

Impact of deer browsing on plant communities in cutover sites on Anticosti Island¹

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Abstract: Browsing by Anticosti Island's large white-tailed deer (*Odocoileus virginianus*) population reduces the abundance of the deer's preferred species and increases the abundance of less preferred species. This alteration of plant communities can lead to degradation of deer habitat. However, the centres of large clear-cuts may be spared from deer browsing given that this animal's instincts regarding predation risk impel it to remain near escape cover. This project assessed the impact of high deer populations on plant communities in large clear-cuts harvested by cutting with protection of regeneration and soils (CPRS). To this end, vegetation growing within fenced-off areas was compared with that growing in adjacent unfenced plots following application of large CPRS in 1995 and 1996. The results show that deer browsing does not have an impact on the number of plant species but does dramatically reduce total species cover. Furthermore, although the number of species was the same in the fenced and unfenced plots, some species were only found in one or the other type of plot. *Rubus idaeus*, *Betula papyrifera*, *Epilobium angustifolium*, *Abies balsamea*, *Maianthemum canadense*, *Cornus Canadensis*, and *Prunus pensylvanica* were all associated with fenced plots, while the cover of true grasses and *Cirsium arvense* increased in the unfenced plots. We conclude that large CPRS-harvested clear-cuts cannot serve to preserve the initial plant communities on Anticosti Island.

Keywords: browsing, clear-cuts, high density, plant communities, white-tailed deer.

Résumé : Le broutement effectué par la forte population de cerf de Virginie (*Odocoileus virginianus*) sur l'île d'Anticosti tend à diminuer l'abondance des espèces favorites du cerf et augmenter celle d'espèces moins utilisées. Cette modification des communautés végétales pourrait mener à un appauvrissement de l'habitat du cerf. Par ailleurs, le centre de grandes coupes pourrait être épargné du broutement par le cerf étant donné que l'instinct du cerf quant au risque de prédation l'incite à demeurer près du couvert de fuite. Ce projet vise à évaluer l'impact des fortes densités de cerf sur les communautés végétales se développant à la suite de coupes avec protection de la régénération et des sols (CPRS) de grandes dimensions. Pour ce faire, la végétation se développant sur des surfaces clôturées a été comparée à celle de parcelles non clôturées adjacentes à la suite de l'application de grandes CPRS en 1995 et 1996. Les résultats montrent que le broutement du cerf n'a pas d'impact sur le nombre d'espèces végétales, mais qu'il a fortement diminué le recouvrement total des espèces. De plus, même si le nombre d'espèces était le même dans les placettes clôturées et non clôturées, certaines de ces espèces se retrouvaient uniquement dans l'un ou l'autre des 2 types de placette. *Rubus idaeus*, *Betula papyrifera*, *Epilobium angustifolium*, *Abies balsamea*, *Maianthemum canadense*, *Cornus canadensis* et *Prunus pensylvanica* étaient toutes associées aux zones clôturées alors que le recouvrement de graminées et de *Cirsium arvense* augmentait dans les zones non clôturées. Il apparaît donc que les CPRS de grandes dimensions ne permettront pas de conserver les communautés végétales initiales sur l'île d'Anticosti.

Mots clés : broutement, cerf de Virginie, communautés végétales, coupe avec protection de la régénération et des sols, densité élevée.

Nomenclature: Marie-Victorin, 1964; Banfield, 1974.

Introduction

The capacity of high-density populations of large herbivores to disturb ecosystems is well recognized. Deer can significantly impair natural or artificial regeneration of many woody species (Stoekeler, Strothmann & Krefting, 1957; Bellingham & Allan, 2003), which can disrupt supply to forest industries and thus affect the economy of a given region (Tilghman, 1989; Conover, 1997). Deer

browsing can also affect other vegetation, reducing both richness (Tilghman, 1989; Cockayne, 1909 in Bellingham & Allan, 2003; Horsley, Stout & deCalesta, 2003) and cover (Augustine & Frelich, 1998; Potvin & Poirier, 2004) of plant species. In addition, plant species composition can be altered (Anderson & Loucks, 1979; Anderson & Katz, 1993; Gill & Beardall, 2001; Morecroft *et al.*, 2001), and extinction of some species can occur when the deer population is high (Case & McCullough, 1987; Augustine & Frelich, 1998; Cornett *et al.*, 2000; Russell & Fowler, 2004).

White-tailed deer (*Odocoileus virginianus*) populations have been increasing in North America for the past century and are presently at levels that have not been seen for several hundred years (Rooney, 2001). White-tailed deer were introduced on Anticosti Island in the late 19th century. The

¹Rec. 2007-07-10; acc. 2008-02-04.

Associate Editor: Hugo Asselin.

