Various effects of forest harvesting and management on stream ecosystems in the vicinity of Fundy National Park, New Brunswick

by

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Abstract

This study examines some effects of clearcutting and road-building on streams in the vicinity of Fundy National Park, New Brunswick. Sixteen streams were selected for sampling: four were reference streams located within mature forest in the park, and twelve streams drained nearby clearcuts 4-20-years old. Streams were selected on the basis of similarities in stream order, size, and accessibility. Along each stream, sampling was centred around a 25-m reach, with at least one riffle, and with depth and width similar across streams. From May to November, 1993, each stream was sampled for temperature, dissolved oxygen, major ions, sedimentation and bedload movement, substrate type and concentration of organic carbon, channel dimensions, riparian vegetation, and invertebrates (the latter sampled using gravel-filled rockballs, placed in sampling reaches for six weeks).

Harvesting within the riparian zone reduced canopy shading, reduced cover of aquatic bryophytes, removed large trees and snags, and increased densities of shrubs (primarily *Alnus rugosa*). Riparian buffer strips reduced these effects, provided they were wide enough to ensure shading of the stream channel and feeder springs, withstand windthrow of riparian trees, and supply inputs of large woody debris. In general, with or without a riparian buffer, streams draining harvested areas had warmer water and greater temperature fluctuations, and larger nutrient concentrations and sedimentation rates than streams draining reference watersheds. Most of the highest rates of sedimentation occurred downstream of logging roads, while most of the lowest rates were recorded in reference streams. Three of the reference streams had similar properties and were fairly distinct from the cutover streams, and from the fourth reference stream, which was logged in the 1920s.

Clearcutting appeared to have a negative influence on the abundance of Chironomidae and Nymphomyiidae, and positive influences on Plecoptera, Elmidae, and Oligochaetes. The overall abundance of invertebrates colonizing rockballs was strongly correlated with age of stand, with the greatest numbers occurring in reference and older cutover streams, and the smallest numbers in the most recently cutover streams. The numbers of invertebrates colonizing rockballs in the older cutover streams were similar to those of undisturbed streams. This occurred despite the slower recovery of stream energetic processes in the older cutovers, as indicated by a principal components analysis of physical, chemical, and biological variables.

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1.0.0. Introduction

This study was designed to provide insights into the effects of forestry-related environmental stressors on the structure and function of stream ecosystems in the vicinity of Fundy National Park, New Brunswick (a study area known as the Greater Fundy Ecosystem). Sixteen first-order streams were selected for sampling, twelve of which drained clearcuts aged 4-20 years old, and four of which drained reference watersheds within Fundy National Park. Each stream was sampled for substrates, water temperature, nutrient concentrations, riparian vegetation, sedimentation, channel morphology, and benthic invertebrates. The choice of study streams represents a time series (or chronosequence), allowing study of the longer- and short-term effects of clearcutting using different-aged cutovers, instead of monitoring particular streams before, during, and after clearcutting. This approach assumes that changes caused by clearcutting and road-building would be detectable despite location-related differences amongst the 16 streams. To reduce the effects of spatial influences, the study streams were selected to be close to each other, reducing differences in climate, geology, geomorphology, and forest type.

The objectives of this study were:

i) To determine how rapidly streams in clearcut and reforested sites in the Greater Fundy Ecosystem recover biotic regulation of nutrient and sediment export, water temperature, and allochthonous inputs of biomass.

ii) To determine if there are substantial changes in the invertebrate community (that is, in terms of the abundance and dominance of taxonomic groups), and to identify environmental factors related to, and possibly influencing those changes.

iii) To assess the effects of clearcutting and plantation establishment on the habitat of fish.

iv) To contribute to the development and evaluation of guidelines for riparian buffer strips, road construction, culvert design, and alternative harvesting and management practices. Such information is relevant to the operational needs of forest, fishery, and national-park managers.

1.1.0. The effects of disturbance on ecosystems

The succession of a forested ecosystem towards a mature, and eventually an old-growth state, is represented by adjustments of structure and function and by changes in species composition (Bormann and Likens 1979). Well-established vegetation increases the efficiency of an ecosystem by regulating energy flows, sediment exports, and water and nutrient cycles, so that relatively predictable conditions are maintained. The functional complexity of such a highly organized, mature system enhances resistance to small or short-term perturbations, such as naturally recurring wildfires, strong wind storms, and insect infestations (Odum 1981; Bormann 1982). Usually, an ecosystem has a built-in ability to recover from such perturbations or stresses, and, over the course of several decades, the ecosystem will return to its predisturbance state of structure and function. In boreal systems, for example, wildfires and wind storms commonly occur at intervals of 50-100 years, leaving a biotic community with species adapted to fire and windthrow (Schindler 1987).

In addition to natural stressors, anthropogenic activities subject ecosystems to stress. Depending on the persistence and intensity of the activity, the system may be able to absorb the stress without undergoing measurable changes. If, however, some threshold of tolerance or resistance is exceeded, the system will display disruptions of ecological processes. Such disruptions include nutrient cycles becoming leaky, energy flows becoming smaller at higher trophic levels, food chains shortening, tolerant species becoming more dominant, and, following severe disturbances, species richness being reduced (Odum 1981; Freedman 1995).

Some disturbances can so damage the ecosystem - through loss of species, ecosystem structure, nutrients and soil, that the capacity of the ecosystem to repair itself is greatly reduced (Odum 1981; Bormann 1982). Succession reverts to earlier stages, with energy being diverted from maintenance and production to repair and recovery. Under such conditions, damaged ecosystems may take centuries or more to achieve predisturbance rates of productivity and levels of complexity.

Anthropogenic disturbances such as log driving and damming have been reported to cause enough damage to stream and river ecosystems that recovery may take hundreds of years (D. Clay pers. comm. 1994). In Fundy National Park, New Brunswick, dams across the major rivers caused local extirpations of anadromous fish by blocking their migration for 150 years; brow sites (where logs were slid into rivers) are still devoid of vegetation and causing major land slides nearly fifty years after their last use in lumbering operations; and the damages caused by logdrives have caused a lingering depletion of the overwintering habitat of salmonids by making riverine habitats wider, more shallow, and more homogeneous, with fewer deep pools and quality spawning beds (Cooper and Clay 1994).

Current practices of clearcutting also cause severe stress to stream ecosystems. Many of the common symptoms include: large losses of dissolved nutrients; a species - rich community of ruderal plants and animals; increased losses of inorganic and organic material; increased water temperatures, streamflows and peakflows; less stable substrates; and changes in the type and timing of energy inputs and production (Freedman 1995).

1.2.0. Forestry Activities

Forests around the world are being cleared at increasing rates to provide land for agriculture, and for lumber, paper products and energy production. In the late 1980s, about 25 million ha/year of forest were cleared globally, yielding approximately 4.1 billion m^3 of wood per year (Freedman 1995). This represents a 25% increase in the area cleared and the volume harvested annually since the late 1970s, and a net global deforestation rate of 1%/year (Freedman 1995). In 1989, international trade in forestry products was worth approximately U.S.$101 billion (Freedman 1995).

In Canada, large areas of mature forest are cut each year to supply a multi-million dollar tree fibre industry. Between 1985-1990, an average of 988 000 ha were harvested annually, 91% by clearcutting (Freedman 1995). New Brunswick, one of the smallest Canadian provinces, has one of the highest rates of deforestation in the country (Canadian Council of Forest Ministers 1994). In 1992 alone, 1.7% of the total available forest land of New Brunswick was harvested for tree fibre. Between 1985 and 1990, an average of 88 725 ha or 1.45% of forest land in New Brunswick were harvested annually (0.45% higher than the global average, 1.25% higher than the national average). Over 6 years, this totalled 8.9% of the total timber-productive forest land available for the harvesting of forest crops, or 8.7% of the total forest land in New Brunswick (Canadian Council of Forest Ministers 1994).

Various authors in the Maritime provinces have researched aspects of forest harvesting and management on watersheds. Englert *et al.* (1982) studied the effects of logging disturbances on salmonid biomass in ten small streams in New Brunswick and Nova Scotia. Researchers in the Nashwaak Experimental Watershed, New Brunswick, monitored stream hydrology, nutrient exports, suspended sediments, riparian inputs, thermal regime, and invertebrate and fish populations before and after commercial clearcutting along Narrows Mountain Brook, and along a spatial control, Hayden Brook (Powell 1981, 1982, 1983). Henderson (1978) looked at the effects of clearcutting on water yields in the Shubenacadie-Stewiacke River Basin. Van Groenewoud (1977) discussed the benefits of buffer strips for the protection of headwater streams against heating and sedimentation, and for the maintenance of riparian inputs of biomass. He further suggested a minimum buffer strip width for the protection of streams in Nova Scotia. Welch *et al.* (1977) sampled macroinvertebrate and trout abundance in clearcut and reference streams of New Brunwick, as well as measuring siltation, channel width and depth, and stream substrates. Sabean (1977) recorded stream temperatures in Nova Scotia in watersheds with various amounts of canopy cover.

While much information can be gleaned from these and other's studies, responses tend to vary with harvesting technique, topography, climate, soil type, vegetation and surficial geology. Thus there arises a need for site-specific research to protect streams. The area of Fundy National Park, New Brunswick, provides an excellent opportunity to study the effects of intensive harvesting activities on stream ecosystems. The surrounding forests have been extensively clearcut, so that the park has become an ecological island of forest land in southern New Brunswick (Woodley 1985). In addition, the New Brunswick government's willingness to heed the advice of researchers in the Greater Fundy Ecosystem provides the opportunity to supply rationale and recommendations for the better protection of streams in this region.

To understand the effects of forestry activities on stream ecosystems it is useful to study both the biotic community and its interactions with the physical environment. Determination of the structure and dynamics of the benthic macroinvertebrate community is one key to understanding the state of a freshwater ecosystem. Residing in the streambed, and feeding upon riparian inputs of litter and organic debris, the benthic community is particularly sensitive to ecosystem disturbances like clearcutting. Community structure reflects the energetics of the ecosystem and can be used to make inferences about changes in ecological processes, such as productivity and decomposition. In addition, the long life histories of various species provides a cumulative indication of conditions over time (Reice and Wohlenberg 1990). By concentrating on the benthos as a community, interacting with and reflecting changes in the ecosystem, important insights into the dynamics of freshwater ecosystems can be made (Reice and Wohlenberg 1990).

1.3.0. Background: Effects of forestry on stream ecosystems

Following is a summary of the known effects of clearcutting and road construction on the physical and chemical characteristics of streams, and the consequent effects on macroinvertebrates and salmonids. Most of the research cited has been done in North America, except for several studies from Great Britain, Australia and New Zealand.

1.3.1. Stream temperature and associated factors

The effects of clearcutting on water temperature are directly related to the surface area of the stream exposed to direct sunlight. With the harvesting of riparian vegetation, the low-intensity, diffused light under the forest canopy is replaced by direct solar radiation (Corbett *et al.* 1978). During the plant growing season, this results in higher maximum stream temperatures and greater daily fluctuations (Krause 1982; Hetherington 1986; Campbell and Doeg 1989). Several studies in North America have documented increases in maximum stream temperatures of 5.7°C to 15°C, with the highest recorded temperatures sometimes exceeding 30°C (Gray and Edington 1969; Hall and Lantz 1969; Brown and Krygier 1970; Lee and Samuel 1976; Feller 1981; Lynch *et al.* 1984; Noel *et al.* 1986; Garman and Moring 1991; Ahtiainen 1992). Daily temperature fluctuations of 15°C have been observed in some streams where little vegetation was left standing (Bjornn and Reiser 1991). In contrast, where riparian buffer strips are maintained, changes in stream temperature regime are smaller, and may not be measurable (Sweeney 1992; Osborne and Kovacic 1993). The buffer strip must, however, be provided along the entire length of stream (Brown 1971).

Higher-than-average streamwater temperatures and large daily temperature fluctuations have many documented effects on the macroinvertebrate and fish communities of streams. These include changes in development rates, behaviour, physiology, reproductive success, and survival (Lynch *et al.* 1984). Although higher-than-average temperatures may enhance growth and survival, extreme temperatures and extreme fluctuations in thermal and related water quality characteristics can be lethal, or can exceed metabolic capacities (e.g., dissolved oxygen) (Fry 1947). For example, in Carnation Creek, British Columbia, higher-than-expected average stream temperatures in the spring produced larger fry of chum salmon (*Oncorhynchus keta*) and coho salmon (*O. kisutch*) and induced earlier migration to the sea, thereby improving marine survival (Hartman and Scrivener 1990). However, these initial beneficial effects were lost when egg production and survival declined in later years, possibly as a consequence of lethally high temperatures.

The optimum temperature range for adult brook trout is 11°C - 16°C, while temperatures warmer than 20°C cause stress, and at 25°C, mortality occurs within several hours (Brett 1956; Raleigh 1982). The preferred temperature range for yearling trout is 10-12°C, and the upper lethal temperature, 25°C (Meehan and Bjornn 1991). Optimum temperatures for Atlantic salmon (*Salmo salar*) are between 14 and 16°C, and lethal levels are 25°C and higher (Alabaster 1967, as cited in Sabean 1977). Smaller salmonids will often protect themselves from sub-optimum temperatures by hiding in the interstices of substrates, or in other forms of cover (Bjornn and Reiser 1991). Larger fish tend to migrate to locations with more desirable temperatures. These lead to changes in the fish community, including invasion by warmwater species into formerly cool, headwater reaches. However, salmonids may not always avoid unsuitable temperatures, especially if the temperature changes are rapid and not part of the usual pattern of environmental change (Bjornn and Reiser 1991).

Physiological stress induced by high temperatures and rapid temperature fluctuations reduces resistance to disease and predation, and inhibits feeding and reproduction (Lynch *et al.* 1984). Various studies carried out on salmonids show that higher-than-average temperatures significantly increase the incidence of various diseases. Increased drift of some mayfly species is also a documented response to higher stream temperatures (Wojtalik and Waters 1970). Lynch *et al.* (1984) found that daily temperature fluctuations of 17°C in a stream draining a clearcut and herbicided plot were large enough to cause thermal shock to brook trout.

In addition to the direct effects of higher temperatures, survival of stream biota is also influenced by low dissolved oxygen concentrations that may be a consequence of elevated temperatures and the addition of organic sediment and logging debris into streams (Ponce 1974). Higher temperatures increase metabolic activity, which then increases oxygen demand; at the same time, however, warm waters hold less oxygen, and decomposition of organic matter consumes oxygen, thereby reducing its availability to stream organisms. In turbulent waters, dissolved oxygen is typically at a tolerable concentration for animal and plant life. However, in streams with low flows, high temperatures, and decomposing organic matter, concentrations may reach critical concentrations for the survival of macroinvertebrates and fish eggs, juveniles, and adults.

While many midges tolerate low concentrations of dissolved oxygen (down to 1 mg/l), cold-water mayflies and stoneflies cannot tolerate oxygen concentrations much below 5 mg/l (Nebeker 1972). Sub-optimal concentrations for salmonid eggs produces small weak embryos, premature or delayed hatching, and an increased incidence of morphological anomalies (Bjornn and Reiser 1991). Dissolved oxygen concentrations below 3.3 mg/l in standing water are lethal for Atlantic salmon (*Salmo salar* Linnaeus), while concentrations less than 5 mg/l induce relocation to other watercourses, and reduce growth rates and food conversion efficiencies of Atlantic salmon and brook trout (Bjornn and Reiser 1991). Oxygen concentrations below 8-9 mg/l can adversely affect the swimming performance of salmonids, with maximum sustained swimming speeds of brook trout declining sharply when dissolved oxygen drops below air-saturation concentrations (Raleigh 1982).

1.3.2. Nutrients

The precedent-setting work of Bormann and Likens (1979) in Hubbard Brook, New Hampshire, documented large exports of dissolved nutrient ions into streams following forest cutting and two years of vegetation suppression by herbicide treatments. These losses were found to be most severe for nutrients that are relatively mobile in soil - particularly nitrate (NO^{3-}) and potassium (K^+). In the three years after cutting, average streamwater concentrations of ions from the deforested watershed exceeded those of the reference forested watershed by: (NO_3^-) 40 times; (K^+) 11 times; (Ca^{2+}) 5.2 times; (Al^{3+}) 5.2 times; (Mg^{2+}) 3.9 times; (H^+) 2.5 times; (Na^+) 1.7 times; (Cl^-) 1.4 times and (Si) 1.4 times. Ion concentrations decreased during the third year following cutting, probably as a result of the progressive exhaustion of easily decomposed organic substrates and leaching of easily exchanged ions in soil. Commercial clearcutting without the use of herbicides in the same area of New Hampshire showed average streamwater concentrations in the three years post-logging to be (NO_3^-) 8.5 times, (K^+) 2.5 times, and (Ca^{2+}) 2 times greater than expected for an uncut watershed (Pierce *et al.* 1972).

Curiously, the White Mountains of New Hampshire appear to be more prone to nutrient losses than most other places studied in North America. While other researchers in New Hampshire also document large losses of nitrates, calcium and potassium (Pierce *et al.* 1972; Hornbeck *et al.* 1975;

both cited in Martin *et al.* 1984), similar studies in Pennsylvania, West Virginia, North Carolina, Maine, Vermont, and Connecticut found relatively minor changes in stream chemistry following logging (Martin *et al.* 1984). In Narrows Mountain Brook, New Brunswick, nitrate concentrations were 3 to 5 times, and potassium concentrations 1.5 to 2 times greater than expected following commercial clearcutting (Powell 1983). Hartman and Scrivener (1990) also found an increase in total ion concentration (mostly NO_3^-) in Carnation Creek, B.C., for 2 to 4 years following logging and debris burning. Losses were also intensified by herbicide treatments upstream, though total losses were far less than those seen in New Hampshire. Similarily, nitrogen losses after clearcutting and slash burning in Oregon were much less than those measured in Hubbard Brook (5.2 kg/ha/yr compared with 142 kg/ha/yr) but nevertheless 3.3- times greater than losses from undisturbed forests (Fredriksen 1971). In general, Frederiksen found nitrate, magnesium and potassium losses were higher from clearcut sites. Patch cuts, strip cuts, and buffer strips had smaller losses of nutrients than clearcuts.

A number of factors influence this variation in nutrient losses. Slope may affect the water table, degree of saturation, and rates of mineralization and nitrification, thus altering the amounts of nutrients reaching streams. Leaching of nutrients is further influenced by climate, which affects the amount of precipitation and evaporation, and soil type, which varies in exchange capacities and bonding strength (Bormann and Likens 1979; Martin *et al.* 1984).

1.3.3. Hydrology

During the growing season, trees evapotranspire large amounts of water into the atmosphere, usually more than is replenished by incoming precipitation (Bormann and Likens 1979). Removal of trees by clearcutting temporarily reduces transpiration, so that water entering the watershed exits as streamflow, seepage to deep groundwater, or evaporation (Freedman 1995). Disturbances to the hydrologic regime by forestry activities have been observed to generate more common and more pronounced peak flows, higher annual water yields, and increased summer low flows, with secondary downstream effects such as flooding and erosion (Hetherington 1986; Hartman and Scrivener 1990). Studies of forest hydrology in Canada have revealed increases of peakflow following harvesting of 0 - 230% (Hetherington 1986). Smaller evapotranspiration losses have resulted in higher summer streamflows of 10-318% in areas of British Columbia, New Brunswick and Ontario (Hetherington 1986). Changes in streamflow and peakflow are less apparent in the winter months, when soil water storage is very similar on deforested and forested watersheds. In the spring, the rate of snowmelt is increased by the relatively unshaded condition of clearcuts (Freedman 1995).

The increase in streamflow following harvesting usually depends on the proportion of the catchment harvested, and the amount of foliar transpiration surface removed (Campbell and Doeg 1989). Usually, the largest increases in streamflow occur in the first postcutting year, with water yields returning to predisturbance levels within several years to several decades (Hetherington 1986).

Increased flows have important effects on surface erosion of cut areas and export of dissolved substances and particulate matter through the watershed, and also on the survival of stream biota. By influencing the number of upstream migrants, as well as the amount of available spawning area, streamflow, particularly low flows in late summer to early winter, is critical for salmonid migration and survival (Raleigh 1982; Bjornn and Reiser 1991). With the elevated water yields following clearcutting, streamflow can increase during the critical low-flow period, thus improving chances of survival (Eschner and Larmoyeux 1963; Englert *et al.* 1982). Alternatively, fish and other stream organisms may be negatively affected because higher peak flows enhance scouring of bottom sediments; create abnormal bedload movements that dislodge eggs, larvae, and nymphs; accelerate bank cutting and stream widening; destabilize debris dams; create physical obstructions to migrating fish; and intensify downstream siltation (Ontario Ministry of Natural Resources 1988).

1.3.4. Sediments

Forest harvesting and the construction of logging roads can greatly increase suspended and deposited sediment loads in streams (Bormann and Likens 1979; Dorcey *et al.* 1980; Hartman and Scrivener 1990). This is primarily due to the exposure of mineral soil to erosion through removal of the forest floor during road construction, log skidding, prescribed burning and scarification. Erosion is particularly severe in wet weather, and in areas with fine soils, erodable bedrocks, and steep slopes (Environment Council of Alberta 1979). Sediment loads also increase with the use of streams as skid trails, with the harvesting of forest adjacent to water bodies, and with the increased capacity of faster-moving water to destabilize, erode, and transport sediments and debris (Krause 1982; Hetherington 1986). Maximum reported sediment concentrations during the construction of logging roads have ranged from 0.2 to 8 g/l (Krause 1982). In Carnation Creek, B.C., suspended sediments temporarily increased 2 to 175 times following the construction of logging roads. Dorcey *et al.* (1980) documented a study of 29 watersheds in Oregon, in which harvesting increased sedimentation an average of four times over pre-harvest rates. In Hubbard Brook, New Hampshire, output of sediment reached a maximum concentration of sixteen times that of the reference watershed in the third year after logging and two years of suppression of vegetation with herbicides (Bormann and Likens 1979). Erosion and transport of sediments decreased sharply after the second year of regrowth (the fifth year after cutting), as stream banks were stabilized by vegetation, and debris dams became re-established. In Carnation Creek, B.C., forestry activities produced persistent sources of sediment that were still accumulating in downstream areas 10 years after logging (Hartman and Scrivener 1990).

