods has affected forest regeneration following disturbance, particularly along edges and roadsides (Bergerud and Manuel 1968). In these affected forests, spruce and larch (*Larix laricina*) grow normally, since they are species not normally found in moose diets, but balsam fir, a heavily-consumed species, can be prevented from reaching heights > 1 m. The resulting open ecotype has been described but not attributed specifically to moose in any general literature on forest management in Newfoundland; the forest succession leading to the ecotype has been termed “old-field spruce succession” (Damman 1964).

In updating the forest site classification (ecological land classification) for Newfoundland, Roberts and Bajzak (1984) used the term “ungulate induced” to describe the succession specific to richer sites, in which a shift occurs from closed canopy balsam fir and white spruce (*Picea glauca*) to open-grown white spruce following disturbances (Fig. 2). Roberts (1989a, 1989b) attributed this change to occupation of young forests by overabundant moose during the

1960s. Roberts’ (1989b) concern was for white spruce associations and other rare forest types. Similar examples of vulnerable forest types are yellow birch (*Betula alleghaniensis*) associations on the Avalon Peninsula and red maple (*Acer rubrum*) associations in central Newfoundland. These types, occurring with balsam fir, are frequently subject to logging and wind disturbance and occur in management areas where moose are kept near target densities to offer steady hunting opportunities. Yellow birch and red maple are preferred species in a moose diet dominated by fir.

In time, additional changes to natural forest succession caused by overabundant moose may become apparent. Since the 1980s, the total area disturbed by logging, as well as the secondary road network, has increased as forestry has become more extensive in Newfoundland. More common ecological associations may be threatened by the coexistence of regenerating commercial forest and overabundant moose, especially with declining interest in hunting. Less accessible areas may be especially prone to a combination of natural disturbance like insect outbreak and moose overabundance. Areas of both natural and logging disturbance that fail to regenerate into closed canopy forest are already at a scale readily visible on aerial photographs (Fig. 2). Our photographs show white spruce associations. Thompson et al. (1992) and Thompson and Mallik (1989) extended their concerns to black spruce (*Picea mariana*) associations.

Observations of forest composition change attributed to moose overabundance have been recorded in the national parks. Terra Nova National Park, 344 km<sup>2</sup>, has been unaffected by logging since the 1950s, but is a special case of forest disturbance, in which small (< 1 km<sup>2</sup>) landscape patches were disturbed by insect outbreaks in the late 1970s, just before a peak in moose

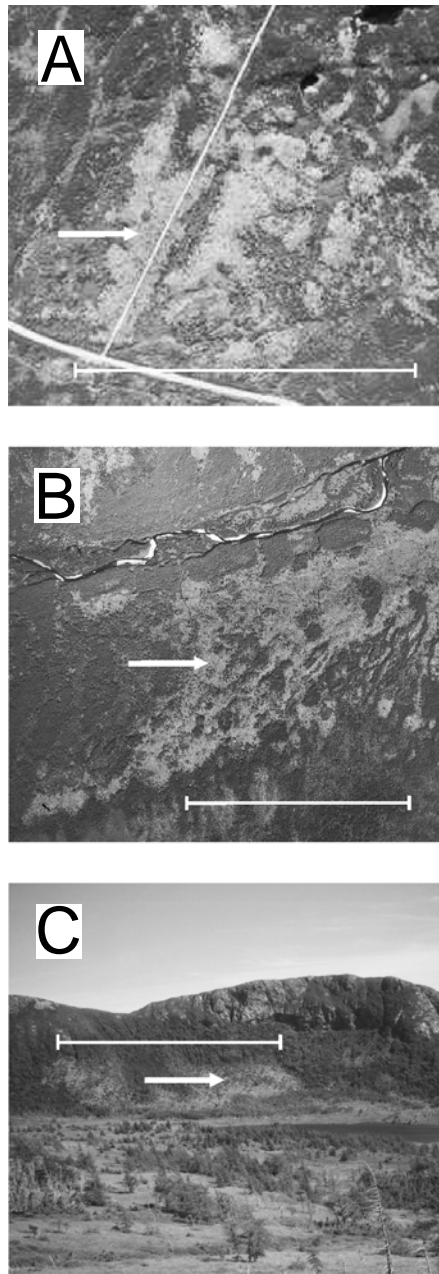


Fig. 2. Ungulate induced changes to the Newfoundland landscape. The aerial photographs show cases where forest succession is interrupted by overabundant moose for (A) an area near Blue Hill in Terra Nova National Park that was disturbed by an insect outbreak in 1978 and has failed to regenerate in 25 years (photo date 1996), (B) a partially forested watershed in 1988, logged in the 1950s, where balsam fir has failed to regenerate in a white spruce

association, following disturbances that also included insect outbreaks, and (C) an area near Halfway Cliff in Gros Morne National Park, where moose have created semi-open cover because they prevent a fir-dominated alpine forest understory from regenerating. The partial canopy in the three photographs (at arrows) is created by open-grown spruce. The scale bars in each photograph measure ca. 1 km.