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DOI 10.2980/15-3-3123

current population is estimated at 125 000 deer, and mean density is estimated at more than 15 deer·km⁻² (Potvin & Poirier, 2004) and can be as high as 30 deer·km⁻² in certain locations, which is much higher than the densities recorded elsewhere in Quebec (1–14 deer·km⁻²; Daigle *et al.*, 2004). At these densities, deer have a marked impact on the environment by limiting development of regeneration and modifying the equilibrium of ecosystems (Potvin, Beaupré & Laprise, 2003; Potvin & Poirier, 2004). Excessive browsing on deer's favourite species can shift the composition of plant communities towards species that deer avoid or consume less of and, in the process, degrade the deer habitat.

In Quebec and elsewhere in the world, studies have shown that the centres of large clear-cuts can be spared from deer browsing due to the long distance from the forest edge and the consequent increase in predation risk (Drolet, 1978; Welch *et al.*, 1991; Kay, 1993; Schmitz, 2005). Anticosti Island is characterized by an absence of predators, but the deer's instinct (Byers, 1997) and the long hunting season—nearly 4 months—might lead to similar behaviours. This study is intended to test the impact of high deer populations on plant communities in large clear-cuts harvested by cutting with protection of regeneration and soils (CPRS) on Anticosti Island. The hypotheses to be tested are that 1) deer browsing reduces species richness and total species cover close to forest edge and 2) deer browsing pressure on the environment affects the dominance of plant communities. In accordance with the first hypothesis, we predict that near the forest edge, both species richness and total species cover will be strongly reduced, whereas in the centre of the clear-cuts, such reductions will be less important, especially in the case of species richness. In regard to the second hypothesis, we predict that intensive deer browsing will decrease the presence of highly palatable species and favour the development of non-palatable species. Therefore, we expect to observe a shift in species dominance from a forest plant community to a prairie-like community in intensively browsed areas. These hypotheses will be tested using an experimental design comprising large cutblocks within which fenced areas were established adjacent to unfenced areas.

Methods

STUDY AREA

Anticosti Island is located in the Gulf of St. Lawrence (49° 28' N, 63° 00' W) on the eastern edge of Quebec. The island is 222 km long and has an area of 7943 km². The topography is generally even, with mean altitude being 126 m. The island's deposits are of post-glacial marine origin along the coast and at lower elevations, while the centre is covered with glacial till. Vast areas of organic deposits in peat bogs are found primarily in the eastern part of the island (Potvin & Poirier, 2004). Snowfall is abundant and approximately 20% higher than that measured in the white-tailed deer's range elsewhere in Quebec (Huot, 1982). The climate corresponds to that of the maritime sub-boreal bioclimatic zone, which is characterized by cool summers and relatively mild winters. The mean temperature is close to -11 °C in January and 16 °C in July (Environment Canada, 2006).

Anticosti Island is located in the balsam fir–white birch bioclimatic domain and in the eastern sub-domain of the lower boreal region (Grondin *et al.*, 1996). The primary tree species are balsam fir (*Abies balsamea*), white spruce (*Picea glauca*), and black spruce (*Picea mariana*). Species such as trembling aspen (*Populus tremuloides*), balsam poplar (*Populus balsamifera*), paper birch (*Betula papyrifera*), and tamarack (*Larix laricina*) occur sporadically. Herbaceous vegetation is rich and diversified and dominated by *Clintonia borealis*, *Maianthemum canadense*, *Cornus canadensis*, *Listera cordata*, *Dryopteris spinulosa*, and *Oxalis montana*. The primary cause of forest disturbance is defoliation by hemlock looper (*Lambdina fiscellaria fiscellaria*). A very serious outbreak occurred from 1931 to 1936, and the most recent infestation, on a smaller scale, occurred from 1971 to 1973 (Dorais *et al.*, 1996). Spruce budworm (*Choristoneura fumiferana*), which appeared for the first time in 1973 (Blais, 1983), is an additional source of damage. Windfall is another significant disturbance, while fire is of only secondary importance.

EXPERIMENTAL DESIGN

In 1995 and 1996, 7 square areas of approximately 3 km² were delimited for CPRS clear-cuts without separation strips. These blocks were logged using tree-length harvesting, a technique that leaves logging debris on the site. Prior to felling, 5 to 9 sampling plots were established per block at different distances from the forest edge (from a minimum of 24 m to a maximum of 919 m, Figure 1), for a total of 52 sampling plots. Each plot included a combination of a fenced area (which excluded deer and snowshoe hare but not other small mammals) and an unfenced area, each of which had a circular surface area of approximately 80 m². Fences 3 m in height were installed immediately after logging in 1995 (3 blocks) and 1996 (4 blocks). Each of these areas contained 10 circular quadrats of 4 m². Following significant windfall in 1996, salvage cuts were applied (Figure 1) and the dimensions of certain cutblocks were enlarged, which obliged us to re-measure the distances of the sampling plots from the nearest forest edges.