Increases in sediment load and bedload transport following clearcutting can result in significant changes in the substrate composition of spawning beds, though the specific effects are highly variable and difficult to predict (Duncan and Ward 1985). Adams (1980, cited in Duncan and Ward 1985) found that the magnitude of peak flows was correlated with composition of streambed gravels, and Duncan and Ward (1985) emphasized the importance of upstream geology and climate as determinants of gravel composition.

Increased erosion and sedimentation during and following construction of forest roads and forest harvesting can have notable impacts on stream benthos. Suspended sediments may clog the food-trapping mechanisms of filter-feeding insects, abrade respiratory organs, and inhibit the ability of insects to cling to silt-covered stones. Tumbling sediments (mostly fines <4 mm in diameter) can dislodge insects living on exposed rock surfaces, sometimes causing catastrophic drift of many taxa. In Carnation Creek, B.C., suspended and saltating fines reduced density and biomass of macroinvertebrates by 50% in 24 hours (Hartman and Scrivener 1990). The bottoms of fast-flowing headwater streams typically consist of a conglomeration of large boulders, cobble, gravel and sand, with little clay or silt (van Groenewoud 1977). Water brings oxygen and suspended organic matter through the interstices, and removes the potentially toxic metabolites of gravel-inhabiting caddisfly, mayfly, stonefly and black fly larvae (Morantz 1988). With the increase in sedimentation caused by clearcutting, accumulating sediments smother larvae and lower the permeability of the streambed. In the southern Appalachians, siltation of bottom gravels has reduced invertebrate populations as much as 70% (Tebo 1955). The mayflies *Pseudocloeon vinosum* and *Paraleptophlebia* spp., and the blackflies *Simulium* spp. are intolerant of silty surfaces (Chutter 1969; Hartman and Scrivener 1990).

Concurrent with a decrease in inputs of allochthonous litter following streambank logging, siltation and scouring of streambeds further decrease the capacity of substrates to retain detritus (Newbold *et al.* 1980). Wallace *et al.* (1988) noted that disturbances such as logging may alter physical characteristics of streams, such as the ability to retain litter, for decades.

High concentrations of suspended sediments may affect fish habitat by: (1) blocking light for photosynthesis, thereby reducing primary productivity, (2) obscuring visibility, thus impairing the ability of salmonids to see and capture prey, (3) reducing benthic insect populations, (4) causing direct harm to fish, and (5) preventing or delaying migration (Dorcey *et al.* 1980; Bjornn and Reiser 1991). Gill membranes are vulnerable to damage by suspended particles when stream velocities are high and little protective cover is available. Decreased foraging efficiency plus gill damage cause reduced growth and resistance to disease, thereby inducing fish to emigrate from or avoid waters with large silt loads (Morantz 1988).

Trout and salmon depend upon clean, cool, well-oxygenated water and silt-free gravel substrates for spawning, egg incubation, and juvenile development. Clogging of gravel interstices by organic and inorganic particles reduces the flow of water and oxygen to eggs, allows accumulations of toxic metabolites surrounding eggs, entombs fry, obliterates gravel substrate habitats, and increases susceptibility to disease and predation (Dorcey *et al.* 1980; Morantz 1988). The degree to which deposited sediments affect spawning gravels depends on the size of substrate in the redd, flow conditions in the stream, and the amount and size of the sediment being transported (Bjornn and Reiser 1992).

Productive salmon and trout spawning beds should not contain more than 5% silt. When the percentage of fine sediment in spawning gravel exceeds 15%, survival to emergence of salmonid eggs, embryos and alevins is sharply curtailed (Morantz 1988). More than 30% silt constitutes a non-productive spawning area for salmonids (Morantz 1988). An influx of 1-3 mm particles (or 'fines') into open gravels can result in a survival rate of only 50%, if the fines make up 30% of the original mass of open gravels (Phillips 1970, as cited in Bray 1988). Even if surface silt is rapidly flushed out by strong water currents, water velocities below the top layer of gravel may not be sufficient to flush out deposited silt. Hence, it may take decades to remove fine sediment from gravel beds, particularly where soil compaction contributes to siltation for many years post-harvesting (Englert *et al.* 1982).

1.3.5. Organic debris

Dead trees falling into streams create a diversity of hydraulic gradients and cover that enhance channel complexity, number and volume of pools, quality of cover, and capacity to store and process organic matter (Hicks *et al.* 1991). The resulting microhabitat heterogeneity allows the coexistence of multispecies fish communities, and is closely linked to salmonid abundance (Hicks *et al.* 1991). Clearcutting, selective-cutting, and road-building in riparian zones cause a long-term reduction in the recruitment of large woody debris to stream channels. While this may lead to a reduction of large, stable debris in stream channels, the rapid addition of large amounts of logging debris may create large, infrequent, unstable and impassable debris dams. In combination with past practices of clearing woody debris from stream channels, and dam construction for bridges and log drives, streams may take many decades to recover from logging-induced changes in large dimension, organic debris (Maser *et al.* 1988).

Large and rapid additions of logging debris to streams does not tend to have the same beneficial effects for stream biota as gradual additions of natural woody debris. Logging debris may (1) block migration routes, (2) destabilize gravel habitat, (3) reduce habitat by filling stream gravel interstices with slash and organic matter, (4) reduce oxygen flows in the interstices, (5) increase biochemical oxygen demand, (6) poison salmonid fry by decomposing and releasing dissolved organic compounds, (7) destroy cover and habitat by damage to streambanks, and (8) reduce flows and thermal conductivity of rocky streambeds, thus increasing daily temperature fluctuations (Dorcey *et al.* 1980; Hartman and Scrivener 1990).

1.3.6. Changes in food source

Low-order streams draining northern temperate forests are known to derive most of their total energy budget from leaf litter (Culp and Davies 1983). Approximately 95% of all food consumed by invertebrates of low order streams is derived from leaves, twigs, branches, and other organic debris that falls into streams from riparian trees and shrubs (Fisher and Likens 1973). In the short-term, clearcutting and selective logging of riparian zones reduce allochthonous inputs of biomass into headwater streams. Subsequent colonization of the harvested area by fast growing herbs, shrubs, and trees, in combination with increased solar radiation, stream temperatures, and nutrient concentrations, all contribute to altering a stream's energy budget.

The quality and quantity of allochthonous inputs may take many years to return to pre-logging values. Typically, leaf input after logging is from species which undergo relatively fast microbial breakdown, like alder (*Alnus spp.*) and salmon berry (*Rubus parviflorus*) in British Columbia. In contrast, leaf input before logging is more evenly distributed among species with fast and slow processing rates. Alder leaves, for example, are more easily decomposed than hemlock (*Tsuga* spp.) needles in B.C., and are thus a preferred energy source by detritivores (Hartman and Scrivener 1990). Results from experimental manipulations with substrates and detritus in Carnation Creek show that a significant decrease in invertebrate biomass and density resulted from the absence of fast-decomposing alder detritus from pebble and cobble substrates. Timing of allochthonous inputs may also influence the distribution and abundance of detritivores, for example, inputs of angiosperm foliage enter streams between September and November, while coniferous inputs enter mostly in the autumn, but also throughout the year.

Increased autochthonous production, in the form of algal growth, is often a direct result of the increased temperatures, direct insolation and increased nutrient concentrations following riparian logging (Burns 1972; Murphy *et al.* 1981). Higher densities of algae-consuming invertebrates have often been observed in cutover streams compared with undisturbed streams (Noel *et al.* 1986; Campbell and Doeg 1989). Alternatively, the potential for greater primary productivity may be reduced by sedimentation, scouring, and turbidity. Results from Carnation Creek, B.C., show lower densities of most invertebrate taxa in streams draining cutovers compared with nearby undisturbed streams (Culp and Davies 1983).

The amount of food available to fish is one of the factors influencing the salmonid carrying capacity of streams (Bjornn and Reiser 1991). Fish living in gravel-bottomed streams feed mostly on the bottom population of large larvae of caddisflies, stoneflies, and mayflies, as well as on stream drift, which includes both terrestrial and aquatic invertebrates. The ability of fish to capture prey varies with habitat complexity: at low levels of complexity (i.e., a small amount of cover and woody debris, and a large amount of fines), foraging efficiency is high but prey availability low (Wilzbach *et al.* 1986). Consequently, the risk of depletion of prey and of predation from piscivores is high. With high habitat complexity, salmonid foraging efficiency is low, but the large number of prey refuges provided by heterogeneous substrates and debris would both increase the amount of prey and drift, and decrease the risk of their depletion. Prolonged reductions in food supply can lead to adverse indirect effects on weakened fish, displacement, and fish starvation (Morantz 1988).

Increases in sedimentation and primary productivity following clearcutting may increase or decrease food availability to fish. Greater short-term growth rates of salmonids has been documented in logged sections of streams (Burns 1972; Englert *et al.* 1982). This has been attributed to a number of factors, including increased invertebrate drift, visibility, higher streamflows, foraging efficiency, primary productivity, and abundance of algae-consuming macroinvertebrates. However, over the longer-term, good fish-rearing habitat is degraded after clearcutting, resulting in declines in abundance and biomass of salmonids in cut-over streams (Murphy *et al.* 1986; Hartman and Scrivener 1990; Hicks *et al.* 1991).

1.3.7. Loss of riparian and in-stream cover

Cover is a basic and essential component of the habitat of salmonid streams. In the sense used here, cover can include overhanging and submerged vegetation, overhanging or undercut banks, in-stream objects (rocks, roots, logging debris), rocky substrates, water depth, and water surface turbulence (Raleigh 1982). Cover provides protection from predation, shelter during stormflows and spring runoff, and resting places for migrating salmonids (Morantz 1988). Cover is also important in determining the suitability of a stream for spawning salmonids. Anadromous fish entering streams weeks or months before they spawn require adequate cover for protection from disturbance and predation (Bjornn and Reiser 1991).

Removal of riparian vegetation, scouring of the stream channel, and siltation of bottom substrates tends to reduce cover for stream organisms. This may result in fry and juvenile salmonids and aquatic invertebrates being swept downstream, where they could face predation, competition, and unsuitable habitats (Henderson 1978; Morantz 1988). In some situations, however, clearcutting of streambank vegetation leads to the addition of logging debris into streams, which may enhance cover, but more often results in clogging of the stream channel (Elliott 1986).

1.3.8. Overall changes in the macroinvertebrate community

Streams draining clearcut watersheds tend to be dominated by macroinvertebrate species with short generation times, rapid colonization rates, and broad tolerances (e.g., *Baetis* spp. and Chironomidae) (Gurtz and Wallace 1984). These attributes enable the macroinvertebrates to cope with the greater fluctuations of the post-logging stream environment, and for certain species, to build up large populations opportunistically (Newbold *et al.* 1980). The greatest increases in macroinvertebrate densities following clearcutting have been observed in small, first-order, high gradient streams, with greater primary production than reference streams (Murphy and Hall 1981). However, suspended and deposited sediments tend to reduce species diversity and biomass, and cause changes in species composition of macroinvertebrates. Densities of taxa requiring solid surfaces are often reduced, and taxa such as oligochaetes and chironomid larvae, which are capable of using fine sediments as habitat, become more abundant (Campbell and Doeg 1989). As the stream recovers from the logging-induced disturbance, the average size of macroinvertebrate species can be expected to increase, with more large shredders and filter feeders, and fewer small collector-gatherers and scrapers (Gurtz and Wallace 1984).

1.4.0. Stream protection

1.4.1. Buffer zones

To a degree, uncut buffer strips can mitigate the effects of forest harvesting. They help control erosion by blocking overland flow of sediment and debris, by stabilizing streambanks and wetland edges, and by promoting infiltration, and they decrease the flow rate of water by resisting channelization. They remove nutrients in runoff by filtering water and by uptake by plants. In addition, buffers supply long-term inputs of large and small woody debris, thereby providing sources of cover, food, and habitat complexity. Finally, buffers provide canopy cover, thus helping to maintain low water temperatures in the summer and high temperatures in the winter (Castelle *et al.* 1994).

While riparian buffer strips reduce the impacts of clearcutting, they are not completely effective in protecting aquatic resources. For example, logging and forest roads often cross stream channels. It is not uncommon for culverts to wash out, resulting in sustained inputs of sediment into stream channels. Furthermore, increased nutrient exports, peak flows, and maximum temperatures have been observed in some cases, even with buffer strips. Hewlett and Fortson (1982) observed that forest cover reductions in areas of gentle land relief may elevate stream temperatures, even with a large buffer strip.

Krause (1981, as cited in Krause 1982) cited a 25-fold increase in nitrate loss following clearcutting with a 75-m-wide buffer strip on either side of the stream channel (though feeder streams had no buffers), and Martin and Pierce (1980) noted greater nitrate and calcium losses from a 70% clearcut watershed with 20-m-wide buffer strips on each side of the stream. Hopman *et al.* (1987) and Aubertin and Patric (1974) documented higher runoff and peakflows with 10 to 30 m-wide vegetation strips along each side of the stream channel.

Riparian zones provide valuable habitat for many aquatic and terrestrial species of wild life. While great attention is paid to the importance of riparian zones in maintaining trout and salmon habitat, they are also highly productive areas for terrestrial fauna and flora: at least 68% of the 52 mammal species in New Brunswick use riparian zones; many species of birds, bats, amphibians and mammals use them as migratory routes; and many species reside in the diverse habitat of coniferous and deciduous trees, snags, shrubs, rushes, sedges and emergents found along the water's edge (R. Stocek, unpublished 1993).

The effectiveness of buffer strips in protecting streams from degradation caused by forest clearing is partly dependent on their length and their width. While undersized buffers may place aquatic resources at risk, oversized buffers may infringe upon the resource-extractor's use of the land. Thus it is useful to determine a minimum buffer width to protect aquatic resources, as well as to protect the habitat of the many terrestrial species living in, or using riparian zones.

1.4.2. Forestry in New Brunswick

Currently, streams in New Brunswick are protected from forestry activities by the N.B. Clean Environment Act, the Crown Lands and Forest Act, and the N.B. Clean Water Act. In 1977, regulation 77-7, Section 32 of the Clean Environment Act prohibited unauthorized changes to watercourses. These changes included the operation of machinery or the removal of vegetation in a streambed, removal of substrates or soils from or within 30 m of a watercourse, and the removal of trees within 10 m of the bank of a watercourse. The Crown Lands and Forest Act (1983) required a 15-100-m buffer zone along watercourses, subject to the regional biologist. While these regulation applied to crown lands, companies were strongly encouraged to use these guidelines on private holdings. The Clean Water Act (1991) required a 15-m `no cut` zone on Crown or private land adjacent to streams greater than 0.5 m in width, and a 75-m buffer strip within municipal watersheds designated for the protection of water supplies.

The N.B. Department of Natural Resources and Energy has recently published a draft of their proposed stream protection guidelines for crown lands, entitled "Watercourse Buffer Zone Guidelines for Crown Forestry Activities" (March 1995). These guidelines suggest a minimum buffer width of 15 m (measured from each side of the watercourse) for stream channels greater than or equal to 0.5 m in width. If the drainage area is > 600 ha, however, the buffer zone cannot be less than 30 m in width on each side of the stream. Furthermore, where slope is >5%, erosion or windthrow hazard moderate, or the habitat critical for fish spawning, the buffer width must be at least 30-m-wide. Other considerations include: no mobile equipment within 15 m of a watercourse 0.5-m or greater in width; no logging roads located within buffer zones; no trees felled across or into a watercourse; no slash or debris from a timber harvest operation entering a watercourse; no ground disturbance within 15 m of a watercourse; no less than 70% of the initial snags and moribund trees be left standing in a buffer zone; and no more than 30% of the merchantable basal area within watercourse buffer zones be harvested within any 10 year period.

2.0.0. STUDY AREA

The study streams are situated in the vicinity of Fundy National Park, New Brunswick (45°37'N, 65°03'W) (Figure 1). Four are located within the park, while the remaining 12 are in clearcuts close to the Park boundaries: within 5 km to the west, 4 km to the north, and 4 km to the northeast. In total, the study area spans approximately 300 km^2. All the study watersheds drain into the Bay of Fundy (Map, Figure 2).

The Park and surrounding area is situated on a rolling plateau, dissected by deep, steep-sided river valleys. The plateau reaches elevations of over 300 m within 4-5 km of the coast, but seldom exceeds 350 m above sea level (Woodley 1985). Most of the plateau is covered by morainic deposits of till, including silt, sand, gravel, and rubble (Woodley 1985). The most common soil type overlying the glacial till is a stony, Lomond loam. The steeper slopes are well-weathered and partially disintegrated, glacially-molded bedrock.

Most of the study area is underlain by highly deformed volcanic and sedimentary rocks of Precambrian age. Streams located to the west and northwest of the Park (including Dustin Bk.,#15, #40, #41, #21 and #45) are underlain by Precambrian deformed granodiorite, granite and minor gabbro, and the remaining streams are underlain by Precambrian basaltic and rhyolitic volcanics deposited in shallow

Figure 1. Map of New Brunswick, Canada, with Fundy National Park in southeast corner.

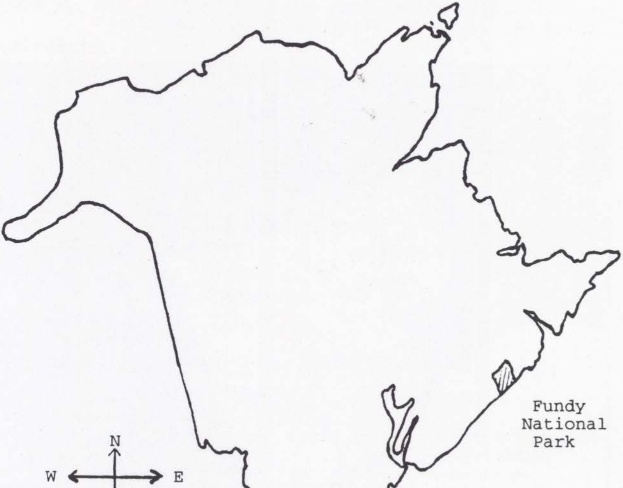
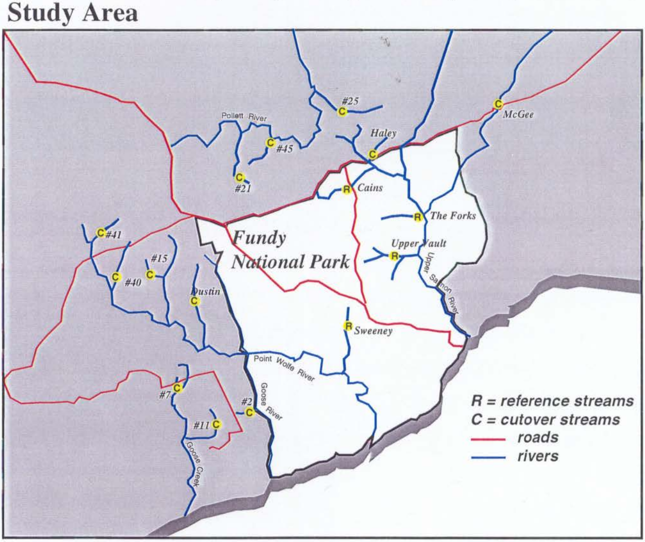


Figure 2. Map of Fundy National Park and surrounding area, including study sites, river systems, and roads.



submarine and subaerial environments (Geological Highway Map of N.B. 1985).

The Park and surrounding clearcuts lie within the Acadian Highlands Natural Region. The forests in the area of study are classified under the Maritimes Uplands Eco-Region of the Sugar Maple-Yellow Birch-Fir Zone. Coastal areas, within 3 or 4 km of the Bay of Fundy, are classified in the Fundy Bay Eco-region of the Spruce-Fir Coast Zone (Woodley 1985). Three streams, Sweeney, #2 and #11, are located in the transitional area between the upland and coastal zones, while the remaining 13 streams are in the Maritime Uplands Eco-Region.

Thirteen of the sixteen streams are located in the Southern New Brunswick climate zone, which is characterized by warm summers, cold winters, and higher precipitation than the Fundy zone. The three streams in the transitional zone may also be affected by the cool summers, mild winters, and high frequency of fog typical of the Bay of Fundy coast (Woodley 1985).

A large portion of the park and surrounding areas have an extensive history of exploitation. The Old Shepody Road was constructed along the northern border of the Park in the 1820s and was settled shortly thereafter. By the 1870s, the Park area was well-settled. Farming and logging were common throughout. Sawmills sprang up along the Point Wolfe and Upper Salmon Rivers. Dams were built along many tributaries of the Point Wolfe, Upper Salmon, and Goose Rivers for the purpose of driving logs downstream to the mills. These log drives "resulted in scouring of the river bed, erosion of the banks and removal of any large woody debris" (Cooper and Clay 1994). By the late 1890s, the forest industry had virtually destroyed the local fishing industry by preventing the migration of anadromous fishes and by polluting the rivers with sawdust, discarded lumber, and bark (Allardyce 1969).

When Fundy National Park was established in 1948, lots along three of the reference streams were owned as freehold lands by the lumber company Hollingsworth and Whitney, having been purchased from the C.T. White company in 1921 (Cooper and Clay 1994). These freehold lands were primarily old farms which had been partially cleared and cultivated at one time, but had since returned to forest (Prov. of N.B., Registry of Deeds). A year prior to 1948, 80 000 cords of pulp were cut from granted lots, mostly along the Shepody Road, and some from within what was to become the Park area.

