MAMMALIAN ABUNDANCE IN RIPARIAN AND ADJACENT TERRESTRIAL HABITATS PRIOR TO IMMEDIATELY FOLLOWING FOREST HARVESTING IN THE COPPER LAKE WATERSHED, NEWFOUNDLAND

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Mammalian abundance in riparian and adjacent terrestrial habitats prior to and immediately following forest harvesting in the Copper Lake watershed, Newfoundland

by

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A thesis submitted to the School of Graduate Studies
in partial fulfillment of the requirements
for the degree of Master of Science

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September, 1998

St. John's, Newfoundland

Abstract

The Copper Lake Buffer Zone Study was initiated in 1994 to evaluate the effects of forest harvesting practices on mammalian, avian and piscine assemblages in a small watershed. This portion of the Copper Lake study evaluates mammalian abundance in riparian habitats both prior to and immediately following forest harvesting. Three headwater streams within the Copper Lake watershed received different harvesting treatments; i.e., harvested with no riparian buffer, harvested with a 20 m riparian buffer and not harvested (essentially a buffer of at least 100 m). The riparian habitats and adjacent interior forests on both sides of each stream were evaluated for mammal usage prior to and after harvesting.

Summer trapping studies of small mammals, i.e., masked shrews (*Sorex cinereus*), meadow voles (*Microtus pennsylvanicus*) and deer mice (*Peromyscus maniculatus*), prior to harvesting indicated that these species were initially low in abundance. Shrew abundances showed a large increase in the summer of 1995 in both harvested and unharvested study sites. Post-harvesting data from the summers of 1995 and 1996 indicated no noticeable, immediate effect of harvesting on *S. cinereus* when the clear-cut sites were compared to uncut sites. Within the 20 m buffers, however, shrew abundances were significantly higher than in the adjacent clear-cuts. *M. pennsylvanicus* and *P. maniculatus* showed slight increases in harvested areas. The cyclical nature of such small mammal species may account, in part, for these increases, particularly since abundances were so low in 1994.

Winter track data for other mammalian species, i.e., pine marten (*Martes americana*), weasel (*Mustela erminea*), red fox (*Vulpes vulpes*), snowshoe hare (*Lepus americanus*) and red squirrel (*Tamiasciurus hudsonicus*), indicated that prior to harvesting, winter activity was generally higher in the forest than at the stream edges. After harvesting occurred, I found that more activity occurred in residual patches of forest, including riparian buffers, than in the clear-cuts. Marten, hare and squirrel tracks were significantly more abundant on forested transects. No marten tracks were observed on the clear-cuts without a buffer, while track abundances increased on the uncut sites. The results of this study raise the question of whether a 20 m buffer is adequate to support increased competition for space and prey due to species packing, especially considering the low diversity and densities of small mammals in insular Newfoundland. This can also be compounded with the potential loss of buffer habitat over time due to tree blowdown.

Acknowledgements

I would like to acknowledge the following people and organizations, without whom/which this study would not have been possible. Many thanks to my former supervisor, the late Dr. G.I. McT. Cowan, whose continuous support and generosity will not be forgotten. Thanks to my current supervisors, Dr. P.J. Scott and E.M. Baggs, for taking over where Dr. Cowan left off and for their considerable time, effort and patience in helping me prepare this thesis. Thanks again to E.M. Baggs for his help in the field, for continuing this project after my research was completed and for many supplies. Advice and suggestions from my other committee members, Drs. W. Threlfall and W.A. Montevecchi, were invaluable, particularly in the initial stages of this study and in reviewing the thesis. Thanks also to Dr. D.C. Schneider for statistical advice throughout this study.

Funding for this study was provided by the Newfoundland Forest Service, through L. Moores (for stipend, supplies, field assistants, and winter data collection) and the Wildlife Division through J. Brazil and C. Butler (for summer field assistants), Government of Newfoundland and Labrador. Corner Brook Pulp and Paper Limited, through M. Churchill and P. Tompkins, provided the loggers and equipment for harvesting treatments as described in the study design. Also thank you to the Biology Department, Memorial University, for providing me with additional support, when this study had to be continued for an additional field season.

M. Ryan (Environment Canada), J. Brazil (Wildlife Division) and L. Moores (Newfoundland Forest Service) were instrumental in the development and original design of this study. Thank you for getting me involved and for helping me to secure funding.

The help of field assistants D. Mercer, J. Hare and H. Gardner was greatly appreciated. Additional field assistance was provided by E. Rolls (Newfoundland Forest Service), P. O' Keefe (Western Newfoundland Model Forest), G. Fifield, D. Delaney, B. Wells and G. Kitchen (latter individuals from Newfoundland Forest Service). Thanks also to the Wildlife Division staff in Pasadena, particularly C. Maloney and H. Frye, for help in establishing the field camp and setting up the original winter track transects. Many thanks to G. Jennings and assistants for collecting the mammal track data each winter. Thank you to B. Sturtevant for loaning me snap-traps when I needed extras on short order.

Special thanks go to the rest of the camp crew, J. McCarthy, D. Whitaker, J. Benoit, S. Balsom and J. Bishop, for their help in the field and for their company. Also to my Aunt Norma and Uncle Eric in Pasadena for providing me with a home away from camp and especially to my parents for their continuing support and encouragement.

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Introduction

Riparian habitats are those that border the shorelines of lakes, rivers and streams, forming the transitional zones between aquatic and terrestrial ecosystems (Gregory *et al.* 1991, Stocek 1994). They have been studied from a variety of perspectives resulting in a diversity of definitions (Gregory *et al.* 1991). For example, a riparian zone can be defined as the immediate water's edge with a distinct community of plants and animals, the banks and floodplain of a stream, or the band of forest that either significantly influences or is significantly influenced by the stream ecosystem (Hunter 1990). Defining the limits of a riparian zone is not easily done (Gregory *et al.* 1991, Binford and Buchenau 1993), but the boundaries extend outward from the stream to the limits of flooding and vertically into the streamside vegetation canopy (Gregory *et al.* 1991).

Ewel (1978) stated that riparian zones have two essential characteristics: laterally flowing water that rises and falls at least once within a growing season and a high degree of connectedness with other ecosystems. Riparian zones are also characteristically patchy with high species diversity and productivity (Gregory *et al.* 1991, LaRue *et al.* 1995). This is often due to their ecotonal nature which produces maximum edge conditions within a small area (Thomas *et al.* 1979, Parker 1993). The linear nature of streams and river valleys plus their associated disturbance regimes also contribute to the patchiness of riparian zones (Gregory *et al.* 1991).

Much of the research on riparian zones in North America has been based on studies in arid western regions where there is a sharp contrast between the vegetation of the riparian zone and that of the adjacent area (Cross 1985, LaRue *et al.* 1995). In the boreal forests of the northeast, riparian zones are not as clearly defined (Cross 1985, Stocek 1994, LaRue *et al.* 1995). This is because soil water is not as limiting (LaRue *et al.* 1995) and therefore the vegetation of riparian and upland areas tends to blend and create a broader ecotone (Cross 1985).

The uniqueness and importance of riparian zones to terrestrial and aquatic ecosystems are well documented (e.g., Thomas *et al.* 1979, Cross 1985, Pike and Racey 1989, Hunter 1990, Parker 1993, LaRue *et al.* 1995). There are many reasons why riparian zones have ecological and economical values. They can protect streams from excessive temperature increases, siltation and nutrient leaching resulting from terrestrial disturbances (Corbett and Lynch 1985) and thus help preserve water quality and fish habitat (Pike and Racey 1989, Parker 1993). Often identified as prime wildlife habitat (LaRue 1995), riparian zones provide a source of food, cover and water for a variety of animal species (Thomas *et al.* 1979). For example, the nutrient-rich, productive vegetation of these areas produces quality forage for browsing animals, and the presence of water provides habitat for species such as beaver and otter which require both aquatic and terrestrial habitat features (Stocek 1994). Riparian zones are also used by wildlife as migration routes (Thomas *et al.* 1979), travel corridors (Gregory *et al.* 1991) and refugia (Spackman and Hughes 1995).

Riparian zones provide productive stands for harvesting timber (Cross 1985, Binford and Buchenau 1993) and are noted for their scenic and recreational values

(Thomas *et al.* 1979, Parker 1993). Thus, they represent a potential area of conflict for a variety of resource users (Thomas *et al.* 1979).

Riparian zones are viewed as sensitive environments (Thomas *et al.* 1979, Stocek 1994) and can be affected by landscape changes caused by physical and biological disturbances (Wissmar and Swanson 1990). Examples of natural disturbances include removal of riparian vegetation by flash floods and landslides, fire, wind, plant disease and insect outbreaks (Wissmar and Swanson 1990, Gregory *et al.* 1991). Human influences include stream channelization (Beschta and Platts 1986), road construction, agricultural practices, recreational use and forest harvesting practices (Thomas *et al.* 1979, Wissmar and Swanson 1990).

Intensive forest harvesting activities have been ongoing in insular Newfoundland since the early 1900s, and much of the merchantable timber in the province occurs in association with riparian habitats (Scruton *et al.* 1995). Forest harvesting in these areas can have a variety of effects. For example, water quality and quantity are often affected. The more significant impacts on water quality involve increases in water temperature, turbidity and sediment levels, and concentration of dissolved nutrients (Corbett and Lynch 1985). Harvesting trees can also increase water output from watersheds (Brown and Beschta 1985, Pike and Racey 1989, Neary and Hornbeck 1994).

Landscape fragmentation results from forest harvesting and can dramatically change the habitat value for some wildlife species while at the same time create new habitats and opportunities for more tolerant species (Freedman 1982, Neary and Hornbeck

1994). Forest floor microclimatic characteristics influenced by light, moisture and temperature, are often changed (Morrison *et al.* 1992) at edges and this may extend for tens of meters into adjacent interior habitats. Such changes can alter the native plant and animal communities present (Collinge 1996). In addition, the availability of cover can change, as well as the types and quality of foodbase (Morrison *et al.* 1992). Competition and predation can be increased through bringing together species which normally have little contact (Morrison *et al.* 1992).

Legislation concerning forest harvesting in riparian habitats often requires that a strip of uncut trees, *i.e.*, a buffer zone, be left along the edge of the water body. Although these riparian buffers are recognized for their usefulness in minimizing the effects of forest harvesting on aquatic and terrestrial habitats, the width necessary for them to be effective is controversial (Freedman 1982). Corbett and Lynch (1985) stated that a 40 ft (12.2 m) riparian buffer may be adequate to prevent temperature increases in small streams, but in order to protect the stream ecosystem a zone of 66 (20.1 m) to 100 ft (30.5 m) is usually required. Within mature, northern hardwood forests in Vermont, Spackman and Hughes (1995) found that the effective riparian buffer width for plant diversity around third and fourth order streams varied among sites. For example, at one site a riparian buffer of at least 30 m above the high water mark was necessary to retain over 90% of the vascular plant species, while at another site a buffer of only 10 to 15 m was needed. They also determined that corridor widths of 150 to 175 m were necessary to include 90 to 95% of the bird species. Darveau *et al.* (1995) found evidence that a 60 m riparian buffer was required for forest-dwelling birds in Quebec. In Newfoundland, Whitaker and Montevecchi (1997) found that many interior forest birds were rare along riparian edges and stated that preserved contiguous blocks of interior forest are necessary in addition to the riparian buffer. For resident squirrel populations in young pine plantations of the southern United States, Dickson (1989) stated that a riparian buffer of at least 50 m is necessary to meet minimal habitat requirements.

Timber harvesting guidelines for riparian buffers along waterways also vary from jurisdiction to jurisdiction. In Newfoundland, all waterways appearing on a 1:50 000 topographic map require a 20 m wide, no harvest buffer when the slope is less than 30% and 12 m + (1.5 x % slope) when the slope is greater than 30% (Newfoundland Forest Service 1993). In special cases, such as protected water supplies, wider, no harvest buffers are required (Newfoundland Forest Service 1993). Similar guidelines are in place for Nova Scotia (Towers and Milton 1990), while in New Brunswick 30 m buffers are required on each stream side, with 75 m buffers on protected water supplies (Hooper 1994). In Oregon, riparian buffers must be three times the width of the stream but not less than 8 m or greater than 31 m (Brown 1994). Such variation in recommended/required riparian buffer widths results from the great variety in riparian zones from region to region making it difficult to develop guidelines that are appropriate in all situations. In addition, riparian buffers are often subject to tree blowdown which can reduce their effectiveness over time (Steinblums *et al.* 1984) and add to the difficulty of determining adequate buffer widths. In order to develop appropriate harvesting guidelines for the boreal forest, there is a need for ecosystem-specific data for wildlife habitat usage that consider local conditions.

The majority of studies on mammalian responses to forest harvesting involves small mammals such as rodents and insectivores. In a review of 21 studies, Kirkland (1990) identified a general pattern of positive initial response of small mammals to conventional clear-cutting of temperate North American forests. Kirkland (1977) found that in northern Appalachian forests of West Virginia small mammals initially increased after harvesting but such changes were not long-lasting. Six to fifteen years after harvesting, small mammal abundance and diversity decreased. Similar results were obtained by Parker (1989) in conifer plantations of New Brunswick - small mammals showed an initial increase in abundance followed by a prolonged decline as microhabitat changed over time (lowest densities in plantations aged 15 - 17 years). In clear-cuts of black spruce forests of northern Ontario the most obvious change was in small mammal species composition rather than density (Martell and Radvanyi 1977). Red-backed voles (*Clethrionomys gapperi*) were common on uncut stands and predominated initially (up to two years) on the new clearcuts, but then declined rapidly, while deer mice (*Peromyscus maniculatus*) were uncommon prior to harvesting and subsequently predominated two years after harvesting occurred.

Small mammal populations in temperate forests characteristically exhibit seasonal and/or multi-annual fluctuations (Kirkland 1990). Microtine rodents are well-known for their cyclical nature (Elton 1965, Batzli 1992), while *Peromyscus* spp. typically exhibit constant low population densities without major yearly fluctuations (Terman 1968, Batzli 1992). Soricid population densities show both considerable seasonal and annual variation, but rarely exhibit the regular peaks and crashes as seen in the microtines (Churchfield 1990). Any changes in small mammal abundance subsequent to clear-cutting may be reflective of natural population fluctuations and such fluctuations should, therefore, be acknowledged.

Microtine rodents occupy a variety of habitats from woodlands to meadows (Whitaker 1992) and have diets consisting mainly of grasses, sedges and other herbaceous vegetation (Northcott 1980, Banfield 1987). Microtines generally show increased abundance on recent clear-cuts compared with uncut forests, with the species most responsible being the red-backed vole (*Clethrionomys gapperi*) (Kirkland 1977, Martell and Radvanyi 1977, Martell 1983, Kirkland 1990). Meadow voles (*Microtus pennsylvanicus*) are often trapped in higher numbers on clear-cuts where favourable wet, grassy patches are created (Ramirez and Hornocker 1981, Swan *et al.* 1984, Probst and Rakstad 1987). Possible factors favouring an increase in microtines on recent clear-cuts include increased herbaceous resources and/or slash cover following clear-cutting (Kirkland 1977, Martell 1983, Yahner 1986).

Deer mice (*Peromyscus maniculatus*) exhibit a broad tolerance for different habitats, from grasslands to coniferous forests, with the requirement that the habitat be dry (Banfield 1987). They are primarily seed eaters, but also eat animal prey, such as insects, insect eggs and larvae, and spiders (Banfield 1987). Deer mice tend to show a pattern of increase in abundance following clear-cutting in coniferous forests and a decrease in abundance following clear-cutting of deciduous forests (Kirkland 1977, Kirkland 1990). Martell (1983) found that deer mice increased on scarified clear-cuts over the first three summers after harvest, while on unscarified clear-cuts they remained rare for two years following clear-cutting. Martell and Radvanyi (1977) found deer mice to be scarce at the end of the first summer after clear-cutting but that they increased by the end of the second and remained high in the third. Deer mice numbers in clear-cuts are affected by the amount of fruit and seed-producing vegetation present (Ramirez and Hornocker 1981).

Soricids are found in a variety of habitats, with the presence of moisture being a major factor influencing their distribution (Getz 1961). Areas with ground cover such as leaves, stumps and herbaceous vegetation seem to be ideal habitats (Banfield 1987) because cover promotes high humidity conditions (Getz 1961). Shrews primarily eat invertebrates, but also include plants, fungi and small mammals in their diet (Whitaker 1992). They show varied responses to forest harvesting but the general pattern is an increase in shrew numbers on recent clear-cuts (Kirkland 1990). Martell (1983) found significantly more soricids on unscarified clear-cuts in Ontario than on scarified clear-cuts or uncut sites. In Nova Scotia, Swan *et al.* (1984) found that although shrew numbers were low during their study, the greatest abundance of shrews occurred on clear-cuts. In contrast, Sullivan and Sullivan (1982) found that in coniferous forests of British Columbia shrew density was consistently higher in the forest than on the clear-cuts. Because shrews are primarily insectivorous they should not be as directly affected by changing plant communities as are herbivores (Sullivan and Sullivan 1982). An increase in shrew abundance after clear-cutting may be a direct response to increased invertebrate prey on the clear-cut (Martell 1983).

Sciurids also show mixed responses to clear-cutting. Kirkland (1977) found that sciurids were excluded from clear-cuts in both coniferous and deciduous forest zones. Thompson *et al.* (1989) also found that red squirrels (*Tamiasciurus hudsonicus*) were more abundant in the uncut forest. For tree-dwelling sciurids, the loss of nest sites and shelter from clear-cutting may be the cause of this trend (Kirkland 1990). Ground-dwelling species such as the chipmunk (*Tamias* spp., *Eutamias* spp.) have been reported to increase in number following clear-cutting (Kirkland 1990). Martell and Radvanyi (1977) trapped plots for three years after clear-cutting occurred and found that least chipmunks (*Eutamias minimus*) did not appear on plots until the second summer. By the third summer after clear-cutting chipmunk numbers were relatively stable.

There are fewer studies addressing the responses of other small mammalian herbivores and carnivores to forest harvesting. Snowshoe hare (*Lepus americanus*) eat grasses, forbs, tree leaves, plus tree buds, twigs and bark in the winter (Banfield 1987). They are found in various habitats, e.g., forests, swamps and riverside thickets (Banfield 1987). Monthey (1986) and Thompson *et al.* (1989) found higher snowshoe hare activity in regenerating clear-cuts than in undisturbed forests, however, the lowest amount of activity occurred immediately after cutting (Monthey 1986, Thompson *et al.* 1989).

Gashwiler (1970) observed that hares near forest edges moved out into clear-cuts when cover was sufficient and resided in available patches of suitable cover. Use of older clear-cuts by snowshoe hares is possibly related to more browse being available after harvesting (Monthey 1986).

For species inhabiting climax coniferous forests, e.g., the pine marten (*Martes americana*), forest harvesting can have negative effects. In Maine, Soutiere (1979) found that clear-cuts were poor habitat for marten and marten densities in these areas were about one-third those of mature forests. For Ontario, Thompson (1991) reported marten densities in clear-cuts to be 67 to 90% lower than in uncut, mature stands. One reason for marten preference of mature forests is that they require substantial vertical structure (Thompson and Curran 1995) and this is not available in recently cut, second-growth stands (Sturtevant *et al.* 1996).

Thompson *et al.* (1989) studied winter tracks of mammalian species in Ontario and found that red foxes (*Vulpes vulpes*) showed higher activity in logged areas. This was expected as red foxes generally prefer semi-open areas and are seldom found in the interior of dense forests (Banfield 1987). In the same study, weasel (*Mustela erminea*) tracks showed no consistent pattern. Gashwiler (1970) and Towers and Milton (1990) found similar results for weasel tracks in Oregon and Nova Scotia, respectively.

The island of Newfoundland has a depauperate fauna with a limited small mammalian prey base for larger predators (Northcott 1980, Tucker *et al.* 1988). This creates an unstable and unbalanced system (Northcott 1980, Dodds 1983). Meadow voles, masked shrews, snowshoe hares and red squirrels are the only small mammalian prey species found in any abundance in the wild (Tucker *et al.* 1988). Other small mammalian species, such as deer mice and striped chipmunks (*Tamias striatus*), are not well-established and are reported only from restricted parts of the island, most notably the west coast (Gould and Pruitt 1969, Northcott *et al.* 1973, Bateman 1983, Tucker *et al.* 1988).

The few studies of small mammals in relation to forest harvesting in insular Newfoundland (predominantly in western Newfoundland) show results comparable to those elsewhere. Tucker (1988) found no difference between shrew abundance in a mature forest and a one-year-old clear-cut in western Newfoundland. However, there were significantly higher numbers of shrews in a 13- and a 23-year-old clear-cut than in the mature forest. Meadow vole and deer mouse numbers were too low to directly determine harvesting effects. Thompson and Curran (1995) caught shrews in all age clear-cuts and in uncut stands, while meadow voles were caught only in uncut stands. Nichols (1995) captured no small mammals on a one-year-old clear-cut. Tucker (1988) noted a slight increase in snowshoe hare abundance after forest harvesting based on their winter trails.

Studies of the Newfoundland pine marten have produced similar findings to those elsewhere - marten seldom use clear-cut areas (Snyder 1984, Bissonette *et al.* 1988). In western Newfoundland, Snyder and Bissonette (1987) captured 90% of all live-trapped marten in residual stands, while only 10% were captured in clear-cuts less than 15 years old. No marten were caught in 16- to 23-year-old clear-cuts. I have uncovered no additional local studies of the reponses of other mammalian carnivores to forest harvesting.

This project is one component of a larger study on riparian buffer zones in the Western Newfoundland Model Forest. The Copper Lake Buffer Zone Study (Scruton *et al.* 1995) was established with a multispecific, interdisciplinary approach to resource management in a small watershed with virgin, old-growth forests. The major foci of the Copper Lake study are the short- and long-term responses of aquatic and terrestrial wildlife to forest harvesting in riparian zones of the boreal forest, the effects of forest harvesting on water quality and the effectiveness of 20 m riparian buffers. Timber harvest regimes in the Copper Lake watershed were manipulated so that pre- and post-harvest data could be collected from three situations: a clear-cut without a riparian buffer, a clear-cut with a 20 m riparian buffer and an unharvested area with an effective buffer of at least 100 m.

This thesis deals specifically with the pre-harvest usage and subsequent initial responses of small mammalian herbivores and carnivores to disturbance associated with forest harvesting in riparian and adjacent terrestrial habitats.

Methods

Site Description

The study area is located in the Copper Lake watershed (Figure 1), approximately 18 km southeast of Corner Brook, Newfoundland, and falls within the Western Newfoundland Model Forest. This area is typical of the Corner Brook Subregion of the Western Newfoundland Ecoregion (Damman 1983). This ecoregion has generally nutrient-rich, productive humic podzols with a very dark B horizon due to humus content (Meades and Moores 1994); slates and limestone underlie most of the area (Damman 1983). The climate is generally cool and wet with a mean annual temperature of 5.2°C (Table 1) and precipitation on more than 180 days a year (Damman 1983).

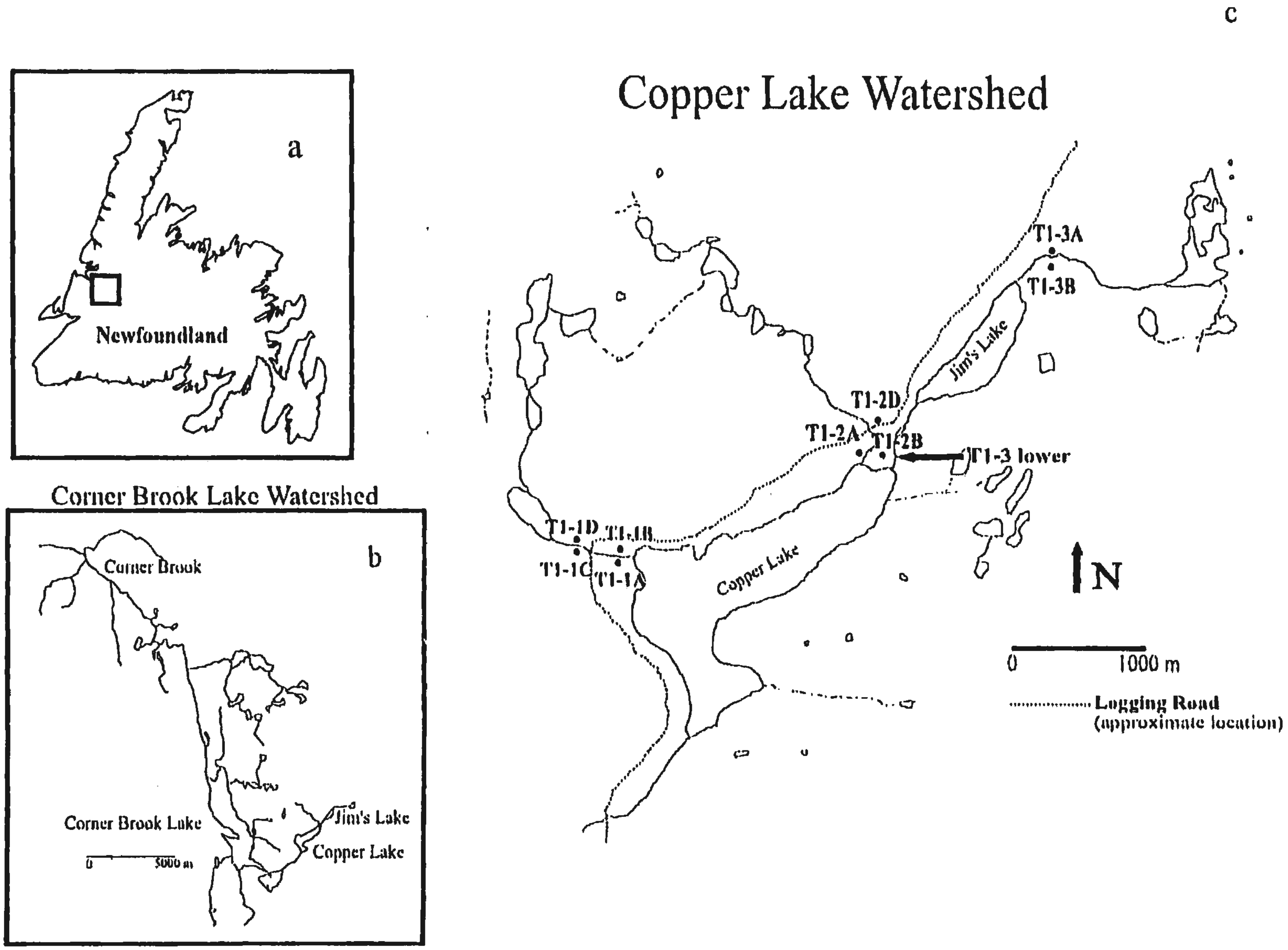
Three first order streams, mean widths of 6.3, 6.1 and 5.8 m, in the Copper Lake watershed were selected for this study and named, by convention, T1-1, T1-2 and T1-3, respectively (Scruton *et al.* 1995). Riparian (stream bank) habitats and adjacent interior habitats on both sides of each stream were used as study sites (Figure 1, Table 2). Stream T1-3 was divided into upper and lower sections. T1-3 upper was used for this study, while T1-3 lower actually cut through my study site at T1-2B (Figure 1). Figures 2 to 4 depict these three streams and the approximate location of the study sites.

The Copper Lake area consists predominantly of mature to overmature balsam fir (*Abies balsamea*) forest with black spruce (*Picea mariana*) fens (Meades and Moores 1994) occurring frequently. White birch (*Betula papyrifera*) and white spruce (*Picea

Figure 1: a) Map of insular Newfoundland with general location of the Copper Lake and Corner Brook Watersheds highlighted.

b) Enlargement of the highlighted area in (a).

c) The Copper Lake Watershed enlarged from (b) showing approximate locations of the Copper Lake study sites and the logging road constructed in 1994/95.



a
Newfoundland
Corner Brook Lake Watershed
b
Corner Brook
Corner Brook Lake
Jim's Lake
Copper Lake
0
5000 m
c
Copper Lake Watershed
T1-3A
T1-3B
Jim's Lake
T1-2D
T1-2A
T1-2B
T1-3 lower
T1-1D
T1-1B
T1-1C
T1-1A
Copper Lake
N
0
1000 m
Logging Road
(approximate location)

Table 1: Climate normals (means from 1933-1990) for Corner Brook (48°57' N 57°57' W) (Environment Canada 1991).

Month	**Mean Temp. (°C)**	**Total Rainfall (mm)**	**Total Snowfall (cm)**
January	-5.4	28.0	111.6
February	-6.8	17.1	74.9
March	-2.8	27.5	55
April	2.6	39.5	23.6
May	7.5	58.6	5.3
June	12.9	83.5	0.2
July	17.4	82.7	0.0
August	16.8	97.4	0.0
September	12.4	95.5	0.0
October	7.2	109.0	6.4
November	2.8	86.0	40.4
December	-2.5	46.0	97.0
Year	5.2	771.0	414.4

Table 2 : Study site locations and treatments.

Site	Location	Elevation (m)	Years Studied	Year Cut/ Treatment
T1-1A	48°49'18"N, 57°46'56"W	375-400	winter 1994 to winter 1997	1996 no buffer
T1-1B	48°49'22"N, 57°46'56"W	375-400	winter 1994 to winter 1997	1996 no buffer
T1-1C	48°49'19"N, 57°47'15"W	375-450	summers of 1995 to 1996	1994 no buffer
T1-1D	48°49'24"N, 57°47'13"W	375-450	summers of 1995 to 1996	1994 no buffer
T1-2A	48°49'38"N, 57°45'13"W	375-425	winter 1994 to winter 1997	1996 20 m buffer
T1-2B	48°49'38"N, 57°45'20"W	350-400	winter 1994 to winter 1997	1996 20 m buffer
T1-2D	48°49'48"N, 57°45'17"W	400-450	summers of 1995 to 1997	1994 20 m buffer
T1-3A	48°50'13"N, 57°45'00"W	375-400	winter 1994 to winter 1997	uncut
T1-3B	48°50'10"N, 57°44'55"W	375-425	winter 1994 to winter 1997	uncut
FEN	48°48'18"N, 57°38"48'W	330-350	summers of 1994 to 1996	no treatment
5YRCO	48°48'30"N, 57°49'55"W	350-425	summers of 1994 to 1996	harvested ~ 1990

Figure 2: Aerial photograph of the stream T1-1 and the approximate locations of the associated study sites, T1-1A, B, C and D.

Figure 3: Aerial photograph of the stream T1-2 and the approximate locations of the associated study sites, T1-2A, B and D.

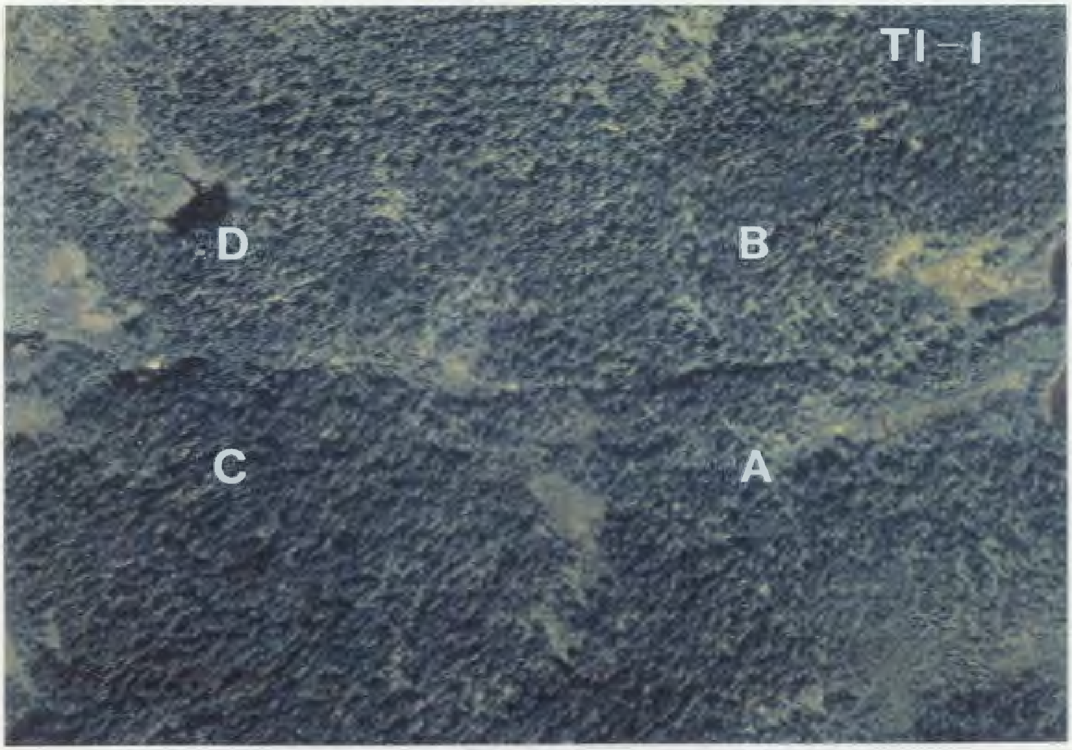


Figure 2

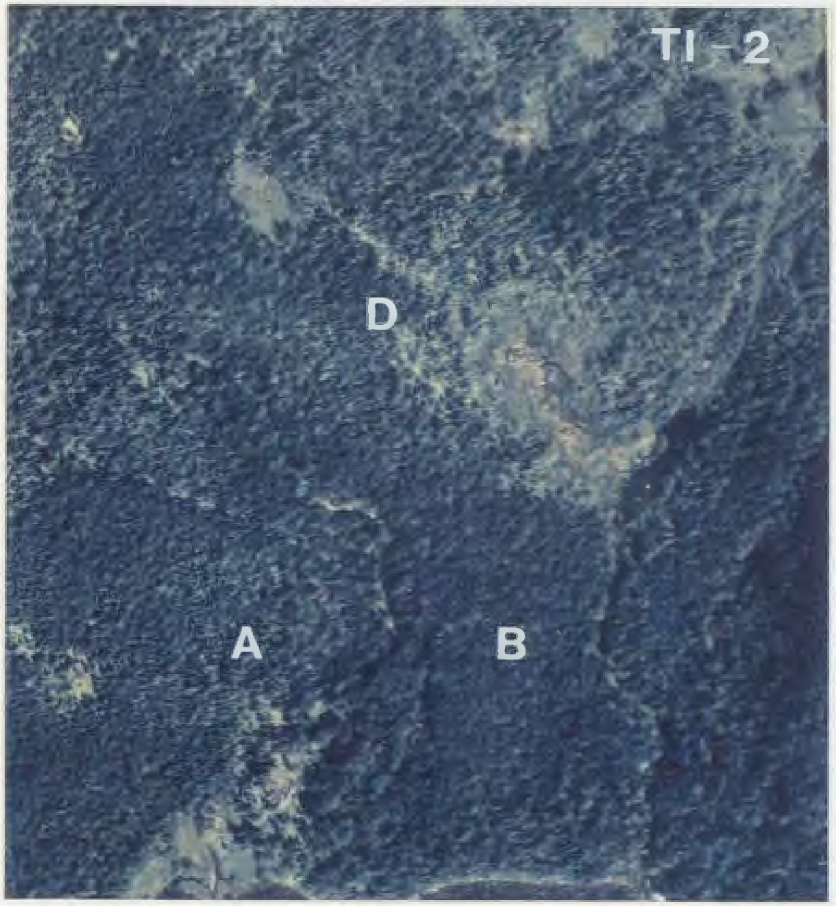


Figure 3

Figure 4: Aerial photograph of the stream T1-3 and the approximate locations of the associated study areas, T1-3A and B.