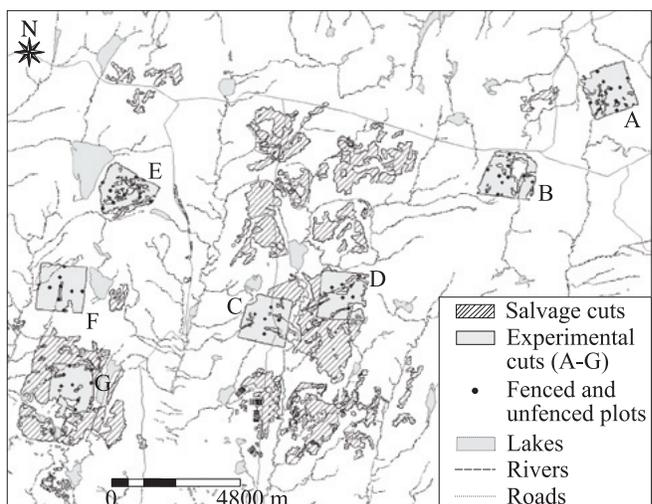


FIGURE 1. Diagram of the spatial arrangement of clear-cut blocks and of plots within each clear-cut block.

Sample plots were surveyed 3 times: soon after cutting (year 0) and 5.5 and 8.5 y after cutting. Because of staff constraints, the first inventory (year 0) was performed 1 to 4 weeks after fencing, which itself was done immediately after cutting. This lag between fencing and the first inventory can explain differences in percent cover of some species at the beginning of the study. The surveys performed in 2001 and in 2004 were respectively 5 or 6 y and 8 or 9 y after cutting, depending on the year the block was logged. The results obtained for these years were pooled because if the years after clear-cutting were taken into account separately, it would result in too many zero values in the data set to perform the statistical analyses. Accordingly, the combined data were assigned to the mean years, that is, 5.5 and 8.5 y after cutting.

VARIABLES MEASURED

The plant diversity survey consisted of assessing the percentage of ground area covered by each species or group of species when only genera were considered. The total percent cover in each quadrat could exceed 100%, as it represented the sum of cover by species of varying heights, which could overlap. The data collected served to calculate species richness, that is, the total number of species found in each quadrat, regardless of cover and species. In addition, total cover for all species combined was calculated in each quadrat. Each species' percent cover was also considered in order to verify whether certain species were more sensitive to browsing than others. When statistical analyses were subsequently performed, a mean of 10 quadrats per plot was used.

To test the floristic similarity between fenced and unfenced areas, the Sørensen similarity index (SI) (Mueller-Dombois & Ellenberg, 1974; Magurran, 2004) was calculated using the number of species observed in each pair of fenced and unfenced plots. This index enabled us to compare fenced and unfenced areas in terms of numbers of species present, on the basis of the following equation:

$$SI = (2c/A + B) \times 100 \quad [1]$$

where c is the number of species common to both areas, while A and B represent the number of species in the fenced and unfenced areas, respectively. A value of 100% signifies that exactly the same species are present in the 2 areas, while a value of 0% indicates that there are no species common to the 2 areas.

STATISTICAL ANALYSES

An analysis of covariance for a repeated measurement design was performed using the MIXED procedure of the SAS software (Milliken & Johnson, 2002) according to the following model:

$$Y_{ijk} = \mu + B_i + (\theta + \alpha_i)d_{ij} + T_{j(i)} + Y_k + \lambda_k d_{ij} + \varepsilon_{ijk} \quad [2]$$

where Y_{ijk} is the difference between fenced and unfenced plots for a given dependent variable measured in block i , transect j , and year k ; μ is the overall mean; B_i is the random effect of block i ; θ is a regression coefficient related to the covariate d_{ij} ; α_i is a regression coefficient related to the interaction between block i and covariate d_{ij} ; d_{ij} is the covariate distance from forest edge; $T_{j(i)}$ is the random

effect of transect j nested within block i ; Y_k is the fixed effect of year k ; λ_k is a regression coefficient related to the interaction between year k and covariate d_{ij} ; and ε_{ijk} is random error.

To take into account the correlation between repeated measurements of a plot, we tested 8 variance-covariance structures (ante-dependence, autoregressive, compound symmetry, factor analytic, heterogeneous compound symmetry, Huynh-Feldt, spatial power, and unstructured). We then selected, for each analysis, the one that minimized the Akaike information criterion (AIC) as suggested by Milliken and Johnson (2002). Thus, we used the ante-dependence structure to analyze difference of percent cover between fenced and unfenced plots and the autoregressive structure to analyze the Sørensen similarity index and the difference in number of species between fenced and unfenced plots. Square-root transformation was used to correct for heteroscedasticity and/or non-normality of the residuals when detected. According to Milliken and Johnson (2002), the model was then simplified by removing fixed-effects variables that were non-significant (beginning with the least significant variable) in order to obtain the best model for explaining the response variables. The fenced and unfenced areas were compared over time while considering the covariate distance from forest edge. The degrees of freedom used by the MIXED procedure were calculated according to the method of Kenward and Roger (1997).