2.1.0. Site Selection

Sixteen streams were selected from 46 candidate sites on the basis of their size (approximately 3-m wide), accessibility, and disturbance history. For the twelve cutover watersheds, disturbance history was determined in terms of the year of cutting, and presence of a logging road and buffer zone. Also, streams draining beaver ponds and lakes high in dissolved organic acids were avoided. Preliminary estimates of stream bankfull width and channel area allowed for selection of first-order streams 2.5-3.8 m in width (Upper Vault, however, is a third-order stream, although it is similar in width to the others). Within these criteria, streams were selected over a wide range of sites to capture the natural biophysical variation of the landscape.

A 25-m sampling reach was established along each stream. Each reach had at least one riffle (that is, an area of turbulence with a relatively high proportion of gravel substrates), with depth and width similar among all streams. Sampling reaches through clearcuts were sometimes located below a logging road. When a road or trail crossed a reference stream, however, sampling was carried out upstream of that potential disturbance.

2.2.0. Site descriptions

The following brief descriptions are of each study stream and their sampling reach. The descriptions are arranged in order of cutting history, with the most recently cutover stream listed first, followed by the next most recently cutover stream, etc. The reference streams are the last on the list. Information about each stream is summarized in Table 1.

Table 1. Summary table of stream descriptions. The first group includes streams draining watersheds clearcut in the mid-late 1980s; the second group includes streams draining watersheds clearcut in the early 1980s; and the third group, streams draining watersheds clearcut in the 1970s. The forth group includes streams draining reference watersheds.

Stream	Reference (R) or Cutover (C)	Year of cutover	Gradient (%)	Channel Width (m)	Stream Cover (%)	Maximum Temp. (Celsius)	Dominant Shrub Spp.	Dominant Tree Spp.	Snag Density (per ha)	Special features
#45	C	1987	5.4	2.9	74.2	20.0	Be.allenghan. Ac.spicatum	Pi.rubens Be.allenghan.	49	Wide buffer strip Logging road crossing upstream of sampling reach
#2	C	1987	9.1	3.1	64.2	16.0	Be.allenghan. Ab.balsamea	Ab.balsamea Be.papyrifera	32	Steep gradient Small tributary crosses logging road Wide buffer strip
#11	C	Upper watershed 1984, Lower watershed 1987	5.0	3.1	62.8	18.4	Pi.rubens Ab.balsamea	Pi.rubens Ab.balsamea	170	Drains muddy pond Selective harvesting within wide buffer strip Logging road crossing upstream of sampling reach
#7	C	Upper watershed 1989, Mid watershed 1986, Lower watershed 1982	5.5	2.5	29.3	18.0	Ab.balsamea Pi.rubens	Pi.rubens Ab.balsamea	45	Variable buffer strip width, 0 m to >30 m Logging road crosses upstream of sampling reach Several huge, dense debris dams upstream
#21	C	1986	4.0	3.0	70.1	14.5	Al.rugosa	Pi.rubens Be.allenghan.	35	Well-vegetated logging road crossing approximately 0.5 km upstream of sampling reach Medium width buffer strip, 7 m to >30 m
Haley	C	1984	3.4	2.7	47.6	19.5	Al.rugosa Pr.virginiana	Pi.rubens Ab.balsamea	23	Medium width buffer strip, 0 m to > 30 m Small tributary of Haley Brook
#25	C	1984	3.6	3.3	61.9	23.6	Al.rugosa Ac.rubrum	Pi.rubens Be.allenghan.	108	Large portion of the stream flows unprotected through clearcut, then enters minimum 15 m buffer
Dustin	C	1983, early 1900s?	1.6	3.8	25.1	21.0	Al.rugosa Ab.balsamea	Pi.rubens Ab.balsamea	56	Remains of a splash dam found along channel Large sawed stumps in riparian zone, ill-defined buffer
#41	C	Upper watershed 1984, Lower watershed 1980	1.6	2.5	26.6	22.0	Ab.balsamea Al.rugosa	Pi.rubens Ab.balsamea	41	Very low flows in August and September Many budworm killed trees in riparian zone, ill-defined buffer Washed out logging road approx. 1 km upstream of sampling reach
#15	C	1979	4.6	2.7	5.1	23.8	Ab.balsamea Al.rugosa	Be.allenghan. Ab.balsamea	16	Clearcutting to streambanks Stream channel full of logging slash Stream channel may have been used as a skid trail
McGee	C	1978	1.8	3.7	23.9	23.0	Al.rugosa	Al.rugosa Ac.spicatum	16	Clearcutting to streambanks Stream channel may have been used as a skid trail
#40	C	1973	1.8	3.3	11.5	22.0	Al.rugosa Ab.balsamea	Pi.rubens So.americana	75	Uncertain buffer strip with some harvesting in riparian area Logging road washed out directly above sampling reach
Cains	R	1920s	1.7	2.7	40.6	19.5	Al.rugosa	Al.rugosa	27	Portable mill situated on Cains in the 1920s Logs with sawed ends observed in debris dams Riparian vegetation dominated by alders
Sweeney	R		4.8	2.9	74.2	15.3	Be.allenghan. Ac.pensylvan.	Pi.rubens Be.allenghan.	223	Portable mill site situated near headwaters, sometime before creation of National Park Portion of land upstream used to cultivate christmas trees
Upper Vault	R		3.1	3.7	59.2	15.6	Ac.spicatum Pr.virginiana	Be.allenghan. Pi. rubens	104	Portable mill cut taken from the head of Upper Vault in the 1920s.
Forks	R		11.3	3.3	67.0	13.5	Pi.rubens Be.allenghan.	Pi.rubens Be.allenghan.	65	Steep gradient

2.2.1. Cutover streams

2.2.1.1. Clearcut in the mid-late 1980s

(1) <u>Stream #45</u>, to the north of the Park, flowed into the Pollett River (Figure 2). Approximately 100 m upstream of the sampling reach, the stream was channeled through several culverts under an actively eroding logging road. Further up, the stream substrates were dominated by fines and small-sized logging debris. Pools were frequent and the thalwag well-defined. Brook trout (Salvelinus fontinalis) were observed in this stream.

Approximately 90% of the watershed was cut in 1987. Along the sampling reach, the buffer strip was at least 30 m in width (buffer strip widths in Appendix 1). Upstream (about 250 m), buffer width was variable and sometimes quite narrow. In places it appeared to have been selectively harvested.

(2) <u>Stream #2</u>, to the southwest of the Park and approximately 4 km from the coast, descended rapidly into the Goose River (Figure 2). Stream #2 had many pools, large boulders and cobbles, and a large proportion of exposed bedrock.

Ninety five percent of site #2's watershed was clearcut in 1987. The wide buffer strip provided intact forest for at least 30 m on each side of the streambank. A small tributary of stream #2 flowed through a culvert below an actively-eroding logging road.

(3) Stream #11, to the southwest of the Park, drained a muddy pond and flowed southward, then descended rapidly from the upland plateau into Goose Creek (Figure 2). Above the sampling reach, the stream divided into many channels with muddy bottoms, crossed under an actively-eroding logging road through several culverts, and then became ill-defined, pond-like, and boggy.

In 1984, 35% of the watershed was cut, and in 1987, 30%. Along the sampling reach, it was forested on one side, though old sawn stumps were observed within 10 m of the streambank. On the other side of the streambank, there was a wide but variable-sized buffer strip (>20 m), dipping to 6 m at one point. Selective cutting was done within this buffer. Debris dams in the stream channel have logs with sawn ends, indicating either streambank cutting in the past, an old dam, or input of felled trees during road construction upstream.

(4) Stream #7, to the west of the Park, rapidly descended into Goose Creek (Figure 2). A number of huge, dense, debris dams were upstream.

Approximately 35% of this watershed was cut in 1982, 25% in 1986, and 30% in 1989. The buffer zone was greater than 20 m on one side, and ranged from 0-13 m on the other.

In the older cut, stumps were found along stream margins. The sampling reach was situated more than 500 m downstream of a road crossing through a culvert that was at least 12 years old, and still eroding into the stream channel.

(5) Stream #21 was situated north of the Park and flowed into the Pollett River (Figure 2). Upstream, the channel widened and narrowed many times, and the stream bottom was primarily organic fines strewn with small woody debris. There were few pools, and many overhanging banks and pieces of woody debris. Trout were observed in this stream.

Eighty to ninety percent of the watershed of stream #21 was cut in 1986. A buffer strip ranging in width from 7-28 m was left standing. Stream #21 flowed through a culvert under the old Grassy Lake Road, approximately 500 m upstream of the sample reach. The streambanks along the road crossing were well-vegetated.

2.2.1.2. Clearcut in the early 1980s

(6) This tributary of Haley Brook flowed southwest into the Park toward the Upper Salmon River (Figure 2). Despite frequent patches of dry streambed by late summer, trout were observed in several pools, protected by the abundant overhanging vegetation.

In 1984, 70-80% of this watershed was clearcut. The buffer strip was typically greater than 20 m in width, though in places was less than 10 m.

(7) <u>Stream #25</u>, north of the Park, flowed west into the Pollett River, which drains into the Petitcodiac (Figure 2).

Seventy five percent of the watershed was cut in 1984. On the north side of the sampling reach, the forest was uncut. On the south side, the buffer strip varied in width from 14-28 m. Further up, the stream flowed unprotected from the clearcut. Trout were observed in the deep fast flowing section through the cut. Downstream, logging slash in the stream channel formed several debris dams and pools.

(8) <u>Dustin Brook</u>, to the west of the Park, slowly descended the upland plateau and plunged toward the Point Wolfe River (Figure 2). Below the sampling reach, the stream widened and was full of woody debris and debris dams with logging slash.

Stumps in the riparian zone indicated recent selective logging or earlier logging in the area. The latter was suggested by remains of a splash dam in the stream valley. This watershed had been the property of Hollingsworth and Whitney. Most recently, 70-80% of the watershed was clearcut in 1983. The buffer strip was difficult to distinguish; it was usually greater than 20 m in width but consisted of alders with larger trees beyond.

(9) Stream #41, located to the west of #15, gradually descended the upland plateau into the Point Wolfe River (Figure 2). Trout were observed in many of the long, deep pools when other parts of the streambed had dried up. Little cover was provided by large woody debris or overhanging vegetation.

Approximately 30% of the watershed was cut in 1984, and another 40% in 1980. The riparian zone consisted of many spruce- budworm killed trees, such that the stream was protected by a thin forest canopy. Approximately 1 km above the sampling reach, a logging road crossing had been washed out, and was still eroding into the stream channel. An old log bridge crossed the stream above the sampling reach, indicating some past disturbance. Also, old logs with sawn ends were seen in the stream channel.

2.2.1.3. Clearcut in the 1970s

(10) Stream #15, to the northwest of the Park, flowed down the high plateau into the Point Wolfe River (Figure 2). The stream channel was full of logging slash, and the bottom was covered by a layer of dark, organic mud. Debris dams of logging slash were frequent. In some places, the main channel divided into two parallel smaller channels, indicating the probable use of skidders within the stream.

This land was once freehold owned by the lumber company Hollingsworth and Whitney. In 1979, 80-90% of the watershed of stream #15 watershed was clearcut. No buffer strip was left.

(11) McGee Brook, to the northeast of the Park, flowed southward toward the Forty Five River (Figure 2). The sampling reach had many overhanging banks and large cobbles. Upstream, however, there were few deep pools and the channel was unusually straight and shallow. In some places, the stream divided into two evenly spaced channels, suggesting that skidders may have used the stream channel as a trail. Trout were observed in McGee Brook.

While 60-70% of the McGee watershed was cut in 1978, it is likely that it was also cut when the plot granted to J. McGee was sold to Alma Lumber and Shipbuilding Co. in the 1800s. No buffer strip was left along the stream during the 1978 harvest. Sawed stumps were observed in and next to the streambank.

(12) Stream '#40' flowed under Shepody road, to the northwest of the Park, and descended to the Point Wolfe River (Figure 2). Stream #40 had few pools, but abundant woody debris and many cobbles and boulders. Trout were observed in this stream.

Eighty to ninety percent of the watershed of stream #40 was clearcut in 1973. From 70 m above to 70 m below the sampling reach, there was a buffer strip greater than 20 m in width (each side), with evidence of selective harvesting. A logging road was washed out directly above the sampling reach.

2.2.2. Reference streams

(13) Cains Brook, at the north end of the Park, flowed along the wet plateau into Haley Brook (Figure 2). The sampling reach was located approximately 100 m upstream of the Laverty Auto Trail. Cains provided good habitat for brook trout (*Salvelinus fontinalis*), including undercut banks, overhead cover by alders and woody debris, and moderately abundant pools.

Lands originally granted to M. Quigley and F.O. Talbot along Cains were later sold to the Alma Lumber and Shipbuilding Co. (1886), and finally to Hollingsworth and Whitney in 1921 (Registry of Deeds, Albert Co., N.B.). A portable mill was situated on Cains in the 1920s (Cooper and Clay 1994). Logs with sawn ends were observed in debris dams along Cains.

(14) Sweeney Brook, in the southern half of the Park, flowed down the high rolling plateau to the steep river valley of the Point Wolfe River (Figure 2). The sampling reach was located just upstream of the Rat Tail Trail crossing.

Upstream, a plot originally granted to M. Sweeney (Lot

121) was later sold to E. Hogan (1893), C.T. White (1899), and Hollingsworth and Whitney (1921). When the Park was established in 1948, a portion (18.2 hectares) of Hogan's land was being used to cultivate Christmas trees. This area has since been allowed to regenerate naturally. At one time, a portable mill was located near the headwaters of Sweeney Brook (Cooper and Clay 1994).

(15) Upper Vault (U.V.), or Third Vault Brook, was located directly south of the Forks. As this stream descended the plateau, it received several other small streams and flowed toward the Upper Salmon River (Figure 2). The sampling reach was located along a third order stream on the high plateau. Woody debris and moss-covered substrates were abundant, debris dams common, pools and overhanging banks frequent, and channel width highly variable.

A portable saw mill cut was taken from the head of Upper Vault in the 1920s (Cooper and Clay 1994). A splash dam, used to drive logs, was located downstream of the sampling reach (Cooper and Clay 1994).

(16) "The Forks", located in the northeast corner of Fundy National Park, descended rapidly from the high rolling plateau to the steep sloping valley walls of the Broad River (Figure 2). The sampling reach was located near the outlet to Broad River, above a hiking trail crossing. The Forks

had a high frequency of pools and debris dams.

The location of a logging brow close to the outlet of the Forks was the only obvious indication of past disturbance near this stream. Otherwise, the Forks was not obviously disturbed by logging or agriculture.

3.0.0. METHODS

Except when noted (Section 3.6.0) all data were collected within the 25-m sampling reaches.

3.1.0. Water Quality

Temperature

Stream water temperature was measured with five Hobo recording thermographs in each of Sweeney, Upper Vault, Stream s#11, #25 and #15; and with 1 maximum/minimum thermometers in each of the remaining 11 study streams. The temperature recorders were placed in streams from the beginning of June to the middle of November, 1993. The thermometers were read and reset every 4 or 5 days from June to early September, and once a month in October and November. Over that period, the thermographs were downloaded in July, September and November. The thermograph on stream #25 operated well in June, but malfunctioned from July to November. The mean, range, and maximum temperatures for stream #25 in July, August, and September were extrapolated using the temperatures measured in June. This was done by calculating the mean range, overall mean, and mean maximum temperature for June for the four other streams with thermographs, then calculating the percent difference between these and the values for stream #25. The differences were then used to estimate the mean, range and maximum temperatures for stream #25 for the months of July, August and September.

Dissolved oxygen

Dissolved oxygen was measured sporadically in sampling reaches in early September, 1993. Sampling was done using a YSI instrument, with a Clark-Type, membrane-covered polarographic sensor. Due to equipment malfunctions, I was unable to measure dissolved oxygen earlier in the season.

Stream chemistry

Nutrient concentrations were measured in water samples collected from each stream in early July, early August, and early September, 1993. Water for phosphorus analysis was collected in a separate acid-washed container, and the water for remaining nutrients was collected in a 1-litre nalgene bottle. All samples were collected while wearing well-rinsed rubber gloves and refrigerated until the time of analysis. The samples were analyzed by the Inland Waters Directorate (Environment Canada, Moncton) for total nitrogen (including nitrate), phosphate, potassium, chloride, magnesium, calcium, sodium, pH, alkalinity, and specific conductance.

Sedimentation

Sedimentation was measured with three 500 ml nalgene bottles, dug into each sampling riffle such that the mouth, which was 45-mm in diameter, was flush with the streambed. Each bottle, or pot, had 2-3 stones (32-64 mm in diameter) weighting it down. The pots were left in the streambed from June until early October. The contents, excluding the gravel weights, were dried, sieved, and weighed into two categories, fines (< 3.3 mm) and larger (> 3.3 mm).

3.2.0. Substrates

Stream substrates were measured using substrate cores in mid-late June, 1993 and by surface estimation of size frequency in late August, 1993. With the former technique, 5 pairs of double-digit numbers were selected from a random number table. Each was translated into a coordinate and assigned a location in the sampling riffle. This spot was then sampled with a metal cylinder, 15 cm in diameter, that was driven 15 cm into the stream bottom. The substrates within the cylinder were emptied into a bucket. Cobbles more than half the way into the cylinder were placed with the rest of the sample in the bucket. An estimation was made of the percent of the core obstructed by bedrock or boulder.

Substrates in the bucket were wet sieved into the following categories described by Hamilton and Bergensen (1984): large cobble (128-256 mm), small cobble (64-128 mm), very coarse gravel (32-64 mm), coarse and medium gravel (8-

32 mm)[a], and fine gravel (3.3-8 mm)[b]. Each category was weighed with a spring balance. Fines (<3.3 mm in diameter) were stored and brought to the lab, where each sample was air dried and mechanically sorted for 20 minutes in a shaker machine. These substrates were sorted into very fine gravels (2-3.3 mm), coarse and very coarse sands (0.5-2 mm), medium sands (0.25-0.5 mm), fine and very fine sands (0.053-0.25 mm), and silt and clay (<0.053 mm).

Composition of surface substrates was visually estimated along transects placed across the stream at 10, 20 and 40 m above and below the sampling reach, and every 5 m within the sampling reach. Along each transect, three 30 cm x 30 cm quadrats were placed at 1/4, 1/2, and 3/4 the distance across the stream channel. Following a methodology outlined by Hamilton and Bergensen (1984), within each quadrat estimates were made of the most dominant and second most dominant substrate, as well as the percent embeddedness (the degree to which the larger particles are surrounded or covered by sediments < 4 mm in size). The latter is a measurement of how much of the surface area of the largest size particles is covered by fine sediment. Substrates were

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[a] For the substrate cores, coarse and medium gravels were grouped together as one category.

[b] In the substrate core analysis, 'fines' included substrates less than 3.3 mm in diameter. For the estimates of surface substrates, fines included all substrates less than 4 mm in diameter. Both measures are commonly used.

broken down into 11 categories: wood; bedrock; boulder (>256 mm); large cobble (128-256 mm); small cobble (64-128 mm); very coarse gravel (32-64 mm); coarse gravel (16-32 mm); medium gravel (8-16 mm); fine gravel (4-8 mm); and very fine gravel, sand, silt and clay (<4 mm). The (percent) frequency of occurrence of each substrate type was used to show the proportion of the total stream covered by, for example, large cobble. Similarly, the frequency of occurrence of each embeddedness class was used to show the proportion of the streambed reach covered with, for example, 50% fines. In each quadrat, macrophyte and moss cover were also estimated.

Surface estimation of stream substrates differs from the substrate cores in that the former technique estimates surface substrates only, includes wood, separates medium and coarse gravels into two categories, combines all fines into one category, and defines fines as all substrates <4 mm in diameter (whereas in the substrate core analysis, fines included all substrates <3.3 mm, because no 4 mm sieve was available).

Organic matter

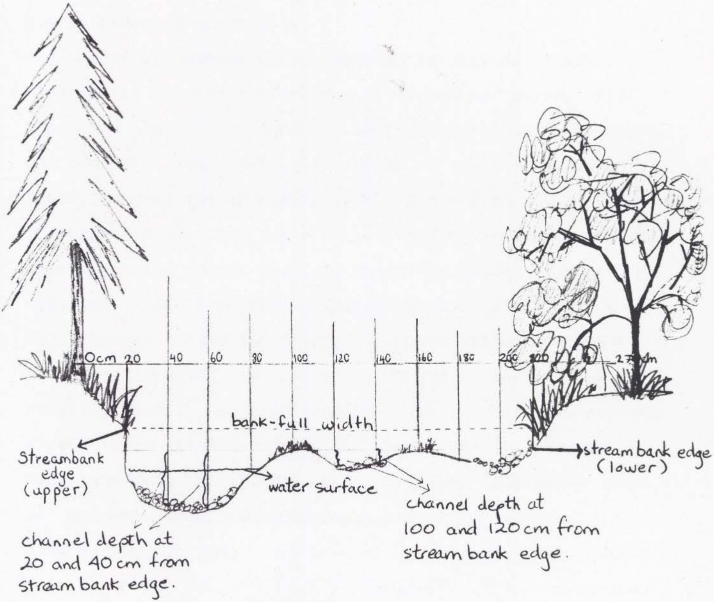
Substrate core samples were used to calculate the organic carbon content of the streambed. Samples taken back to the lab for dry sieving were picked through for pieces of organic matter greater than 2 mm in size. These larger pieces were fed into a grinder then remixed with the portion of each sample less than 2 mm in size. Two subsamples of 2 g each (for a total of 10 per stream) were placed in a beaker and the organic carbon content determined using the Walkley-Black method of wet oxidation (Page *et al.* 1982).

3.3.0. Channel dimensions

Using a technique described by Hamilton and Bergensen (1984), channel measurements were made along transects placed 10, 20, and 40 m above and below the sampling reach, and every 5 m within the sampling reach in early September, 1993. At each sampling point, a measuring tape was secured level above and across the stream channel. At 20 cm intervals from the edge of the streambank, the distance from the measuring tape to the streambed was recorded. The stream channel was defined as the cross-sectional area under water at bank-full depth, which was estimated as the cross-sectional area below the line of permanent vegetation. In the calculation of stream channel cross-sectional area, the distance below the lowest stream edge was subtracted from the vertical distance between the measuring tape and the streambed (Figure 3). Any boulders projecting above the lower stream edge were assigned zero depth. The average channel depth was multiplied by the channel width to derive channel area. Channel width was divided by average channel depth to estimate width:depth ratios for each stream.