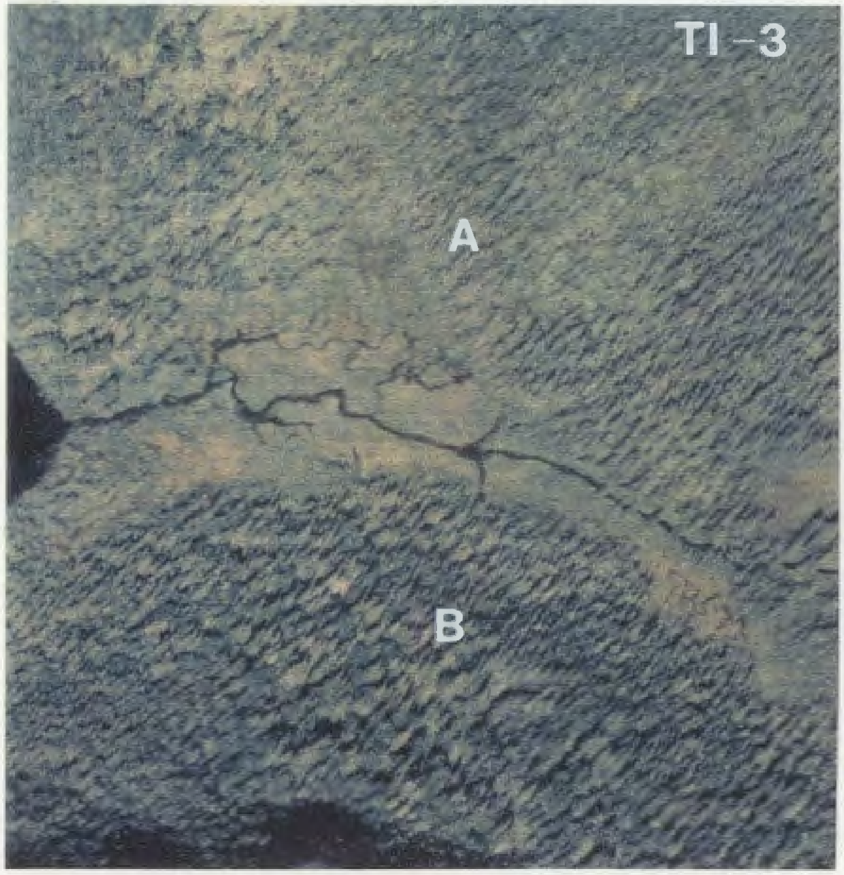


Figure 4

glauca) are also present in the area. Tree heights obtained from the Newfoundland Forest Service forest type maps range from 9.6 to 15.5 m. Site elevations range from 350 to 450 m (Table 2). A list of mammalian species (Northcott 1980) potentially found in the Copper Lake area is provided in Appendix A.

Study Design

The Copper Lake watershed was chosen by the various participants in the Buffer Zone Study because it contained riparian habitats located within old-growth (>80 years old) forests that were scheduled to be harvested during 1994 to 1996. Harvesting was scheduled for fall 1994 so that one year of pre-harvest data (winter through summer 1994) and then one year of post-harvest data (winter through summer 1995) could be collected. The cutting plan was designed so there would be two uncut areas (T1-3A and T1-3B), two clear-cuts with no riparian buffer (T1-1A and T1-1B), and two clear-cuts with a 20 m riparian buffer (T1-2A and T1-2B) (Table 2). For the small mammal portion of the study only, two additional sites outside of the Copper Lake study area were included: a five-year-old cutover (5YRCO) and a fen, both near Corner Brook Lake (Table 2). These sites were used to check small mammal numbers in other habitat types and to compare capture frequencies inside and outside of the Copper Lake watershed.

When the sites at Copper Lake were revisited in 1995 it was discovered that harvesting had not occurred on the correct areas. Instead, portions of forest further upstream from sites T1-1A, T1-1B, T1-2A and T1-2B had received the treatments. The study design was then altered in order to include some of these harvested areas. Two clear-cuts without a riparian buffer, sites T1-1C and T1-1D, and one clear-cut with a 20 m riparian buffer, T1-2D, were added only for the small mammal portion of the study. Figures 5 to 7 depict the study sites at Copper Lake in their pre- and post-harvest conditions, while Figure 8 shows the fen and 5YRCO site used for the small mammal study. It should be noted that prior to the late summer of 1994 there was no road access to any of the sites at Copper Lake. Subsequently, the logging road that was constructed passed through the interior of the plot on T1-1B and across the northern corner of the plot on T1-2A (Figure 1). The logging road also curved around the western/southwestern boundaries of T1-1A and paralleled the northern boundary of T1-3A (Figure 1).

This study was extended to summer 1996 and winter 1997 as sites T1-1A, T1-1B, T1-2A and T1-2B were harvested in 1996 according to the original cutting schedule. This allowed a comparison of baseline and post-harvest data as originally planned.

Small Mammals

Small mammals were trapped from June 27 to August 25, 1994, July 6 to August 20, 1995 and June 15 to August 20, 1996. Starting at the stream edge of each site, 10 parallel traplines, 15 m apart, were set up with 10 stations per line. Stations were placed at 15 m intervals resulting in an effective grid size of 150 x 150 m (2.25 ha). Traps were placed within 2 m of the station marker. Sites were trapped for a maximum of five days. Two smaller grids (75 m x 75 m) were established outside the study area in a fen and in

Figure 5: a) Photo depicting forest conditions at T1-1A, prior to harvesting.

b) Photo depicting the fen on T1-1B with the forest in the background (prior to harvesting). Note the pink and yellow flagging tape representing small mammal trapping stations.

Figure 5

Figure 5: c) Photo depicting pre-harvest forest conditions at T1-1B. Notice the deadfall typical of an older forest.

d) Photo depicting post-harvest conditions at both T1-1A (right of stream) and T1-1B (left of stream and above road). Note that the logging road passes through the interior of the plot on T1-1B and that a portion of the plot remains forested. Also note skidder trails along T1-1A. Copper Lake is at the far end of the picture.

Figure 5

Figure 5: e) Photo depicting post-harvest conditions at T1-1C.

f) Photo depicting post-harvest conditions at T1-1D. Notice in both pictures the amount of debris, slash and scrub spruce remaining.

Figure 5

Figure 6: a) Photo depicting pre-harvest forest conditions at T1-2A.

b) Photo depicting post-harvest conditions at T1-2A. The 20 m buffer on Copper Lake is in the background. Note the slash and the sun-curing of the remaining vegetation.

Figure 6

Figure 6: c) Photo depicting pre-harvest forest conditions at T1-2B.

d) Photo depicting post-harvest conditions at T1-2B. The 20 m stream buffer is shown on the right and the 20 m buffer on Copper Lake is at the far end of the picture.

Figure 6

Figure 6: e) Photo depicting post-harvest conditions at T1-2B. The 20 m buffer seen here is that from T1-3 lower, the stream that passes through the southeast side of the plot at T1-2B. Note the blowdown occurring at the edge of the buffer (in contrast to the buffer shown in Figure 6d which was sheltered from the wind).

f) Photo depicting post-harvest conditions at T1-2D, looking toward the 20 m buffer. Note the piles of slash left on the edge of the plot.

Figure 6

Figure 7: a) Photo depicting forest conditions at T1-3A.

b) Photo depicting forest conditions at T1-3B.

Figure 7

Figure 7: c) Photo depicting the grassy streamsides of T1-3A (left and far end of photo) and T1-3B (right of photo).

Figure 7

Figure 8: a) Photo depicting the fen near Corner Brook Lake used as an additional small mammal trapping site.

b) Photo depicting the five-year-old cutover on the side of Corner Brook Lake (background) used as an additional small mammal trapping site.

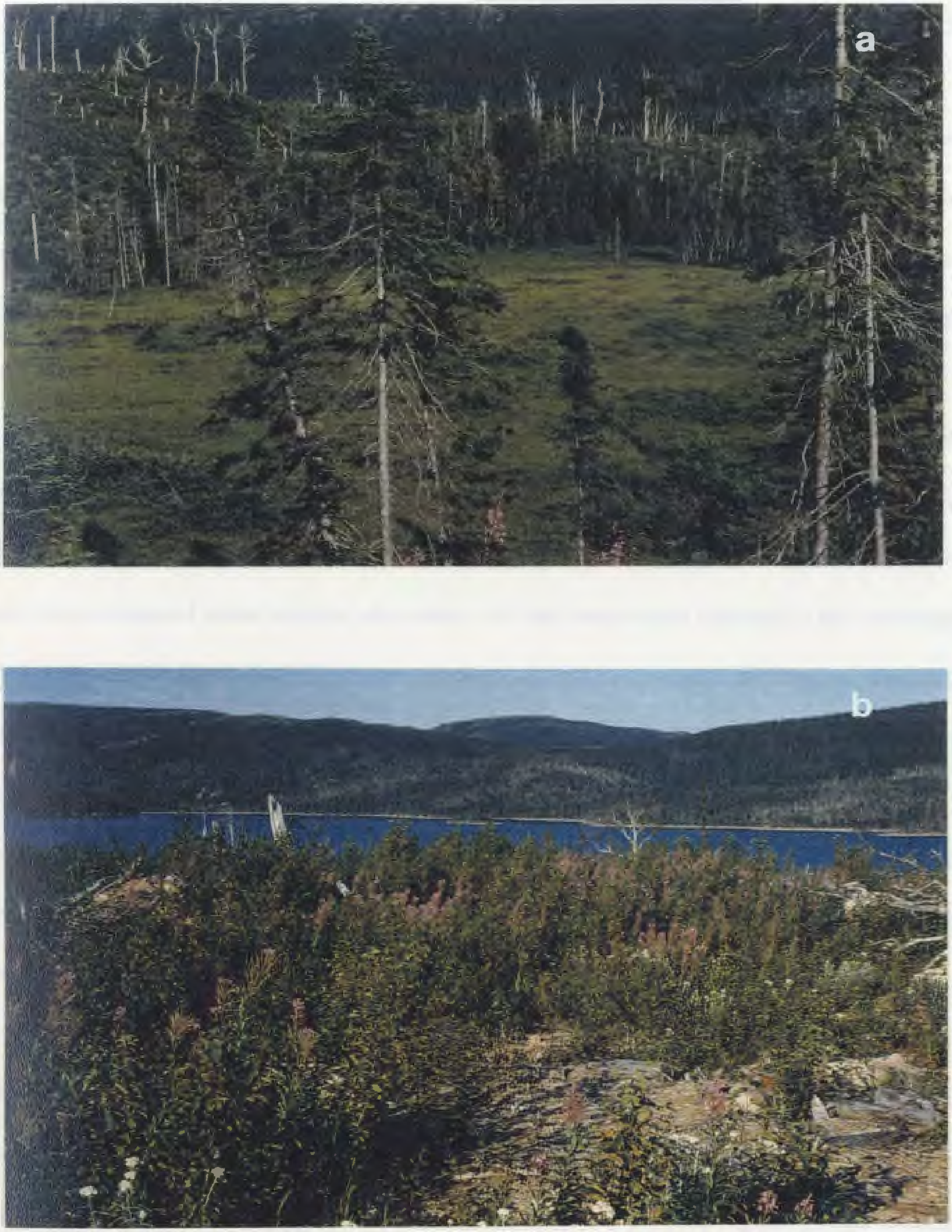


Figure 8

a five-year-old clear-cut.

In 1994, small Sherman live-traps were used initially on T1-1A and T1-1B. One trap was set per station and each trap was baited with a mixture of rolled oats, peanut butter, brown sugar, honey and bacon fat. Sheep's wool was used as bedding material. The traps were set for five days and checked twice daily, at 6:00 a.m. and at 8:00 p.m.. Any traps missing bait or with wet bedding were reset with new material.

No small mammals were caught in the live-traps over 1000 trap-nights (TN), so the first two sites were immediately re-trapped using Victor snap-traps and the same bait mixture. Fifty snap traps were used with two traps at every second station. The Victor snap-traps captured some shrews and voles, so they were used instead of the live-traps on the remaining sites. Initially, large numbers of snap-traps were difficult to obtain so trapping effort was increased as more traps became available. This also served as a check on whether trapping effort was affecting the results.

The trapping effort was doubled on the next two sites, T1-2A and T1-2B, with 100 snap traps used per site (1 trap per station). At T1-3A and T1-3B there were 200 snap traps used per site (2 traps per station).

In 1995 and 1996, all grids were set with two traps per station, including T1-1C, T1-1D and T1-2D. The total number of TN was adjusted for traps damaged by other animals, traps set off by heavy rainfall and missing traps. Catches were standardized to numbers of small mammals per 100 TN because of unequal trapping effort (cf. Clough 1987, Parker 1989).

Winter Tracks

Series of line transects parallel to each stream were set up on T1-1A, T1-1B, T1-2A, T1-2B, T1-3A and T1-3B in late January, 1994 and these were surveyed for mammal tracks from February 4 to April 8, 1994, January 2 to April 12, 1995, January 27 to March 30, 1996 and January 9 to March 29, 1997. The transects ranged from 150 m to 250 m in length depending upon the topography of each site. The first transect was located at the stream's edge, while the second and third transects were set 20 m and 50 m back respectively. The remaining transects were spaced at 50 m intervals up to a distance of 150 m from the stream where possible (i.e. all sites except T1-2B). Transects were walked within 24 hours of fresh snowfall. Any new track sets which occurred within 5 m on either side of the transect lines were counted and identified to species. The direction of animal travel and the location of tracks along the transect were also recorded.

Habitat

Overstory vegetation was described during summer 1994 using the point quarter method as outlined in Smith (1990). This plotless method involves selecting and marking a series of random points along a line transect through a stand of trees. The working area around each point is divided into four 90° quadrants and then the distance to the nearest tree in each quadrant is measured. The species of tree and diameter at breast height are also recorded. Six to seven point quarter transect lines 150 m long and spaced 25 m apart were set up on each site. Seventeen random points from which to measure trees were selected from a random number table (numbers picked between 0 and 150) and the same series of random points was used for each transect. Tree densities and frequencies were calculated using the formulae: relative density = number of individuals of species A/total number of individuals of all species x 100; frequency = number of points at which species A occurs/total number of points sampled x 100 (Smith 1990). Approximate tree densities and frequencies were obtained for T1-1C, T1-1D and T1-2D in 1995 from tree stumps and residual trees.

Understory vegetation cover was estimated using Braun-Blanquet cover classes (0-1%, 2-5%, 6-25%, 26-50%, 51-75%, 76-100%; modified from Smith 1990). The midpoints of these cover classes (0.5, 3.5, 15.5, 38.0, 63.0 and 88.0%, respectively) were then used for statistical purposes and for calculations of mean cover. For each site, 25 randomly selected stations from the small mammal grids were used as plots. Nested plots with sizes relative to the size of vegetation type were used: a 1.4 x 1.4 m plot for woody plants, a 1 x 1 m plot for herbaceous vegetation and a 0.1 x 0.1 m plot for bryophytes. Plants were identified to genus and species where possible (using Ryan 1978, Scott 1987, Niering and Olmstead 1993 and Meades and Moores 1994) and were also grouped into the following categories: shrubs, forbs, grasses, sedges, feather mosses, *Sphagnum* spp. and lichens. This was done to compare the physiognomic characteristics of the sites.

Statistical Analysis

Statistical analyses were carried out using the SPSS statistical package for Windows. To determine similarities and differences between sites a hierarchical cluster analysis (average linkage method) was performed using tree stand and understory vegetation characteristics as variables. Shannon-Wiener diversity indices and jacknife estimates of plant species richness (Kent and Coaker 1992) were calculated for each site.

Comparisons of small mammal abundances by sites and harvesting treatments were limited to *Sorex cinereus*, owing to the infrequent capture of both *Microtus pennsylvanicus* and *Peromyscus maniculatus* throughout this study. Shrew data from 1994 were examined separately because there were only uncut sites trapped in this year. A single sample t-test ($\alpha = 0.05$) was performed on these data to see if observed shrew abundance was affected by the progressive increase in trapping effort on T1-2A and B and T1-3A and B.

A one-way ANOVA, followed by a Tukey's multiple comparison test ($\alpha = 0.05$) was used to test for differences in shrew abundance over the three summers. In addition, shrew abundances from 1995 and 1996 were grouped by harvesting treatment type (i.e., uncut, clear-cut, clear-cut with 20 m buffer) and a two-way ANOVA ($\alpha = 0.05$) was used to test for the effects of both year and harvesting treatment. For 20 m buffer sites, the catch per 100 TN was also calculated separately for the buffers and adjacent clear-cuts. A paired sample t-test ($\alpha = 0.05$) was then used to compare shrew abundances between the 20 m buffers and adjacent clear-cuts.

Pearson correlation coefficients were calculated to determine the amount of correlation between 1995 shrew and meadow vole abundances and the following vegetation variables: percent cover of shrubs, forbs, grasses, sedges, feather mosses, *Sphagnum* spp., and lichens, tree density, tree frequency, plant species richness and Shannon-Wiener diversity. Deer mice captures were too infrequent to attempt any correlation analysis. A stepwise multiple regression of shrew and meadow vole abundances on vegetation characteristics was run using the previous variables.

Winter track data were very variable with many zero counts for most mammal species and therefore, nonparametric statistical methods were used on these data sets (cf. Thompson *et al.* 1989). Track counts were standardized to numbers per 100 m. Kruskal-Wallis and Tukey tests (Zar 1996) were then used to compare total and individual species track abundances by year, by distance from the stream and by site. The 1997 track data (track sets/100 m) were categorized on the basis of whether individual transects were forested or open (clear-cut or no tree cover). Kruskal-Wallis and Tukey tests were then used to compare mammal usage of forested and open areas. As an additional measure of mammal activity, track frequencies were calculated for each species by site as the number of times a species was present divided by the number of days a site was surveyed (cf. Cameron 1997). Track frequencies give an indication of how often an individual or group of individuals was present in a site, while track abundances (track sets/100 m) indicate how intense the activity was in that particular site.

Results

Vegetation

Mean vegetation cover and frequency data for each site in 1995 are presented in Tables 3 and 4. The frequencies of coniferous and hardwood trees display some consistency within the sites as indicated by the small standard deviations (Table 3). The post-harvesting analysis of T1-1C, T1-1D and T1-2D based on tree stumps and residual vegetation (e.g., birch) yielded similar results (Table 3). In addition, conifer and hardwood frequencies are comparable among sites, with the exception of the hardwood frequency for T1-1A which was only 0.84 (Table 3). See Appendix B for frequencies by species.

The understory vegetation exhibited high variation within sites as indicated by the large standard deviations in most cases (Table 4). Lists of plant species and Braun-Blanquet cover classes for each site are provided in Appendix C. A hierarchical cluster analysis (Figure 9), based on both overstory and understory vegetation characteristics in 1995, shows a grouping of the three harvested sites, T1-1C, T1-1D and T1-2D, with T1-2D clustering closer to T1-1C. Sites T1-1B and T1-3B cluster together as do T1-1A and T1-3A. The cluster of T1-1A and T1-3A is separated from the harvested areas, T1-1B, T1-3B and T1-2A. This is reflected in the greater percentages of *Sphagnum* spp. and sedge cover found in these two sites (Table 4). T1-2B, having the greatest amount of feather moss cover and little other understory (Table 4), was the least similar to the other

Table 3: Mean conifer and hardwood frequencies (percentage ± standard deviation) obtained from point quarter transects in 1994. Values for sites T1-1C, T1-1D and T1-2D are estimates based on tree stumps and trees remaining in the clear-cuts in 1995.

Site	**No. of Transects**	**Conifer Frequency**	**Hardwood Frequency**
T1-1A	7	97.5 ± 6.7	0.8 ± 2.2
T1-1B	6	84.3 ± 11.6	10.8 ± 12.6
T1-1C	6	100.0 ± 0.0	19.6 ± 4.8
T1-1D	6	92.2 ± 12.7	7.8 ± 7.1
T1-2A	7	100.0 ± 0.0	6.7 ± 4.1
T1-2B	7	97.5 ± 4.6	16.0 ± 7.4
T1-2D	6	94.1 ± 14.4	8.8 ± 7.2
T1-3A	7	90.8 ± 11.7	11.8 ± 11.8
T1-3B	6	83.3 ± 5.8	14.7 ± 12.2

Table 4: Mean understory vegetation coverage (percentage ± standard deviation) using the mid-points from Braun-Blanquet cover classes (p. 32).

Site	**No. of Plots**	**Shrub Cover**	**Forb Cover**	**Grass Cover**	**Sedge Cover**	***Sphagnum* Cover**	**Feather Moss Cover**	**Lichen Cover**
T1-1A	24	32,4 ± 29,8	21,8 ± 24,9	3,1 ± 8,6	21,5 ± 33,7	49,9 ± 42,4	18,7 ± 31,5	0
T1-1B	22	29,1 ± 37,5	20.2 ± 25,4	4,5*10^{-2} ± 0,14	5,5 ± 13,9	28,0 ± 41,9	43,6 ± 44,6	6,8*10^{-2} ± 0,18
T1-1C	25	7,4 ± 10,4	9,1 ± 14,3	0,64 ± 3,1	3,2 ± 12,9	21,4 ± 36,4	27,3 ± 37,8	0
T1-1D	24	22,3 ± 27,3	16,2 ± 25,3	7,0 ± 21,7	4,4 ± 18,1	19,0 ± 36,3	37,0 ± 43,4	0
T1-2A	25	42,2 ± 36,1	10,9 ± 19,1	0	0,16 ± 0,70	9,2 ± 25,1	43,6 ± 38,8	3,5 ± 17,6
T1-2B	24	7,3 ± 15,6	13,1 ± 19,1	4,0*10^{-2} ± 0,14	0,14 ± 0,70	2,5 ± 12,6	60,7 ± 45,6	0
T1-2D	24	10,1 ± 19,8	8,3 ± 13,9	2,0*10^{-2} ± 0,10	4,3 ± 17,7	3,7 ± 17,6	32,4 ± 39,6	0
T1-3A	25	27,0 ± 28,2	15,3 ± 14,1	16,2 ± 31,4	22,8 ± 27,6	55,8 ± 39,5	1,6 ± 7,8	0
T1-3B	25	27,9 ± 36,5	11,4 ± 19,4	3,8 ± 18,0	1,6 ± 4,4	15,9 ± 31,5	31,6 ± 36,9	0

Figure 9: Cluster diagram showing vegetation similarity between sites using the average linkage method of a hierarchical cluster analysis.

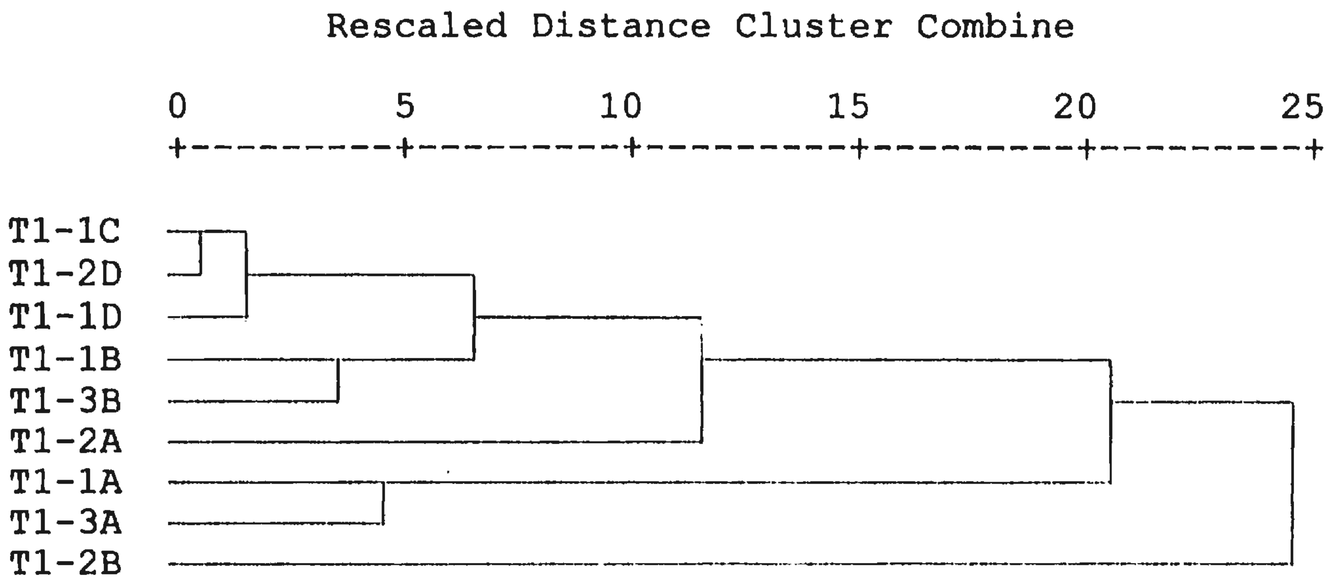

Rescaled Distance Cluster Combine
0
5
10
15
20
25
T1-1C
T1-2D
T1-1D
T1-1B
T1-3B
T1-2A
T1-1A
T1-3A
T1-2B

Figure 9

sites.

Table 5 gives the Shannon-Wiener diversity indices and species richness values for the understory vegetation at each site. All sites had very similar diversities (range of 1.0 to 1.2) with the exception of T1-3A which had a much lower diversity (0.6). T1-3A and T1-3B had the highest species richness (50.6 and 51.5, respectively). T1-2D had the lowest species richness at 31.8.

Small Mammals

A total of 966 *Sorex cinereus*, 41 *Microtus pennsylvanicus*, and 10 *Peromyscus maniculatus* were captured over the summers of 1994 to 1996 inclusive (Table 6) with a total of 20,581 trap-nights (TN). All small mammals were caught in Victor snap-traps.

Sorex cinereus:

The six sites within the Copper Lake watershed trapped in 1994 yielded a total of seventeen *Sorex cinereus*. Shrew captures ranged from 0.10 to 2.40 per 100 TN (Table 6). A one sample t-test to compare shrew abundances among these sites revealed no significant differences ($t_{0.05, 5} = 2.05$, $p = 0.095$) and, therefore, no change in shrew abundance with an increase in the number of traps used.

Sorex cinereus abundances in 1995 and 1996 were significantly higher than in 1994 in all Copper Lake sites (including T1-1C, T1-1D and T1-2D; $F = 14.73$, $p < 0.001$). Captures per 100 TN ranged from 3.59 to 7.80 shrews in 1995 and from 2.24

Table 5: Understory plant species diversity and total species richness for the Copper Lake sites.

Site	Shannon-Weiner Diversity Index (Log_{10})	Jacknife Estimate of Species Richness
T1-1A	1.0	48.6
T1-1B	1.2	46.8
T1-2A	1.1	40.8
T1-2B	1.0	48.5
T1-3A	0.6	50.6
T1-3B	1.1	51.5
T1-1C	1.1	39.7
T1-1D	1.2	44.8
T1-2D	1.0	31.8

Table 6: Number of small mammal captures per 100 trapnights at sites both inside and outside of the Copper Lake watershed (actual catch in parentheses). 5YRCO = five-year-old cutover.

Site	Year	*Sorex cinereus*	*Microtus pennsylvanicus*	*Peromyscus maniculatus*
T1-1A	1994	2.40 (6)	0.00 (0)	0.00 (0)
	1995	6.47(64)	0.10 (1)	0.00 (0)
	1996	4.02(40)	0.10 (1)	0.00 (0)
T1-1B	1994	0.42 (1)	1.26 (3)	0.00 (0)
	1995	7.58(54)	0.14 (1)	0.00 (0)
	1996	8.54(76)	1.35(12)	0.11 (1)
T1-1C	1995	6.37(63)	0.00 (0)	0.00 (0)
	1996	6.38(63)	0.30 (3)	0.10 (1)
T1-1D	1995	7.80(62)	0.13 (1)	0.00 (0)
	1996	5.94(58)	0.41 (4)	0.00 (0)
T1-2A	1994	0.21 (1)	0.00 (0)	0.00 (0)
	1995	6.21(44)	0.14 (1)	0.00 (0)
	1996*	4.44(43)	0.00 (0)	0.10 (1)
T1-2B	1994	1.20 (6)	0.00 (0)	0.00 (0)
	1995	3.59(28)	0.00 (0)	0.00 (0)
	1996*	2.33(22)	0.11 (1)	0.21 (2)
T1-2D	1995*	4.44(35)	0.00 (0)	0.00 (0)
	1996	10.49(102)	0.61 (6)	0.10 (1)
T1-3A	1994	0.10 (1)	0.30 (3)	0.00 (0)
	1995	5.21(41)	0.25 (2)	0.00 (0)
	1996	4.20(41)	0.20 (2)	0.00 (0)
T1-3B	1994	0.20 (2)	0.00 (0)	0.00 (0)
	1995	6.53(50)	0.00 (0)	0.00 (0)
	1996	2.24(21)	0.00 (0)	0.00 (0)
Fen	1994	2.02 (3)	0.00 (0)	0.00 (0)
	1995	5.61(11)	0.00 (0)	0.00 (0)
	1996	2.81 (7)	0.00 (0)	0.00 (0)
5YRCO	1994	3.00 (3)	0.00 (0)	0.00 (0)
	1995	8.51(16)	0.00 (0)	0.53 (1)
	1996	0.82 (2)	0.00 (0)	1.23 (3)

*indicates years in which a 20 m riparian buffer was created.

to 10.49 shrews in 1996 (Table 6). When these data were analysed according to forest harvesting treatment received (i.e., uncut, clear-cut or clear-cut with 20 m buffer), I found no significant difference in shrew abundance with treatment type ($F = 1.297$, $p = 0.309$).

For sites that had a forested buffer, T1-2A and T1-2B in 1996 and T1-2D in 1995 and 1996, the shrew abundances were significantly higher within the buffer than on the adjacent clear-cuts ($t_{0.05,3} = 5.20$, $p = 0.014$; Table 7). Catches in the buffers on T1-2A and T1-2B, in particular, were two to three times higher than on the clear-cuts.

In the fen and 5YRCO site more shrews were captured in 1995 than in 1994 or in 1996. Captures per 100 TN ranged from 2.02 to 5.61 in the fen and 0.82 to 8.51 in the cutover (Table 6).

Microtus pennsylvanicus:

Microtus pennsylvanicus captures were low in all three summers and thus precluded most statistical analyses. However, there was an increase in the number of voles caught in 1996 compared with 1994 and 1995 (six voles in both 1994 and 1995 and 29 voles in 1996; Table 6). In 1994, voles were captured in two of the six Copper Lake sites (T1-1B and T1-3A) with capture rates of 0.30 to 1.26 voles per 100 TN (Table 6). In 1995, the same total number of voles was caught even with the addition of three extra sites (voles caught in T1-1A, T1-1B, T1-2A, T1-3A and T1-1D). Captures per 100 TN ranged from 0.10 to 0.25 voles (Table 6). In 1996, after forest harvesting had occurred,

Table 7: Abundance (catch per 100 trapnights, TN) of *Sorex cinereus* within and outside of the 20 m stream and lake buffers on T1-2A, T1-2B and T1-2D. The actual number caught is in parentheses. Note: abundances within the buffers were significantly higher than in the adjacent clearcuts (paired sample t-test, $t_{0.05,3} = 5.20$, $p = 0.014$).

Site/Year	**Number/100 TN Within Buffer**	**Number/100 TN Outside Buffer**
T1-2A 1996	6.99 (13)	3.84 (30)
T1-2B 1996	3.06 (17)	1.02 (4)
T1-2D 1995	4.28 (17)	3.05 (18)
1996	12.21 (48)	9.15 (54)

voles were caught in T1-1A, T1-1B and in T1-2B. Voles were also captured in the cutovers at T1-1C and T1-1D, T1-2D and in T1-3A. Captures per 100 TN ranged from 0.10 to 1.35 voles (Table 6).

The number of *M. pennsylvanicus* captures increased on T1-1C, T1-1D and T1-2D from the summers of 1995 to 1996 (Figure 10). In T1-1C the catch per 100 TN went from 0 to 0.30 voles, 0.13 to 0.41 voles in T1-1D and 0 to 0.61 voles in T1-2D (Table 6). With the exception of T1-2A, the new clear-cuts/buffer sites showed an increase or no change in vole abundance (Figure 11, Table 6). There were too few vole captures on any of the 20 m buffer sites to determine if there were differences in abundance between buffers and clear-cuts. In T1-3A and T1-3B there was little or no change in vole captures from 1995 to 1996: T1-3A - from 0.25 to 0.20 voles per 100 TN and T1-3B - no voles in either year (Table 6). No meadow voles were caught on either the fen or 5YRCO during the three summers of trapping.

Peromyscus maniculatus:

Peromyscus maniculatus was not captured in the Copper Lake area until 1996. The number of deer mice captures was very low in this year with six captures in total and catches per 100 TN ranging from 0.1 to 0.21 (Table 6). Deer mice were captured only on clear-cuts or 20 m buffer sites - two of the sites cut in 1994 (T1-1C and T1-2D) and three of the sites cut in 1996 (T1-1B, T1-2A and T1-2B). As with the voles, too few deer mice were captured on the buffer sites to determine if there was any effect of a 20 m

Figure 10: *Microtus pennsylvanicus* captures per 100 trapnights (TN) in 1995 and 1996 on T1-1C, T1-1D and T1-2D.