A principal component analysis (Legendre & Legendre, 1984) was performed using the CANOCO software program (ter Braak & Šmilauer, 2002). This multivariate analysis determined the preferences of each plant species according to the 2 most contrasting environmental conditions in our study: fenced and unfenced areas (Lepš & Šmilauer, 1999). The purpose of this analysis was to determine whether deer browsing altered species composition. To this end, data from fenced and unfenced areas 8.5 y after cutting were compared. The variable analyzed was percent cover of each species covering at least 0.5% of the plot (mean of 10 quadrats). Species with less than 0.5% mean cover were eliminated from the analysis as they were of marginal significance within the plant community. This analysis served to create a chart depicting the relationship of species to fenced and unfenced areas and thereby bring out the species most strongly associated with either of these 2 main conditions. To determine if a species was significantly associated with fenced or unfenced areas, we selected the species located close to fenced and unfenced plots in Figure 3 and submitted their percent cover at 8.5 y after cutting as the dependent variable in a model of analysis of covariance. This model was similar to equation [2] except that the fixed effect "year of measurement" was replaced by the fixed effect "fencing". A threshold value $\alpha = 0.05$ was then used to detect a significant effect of the fencing factor, *i.e.*, differences between fenced and unfenced areas.

Results

Differences in species richness ($P = 0.7485$), in total percent cover of species ($P = 0.6610$), and in the Sørensen similarity index ($P = 0.9669$) did not vary as a function of

distance from forest edge (Table I). In addition, the difference in species richness varied over time ($P = 0.0001$), fewer species being observed during the years when the cuts were made (Table II). Moreover, in those years, the number of species in the fenced areas tended to be slightly higher ($P = 0.089$) than in unfenced areas, but the species varied among plots, and this difference was not explained by any particular species. This difference declined over time, with no difference remaining between fenced and unfenced areas 5.5 and 8.5 y after cutting (Table II), at which time the mean number of species was 25.

The calculated Sørensen similarity index values rose from 80.5% immediately after cutting to 90.5% 5.5 y after cutting. The index slipped back to 88.2% 8.5 y after cutting, which means that 11.8% of species were found in only 1 of the 2 main areas (fenced or unfenced).

Total plant species cover was significantly different between fenced and unfenced areas over time ($P < 0.001$), and this difference was pronounced in every year analyzed ($P < 0.001$). The difference in plant species cover between fenced and unfenced areas was approximately 20% immediately after cutting and increased to 70% 8.5 y after cutting (Figure 2).

Principal component analysis brought out a difference in species composition between fenced and unfenced areas (Figure 3). Species falling on the left side of Figure 3 were more closely associated with unfenced areas, while species falling on the right side of Figure 3 were more closely associated with fenced areas. Axis 1 explains 33.1% of species composition variation observed, while axis 2 explains 16.2%. According to analyses of covariance performed to support the visual information of Figure 3, true grasses, Canada thistle (*Cirsium arvense*), and violet (*Viola* sp.) were the species significantly ($P < 0.05$) associated with unfenced plots and, as a result, with deer browsing. Conversely, raspberry (*Rubus idaeus*), paper birch, fireweed (*Epilobium angustifolium*), bunchberry (*Cornus canadensis*), wild lily-of-the-valley (*Maianthemum canadense*), and pin cherry (*Prunus pensylvanica*) were significantly associated with fenced areas, which suggests that these species are sensitive

to deer browsing. The strong association of balsam fir with the fenced area (Figure 3) shows that deer also exert significant pressure on this species.

Table III presents a more detailed analysis of mean percent cover for each species in the fenced and unfenced areas. Among species with the most significant cover, the highest values are generally found in fenced areas 8.5 y after cutting (e.g., BEP, PRP, CON, EPA, RUI, LIB, MAC, POT, RUP, and ABB). Only 3 species (CIC, PIG, and GRS) associated with unfenced areas had significant cover at that time. The increase in total cover in fenced areas (Figure 2) is essentially explained by the growth of a few species. In addition, mean total cover of all species had a higher value in fenced areas (176.1%) than in unfenced areas (98.6%) (Table III). Violet (VIS), which is more associated with unfenced areas (Figure 3), had very minimal cover in both areas. The increase in total percent cover between 0 and 8.5 y after cutting in both fenced and unfenced areas is explained by the disappearance of tree cover, which produces a pronounced increase in the light available for herb and shrub strata.

Discussion

IMPACT OF DEER ON RICHNESS AND DIVERSITY OF PLANT SPECIES

Plant diversity was not influenced by proximity of forest edge. The number of species, total species cover, and Sørensen similarity index were all unaffected by distance from escape cover. The initial hypothesis that plant diversity in the centres of large clear-cuts would be spared from deer browsing was thus disproved. This hypothesis was also partially disproved by Casabon and Pothier (2007), who considered the specific case of balsam fir regeneration.

Contrary to our initial hypothesis, deer did not have a significant medium-term effect on plant species richness, as the number of species was roughly the same in fenced and unfenced areas (Table II). This finding is borne out by the high Sørensen similarity index values for 5.5 and 8.5 y after cutting (90.5% and 88.2%, respectively), which seem to support the deduction that deer have little impact on species composition regardless of the type of area con-

TABLE I. Results of analyses of covariance using the SAS MIXED procedure prior to removing non-significant terms.