density during the 1980s. Moose density is low today, ca. 0.7 / km<sup>2</sup>, but very low recruitment observed in all of the last 5 midwinter surveys suggests a declining population, exceeding carrying capacity (Gosse et al. 2002). Carrying capacity in the park has itself declined over time as forests have matured and moose have nearly completely removed hardwoods, such as red maple and mountain ash (*Sorbus americana*) from mature-forest understories. In the forage-limited, disturbed patches, pioneer hardwoods, such as white birch (*Betula* spp.), aspen (*Populus tremuloides*), and pin cherry (*Prunus pensylvanica*), have also been affected. To illustrate the effect moose continue to have on limiting growth and survival of understory trees, park managers constructed several fenced areas or “exclosures” (Terra Nova National Park, unpublished). After only 3 growing seasons, red maple density inside exclosures was up to 3 times higher than outside. For white birch, similar densities occur inside and outside, because stump and root sprouting is common. However, growth of white birch has been affected. For example, up to 90% of white birch stems are > 0.5 m inside the exclosures (this includes 14% > 1 m after 3 growing seasons), but outside, only 5% are > 0.5 m. Outside the exclosures, reductions in stem density of 2–17% for balsam fir also occurred between 1999 and 2003. Measured annually, this effect was highest when snow cover was low and moose were able to uproot young stems. In

Gros Morne National Park, Canada yew (*Taxus canadensis*), once common in coastal forest associations (Robertson and Roberts 1982), is nearly completely removed from the forest understory, presumably by moose. In addition, areas with once fairly closed forest now have frequent understory openings and associated changes in forest structure (Connor et al. 2000).

Experimental introduction of moose to offshore islands in eastern Newfoundland has provided other examples of their capacity to change forest composition very soon after arrival. On Brunette Island, off the south coast, several moose were introduced for the first time in 1974 and plant measurements using exclosures to assist comparisons were undertaken during the 10 following years (Butler 1986). In this study, the ratio in annual production of balsam fir to faster growing hardwoods declined during the first 5 years from 39:1 to 15:1, measured in stems/ha (Wildlife Division, unpublished data). Less common species, like mountain ash and wild raisin (*Viburnum cassinoides*), declined faster than balsam fir as a ratio to the fastest growing hardwoods in the same period. These measurements were corrected for the “natural” succession changes observed inside the exclosures. The removal of forbs, grasses, and alder (*Alnus* spp.), not normally observed in retrospective studies, was significant in observations of a tame moose, which consumed these plants in > 80% of observed bites in a summer period (Butler 1986). Recognizing the earliest changes created by an introduced species, often difficult to quantify, is important to biodiversity management.

**Soils** — A review of the effects of herbivores on multi-trophic interactions including soil effects has recently been provided by Bardgett and Wardle (2003). Pastor and Naiman (1992) discussed the subject for moose. We provide one unique Newfoundland example. In many moun-

tainous, serpentine plateaus in Gros Morne National Park, toxic levels of magnesium and heavy metals occur in soils (Roberts 1980, 1992). Before 1980, moose were rarely encountered in these areas of the park, but their use of the serpentine plateaus is now prevalent (B. A. Roberts, unpublished observations). In plants adapted to the specialized plateau ecosystems, such as dwarf birch (*Betula glandulosa*) and alder (*Alnus crispa*), nickel concentrations range from 48–77 ppm (Roberts 1992, Roberts and Proctor 1992). While not normally including these species in their diet, moose are now consuming these toxic plants, presumably because of the lack of preferred forage species, a consequence of overabundance. This interaction may ultimately be detrimental both to the animals and to the long-term persistence of some plant species. Our concern for landscape effects is that even the sparse cover of plants acts as a soil stabilizer. As moose consume and trample slowly growing plants, the rate of soil erosion increases, as once described by Leopold (1949), in “thinking like a mountain.”