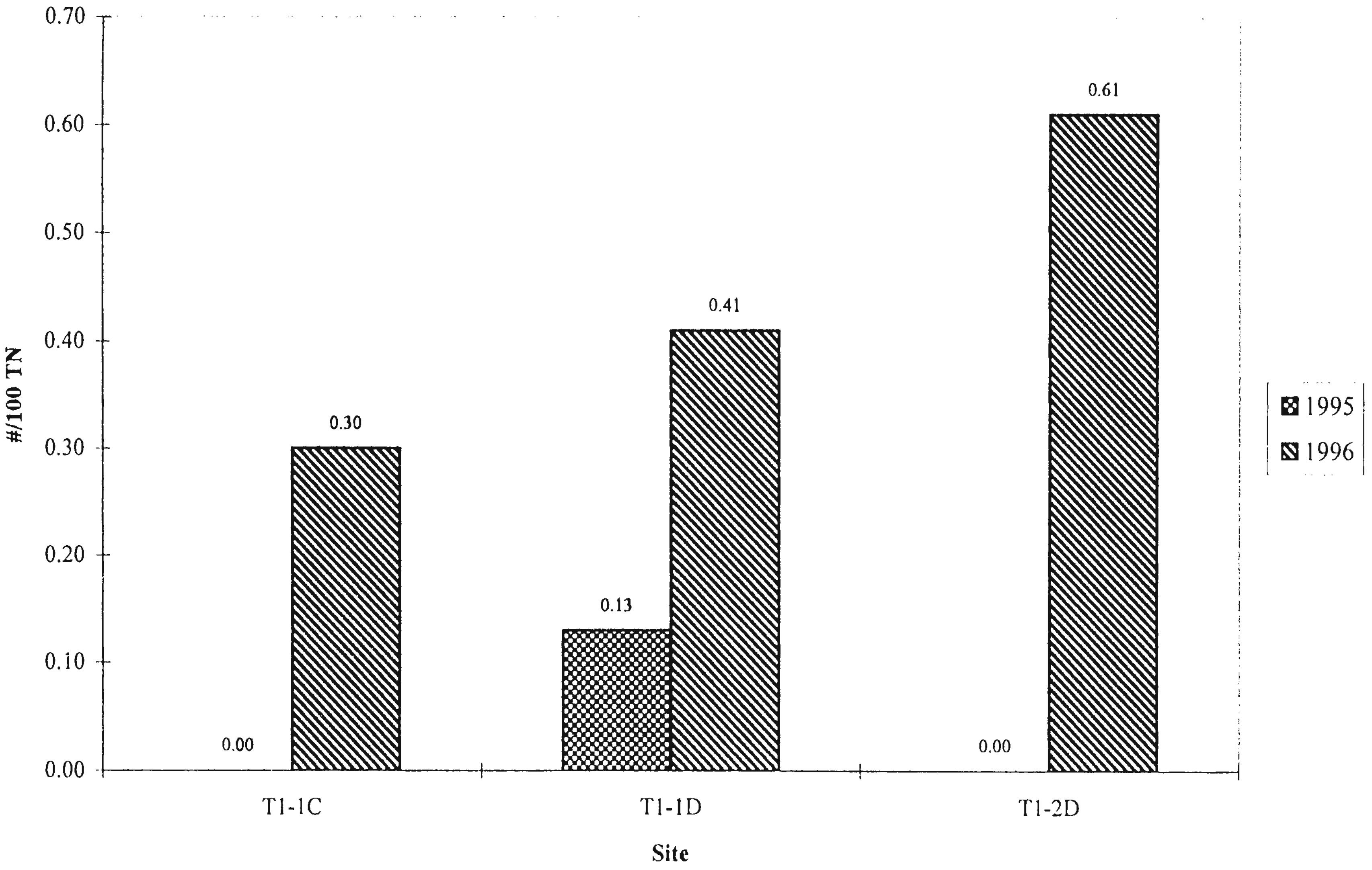

0.70
0.60
0.50
0.40
0.30
0.20
0.10
0.00
#/100 TN
0.00
0.30
0.13
0.41
0.00
0.61
T1-1C
T1-1D
T1-2D
Site
▩ 1995
▧ 1996

Figure 10

Figure 11: *Microtus pennsylvanicus* captures per 100 trapnights (TN) in 1994, 1995 and 1996 on T1-1A, T1-1B, T1-2A and T1-2B.

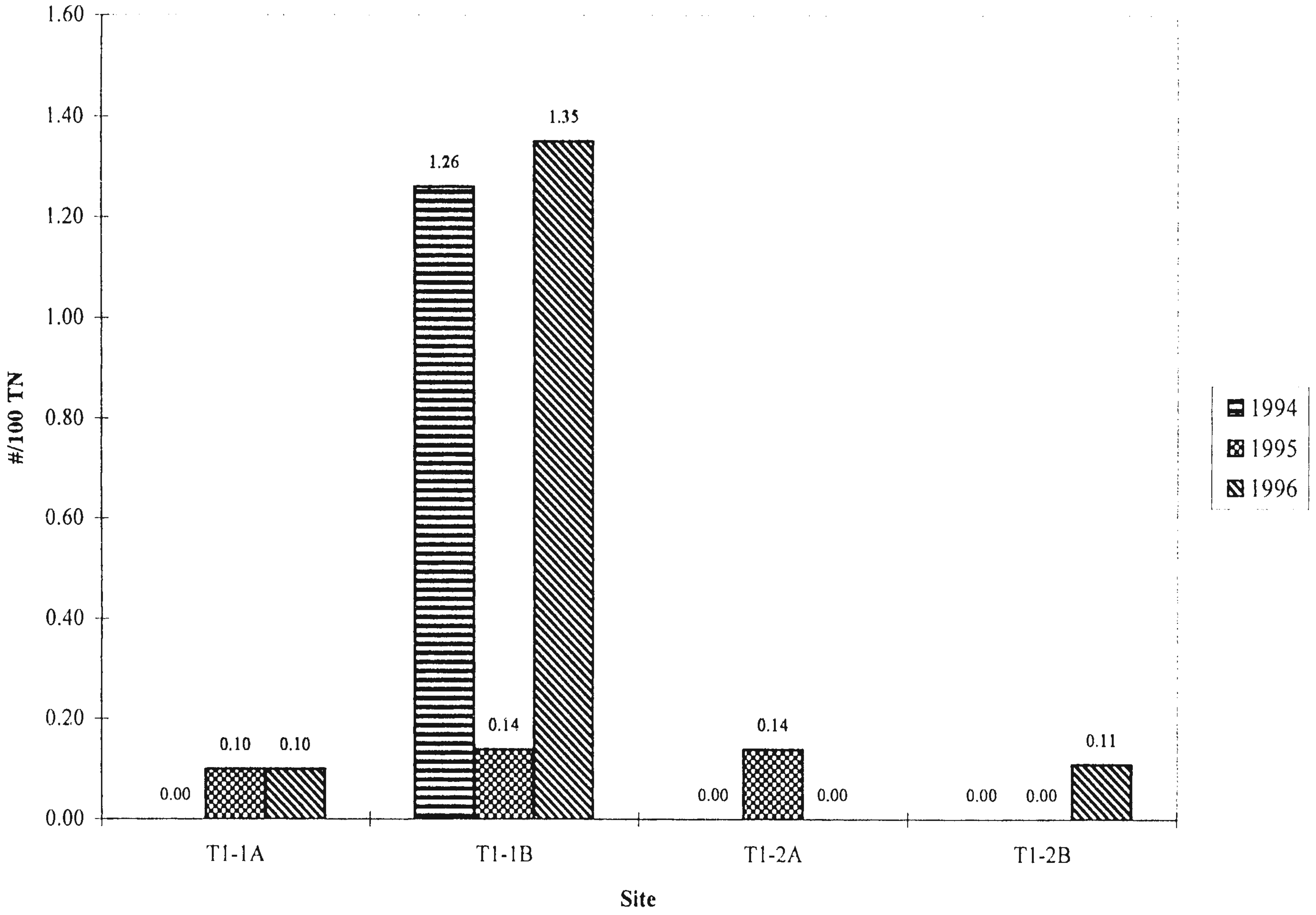

1.60
1.40
1.20
1.00
0.80
0.60
0.40
0.20
0.00
#/100 TN
0.00
0.10
0.10
1.26
0.14
1.35
0.00
0.14
0.00
0.00
0.00
0.11
T1-1A
T1-1B
T1-2A
T1-2B
Site
▤ 1994
▩ 1995
▨ 1996

Figure 11

buffer. Outside the study area, a total of four deer mice were caught on 5YRCO - one in 1995 and three in 1996 (Table 6). None were captured on the fen.

Habitat Associations

Pearson correlation coefficients for vegetation variables and shrew and vole captures in 1995 are presented in Table 8. Only one significant correlation was found - *Microtus pennsylvanicus* had a significant, positive correlation with grass cover ($p < 0.05$). Other correlations of meadow voles with vegetation approached significance; for example, there were high positive correlations with forb, sedge, *Sphagnum* and shrub cover. A stepwise, multiple regression of voles on vegetation variables showed similar results - a significant ($p < 0.05$) relationship between vole captures and grass cover. *Sorex cinereus* was not significantly correlated with any vegetation feature, although the correlation coefficients for shrub cover and Shannon-Wiener diversity approached significance. The stepwise, multiple regression also showed no significant relationships between shrews and vegetation features.

Winter Track Data

Although not consistently measured, limited snow depth data from T1-1A and B indicate that snowfall in the Copper Lake area is typical of the Corner Brook area (Table 1). Snowfalls were much greater in 1994 and 1995 than 1996 or 1997 (ranges of 61 - 150 cm, 84 - 145 cm, 12 - 58 cm and 20 - 80 cm, respectively).

Table 8: Pearson correlation coefficients for *Sorex cinereus* and *Microtus pennsylvanicus* in 1995 with site vegetation characteristics.

Variable	***Sorex cinereus***	***Microtus pennsylvanicus***
Trees/m^2	-0.451	-0.215
Tree Frequency	-0.313	0.261
Shrubs	0.508	0.628
Forbs	0.488	0.616
Grasses	0.047	0.694*
Sedges	0.093	0.643
Sphagnum moss	0.283	0.661
Feather mosses	-0.246	-0.473
Lichens	0.060	0.237
Species Richness	0.178	0.340
Shannon-Weiner Diversity	0.513	-0.405

*significant at $\alpha = 0.05$

Grouped Species:

Tracks of *Martes americana*, *Mustela erminea*, *Mustela vison*, *Vulpes vulpes*, *Tamiasciurus hudsonicus*, *Lepus americanus*, *Sorex cinereus*, *Microtus pennsylvanicus* and *Peromyscus maniculatus* were observed in the winters of 1994 to 1997 (Table 9). Of these species, only *M. americana*, *M. erminea*, *V. vulpes*, *L. americanus* and *T. hudsonicus* tracks were used for statistical analysis because they occurred most frequently over the four years. For a summary of track data on large mammals see Appendix D.

A Kruskal-Wallis test on data pooled from these five species at all sites and years showed a significant difference in track counts with distance of transect from the stream ($p < 0.001$; Figure 12). Tracks occurred most frequently on the transect farthest from the stream (150 m) and least frequently at the stream edge. There was a gradual increase in the number of tracks with distance from the stream. Observing grouped track data from 1997 (i.e., post-harvesting) by site shows more variable results. On both T1-1A and T1-1B there were no tracks at the streamside and numbers increased with distance from the stream (Figure 13). T1-2A shows the most tracks within the 20 m stream buffer, while T1-2B shows a similar number of tracks throughout (Figure 13). Both T1-3A and T1-3B have fewer tracks at the stream edge, more in the interior, but fewer on the 150 m transect (Figure 13).

Pooled track data were also found to have significant differences by year and site ($p < 0.001$). Both 1994 and 1996 data showed more mammal tracks than 1995 and 1997 (Table 10). T1-1A had the fewest tracks, while T1-3B had the most.

Table 9: The number of mammal winter tracks per 100 m of transect averaged for each site in the Copper Lake watershed. 1994 to 1996 data are prior to forest harvesting, while 1997 data are post-harvest. Note: years in which there were no tracks for a particular species are omitted.

Species	Year	TI-1A	TI-1B	TI-2A	TI-2B	TI-3A	TI-3B
Sorex cinereus	1996	0.03	0.14	0.01	0.08	0.29	0.00
Microtus pennsylvanicus	1996	0.00	0.00	0.00	0.00	0.01	0.00
Peromyscus maniculatus	1996	0.01	0.19	0.01	0.02	0.00	0.00
Tamiasciurus hudsonicus	1994	0.04	0.00	0.00	0.00	0.02	0.05
	1995	0.02	0.01	0.00	0.00	0.00	0.20
	1996	0.34	0.36	0.21	0.16	0.19	0.49
	1997	0.00	0.08	0.05	0.04	0.07	0.20
Lepus americanus	1994	0.00	0.00	0.43	0.34	0.43	1.48
	1995	0.01	0.20	0.21	0.09	0.12	0.26
	1996	0.07	0.01	0.08	0.03	0.18	0.15
	1997	0.21	0.34	0.69	0.17	0.20	0.07
Martes americana	1994	0.30	0.56	0.43	0.38	0.38	0.13
	1995	0.15	0.11	0.18	0.14	0.20	0.24
	1996	0.10	0.11	0.45	0.53	0.36	0.28
	1997	0.00	0.06	0.15	0.15	0.11	0.37
Mustela erminea	1994	0.17	0.38	0.08	0.03	0.15	0.23
	1995	0.02	0.01	0.05	0.00	0.00	0.00
	1996	0.03	0.09	0.04	0.03	0.19	0.24
	1997	0.11	0.04	0.06	0.04	0.06	0.06
Mustela vison	1995	0.01	0.01	0.01	0.14	0.02	0.08
	1997	0.00	0.00	0.03	0.05	0.04	0.00
Vulpes vulpes	1994	0.07	0.14	0.00	0.00	0.15	0.28
	1995	0.07	0.03	0.13	0.09	0.22	0.44
	1996	0.06	0.01	0.11	0.11	0.09	0.04
	1997	0.00	0.02	0.13	0.06	0.07	0.00

Figure 12: The number of mammal winter track sets per 100 m (group average for *Martes americana, Mustela erminea, Vulpes vulpes, Lepus americanus* and *Tamiasciurus hudsonicus*) with increasing distance from the stream edge.

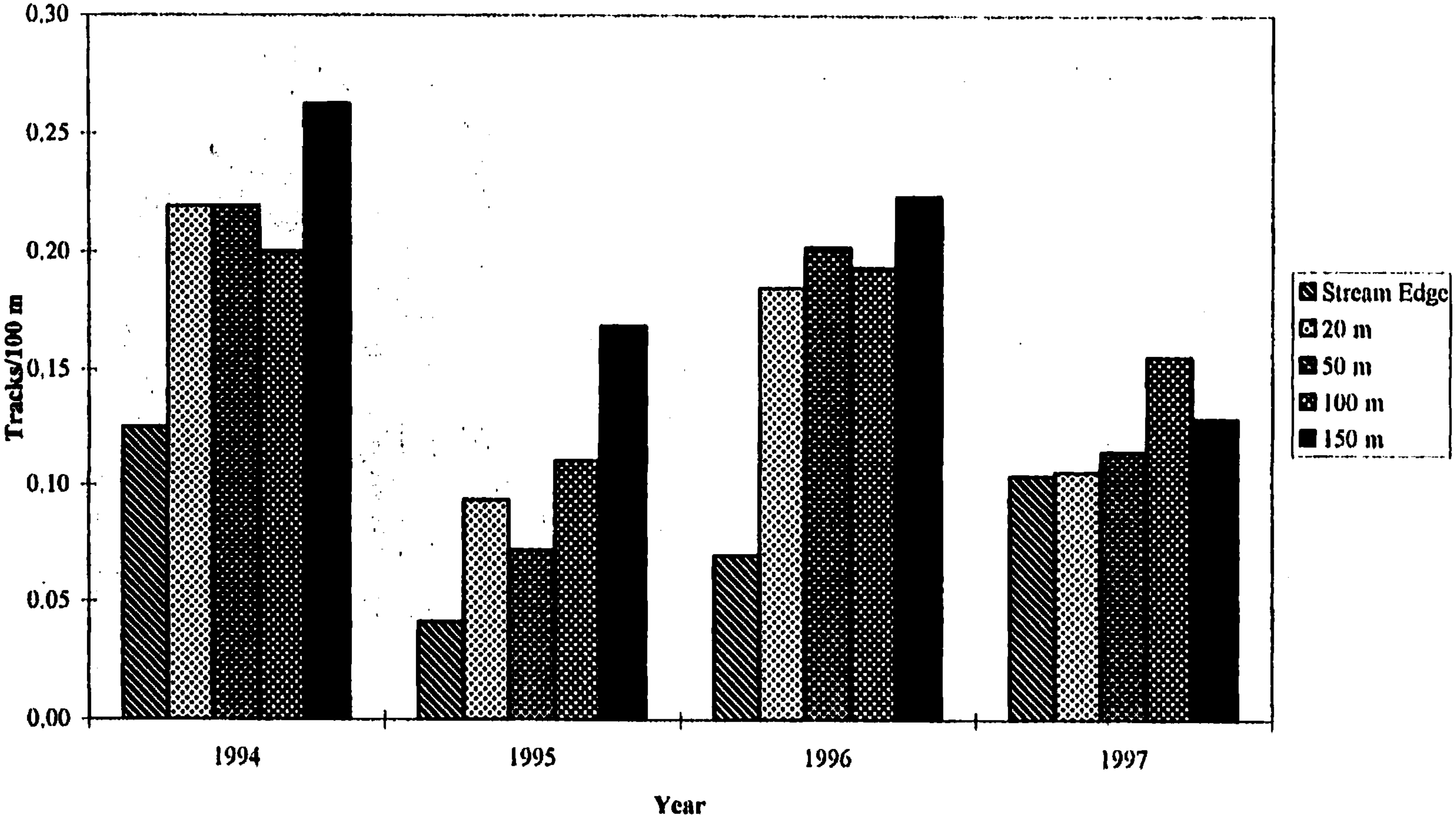

0,30
0,25
0,20
0,15
0,10
0,05
0,00
Tracks/100 m
1994
1995
1996
1997
Year
▧ Stream Edge
▩ 20 m
▩ 50 m
▩ 100 m
■ 150 m

Figure 12

Figure 13: The number of mammal winter track sets per 100 m (group average for *Martes americana, Mustela erminea, Vulpes vulpes, Lepus americanus* and *Tamiasciurus hudsonicus*), by site in 1997, with increasing distance from the stream edge.

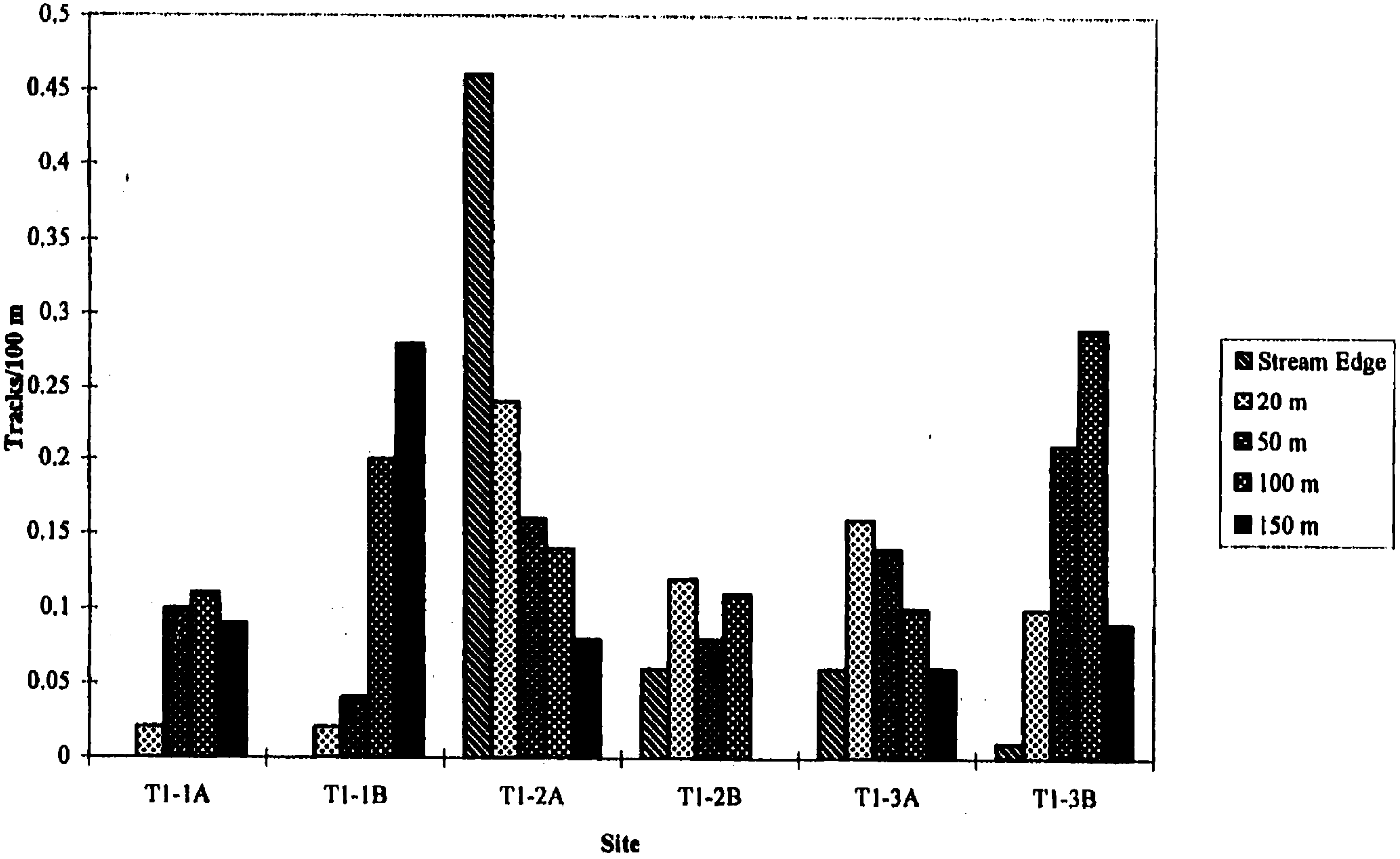

0,5
0,45
0,4
0,35
0,3
0,25
0,2
0,15
0,1
0,05
0
Tracks/100 m
T1-1A
T1-1B
T1-2A
T1-2B
T1-3A
T1-3B
Site
▧ Stream Edge
▨ 20 m
■ 50 m
▨ 100 m
■ 150 m

Figure 13

Table 10: Results of nonparametric multiple comparison tests showing the years with highest and lowest winter track counts for selected mammal species in the Copper Lake watershed.

Species	**Years with Highest Track Counts ($p < 0.05$)**	**Years with Lowest Track Counts ($p < 0.05$)**
Martes americana	1994, 1996	1997, 1995
Mustela erminea	1994, 1996	1995
Vulpes vulpes	1995	1997
Lepus americanus	1994	1996
Tamiasciurus hudsonicus	1996	1994, 1995, 1997

Due to the patchy nature of the sites, individual transects were distinguished either as being open (i.e., if they occurred on a clear-cut or fen) or as forested. There were significantly more mammal tracks (Kruskal-Wallis test, $p < 0.01$) on the forested transects. Analysis by species showed that *M. americana, L. americanus* and *T. hudsonicus* tracks were significantly more frequent on forested transects (Kruskal-Wallis test, $p < 0.05$) (Figure 14). There was no significant difference in *V. vulpes* or *M. erminea* activity on the open or forested transects.

Individual Species - Small Mammalian Predators:

Martes americana:

Pine marten track abundance showed no significant differences among sites or by distance from the stream over the four years combined. However, there was a difference between years (Kruskal-Wallis test, $p < 0.01$) with 1994 and 1996 having significantly more tracks than 1995 or 1997 (Table 10). Examination of the track abundances on individual sites shows a decline in activity from 1994 through to 1997 for both T1-1A and T1-1B (Figures 15a and b), fluctuating abundances for T1-2A, T1-2B and T1-3A (Figures 15c, d and e) and a steady increase in activity in T1-3B (Figure 15f). For 1997 post-harvest data there was a significant difference by site (Kruskal-Wallis test, $p < 0.01$). T1-1A had significantly fewer tracks than T1-3B. Multiple comparison procedures failed to detect any differences between other sites.

Marten track frequencies (i.e., percent of total days surveyed that tracks were

Figure 14: The average number of winter track sets of *Martes americana*, *Mustela erminea*, *Vulpes vulpes*, *Lepus americanus* and *Tamiasciurus hudsonicus* per 100 m occurring on forested and non-forested (i.e., open) transects. Note: asterisk indicates significant differences ($p < 0.05$) in track abundances between forested and open transects.

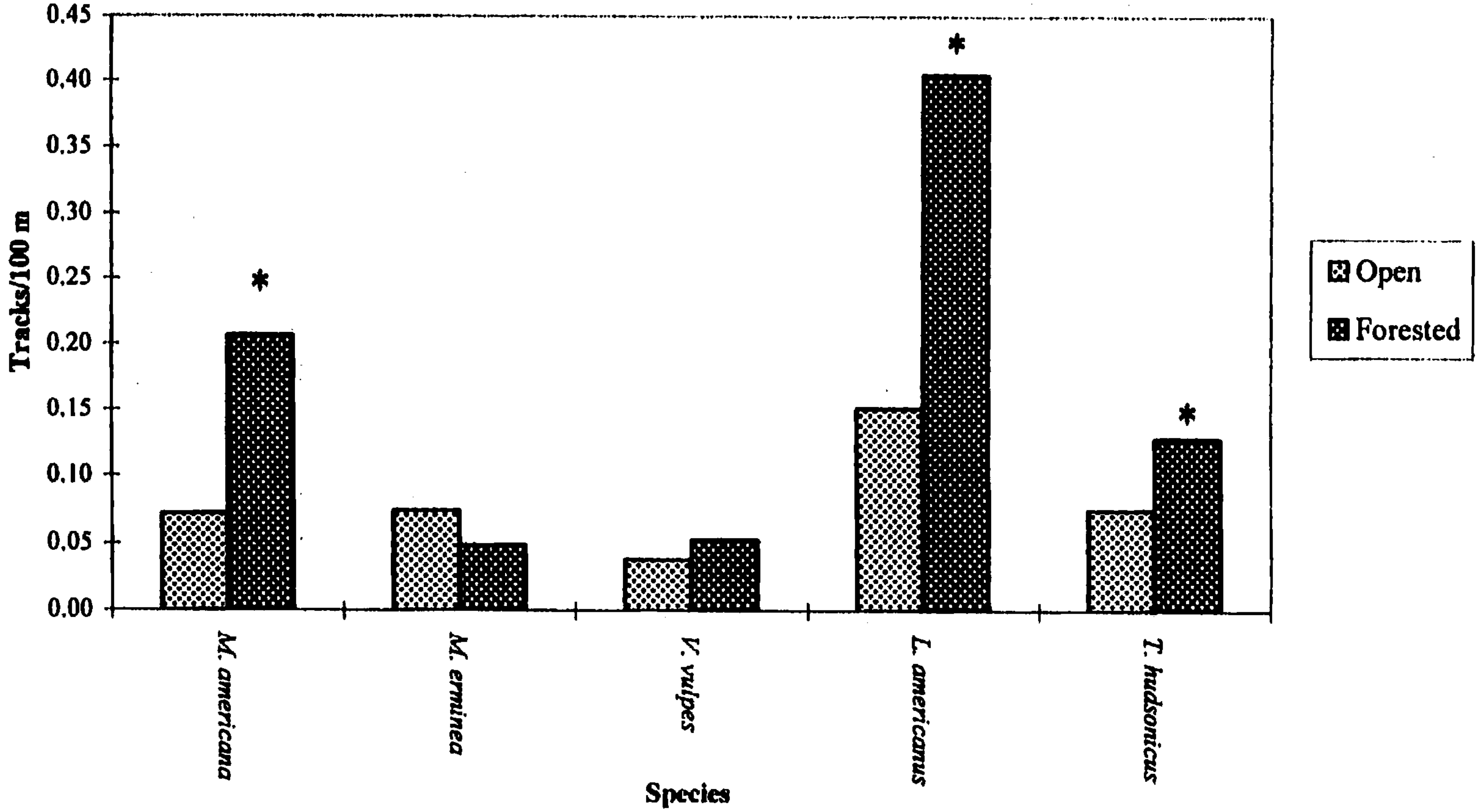

0.45
0.40
0.35
0.30
0.25
0.20
0.15
0.10
0.05
0.00
Tracks/100 m
*
*
*
M. americana
M. erminea
V. vulpes
L. americanus
T. hudsonicus
Species
▨ Open
▨ Forested

Figure 14

Figure 15: a) Winter track abundance (tracks/100 m) of *Martes americana* on T1-1A from 1994 to 1997.

b) Winter track abundance (tracks/100 m) of *Martes americana* on T1-1B from 1994 to 1997.

c) Winter track abundance (tracks/100 m) of *Martes americana* on T1-2A from 1994 to 1997.

Note: values are means calculated from all transects and days surveyed for each site.

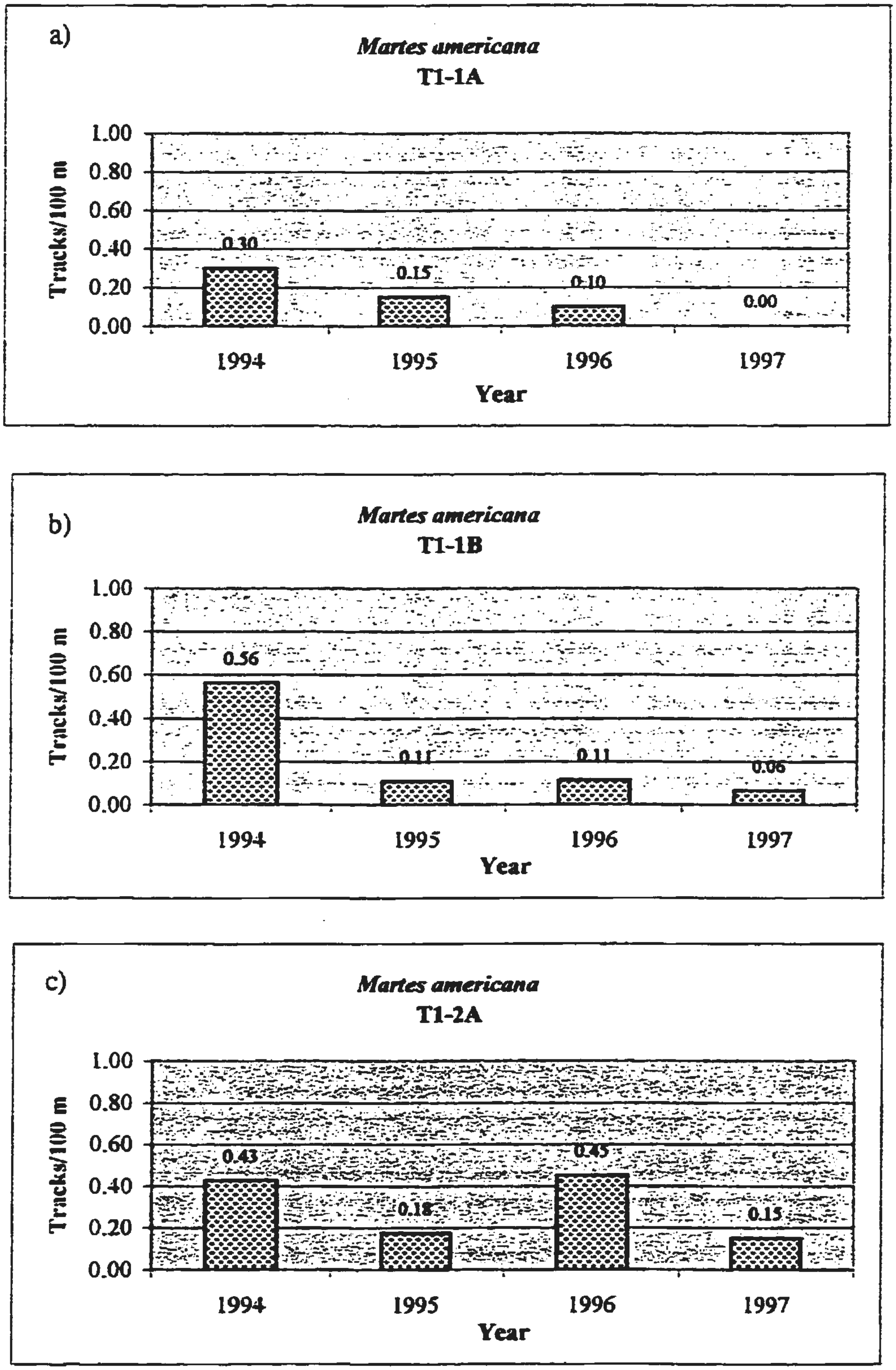

a)
Martes americana
T1-1A
Tracks/100 m
1.00
0.80
0.60
0.40
0.20
0.00
0.30
0.15
0.10
0.00
1994
1995
1996
1997
Year
b)
Martes americana
T1-1B
Tracks/100 m
1.00
0.80
0.60
0.40
0.20
0.00
0.56
0.11
0.11
0.06
1994
1995
1996
1997
Year
c)
Martes americana
T1-2A
Tracks/100 m
1.00
0.80
0.60
0.40
0.20
0.00
0.43
0.18
0.45
0.15
1994
1995
1996
1997
Year

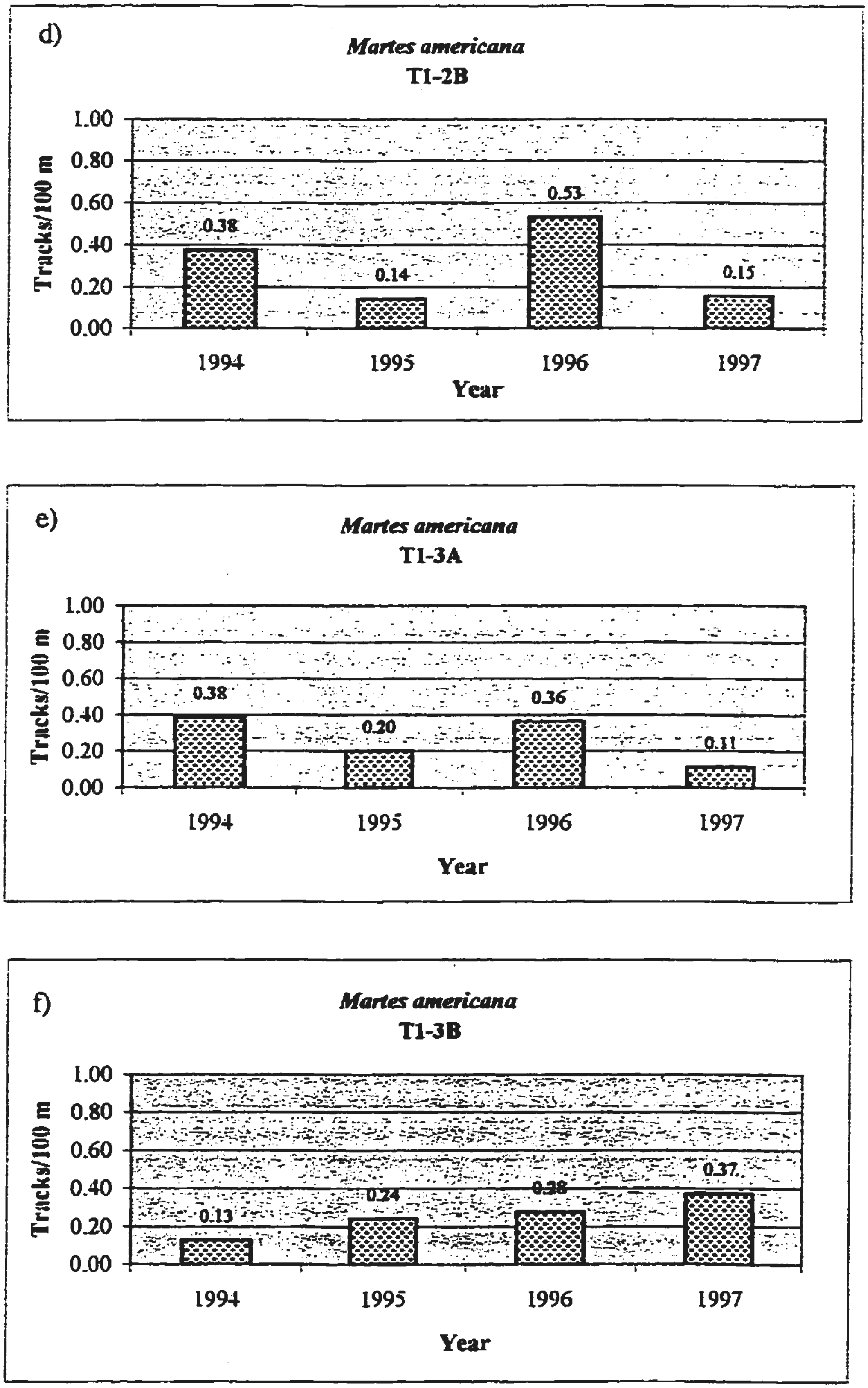
Figure 15

Figure 15: d) Winter track abundance (tracks/100 m) of *Martes americana* on T1-2B from 1994 to 1997.

e) Winter track abundance (tracks/100 m) of *Martes americana* on T1-3A from 1994 to 1997.

f) Winter track abundance (tracks/100 m) of *Martes americana* on T1-3B from 1994 to 1997.