Figure 3.

Diagram showing measurements of channel dimensions. A level tape was secured across the stream channel, and the vertical distance from the tape to the streambed recorded at 20 cm intervals along the tape. Stream channel cross-sectional area was approximated as the area below the lower streambank edge. This was calculated by multiplying the bank-full width by the average channel depth.



Channel Area =
(bank-full width) × (average channel depth)

Channel gradient above and along the sampling reach was estimated from a 1:50,000 topographical map by dividing the horizontal distance by the drop in elevation, measured by contour lines.

3.4.0. Riparian vegetation

Riparian vegetation was sampled in July and August, 1993, using the point-centred quarter method (Smith 1980). This is a useful technique for sampling communities in which the dominant plants are large shrubs or trees (Smith 1980). Transects were set up approximately 1.5 m from the streambank along both sides of the stream, beginning at the downstream end of the sampling reach and extending 95 m upstream. Data from the streambank side of the transect were ignored. Sampling points were established every 5 m along the sampling reach, and 10, 20 and 40 m above the sampling reach. The distances from the point to the nearest shrub (<5 cm diameter at breast height [DBH]), tree ($\geq$5 cm DBH), and snag ($\geq$5 cm DBH, >4 m tall) were measured in both 90° sectors around the sampling point on the non-streambank side of the transect. Shrub-sized species and tree species were recorded, and the diameter at breast height of shrubs, trees, and snags. Due to time constraints, plants beyond 15 m of the sampling point were not measured. Plants were measured more than once when they were nearest to more than one sampling point. Following Cox (1990), calculations of basal area per hectare, average dominance, density, relative density, dominance, relative dominance, frequency, relative frequency, and importance value were made for all shrub and tree species. Density was calculated by assigning a distance of 25 m to those plants beyond 15 m from the sampling point (thereby overestimating the density of trees and snags in areas with no buffer strip). Shrub and tree diversity were calculated using Brillouin's Index of species diversity, following the methodology described by Krebs (1989) in Ecological Methodology.

Riparian canopy cover was estimated at each sampling point. A funnel was positioned approximately 1 m above the ground, and an estimate of overhanging vegetation, or cover, was made by looking up through the funnel. Overhead cover was estimated at each quarter distance across the stream channel, along the same 12 transects set up for channel measurements.

3.4.1. Buffer strip width

Buffer strip width was measured along streams draining cutovers at each of the transects set up for channel measurements. Width was measured to a maximum of 30 m from the streambank in either direction.

3.5.0. Stream Invertebrates

Stream invertebrates were sampled using 'rockballs', which are rounded mesh bags, 10-15-cm in diameter and 5-10-cm high, with 6-mm wide holes. Following a methodology described by D. Eidt (in verbis 1993), approximately 1 kg of medium to coarse gravels were used per ball. Five rockballs were distributed in the sampling riffle of each stream. Each rockball was partially buried in the streambed and fully submerged. The rockballs were left there from early October to mid-November, 1993, to avoid the emergence of insects in the summertime. At the end of 6 weeks, rockballs were removed with a fine-meshed D net, and later untied, rinsed, the gravel removed, and the invertebrates and organic detritus separated with a sieve and preserved in 70% alchohol. Some months later, these samples were rinsed in a 300 μm sieve and placed in an enamel tray. All macroinvertebrates were picked from the tray with forceps and identified to order only, except for Diptera, which were identified to family. Merritt and Cummins (1984) was used to identify the invertebrates. The average abundance of each identified group was calculated for each stream.[c]

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[c] The invertebrate samples from the The Forks and one from Cains Brook were mistakenly rinsed with a 500 μm sieve, resulting in the loss of organisms smaller than 500 μm. The remaining 4 samples from Cains were rinsed with both the 500 μm and 300 μm sieves, and the proportion retained by the 300 μm sieve calculated and used to derive the approximate number of each group of organisms in the first sample. Using these same proportions, approximate numbers for The Forks were derived, however little value was placed in these numbers.

Rockballs provide a homogeneous, artificial habitat that reduces variability, thereby facilitating comparisons of different streams and enhancing statistical analyses (D. Giberson pers. comm. 1993). However, rockballs do not give as true a representation of the stream fauna as do Surber samplers, which require stirring up the substratum upstream of a fine-meshed net. Instead, rockballs attract opportunistic species that are able to rapidly colonize new gravel habitats.

3.6.0. Other observations

Each stream was checked for upstream disturbances by exploring for a distance of approximately 1 km. Descriptions were made of the buffer strip, substrates, riparian vegetation, pool:riffle ratio, flow rate, abundance of large organic debris, and available salmonid cover.

3.7.0. Data Analysis

Two approaches were taken in the analysis of the data. The first approach assumed a univariate data set, to allow examination of changes in variables along the chronosequence of streams (from the stream draining the most recent clearcut, to the oldest clearcut, to the reference streams). Regressions were employed to test the ecological "recovery" of cutover streams, and independent-samples t-tests were used to demonstrate differences between the group of 4 reference streams and the 12 cutover streams. The second approach assumes no differences between reference and cutover streams. Multivariate ordination techniques were used to cluster streams on the basis of patterns of covariation of environmental and invertebrate data, allowing inferences to be made about disturbance history.

Each of the measured variables was tested for normality using a probability plot. If a transformation (log, inverse, arcsine, or square root) made a variable more normal in distribution, the transformed state was used in the multivariate analysis. However, some variables could not be normalized in this way.

3.7.1. Univariate approach

Single, forward-stepwise and multiple regressions were performed on variables suspected of having an influence on other variables. In each case, the assumptions of the regression model were checked. For the forward stepwise regression, alpha-to-enter and alpha-to-remove were set at 0.05.

Independent samples T-tests were performed on variables that showed a pattern over the chronosequence of stands. Sites were divided into two groups: the four park streams and the 12 cutover streams. Where variances were unequal, the separate variances t-value was used. When the variances of the two groups were similar, the pooled variance t-value was used (Wilkinson 1990).

3.7.2. Multivariate approach

Principal components analysis was used to simplify the data matrix. The temperature, water quality, vegetation and substrate data were analysed separately. Redundant variables were identified and removed from a final set of variables. For each analysis, an ordination was made of the stream sites.

Discriminant function analysis was used to check for significant groupings of sites based on environmental data.

Correspondence analysis was employed to graphically position streams based on the invertebrate data alone, with no prior grouping of sites.

A final set of 12 environmental variables, 20 invertebrate taxa, and 16 stream sites was analysed with an exploratory canonical correspondence analysis (CANOCO). This technique was useful when trying to identify the environmental variables responsible for the distribution and abundance of taxa. The technique also allows for the ordination of species, environmental variables, and sites on the same graph. The forward selection option of CANOCO was utilized to select those variables that accounted for a significant ($P < 0.05$) additional proportion of the total variance. Significance was assessed on the basis of the Monte Carlo permutation test (Walker *et al.* 1991).

A Pearson correlation matrix of all physical and chemical variables was used to identify highly correlated variables. There were too many variables to test all correlations, so two smaller matrices were calculated in order to derive significance values. The variable 'age of stand' was calculated as the number of years before 1993 that the watershed was most recently cutover. Cains, which was cutover in the 1920s, was assigned a value of 70 years, while the other park streams were assigned values of 90 years. Watersheds clearcut in stages were given an average age calculated from the size and date of each partial cut.

4.0.0. RESULTS

4.1.0. Habitat data

4.1.1. Temperature

Water temperatures for streams #11, 15, 25, Sweeney Brook and Upper Vault Brook were measured with Hobo thermographs (for raw data, see Appendix 2). The maximum temperature for the references Sweeney Brook and Upper Vault Brook did not exceed 15.6°C (Table 2). The daily temperature range for Sweeney Brook never exceeded 4°C, and for Upper Vault exceeded 4°C five out of 161 days (3% of the total recorded days) (Table 3). For stream #11, partially cut with a wide buffer strip, the daily temperature range exceeded 4°C three out of 154 days, or 2% of the total recorded days, and the daily maximum temperature exceeded 17°C 5% of the total recorded days. For stream #15, with no buffer, the daily temperature range exceeded 4°C 55% of the total days, and the daily maximum temperature exceeded 17°C, 34% of the total days. The highest maximum temperature recorded on all streams was 23.8°C, on stream #15 (Table 2).

Water temperatures of streams #45, 2, 7, 21, Haley, Dustin, 41, McGee, 40, Cains and the Forks were recorded with maximum/minimum thermometers (See Appendix 3). The 4-5 day range exceeded 4°C more than 80% of the time for streams #45, #41, #40, Dustin, and Haley (cutover in the 1980s), McGee (cutover in 1979), and the reference, Cains; 60% of

Table 2. Monthly temperature records for streams #11, 15 and 25, Sweeney Brook and Upper Vault Brook, with the daily mean, maximum, minimum and mean range (in degrees Celsius).

	June	July	August	September	October	November
STREAM #11 (cut 1984, 1987)						
Mean	12.4	14.6	14.7	12.0	6.9	4.4
Maximum	16.1	18.4	18.0	15.6	10.9	7.7
Minimum	7.2	11.1	8.9	6.8	2.7	1.7
Mean Range	2.4	1.8	2.6	1.8	1.7	1.5
STREAM #15 (cut 1979)						
Mean	10.4	13.8	15.5	12.0	6.5	4.6
Maximum	17.4	21.4	23.8	21.6	11.3	7.5
Minimum	5.2	8.1	1.8	4.6	3.2	2.5
Mean Range	5.1	5.8	9.2	5.8	2.4	1.5
STREAM #25 (cut 1984)						
Mean	12.5	15.5	16.6	13.6	6.5	3.7
Maximum	19.0	22.3	23.6	21.0	10.9	8.1
Minimum	6.8				2.2	0.6
Mean Range	4.4	4.1	5.8	4.1	2.3	2.0
Sweeney Brook (reference)						
Mean	9.3	11.5	12.6	11.0	6.6	4.9
Maximum	12.0	14.1	15.3	14.1	10.2	7.5
Minimum	6.0	8.8	9.7	6.6	4.1	3.0
Mean Range	1.8	1.4	1.9	1.6	1.4	1.0
Upper Vault Brook (reference)						
Mean	9.5	11.8	12.5	10.4	5.8	4.4
Maximum	13.3	14.8	15.6	13.4	9.7	7.5
Minimum	5.5	8.3	9.7	6.2	2.7	1.9
Mean Range	2.8	2.0	2.1	1.9	1.9	1.4

Note: July, August and September thermograph data for Stream #25 were extrapolated using June data and the relative values of other streams

Table 3. Thermograph data, with the proportion of days the temperature range exceeded 4C, and the proportion of days the maximum exceeded 17C. These calculations could not be made for Stream #25 as the thermograph malfunctioned from July to September.

STREAM	#11	#15	Sweeney	Upper Vault
Days Range >4C (%)	2	55	0	3
Days Temp >17C (%)	5	34	0	0

the time for stream #7 (cutover 1982-1989); 40% for #21 (cutover in 1986); 25% for #2 (cutover in 1987); and 10% for the reference, Forks (Table 4). The mean 4-5 day range was greatest for McGee at 9.5°C, with no buffer strip, followed by #40 at 8.1°C, Dustin at 7.8°C, and #41 at 7.7°C. The lowest mean 4-5 day range was recorded on the reference stream, Forks, at 3.1°C, followed by #2 at 3.6°C. Stream temperature did not once exceed 17°C for #21, #2 and the Forks, whereas it was equal to or exceeded 19°C more than 35% of the time for Dustin, #41, #40, and McGee. The highest maximum temperatures recorded were 23°C for McGee, 22°C for #40 and #41, 21°C for Dustin, and 20°C for #45. The lowest maximum temperatures were 13.5°C for the Forks, 14.5°C for #21, and 16°C for #2.

In summary, three of the reference streams (The Forks, Upper Vault, and Sweeney) tended to have smaller fluctuations in temperature and lower maximum temperatures than streams draining clearcuts. Streams draining partially cutover watersheds, or with wide buffer strips along their length, had similar thermal regimes to the reference watersheds (especially stream #2). Where there were no buffer strips (stream #15 and McGee), high maximum temperatures were reached, and diurnal fluctuations were large. Cains, a reference stream, behaved more like the older cuts than the other 3 reference streams, possibly as a consequence of past logging in the 1920s, or some other more

Table 4. Maximum / minimum thermometer data (in degrees Celsius), for streams #45, #2, #7, #21, Haley, Dustin, #41, McGee, #40, and the reference streams, Cains and the Forks.

STREAM	#45	#2	#7	#21	Haley	Dustin	#41	McGee	#40	Cains	Forks
Cutting history	cut 1987	cut 1987	cut 1982, 1986,1989	cut 1986	cut 1984	cut 1983	cut 1984, 1980	cut 1978	cut 1973	reference	reference
Maximum	20.0	16.0	18.0	14.5	19.5	21.0	22.0	23.0	22.0	19.5	13.5
Minimum	5.0	6.0	8.5	5.0	4.5	5.0	5.0	1.5	4.0	5.0	4.5
Mean (May-October)	11.4	11.5	13.5	9.8	12.1	13.2	14.2	13.3	13.7	13.2	9.7
Mean (August)	12.8	12.6	14.5	10.6	13.2	15.2	16.3	15.0	15.5	14.3	11.3
Mean Range	6.2	3.6	4.7	4.4	7.5	7.8	7.7	9.5	8.1	6.8	3.1
Times 4-day Range >4C (%)	84.0	25.0	60.0	40.0	86.0	94.0	95.0	100.0	95.0	90.0	10.0
Times Temp >17C (%)	5.0	0.0	25.0	0.0	27.3	47.4	50.0	60.0	54.6	30.0	0.0

recent disturbance.

4.1.2. Dissolved oxygen

Malfunctioning equipment delayed recording of dissolved oxygen to early September, before the fall rains but after the highest recorded summer temperatures. The lowest O_2 concentrations were observed in streams #41 and #15 (7.7 and 8.2 mg/l, respectively) (Table 5). The latter stream was full of logging slash, which may have obstructed flow, as well as depleted available oxygen. Furthermore, with no overhead canopy cover, #15 reached relatively high water temperatures, further reducing O_2 concentrations. Stream #41, on the other hand, had dried up in many places, except where some deep pools remained. This stream also had poor shading. High temperatures and near absent streamflow could have contributed to the low dissolved oxygen concentrations.

4.1.3. Water chemistry

Total nitrogen (N) in stream water had a strong negative correlation with age of stand, demonstrated by a Pearson $r = -0.80$, $p < 0.001$ (Appendix 4), and a highly significant simple regression (F-ratio = 10.7, $p = 0.006$) (Figure 4). For the four park streams, total nitrogen did not exceed 0.21 mg/l, and the concentrations increased slightly among sampling dates as the growing season progressed (Figure 5). A T-test with separate variances

Table 5. The date, time and water temperature of dissolved oxygen readings, as well as percent saturation.

Stream	45	45	2	11	7	21	Haley	25	Dustin	41	41	15	40	40	Cains	Sween	U.V.
Date	04/09/93	08/09/93	06/09/93	06/09/93	06/09/93	04/09/93	05/09/93	05/09/93	03/09/93	03/09/93	08/09/93	06/09/93	03/09/93	08/09/93	05/09/93	05/09/93	04/09/93
Time	10:15 AM	10:20 AM	10:40 AM	01:30 PM	02:03 PM	10:50 AM	01:10 PM	12:00 PM	04:52 PM	06:15 PM	03:30 PM	05:40 PM	05:30 PM	12:05 PM	01:35 PM	03:05 PM	12:10 PM
Temperature (Celsius)	13.5	11.0	12.0	15.1	15.5	11.4	14.0	12.5	13.0	13.8	16.5	20.2	13.3	15.5	15.5	13.7	12.2
Dissolved Oxygen (mg/l)	9.7	11.1	10.6	9.5	9.4	10.2	9.8	9.2	9.6	7.7	9.0	8.2	10.0	10.4	9.6	9.7	9.8
Saturation (%)	96.0	106.0	101.0	97.0	97.0	97.0	99.0	90.0	95.0	76.0	95.0	92.0	98.0	103.0	100.0	96.0	95.0

Figure 4. Mean nitrogen in stream water for 3 collection dates: July 1, August 4, and September 6, 1993. Streams #45 - #21 drained watersheds clearcut in the mid-late 1980s; Haley - #41 drained watersheds clearcut in the early 1980s; #15 - #40 drained watersheds clearcut in 1970s; and Sweeney - Upper Vault were reference streams.

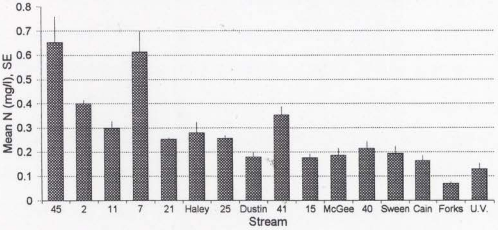
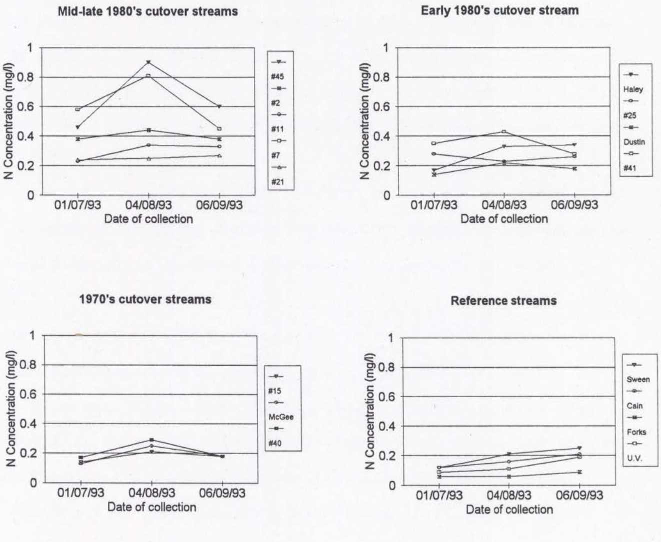


Figure 5. Total nitrogen in stream water for three collection dates: July 1, August 4, and September 6, 1993, for all 16 streams.



indicated a significant difference in N concentrations between the four reference streams and the twelve streams draining clearcuts ($\bar{x}_4$ = 0.14 and $\bar{x}_{12}$ = 0.32; t = 3.4, p = 0.004). The streams draining early 1980s cuts had N concentrations of 0.14 to 0.43 mg/l, with large variations in concentration among sampling dates (Figure 5). The streams draining the five most recent cuts had minimum N concentrations of 0.23 to 0.45 mg/l. Stream #45, cut in 1987, and #7, most recently cut in 1989, had N concentrations of 0.45 mg/l to 0.90 mg/l. Both of these streams had much larger concentrations in early August than in July or September - a pattern shared to various degrees by most of the study streams. Total nitrogen included, and was strongly correlated with nitrate-N (Pearson r = 0.95), and had a similar pattern over the chronosequence of streams (Figure 6).

Magnesium concentration in streamwater was greater in recently disturbed streams (Figure 7) (significant negative correlation with the age of stand; Pearson r = -0.54, p = 0.03), and showed a significant difference in concentration among the four park streams and the twelve cutover streams (independent samples T-test, $\bar{x}_4$ = 0.43 and $\bar{x}_{12}$ = 0.55; t = 2.8, p = 0.014). For the four park streams, Dustin Brook and #11, dissolved Mg did not exceed 0.54 mg/l, and varied little in concentration over the three sampling dates (Figure 8). The streams draining the most recent cuts

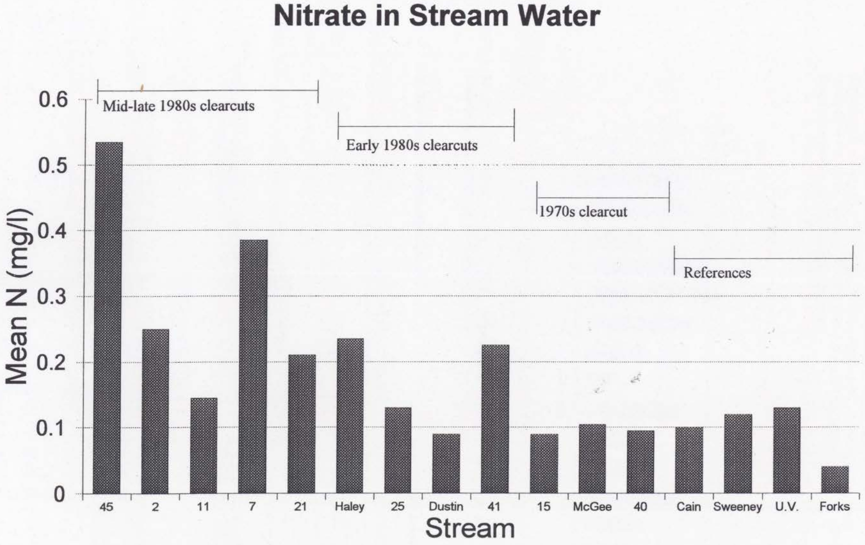


Figure 6. Nitrate concentrations in stream water. Samples collected in July and August, 1993. Stream #45 drained the most recently clearcut watershed, and #40 the oldest clearcut watershed. Cains, Sweeney, Forks and Upper Vault were reference streams.

Figure 7. Mean magnesium in stream water for 3 collection dates. Streams #45 - 21 drained watersheds clearcut in the mid-late 1980s; Haley - #41 drained watersheds clearcut in the early 1980s; #15 - #40 drained watersheds clearcut in the 1970s; and Cain - Forks were reference streams.