**Other forest wildlife** — A review of deer overabundance in many parts of the world and its cascading (indirect) effects on other plant and animal species is provided by Côté et al. (2004). Changes to forests caused by overabundant moose in Newfoundland are also very likely to affect many more forest-dwellers than the plants directly affected by browsing. For example, forest songbirds dependent on hardwood and balsam fir trees (Settingington et al. 2000) and epiphytic tree lichens with specific habitat requirements (Yetman 1999) may actually be eliminated from portions of the landscape with overabundant moose. Alternatively, ungulate-induced modification of forest structure and composition after natural disturbance or logging may result in changes to habitat selection among

forest-dependent species. Our examples below further illustrate these hypotheses.

Lichen diversity is generally related to the availability of different microhabitats (Gustafsson et al. 1992, Kuusinen 1995, Rosentreter 1995, Neitlich and McCune 1997). A mixture of deciduous and coniferous tree species provides an array of bark acidity and texture, offers diverse trunk structures, and creates a mosaic of moisture and light conditions for lichen growth. Consequently, altered forest composition and structure caused by moose, such as the elimination of hardwoods, may have an indirect effect on epiphytic lichen community composition. This effect was illustrated through principal component analysis (PCA) of lichen cover in various forest types of Terra Nova National Park (Yetman 1999). Red maple in particular, in stands mixed with balsam fir, supports a unique lichen community according to the PCA, in which the principle components correspond to site richness and bark acidity. As discussed above, maple is one of the trees being limited in both density and height growth as a result of moose in the park. An end result may be the loss of the lichen community this tree supports. In the Avalon Peninsula, yellow birch is known to be a specific host for the rare cyanolichen, *Degelia plumbia*. As yellow birch is selectively browsed in this area, its subsequent failure to regenerate may limit opportunities for this host-specific epiphyte.

Boreal felt lichen, *Erioderma pedicellatum*, a globally rare species, is now restricted to coastal Nova Scotia and the island of Newfoundland. It was recently listed as vulnerable under the Newfoundland and Labrador Endangered Species Act. A species of oceanic affinity, it is found in moist, cool forests where it grows predominantly on balsam fir and forms part of a characteristic cyanolichen community (Ahti 1983, Maass 1983). The known New-

foundland population of boreal felt lichen is concentrated in the Avalon Peninsula and Bay d'Espoir areas. The Avalon Peninsula has had a long history of land development, logging, insect outbreaks, wind disturbance, and fires; it is also an area frequently used by moose in winter. The percentage of juvenile lichens is much lower in this area than in the second area, where forest cover is more complete and moose are less abundant (N. Djan-Chékar, unpublished data). We speculate that in the Bay d'Espoir area more opportunity for lichen colonization exists as a result of fewer moose and better balsam fir regeneration. To assess this hypothesis, biologists now monitor both the boreal felt lichen by recording its occurrence and abundance on the landscape, and also the use by moose of its critical habitat by annually counting pellet groups, an ecosystem approach to biodiversity monitoring.

Changes to vertebrate communities as a result of forest succession are probably best understood for birds (Helle and Niemi 1996). Plant structure and diversity influence avian assemblages (e.g., MacArthur and MacArthur 1961, MacArthur et al. 1962, James and Rathbun 1981, Cody 1985, Willson and Comet 1996), foraging behaviour (Parrish 1995), and nest site selection (Martin 1992). In addition, predation of nests can increase in areas with low foliage density (Martin 1993). However, the indirect effects that herbivores exert on avian assemblages through modifying vegetation are not well known (Rotenberry et al. 1995, McShea and Rappole 1997) and only a few researchers have explicitly studied the effects of herbivores on birds (DeGraaf et al. 1991, Popotnik and Giuliano 2000).

Indirect effects of moose overabundance on songbirds in the forest–heathland ecotone in central Newfoundland were considered in a model of avian richness and abundance in black spruce–feathermoss forest and *Kalmia angustifolia* heath. Fire

suppression, logging, and plant consumption by moose all contribute to a conversion from black spruce-feathermoss forest to transitional black spruce-*Kalmia* forests and, in cases of very poor regeneration, *Kalmia* heath (cf. Thompson and Mallik 1989, Thompson et al. 1992). A forest inventory describing the extent of this conversion has not been completed but the problem has demanded attention by silviculturists in the province (English and Hackett 1994). We observed that songbird abundance and species richness is significantly lower in *Kalmia* heath compared to black spruce-*Kalmia* forest (Lewis 2004). Most birds in *Kalmia* heath tended to be habitat generalists and were also common in the forests. Species abundance increased with increasing vertical structure in the *Kalmia* heath, indicating the importance of fire skips and snag retention, current elements of forest management. For example, Common Yellow Throat (*Geothlypis trichas*) and Lincoln's Sparrow (*Melospiza lincolni*) were associated with the fire skips. We were able to conclude that the continued suppression of many plant species by overabundant moose, as well as the invasiveness of *Kalmia* with logging and disruption of a natural fire regime, has variable but significant effects. Initial conversion of forest to black spruce-*Kalmia* transition types will result in increased songbird abundance and species richness. However, if severe restriction of regeneration in black spruce forest continues and heath increases in area, moose will have contributed to an increasingly impoverished avifauna.