Note: values are means calculated from all transects and days surveyed for each site.



d)
Martes americana
T1-2B
1.00
0.80
0.60
0.40
0.20
0.00
Tracks/100 m
0.38
0.14
0.53
0.15
1994
1995
1996
1997
Year
e)
Martes americana
T1-3A
1.00
0.80
0.60
0.40
0.20
0.00
Tracks/100 m
0.38
0.20
0.36
0.11
1994
1995
1996
1997
Year
f)
Martes americana
T1-3B
1.00
0.80
0.60
0.40
0.20
0.00
Tracks/100 m
0.13
0.24
0.28
0.37
1994
1995
1996
1997
Year

Figure 15

present) showed a decline from 1994 through to 1997 in both T1-1A and T1-1B (Figures 16a and b). Track frequencies in both T1-2A and T1-2B were variable by year (Figures 16c and d), while in T1-3A the frequency was constant and in T1-3B there was an increase from 1994 to 1996 (Figures 16e and f, respectively). In the case of T1-2A and T1-2B in 1997, all of the marten track sets observed, except one, were recorded either within the stream buffers or the buffer left on the lakeside (Table 11).

Mustela erminea:

Weasel track data pooled from all years showed no significant difference by site or with distance from the stream. This was also the case with the 1997 post-harvest data when tested alone. There were significant differences by year (Kruskal-Wallis test, $p < 0.001$) with 1994 and 1996 having significantly more tracks than 1995 (Table 10). Both 1994 and 1996 had more tracks than 1997 but the difference was not significant. On examination of individual sites it can be seen that in most instances weasel tracks were not abundant (Figures 17a - f). In particular, there were no tracks observed on either T1-2B, T1-3A or T1-3B in 1995. All of the weasel tracks on T1-2A and T1-2B in 1997 occurred within the stream and lakeside buffers (Table 11). Weasel track frequencies were highly variable within sites from 1994 to 1997, with the highest frequencies occurring in 1994 (Figures 18a - f).

Figure 16: a) Winter track frequency (total tracks recorded/number of days surveyed) of *Martes americana* on T1-1A from 1994 to 1997.

b) Winter track frequency (total tracks recorded/number of days surveyed) of *Martes americana* on T1-1B from 1994 to 1997.

c) Winter track frequency (total tracks recorded/number of days surveyed) of *Martes americana* on T1-2A from 1994 to 1997.

Note: values are means calculated from all transects surveyed on each site.

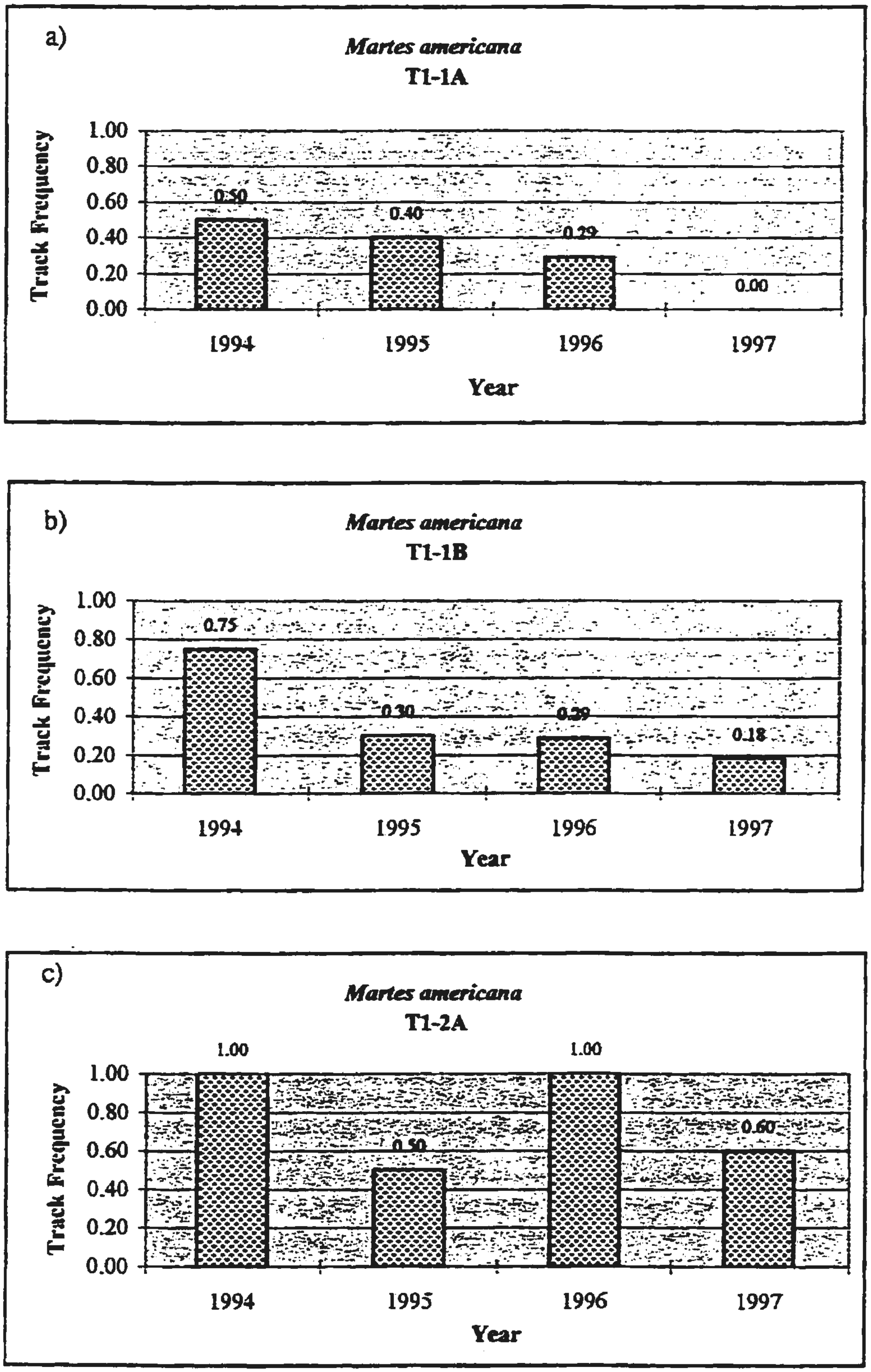

a)
Martes americana
T1-1A
Track Frequency
1.00
0.80
0.60
0.40
0.20
0.00
0.50
0.40
0.29
0.00
1994
1995
1996
1997
Year
b)
Martes americana
T1-1B
Track Frequency
1.00
0.80
0.60
0.40
0.20
0.00
0.75
0.30
0.29
0.18
1994
1995
1996
1997
Year
c)
Martes americana
T1-2A
Track Frequency
1.00
0.80
0.60
0.40
0.20
0.00
1.00
0.50
1.00
0.60
1994
1995
1996
1997
Year

Figure 16

Figure 16: d) Winter track frequency (total tracks recorded/number of days surveyed) of *Martes americana* on T1-2B from 1994 to 1997.

e) Winter track frequency (total tracks recorded/number of days surveyed) of *Martes americana* on T1-3A from 1994 to 1997.

f) Winter track frequency (total tracks recorded/number of days surveyed) of *Martes americana* on T1-3B from 1994 to 1997.

Note: values are means calculated from all transects surveyed on each site.

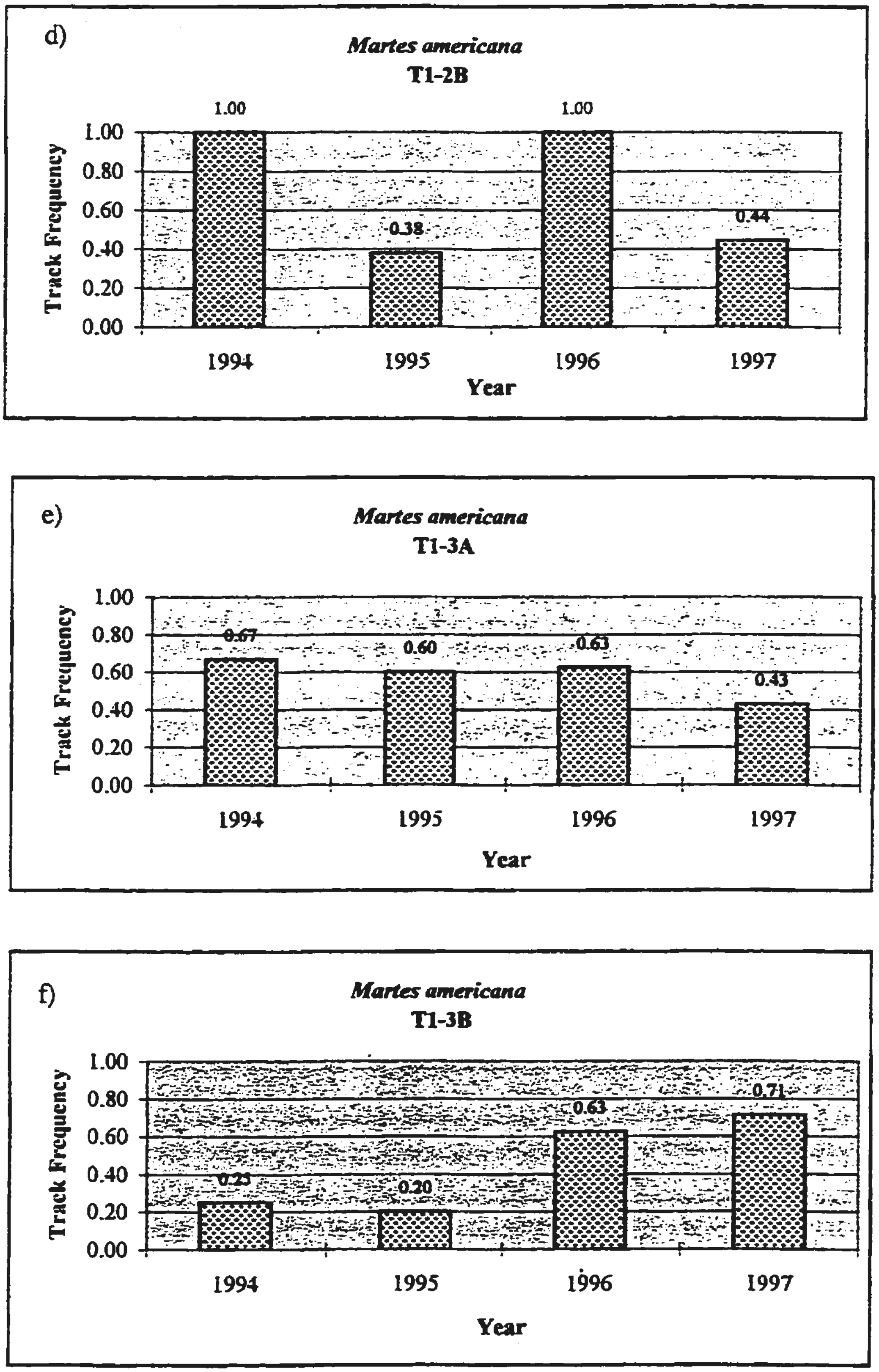

d)
Martes americana
T1-2B
Track Frequency
1.00
0.80
0.60
0.40
0.20
0.00
1.00
0.38
1.00
0.44
1994
1995
1996
1997
Year
e)
Martes americana
T1-3A
Track Frequency
1.00
0.80
0.60
0.40
0.20
0.00
0.67
0.60
0.63
0.43
1994
1995
1996
1997
Year
f)
Martes americana
T1-3B
Track Frequency
1.00
0.80
0.60
0.40
0.20
0.00
0.25
0.20
0.63
0.71
1994
1995
1996
1997
Year

Figure 16

Table 11: Winter track data for 1997 showing the percentage of tracks occurring inside and outside the stream and lake buffers on T1-2A and T1-2B (actual number of tracks in parentheses). Note: T1-2A and T1-2B were the only buffer sites surveyed for winter tracks.

Species	**T1-2A**		**T1-2B**	
	Inside Buffer	**Outside Buffer**	**Inside Buffer**	**Outside Buffer**
Martes americana	100 (11)	0 (0)	89 (8)	11 (1)
Mustela erminea	100 (5)	0 (0)	100 (3)	0 (0)
Mustela vison	100 (3)	0 (0)	80 (4)	20 (1)
Vulpes vulpes	100 (8)	0 (0)	0 (0)	100 (1)
Lepus americanus	100 (58)	0 (0)	100 (9)	0 (0)
Tamiasciurus hudsonicus	100 (4)	0 (0)	100 (3)	0 (0)

Figure 17: a) Winter track abundance (tracks/100 m) of *Mustela erminea* on T1-1A from 1994 to 1997.

b) Winter track abundance (tracks/100 m) of *Mustela erminea* on T1-1B from 1994 to 1997.

c) Winter track abundance (tracks/100 m) of *Mustela erminea* on T1-2A from 1994 to 1997.

Note: values are means calculated from all transects and days surveyed for each site.

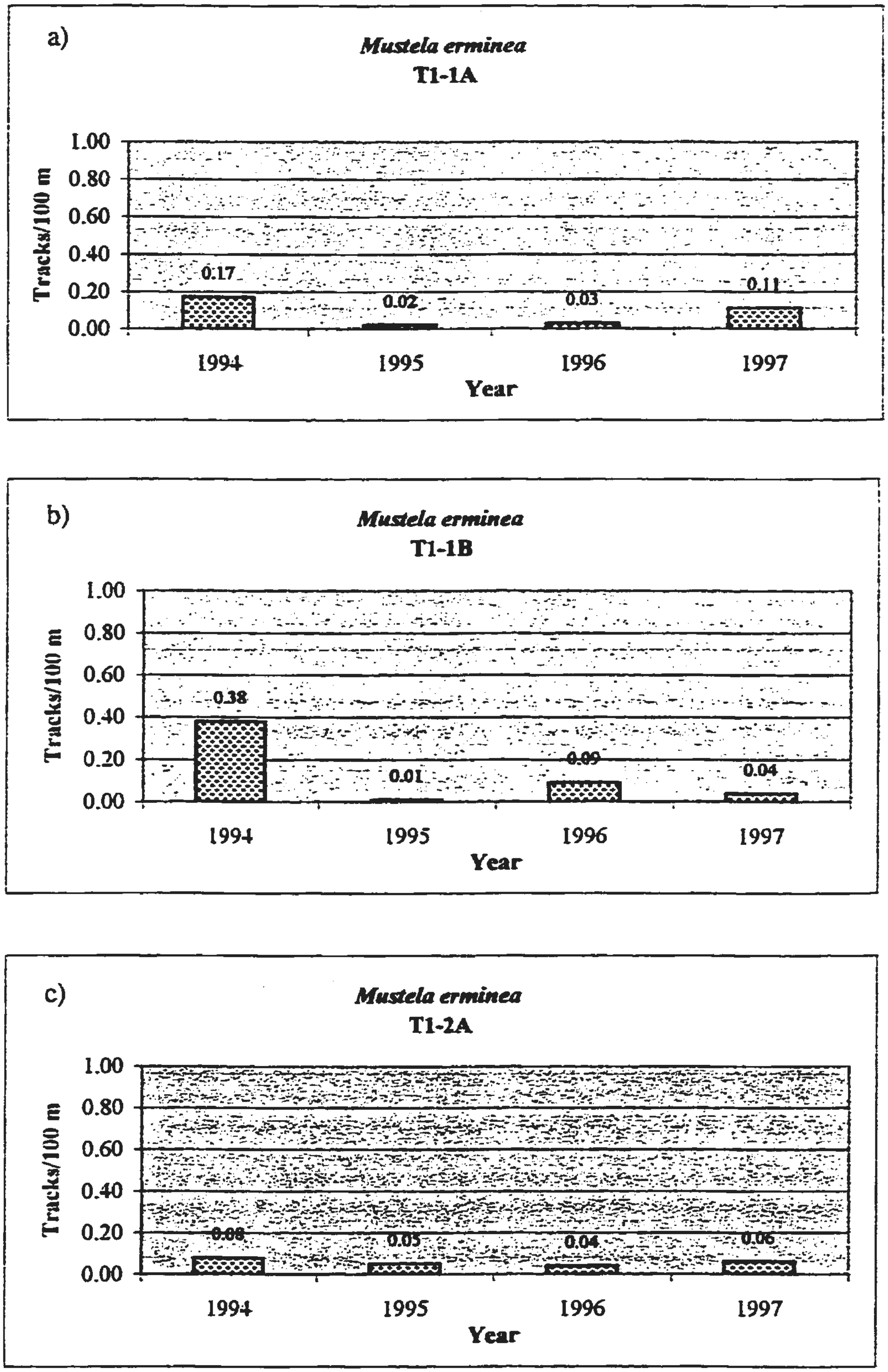

a)
Mustela erminea
T1-1A
Tracks/100 m
1.00
0.80
0.60
0.40
0.20
0.00
0.17
0.02
0.03
0.11
1994
1995
1996
1997
Year
b)
Mustela erminea
T1-1B
Tracks/100 m
1.00
0.80
0.60
0.40
0.20
0.00
0.38
0.01
0.09
0.04
1994
1995
1996
1997
Year
c)
Mustela erminea
T1-2A
Tracks/100 m
1.00
0.80
0.60
0.40
0.20
0.00
0.08
0.05
0.04
0.06
1994
1995
1996
1997
Year

Figure 17

Figure 17: d) Winter track abundance (tracks/100 m) of *Mustela erminea* on T1-2B from 1994 to 1997.

e) Winter track abundance (tracks/100 m) of *Mustela erminea* on T1-3A from 1994 to 1997.

f) Winter track abundance (tracks/100 m) of *Mustela erminea* on T1-3B from 1994 to 1997.

Note: values are means calculated from all transects and days surveyed for each site.

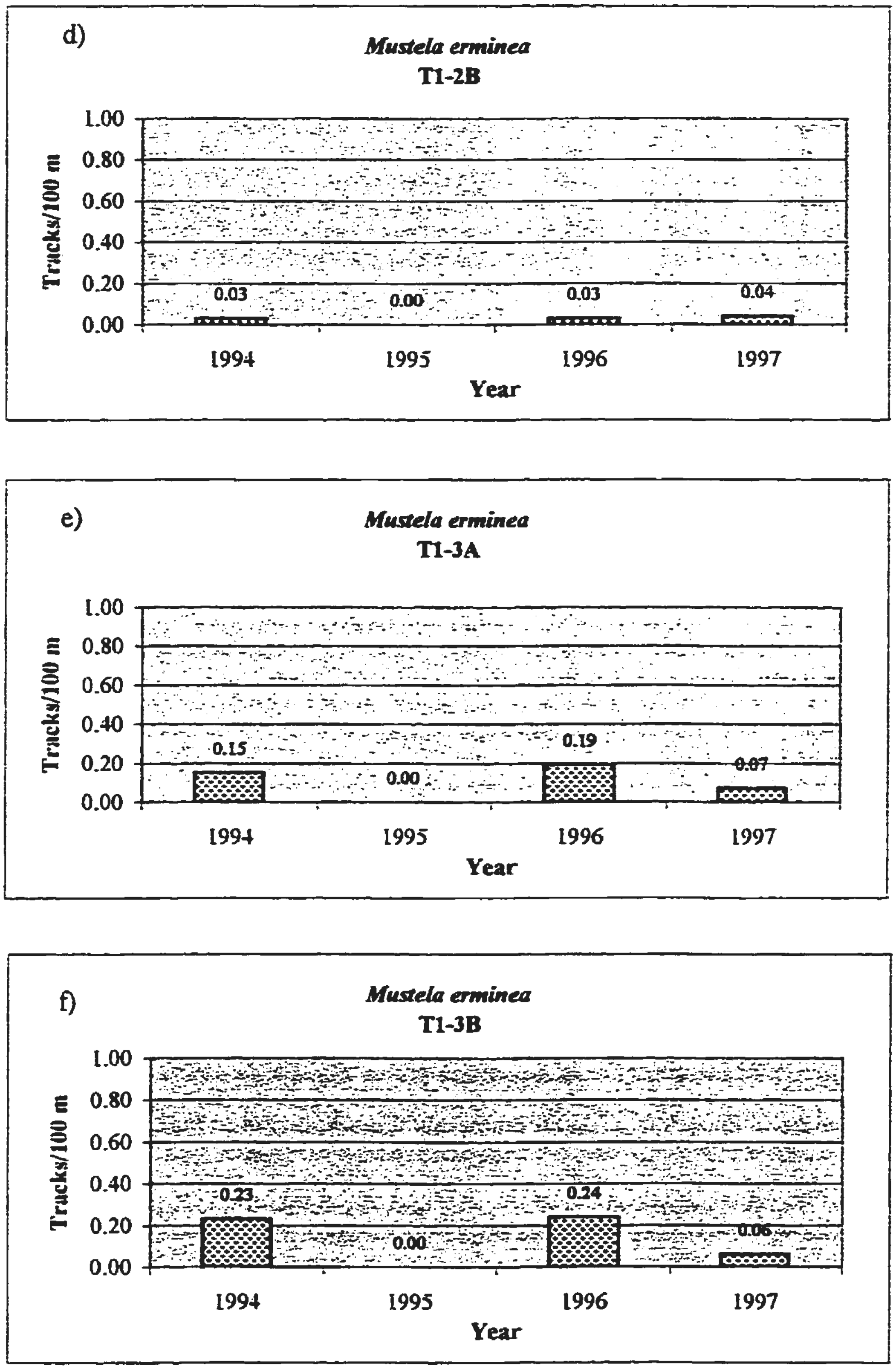

d)
Mustela erminea
T1-2B
Tracks/100 m
1.00
0.80
0.60
0.40
0.20
0.00
0.03
0.00
0.03
0.04
1994
1995
1996
1997
Year
e)
Mustela erminea
T1-3A
Tracks/100 m
1.00
0.80
0.60
0.40
0.20
0.00
0.15
0.00
0.19
0.07
1994
1995
1996
1997
Year
f)
Mustela erminea
T1-3B
Tracks/100 m
1.00
0.80
0.60
0.40
0.20
0.00
0.23
0.00
0.24
0.06
1994
1995
1996
1997
Year

Figure 17

Figure 18: a) Winter track frequency (total tracks recorded/number of days surveyed) of *Mustela erminea* on T1-1A from 1994 to 1997.

b) Winter track frequency (total tracks recorded/number of days surveyed) of *Mustela erminea* on T1-1B from 1994 to 1997.

c) Winter track frequency (total tracks recorded/number of days surveyed) of *Mustela erminea* on T1-2A from 1994 to 1997.

Note: values are means calculated from all transects surveyed on each site.

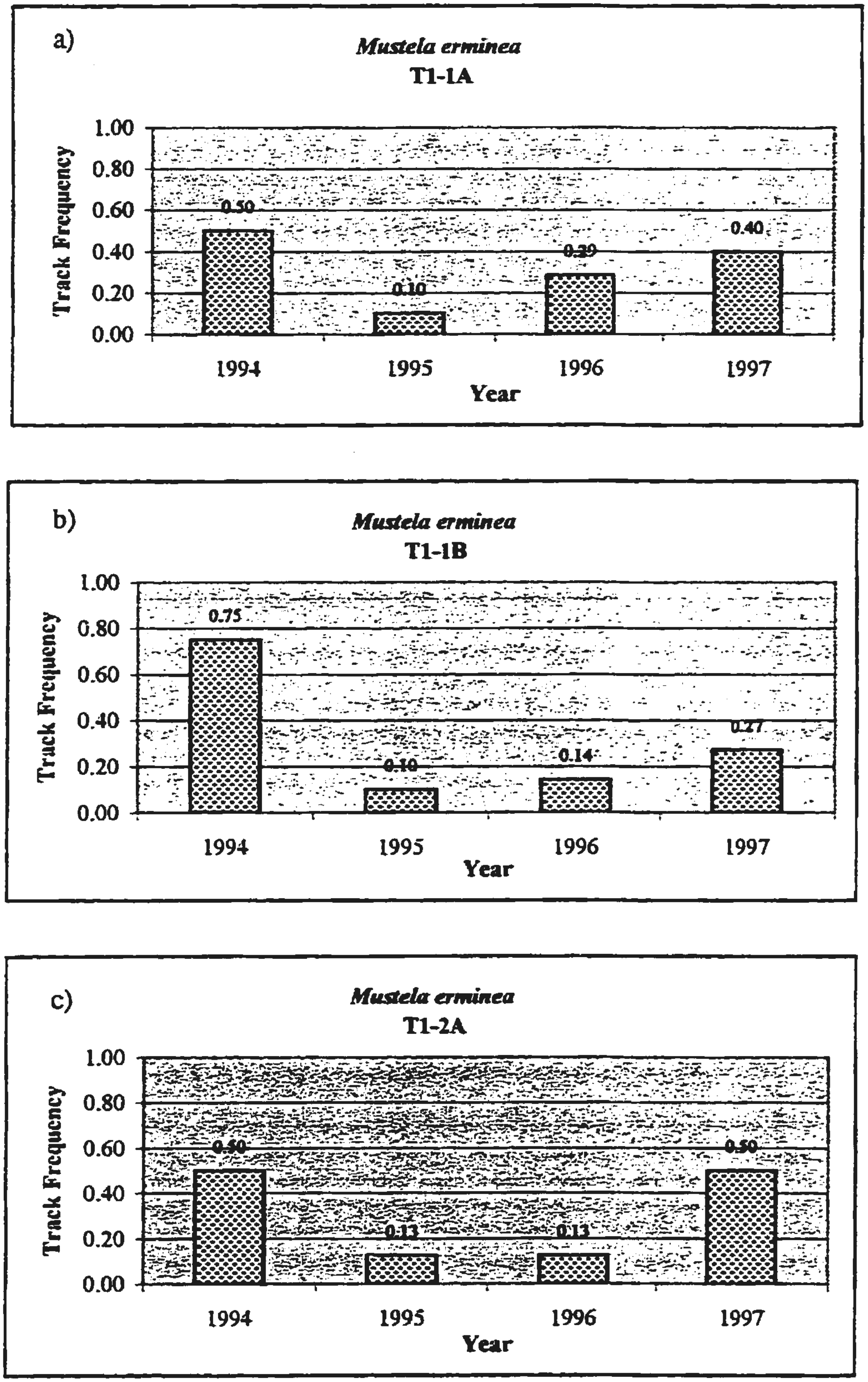

a)
Mustela erminea
T1-1A
Track Frequency
1.00
0.80
0.60
0.40
0.20
0.00
0.50
0.10
0.29
0.40
1994
1995
1996
1997
Year
b)
Mustela erminea
T1-1B
Track Frequency
1.00
0.80
0.60
0.40
0.20
0.00
0.75
0.10
0.14
0.27
1994
1995
1996
1997
Year
c)
Mustela erminea
T1-2A
Track Frequency
1.00
0.80
0.60
0.40
0.20
0.00
0.50
0.13
0.13
0.50
1994
1995
1996
1997
Year

Figure 18

Figure 18: d) Winter track frequency (total tracks recorded/number of days surveyed) of *Mustela erminea* on T1-2B from 1994 to 1997.

e) Winter track frequency (total tracks recorded/number of days surveyed) of *Mustela erminea* on T1-3A from 1994 to 1997.

f) Winter track frequency (total tracks recorded/number of days surveyed) of *Mustela erminea* on T1-3B from 1994 to 1997.

Note: values are means calculated from all transects surveyed on each site.

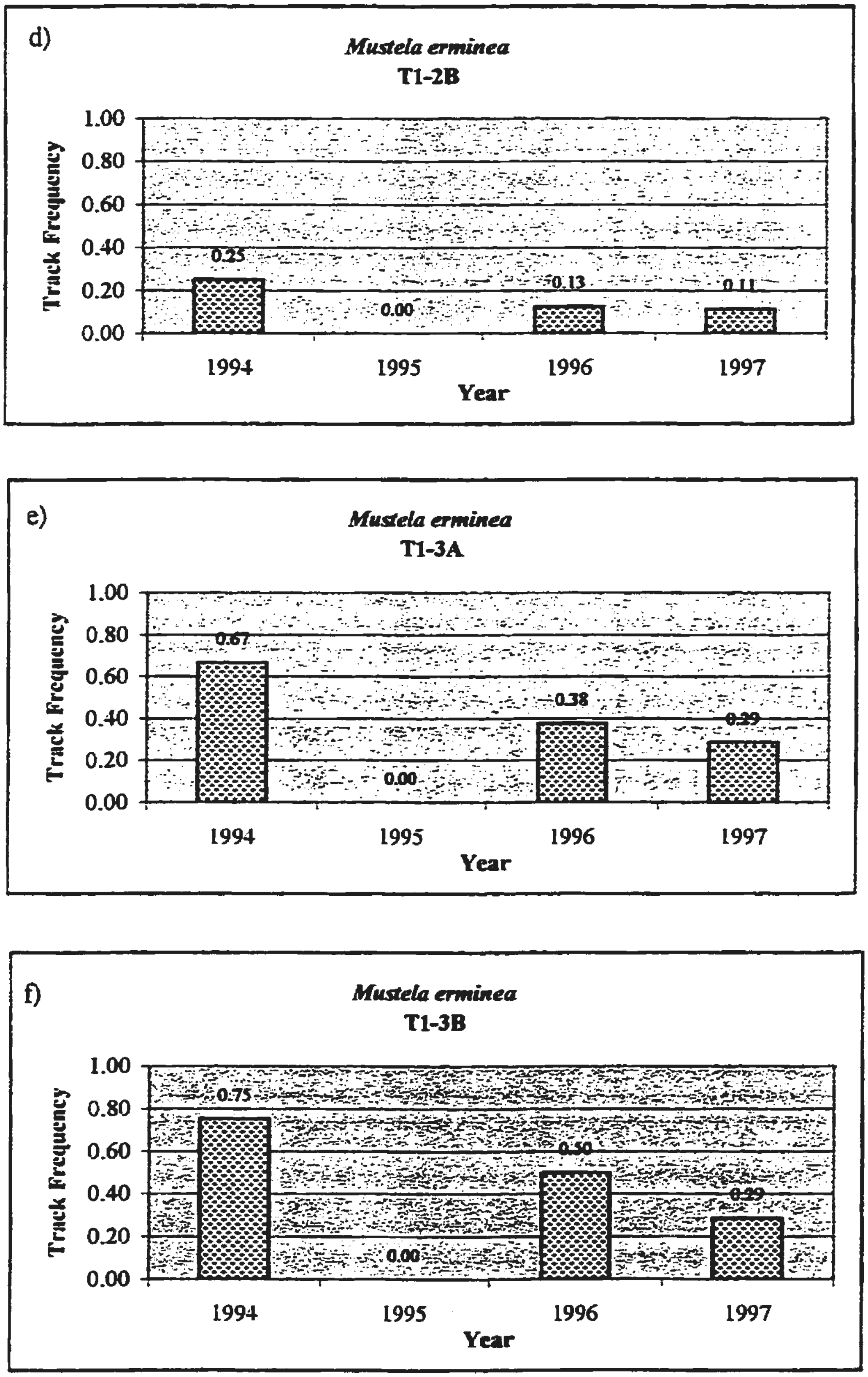

d)
Mustela erminea
T1-2B
Track Frequency
1.00
0.80
0.60
0.40
0.20
0.00
0.25
0.00
0.13
0.11
1994
1995
1996
1997
Year
e)
Mustela erminea
T1-3A
Track Frequency
1.00
0.80
0.60
0.40
0.20
0.00
0.67
0.00
0.38
0.29
1994
1995
1996
1997
Year
f)
Mustela erminea
T1-3B
Track Frequency
1.00
0.80
0.60
0.40
0.20
0.00
0.75
0.00
0.50
0.29
1994
1995
1996
1997
Year

Figure 18

Vulpes vulpes:

Red fox tracks were generally not abundant in any year or site (Figures 19a -f), however, the data showed a significant difference by site over all years (Kruskal-Wallis test, $p < 0.05$). T1-3A had the most tracks while T1-1B had the fewest. There were also significant differences by year (Kruskal-Wallis test, $p < 0.01$) with the 1995 data showing significantly more tracks than 1997 (Table 10). No other significant differences were detected among years. The 1997 post-harvest data showed significant differences by site (Kruskal-Wallis test, $p < 0.01$). The clear-cuts at T1-1A and B had the least amount of activity while T1-2A had the most. On T1-2A in 1997, all of the tracks occurred within the buffers, while on T1-2B, the one set of tracks observed was on the clear-cut (Table 11). On the least disturbed site, T1-3B, there were no red fox tracks in this year (Figure 19f). Red fox track frequencies were variable from site to site and year to year (Figures 20a -f).