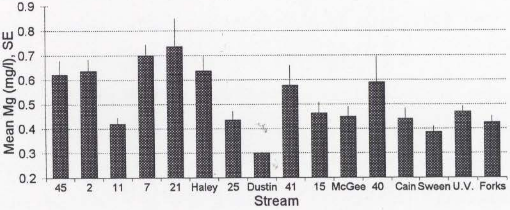
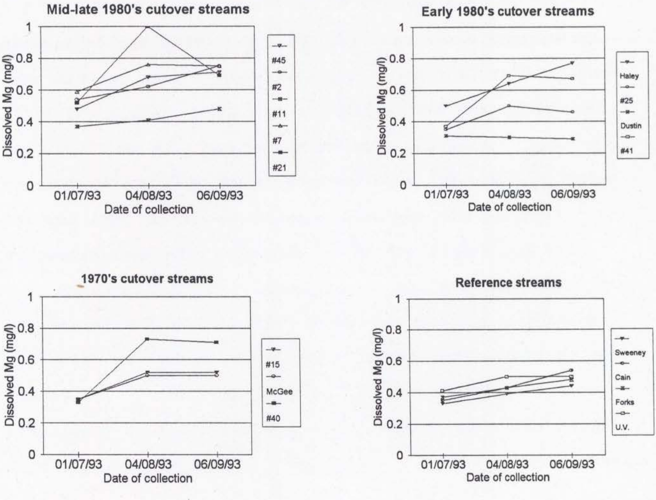


Figure 8. Magnesium in stream water for three collection dates: July 1, August 4, and September 6, 1993, for all 16 streams.



tended to have larger and less stable Mg concentrations. Stream #25, McGee and #15 had comparable concentrations to the reference streams, but they fluctuated more over the three sampling dates. The concentration of dissolved Mg tended to increase over the summer, probably as a consequence of low summer streamflows.

Potassium (K) showed a similar pattern to magnesium and total nitrogen (Figures 9 and 10). Dissolved K had a strong negative correlation with age of stand (Figure 9) (Pearson r = -0.747, p = 0.001), and showed a significant difference in concentration between the park and cutover streams (independent samples T-test, $\bar{x}_4$ = 0.14 and $\bar{x}_{12}$ = 0.22; t = 3.4, p = 0.005). The 4 reference streams did not have K concentrations exceeding 0.21 mg/l, and the 5 most recent cuts did not have K concentrations below 0.20 mg/l (Figure 10). The highest potassium concentrations recorded were 0.41 mg/l, on #7. Again, streamwater nutrient concentrations were least variable over the sampling dates for the four reference streams, although the largest concentrations were recorded on the third sampling date.

Mean calcium (Ca), specific conductance, and alkalinity all showed similar patterns among the 16 streams (Figure 11), with an increase in value over the three sampling dates (Table 6). Like total N, the specific conductance and alkalinity of the reference streams fluctuated only slightly from one sampling date to the next. Cains, however, behaved

Figure 9. Mean potassium in stream water for 3 collection dates. Streams #45 - 21 drained watersheds clearcut in the mid-late 1980s; Haley - #41 drained watersheds clearcut in the early 1980s; #15 - #40 drained watersheds clearcut in the 1970s; and Cain - Forks were reference streams.

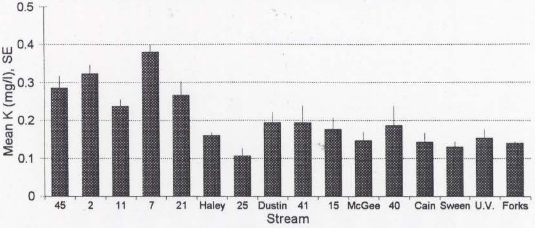


Figure 10. Potassium in stream water for three collection dates: July 1, August 4, and September 6, 1993, for all 16 streams.

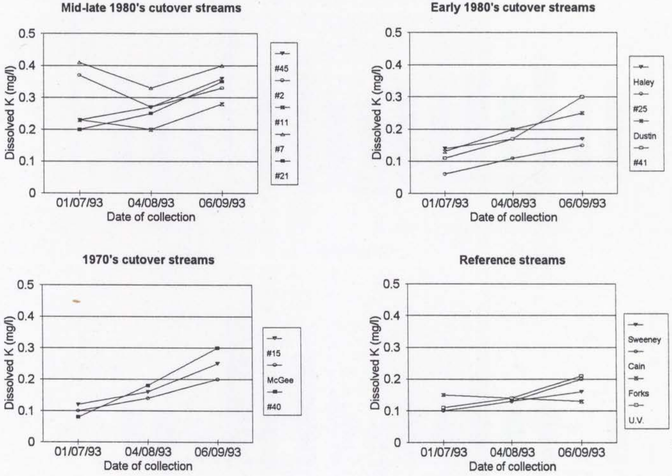
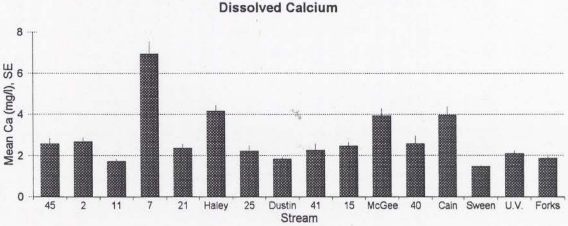
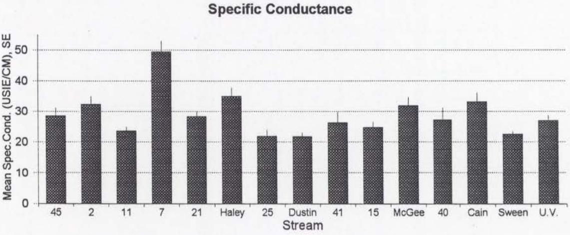


Figure 11(a)-(c). Mean calcium, specific conductance and alkalinity of stream water for three collection dates: July 1, August 4 and September 6, 1993. Streams #45 - 21 drained watersheds clearcut in the mid-late 1980s; Haley - #41 drained watersheds clearcut in the early 1980s; #15 - #40 drained watersheds clearcut in the 1970s; and Cain - Forks were reference streams.

11(a)



11(b)



11(c)

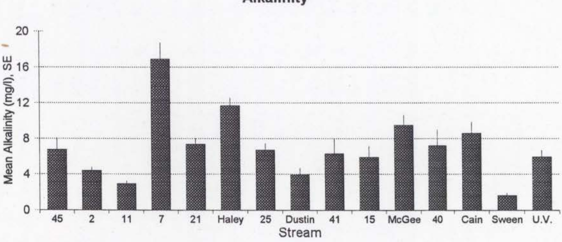


Table 6. Water chemistry measures for the the chronosequence of stands. Samples were collected over three periods: 1. 25/06/93 - 01/07/93 2. 03/08/93 - 06/08/93 3. 03/09/93 - 06/09/93.

STREAM	Date of Collection	Colour REL UNIT	Specific Conductance USIE/CM	Dissolved Organic Carbon Mg/L	Nitrate Mg/L	Total Nitrogen Mg/L	Alkalinity Mg/L	pH PH UNIT	Sodium Mg/L	Magnesium Mg/L	Total Phosphorus Mg/L	Sulphur Mg/L	Chloride Mg/L	Potassium Mg/L	Calcium Mg/L	Anions - Cations %	Organic Ions ueq/L
45	1	45.00	23.00	3.40		0.46	3.70	6.60	1.60	0.48	0.008	2.30	1.30	0.23	2.00	15.14	32.70
45	2	20.00	28.50	2.50	0.58	0.90	7.10	6.90	2.10	0.54	0.005	2.40	1.40	0.27	2.50	1.15	24.37
45	3	20.00	34.00	2.50	0.49	0.60	9.40	7.00	2.40	0.37	0.005	3.30	1.50	0.36	3.20	0.31	24.46
2	1	45.00	28.40	4.90		0.38	3.70	6.80	2.30	0.59	0.008	4.50	1.90	0.37	2.30	9.72	47.59
2	2	30.00	29.50	4.80	0.25	0.44	4.10	6.60	2.30	0.52	0.006	4.60	2.00	0.27	2.50	5.79	46.17
2	3	15.00	39.00	4.80	0.25	0.38	5.40	6.90	2.90	0.50	0.004	6.60	2.40	0.33	3.20	3.67	46.80
11	1	40.00	21.80	4.40		0.23	2.10	6.40	1.90	0.35	0.013	3.00	2.00	0.23	1.50	9.40	41.81
11	2	40.00	22.00	4.50	0.14	0.34	3.10	6.40	1.90	0.31	0.003	2.70	1.90	0.20	1.90	8.77	42.76
11	3	15.00	27.00	3.80	0.15	0.33	3.60	6.70	2.70	0.37	0.005	3.00	2.30	0.28	1.80	9.51	36.74
7	1	40.00	43.50	4.10		0.58	13.30	7.30	1.90	0.35	0.007	3.00	1.80	0.41	5.80	6.44	40.43
7	2	30.00	46.90	3.60	0.44	0.81	16.40	7.30	2.10	0.35	0.007	2.80	1.60	0.33	6.80	4.06	35.50
7	3	15.00	58.00	3.30	0.33	0.45	21.00	7.60	2.40	0.33	0.008	3.40	1.80	0.40	8.30	2.23	32.71
21	1	35.00	25.00	2.40		0.24	5.90	6.80	1.90	0.33	0.005	2.60	1.50	0.20	1.90	2.54	23.31
21	2	10.00	27.90	1.90	0.20	0.25	7.30	6.90	2.10	0.35	0.002	2.60	1.60	0.25	2.40	7.24	18.52
21	3	5.00	32.00	1.50	0.22	0.27	8.90	7.00	2.30	0.37	0.002	2.70	1.80	0.35	2.80	0.83	14.67
Haley	1	35.00	28.60	3.60		0.17	9.50	7.10	1.80	0.41	0.004	2.70	0.90	0.14	3.60	5.46	35.33
Haley	2	10.00	35.10	3.00	0.17	0.33	13.00	7.10	2.10	0.64	0.002	2.80	0.90	0.17	4.10	0.36	29.44
Haley	3	10.00	41.00	2.20	0.30	0.34	12.60	7.30	2.40	0.77	0.002	3.70	1.50	0.17	4.80	2.38	21.69
25	1	95.00	16.80	7.20		0.28	4.90	6.50	1.20	0.35	0.013	1.10	0.50	0.06	1.60	14.77	68.86
25	2	35.00	22.70	6.40	0.07	0.23	8.00	6.80	1.70	0.50	0.008	1.00	0.70	0.11	2.50	8.34	62.15
25	3	15.00	26.00	5.90	0.19	0.26	7.20	6.80	2.10	0.46	0.005	2.00	1.90	0.15	2.60	1.99	57.30
Dustin	1	60.00	18.40	4.10		0.14	2.00	6.30	1.60	0.31	0.003	2.60	1.60	0.13	1.60	12.43	38.68
Dustin	2	30.00	22.80	3.00	0.08	0.22	4.60	6.80	1.90	0.30	0.002	2.40	1.90	0.20	1.90	1.52	29.13
Dustin	3	10.00	24.00	2.20	0.10	0.18	5.20	6.80	2.20	0.29	0.001	2.40	1.90	0.25	2.00	2.57	21.36

Table 6. Continued.

STREAM	Date of Collection	Colour REL UNIT	Specific Conductance USIE/CM	Dissolved Organic Carbon Mg/L	Nitrate Mg/L	Total Nitrogen Mg/L	Alkalinity Mg/L	pH PH UNIT	Sodium Mg/L	Magnesium Mg/L	Total Phosphorus Mg/L	Sulphur Mg/L	Chloride Mg/L	Potassium Mg/L	Calcium Mg/L	Anions - Cations %	Organic Ions ueq/L
41	1	30.00	18.30	3.40		0.35	2.30	6.40	1.40	0.37	0.005	2.50	1.20	0.11	1.50	12.46	32.31
41	2	5.00	27.80	1.70	0.25	0.43	7.90	7.10	2.00	0.69	0.003	2.50	1.40	0.17	2.50	1.05	16.68
41	3	5.00	33.00	2.50	0.20	0.28	8.70	6.70	2.30	0.67	0.010	3.30	1.70	0.30	2.80	0.33	24.17
15	1	60.00	20.10	4.10		0.14	3.00	6.50	1.50	0.35	0.007	2.70	1.80	0.12	2.00	8.32	39.21
15	2	45.00	26.20	3.00	0.05	0.21	6.50	6.80	1.90	0.52	0.005	2.60	2.00	0.16	2.60	3.06	29.13
15	3	30.00	28.00	3.80	0.13	0.18	8.20	6.90	2.10	0.52	0.008	2.80	2.00	0.25	2.80	1.31	37.05
McGee	1	50.00	25.60	3.20		0.13	6.80	7.00	1.60	0.35	0.007	3.10	1.30	0.10	3.20	4.75	31.31
McGee	2	40.00	31.70	3.60	0.10	0.25	9.80	7.00	1.90	0.50	0.006	2.90	1.40	0.14	3.90	3.08	35.22
McGee	3	25.00	38.00	2.50	0.11	0.18	11.80	7.10	2.10	0.50	0.009	3.00	1.60	0.20	4.70	2.90	24.53
40	1	65.00	18.00	4.50		0.17	2.80	6.40	1.40	0.33	0.008	2.20	1.40	0.08	1.60	9.34	42.76
40	2	30.00	29.80	3.40	0.10	0.29	9.00	7.00	1.90	0.73	0.012	2.40	1.70	0.18	2.80	0.39	33.26
40	3	20.00	34.00	2.60	0.09	0.18	9.80	7.10	2.30	0.71	0.008	2.90	1.90	0.30	3.30	2.27	25.51
Cains	1	40.00	26.90	3.10		0.12	5.80	6.80	1.60	0.35	0.002	3.40	1.70	0.10	3.10	4.32	30.11
Cains	2	25.00	32.20	2.80	0.09	0.16	8.50	7.00	1.90	0.43	0.002	3.50	1.70	0.13	3.90	3.09	27.39
Cains	3	15.00	40.00	2.50	0.11	0.21	11.40	7.10	2.20	0.54	0.003	4.10	2.00	0.20	4.90	1.62	24.53
Sweeney	1	35.00	20.60	2.80		0.12	0.90	6.10	1.90	0.33	0.005	3.20	2.50	0.10	1.30	6.87	25.96
Sweeney	2	40.00	21.90	3.30	0.10	0.21	2.10	6.20	2.20	0.39	0.003	3.00	2.60	0.13	1.50	5.54	30.88
Sweeney	3	20.00	25.00	3.60	0.14	0.25	2.00	6.30	2.30	0.44	0.003	3.80	2.60	0.16	1.60	4.31	33.96
Forks	1	10.00	22.10	1.30		0.06	2.40	6.50	1.70	0.37	0.002	3.70	2.30	0.15	1.70	0.91	12.43
Forks	2	5.00	23.20	1.90	0.03	0.06	3.10	6.60	1.90	0.43	0.001	3.80	2.30	0.14	1.80	0.86	18.28
Forks	3	5.00	25.00	1.40	0.05	0.09	3.80	6.70	2.10	0.48	0.001	3.70	2.20	0.13	2.10	4.49	13.54
Upper Vault	1	40.00	23.60	2.30		0.09	4.60	6.80	2.00	0.41	0.004	2.80	1.90	0.11	1.70	1.15	22.34
Upper Vault	2	10.00	26.40	1.20	0.11	0.11	5.70	6.90	2.40	0.50	0.004	2.90	2.10	0.14	2.20	3.53	11.70
Upper Vault	3	10.00	31.00	1.10	0.15	0.19	7.60	7.00	2.30	0.50	0.012	2.70	2.00	0.21	2.40	1.62	10.76

less like the other park streams and more like the cutover streams. The highest values of all 3 variables were observed from watershed #7, which was most recently cut in 1989.

Total phosphorus (P) concentrations in stream water were extremely variable amongst streams, and tended to fluctuate considerably from one sampling date to the next (Figure 12a, Table 6). In general the park streams appeared to have smaller, more stable phosphorus concentrations, except for a sudden increase of P in Upper Vault on the September sampling date. No consistent pattern in concentration over the three sampling dates was observed.

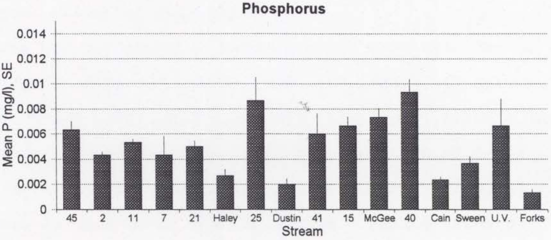
Dissolved chloride (Cl) concentrations are shown in Figure 12b. The three streams sites closest to the Fundy coast (Sweeney, #2 and #11) (Figure 2), had some of the largest chloride concentrations, suggesting that the sea had some influence on stream ion content.

4.1.4. Sedimentation

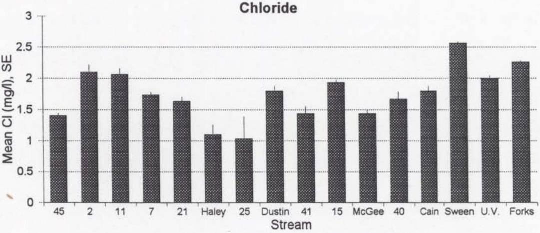
Some of the sediment pots buried into the stream bottom were not found again, presumably because they were covered by substrates, or possibly dislodged during the heavy autumn flows. Three pots, one each from Haley, #15 and #40, were found empty on the streambank beside where they had been buried - presumably emptied by curious passers-by. Where it was known that pots had been removed, the remaining pots

Figure 12(a)-(b). Mean phosphorus and chloride of stream water for three collection dates: July 1, August 4 and September 6, 1993. Streams #45 - 21 drained watersheds clearcut in the mid-late 1980s; Haley - #41 drained watersheds clearcut in the early 1980s; #15 - #40 drained watersheds clearcut in the 1970s; and Cain - Forks were reference streams.

12(a)



12(b)



were used to estimate the rate of sedimentation. Three lost pots in #25, and one lost pot in McGee and #40 were assumed full, and assigned an approximate weight based on the known weight of full pots.

The highest rates of sedimentation were recorded on #2 (478 g/pot), #40 (437 g/pot), #25 (400 g/pot), #7 (177 g/pot), McGee (180 g/pot), and #45 (118 g/pot) (Figure 13). The latter had a particularly large proportion of fines (<3.3 mm). The sampling reaches of streams #2, #40, #7 and #45 were located below road crossings. Medium rates of deposition were observed on streams #41 (65.9 g/pot) and Haley (40.2 g/pot). The greatest quantity of sediment recorded in any reference stream was from Cains, in which one pot had accumulated more than 28 g (21.7 g >3.3 mm; 6.4 g <3.3 mm) between June and October (Figure 13). The other park streams had no more than 7.5 g of sediment in any one pot.

T-tests for the deposition of sediment <3.3 mm and >3.3 mm showed significant differences between the cutover streams and the park streams ($t = 3.1$, $p = 0.01$ for <3.3 mm; $t = 2.6$, $p = 0.025$ for >3.3 mm). The Pearson correlation matrix also showed a significant negative relationship between the deposition of fines (<3.3 mm) and the age of stand (inverse transformed) ($r = 0.62$, $p = 0.011$), and a simple regression showed a significant linear relationship between the amount of sediment (<3.3 mm) and the age of the

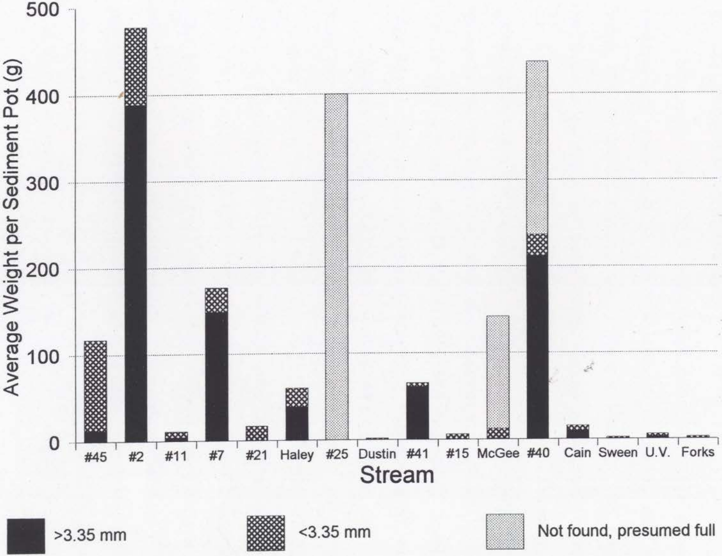


Figure 13. Sedimentation (deposition of materials <4.5 cm) measured with sediment pots placed in each stream from June to Ocober, 1993. Streams #45 - 21 drained watersheds clearcut in the mid-late 1980s; Haley - #41 drained watersheds clearcut in the early 1980s; #15 - 40 drained watersheds clearcut in the 1970s; and Cains - Forks were reference streams. Lost bottles were assumed to have been buried and were assigned the weight of full pots.

stand (F = 9.2, p = 0.009).

4.1.5. Streambed composition

The results of the substrate core analysis showed that medium and very coarse gravels, small cobbles, and bedrock were the dominant substrates (Figure 14a-d). Of the four reference streams (Figure 14d), Cains appeared to be anomalous, with no exposed bedrock and a relatively large proportion of fines (12.4%), and medium and very coarse gravels being the most dominant substrate sizes. Upper Vault had a total of 7% fines and 35% bedrock. Sweeney and the Forks had smaller proportions of fines and of bedrock.

McGee (Figure 14c) and stream #21 (Figure 14b) had similar substrate size distributions to that of Cains (Figure 14d), with a low proportion of bedrock (7.2% and 0%), a high proportion of medium and very coarse gravels (10.2%), and a relatively high proportion of fines (20%). The high proportion of fines for stream #21 (Figure 14b) was mostly due to an exceptionally high percentage of coarse sands. The lowest proportion of fines (<3.3 mm) was measured in Dustin (2.3%), followed by #11 (2.5%), and #40 (2.8%). Overall, there was no pattern found with the proportion of streambed fines and the age of stand.