### Monitoring Moose Overabundance

Given that herbivores can influence forested ecosystems in a variety of ways, it is important to have programs to monitor their abundance and effects. Increasingly, moose management plans must ask for local knowledge and advice on appropriate

mitigative measures in cases of overabundance. We offer some suggestions to improve or refine monitoring of moose populations, especially at local scales, in Newfoundland and in other jurisdictions where overabundance may be a concern.

#### Improved use of aerial surveys —

Surveys are not generally useful in assessing moose overabundance, because they are prioritized to areas of low, not high abundance, providing information to justify changes to licence quotas, the largest concern of hunters. In addition, survey areas are often much larger than areas of local overabundance. However, data from past surveys are often readily available and their innovative use, such as for identification of very high density areas or for location of census blocks of high individual counts, can be made to assess local overabundance and to verify interpretations from land capability indices.

#### Land inventories —

Management for forest sustainability must recognize land capability as the critical long-term factor determining productivity of any biological species. Land inventories can assist in ecosystem management at various scales. Most readily available maps are based on timber inventory, i.e., predicting economic value of the trees, and have limited applicability to predicting wildlife habitat (e.g., Proulx and Joyal 1981, Potvin et al. 1999, McLaren and Mahoney 2001). However, the soil-based Canada Land Inventory (CLI) is a more comprehensive classification that includes specific references to the habitat requirements of deer and other wildlife. It offers ecological comparisons of areas with varying forest capability and consequently shows relative capability to support ungulate populations. The inventory can become a reference for predicting “ungulate-induced” changes to forest succession on the landscape. For example, *Dryopteris*–*Lycopodium*-balsam fir and *Hylocomium*-bal-



sam fir, classed uniquely in the CLI, are the dominant types where stand conversion to white spruce has taken place as a result of disturbance and overabundant moose (Roberts 1989a, b).

**Pellet-group counts** — Neff (1968) first described pellet-group (deer defecation) counts as a means of assessing local deer density. Jordan et al. (1993) advise that pellet-group counts for moose are an effective means to make relative density comparisons over short periods and that long-term averages also compare well to information obtained from aerial surveys. The fact that moose can migrate seasonally, sometimes over long distances (McLaren et al. 2000b), may make pellet groups a more relevant index of local overabundance than an aerial survey conducted in one season. Important factors in designing a monitoring program using pellet-group counts are to replicate counts in each area of interest and to conduct counts at the same time in successive years (Jordan et al. 1993). Pellet-group counts are subjective, because group definition is variable between observers and the season in which defecation occurs in a pellet form is variable between years and individuals. Pellet groups also preserve longer in dry relative to humid soil conditions that vary with topography. Simple attempts at calibration to correct for such errors are likely to be unsuccessful.

**Browse surveys** — Often, plant-based protocols for monitoring herbivore effects are too labour-intensive or they include inaccurate assessments like browsing “severity indices” determined by visual inspection. Indices designed for one plant architecture may not apply well to another. Telfer (1967, 1972) advises on more accurate measurement of forage yield and browsing effects based on twig counts. Literature on optimal foraging theory (e.g., Gross et al. 1993) offers additional advice.

**Dendrochronology** — Roberts (1989a) used dendrochronology to describe “old-field spruce succession” by means of sampling balsam fir trees from several areas within one forest type to measure stem growth and ring width. From this work, a specific period of moose overabundance was determined. Dendrochronology has since been applied to several studies of the long-term effects of mammals on ecosystems (e.g., Sinclair et al. 1993, McLaren and Peterson 1994). The technique can provide an accurate description of the cumulative effects of moose consumption on tree or shrub biomass (McLaren and Peterson 1996).

**Experimental enclosures** — Several ecologists have constructed enclosures (fences to keep mammalian herbivores out of an experimental area, e.g., McInnes et al. 1992). Enclosures may not produce immediate changes in forest succession but they may reveal other ecological effects. Changes in the trophic pyramid may be most apparent at levels directly above and below the plants, i.e., in soil or in the abundance of herbivores that are not excluded by the fence. For example, McInnes et al. (1992) gained insight into effects of moose on the boreal forest by measuring not only browsing of trees and shrubs, but also changes to leaf litter.