Sources of variation	Difference in species richness			Difference in percent cover			Sørensen similarity index		
	Degrees of freedom num/den	F value	P	Degrees of freedom num/den	F value	P	Degrees of freedom num/den	F value	P
Distance from forest edge (D)	1/49	0.10	0.7485	1/49.8	0.19	0.6610	1/49.6	0.00	0.9669
Year (Y)	2/67.3	10.07	0.0001	2/49	28.84	< 0.0001	2/72	0.45	0.6401
Y × D	2/67.3	0.89	0.4157	2/49	1.04	0.3614	2/72	0.09	0.9098

TABLE II. Significance level (P) of the fencing effect on the number of species as calculated from an analysis of covariance performed with the difference of number of species between fenced and unfenced plots as the dependent variable.

Years after cutting	P	Average number of species (\pm SD) in fenced areas	Average number of species (\pm SD) in unfenced areas
0	0.089	23.9 \pm 6.9	21.4 \pm 5.9
5.5	0.730	25.3 \pm 6.7	24.7 \pm 5.6
8.5	0.521	25.1 \pm 6.5	25.7 \pm 5.7

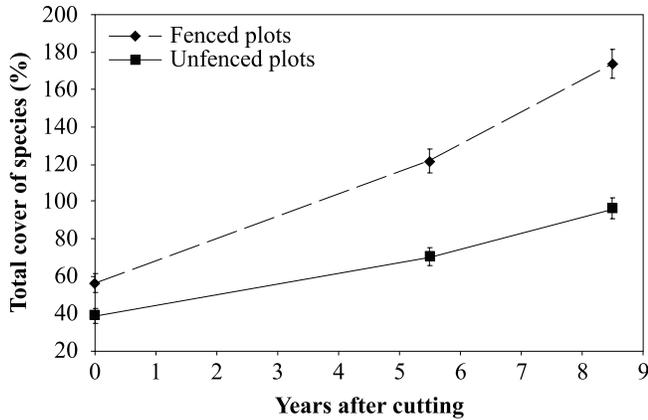


FIGURE 2. Change in total percent cover of species in fenced and unfenced areas. Vertical lines represent standard error.

cerned. However, these same SI values also mean that 10–12% of species are unique to either fenced or unfenced areas, a proportion that can be considered to be quite large. Nevertheless, because this 10–12% of species is equally distributed between fenced and unfenced areas, it cannot indicate a trend towards a change in species composition.

The absence of any impact on number of plant species by deer fits with the findings of other studies showing that in open environments or clear-cuts, the mean number of plant species does not vary significantly between areas with and without deer access, whenever this animal is present in high densities (Bowers, 1997; Bergquist, Örlander & Nilsson, 1999). The same type of result was observed under the cover of mature stands subject to the impacts of various herbivores (McInnes *et al.*, 1992; Potvin & Poirier, 2004; Schmitz, 2005).