The highest proportion of bedrock was measured in stream #45, with 65.6%; followed by #2, with 51.1%; Dustin Brook, 45.1%; #40, 43.7%; and #11, 39.2%.

Figure 14(a). The range of particle sizes in the 1993 substrate cores taken from the sampling reaches of streams draining watersheds clearcut in the mid-late 1980s.

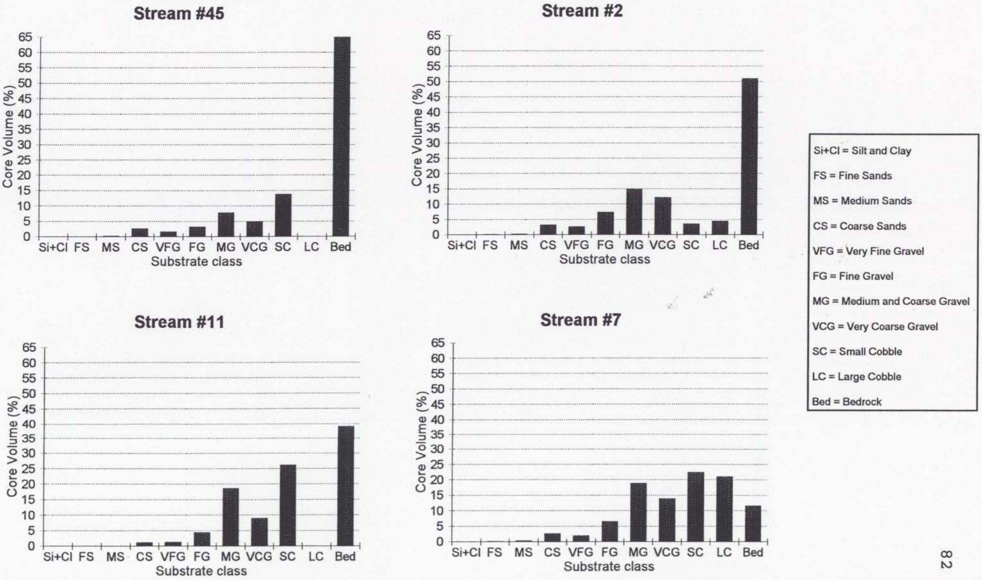


Figure 14(b). Substrate cores cont'd. Streams #21, #25, Haley and Dustin drained watersheds clearcut in the early 1980s.

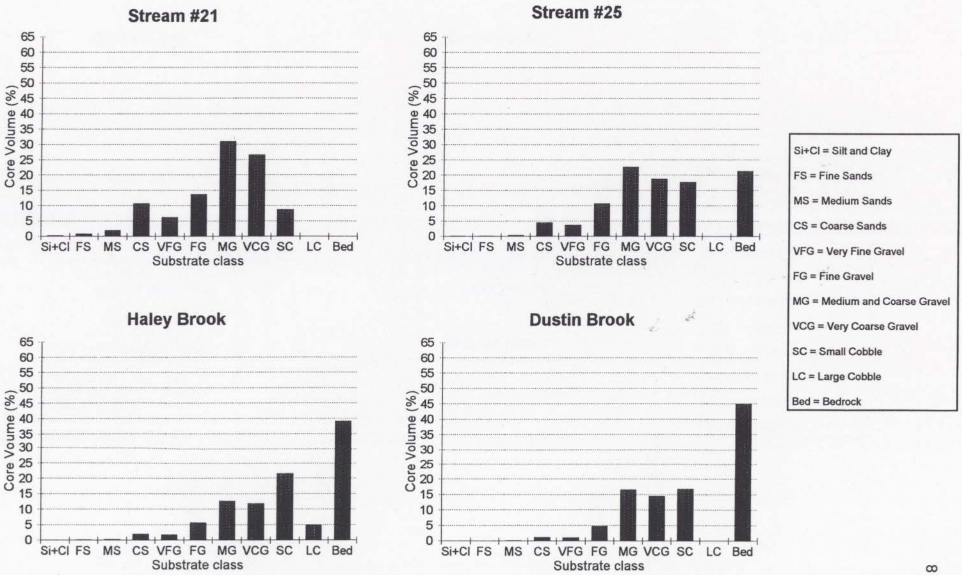


Figure 14(c). Substrate cores cont'd. Stream #41 drains a watershed clearcut in the early 1980s, and streams #15, #40, and McGee drain watersheds clearcut in the 1970s.

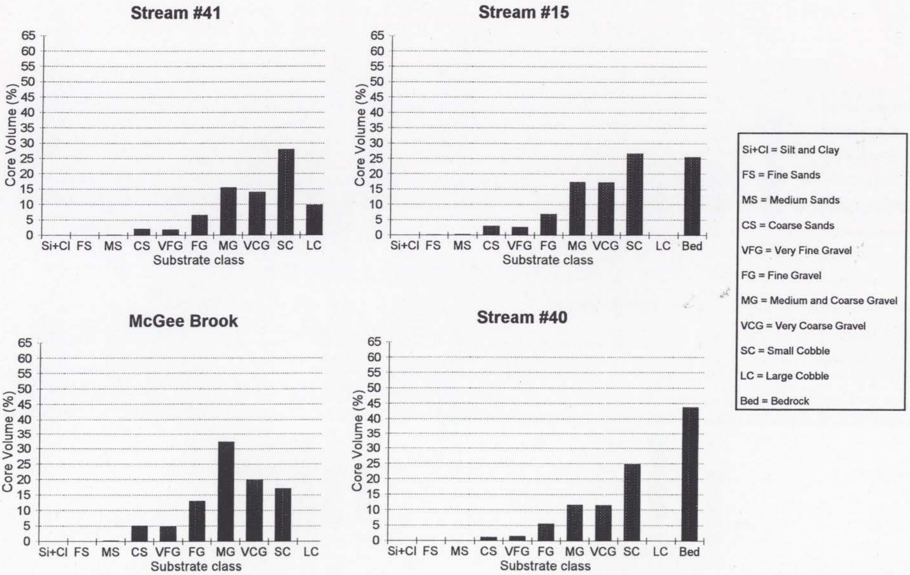
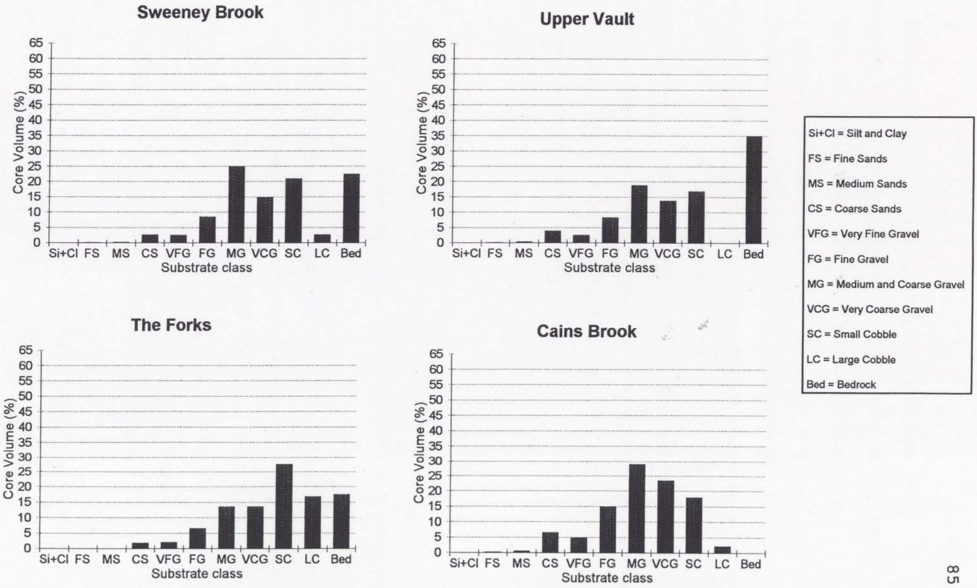


Figure 14(d). Substrate cores cont'd. Sweeney, Upper Vault, The Forks and Cains were reference streams.



Results of the analyses for organic carbon in substrates are given in Figure 15. No trend was evident. The three streams with relatively high organic carbon, Forks, Dustin, and #11, also had very large standard error bars, indicating considerable variance within the sites. Streams #15 and #25 had moderately uniform amounts of organic carbon from one sample to the next. The low concentrations of organic carbon in streams #15 and #21 were somewhat surprising, considering that the former was choked with logging slash and coated by a layer of dark organic silt, and the streambed of the latter had a dark-black, mucky consistency.

4.1.6. Surface substrates

The differences between the two techniques for estimating streambed substrates - core sampling and surface estimation - gave rise to some dissimilar results for each stream. In particular, bedrock was less frequently reported as a surface substrate. In the four most recently cutover streams (Figure 16a), bedrock was often the most dominant or subdominant substrate, but in the other streams (Figure 16b-d), surface exposures of bedrock were nearly absent. There was a significant Pearson correlation between surface bedrock and age of stand ($r = 0.7$, $p = 0.004$). The Forks was the only other stream with a noticeable proportion of bedrock (Figure 16d), no doubt due to its high gradient and

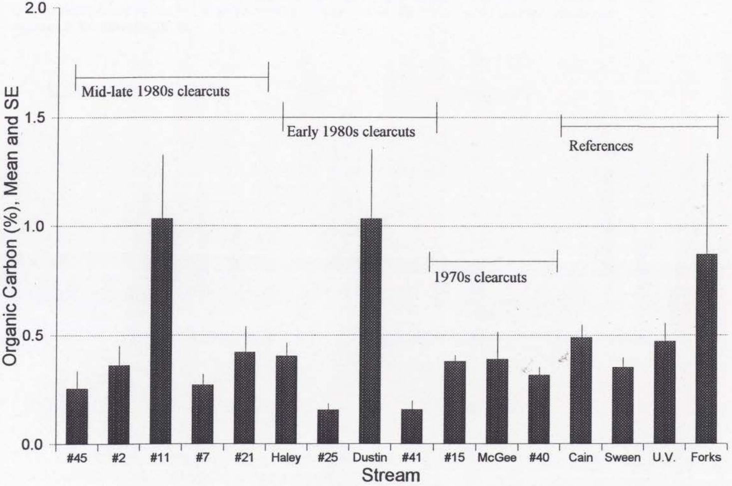


Figure 15. Organic carbon concentration of streambed substrates.

Figure 16(a). The dominant and subdominant surface substrates in the sampling reach of each stream, measured in July, 1993. The dominant substrate has the largest proportion of surface cover, and the subdominant substrate has the next largest proportion of surface cover. Streams #45, #2, #11, and #7 drained watersheds clearcut in the mid-late 1980s.

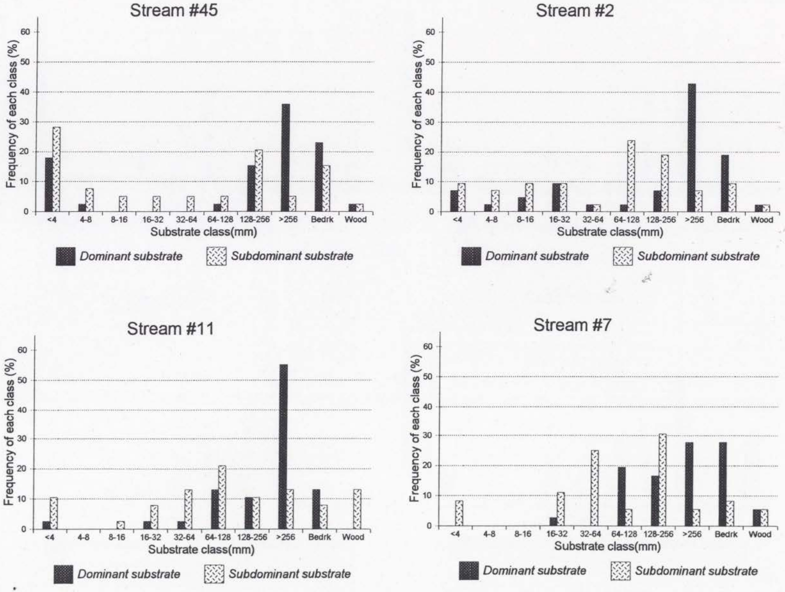


Figure 16(b). Surface substrates continued. Streams #21, #25, Haley and Dustin drain watersheds clearcut in the early-mid 1980s.

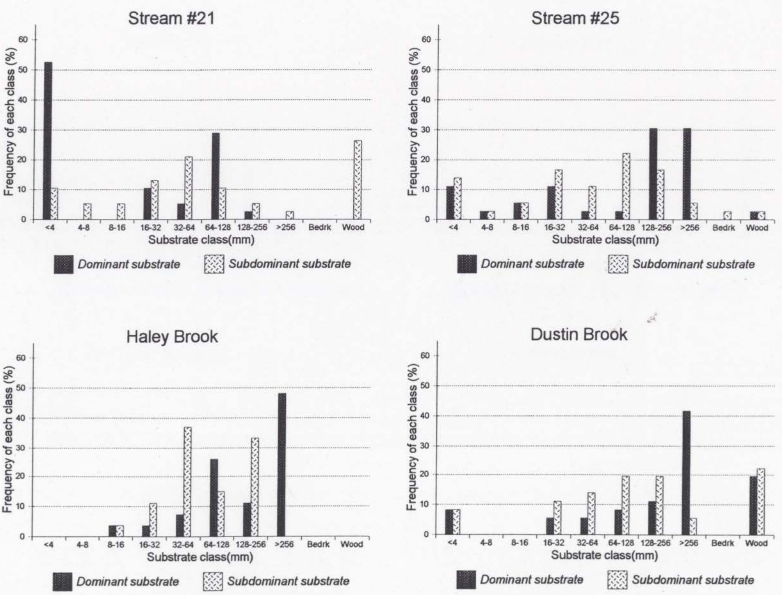


Figure 16(c). Surface substrates continued. Stream #41 drained a watershed clearcut in the early 1980s, and streams #15, McGee and #40 drained watersheds clearcut in the 1970s.

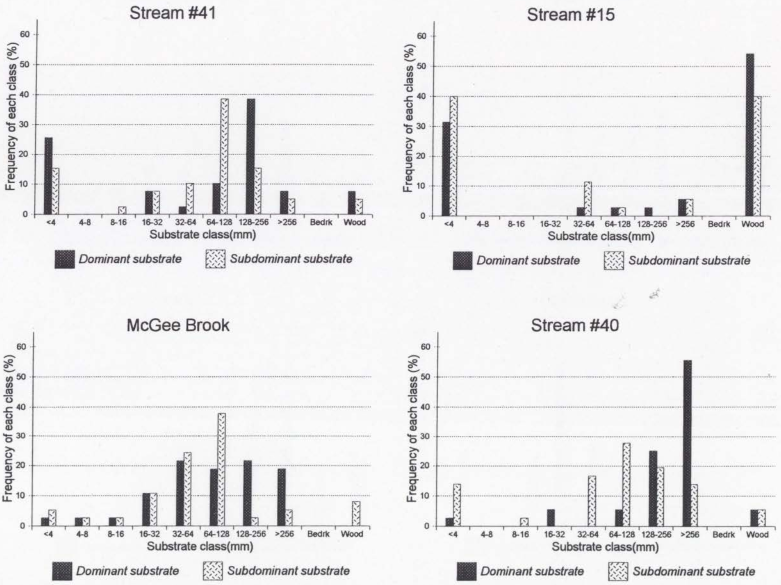
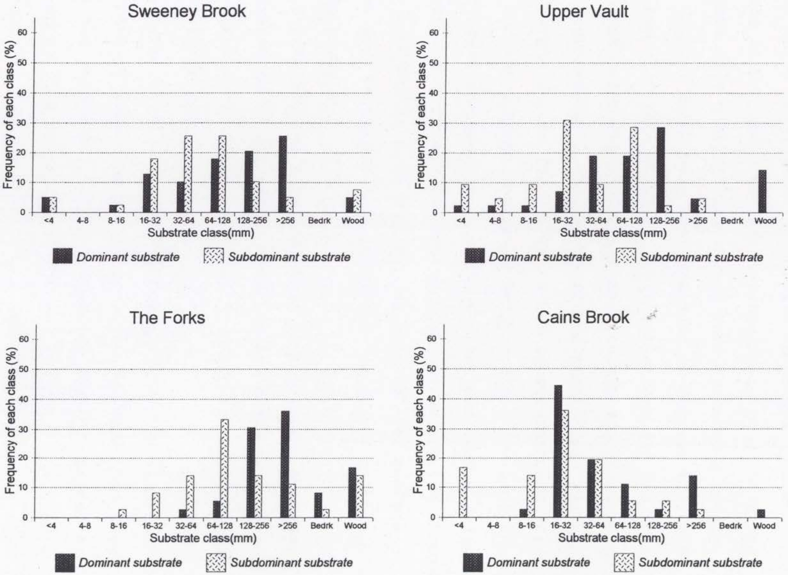


Figure 16(d). Surface substrates continued. Sweeney, Upper Vault, The Forks, and Cains were reference streams.



consequent rapid discharge and erosiveness. Also, the proportion of large cobbles and boulders appeared much higher using the surface estimation technique, probably because it was difficult to distinguish large cobbles and boulders from bedrock when coring into the stream bottom.

Surface fines were most abundant in streams #21, #15, #45, and #41 (Figure 16a-c). All but stream #15 had road crossings above the sampling reach, though #15 appeared to have been used as a skid trail at one time. Only #21 had a high proportion of fines both in the substrate cores (Figure 14b) and the surface estimates (Figure 16b). T-tests showed significant differences in surface fines and coarse gravels between the four park streams and the twelve cutover streams ($\bar{x}_4$ = 1.87 and $\bar{x}_{12}$ = 13.5; t = 2.4, p = 0.03 for surface fines; $\bar{x}_4$ = 40.37 and $\bar{x}_{12}$ = 24.62; t = -2.2, p = 0.044 for coarse gravels).

Stream #15 had a particularly high coverage of woody substrates as it was choked with logging slash (Figure 16c). Wood was also a common substrate in Dustin Brook, #21, and the Forks (Figure 16b, 16d).

Figure 17a-d shows the embeddedness of the streambed as a percent of streambed area (the degree to which the large particles [boulders, cobbles, gravels] were surrounded or covered by fine sediments <4 mm in diameter). The highest embeddedness category (>75%) was strongly correlated with the proportion of surface substrates less than 4 mm in size

Figure 17(a). Embeddedness of the streambed in the sampling reach of each stream (the degree to which the large-sized particles are surrounded or covered by fines <4 mm). Streams #45, #2, #11 and #7 drained watersheds clearcut in the mid-late 1980s.

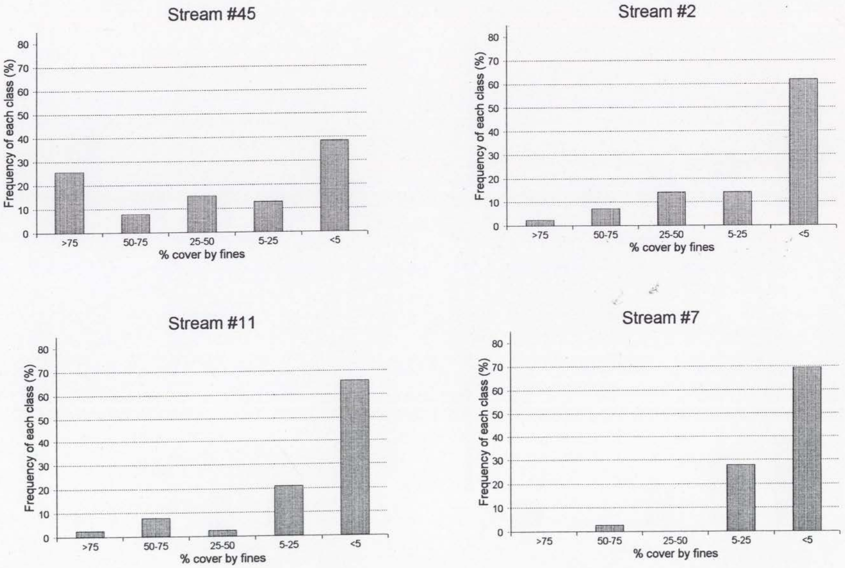


Figure 17(b). Embeddedness continued. Streams #21, #25, Haley and Dustin drained watersheds clearcut in the early-mid 1980s.

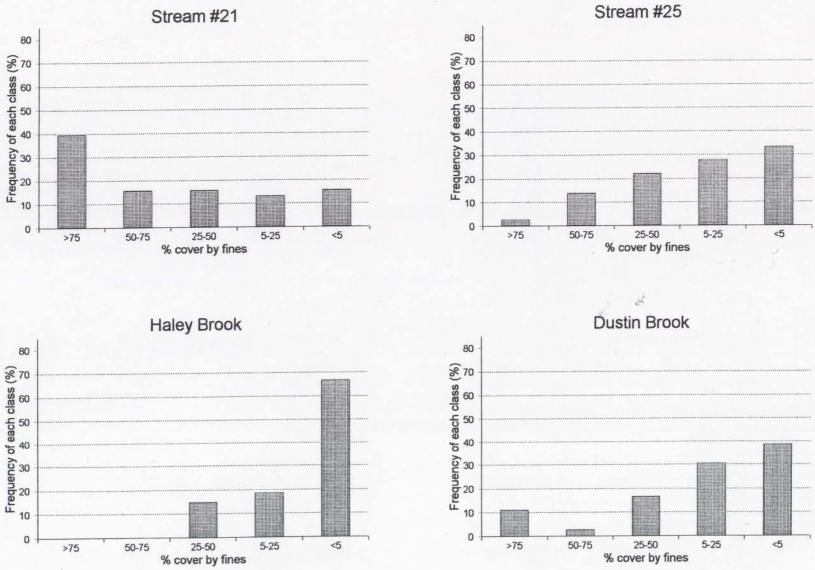


Figure 17(c). Embeddedness continued. Stream #41 drained a watershed clearcut in the early 1980's, and Streams #15, #40 and McGee drained watersheds clearcut in the 1970s.