Mustela vison:

Although mink tracks were too infrequent for statistical analysis, having been recorded only in 1995 and 1997, they were considered, because the mink is a riparian species. Mink tracks were observed on all sites in 1995 and 14 of the 17 sets of tracks occurred within 20 m of the stream edges. In 1997, mink tracks were only found on T1-2A, T1-2B and T1-3A. Of the eight sets of tracks occurring on T1-2A and T1-2B, only one set was outside of the stream and lakeside buffers (Table 11).

Figure 19: a) Winter track abundance (tracks/100 m) of *Vulpes vulpes* on T1-1A from 1994 to 1997.

b) Winter track abundance (tracks/100 m) of *Vulpes vulpes* on T1-1B from 1994 to 1997.

c) Winter track abundance (tracks/100 m) of *Vulpes vulpes* on T1-2A from 1994 to 1997.

Note: values are means calculated from all transects and days surveyed for each site.

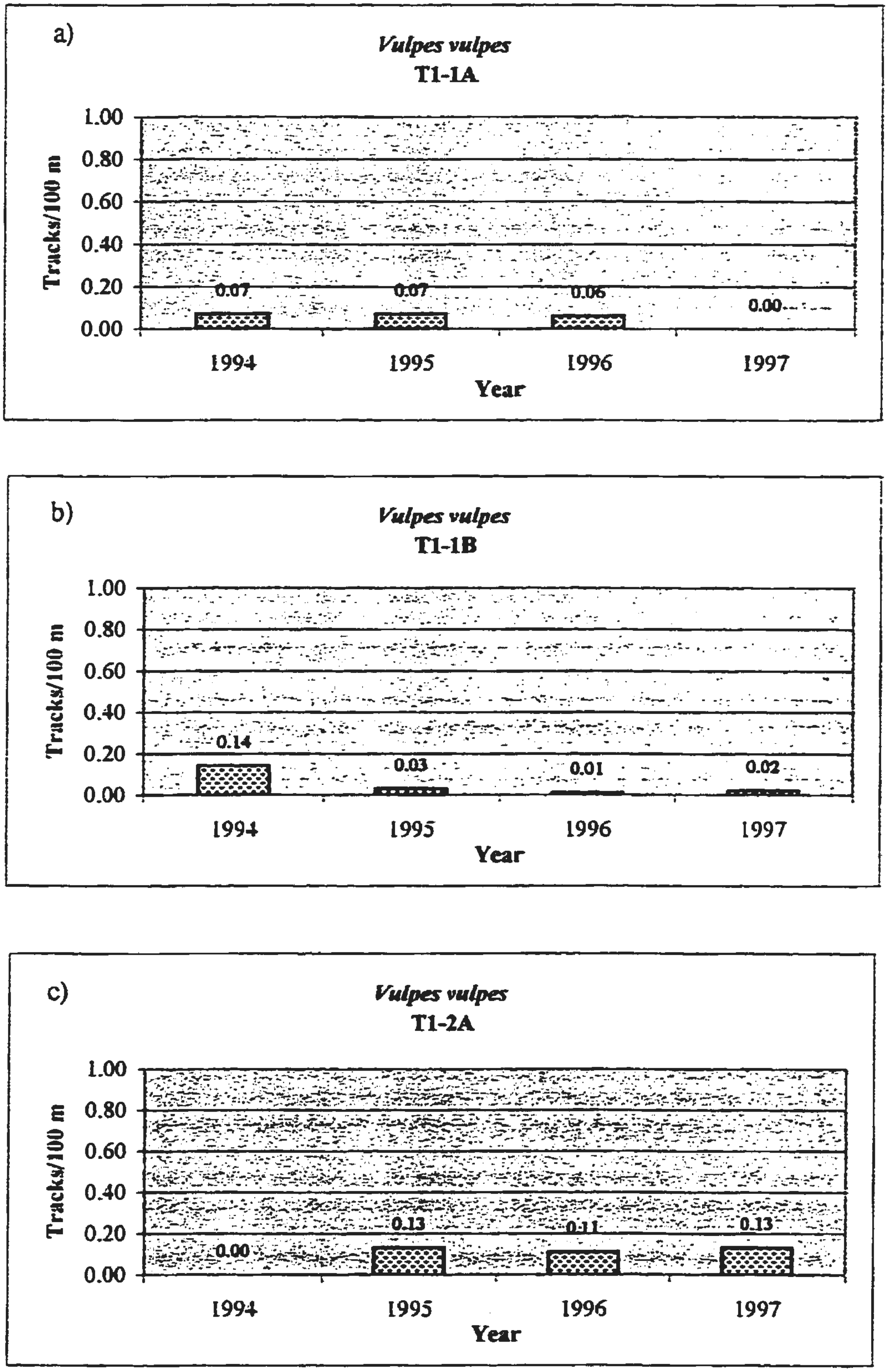

a)
Vulpes vulpes
T1-1A
Tracks/100 m
1.00
0.80
0.60
0.40
0.20
0.00
0.07
0.07
0.06
0.00
1994
1995
1996
1997
Year
b)
Vulpes vulpes
T1-1B
Tracks/100 m
1.00
0.80
0.60
0.40
0.20
0.00
0.14
0.03
0.01
0.02
1994
1995
1996
1997
Year
c)
Vulpes vulpes
T1-2A
Tracks/100 m
1.00
0.80
0.60
0.40
0.20
0.00
0.00
0.13
0.11
0.13
1994
1995
1996
1997
Year

Figure 19

Figure 19: d) Winter track abundance (tracks/100 m) of *Vulpes vulpes* on T1-2B from 1994 to 1997.

e) Winter track abundance (tracks/100 m) of *Vulpes vulpes* on T1-3A from 1994 to 1997.

f) Winter track abundance (tracks/100 m) of *Vulpes vulpes* on T1-3B from 1994 to 1997.

Note: values are means calculated from all transects and days surveyed for each site.

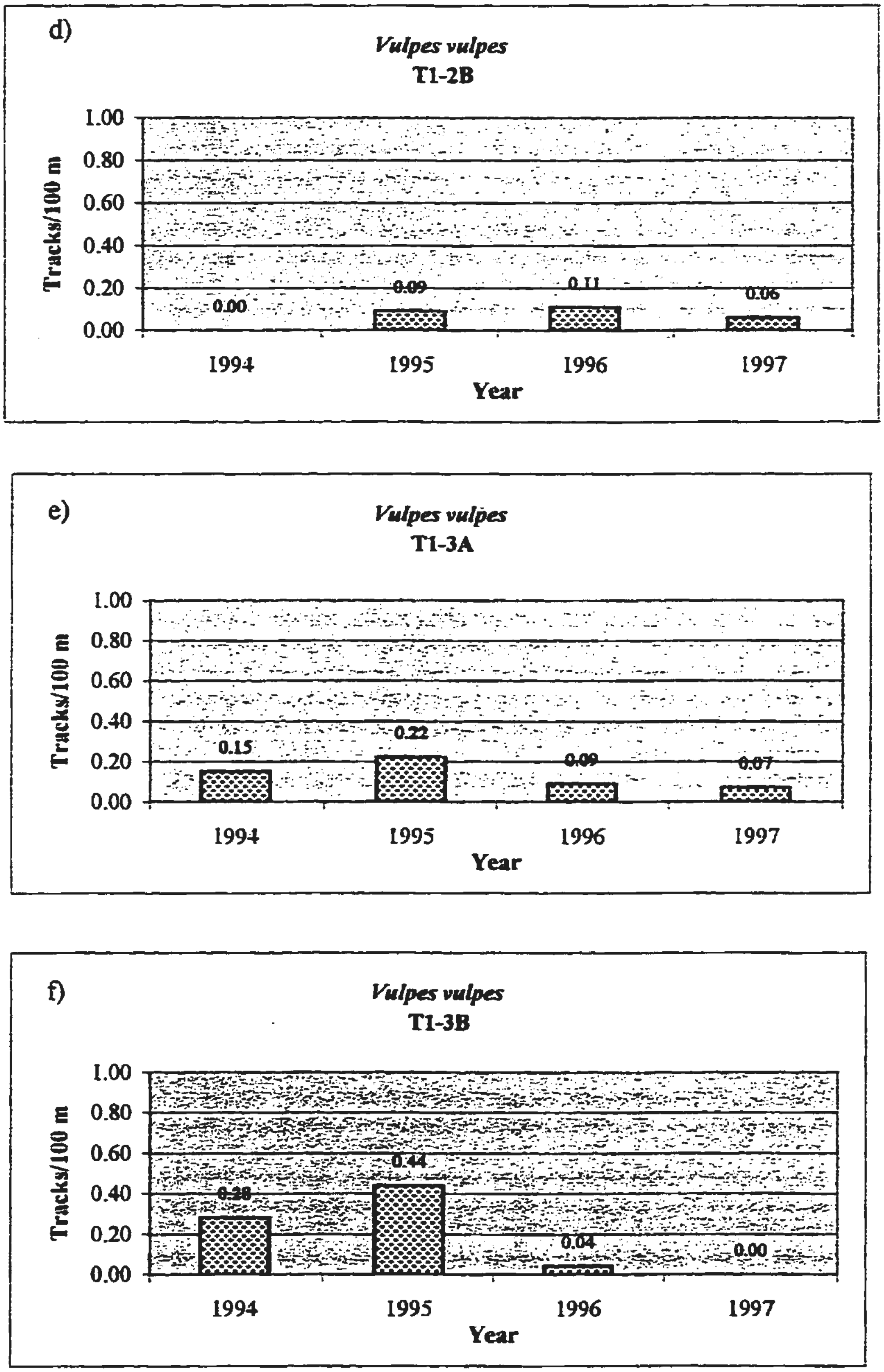

d)
Vulpes vulpes
T1-2B
Tracks/100 m
1.00
0.80
0.60
0.40
0.20
0.00
0.00
0.09
0.11
0.06
1994
1995
1996
1997
Year
e)
Vulpes vulpes
T1-3A
Tracks/100 m
1.00
0.80
0.60
0.40
0.20
0.00
0.15
0.22
0.09
0.07
1994
1995
1996
1997
Year
f)
Vulpes vulpes
T1-3B
Tracks/100 m
1.00
0.80
0.60
0.40
0.20
0.00
0.28
0.44
0.04
0.00
1994
1995
1996
1997
Year

Figure 19

Figure 20: a) Winter track frequency (total tracks recorded/number of days surveyed) of *Vulpes vulpes* on T1-1A from 1994 to 1997.

b) Winter track frequency (total tracks recorded/number of days surveyed) of *Vulpes vulpes* on T1-1B from 1994 to 1997.

c) Winter track frequency (total tracks recorded/number of days surveyed) of *Vulpes vulpes* on T1-2A from 1994 to 1997.

Note: values are means calculated from all transects surveyed on each site.

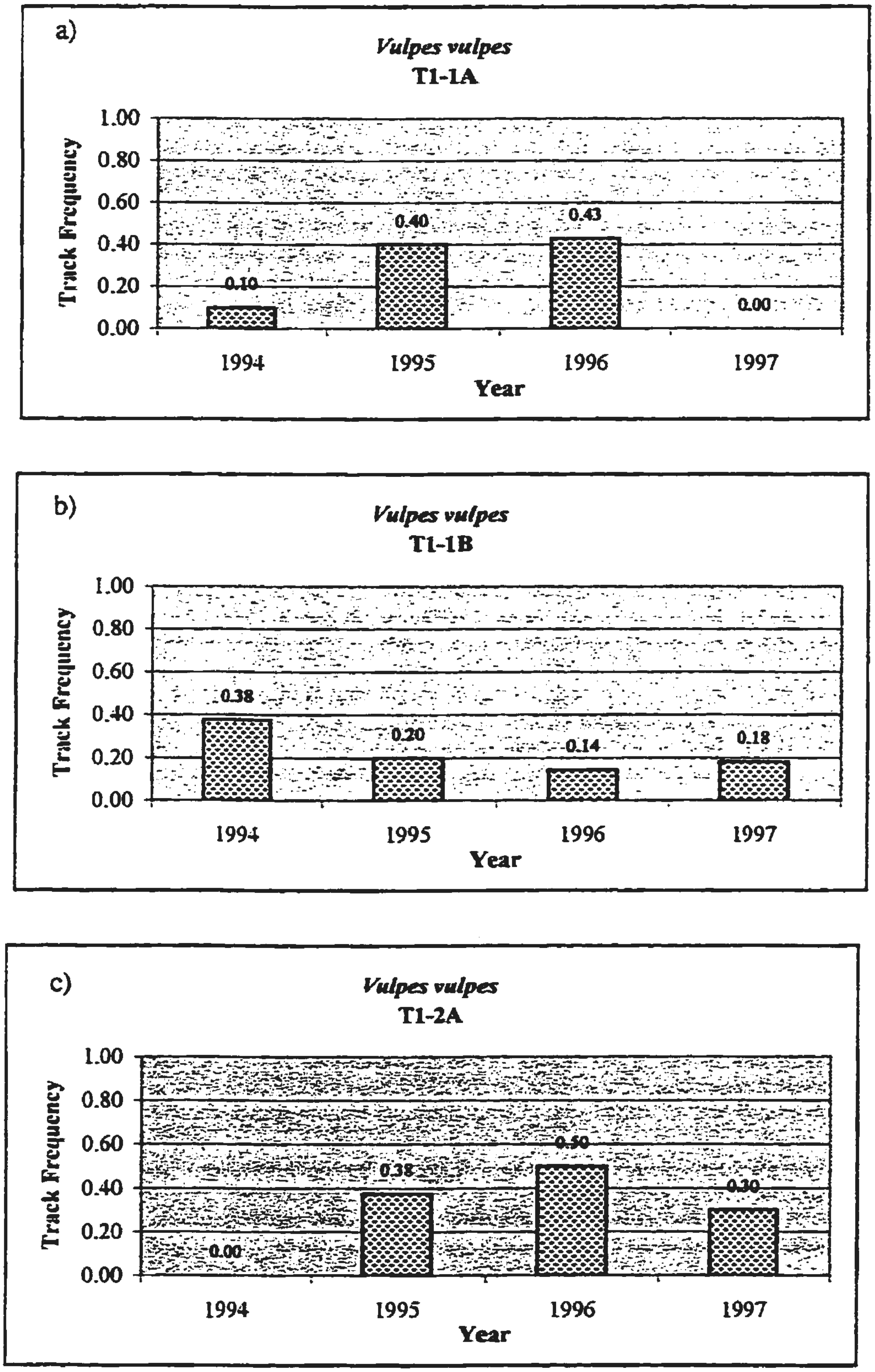

a)
Vulpes vulpes
T1-1A
Track Frequency
1.00
0.80
0.60
0.40
0.20
0.00
0.10
0.40
0.43
0.00
1994
1995
1996
1997
Year
b)
Vulpes vulpes
T1-1B
Track Frequency
1.00
0.80
0.60
0.40
0.20
0.00
0.38
0.20
0.14
0.18
1994
1995
1996
1997
Year
c)
Vulpes vulpes
T1-2A
Track Frequency
1.00
0.80
0.60
0.40
0.20
0.00
0.00
0.38
0.50
0.30
1994
1995
1996
1997
Year

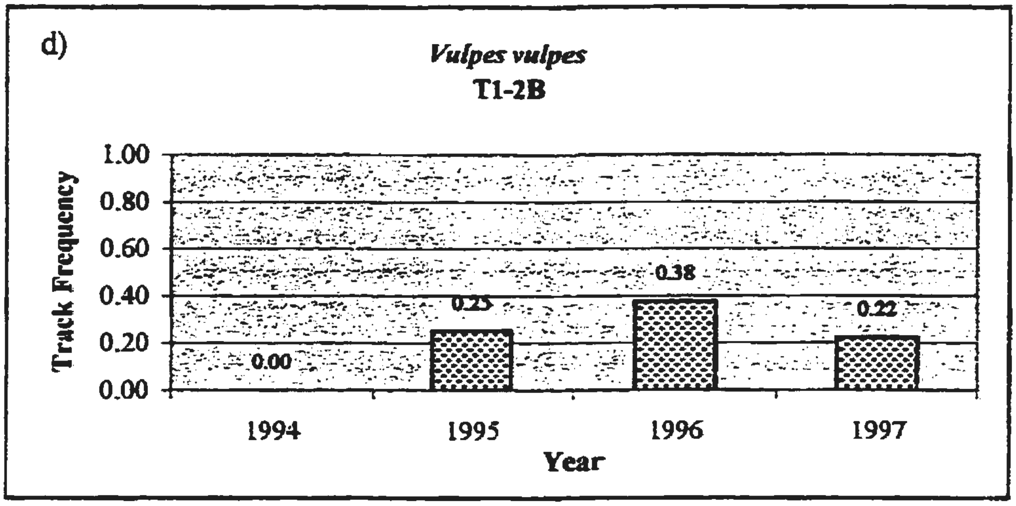
Figure 20

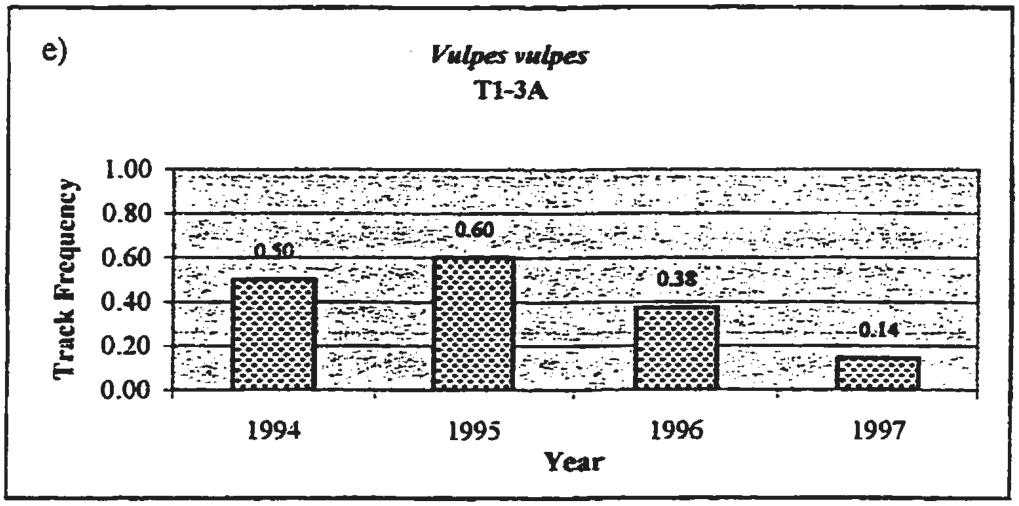
Figure 20: d) Winter track frequency (total tracks recorded/number of days surveyed) of *Vulpes vulpes* on T1-2B from 1994 to 1997.

e) Winter track frequency (total tracks recorded/number of days surveyed) of *Vulpes vulpes* on T1-3A from 1994 to 1997.

f) Winter track frequency (total tracks recorded/number of days surveyed) of *Vulpes vulpes* on T1-3B from 1994 to 1997.

Note: values are means calculated from all transects surveyed on each site.


d)
Vulpes vulpes
T1-2B
Track Frequency
1.00
0.80
0.60
0.40
0.20
0.00
0.00
0.25
0.38
0.22
1994
1995
1996
1997
Year


e)
Vulpes vulpes
T1-3A
Track Frequency
1.00
0.80
0.60
0.40
0.20
0.00
0.50
0.60
0.38
0.14
1994
1995
1996
1997
Year

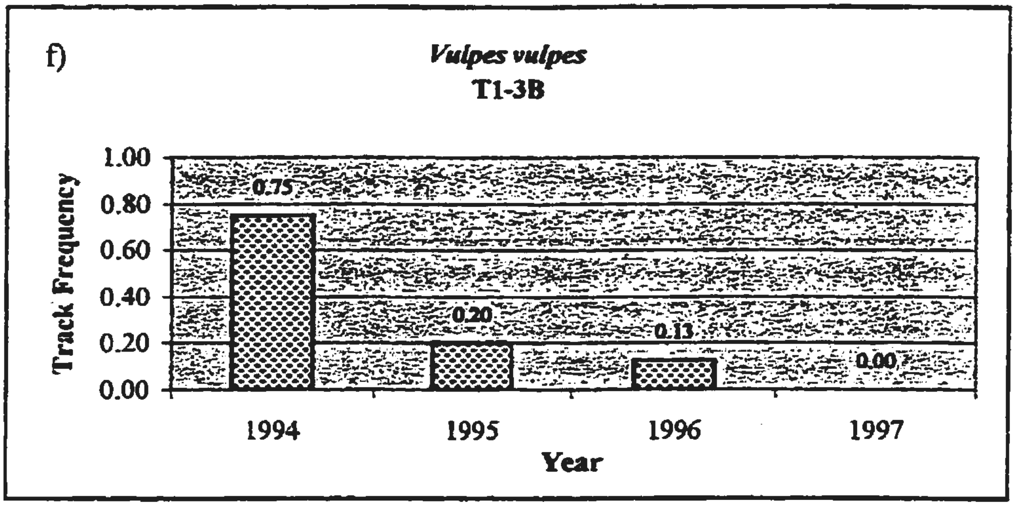

f)
Vulpes vulpes
T1-3B
Track Frequency
1.00
0.80
0.60
0.40
0.20
0.00
0.75
0.20
0.13
0.00
1994
1995
1996
1997
Year

Figure 20

Individual Species - Small Mammalian Prey:

Lepus americanus:

Snowshoe hare track data pooled from 1994 to 1997 showed no significant difference in abundance with distance from the stream, but the difference among sites was significant (Kruskal-Wallis test, $p < 0.001$). T1-1A and B had significantly fewer tracks than T1-3B. There were also significant differences by year (Kruskal-Wallis test, $p < 0.01$). 1994 and 1997 showed significantly more tracks than 1995 and 1996 (Table 10). Track abundances were variable within each site from 1994 to 1997 (Figures 21a - f) with the greatest range in abundance in T1-3B (dropping from 1.48 tracks/100 m in 1994 to 0.07 tracks/100 m in 1997) (Figure 21f). The 1997 data showed no significant differences in hare activity on the clear-cuts, buffer sites or the uncut sites. However, on T1-2A and T1-2B, all of the tracks were recorded within the stream and lake buffers and not on the cut-overs (Table 11). Snowshoe hare track frequency was highly variable (Figures 22a - f), but did show an increase over time in both T1-1A and T1-2A (Figures 22a and c) and a decrease in T1-3B (Figure 22f).

Tamiasciurus hudsonicus:

Red squirrel track abundance showed no difference by site or with distance from the stream for data combined over the four winters. However, there were significant differences by year (Kruskal-Wallis test, $p < 0.0001$). 1996 had significantly more activity than all other years (Table 10). No squirrel tracks occurred on T1-1B, T1-2A or

Figure 21: a) Winter track abundance (tracks/100 m) of *Lepus americanus* on T1-1A from 1994 to 1997.

b) Winter track abundance (tracks/100 m) of *Lepus americanus* on T1-1B from 1994 to 1997.

c) Winter track abundance (tracks/100 m) of *Lepus americanus* on T1-2A from 1994 to 1997.

Note: values are means calculated from all transects and days surveyed for each site.

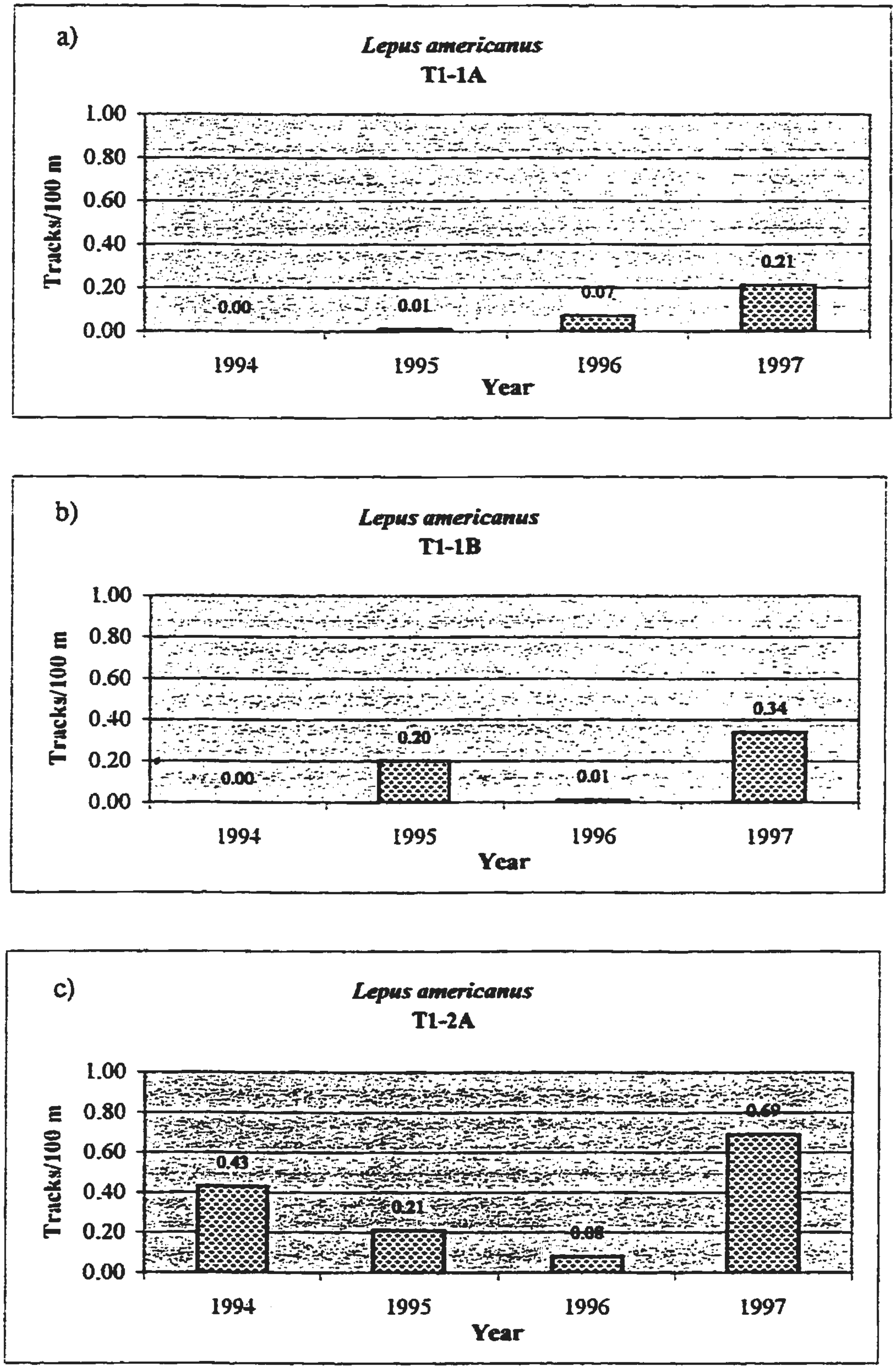

a)
Lepus americanus
T1-1A
Tracks/100 m
1.00
0.80
0.60
0.40
0.20
0.00
0.00
0.01
0.07
0.21
1994
1995
1996
1997
Year
b)
Lepus americanus
T1-1B
Tracks/100 m
1.00
0.80
0.60
0.40
0.20
0.00
0.00
0.20
0.01
0.34
1994
1995
1996
1997
Year
c)
Lepus americanus
T1-2A
Tracks/100 m
1.00
0.80
0.60
0.40
0.20
0.00
0.43
0.21
0.08
0.69
1994
1995
1996
1997
Year

Figure 21

Figure 21: d) Winter track abundance (tracks/100 m) of *Lepus americanus* on T1-2B from 1994 to 1997.

e) Winter track abundance (tracks/100 m) of *Lepus americanus* on T1-3A from 1994 to 1997.

f) Winter track abundance (tracks/100 m) of *Lepus americanus* on T1-3B from 1994 to 1997.

Note: Values are means calculated from all transects and days surveyed for each site.

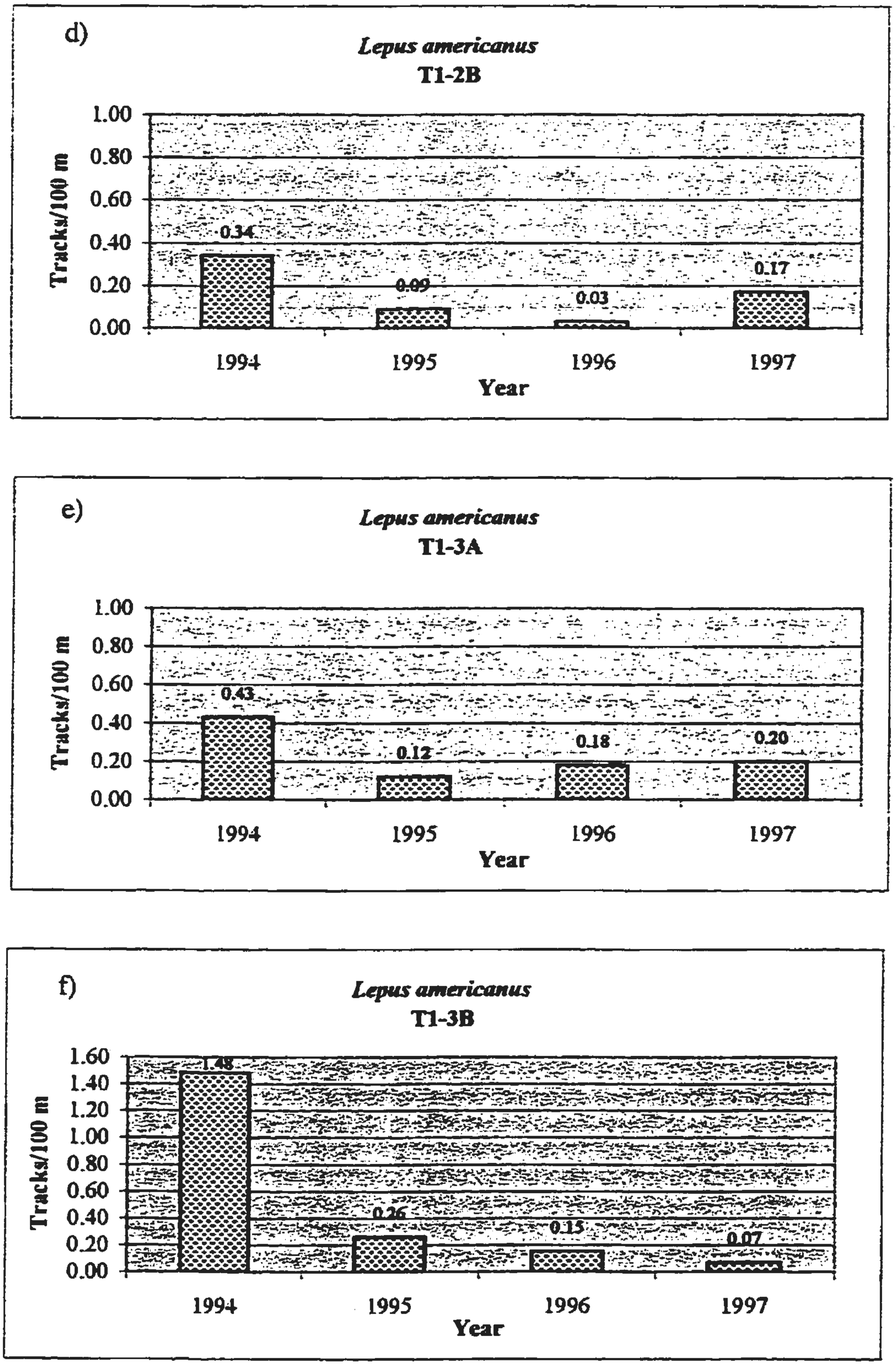

d)
Lepus americanus
T1-2B
Tracks/100 m
1.00
0.80
0.60
0.40
0.20
0.00
0.34
0.09
0.03
0.17
1994
1995
1996
1997
Year
e)
Lepus americanus
T1-3A
Tracks/100 m
1.00
0.80
0.60
0.40
0.20
0.00
0.43
0.12
0.18
0.20
1994
1995
1996
1997
Year
f)
Lepus americanus
T1-3B
Tracks/100 m
1.60
1.40
1.20
1.00
0.80
0.60
0.40
0.20
0.00
1.48
0.26
0.15
0.07
1994
1995
1996
1997
Year

Figure 21

Figure 22: a) Winter track frequency (total tracks recorded/number of days surveyed) of *Lepus americanus* on T1-1A from 1994 to 1997.

b) Winter track frequency (total tracks recorded/number of days surveyed) of *Lepus americanus* on T1-1B from 1994 to 1997.

c) Winter track frequency (total tracks recorded/number of days surveyed) of *Lepus americanus* on T1-2A from 1994 to 1997.

Note: values are means calculated from all transects surveyed on each site.