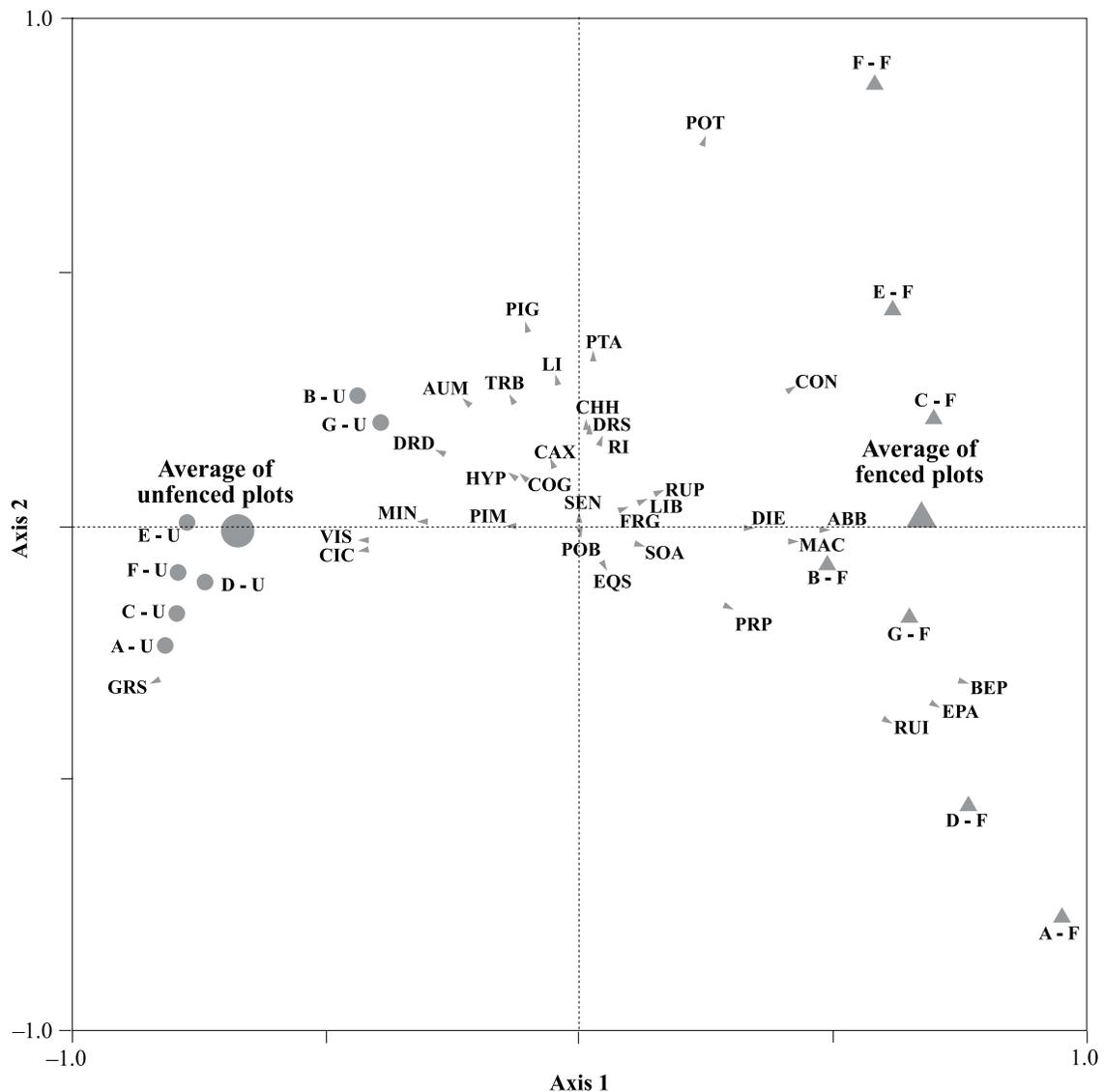


FIGURE 3. Distribution of species with an average cover of 0.5% or more in fenced and unfenced areas 8.5 y after cutting as calculated using CANOCO software program's principal component analysis (refer to codes in Table III). Axis 1 is the horizontal axis and explains 33.1% of species variability, while Axis 2 is the vertical axis and explains 16.2% of the variability. The site names consist of a letter for each block (clearcuts A, B, C, D, E, F, and G) followed by the letter F if it is a fenced area (triangle) or U if it is an unfenced area (circle). There is one fenced area and one unfenced area for each block. As there are 7 blocks, there are 14 points representing areas. The position of the average of fenced areas and of unfenced areas is also identified.

TABLE III. Mean percent cover of species covering 0.5% or more on average in fenced and unfenced areas 8.5 y after cutting.

Species	Latin name	Code	Immediately after cutting		8.5 y after cutting	
			Fenced area	Unfenced area	Fenced area	Unfenced area
Other mosses		AUM	1.7	1.8	0.7	2.1
Paper birch	<i>Betula papyrifera</i>	BEP	2.5	0.5	24.3	0.5
Carex sp.	<i>Carex</i> sp.	CAX	2.3	3.4	1.6	1.7
Pin cherry	<i>Prunus pensylvanica</i>	PRP	0.6	0.0	4.8	0.0
Canada thistle	<i>Cirsium arvense</i>	CIC	1.6	2.4	1.4	8.9
Creeping snowberry	<i>Gaultheria hispidula</i>	CHH	0.2	0.3	1.2	0.3
Goldthread	<i>Coptis groenlandica</i>	COG	1.4	1.5	2.2	2.3
Bunchberry	<i>Cornus canadensis</i>	CON	2.9	1.1	9.8	1.6
Bush honeysuckle	<i>Diervilla lonicera</i>	DIE	0.2	0.0	2.4	0.1
Oak fern	<i>Dryopteris disjuncta</i>	DRD	0.9	1.0	0.6	1.4
Spinulose wood fern	<i>Dryopteris spinulosa</i>	DRS	0.8	0.3	0.8	0.7
Fireweed	<i>Epilobium angustifolium</i>	EPA	6.7	0.1	25.7	0.2
White spruce	<i>Picea glauca</i>	PIG	4.8	4.7	13.5	21.0
Black spruce	<i>Picea mariana</i>	PIM	0.9	1.4	4.6	5.5
Strawberry sp.	<i>Fragaria</i> sp.	FRG	1.6	0.2	2.6	0.7
Raspberry	<i>Rubus idaeus</i>	RUI	1.8	0.2	13.6	0.2
Currant sp.	<i>Ribes</i> sp.	RI	0.2	0.0	0.8	0.2
True grass sp.		GRS	5.5	8.8	7.5	37.6
Bunchberry moss sp.		HYP	3.1	4.6	0.9	2.9
Lichen sp.		LI	0.1	0.1	0.7	0.8
Twinflower	<i>Linnaea borealis</i>	LIB	0.7	0.2	6.0	0.3
Wild lily-of-the-valley	<i>Maianthemum canadense</i>	MAC	2.6	1.2	5.4	1.3
Naked miterwort	<i>Mitella nuda</i>	MIN	1.8	2.1	0.9	2.0
Balsam poplar	<i>Populus balsamifera</i>	POB	0.7	0.0	2.3	0.0
Trembling aspen	<i>Populus tremuloides</i>	POT	3.1	0.6	13.3	0.0
Horsetail sp.	<i>Equisetum</i> sp.	EQS	0.3	0.1	1.1	0.3
Bracken	<i>Pteridium aquilinum</i>	PTA	0.8	0.0	1.5	0.5
Hairy raspberry	<i>Rubus pubescens</i>	RUP	1.0	0.3	7.4	0.5
Balsam fir	<i>Abies balsamea</i>	ABB	1.6	0.8	11.6	0.5
Groundsel sp.	<i>Senecio</i> sp.	SEN	0.4	0.0	0.9	0.0
American mountain ash	<i>Sorbus americana</i>	SOA	0.1	0.0	1.1	0.0
Northern starflower	<i>Trientalis borealis</i>	TRB	1.5	0.5	0.3	0.5
Violet sp.	<i>Viola</i> sp.	VIS	0.5	0.7	0.3	0.7
Other species			3.4	1.7	4.5	3.2
Mean total cover			58.3	40.5	176.1	98.6