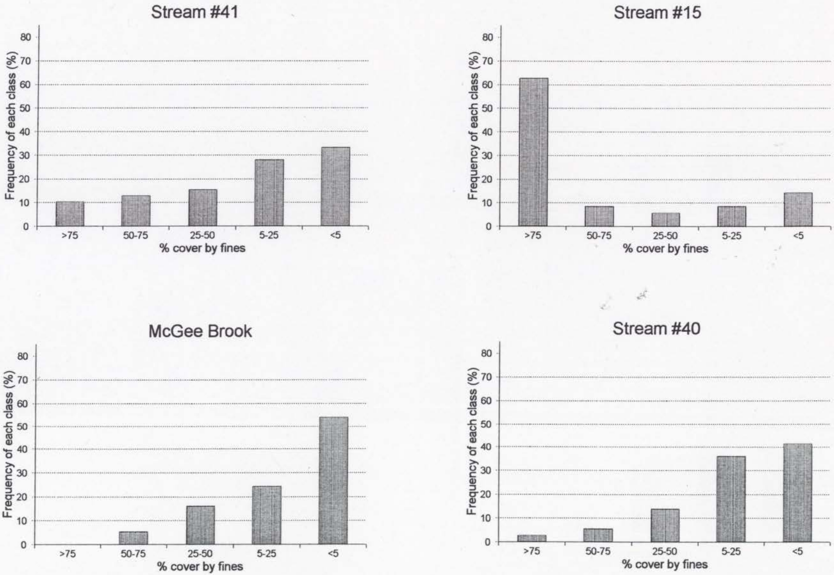
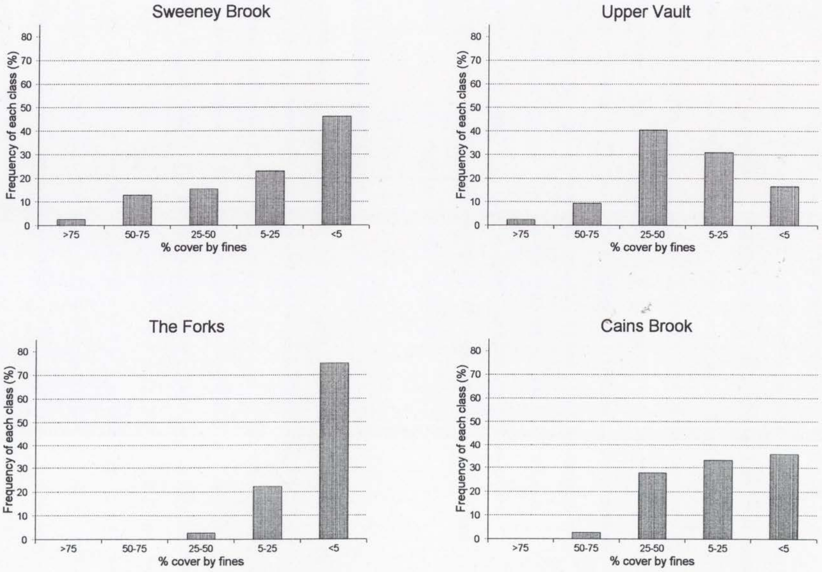


Figure 17(d). Embeddedness continued. Sweeney, Upper Vault, The Forks and Cains were reference streams.



(Pearson r = 0.94). The most frequent category was <5% embeddedness, and the least frequent 50-75%, closely followed by >75%. The lowest embeddedness was seen on #2, #11, #7, and the Forks (Figure 17a, 17d), while the greatest embeddedness was seen on #21, #15, and #45 (Figure 17a-c). These results suggest that the quantity of fines covering the streambed is not necessarily related to the quantity of fines below the surface. For example, while Cains, McGee, and #21 had the greatest volume of fines in the substrate cores, neither Cains nor McGee had many surface fines. Similarly, #45 and #15 had a high frequency of surface fines, but an average proportion of substrate core fines compared with the other study streams.

4.1.7. Stream channel

Mean channel bank-full width and cross-sectional area which are indicators of stream size and the size of the watershed, are shown in Figures 18 and 19. The streams with the greatest bank-full width were Dustin (3.8 m), McGee (3.7 m), and Upper Vault (3.6 m), and those with the least width were #41 (2.5 m), #7 (2.5 m), #15 (2.7 m), Cains (2.7 m) and Haley (2.7 m). At least 2 of the streams with the greatest recorded channel widths were heavily disturbed in the past: McGee appeared to have been used as a skid trail, and Dustin for logdriving. The standard error bars indicated that certain streams, particularly Dustin and #21, had a highly

Figure 18. Channel width measured over the chronosequence of 16 streams, August, 1993.

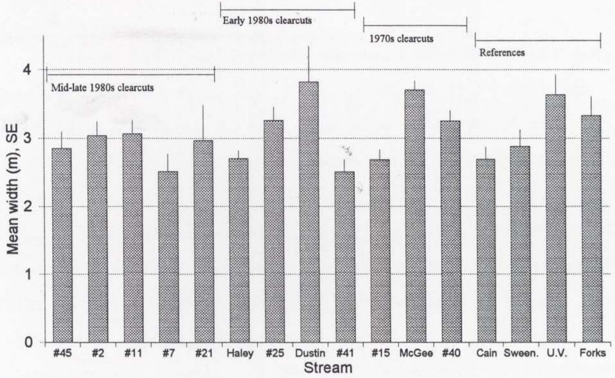
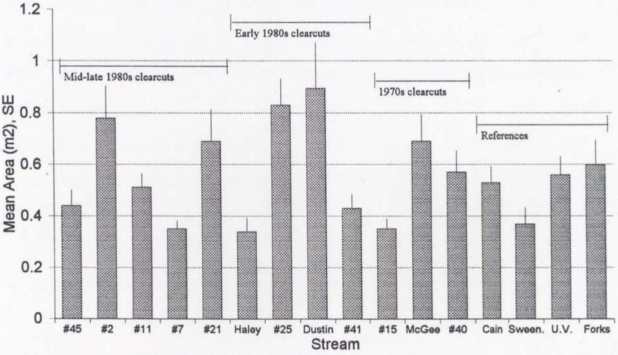


Figure 19. Channel area measured over the chronosequence of 16 streams, August, 1993.



variable channel width along the sampling reach, while others, such as McGee and Haley, had a somewhat uniform channel width.

Streams with the greatest channel cross-sectional area were Dustin (0.89 m^2), #25 (0.83 m^2) and #2 (0.78 m^2), and those with the least channel area were Haley (0.34 m^2), #7 (0.35 m^2), #15 (0.35 m^2) and Sweeney (0.37 m^2) (Figure 19). These estimates of stream cross-sectional area gave us a rough idea of watershed size as well as flood frequency. It appeared that Dustin and #25 drained relatively large areas, while Haley, #7, #15 and Sweeney drained smaller-sized watersheds. Stream #2 did not drain a large area, however it was a steep stream, and the large cobble banks suggested high peakflows.

Streams #2, #21, #25 and Cains were consistently narrow and deep, with average width:depth ratios of 12.1 to 17.2, and standard errors of less than 3.8 (Table 7). Streams #45, #11, Dustin, #41, #15, McGee, #40 and Sweeney had relatively consistent width:depth ratios of 21.1 to 28.9, with standard errors between 2.1 to 5.2. The remaining streams, #7, Haley, Upper Vault and the Forks, had highly variable width:depth ratios.

4.1.8. Vegetation

Shrub density was very low, between 0.40 x 10^3 to 1.4 $\overline{x}$ 10^3 per ha, for the more recent cuts with wide buffer strips

Table 7. The range of width: depth ratios for each stream, as well as the average and the standard error.

Stream	Range	Average width: depth ratio	Standard error	
#45	7.0 - 42.4	21.1	3.3	(n=12)
#2	4.3 - 25.2	12.1	1.9	(n=12)
#11	9.7 - 57.9	22.3	4.4	(n=12)
#7	8.8 - 105.8	29.0	9.1	(n=12)
#21	6.1 - 35.1	13.4	2.2	(n=12)
Haley	10.9 - 143.1	36.7	14.1	(n=9)
#25	6.9 - 35.3	15.1	2.1	(n=12)
Dustin	7.2 - 64.1	28.9	3.9	(n=12)
#41	11.4 - 31.8	21.9	2.1	(n=12)
#15	10.6 - 60.5	26.3	5.2	(n=11)
McGee	12.1 - 50.0	24.7	3.1	(n=12)
#40	8.4 - 68.5	23.5	4.4	(n=12)
Cain	7.9 - 31.8	17.2	3.8	(n=6)
Sweeney	12.9 - 50.9	28.5	4.4	(n=12)
Upper Vault	11.4 - 120.7	31.5	8.2	(n=12)
Forks	4.8 - 298.2	45.3	24.3	(n=11)

(#45, #2 and #11), and for three of the four reference streams (all but Cains) (Figure 20). By contrast, shrub density was much higher for all the other streams, especially Haley (21.5×10^3 per ha), Cains (15.0×10^3 per ha) and Dustin (13.3×10^3 per ha). The six streams with low shrub density tended to be dominated by non-alder species, such as *Betula allenghaniensis, Abies balsamea, Picea rubens*, and *Betula papyrifera* (Table 8). The other streams, however, were almost all dominated by *Alnus rugosa* or, occasionally, *Abies balsamea*. The highest shrub diversity (using the Brillouin's Index) was recorded on #40, #41, and #25, and the lowest on McGee and Cains.

Snag density (the density of standing, dead trees) followed a different pattern to that of shrub density. In general, where snag density was high, shrub density was low, and vice versa (Figures 20 and 21). This was particularly obvious for the four park streams, as well as #11, Haley, #25, and McGee. This trend was further illustrated by a strong negative Pearson correlation between snag and shrub density ($r = -0.70$, $p = 0.003$). Furthermore, the Pearson correlation coefficients between these two variables and many other variables were almost always opposite in direction. For example, the Pearson r between density of shrubs and moss cover on the streambed was -0.80 ($p < 0.001$), compared with $r = 0.71$ ($p = 0.002$) for snag density and moss cover; similarly the Pearson r between alkalinity

Figure 20. The density of shrubs in the riparian zone of each stream, measured in August, 1993. Alders were the dominant shrub species.

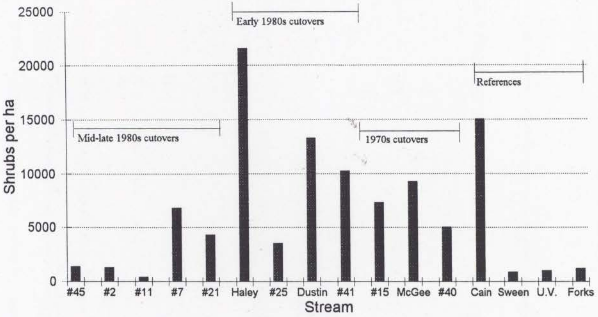


Figure 21. The density of snags in the riparian zone of each stream, measured in August, 1993. Snag density had a strong negative correlation with shrub density.

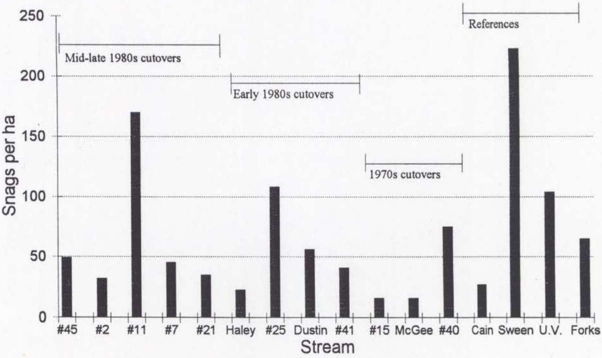


Table 8. The importance value of each shrub species (<5 cm dbh) at each stream site, measured in August, 1993. Importance value is a measure of basal area, density and frequency.

Shrub species	#45	#2	#11	#7	#21	Haley	#25	Dustin	#41	#15	McGee	#40	Cains	Sween	U.V.	Forks
Abies balsamea	30.8	95.2	135.7	101.0	7.5	42.9	27.3	78.0	91.5	131.1	7.9	50.4				17.8
Acer pensylvanicum	3.7						22.5					5.8		63.8	23.0	21.7
Acer rubrum	31.2		3.9	11.4			59.9	4.6	19.0			3.7	13.4	13.2	7.5	25.0
Alnus rugosa	30.2		28.8	46.9	185.6	122.2	73.6	138.3	67.4	115.9	257.5	114.8	252.2		37.8	
Acer saccharum	25.0	4.2			9.7							4.1		11.0	31.6	
Acer spicatum	58.5	9.8			29.8		10.8		11.8			8.8		33.9		32.3
Amelanchier							3.8	16.1	10.5	4.4	7.4					
Betula allenghaniensis	66.7	110.8		3.7	15.3		13.6							122.6	126.8	66.1
Betula papyrifera	32.7	34.0	6.2	22.1	17.8	20.8	40.3		28.0	24.2		25.7	6.8		20.3	9.9
Betula populifolia												4.1				
Cornus rugosa													14.6			
Cornus spp.						4.0			10.5							
Corylus cornuta						16.5	5.2		7.7			3.8		5.9		
Fagus grandifolia														7.1		
Fraxinus nigra							8.0									
Lonicera spp.															11.8	
Nemopanthus mucronata						8.5			5.3							
Picea rubens	17.8	33.8	113.8	97.4	26.5		31.7	49.4	29.6	11.1	15.3	41.0	13.1	42.6	37.5	123.5
Prunus pensylvanica						5.1										
Prunus virginiana						70.6										
Salix bebbiana											12.0	5.9				
Salix discolor									3.8							
Salix spp.										4.4		3.7				
Sambucus racemosa		4.1														
Sorbus americana				8.4			3.4		8.3			21.1				
Spirea spp.						9.5		13.6	3.4	8.9		7.3				296.3
Thalictrum polygamum					7.7											
Viburnum cassinoides				5.6												
Viburnum trilobum									3.4							

Note: See Appendix 7 for common names.

and shrub density was 0.66 ($p = 0.006$), compared with -0.63 ($p = 0.009$) for snag density and alkalinity.

The three park streams other than Cains had high snag densities (65-223 per ha), as did #11 (170/ha) and #25 (108/ha), which had fairly wide buffers and were still forested on one side of the sampling reach. The average snag size for #11 was relatively small - less than 0.01 m^2 in basal area. Tree density was high along #11, and it appeared that the forest was going through a self-thinning process in which many smaller sized trees were dying. The presence of snags in these riparian zones ensured a long-term supply of woody debris into the stream channels.

Riparian tree density was greatest for stream #11, at 3.02×10^3 per ha (Figure 22). The remaining 15 streams had fewer than 10^3 trees per ha. Cains, McGee, #15 and Dustin had particularly low tree densities. The dominant species for most streams was *Picea rubens*, followed by *Abies balsamea* and *Betula allenghaniensis*. *Betula allenghaniensis* was common along Sweeney, Upper Vault, and the Forks (Table 9). The fourth reference stream, Cains, was dominated by alders. The highest tree diversity (using Brillouin`s index) was observed along #25, followed by Haley, #40, and #2. The lowest diversity was measured along Cains and #15.

Tree density was negatively correlated with shrub density ($r = -0.65$, $p = 0.006$) and positively correlated with snag density ($r = 0.64$, $p = 0.007$). These correlations

Figure 22. The density of trees (>5 cm dbh) in the riparian zone of each stream, measured in August, 1993.

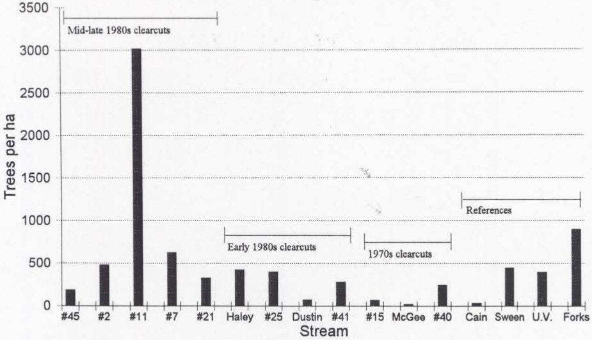


Table 9. The importance value of each tree species (>5 cm dbh) at each stream site, measured in August, 1993. Importance value is a measure of basal area, density and frequency.

Tree Species	#45	#2	#11	#7	#21	Haley	#25	Dustin	#41	#15	McGee	#40	Cains	Sween	U.V.	Forks
Abies balsamea	7.0	123.5	96.7	65.8	24.8	86.6	7.0	46.6	105.6	94.4	33.1	36.9				
Acer pensylvanicum															13.0	
Acer rubrum							29.4					22.2	16.1			13.9
Alnus rugosa				7.0	49.7	53.8	52.2	33.0	21.6		113.2	7.1	131.5			
Acer saccharum														33.5		
Acer spicatum																13.5
Betula allenghaniensis	90.7	33.5	32.5		20.2		37.1			125.9				101.8	148.5	74.1
Betula papyrifera	10.9	93.4	22.2	24.4	10.0		20.4		24.0	6.0					36.2	24.3
Picea glauca						27.1	25.8				76.3	37.2				
Prunus pensylvanica						14.1										
Picea rubens	147.7	37.5	148.6	190.4	170.0	105.6	94.9	140.8	121.1			132.0	23.8	164.7	96.2	167.8
Sorbus americana							14.6					51.9				

Note: See Appendix 7 for common names.

were not as strong as those between snags and shrub-sized species, but it is intuitive that there were more snags where tree density is highest. These results indicated that the least disturbed riparian forests had relatively low shrub density and higher tree and snag density, and was further reinforced by the positive correlation between buffer width and density of snags ($r = 0.67$, $p = 0.005$).

Canopy cover was estimated above the stream channel and in the riparian zone. The results of both estimates were similar, and were strongly correlated (Pearson $r = 0.84$, $p < 0.001$), although stream canopy cover tended to be lower than riparian canopy cover (measured 1.5 m from the streambank) (Figures 23 and 24). Streams #15 (no buffer) and #40 (an uncertain buffer) had less than 12% cover shading the watercourse, while #7 (variable-sized buffer), Dustin (alder-dominated buffer), #41 (budworm-infested buffer) and McGee (no buffer) had 20-30% canopy cover. The remaining streams had greater than 58% cover except for Haley (10-20 m buffer) and Cains (alder-dominated buffer), with 40-50%. Pearson correlations indicated that tree and snag density were positively correlated with canopy cover, and shrub density negatively correlated with canopy cover ($r = 0.50$, $p = 0.049$ for snag density and stream cover). Stream canopy cover played a significant role in predicting the maximum August temperature: the simple regression F-ratio = 12.41, $p = 0.003$, and the Pearson correlation between cover and

Figure 23. Canopy cover over the stream channel, measured in September, 1993.

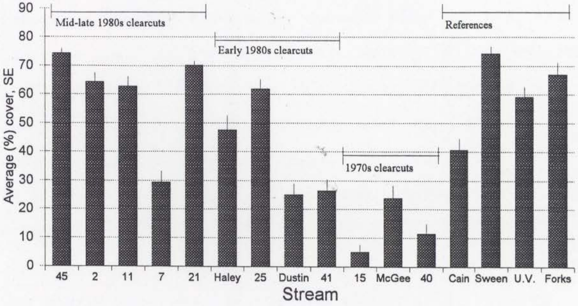
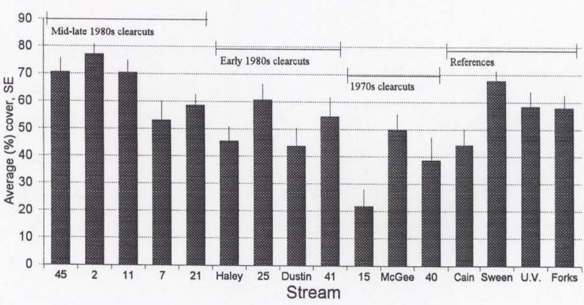


Figure 24. Riparian canopy cover (measured in August, 1993, from within the riparian zone, 1.5 m away from the streambank).



maximum August temperature was r = -0.69, p = 0.003.

Relatively large moss cover was observed in three of the reference streams (excluding Cains), #45, and #11, while little or no moss was recorded in #2, Haley, #41, McGee, and Cains (Figure 25). The percent moss cover followed a similar trend to the density of snags over the chronosequence of sites (Pearson r = 0.71, p = 0.002), and an opposite trend to the density of shrubs (Pearson r = -0.80, p = 0.00).

Average aquatic macrophyte cover varied from 0 to 8.6%, and was highly variable between and within sites (Figure 26). Less than one percent macrophyte cover was observed in four of the five most recently clearcut streams - #2, #11, #7, and #21, as well as in Dustin, #40, Sweeney, and the Forks.

4.2.0. Invertebrate data

The average number of macroinvertebrates collected in each rockball is depicted in Figure 27. Totals are not presented as several rockballs were not recovered, including 2 from McGee, and 1 from each of #25, Haley, Sweeney and #11. The mistaken use of an oversized sieve when rinsing the samples from The Forks may have led to the loss of some smaller organisms and the possible underestimation of totals for this stream. As a result, these data were omitted from

Figure 25. Percent moss cover on the streambed, measured in September, 1993.