There are several cautions in designing an enclosure system:

1. The fence itself introduces ecological effects. Attracted by the forage regeneration inside the enclosure, herbivores may circle the fence and cause more extreme damage to plants adjacent to it. Consequently, the “control” (unfenced) monitoring area should be well outside the fence. As vegetation in the area outside the fence continues to be browsed, it may retain an open or semi-open canopy depending on conditions at the time of fence construction. Plants just inside the fence thus benefit

from a higher light supply than in the absence of herbivores and their measurement would constitute a bias. The “experimental” (fenced) monitoring area should therefore be located far enough inside the fence to eliminate this bias. Finally, the fence material should be chosen so that it produces minimum soil change; e.g., galvanized material adds toxic zinc to soils. Cribbing works well on fence posts in shallow soil.

2. Sufficient replication to control for differences in site history and productivity is often difficult, both because of the difficulty in placing enclosure sites randomly and because of the expense of fence construction and maintenance. An obvious compromise is to construct larger enclosures; we suggest that 35 m on a side is minimum construction because it allows at least a 500-m<sup>2</sup> unbiased sampling area. Larger enclosures may be required to incorporate topographic variation. Numerous plots inside enclosures may be unnecessary and they do not provide real landscape-scale replication.

3. Monitoring protocols are often too ambitious or flawed. Good advice is to begin fence construction only after very specific research hypotheses have been outlined.

4. Enclosures will not mimic the forest succession that would occur as if the herbivore had never been present in the ecosystem. While this statement is a truism, it is often ignored in discussions of the results of enclosure monitoring. For example, trees regenerating inside an enclosure often grow vigorously from rhizomes, stump sprouts, or layering. Such vegetative reproduction is obviously enhanced when herbivores dominate an ecosystem for many years allowing plants to allocate biomass below ground or to lateral branch growth. Meanwhile, seed fall into an enclosure declines when plants outside the enclosure are not replaced because herbivores continue to suppress api-

cal or floral growth, and seeds with longer persistence will have a competitive advantage inside the enclosure area.

**Local ecological knowledge** — Forest managers rely on conservation officers, field technicians, and the general public, especially hunters and naturalists, to collect new field observations. Access to wilderness areas and our ability to change them through industry, sport, and deliberate or accidental species introductions increase in tandem. Thus, we have two related reasons to improve the reporting and analysis of local ecological knowledge. In many instances, descriptions of changing forest structures or landscapes through natural succession can only be made following direct field observations repeated over many years, usually by naturalists and field technicians (e.g., Robertson and Roberts 1982, Crête et al. 2001). Good examples of the systematic collection of this information already exist (e.g., Ecological Monitoring and Assessment Network, [www.eman-rese.ca](http://www.eman-rese.ca), Bird Map Canada, [www.bsc-eoc.org/birdmap\\_e.htm](http://www.bsc-eoc.org/birdmap_e.htm), etc.). These can serve as baselines for new or improved monitoring systems. Similarly, diligent reporting by hunters has allowed efforts to summarize overabundance at a continental scale (e.g., Crête and Daigle 1999), or at local scales (e.g., Mercer and McLaren 2002). Local ecological knowledge is increasingly incorporated into research and management plans (e.g., Ferguson and Messier 1997).

### **Moose in Newfoundland: Some Management Recommendations**

Côté et al. (2004) challenged ecologists and wildlife managers to reduce deer numbers before and not after long-term impacts become difficult to reverse. Angelstam et al. (2000) specifically reviewed management issues involving high moose densities. Mercer (1995: 92) correctly took the posi-

tion for Newfoundland that more emphasis be placed on relationships between hunter-moose-kill and moose density to stabilize populations. In areas of similar habitat, hunter kill density should be similar and proportional to moose density. In one area of Newfoundland, local moose overabundance was effectively managed by temporarily directing resident hunting into the area (McLaren et al. 2000a). In other areas, where resident hunter demand is not high, or in less accessible areas, non-resident hunters might be encouraged to use permanent or temporary hunting camps. Other innovations that may be part of future management of overabundant moose in Newfoundland include implementation of a commercial hunt for the restaurant and luxury export trade and targeted areas of reduction, currently being considered for the national parks.

### CONCLUSION

Considerable interest was generated by the ecological literature discussing when and where a species becomes a “keystone” for its effect on structuring ecosystems (Paine 1995, Power et al. 1996). We suggest that this discussion also be applied to introduced herbivores to determine where the effects of their introduction on the structure and function of native communities accumulate to the extent that they can be described as “wrecking balls”. Understanding and predicting the continuous effects of herbivores on forest ecosystems following disturbance and managing these effects must be part of sustainable forest management.

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