Note: Species in bold characters are those statistically associated with fenced or unfenced plots.

In contrast, some authors have shown that deer browsing reduces the richness of herbaceous, shrub (Cockayne, 1909 in Bellingham & Allan, 2003), and tree (Stewart & Burrows, 1989; Tilghman, 1989; deCalesta, 1997; Gill & Beardall, 2001; Horsley, Stout & deCalesta, 2003) species. This loss of richness is greater when the impact of high deer densities is combined with certain local ecological conditions, such as moisture, light, and nutrient regimes (Gill & Beardall, 2001). For instance, Tilghman (1989) emphasized that dense fern cover coupled with the presence of high deer densities in clear-cuts can affect the growth and survival of tree seedlings and thereby reduce species richness. However, this combination of high deer density and restrictive ecological conditions does not apply to Anticosti Island, which may explain why plant species richness did not decrease.

Our findings do show that immediately after cutting, the number of species tended to be higher in fenced areas than in unfenced areas. However, only 3 more species were counted in fenced areas than in unfenced areas, and these species were minimally represented. While fences were installed immediately after cutting, the regeneration inventory in both fenced and unfenced plots was delayed from 1 to up to 4 weeks after cutting because of staff constraints. This period of time between fence installation and regenera-

tion inventory may have been sufficient for deer to browse partially or totally some individuals, reducing the likelihood of observing some species in unfenced areas. Moreover, the large quantity of woody debris over the ground shortly after cutting may have contributed to these observation-related difficulties. Finally, the appearance of a species in a fenced area (but not in an unfenced area), as in the case of red baneberry (*Actea rubra*) and red-osier dogwood (*Cornus stolonifera*), could simply have been a result of chance due to the relatively small size of the regeneration plots.

For every year surveyed, deer exclusion was associated with plant cover, which was significantly greater than that in the open plots (Figure 2). Thus, although the number of species is similar, their cover is greatly affected by the presence of deer in this habitat (Table III). The same result was obtained by Potvin and Poirier (2004) in a mature fir stand on Anticosti Island. This result may be explained by deer browsing combined with trampling, which can compact the soil or cause death directly (Vare, Ohtonen & Mikkola, 1996; Persson, Danell & Bergström, 2000; Kirby, 2001; Watkinson, Riding & Cowie, 2001).

These differences in species composition between deer exclusion areas and areas where deer have free run have already been noted by several authors (Anderson & Loucks, 1979; Anderson & Katz, 1993; Gill & Beardall, 2001;

Morecroft *et al.*, 2001). The species preferred by deer grow scarce or disappear and are replaced by less sought after species (Case & McCullough, 1987; Stewart & Burrows, 1989; deCalesta, 1997; Cooke & Farrell, 2001; Reyes & Vasseur, 2003; Rooney & Waller, 2003), and the deer's diet changes in accordance with this alteration in vegetation (Veblen *et al.*, 1989; Cross, 1998). In this study, the species sensitive to the presence of deer and fated to disappear if deer densities are not reduced over time are paper birch, fireweed, raspberry, balsam fir, wild lily-of-the-valley, bunchberry, and pin cherry (Figure 3; Table III). Other authors have also noted the pronounced negative impact of deer on paper birch (Horsley, Stout & deCalesta, 2003), fireweed (Bergquist, Örlander & Nilsson, 1999), raspberry (Cross, 1998; Kirby, 2001; Horsley, Stout & deCalesta, 2003), balsam fir (Cornett *et al.*, 2000), wild lily-of-the-valley (Waller & Alverson, 1997; Rooney, 1997), bunchberry (Spalinger, Hanley & Robbins, 1988), and pin cherry (Behrend & Patric, 1969). Several other species found on Anticosti Island have proven to be very sensitive to the presence of deer and its impact on the environment: bush honeysuckle (*Diervilla lonicera*) (Kohn & Mooty, 1971), blackberry sp. (*Rubus* spp.) (Tilghman, 1989), wild sarsaparilla (*Aralia nudicaulis*), and yellow clintonia (*Clintonia borealis*) (Waller & Alverson, 1997).