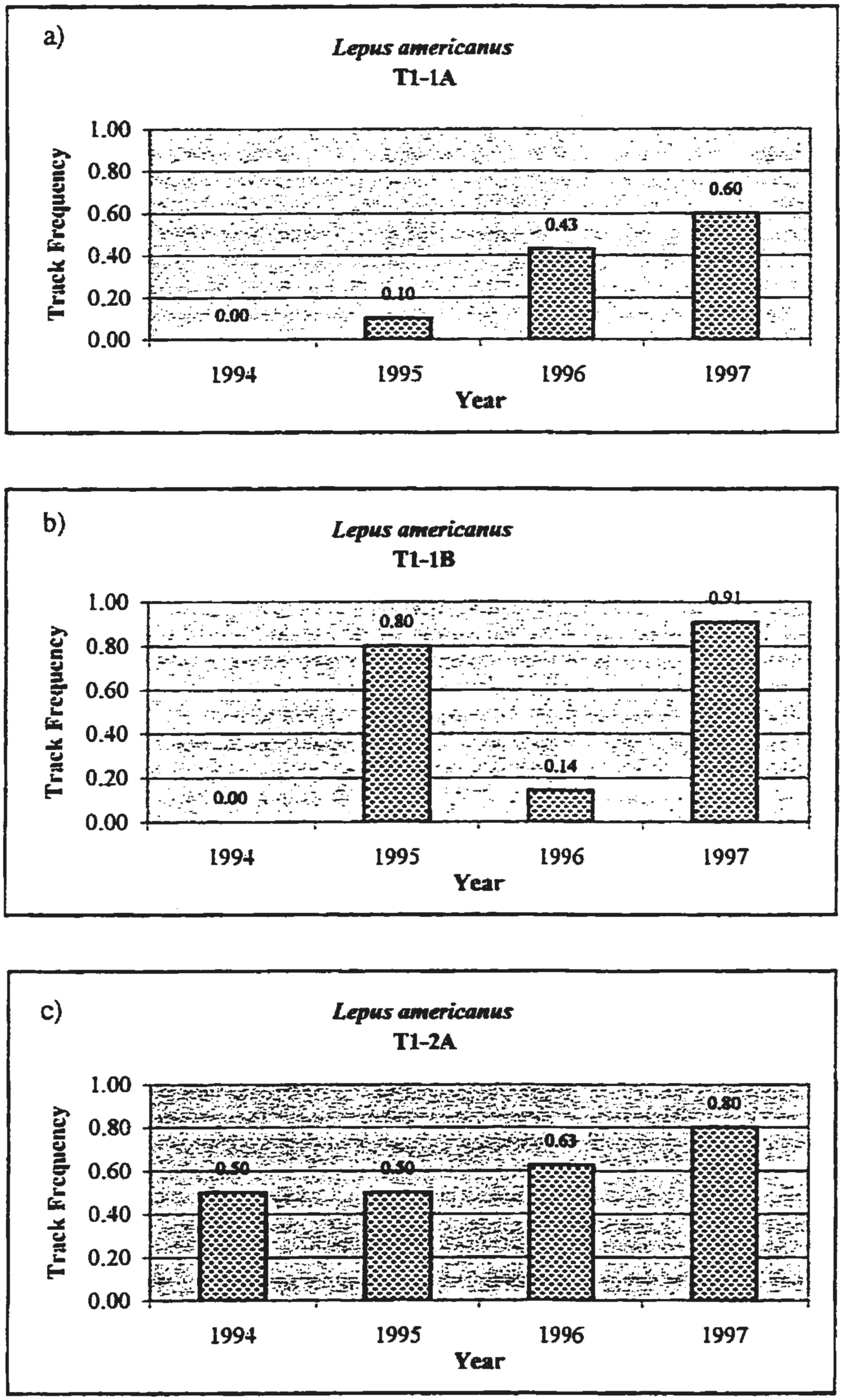

a)
Lepus americanus
T1-1A
Track Frequency
1.00
0.80
0.60
0.40
0.20
0.00
0.00
0.10
0.43
0.60
1994
1995
1996
1997
Year
b)
Lepus americanus
T1-1B
Track Frequency
1.00
0.80
0.60
0.40
0.20
0.00
0.00
0.80
0.14
0.91
1994
1995
1996
1997
Year
c)
Lepus americanus
T1-2A
Track Frequency
1.00
0.80
0.60
0.40
0.20
0.00
0.50
0.50
0.63
0.80
1994
1995
1996
1997
Year

Figure 22

Figure 22: d) Winter track frequency (total tracks recorded/number of days surveyed) of *Lepus americanus* on T1-2B from 1994 to 1997.

e) Winter track frequency (total tracks recorded/number of days surveyed) of *Lepus americanus* on T1-3A from 1994 to 1997.

f) Winter track frequency (total tracks recorded/number of days surveyed) of *Lepus americanus* on T1-3B from 1994 to 1997.

Note: values are means calculated from all transects surveyed on each site.

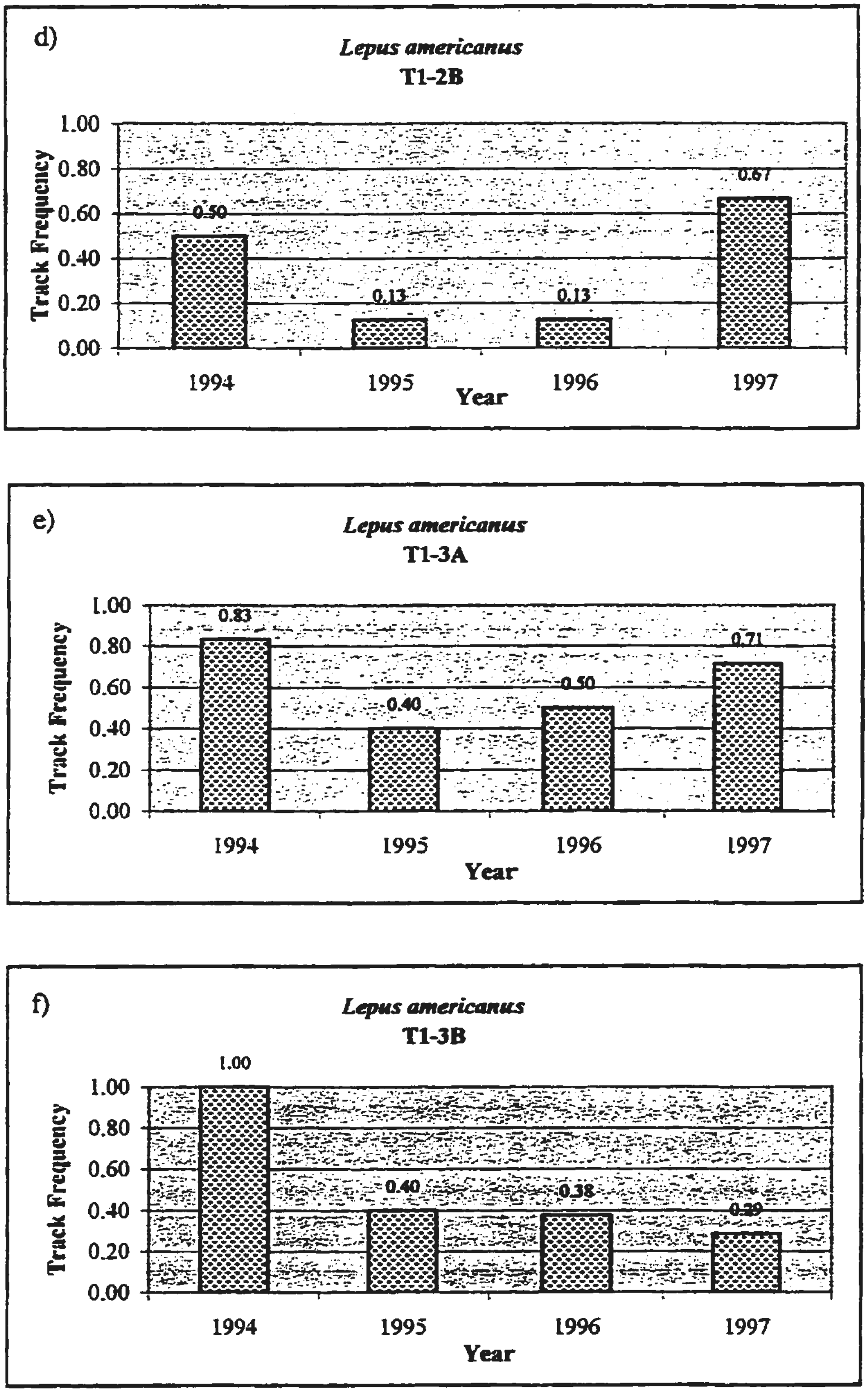

d)
Lepus americanus
T1-2B
Track Frequency
1.00
0.80
0.60
0.40
0.20
0.00
0.50
0.13
0.13
0.67
1994
1995
1996
1997
Year
e)
Lepus americanus
T1-3A
Track Frequency
1.00
0.80
0.60
0.40
0.20
0.00
0.83
0.40
0.50
0.71
1994
1995
1996
1997
Year
f)
Lepus americanus
T1-3B
Track Frequency
1.00
0.80
0.60
0.40
0.20
0.00
1.00
0.40
0.38
0.29
1994
1995
1996
1997
Year

Figure 22

T1-3A in 1994, and tracks occurred only on T1-3B in 1995 (Figures 23a - f). T1-3B was the only site to have tracks present in all years (Figure 23f). The 1997 post-harvest data showed no significant relationships when analysed separately. On T1-2A and T1-2B, all of the tracks occurred within the stream and lakeside buffers (Table 11). Squirrel track frequency was variable in each site from 1994 to 1997 (Figures 24a - f), with T1-3B having the highest frequencies (Figure 24f).

Other Small Mammals:

Microtus pennsylvanicus, *Peromyscus maniculatus* and *Sorex cinereus* tracks only occurred on the transects in 1996, prior to forest harvesting. A single set of vole tracks occurred near the streamside on T1-3A, while deer mice tracks were recorded on all sites except T1-3A and B. Shrew tracks occurred on every site except for T1-3B.

Figure 23: a) Winter track abundance (tracks/100 m) of *Tamiasciurus hudsonicus* on T1-1A from 1994 to 1997.

b) Winter track abundance (tracks/100 m) of *Tamiasciurus hudsonicus* on T1-1B from 1994 to 1997.

c) Winter track abundance (tracks/100 m) of *Tamiasciurus hudsonicus* on T1-2A from 1994 to 1997.

Note: values are means calculated from all transects and days surveyed for each site.

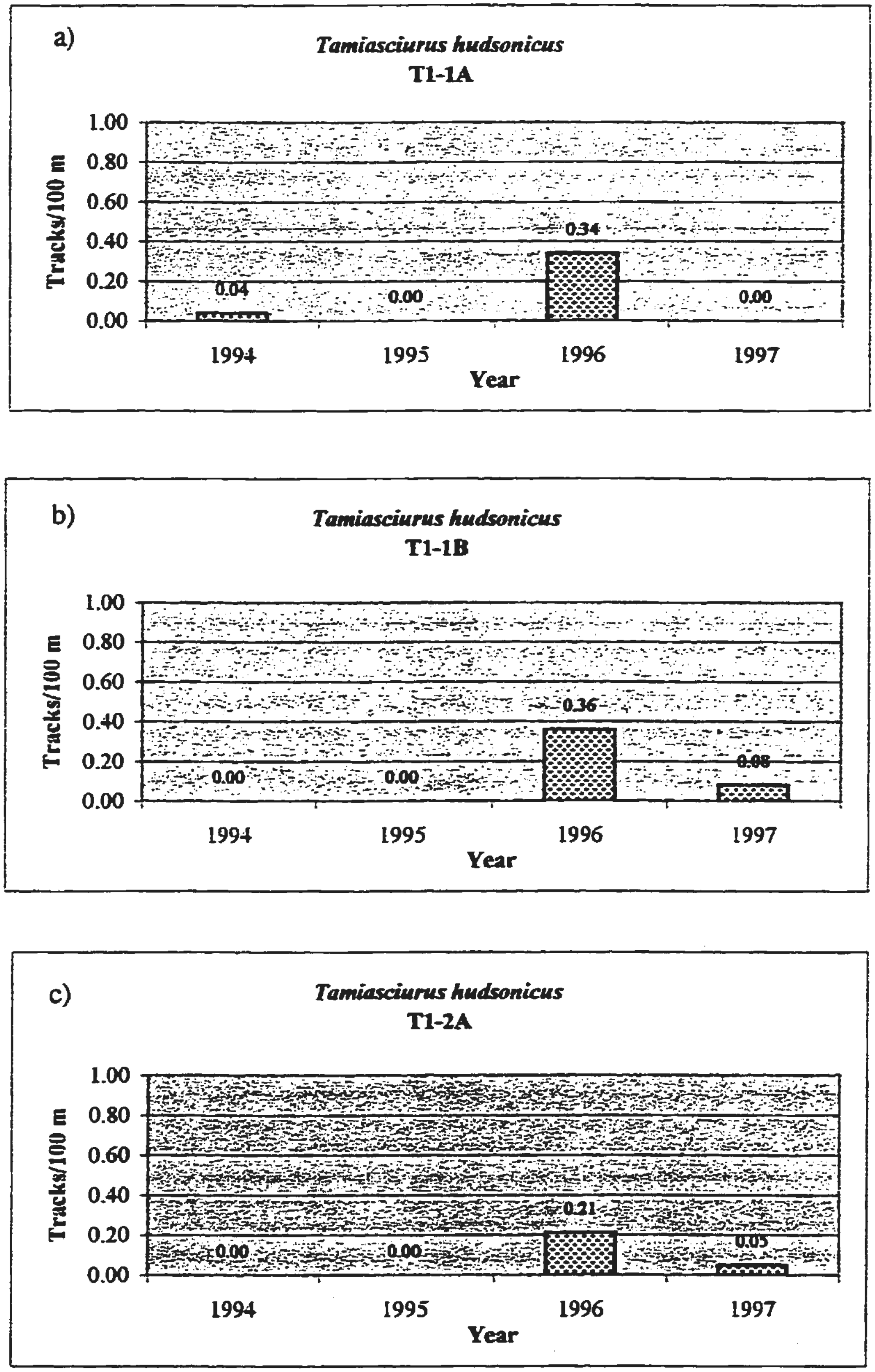

a)
Tamiasciurus hudsonicus
T1-1A
Tracks/100 m
1.00
0.80
0.60
0.40
0.20
0.00
0.04
0.00
0.34
0.00
1994
1995
1996
1997
Year
b)
Tamiasciurus hudsonicus
T1-1B
Tracks/100 m
1.00
0.80
0.60
0.40
0.20
0.00
0.00
0.00
0.36
0.08
1994
1995
1996
1997
Year
c)
Tamiasciurus hudsonicus
T1-2A
Tracks/100 m
1.00
0.80
0.60
0.40
0.20
0.00
0.00
0.00
0.21
0.05
1994
1995
1996
1997
Year

Figure 23

Figure 23: d) Winter track abundance (tracks/100 m) of *Tamiasciurus hudsonicus* on T1-2B from 1994 to 1997.

e) Winter track abundance (tracks/100 m) of *Tamiasciurus hudsonicus* on T1-3A from 1994 to 1997.

f) Winter track abundance (tracks/100 m) of *Tamiasciurus hudsonicus* on T1-3B from 1994 to 1997.

Note: values are means calculated from all transects and days surveyed for each site.

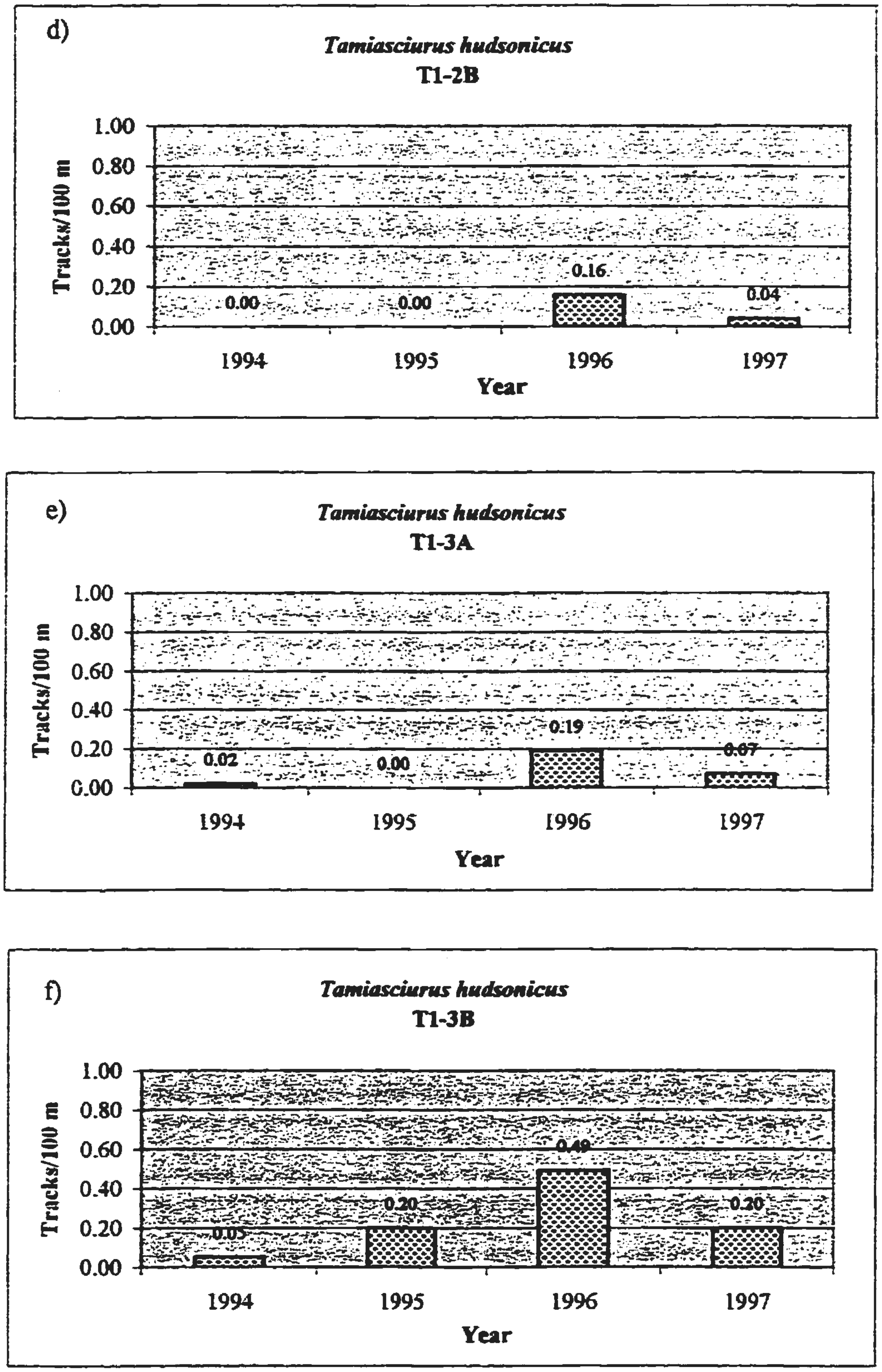

d)
Tamiasciurus hudsonicus
T1-2B
Tracks/100 m
1.00
0.80
0.60
0.40
0.20
0.00
0.00
0.00
0.16
0.04
1994
1995
1996
1997
Year
e)
Tamiasciurus hudsonicus
T1-3A
Tracks/100 m
1.00
0.80
0.60
0.40
0.20
0.00
0.02
0.00
0.19
0.07
1994
1995
1996
1997
Year
f)
Tamiasciurus hudsonicus
T1-3B
Tracks/100 m
1.00
0.80
0.60
0.40
0.20
0.00
0.05
0.20
0.49
0.20
1994
1995
1996
1997
Year

Figure 23

Figure 24: a) Winter track frequency (total tracks recorded/number of days surveyed) of *Tamiasciurus hudsonicus* on T1-1A from 1994 to 1997.

b) Winter track frequency (total tracks recorded/number of days surveyed) of *Tamiasciurus hudsonicus* on T1-1B from 1994 to 1997.

c) Winter track frequency (total tracks recorded/number of days surveyed) of *Tamiasciurus hudsonicus* on T1-2A from 1994 to 1997.

Note: values are means calculated from all transects surveyed on each site.

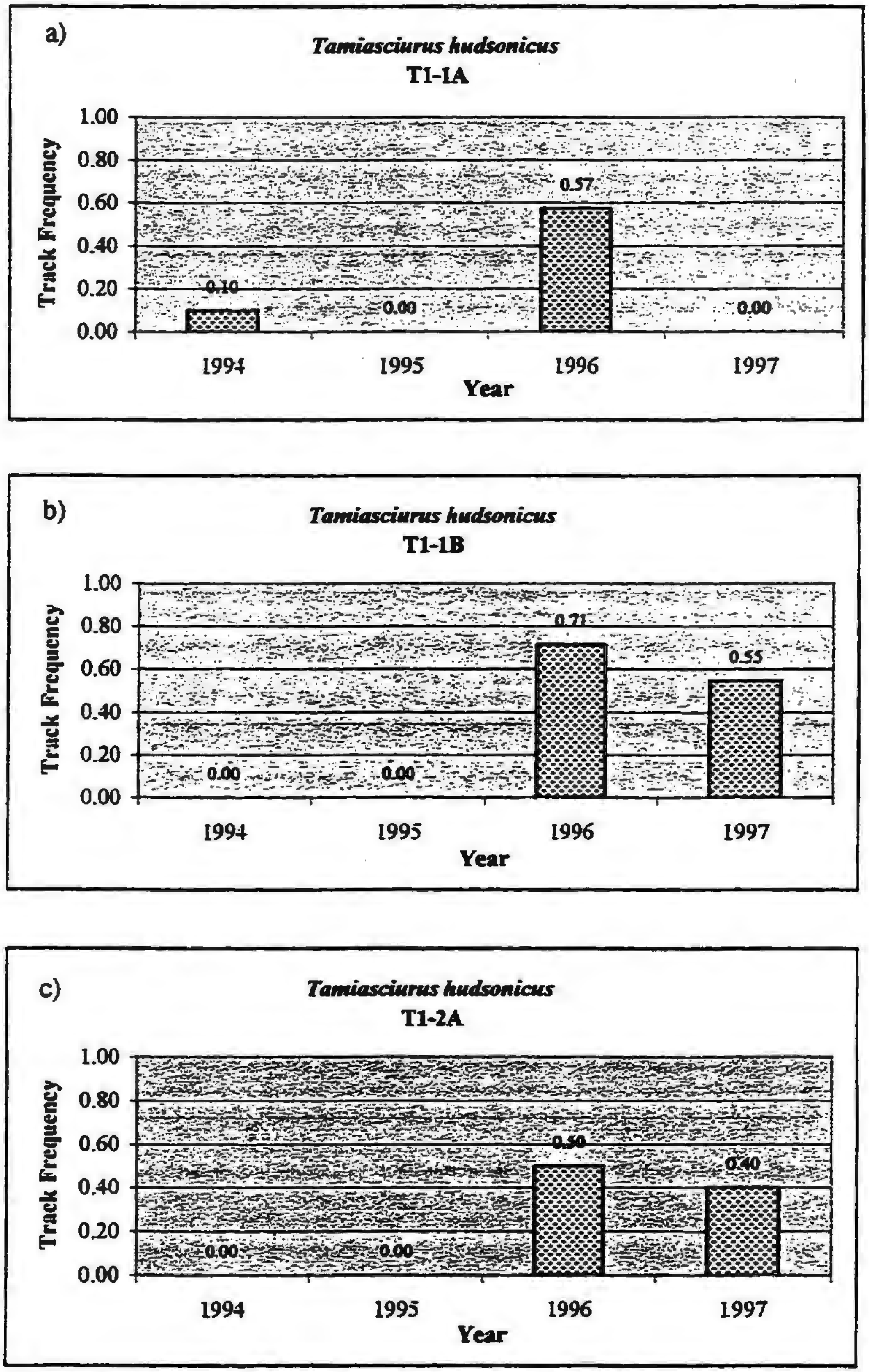

a)
Tamiasciurus hudsonicus
T1-1A
Track Frequency
1.00
0.80
0.60
0.40
0.20
0.00
0.10
0.00
0.57
0.00
1994
1995
1996
1997
Year
b)
Tamiasciurus hudsonicus
T1-1B
Track Frequency
1.00
0.80
0.60
0.40
0.20
0.00
0.00
0.00
0.71
0.55
1994
1995
1996
1997
Year
c)
Tamiasciurus hudsonicus
T1-2A
Track Frequency
1.00
0.80
0.60
0.40
0.20
0.00
0.00
0.00
0.50
0.40
1994
1995
1996
1997
Year

Figure 24

Figure 24: d) Winter track frequency (total tracks recorded/number of days surveyed) of *Tamiasciurus hudsonicus* on T1-2B from 1994 to 1997.

e) Winter track frequency (total tracks recorded/number of days surveyed) of *Tamiasciurus hudsonicus* on T1-3A from 1994 to 1997.

f) Winter track frequency (total tracks recorded/number of days surveyed) of *Tamiasciurus hudsonicus* on T1-3B from 1994 to 1997.

Note: values are means calculated from all transects surveyed on each site.

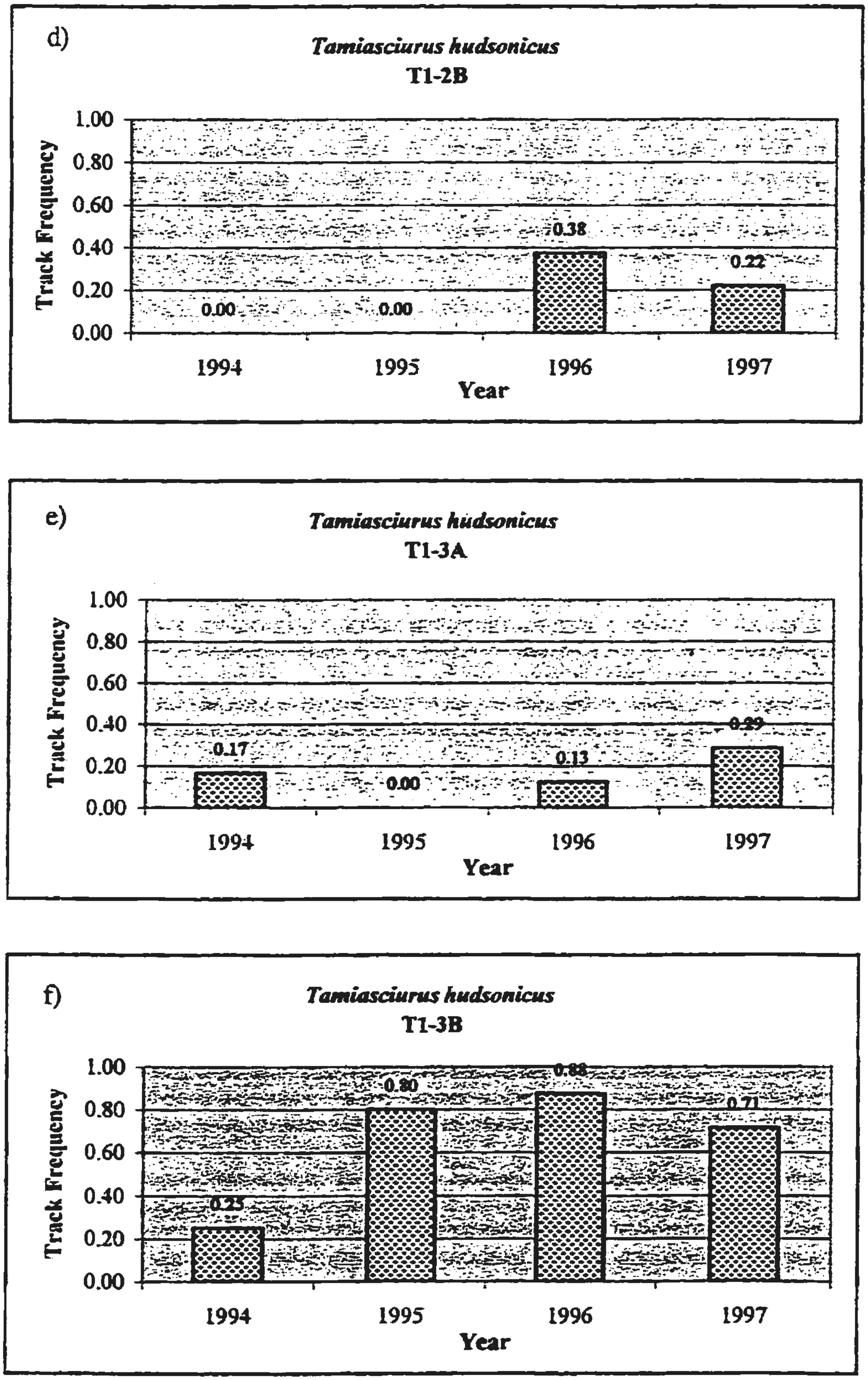

d)
Tamiasciurus hudsonicus
T1-2B
Track Frequency
1.00
0.80
0.60
0.40
0.20
0.00
0.00
0.00
0.38
0.22
1994
1995
1996
1997
Year
e)
Tamiasciurus hudsonicus
T1-3A
Track Frequency
1.00
0.80
0.60
0.40
0.20
0.00
0.17
0.00
0.13
0.29
1994
1995
1996
1997
Year
f)
Tamiasciurus hudsonicus
T1-3B
Track Frequency
1.00
0.80
0.60
0.40
0.20
0.00
0.25
0.80
0.88
0.71
1994
1995
1996
1997
Year

Figure 24

Discussion

Vegetation

In terrestrial ecosystems, the composition of plant communities is influenced by physical factors such as elevation, slope, aspect, soil texture and soil chemistry (White and Harrod 1997). Disturbance can also interact with the physical environment to produce heterogeneity in the landscape (White and Harrod 1997). The composition of riparian plant communities is affected by both fluvial (e.g., flooding) and nonfluvial disturbances (e.g., fire, wind, insect outbreaks) (Gregory *et al.* 1991). In addition, valley floors vary greatly in soil properties and topography (Gregory *et al.* 1991) which can result in considerable differences in riparian vegetation both within and among sites (Bendix 1994).

Results from the Copper Lake sites show that overstory vegetation was similar within and among sites, prior to harvesting (Table 3, Appendix B). Most of the variation in vegetation composition in these sites was observed in the understory (Table 4) and resulted from the interspersion of forest and fen habitats. Low species diversities combined with high species richness in these sites (Table 5), particularly in T1-3B, indicate that many plant species present were rare.

The cluster analysis of the Copper Lake sites by 1995 vegetation characteristics (Figure 11) not only reflects similarities and differences in vegetation types and covers among the sites, it also indirectly reveals similarities and differences in site topography.

For example, the cut-over areas (T1-1C, T1-1D and T1-2D) as a group were most similar to T1-1B, T1-3B and T1-2A. Although this was based on vegetation similarity, each of these sites had a greater slope than T1-1A, T1-3A or T1-2B. T1-1C, T1-1D and T1-2D were steep at the stream edge, while T1-1B, T1-3B and T1-2A had greater slopes 150 m from the stream. T1-1A, T1-3A and T1-2B were flatter and generally wetter. Thus, the vegetation is reflective of the physical conditions present in each site.

Small Mammals

Many North American small mammal species can successfully exploit clear-cut areas because such species have evolved in an environment characterized by periodic ecosystem disturbances, both short-term (e.g., fire, windstorms, floods) and long-term (e.g., effects of the Pleistocene glaciation) (Kirkland 1990).

Factors such as the time and method of sampling (Clough 1987), forest type, variation between sites (Kirkland 1990) and the nature of forest manipulation (Clough 1987) can influence small mammal populations. This may account for the wide variation of reported results on the impacts of forest harvesting practices on small mammals (Clough 1987). In addition, small mammal populations themselves exhibit density-dependent fluctuations, varying reproductive potential and behavioural interactions (Clough 1987).

The results from this study show that small mammal abundance in the Copper Lake area from the summers of 1994 to 1996 was low, with *Sorex cinereus* being the most abundant species. Captures per 100 trapnights, ranging from 0.10 to 10.49 for shrews, 0.10 to 1.35 for *Microtus pennsylvanicus* and 0.10 to 0.21 for *Peromyscus maniculatus*, are similar to those reported from previous studies in insular Newfoundland. Folinsbee *et al.* (1973) recorded a mean abundance of 1.76 meadow voles per 100 trapnights from various locations on the island (Northern Peninsula, central, southwest coast and Avalon Peninsula). Bateman (1986) captured voles and shrews in a variety of coniferous habitats and in open and deciduous habitats in western Newfoundland during the late summer to fall of 1980 and 1981. Abundances ranged from 0.5 to 1.4 voles and 0.3 to 4.5 shrews per 100 trapnights. Tucker (1988) noted a sharp decrease in vole numbers in western Newfoundland from 1986 to 1987. The catch per 100 trapnights went from 1.61 in 1986 to 0.017 in 1987. Shrew captures were reported as mean densities ranging from 15.7 to 51.3 shrews per hectare. Also in western Newfoundland, Thompson and Curran (1995) reported ranges of 0.01 to 0.54 voles, 0.01 to 0.18 shrews and 0.02 to 0.22 deer mice per 100 trapnights from 1990 to 1993, while Sturtevant (1996) obtained vole abundances ranging from 0.04 to 3.06 from 1993 to 1994 and noted that the abundance was higher in 1993.

Sorex cinereus:

Shrews were ubiquitous across all sites and increased in abundance from 1994 to 1995 with numbers remaining stable in 1996. Shrew numbers were very low throughout the trapping sessions in 1994. A smaller effort on the first sites trapped, T1-1A and T1-

1B, was initially considered a possible reason for the small number of shrews captured, yet when more traps were obtained and trapping effort was increased on sites T1-2A, T1-2B, T1-3A and T1-3B there were no corresponding increases in shrew captures. The few captures of shrews on the fen and 5YRCO site near Corner Brook Lake in 1994 reinforce the idea that shrew populations were low during this period.

In 1995, with a trapping effort on all Copper Lake sites equal to that used on T1-3A and B in 1994 (200 traps per night), the relative abundance (number per 100 trap-nights) of shrews on each site was significantly greater than in 1994. Shrews were captured on every site and similar numbers of shrews were captured on the newly cut-over sites, T1-1C, T1-1D and T1-2D. Because these sites were adjacent to T1-1A, T1-1B and T1-2B and had similar types of understory vegetation remaining, the results are comparable with the aforementioned sites. This shows that there was little effect of clear-cutting on shrews in T1-1C, T1-1D and T1-2D.

There were no significant differences in shrew captures per 100 trapnights between 1995 and 1996, or among sites, even on T1-1A, T1-1B, T1-2A and T1-2B where trapping occurred immediately after these sites were harvested in 1996. It is possible that new individuals could have immigrated to these sites with the removal of residents by snap-trapping (Sekgororoane and Dilworth 1995), yet most of these sites were bounded by roads or water, leaving few adjacent areas from which new individuals could move. It is also possible that not enough time had passed between the harvesting and trapping periods in 1996 for any effects of clear-cutting to be detected.

The immediate effects of harvesting on shrew abundance in the Copper Lake area apparently were minimal. However, while shrew abundances were significantly higher in the buffers on T1-2A, T1-2B and T1-2D than in the adjacent clear-cuts, no significant changes in shrew abundance occurred on these sites from 1995 to 1996. Since the buffer area of these trapping grids was much smaller (20 x 150 m) than the clear-cut area (130 x 150 m), results are suggestive of a crowding of shrews into the buffers. In addition, trapping data collected from the Copper Lake sites in the summer of 1997 (Baggs, unpublished data) showed that shrew abundance was lower on the cut-over sites (except on T1-2B) in 1997 than it had been in 1995 or 1996, while the abundance on T1-3A and T1-3B remained stable.