On the other hand, certain species benefited from the presence of deer in the large clear-cuts studied and became more abundant. This is particularly true of grasses and Canada thistle, which were present in large numbers (Table III). These findings agree with those of Horsley, Stout, and deCalesta (2003), who noted a linear increase in percent cover of grasses with deer density in clear-cuts. According to Kirby (2001), grasses, with their basal meristem and high silica content, are often found in locations accessible to deer because among the higher plants, they are best equipped to endure continual browsing (Marie-Victorin *et al.*, 1995). In addition, grasses can take advantage of the elimination of many individual plants from species subject to strong deer browsing to spread and occupy larger areas. In contrast, the dense vegetation in the fenced areas did not permit the expansion of grasses, which therefore occurred in smaller numbers. Canada thistle is a shade-intolerant species that grows on soils having average to good drainage. Once established, it spreads by rhizomes (Marie-Victorin *et al.*, 1995). The fairly dense vegetation in the fenced areas produces light and moisture conditions that are not favourable to the establishment and growth of this species. In contrast, the strong light conditions in heavily deer-browsed areas encourage proliferation of thistle. Violet is also associated with unfenced areas (Figure 3), but its minimal percent cover suggests that its presence is more likely due to chance than to deer impact (Table III).

REVERSIBILITY AND LOCAL EXTINCTION OF PLANT SPECIES

Given that paper birch, fireweed, raspberry, and pin cherry are seriously impacted by deer, it is worth considering whether these species are destined to disappear and, if so, whether this disappearance is reversible. Could they recolonize the area following some future reduction in deer densities? Paper birch and fireweed seeds can travel many kilometres, which should enable them to recolonize the

area as long as some individual plants are present on the island. Raspberry and pin cherry seeds remain viable for many decades (Marquis, 1975), and in the case of raspberry, asexual reproduction enables this species to spread quickly once established (Whittle, Duchesne & Needham, 1997). Furthermore, it has already been noted that disappearance of raspberry is easily reversible (Horsley & Marquis, 1983). As a result, as long as some individuals survive the deer or there is a seedbank in the soil in the case of raspberry and pin cherry, these species should be able to recolonize the area following a decline in deer density (Horsley, Stout & deCalesta, 2003).

However, balsam fir's seeds only disperse a short distance (Brandner, Peterson & Risenhoover, 1990; Hughes & Bechtel, 1997), which makes it difficult for this species to recolonize areas after destruction of the seedling bank present subsequent to clear-cutting. Plans for artificial reforestation will have to be laid to compensate for the possible lack of natural regeneration in the area once the deer population has been reduced. Recolonization of wild lily-of-the-valley and bunchberry is also likely to be difficult because although they are both rhizomatous plants that can spread vegetatively across the forest floor, their seeds are short-lived, with distance-limited dispersal (Silva, Kana & Solbrig, 1982; Burger, 1987). According to Anderson and Katz (1993), the time required to re-establish the initial biodiversity of a stand newly protected from deer is proportional to the length of time during which the forest was subject to intense browsing pressure, which on Anticosti Island amounts to a period of several decades. In addition, if deer densities are not reduced in the short term, certain species could become locally extinct. Examples of this phenomenon are found in the literature, particularly in the case of trillium (Augustine & Frelich, 1998) and hemlock (Case & McCullough, 1987).

Conclusion

Our study has shown that Anticosti Island's characteristic plant communities cannot be maintained at current deer population levels in large CPRS-harvested clear-cuts. Deer use the entire clear-cut and are not influenced by distance from escape cover. The deer profoundly disturb the ecosystem by modifying the mix of plant species within their reach. The large deer population and the uncertainty surrounding feed sources explain the failure of large clear-cuts to preserve and perpetuate Anticosti Island's initial ecosystems. In order for the plant species initially found on the island to be maintained, deer densities must be reduced. Once the deer population has fallen, it should be possible to preserve the plant succession dynamics usually observed following the cutting of fir stands. However, such measures may not be necessary in regions where predators regulate deer populations, unlike Anticosti Island.

Acknowledgements

We would like to thank the Natural Sciences and Engineering Research Council of Canada, Produits forestiers Anticosti, and the Ministère des Ressources naturelles et de la Faune du Québec for funding the project. We would also like to thank A. Goupil for technical assistance in the field and N. Thiffault, V. Roy,

H. Asselin, and two anonymous reviewers for having revised earlier versions of this manuscript.

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