Tucker (1988) found similar results in western Newfoundland. He found no difference in shrew abundance before or immediately after clear-cutting. However, he did find greatest shrew abundance in older clear-cuts (13 and 23 years post-harvest). Martell (1983) found a similar result in Ontario - there was little difference in numbers of masked shrews in uncut and cut stands up to 13 years old. Kirkland (1977) found that shrew numbers increased after clear-cutting of both coniferous and deciduous forests, but that after six to 15 years, post-harvest numbers declined. In spruce plantations in New Brunswick, Parker (1989) found that soricid numbers declined with an increase in plantation age.

Clough (1987) stated that clear-cutting probably affects small mammal populations mainly through its alteration of shrub and ground vegetation rather than through removal of the tree canopy. Harvesting of the Copper Lake sites was patchy, with clumps of vegetation and slash left behind and small strips of forest remaining (Figures 5d - f, 6b and 6d-f). It was in these clumps of vegetation that the shrew captures were more frequent. Thus, there was adequate cover remaining on the cut-overs for the shrews.

The presence of shrews on every site each summer, despite some differences in vegetation among sites (Figure 9), shows that the shrews had no particular habitat preference. This is further supported by the lack of correlation between shrew abundance and vegetation type. Getz (1961) found shrews in abundance in a variety of habitats: areas lacking cover, in forested areas, grassy areas and various intermediate types. He cited moisture as the more important factor affecting shrew distribution and that the major role of cover for shrews is its influence on humidity. Bateman (1986) captured shrews in different habitat types but more often in coniferous forests with high canopy density than in open areas and stated this may have been due to increased moisture levels in the forest. In the present study, shrews, although ubiquitous, were rarely captured on stations in areas inundated with water (i.e., portions of the floodplain).

Microtus pennsylvanicus:

Meadow vole captures were rare over all three summers, although there were more captures in 1996. Voles were most likely at a population low, at least from 1994 to 1995. Similar low captures of voles in western Newfoundland were reported by Thompson and Curran (1995) for 1990 to 1993 and by Sturtevant (1996) for 1994. There was a slight increase in vole abundance on T1-1C, T1-1D and T1-2D from 1995 to 1996 and an increase or no change in abundance on T1-1A, T1-1B and T1-2B immediately after harvesting. Further trapping studies in these areas in 1997 (Baggs, unpublished data) suggested that the vole populations were still increasing; for example, the cut-over sites had more vole captures in 1997 than in 1995 or 1996. While vole populations may have been experiencing a natural population increase, the fact that most of these increases occurred after clear-cutting may indicate an initial positive effect of harvesting.

No effect of a buffer zone could be detected as too few voles were captured on the buffer sites (none on T1-2A in 1996, one on T1-2B in 1996, none in 1995 and six in 1996 on T1-2D). Yet, of the six voles captured on T1-2D in 1996, five were caught on the clear-cut.

There were no voles caught on the 5YRCO site near Corner Brook Lake during this study. This area may not have previously supported a vole population, but it is also possible that the successional changes which occurred since harvesting created an unsuitable habitat for voles. Additional trapping studies in this area in 1997 resulted in the capture of one vole (Baggs, unpublished data).

Tucker's (1988) study of the Grand Lake - Little Grand Lake regions of western Newfoundland showed low numbers of voles. He stated, however, that in the fall of 1986, three weeks after cutting, average numbers of voles captured were almost twice as high in the cut as in the uncut areas. In contrast, Thompson and Curran (1995) captured voles only in the uncut forest, and Sturtevant (1996) captured significantly more voles in over-mature locations than all other forest age classes.

Studies elsewhere also show varying results. Kirkland (1977) found that microtine abundance increased after the clear-cutting of northern Appalachian coniferous and deciduous forests, but that the numbers declined six to 15 years post-harvest. This may have been a result of the initial increase in herbaceous vegetation after clear-cutting and its subsequent loss as succession proceeded to the sapling-young pole stage (Kirkland 1977). In Ontario, Martell and Radvanyi (1977) captured voles only on clear-cuts that were at least one year old and stated that voles will possibly appear on clear-cuts once sufficient cover has been established.

I found that vole numbers were significantly correlated with the amount of grass cover present. This was most noticeable in site T1-1B which had the highest vole captures. The voles on this site were usually captured in the same part of the grid - an area that was a mixture of open woods and wet, grassy grounds. This was also the case with captures on the other sites as voles were typically recovered from the same stations each year. These results are consistent with findings from other studies that the meadow vole is normally associated with herbaceous habitats (Kirkland 1977, Kirkland and Schmidt 1982) and is more abundant on clear-cuts where favourable patches of wet, grassy habitat are created (Ramirez and Hornocker 1981, Swan *et al.* 1984, Probst and Rakstad 1987). In direct contrast to these findings, I captured no voles in either year on T1-3B, which had a high percentage of grassy, fen habitat or on the fen site near Corner Brook Lake. The environmental conditions at T1-3B may explain the lack of vole captures in an area that would otherwise seem an ideal habitat. This site remained in the shadow of the surrounding mountains for the most part of the day and was cooler than the other Copper Lake sites. It was also the last site to lose snow cover in the spring.

Peromyscus maniculatus:

Deer mice are widely distributed in North America but are a recent introduction to insular Newfoundland, with the first specimen captured on the southwest coast in 1968 (Gould and Pruitt 1969). They mainly inhabit woodlands and brushlands, with grains, seeds, fruits and insects making up most of their diet (Baker 1968). In western Newfoundland, Tucker *et al.* (1988) captured 44 deer mice with most occurring on well-drained forest sites, several taken from a 13 year-old regenerating cut-over and single specimens from a grassy habitat and forest edge/alder bed.

The abundance of deer mice was extremely low in the Copper Lake area during this study, with specimens captured only in 1996 on sites harvested immediately before trapping or on sites harvested the previous year. They were caught in small numbers on the 5YRCO site in both 1995 and 1996. These results agree with findings of other researchers in western Newfoundland (e.g., Bateman 1986, Tucker 1988, Thompson and Curran 1995) that deer mice are rare on the island. *Peromyscus* spp. populations in general do not reach very high densities and tend to have more stable annual population levels than other species of small rodents (Terman 1968, Batzli 1992).

Further trapping studies in the Copper Lake sites, fen and 5YRCO site in 1997 showed that deer mice populations were still at the same levels, while showing slight increases in some sites (Baggs, unpublished data). No deer mice have been captured on the uncut sites, T1-3A and T1-3B, to date. There were too few deer mice captured on T1-2D in 1995 and 1996 and on T1-2A and T1-2B in 1996 to determine any effect of a buffer zone. The few mice captured on these sites were generally taken from slash rows left on the plot edges. The lack of deer mice captures on uncut sites with the slight increases on the clear-cut sites may indicate an initial positive effect of harvesting.

In North American coniferous forests, deer mice are found to increase after harvesting (Martell 1983, Kirkland 1990) and they are frequently the first species to colonize recently disturbed areas (Wywialowski 1987). Martell (1983) stated that in coniferous forests, where deer mice are a rare or uncommon component of the small mammal community, timber harvest changes the environment to one that is more suitable for deer mice than in the uncut forest. Deer mice invade clear-cuttings in those conifer stands and increase in number. This may be related to the size and density of the seed crop brought to the ground during clear-cutting (Kirkland 1990).

Winter Tracks

The method of track counting as an index to animal abundance has been used in many studies (e.g., Conroy *et al.* 1979, Monthey 1986, Milton and Towers 1989, Thompson *et al.* 1989, Cameron 1997). Smith (1990) stated that with this type of estimation all data are relative and must be compared with data from other areas or times.

Although they do not provide absolute numbers, such data can indicate trends of populations from year to year and habitat to habitat (Smith 1990). For snowshoe hare in Maine, Monthey (1986) made the assumption that occurrence of tracks in the winter indicated the extent of activity in and preference for harvested or undisturbed forests and the respective cover types. Thompson *et al.* (1989) found that track counts for marten, snowshoe hare and red squirrels were correlated with live-trapping data suggesting that track counts correctly described population changes for these species.

An abundance or lack of animal tracks in a particular area may not necessarily reflect habitat quality. Van Horne (1983) stated that there may be yearly variation in population numbers that reflects variability in food items, predator populations or abiotic environmental factors. Thus, densities may reflect conditions in the recent past or temporary present, rather than long-term habitat quality. This could result in variability in the occurrence of winter tracks. For example, the severity of the winter, including snow depths and hardness, will affect the number and kinds of animal tracks seen. It is also important to consider the behaviours of each mammal species, e.g., territoriality, subniveal activity and reduced winter activity because these will affect the number of tracks observed. It is acknowledged that track groups observed on the winter transects may represent the activity of an individual or several animals and therefore the results can only be used as an index.

Pooled winter track data for the most commonly observed mammal species (pine marten, weasel, red fox, red squirrel and snowshoe hare) on the Copper Lake transects showed that most activity occurred away from the stream edge towards the interior of the forest. The lowest amount of mammal activity occurred at the stream edge. This was also the case for the 1997 data when looking at all sites combined, but does not reflect what occurred on the individual sites after forest harvesting had occurred in 1996.

The grouped activity on T1-3A and T1-3B was greater away from the streamside except for the transects farthest from the streams. On T1-3A this may be due in part to the fact that the northernmost transect was fairly close to the Copper Lake logging road (Figure 1), so that disturbance could be playing a role here. On T1-3B, the transect farthest away from the stream was on a very steep incline and not likely to have much animal activity. The stream edges of these sites were mostly fens and did not provide much cover in the winter months.

For T1-2A and T1-2B, different results can be explained by considering how these sites were harvested. On T1-2A the greatest number of tracks occurred within the riparian buffer and decreased with distance from the stream. Apart from the stream and lake buffers left on T1-2A, little cover remained. There was, however, little difference in track counts among the four transects on T1-2B. This could be the result of this site having 20 m buffer zones left on the stream edge, lakeshore and on the back edge of the plot (buffer zone on the lower section of T1-3). Most tracks found on T1-2B were within a buffer.

On T1-1A and T1-1B, some mammal tracks were observed in clearcuts, however, none were observed on streamside transects. These transects had the least amount of residual cover due mainly to skidder damage (Figure 5d). The large number of tracks found on the fourth and fifth transects of T1-1B reflects the patchy harvesting that resulted in much residual cover on these transects.

Because of the patchiness of these cuts and the presence of fens in some sites, the winter track transects were grouped as either open (very little cover) or forested. As a group, the pine marten, weasel, red fox, snowshoe hare and red squirrel showed significantly more activity on the forested transects than on the open ones. Pine marten, red squirrel and snowshoe hare showed a particular preference for cover. Milton and Towers (1989) had similar findings on their winter track transects in the St. Mary's River area of Nova Scotia. The total wildlife use was higher in uncut than cut areas.

Small Mammalian Predator Species:

Martes americana:

The preference of pine marten for forested areas is not an unexpected result. The pine marten is known to prefer old-growth forest habitat, particularly coniferous forests (Soutiere 1979, Steventon and Major 1982, Taylor and Abrey 1982, Spencer *et al.* 1983, Thompson 1988, Thompson *et al.* 1989, Forsey *et al.* 1995, Sturtevant *et al.* 1996). Marten are also known to avoid clear-cut and open areas, especially in winter (Soutiere 1979, Steventon and Major 1982, Spencer *et al.* 1983). In insular Newfoundland, marten have an even closer tie to old-growth forests than in other parts of North America as a result of the limited prey base on the island (Bissonette *et al.* 1997).

In the Copper Lake area, winter habitat usage by pine marten declined on sites T1-1A and T1-1B from 1994 to 1997, although harvesting of these areas had not occurred until the summer of 1996. This may be related to the progressive amount of disturbance occurring at these sites during this time. Marten activity was greatest in 1994 before road access to these sites was available. Roads fragment habitat connectivity by acting as barriers for many species and may be the most destructive aspect of habitat fragmentation (Noss 1993). By 1995, the logging road had passed near T1-1A and through the plot on T1-1B (Figure 1). Also by this time the adjacent areas, T1-1C and T1-1D had been clear-cut, increasing the amount of disturbance in the general area of T1-1. Corresponding to the increase in disturbance, there was a decline in marten activity in this year (Figures 15a and b, 16a and b). Marten activity remained low in 1996, and then dropped to almost zero in 1997 after T1-1A and T1-1B were clear-cut. The only tracks recorded from these sites in 1997 occurred on the northernmost transects of T1-1B, where the forest had not been cut.

Pine marten activity on T1-2A and T1-2B fluctuated over the four winters (Figures 15c and d, 16c and d). Most tracks were observed in 1994 and 1996 (Table 10). Again, there was no road construction in this area until the late summer/early fall of 1994 and by the winter of 1995 the logging road had crossed the northwest edge of T1-2A and passed by T1-2B (Figure 1). The adjacent area on T1-2D had also been clear-cut, so the amount of disturbance was increased in this area. Track frequencies showed a corresponding decrease. In 1996, track frequencies were higher than they had originally been in 1994, but dropped again in 1997 after T1-2A and T1-2B had been harvested to within 20 m of the stream edge. Except for one set of tracks, all marten activity in these areas in 1997 occurred within the riparian buffers.

On T1-3A, marten track frequencies were generally consistent from 1994 to 1997 (Figure 16e) while track abundances were higher in 1994 and 1996 than they were in either 1995 or 1997 (Figure 15e). T1-3B showed a steady increase in both track frequency and abundance from 1994 to 1997 (Figures 16f and 15f, respectively). T1-3A, although left uncut, had some disturbance in the area with the construction of the logging road in the late summer/early fall of 1994 that passed by its northern side (Figure 1). This may correspond to the decrease in marten track abundance in 1995 (i.e, marten remained in the area but the intensity of activity decreased). T1-3B had the least amount of disturbance with no logging roads in its vicinity, yet had low track frequencies and abundances until 1996 and 1997. This may reflect marten being displaced into a previously less used area due to the disturbance at other sites. The increase in track abundance on this site from 1994 to 1997 may indicate an increase in the level of searching activity by marten for prey. It is noteworthy that no meadow voles or deer mice were captured on this site.

Winter track data collected from the Copper Lake sites in 1998 were similar to those of 1997. No marten tracks occurred on T1-1A or T1-1B, except in residual forest; marten tracks on T1-2A and T1-2B occurred within the buffers; and track abundances/frequencies on T1-3A and T1-3B were stable or increasing (Baggs, personal communication).

The lack of marten winter activity on the clear-cuts in the Copper Lake area corresponds with results of other studies from Newfoundland and elsewhere. From observations of winter tracks in western Newfoundland, Snyder (1984) reported that marten seldom used clear-cuts or residual stands less than 15 ha but did use larger residual stands. Snyder and Bissonette (1987) had greater capture rates of marten in residual fir stands (89.5%) in western Newfoundland than in clear-cuttings (10.5%). They reported that in winter, marten tracks frequently occurred in residual stands and undisturbed forest (74%) while fewer were in clear-cuts (25%). Marten travel patterns were also different between the residual stands and clear-cuts. Trails in the residual stands were often in a zig-zag pattern while the clear-cuts trails were fairly straight, moving from one residual stand to another. In another study (Bissonette *et al.* 1988), marten rarely used clear-cuts during the first year after cutting. Use of residual forest patches in clear-cuts up to 23 years old was greater, but still infrequent. In Maine, Soutiere (1979) found lower marten track counts and had less live-trapping success in commercial clear-cuts than in the uncut forest.

Steventon and Major (1982) stated that underuse of clear-cuts may be a result of poor hunting conditions for marten in this habitat. Marten use woody debris in winter as a means to access the subniveal tunnels of their prey. The structure of slash from clear-cutting may be very different from that due to natural tree mortality in an uncut forest (Steventon and Major 1982). Winter thaws in Newfoundland often result in ice layers forming in the snow (Bissonette *et al.* 1988). In addition, increases in snow depths on clear-cuts in winter make ground access more difficult for marten (Bissonette *et al.* 1988). Prey may therefore be abundant on a clear-cut, but not accessible.

The results of the small mammal trapping in this study show that some prey was available to the marten on each site, although the prey items varied with the sites. Bateman (1986) found that the most important winter food items in marten scats from western Newfoundland were snowshoe hares and meadow voles with frequencies of 51.8 and 30.4%, respectively. Shrews had a frequency of 10.7%. Tucker (1988) also examined marten scats from western Newfoundland and found slight variations in diet with season. Voles were the most frequent prey item in all seasons (average of 91.2%), but their lowest frequency occurred in the winter (66.7%). Shrews were the second major prey item with frequencies ranging from 5.7% (summer) to 30.0% (spring). Berries were a major food source during the summer only, while red squirrels were the most frequent in winter scats. Snowshoe hare remains were found only in winter scats. Bird remains had low frequencies but were found in all seasons.

The high abundance of shrews in the current study suggests that these could be a more important prey species when the preferred species are lacking. For example, meadow voles were low in abundance in all sites during this study, yet marten tracks were still observed on all sites before harvesting occurred. Deer mice were also too infrequent to be a major prey item. Snowshoe hare and red squirrel tracks were variable throughout the study, but do indicate that these prey items were available in the study area.

Mustela erminea:

In this study weasel track abundances were significantly higher in 1994 and 1996 than in the other two years (Table 10). This could be related to the disturbance levels in the Copper Lake area prior to the 1995 and 1997 winter seasons. Weasels showed no significant preference for either forested or open habitats (Figure 14), although tracks were more frequent on the open transects. In contrast to this finding, track data from T1-2A and T1-2B in 1997 showed that all weasel tracks occurred within the 20 m stream and lake buffers (Table 11).

Simms (1979) found that weasels in southern Ontario showed a significant preference for early successional communities and avoided forested habitats. Where they did occur within a forested region, weasels used bog-meadows, thickets along streams and disturbed sites. Also in Ontario, Thompson (1988) found somewhat similar results; weasel tracks were more abundant in early successional stands and also in uncut stands but less common in older, regenerating sites. In contrast, Thompson *et al.* (1989) did not observe a preference by weasels for a certain stand age and reported that weasel tracks were the most erratic and highly variable of all animal species recorded. In Nova Scotia, Milton and Towers (1989) found similar weasel track abundances in cut and uncut riparian zones and determined that weasels did not seem affected by the cover variables measured. Track abundances also showed no correlation with riparian buffer width (tested from 3 to 45 m)

(Towers and Milton 1990).

Weasels are vole specialists which may be related to their size, enabling them to enter the snow tunnels and burrows of the voles (Simms 1979). In Nova Scotia, Cameron (1997) found that weasel tracks were positively correlated with crown closure and tree diameter at breast height. He stated that this is not likely due to weasel habitat preference and instead may reflect the weasels occupying the habitats of their prey. Weasel populations can fluctuate drastically in response to their prey populations (Banfield 1987). Thus, a lot of the variability observed in weasel activity could reflect what is happening to their prey. The subniveal behaviour of weasels in winter may be a more likely cause of track variability. Deeper snow means more use of tunnels would be necessary to access prey resulting in less visible activity. Tracks could be more abundant in years with less snowfall, as was the case in 1996.

Vulpes vulpes:

Red fox tracks were variable over the four winters and in several instances no tracks were observed. The greatest track abundance was observed in 1995 (Table 10), in contrast to the high abundances in 1994 and 1996 for the two previously mentioned species. Fox tracks were not significantly related to overstory cover although more tracks were observed on the forested transects (Figure 14). Also, on T1-2A and T1-2B in 1997 all fox tracks, except for one set, occurred within buffers (Table 11). A similar result was found with the 1998 track data from the buffer sites (Baggs, personal communication).

In contrast to the 1997 data, 1998 showed higher abundances of red fox tracks in all sites, with increasing track abundance from the clearcuts to the buffer sites to the uncut sites (Baggs, personal communication).

Ables (1975) stated that red foxes tend to avoid large homogeneous tracts of any single type and instead select areas with the greatest habitat diversity. In Ontario, Thompson (1988) found that red foxes were more abundant on older, regenerating stands and states that in the boreal forest red foxes prefer intermediate-aged successional habitats to uncut mature forest. Generally the numbers of fox tracks were higher in logged areas than in uncut sites (Thompson 1989).

Red foxes are opportunistic feeders, using whatever is most readily available (Ables 1975, Banfield 1987). Diet varies seasonally, with berries and invertebrates more commonly consumed in summer, and meat in winter (Banfield 1987). Small mammals are a major source of prey, with microtines being preferred to deer mice or shrews (Ables 1975, Banfield 1987). Foxes hunt small mammals in the winter by digging them out of their tunnels (Banfield 1987). In Newfoundland, a major food source is the snowshoe hare (Dodds 1955). The concentration of fox tracks in the buffers on T1-2A and T1-2B may therefore parallel the higher abundances of shrews and higher track abundances of snowshoe hares in the buffers.

Mustela vison:

Mink tracks were seldom encountered but when found were mostly at the stream edges. This is not unexpected owing to their typical habitats, such as stream banks, along lakes and marshes, especially if forested (Northcott 1980) and that, in addition to small mammals, fish form a major part of their diet (Banfield 1987). No mink tracks occurred on T1-1A or T1-1B after clear-cutting (Table 9) and on T1-2A and T1-2B the majority of tracks were within the forested buffers (Table 11). Clear-cutting may have had a negative effect on the mink. There appears to be limited information on mink winter track occurrences in similar studies. In Nova Scotia, Cameron (1997) found mink tracks in forested corridors but not in adjacent cut habitat.

Small Mammalian Prey Species:

Lepus americanus:

Snowshoe hare activity was variable by site and by year, with 1994 and 1997 having significantly higher track numbers than the other two years (Table 10). Thus, track abundance may have fluctuated independently of the disturbances from road construction and forest harvesting in the Copper Lake area. As a further example of this, hare tracks increased in abundance on T1-1A from 1994 to 1997 (Figure 21a). No tracks were observed here in 1994 and the highest abundance occurred after clear-cutting while on T1-1B numbers were up and down over the four winters (Figure 21b) and after clear-cutting the only hare tracks observed were in the residual forest left above the logging road. On T1-2A and T1-2B all of the hare tracks in 1997 were within the buffers (Table 11), yet on T1-3B, the most isolated site, track abundances declined from 1994 to 1997

(Figure 21f).

Researchers report that a dense woody understory is a common physical feature in habitats that support hare populations (Buehler and Keith 1982, Keith 1990, Towers and Milton 1990, Cameron 1997). This type of cover is most extensive on burned areas, clear-cuts and some lowlands (Keith 1990). Conroy *et al.* (1979) also reported heaviest snowshoe hare activity in areas with much habitat interspersion, especially around clear-cut edges with dense understories. Browse species appeared to shift to more palatable and nutritious species on clear-cuts (Conroy *et al.* 1979)

In western Newfoundland, Tucker (1988) counted winter hare trails before and after forest harvesting occurred. He reported that only nine trails were seen over a period of 81 days, while after harvesting, 53 trails were seen in 65 days. This indicated a slight increase in snowshoe hare after harvesting. In Maine, Monthey (1986) found snowshoe hare activity greater than expected in a commercially clear-cut forest and less than expected in undisturbed, partially harvested forests. Activity was greater in 12- to 15-year-old clear-cuts than in younger stages. Thompson (1988) and Thompson *et al.* (1989) found similar results in Ontario. The most activity occurred on clear-cuts approximately 20 years old, while the lowest activity was recorded immediately after cutting.

Tamiasciurus hudsonicus:

Squirrel tracks were variable throughout the study and had their highest occurrences in 1996 (Table 10). Red squirrels tend to limit their outside activity in cold weather to the warmest part of the day and can remain inactive in their burrows for several days during harsh winter weather (Banfield 1987). Thus, small occurrences of squirrel tracks in winter would not be unexpected. Also, red squirrels are preyed upon by a variety of predators such as marten, weasels, foxes and some birds. Low numbers of squirrel tracks on the Copper Lake transects could possibly reflect heavy predation, since all of these predators were present each winter.

The finding that red squirrel tracks were significantly higher on forested transects (Figure 14) is expected (Northcott 1980, Banfield 1987). Red squirrels are so dependent on coniferous trees that in some parts of their range the population density changes in response to the amount of conifer cones produced (Flyger and Gates 1982). Lack of cone production in older, scenescent forests could also affect the number of winter tracks seen in such areas.

Track abundances from 1997 were low on all sites (Figures 23a - f), but no tracks occurred on T1-1A, and the few tracks that occurred on T1-1B were actually above the logging road, in the residual forest left from the clear-cutting. The tracks observed on T1-2A and T1-2B all occurred within the riparian buffers (Table 11), again suggesting a preference of red squirrels for forested areas.

Similar findings by Thompson (1988) and Thompson *et al.* (1989) in Ontario showed that red squirrels were more abundant in uncut forest than in regenerating stands and significantly fewer tracks occurred in stands less than five years old compared with other stand age groups. In Nova Scotia, both Towers and Milton (1990) and Cameron

(1997) found that red squirrel tracks were positively correlated with canopy closure. Cameron (1997) found red squirrel tracks in forested corridors but not in adjacent cut habitat.

Summary

Effects of Clear-cutting - No Buffer

On the sites at T1-1, small mammal abundance was initially low, but increased slightly over the course of the study. In the first one to two years after clear-cutting, abundances remained stable (masked shrews) or slightly increased (meadow voles, deer mice). There appears to have been no major immediate effect of clear-cutting on small mammal populations at T1-1. Examination of winter tracks of other prey species reveals that snowshoe hare tracks increased in abundance on T1-1A over time and were highest after clear-cutting, while red squirrels only occurred in residual patches of forest on T1-1B. Thus, despite the lack of red squirrel activity on clear-cuts in winter, there were at least four mammal prey species available.

Red fox and weasel tracks were always low in abundance and appeared to show no effects of clear-cutting. This was not the case with pine marten tracks; they were more abundant in years prior to harvesting, but were absent after clear-cutting. The only marten tracks observed in this area after harvesting were in the residual forest patches left on T1-1B. Even though mammal prey species were present on the clear-cuts the pine marten was not using these areas.

Effects of Clear-cutting - 20 m Buffer

Small mammal abundances in the sites associated with T1-2 showed no apparent effects of harvesting. But when abundances were compared between the 20 m buffer strip and the remaining clear-cut I found that shrew abundances were significantly higher in the buffers. Winter tracks of other mammal prey species, viz., snowshoe hare and red squirrel, indicated that these animals were only using the buffers and not the adjacent clear-cuts. The predator species, pine marten, weasel and red fox, apparently also limited their winter activity to the stream and lakeshore buffers. The buffers may have been used as corridors or as actual habitat.

Noss (1993) stated that a potential function of corridors in dynamic landscapes is to allow animals to escape disturbances. This could be the case with the buffers left on T1-2, particularly since mammal winter tracks were generally more abundant away from the stream prior to harvesting. Forcing animals from the interior forest into a narrow 20 m buffer may have some negative consequences. Narrow corridors are entirely edge habitat and can be expected to produce high rates of mortality for sensitive species (Noss 1993). Competition and predation can be increased through bringing together species which normally have little contact (Morrison *et al.* 1992). In insular Newfoundland, increased competition and predation in a narrow riparian buffer may have a more pronounced effect due to the limited prey base shared by the various mammal predators.

An important point to consider is that T1-2 is a narrow, intermittent stream. Similarities in winter track abundances and frequencies between T1-2A and B and directions of track paths over the four winters suggests that animals are crossing the stream. It is possible that the combination of two 20 m buffers and a narrow stream could be creating the effect of a larger buffer. Therefore, if the stream were wider, would mammal activity still be high in the buffers? Also the possibility of blowdown at the edge of the buffer could reduce the effectiveness of the buffer as a corridor or as habitat over time (See Figure 6e for an example of blowdown on the cut edge of a buffer, shortly after harvesting occurred.).

Effects of No Cutting (effective buffer of at least 100 m)

Small mammal abundance was low for both T1-3A and B. Although shrew abundance was greater in 1995 and 1996 than in 1994 on both sites, meadow voles were only captured on T1-3A and no deer mice were ever captured in this area. Winter track data indicated that while snowshoe hare activity was constant on T1-3A, it was highest on T1-3B in 1994 and declined thereafter. Red squirrel track abundances and frequencies were generally higher in T1-3B.

Weasel and red fox tracks were variable, yet red fox track frequencies declined on both T1-3A and B over the four winters. In contrast, pine marten activity remained constant on T1-3A and noticeably increased on T1-3B. The marten may have shifted their winter activity to an area with less disturbance and possibly less competition, even though prey may be less abundant.

Riparian Zone Usage by Small Mammals

In Newfoundland there is a low prey diversity relative to that of the predators, resulting in a high demand for a limited resource base. This study shows that, at least in the winter, mammalian activity was greater in the interior forest than at streamside. The exception to this occurred after clear-cutting. The remaining 20 m riparian buffers demonstrated the highest mammalian activity. Population cycles in conjunction with diminishing suitable habitat can only result in decreases in species diversity over time. A riparian buffer may be enough to allow animals to escape disturbance but may not be enough to maintain animal species adapted to interior forest conditions. Furthermore, results of this study indicate that for a sensitive/endangered species (e.g., Newfoundland pine marten), any disturbance, regardless of scale, can affect habitat utilization.

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Appendices

Appendix A

Table A.1: Checklist of mammal species in the western region of Newfoundland.

Scientific Name	Common Name
Sorex cinereus	masked shrew
Microtus pennsylvanicus	meadow vole
Peromyscus maniculatus	deer mouse
Tamiasciurus hudsonicus	red squirrel
Tamias striatus	eastern striped chipmunk
Castor canadensis	beaver
Lepus americanus	snowshoe hare
Martes americana	pine marten
Mustela erminea	short-tailed weasel
Mustela vison	mink
Lutra canadensis	river otter
Vulpes vulpes	red fox
Canis latrans	coyote
Ursus americanus	black bear
Myotis lucifugus	little brown bat
Alces alces	moose
Rangifer tarandus	woodland caribou

Appendix B

Table B.1: Mean frequencies (percentage ± standard deviation) of tree species for the Copper Lake sites in 1995.

Site	*Abies balsamea*	*Picea mariana*	*Picea glauca*	*Larix laricina*	*Betula papyrifera*
T1-1A	36.16 ± 16.63	63.33 ± 16.67	0	0	0.51 ± 1.35
T1-1B	60.21 ± 15.59	32.06 ± 17.75	0	0	7.73 ± 7.73
T1-1C	50.98 ± 16.91	24.51 ± 33.36	0	1.96 ± 4.80	16.67 ± 7.82
T1-1D	48.04 ± 21.18	26.47 ± 15.67	0	0	8.82 ± 8.11
T1-2A	70.50 ± 7.20	24.90 ± 6.65	0	0	4.60 ± 2.54
T1-2B	67.13 ± 6.75	20.14 ± 7.81	0	0	10.87 ± 6.22
T1-2D	59.80 ± 18.76	34.31 ± 21.51	0.98 ± 2.40	0	5.88 ± 7.44
T1-3A	50.82 ± 4.41	36.43 ± 12.03	5.10 ± 6.42	0	7.66 ± 6.32
T1-3B	58.16 ± 8.31	30.20 ± 4.80	1.45 ± 2.28	0	10.18 ± 7.84

Appendix C

The following tables are plant species lists and associated Braun-Blanquet cover classes for the plant plots on the Copper Lake sites. The Braun-Blanquet cover classes and abbreviations used in the tables are defined below. The first number/symbol given in the tables represents the cover class of the plant species, while the number after the decimal represents the sociability class.

Braun-Blanquet Cover Classes (modified from Smith 1990):

\+ - individuals are sparsely present

1 - individuals are plentiful, but small coverage (Note: for the purposes of analysis, the + and 1 categories were grouped and considered to have a cover from 0 - 1 %.)

2 - individuals covering between 2 - 5 % of plot area

3 - individuals few or many, collectively covering 6 - 25 % of plot area

4 - individuals few or many, collectively covering 26 - 50 % of plot area

5 - individuals cover 51 - 75 % of plot area

6 - individuals cover 76 - 100 % of plot area

Braun-Blanquet Sociability Classes (Smith 1990)

1 - shoots growing singly

2 - scattered groups or tufts of plants

3 - small, scattered patches or cushions

4 - large patches or broken mats

5 - large mats of stands or nearly pure populations that almost blanket the area

Note: NA - indicates that no information is available for that plot
road - indicates that the logging road passed through the area where the plot would have occurred

Table C.1: Braun-Blaunquet cover and sociability classes for plant species in T1-1A in the summer of 1995.

	Quadrat Number																								
Species	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25
Abies balsamea	+.1		+.1		3.1	+.1	3.1					2.1							+.1		+.1				
Amelanchier spp.			+.1	+.1			+.1	+.1	+.1			+.1	+.1	+.1		+.1					+.1	+.1	+.1	NA	
Aster spp.	+.1	+.1		+.1				1.2	+.1		+.1	1.2	+.2	+.1	+.1									NA	
Betula papyrifera			+.1																			+.1		NA	
Betula spp.																	2.1							NA	
Carex spp.	3.3	2.2	4.3	6.4				5.3	6.5	2.2	3.3		6.4	3.3	1.2	1.3		6.4	1.2		2.2		2.2	NA	1.2
Clintonia borealis	2.2			+.1		+.1	+.1	+.1				1.2			1.2				+.1		+.1		+.2	NA	+.1
Coptis groenlandica					+.1	1.1	1.2	1.2	1.2		+.2	2.2	1.2	+.1	1.2	2.2	+.1	3.2	1.2		1.1	1.2	2.2	NA	2.2
Cornus canadensis	1.2	3.3	1.2	1.2		4.2	4.3			+.2		1.2	+.2	+.1	+.1	1.2	1.2	1.2	1.2	1.2	1.2	1.2	1.2	NA	+.1
Dicranum spp.							4.3										+.1			+.1			4.4	NA	+.1
Drosera sp.								1.1	2.2	+.2	1.2	+.1	1.2	+.2	+.2	+.1								NA	
Dryopteris noveboracensis				+.2																				NA	
Epigaea repens						+.1	+.1										1.1			3.2		+.1	+.1	NA	+.1
Equisetum sylvaticum				+.1						+.2									+.1					NA	
Gaultheria hispidula	2.2					3.3	3.4	+.1	+.1			2.2		1.2	+.1		+.1	1.2	+.1	1.2	1.2	1.2	1.2	NA	+.1
Hepaticae							3.2										+.2						+.1	NA	+.1
Hylocomium spp.			+.1																					NA	
Kalmia angustifolia													+.1									+.1		NA	
Kalmia polifolia													+.1	+.1									+.1	NA	+.1
Ledum groenlandicum													+.1	+.1										NA	
Linnaea borealis	2.2			+.1		+.1	+.1	+.1				1.2			1.2				+.1		+.1		+.2	NA	+.1
Listera cordata						+.1																		NA	
Lycopodium spp.		1.2	+.2						+.2						+.1									NA	
Maianthemum canadense				+.1	1.2	+.1	+.1			+.1				+.1	2.2				1.1		1.2			NA	
Myrica gale								3.2	4.2		2.1	+.1	+.1	+.1	+.1								+.1	NA	
Nemopanthus mucronata															+.1									NA	
Osmunda cinnamomea		3.1	+.1								+.1				+.1				5.1		5.1			NA	
Picea glauca												+.1												NA	
Picea mariana	+.1	6.1		3.1	6.1	2.1		4.1	+.1	+.1	+.1			3.1	6.1	+.1	3.1	3.1	3.1	5.1	+.1	+.1	3.1	NA	+.1
Pleurozium schreberi						5.1	+.1										5.4			5.4		4.4		NA	
Poaceae	2.2	1.2	3.3					4.2	3.3	1.1		1.2		1.2										NA	

Table C.1 continued:

Species	Quadrat Number																								
	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25
Prenanthes trifoliolata										+,1														NA	
Ptilium crista-castrensis						4,4														3,2		+,1		NA	+,1
Ranunculus sp.										+,1														NA	
Rhododendron canadense	+,1						+,1	+,1	+,1		+,1	3,1		+,1	+,1	2,1	3,1	2,1				5,2	3,1	NA	+,1
Rhytidiadelphus spp.			5,3											6,4										NA	
Rubus chamaemorus								+,1		+,1						+,1	+,1	3,2	+,2	+,1				NA	
Rubus pubescens		+,1																						NA	
Sanguisorba canadensis	3,2	2,2	3,1	4,2				+,1	+,2	6,3	+,2	2,1		+,2										NA	
Solidago macrophylla		+,1																						NA	
Sorbus spp.						+,1	+,1											+,1						NA	+,1
Sphagnum spp.	6,5				6,5		3,3	6,5	6,5	6,5	6,5	6,5	6,5			6,5		6,5	6,5		6,5	5,4	5,4	NA	6,4
Taxus canadensis												+,1			+,1									NA	
Trientalis borealis		2,2			+,1	+,2	+,1		+,1				+,1	+,1					+,1			+,1		NA	
Thalictrum polygamum		+,1																						NA	
Vaccinium angustifolium						3,2							+,1	+,1	+,1	+,1	+,1			+,1	+,1	2,1	+,1	NA	
Vaccinum ovalifolium						+,1	+,1					+,1					+,1	+,1	+,1					NA	2,1
Viola spp.										1,2	+,1													NA	

Table C.2: Braun-Blaunquet cover and sociability classes for plant species in T1-1B in the summer of 1995.

	Quadrat Number																								
Species	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25
Abies balsamea	+,1	+,1	+,1		+,1				+,1		+,1		2,1	3,1	+,1	4.1	+,1	+,1		road	road		+,1	NA	1,1
Amelanchier spp.	+,1	3,1	+,1			+,1						+,1	2,1	+,1						road	road			NA	
Aster spp.		+,1						1,2				+,1								road	road			NA	
Betula papyrifera													+,1						+,1	road	road			NA	
Betula spp.									+,1											road	road			NA	+,1
Carex spp.	1,2		3,3		2,2	3,2	5,4	3,2	2,2	1.3	+,1	2,3								road	road			NA	
Chamaedaphne calyculata	+,1	+,1																							
Cladonia spp.											+,1				+,1	+,1				road	road			NA	
Clintonia borealis			1,2	+,1	+,2	1.1	2,2		+,2	+,2	+,1		1,2	+,1	1,2		+,1	1,2		road	road		+,2	NA	+,1
Coptis groenlandica	+,2		+,2	2,2	3,2	2,3	2,2	1,2	2,2	1,2		2,2	2,2	1,2	1,2					road	road			NA	1,2
Cornus canadensis	2,1	+,1	+,1	+,1	2,2	1,2	3,2	1,2	1,2		+,1	+,1	1,2	2,2	1,2					road	road	2,2		NA	
Dicranum spp.					4,4		+,1				6.5			6,5	6,5					road	road			NA	3,2
Drosera sp.					+,1	1,1				+,1		+,2								road	road			NA	
Dryopteris novaboracensis			3,2	3,1	+,1	+,1														road	road		+,1	NA	
Dryopteris phegopteris			+,1																	road	road			NA	
Dryopteris spinulosa														+,1	+,1		2,1		+,1	road	road	4,1	2,1	NA	3,1
Epigaea repens					+,1		2,2		+,1			+,1	+,1	2.2	1,2			+,1		road	road			NA	
Equisetum sylvaticum			+,1	+,1	2,2				+,1	+,1										road	road			NA	
Gaultheria hispidula	2,3	+,1	+,1		2,3	+,1		+,1	1,2		+,1	2,2	2,3	+,2	3,2		+,1			road	road			NA	
Hepaticae													+,1				6,5			road	road			NA	3,3
Hylocomium spp.	6,5	6,5											5,4							road	road			NA	
Kalmia polifolia								+,1	+,1			+,1								road	road			NA	
Ledum groenlandicum	+,1	+,1																		road	road			NA	
Linnaea borealis	2,3	+,1	+,1		+,1	+,1			+,2	+,1		+,1	+,1	+,2	1,2					road	road			NA	
Maianthemum canadense					+,1	+,1	1,2		+,1		+,1	+,2	+,1	+,1	+,1	+,1	1,2	+,1		road	road		+,2	NA	+,1
Myrica gale								2,2				5,2								road	road			NA	
Nemopanthus mucronata														+,1						road	road			NA	
Osmunda cinnamomea				6,1		4,1	6,1			3,1		3,1								road	road			NA	
Picea glauca		5,1																		road	road			NA	
Picea mariana	6,1				+,1	4,1	+,1	+,1	2,1	+,1	6,1	4,1	4,1	+,1	+,1					road	road			NA	
Pleurozium schreberi					6,4											5,4				road	road			NA	4,4

Table C.2 continued:

	Quadrat Number																								
Species	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25
Poaceae												1.2	+.1							road	road			NA	
Ptilium crista-castrensis					+.1								3.3			4.3				road	road	5.1		NA	
Rhododendron canadense	5.2	3.1						+.1				+.1	+.1	3.1		1.2		1.1		road	road			NA	
Rhytidiadelphus spp.							+.1				4.4		3.1							road	road	4.4		NA	4.4
Rubus chamaemorus						+.1				+.1										road	road			NA	
Sanguisorba canadensis		+.1						+.1				2.1								road	road			NA	
Solidago macrophylla				+.1																road	road		2.2	NA	
Sorbus spp.		+.1			+.1										+.1										
Sphagnum spp.			6.5	6.5		6.5		6.5	6.5	6.5		6.5	+.1							road	road			NA	
Streptopus roseus									+.1											road	road			NA	
Taxus canadensis	+.1		3.1																	road	road			NA	
Trientalis borealis	2.1	+.1	+.1	+.1		+.1		+.1				+.1	+.1	+.1	+.1					road	road	+.1	+.1	NA	+.2
Vaccinium angustifolium	+.1	+.1											+.1		+.1					road	road			NA	
Vaccinium ovalifolium									+.1					3.1	+.1		+.1			road	road	+.1		NA	+.1
Viola spp.								1.2												road	road			NA	

Table C.3: Braun-Blaunquet cover and sociability classes for plant species in T1-1C in the summer of 1995.

	Quadrat Number																								
Species	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25
Abies balsamea		+,1	1,1	3.1	+,1	3,1	+,1	2.1	+,1		+,1	+,1			2.1				2,1	2,1	+,1	+,1		3,1	+,1
Acer rubrum														+,1											
Amelanchier spp.		2,1				+,1	+,1			+,1		+,1	+,1								+,1	+,1		+,1	
Athyrium felix-femina								+,1					5,4	4,4	1,1		+,1	3,3						+,1	
Betula papyrifera												+,1								+,1	+,1			+,1	
Betula spp.		+,1				+,1	+,1				+,1				2,1										+,1
Carex spp.							5,4			3,3				+,1											
Clintonia borealis	1,2	1,2	2.2	+,1	2.2	+,1		2,2	+,2	+,1	+,1	1.1	+,1		+,1	+,1	+,1	+,1	+,1		1,2	+,1	+,2		1,2
Coptis groenlandica			1,1		+,1	+,2	1,2		1,2											+,1	+,2		1,2	1,2	1,2
Cornus canadensis		1,2	1.1		2,2		1,2	+,1	+,1		+,2	1,2	+,1	+,1	1,2	+,1	+,1	+,2	2,2	+,1	1,2	+,1	2,3	2,3	1,2
Cornus stolonifera											+,1														
Dicranum spp.			4,4								3,1								+,1		3,3				
Dryopteris phegopteris								+,1									+,2					+,1			
Dryopteris spinulosa		3,1	+,1	+,2		+,1		+,1	+,1		+,1	2,2			1,1	+,1			+,1	+,1	+,1		+,1	2,1	+,1
Epigaea repens					+,2														+,1		+,1				
Equisetum sylvaticum	+,1													+,1											
Gaultheria hispidula			+,1			1,3	1,2		1,2												1,2				
Hylocomium spp.			3,1				3,1	6,5												+,1					3,3
Linnaea borealis		1,2	+,1		2,2		1,2	+,1	+,1		1,2	1,2			1,2				+,2	+,1	+,1	+,1	+,1	1,3	1.2
Listera cordata																									+,1
Maianthemum canadense			+,1	1,1	+,1	1,2		1,1	+,1		+,1					+,1									
Moneses uniflora																									+,1
Nemopanthus mucronata													+,1		+,1										
Osmunda cinnamomea	+,1				+,1				+,1	+,1															
Picea mariana	3,1														+,1				+,1		4,1				
Pleurozium schreberi		6,5	3,1				5,4																		
Poaceae	+,2						3,3																		
Ptilium crista-castrensis			5,4	3,1				4,4													3,3				5,4
Rhododendron canadense					3,2																				
Rhytidiadelphus spp.								4,3			4,4					+,1			+,1	6,4		6,4			

Table C.3 continued:

	Quadrat Number																								
Species	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25
Rubus chamaemorus							1,2			+,1															
Sanguisorba canadensis	+,1						+,1			+,1				1,2											
Solidago macrophylla				+,1							+,2	+,1				+,1				1.2			+,1		+,1
Streptopus amplexifolius												+,1													
Streptopus roseus																						+,1			
Trientalis borealis			+,1	+,1		+,1			+,1											+,2		+,1			
Vaccinium angustifolium					+,1																				
Vaccinium ovalifolium				2,1	+,1		+,1		+,1		+,1				2,1						2,1			+,1	
Viola spp.		1.2												+,1											

Table C.4: Braun-Blaunquet cover and sociability classes for plant species in T1-1D in the summer of 1995.

	Quadrat Number																								
Species	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25
																								NA	
Abies balsamea	+,1	3,1	+,1					+,1	6,1		+,1	3,1	4,1	+,1	3,1		3,1	2,1			3,1		3,1	NA	
Amelanchier spp.	+,1			2,1	+,1			+,1				+,1			+,1		+,1	3,1			+,1			NA	+,1
Aster spp.							2,2													1,1				NA	
Betula papyrifera														+,1									+,1	NA	
Carex spp.				+,1			6,5			3,4							1,2		1,2		1,2			NA	
Chamaedaphne calyculata							+,1													+,1				NA	
Clintonia borealis	+,1	+,1		1,2	+,1	+,1				+,1	+,1	+,1	+,1	+,1	+,1	1,1	2,2	+,1			+,2	1,2	+,2	NA	+,1
Coptis groenlandica	1,1	+,1	1,2	1,2	+,1	+,1			+,1	1,1			+,1		+,1	+,1			+,1	+,1	1,2			NA	1,1
Cornus canadensis	3,3	1,2	+,1	4,3	1,2	2,2				+,1	+,1	1,2	+,2	+,2	3,2		1,1	2,2	+,1	1,2	2,3		1,2	NA	2,2
Dicranum spp.					6.5	2,1			+,1			3,1												NA	4,3
Drosera sp.							+,2													1,2		+,1		NA	
Dryopteris disjuncta			+,1																						
Dryopteris novaboracensis										4,2						6,4								NA	
Dryopteris phegopteris				+,1						+,1						+,1								NA	
Dryopteris spinulosa				+,1				+,1			+,1												+,1	NA	
Epigaea repens	+,2	+,1			1,3					+,1		+,1		+,1	2,1			+,1			3,2			NA	+,1
Eriophorum spp.							6,5															1,2		NA	
Gaultheria hispidula													1,2	+,2	3,4			2,3		+,1	2,2		+,2	NA	
Hylocomium spp.		3,1	6,5	5,4		3,1																	6,5	NA	
Kalmia polifolia							+,1																	NA	
Ledum groenlandicum																				+,1				NA	+,1
Linnaea borealis	1,2	1,2	+,1							1,2				+,1	2,3	+,1				+,1	2,2		+,1	NA	
Maianthemum canadense		1,1	1,1	1,2	1,1	1,1				+,1	+,1	+,1	+,1	+,1	+,1	1,1	+,1	1,2			+,1		1,2	NA	
Myrica gale						+,1																		NA	
Osmunda cinnamomea	+,1									5,2								2,1	3,1			+,1		NA	4,1
Picea glauca																								NA	+,1
Picea mariana	3,1				3,1	5,1				+,1										3,1				NA	
Pleurozium schreberi													6,4								4,3			NA	5,4
Poaceae	1,2									3,3										6,4		5,4		NA	
Ptilium crista-castrensis		5,4				6,5			+,1									6,4			5,4			NA	

Table C.4 continued:

	Quadrat Number																								
Species	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25
Rhododendron canadense					5.2	2.1														3.1				NA	+.1
Rubus chamaemorus																				+.1		+.1		NA	
Rhytidiadelphus spp.				5.4	5.4				+.1															NA	
Sanguisorba canadensis				2.1						3.2										2.1		+.1		NA	
Solidago macrophylla		+.1		+.1																				NA	
Solidago spp.	+.1																					+.1		NA	
Sorbus spp.	+.1	+.1							+.1					+.1	+.1		+.1							NA	
Sphagnum spp.	3.4						6.5									6.5	6.5			6.5		6.5		NA	
Streptopus roseus				+.1													+.1							NA	
Trientalis borealis	+.1	+.1	+.1	+.1		+.1			+.1	1.1		+.1	+.1	+.2	+.1	+.1	1.2		+.1		+.1			NA	+.1
Vaccinium angustifolium	+.1				2.2	2.2														+.1				NA	+.1
Vaccinium ovalifolium	+.1								+.1						+.1						+.1			NA	+.1
Vaccinium vitis-idaea					1.2																			NA	
Viola spp.				+.1													1.2					1.2		NA	

Table C.5: Braun-Blaunquet cover and sociability classes for plant species in T1-2A in the summer of 1995.

	Quadrat Number																								
Species	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25
Abies balsamea		2,1	3,1	+,1	1,1	5,1	2,1	3,1	4,1	5,1	5,1	6,1		5,1	5,1	3,1	4,1	5,1	6,1	3,1	3,1	+,1	2,1	+,1	+,1
Amelanchier spp.					+,1					+,1					+,1			+,1				+,1			
Acer spicatum			2,1																						
Betula papyrifera							+,1						+,1	+,1			+,1	+,1							
Betula sp.	+,1																								
Carex spp.	1,3	2,3																							
Cladonia spp.				6,4								+,1													
Clintonia borealis	+,1		+,1	+,1		+,1		1,2	+,2	3,2	+,1	+,1	3,2	+,2	+,1		+,1	+,1	+,1		+,1		+,1	+,2	+,1
Coptis groenlandica	+,1							1,1	+,2	1,2	+,1		1,2	1,2	1,2	+,1		+,1	+,2					+,2	1,2
Cornus canadensis	+,1	1,2	3,3					3,2	1,2	2,2	+,1		3,2		3,2		1,1	1,2	+,2	+,1	1,2	1,2	1,2		+,2
Dicranum spp.					2,1				2,3					4,3				3,3	4,3						
Dryopteris disjuncta	+,1	+,1																							
Dryopteris phegopteris	2,2	2,1	+,1																						
Dryopteris spinulosa	+,1			2,1	2,1		+,1				+,1	+,1		+,1		+,1						+,1	+,1	+,1	+,1
Epigaea repens								+,1	1,2	+,2			+,2	+,2	+,2	+,2	+,1	+,1	+,2		+,2		+,1	+,1	
Equisetum sylvaticum	1,3																								
Gaultheria hispidula							3,3		+,2	3,3				1,3	2,2	1,2	1,2	+,1	+,1	1,2	1,2			1,2	
Hepaticae										4,4						+,1	6,5	+,1							
Hylocomium spp.				5,3	4,3		3,1		+,1		3,1			+,1	5,4	3,3									
Linnaea borealis	+,1	+,1			+,1	+,1	+,1			+,1			+,1		1,2	1,2	+,1	+,2		1,2	1,3	1,2	1,2		
Maianthemum canadense			1,2	+,2			+,1	+,1	+,1		+,1	+,1	+,1	+,1	+,1	1,1	+,1	1,2	1,2	1,1	1,2	+,1	+,1	+,1	1,1
Onoclea sensibilis		2,1	2,1																						
Osmunda cinnamomea								6,2	2,1																
Picea mariana	+,1	6,1	+,1	+,1	3,1			3,1	4,1		+,1	3,1	+,1		3,1			3,1	3,1			+,1		3,1	+,1
Ptilium crista-castrensis					5,4			5,4	6,5	3,3			3,3		3,3										
Prenanthes trifoliolata			+,1																						
Rhododendron canadense								+,1											+,1						
Rhytidiadelphus spp.					6,5				5,4	+,1	5,4					6,4	3,3	5,4	4,3	6,4	6,4				
Rubus idaeus	+,1																								
Sanguisorba canadensis	+,1	3,2	3,1																						
Solidago macrophylla			+,1	+,1																					

Table C.5 continued:

Species	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25
	Quadrat Number																								
Sorbus spp.																		+,1					+,1	+,1	
Sphagnum spp.		6,5								6,5				4,3								3,1			
Streptopus roseus													+,1												
Thalictrum polygamum			+,1																						
Trientalis borealis		+,1		+,1			+,1						+,1	+,1	+,1	+,1	+,1	+,1		+,1	+,1	+,2	+,1		+,1
Vaccinium angustifolium			1,2				+,1	+,1		2,2			2,2	+,1	+,1		+,1				+,1			+,1	
Vaccinium ovalifolium									4,1					+,1	2,1	+,1	+,1		+,1	+,1	+,1	2,1	2,1	+,1	+,1
Viburnum edule		+,1	2,2																						
Viola spp.	1,2		1,2																			+,1			

Table C.6: Braun-Blaunquet cover and sociability classes for plant species in T1-2B in the summer of 1995.

	Quadrat Number																								
Species	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25
Abies balsamea	+,1		+,1	+,1	+,1	+,1	+,1	+,1	1,2	1,2	1,2	+,1	1,2	+,2	+,1	+,1	1,3	+,1	1,2	+,1	1,2	+,1	4,1	1,1	3,1
Alnus rugosa	+,1																								
Amelanchier spp.					+,1				+,1																
Anaphalis margaritacea	+,1																								
Athyrium filix-femina	+,1		4,1																						
Betula papyrifera																						+,1			
Betula spp.											+,1								+,1		+,1				
Campylium polygamum													6,4												
Carex spp.	2,2																								
Clintonia borealis			4,2		+,1	+,1	2,1	+,1	1,1	+,1	+,1	+,1				+,2	+,1	2,1	+,2	1,2	+,1	1,2	+,1		3,2
Coptis groenlandica										1,1		1,2				2,2	+,1	+,1				+,2			2,2
Cornus canadensis	3,2			3,2	+,1		+,1	+,1	+,1		+,1	1,2					+,1		1,2			+,2	+,2		
Cornus stolonifera	+,1																								
Dicranum spp.				6,5	6,5		5,4								6,5	5,4		4,4			4,3				3,3
Dryopteris disjuncta			2,1																						
Dryopteris phegopteris	+,1	+,1																							
Dryopteris spinulosa						+,1	3,2	+,1	2,1	2,1	4,1	+,1	+,1		3,1		+,1				4,1		2,1	+,1	
Epigaea repens																		3,3				+,1			
Gaultheria hispidula						+,1		1,2					2,2			1,1	1,3	1,2				1,3	3,4		
Hylocomium spp.			4,3		6,5			4,4					6,5												
Linnaea borealis								1,3									+,1	+,1				+,1	1,2		2,2
Maianthemum canadense		+,1	+,1		1,1		+,1	2,1		1,2	2,2	1,1	1,1		+,1	1,2	+,1	+,1	2,2			+,1	+,2		+,2
Moneses uniflora					+,1		+,1																		
Onoclea sensibilis		+,1	+,1																						
Picea glauca					3,1																				
Picea mariana	2,1																				+,1	+,2			
Pleurozium schreberi								6,4					6,5		6,4							6,5	3,3		6,5
Pteridium aquilinum				+,1																					
Ptilium crista-castrensis						4,4											5,4			5,3			3,3	4,3	3,3
Poaceae	+,1	+,1																							

Table C.6 continued:

	Quadrat Number																								
Species	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25
Ranunculus sp.	+,1																								
Rhododendron canadense																						+,1			
Rhytidiadelphus spp.					6,5	6,5	6,5									5,4	3,5	6.5	6,5		6,5			4,3	
Rubus pubescens	+,2	+,1	2,1	+,1																					
Rubus sp.	+,1		+,1	+,1																					
Sambucus pubens		+,1		2,1	3,1																				
Sanguisorba canadensis	1,2	+,1																							
Solidago macrophylla												+,1													2,1
Solidago spp.	+,1	2,2																							
Sorbus spp.													+,1					+,1	+,1				2,1		+,1
Sphagnum spp.																				5,3					
Streptopus roseus						+,1																	+,1		+,1
Taxus canadensis			+,1																						
Thalictrum polygamum	2,2																								
Trientalis borealis				+,1	+,1		+,1	1,2		+,1	1,2	+,1					+,2	+,1	+,2			+,1			+,1
Vaccinium angustifolium																						+,1			
Vaccinium ovalifolium							+,1		+,1				+,1							2,1	+,1		3,1	1,1	
Viola spp.	2,1	+,1																							

Table C.7: Braun-Blaunquet cover and sociability classes for plant species in T1-2D in the summer of 1995.

	Quadrut Number																								
Species	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**13**	**14**	**15**	**16**	**17**	**18**	**19**	**20**	**21**	**22**	**23**	**24**	**25**
Abies balsamea	3,1	3,1	+,1	+,1	+,1	5,1	+,1					1,1	2,1	+,1	+,1			+,1		+,1	2,1		4,1		+,1
Amelanchier spp.						+,1										+,1		+,1							
Aster spp.																	+,2								
Carex spp.							+,3			1,3						2,3	3,4						6,4		
Clintonia borealis	1,2	+,2	3,2	+,1		3,2	+,1					+,1				+,1	1,1	1,2				2,2	+,2	+,2	+,2
Coptis groenlandica		1,3				2,2	1,2		+,1	1,2													3,4		
Cornus canadensis			+,2	+,1		+,2	3,3		+,1	+,1		+,1	1,2										+,1	+,1	+,1
Dicranum spp.	4,4		6,5		+,1	5,4						3,3	6,4	6,4							+,3			3,3	6,4
Dryopteris spinulosa	+,1	2,1	3,1	+,1		+,1					+,1	3,3	+,1	+,1	+,1			+,1	+,1	+,1		+,1		+,1	+,1
Epigaea repens		+,1	+,1	+,1																					1,2
Equisetum sylvaticum																	2,1						+,1		
Gaultheria hispidula	+,2				1,3											+,1							1,2		
Hylocomium spp.	4,4	3,3			4,3								3,3												
Linnaea borealis												1,3											+,2	+,1	
Maianthemum canadense	1,2		1,2	+,2		+,1					+,1	+,1	2,1	+,1		+,1		+,1	+,1					+,2	+,2
Moneses uniflora		1,2	+,1																						
Osmunda cinnamomea							4,2			+,1						+,1						+,1			+,1
Picea mariana	+,1						5,2									+,1							3,1		
Pleurozium schreberi															+,1										
Ptilium crista-castrensis																		3,3							
Poaceae																						+,3			
Rhytidiadelphus spp.		5,4		6,5	6,5							3,3								+,1					
Rubus pubescens									+,1								+,1								
Sanguisorba canadensis																						1,2	3,2		
Sorbus spp.																	2,2								
Sphagnum spp.							2,1			+,1								6,4					+,1		
Taxus canadensis																							+,1		
Thalictrum polygamum																	+,1						+,1		
Trientalis borealis	+,1	+,1	+,1	+,2	+,1	+,1			+,1	+,1			2,1					+,1					+,1	+,1	
Vaccinium ovalifolium				+,1		2,2							+,1					+,1	+,1					3,1	
Viola spp.																	1,2					1,2	2,2		

Table C.8: Braun-Blaunquet cover and sociability classes for plant species in T1-3A in the summer of 1995.

	Quadrat Number																								
Species	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25
Abies balsamea	4,1					3,1	5,1				2,1				+,1			+,1	4,1		+,1		+,1	NA	4,1
Acer rubrum																							+,1	NA	
Alnus crispa										+,1			+,1		+,1							+,1		NA	
Amelanchier spp.					+,1	+,1												+,1						NA	
Aster spp.	1,1	1,1	1,1	1,1	+,1			1,1			1,2	1,2	1,2								+,1		+,1	NA	
Athyrium filix-femina																2,1								NA	
Betula papyrifera																							+,1	NA	
Betula spp.																		+,1		+,1				NA	
Carex spp.		1,2	4,3		4,3	2,2		3,2	2,3	6,4	4,3	1,2		6,4	4,3		5,3	4,3	1,2	2,2	4,3	4,4	3,3	NA	
Chamaedaphne calyculata		1,1	+,1	+,1	+,1			+,1																NA	
Clintonia borealis	1,2						1.2	1,2	1,2	1,2	+,2	2,2	+,2	1,2	+,1		1,2	+,1	1,2	1,2	1,2			NA	1,2
Coptis groenlandica	1,2									1,2			1,2		+,1		+,2	1,2		1,2	1,2			NA	1,2
Cornus canadensis	1,2				1,2		3,2							+,1	+,1	+,2			1,2	1,2	1,2	1,2		NA	1,2
Cornus stolonifera																	1,2						+,1	NA	
Dicranum spp.							4.3																	NA	
Drosera sp.			+,1					1,2	1,2	1.2		1,1	1,2							1,2				NA	
Dryopteris disjuncta						+,1								1,2	+,1	1,2			+,1			+,1	+,1	NA	
Dryopteris phegopteris																+,1		1,1			+,1			NA	4,1
Dryopteris spinulosa	+,1														+,1	+,1		2,1					3.1	NA	
Epigaea repens						+,2	+,2													+,1				NA	
Equisetum sylvaticum	1,2		+,1					2.1	+,1	+,1	+,1		1,2	+,1	+,1	+,2	1,2		+,1				1,1	NA	
Gaultheria hispidula						1,2				+,2							1,2	+,1	1,2	1,2	1,2		1,2	NA	1,2
Platanthera dilatata										+,1	+,1	+,1					+,1				+,1		+,1	NA	
Hylocomium spp.						+,1																		NA	
Kalmia polifolia				+,1																				NA	
Ledum groenlandicum					+,1						+,1		+,1											NA	
Linnaea borealis						1,2	1,2			+,2	+,1			1,1		+,1	+,1	+,1	+,1	+,1				NA	+,1
Lycopodium spp.	2.3																							NA	
Maianthemum canadense	+,1																							NA	
Myrica gale	+,1	5,2	+,1	4,2	+,1							2,2	3,2											NA	
Osmunda cinnamomea														3,1			4,1			4,1			+,1	NA	

Table C.8 continued:

	Quadrat Number																								
Species	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25
Picea glauca																						3.1		NA	
Picea mariana					6.1	3.1	+.1	+.1	+.1	+.1	+.1	+.1		3.1				4.1		6.1	2.1		3.1	NA	
Poaceae	3.3	6.4	5.4	6.4	1.2				+.1	2.2	1.2	6.4	4.3	1.1				+.1	1.1		+.1		1.2	NA	
Rhododendron canadense				+.1	2.1						1.1													NA	
Ribes glandulosum																							+.1	NA	
Rubus chamaemorus											+.1													NA	
Rubus idaeus																						+.1	3.2	NA	
Rubus pubescens						2.2			+.1						2.2									NA	
Sanguisorba canadensis	3.2	4.2	2.1	1.1	2.1				2.2	3.2	+.2	+.1	2.2	2.1	2.1		2.2	3.2	3.2	2.1	2.2			NA	
Solidago macrophylla						1.2	+.1																	NA	
Solidago spp.	+.2							+.1																NA	
Sphagnum spp.	6.5	3.2	6.5	6.5	6.5	3.2		5.4	6.5	6.5	6.5			6.5	6.5		4.4	6.5	6.5	5.4	6.5			NA	6.5
Taxus canadensis							+.1													+.1				NA	
Thalictrum polygamum	+.1																							NA	
Trientalis borealis	+.1				+.1		1.1						+.1			+.2	+.1						+.1	NA	+.1
Vaccinium angustifolium											+.1							1.2						NA	
Vaccinium ovalifolium																+.1							+.1	NA	
Vaccinium oxycoccus				+.1								+.1												NA	
Viola spp.	+.1					1.2				1.1	1.2			+.1	1.2	+.1	1.2		1.2			1.2	2.2	NA	

Table C.9: Braun-Blaunquet cover and sociability classes for plant species in T1-3B in the summer of 1995.

	Quadrat Number																								
Species	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25
Abies balsamea								+,1			+,1	+,1	+,1	+,1	1,1	+,1	+,1	+,1	1,1	+,1	1,1	+,1	+,1	NA	+,1
Acer rubrum																						2,1	+,1	NA	
Amelanchier spp.			+,1		+,1													+,1						NA	
Aster spp.		2,1	+,1	+,2	+,1	+,1	1,1																	NA	
Betula papyrifera													+,1											NA	
Betula spp.							+,1							+,1			+,1				+,1		+,1	NA	+,1
Carex spp.		3.3	3,3	1,2					2,3	2,2														NA	
Chamaedaphne calyculata		+,1	+,1	+,1	+,1																			NA	
Clintonia borealis	+,2			+,1	+,1			1,1	2.2	+.1	1,2	+.1	+,1		+,1	1,2	+,2	1,1		+,1		1,2		NA	
Coptis groenlandica				1,2	+,2	1,2	1,2	1.2	+,2	2,2		1,1	1,2	+,2	1,2		2,2	1,2		1,2	1,2	+,2		NA	
Cornus canadensis	1,1		+,2	1,1		1,2	1,2	1,2	1,2	1,2		1,1	1.2	+,2	1,2	1,2	+.1	1,3	1,1	3,3		1,2		NA	3.2
Dicranum spp.						3,3		6,4		5.4		3,2	4,3	4,3			3,2		6,5		4.4			NA	
Drosera sp.		1,2			+,1		1,2																	NA	
Dryopteris phegopteris	+,1																							NA	
Dryopteris spinulosa							3,1									3,1				4,1		2,1	4,1	NA	5,1
Equisetum sylvaticum					+,1	+,1	+,1																	NA	
Gaultheria hispidula							1.2		1,2	+,1		+,1									1,2			NA	
Hylocomium spp.			5,4	+,1									3,2											NA	
Kalmia polifolia			+,1		+,1																			NA	
Ledum groenlandicum					+,1																			NA	
Linnaea borealis	+,2			+,1	+,1			1,1	2,2	+,1	1,2	+,1	+,1		+,1	1,2	+,2	1,1		+,1		1,2		NA	
Listera cordata													+,1											NA	
Loiseleuria procumbens					+,1																			NA	
Lycopodium spp.					+,2																			NA	
Maianthemum canadense			+,2	+,2				1,2	+,1		+,1	1,2	1,2	+,1	1,2		+,1	+,1		+,2	1,2	1,2		NA	
Moneses uniflora														+,1										NA	
Myrica gale	+,1	3,2	5,2	2,2	5,2	+,1																		NA	
Picea glauca																	5,1	+,1		+,1				NA	5,1
Picea mariana	6,1		2,1	6,1	3,1	6,1	5,1									+,1							2,1	NA	
Pleurozium schreberi					+,1			4,3	6,5	4,3				6,5	4,3			5,4	1,1					NA	
Poaceae					6,5		2,2																	NA	

Table C.9 continued:

	Quadrat Number																								
Species	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25
Ptilium crista-castrensis											+,1								+,1					NA	
Rhamnus alnifolius						+,1																		NA	
Rhododendron canadense			2,1	+,1		+,1	+,1																	NA	
Rhytidiadelphus spp.																		3,2						NA	
Rubus chamaemorus			1,1	1,1			1,1																	NA	
Rubus idaeus																						+,1		NA	
Rubus pubescens	+,1																							NA	
Sanguisorba canadensis	+,1			+,1	+,1	+,1	+,1																	NA	
Solidago macrophylla								+,1			+,1						+,1							NA	
Sorbus spp.								+,1					+,1											NA	
Sphagnum spp.	6,5		4,3		6,5	5,4	6,5			3,2			+,1											NA	
Spiraea latifolia													+,1											NA	
Streptopus roseus																+,1	+,1	+,1				+,1	+,1	NA	
Taxus canadensis	+,1					+,1	+,1		+,1															NA	
Trientalis borealis			+,2				+,1	1,1	+,1	+,1	1,1	+,1	1,2		+,1	1,2	1,2	+,1	+,1		1,2	+,1	1,1	NA	+,1
Vaccinium angustifolium					+,1	+,1	+,1																	NA	
Vaccinium ovalifolium											+,1	+,1		+,1		+,1	+,1			+,1	+,1			NA	
Vaccinium oxycoccus					+,1																			NA	
Viburnum cassinoides						+,1																			
Viola spp.											+,1													NA	

Appendix D

Table D.1: The average number of large mammal (*Alces alces* and *Rangifer tarandus*) tracks per 100 m for each site in the Copper Lake Watershed. 1994 to 1996 data are prior to forest harvesting, while 1997 data are post-harvest.

Species	Year	Site	Tracks/100 m
Alces alces	1994	T1-1B	0.02
	1995	T1-2B	0.02
		T1-3A	0.1
	1996	T1-1A	0.01
		T1-2B	0.05
		T1-3A	0.01
		T1-3B	0.05
	1997	T1-2A	0.11
		T1-2B	0.08
		T1-3B	0.14
Rangifer tarandus	1994	T1-1A	0.51
		T1-1B	0.11
	1995	T1-1B	0.01
		T1-2A	0.03
		T1-2B	0.02
		T1-3B	0.12
	1996	T1-3A	0.04
		T1-3B	0.3
	1997	T1-1A	0.03
		T1-1B	0.01
		T1-2B	0.01
		T1-3B	0.03