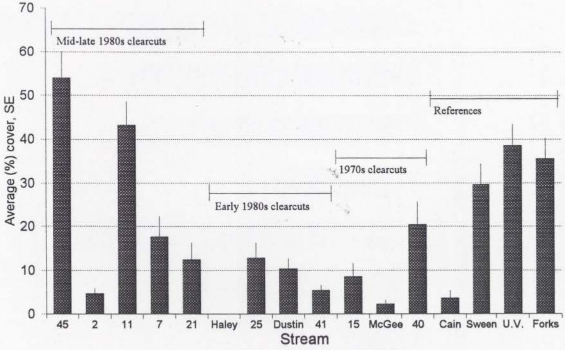
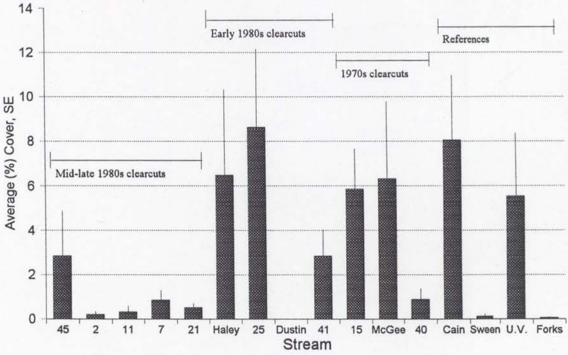


Figure 26. Percent macrophyte cover on the streambed, measured in September, 1993.



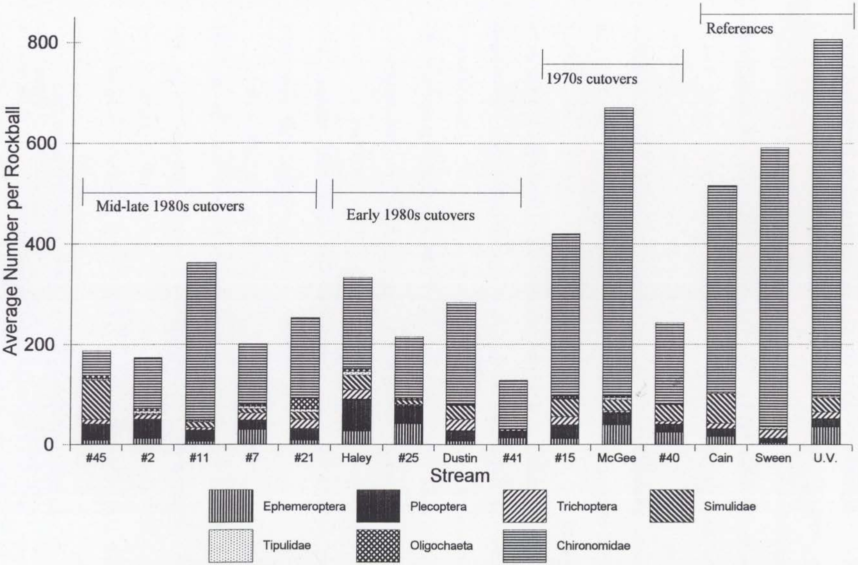


Figure 27. The average number of macroinvertebrates per rockball over the chronosequence of streams. Invertebrates were collected in five gravel-filled bags left in each stream for six weeks, from October - November, 1993. Only the seven most common taxa are included in this figure. See Appendix for further details.

the analysis[d]. There is a steady increase in the number of invertebrates per rockball since time of cutting, with the greatest abundance found on McGee, Sweeney, Cains, and Upper Vault. An independent samples T-test on invertebrate abundance indicates a significant difference between the three reference streams (excluding the Forks) and the cutover streams ($\overline{x}_3$ = 640 and $\overline{x}_{12}$ = 290; $t = -3.7$, $p = 0.003$), and a simple regression of invertebrate abundance against age of stand (log transformed) indicates a highly significant F-ratio of 20.1 with $p = 0.001$. A forward stepwise regression with invertebrate totals as the dependent variable shows that total nitrogen in water (log transformed) was the best predictor of invertebrate abundance (F-ratio = 14.5, $p = 0.002$). In fact, while both nitrogen concentrations and invertebrate abundance followed opposite patterns over the chronosequence of sites, it was not likely that abundance was directly influenced by nitrogen concentrations.

Chironomids were the most dominant group (totalling 19.1×10^3 out of a grand total of 25.4×10^3 in all samples for all streams), and it is apparent that average abundance per rockball would be much lower without them (Table 10). It is also obvious that chironomids were largely responsible

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[d] Alternatively, the large drop in elevation over the drainage basin of the Forks would have resulted in low base flows and high flow variability, which may have reduced habitat quality and abundance of invertebrates (Lanka et al. 1987).

Table 10. The relative abundance of the eight most common taxa. Values are percentages of the total number of invertebrates for each stream.

STREAM	EPHEMEROPTERA	PLECOPTERA	COLEOPTERA Elmidae	TRICHOPTERA	DIPTERA Chironomidae	Simuliidae	Tipulidae	OLIGOCHAETE
45	4.18	16.92	0.32	5.14	26.02	42.40	1.18	3.85
2	6.67	22.18	0.00	4.94	58.74	1.61	2.53	3.33
11	1.66	5.93	0.21	1.38	86.48	3.17	0.41	0.76
7	14.37	9.18	0.40	7.19	60.68	4.19	2.20	1.80
21	3.11	9.24	0.00	7.17	63.75	6.06	2.07	8.61
Haley	7.84	18.85	0.60	5.51	54.52	8.90	2.04	1.73
25	19.23	14.57	2.45	0.93	57.11	3.96	0.58	1.17
Dustin	2.41	6.45	0.92	8.16	71.91	9.50	0.28	0.35
41	10.12	9.19	1.71	2.49	75.08	0.93	0.16	0.31
15	2.86	4.34	2.10	3.77	77.21	8.49	0.38	0.86
McGee	5.87	2.79	0.80	0.60	85.72	3.23	0.70	0.30
40	9.67	6.17	0.51	2.49	66.51	14.19	0.18	0.28
Cains	3.02	2.91	0.12	2.09	79.78	11.55	0.43	0.12
Sweeney	0.47	1.48	0.13	2.83	93.95	0.30	0.68	0.17
U.V.	4.34	1.93	0.15	1.56	87.86	3.49	0.47	0.20

for the increasing trend in abundance per rockball with age of stand (Table 10). Other common groups colonizing rockballs include Simuliids (1.8×10^3), Plecoptera (1.6×10^3), Ephemeroptera (1.2×10^3), and Trichoptera (0.83×10^3).

Plecoptera displayed a relatively steady decrease in abundance and relative abundance with increasing age of stand (Figure 28; Table 10). Average numbers ranged from 39 in stream #2 to 9 in Sweeney. Haley, however, had the highest number of Plecoptera, with an average of 63 per rockball. This unusually high average is due to a very high number of Plecoptera in one rockball (151 individuals). A T-test showed a significant difference in Plecoptera abundance colonizing rockballs in the reference and the cutover streams ($\bar{x}_4 = 12.3$ and $\bar{x}_{12} = 25.7$; $t = 3.1$, $p = 0.009$).

Oligochaete abundance and relative abundance also decreased with age of stand (Figure 29, Table 10). The largest numbers were found on #21 (with an average of 22 per rockball), followed by #45 (7.2), #2 (5.8), and Haley (5.8). Abundance was highly variable amongst samples. Stream #21, for example, had 3 oligochaetes in one rockball, and 60 in another. The lowest numbers were collected on #41 (0.4), #40 (0.6), Cains (0.5) and the Forks (0). There was a significant difference in oligochaete abundance between the 4 park streams and the 12 cutover streams ($t = 2.4$, $p =$

Figure 28. The average number of Plecoptera per rockball collected from 16 streams in November, 1993. Streams #45 - 21 drained watersheds clearcut in the mid-late 1980s; Haley - #41 drained watersheds clearcut in the early 1980s; #15 - 40 drained watersheds clearcut in the 1970s; and Cain - Upper Vault were reference streams.

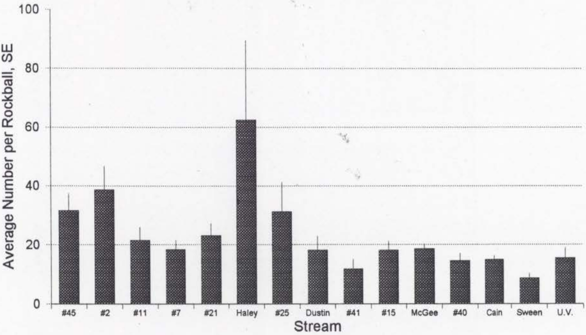


Figure 29. The average number of Oligochaeta per rockball collected from 16 streams in November, 1993. Streams #45 - 21 drained watersheds clearcut in the mid-late 1980s; Haley - #41 drained watersheds clearcut in the early 1980s; #15 - 40 drained watersheds clearcut in the 1970s; and Cain - Upper Vault were reference streams.

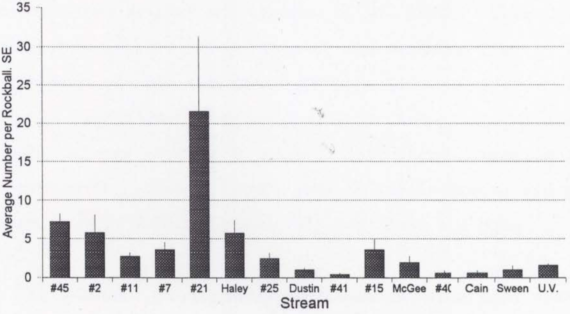
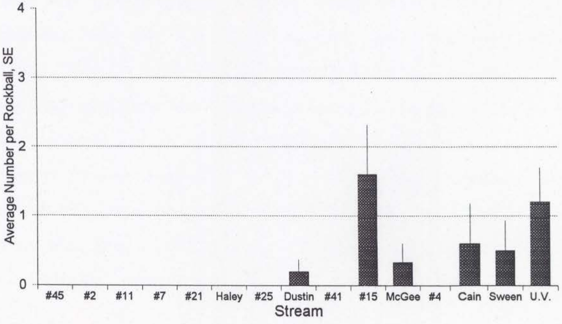


Figure 30. The average number of Nymphomyiidae per rockball collected from 16 treams in November, 1993. Streams #45 - 21 drained watersheds clearcut in the mid-late 190s; Haley - #41 drained watersheds clearcut in the early 1980s; #15 - 40 drained waterheds clearcut in the 1970s; and Cain - Upper Vault were reference streams.



0.038).

One species of the Nymphomyiidae family (Order Diptera), is found in northeastern North America - *Palaeodipteron walkeri* Ide (Borror *et al.* 1989). This organism had an opposite pattern of abundance to the Oligochaetes and the Plecoptera (Figure 30).
While the totals were small - no greater than 4 in any one rockball - there were no *P. walkeri* colonizing rockballs in the seven most recent cutovers, one in Dustin Brook, and one or more in #15, McGee, Sweeney, Cains, Forks and Upper Vault.

The Elmidae (Coleoptera) and Tipulidae (Diptera) showed opposite patterns to each other (Figures 31 and 32). While there were few Elmidae in the reference and most recently cutover streams, and many more in the older cuts, there were few Tipulidae in the older cuts and many more in the recently cutover and reference streams.

The two invertebrate orders, Ephemeroptera and Trichoptera, and the two Dipteran families Simuliidae and Empididae, had no discernible pattern over the chrono-sequence of cutover and reference streams (Appendix 5). The greatest abundance of Simuliidae was found on #45, with an average of 79 per rockball. Nearly all these, however, were collected from one rockball, which had 391 individuals. Three of the five rockballs had no Simuliids, while one other had five. Other streams also tended to have large

Figure 31. The average number of Elmidae per rockball collected from 16 streams in November, 1993. Streams #45 - 21 drained watersheds clearcut in the mid-late 1980s; Haley - #41 drained watersheds clearcut in the early 1980s; #15 - 40 drained watersheds clearcut in the 1970s; and Cain - Upper Vault were reference streams.

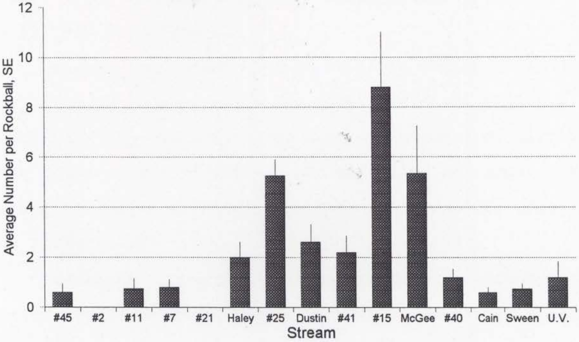
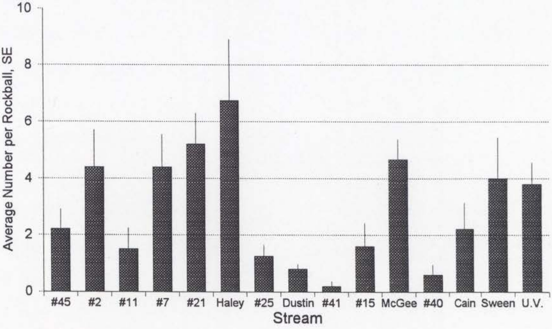


Figure 32. The average number of Tipulidae per rockball collected from 16 streams in November, 1993. Streams #45 - 21 drained watersheds clearcut in the mid-late 1980s; Haley - #41 drained watersheds clearcut in the early 1980s; #15 - 40 drained watersheds clearcut in the 1970s; and Cain - Upper Vault were reference streams.



variations in abundance of Simuliidae, though less so than #45, including Cains, Haley, Dustin, #15, #40, and Upper Vault. Very few simuliids were found colonizing rockballs in #2, #41 and Sweeney.

Ephemeroptera were also variable in abundance amongst rockballs but tended to be far more consistent than the Simuliidae (Appendix 5). The greatest numbers of mayflies were found in stream #25 (41 per rockball) and McGee (39 per rockball), while the fewest were found in Sweeney (2.8) and the Forks (1.4).

Trichoptera were found in all rockballs, though numbers varied widely between streams and sometimes within streams (Appendix 5). The greatest abundance was recorded in Dustin Brook (23 per rockball), while the lowest abundance was recorded on #25 (2).

The remaining invertebrate groups had fewer than ten individuals collected from any one stream, including Pelecypoda (clams), Anisoptera, Arachnida, Coleoptera (excluding the family Elmidae), Collembola, Copepoda, Athericidae, Ceratopogonidae, Empididae, and Psychididae.

4.3.0. Invertebrate and habitat relationships

4.3.1. Principal components analyses

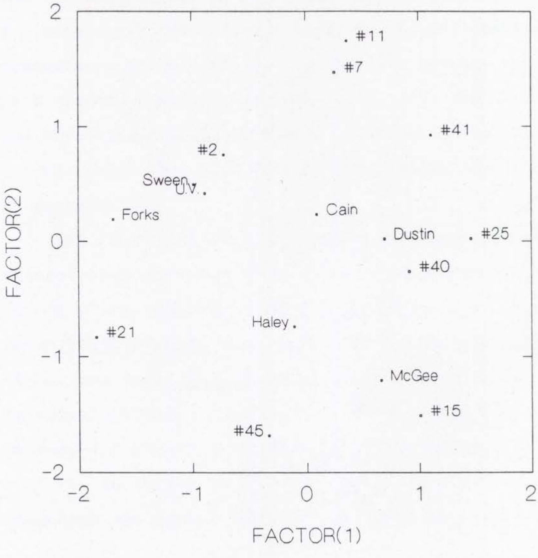
Principal components analysis (PCA) was employed to simplify the data matrix, and to ordinate the stream sites based on different sets of environmental variables. The purpose of this analysis was to observe the arrangement of sites on the basis of covariation of environmental variables alone, i.e., with no *a priori* groupings by clearcut age. Five sets of environmental variables were used to analyze this arrangement of sites. The first four sets included (1) temperature variables, (2) water chemistry variables, (3) vegetation and channel variables, and (4) substrate variables. The final set was selected from these four sets to summarize the environmental data into one ordination of stream sites.

The first PCA utilized four temperature variables, including maximum temperatures for July and August, average August temperature, and average summer temperature (June, July, and August). The first principal component, which accounts for 92.5% of the total variance explained, is highly positively correlated with the four maximum and average temperature measurements (r values of 0.92 to 0.99) (Table 11). In Figure 33, stream #2 and the three reference streams, the Forks, Sweeney, and Upper Vault, are closely ordinated in space, and are negatively associated with maximum stream temperatures. McGee and #15 (both with no buffer strip) are in the bottom right corner, positively associated with maximum temperatures; #11 and 7 in the top centre; and the remaining streams dispersed across the diagram. Cains lies between the other reference streams and Dustin and #40.

Table 11. Principal components analysis of temperature variables with correlations between the first four principal components and the temperature variables.

	Principal component			
	1	2	3	4
Eigenvalue	3.70	0.20	0.08	0.02
% Total Variance	92.54	5.10	1.99	0.38
Average Temp. for June, July, August	0.96	0.22	0.14	0.06
Maximum August Temp.	0.92	-0.38	0.07	0.01
Ave. Temp. for August	0.99	0.11	0.02	-0.10
Maximum July Temperature	0.97	0.02	-0.23	0.03

Figure 33. Principal components analysis of temperature variables. The diagram depicts the ordination of sites based on maximum and average temperatures.



A second PCA was employed to analyze the 10 water quality parameters from the water samples. The first component, explaining 48.9% of the variance, was strongly correlated with dissolved N, Mg, K, specific conductance (sp.cond.), Ca, and alkalinity (alk.) (Table 12). Plots of the principal components depicted the close associations of the latter three variables, and of total N and nitrate-N. The second component, accounting for 22.5% of the variance, was most strongly related to Na and Cl. The four reference streams, along with #11 and Dustin Brook, are situated in the top left corner of the ordination diagram Figure 34, and are negatively associated with N, Mg, K, Ca, specific conductance and alkalinity. Cains lies between the other park streams and the older cuts (#15, #41, #40, and McGee). The more recent cuts are mostly situated in the top right corner, positively associated with nutrient concentrations in streamwater.

The first component from a PCA of vegetation and channel data explained 37.9% of the total variance, the second 19.6%, and the third, 14.0% (Table 13). Canopy cover, buffer width, moss cover, and density of shrubs, trees, and snags were strongly correlated with the first principal component (Table 13). Shrub diversity, organic carbon, and channel area and width were strongly correlated with the second principal component. The third principal component was correlated with tree diversity and channel

Table 12. Principal components analysis of water chemistry variables with correlations between the first four principal components and the water chemistry variables.

	Principal component			
	1	2	3	4
Eigenvalue	3.63	2.14	0.94	0.54
% Total Variance	45.37	26.80	11.75	6.70
Nitrogen	0.82	0.07	0.29	-0.44
Sodium	0.41	0.78	0.10	0.24
Magnesium	0.88	-0.01	0.08	0.17
Specific Conductance	0.85	0.04	-0.41	0.20
Chloride	-0.34	0.83	0.06	0.23
Potassium	0.80	0.44	0.22	-0.16
Phosphorus	0.12	-0.58	0.71	0.34
Alkalinity	0.72	-0.55	-0.34	0.15

Figures 34. Principal components analysis of water chemistry data. Stream sites are positioned with respect to factors 1 and 2.

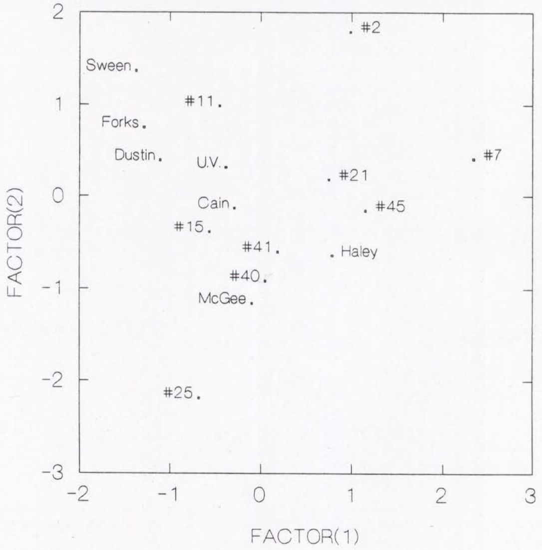
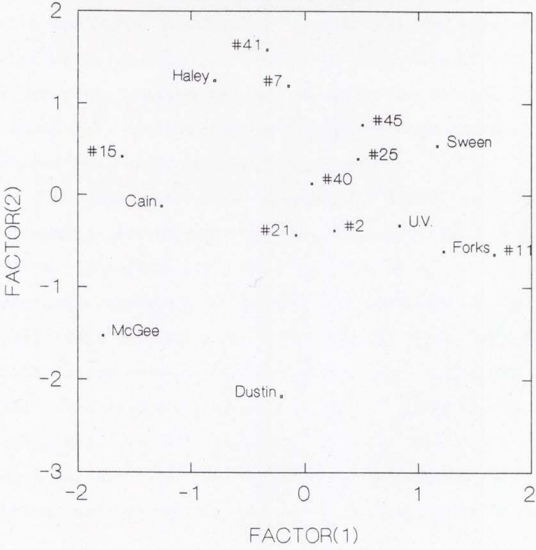


Table 13. Principal components analysis of vegetation and channel variables with correlations between the first four principal components and the vegetation and channel variables.

	Principal component			
	1	2	3	4
Eigenvalue	4.55	2.35	1.68	0.95
% Total Variance	37.90	19.62	13.99	7.89
Shrub Diversity	0.45	0.61	0.40	0.21
Tree Diversity	0.39	0.15	0.87	0.12
Stream Canopy Cover	0.70	0.02	0.04	-0.41
Buffer Width	0.68	0.20	-0.14	0.20
Channel Width	0.18	-0.83	0.35	-0.14
Shrub Density	-0.85	0.10	0.17	0.36
Snag Density	0.88	-0.01	0.02	-0.03
Tree Density	0.83	0.29	0.06	0.22
Instream Moss Cover	0.78	-0.05	-0.31	-0.28
Instream Macrophyte cover	-0.56	0.30	0.25	-0.51
Organic carbon in substrate cores	0.32	-0.70	-0.34	0.37

Figure 35. Principal components analysis of vegetation and channel data. Stream sites are positioned with respect to factors 1 and 2.



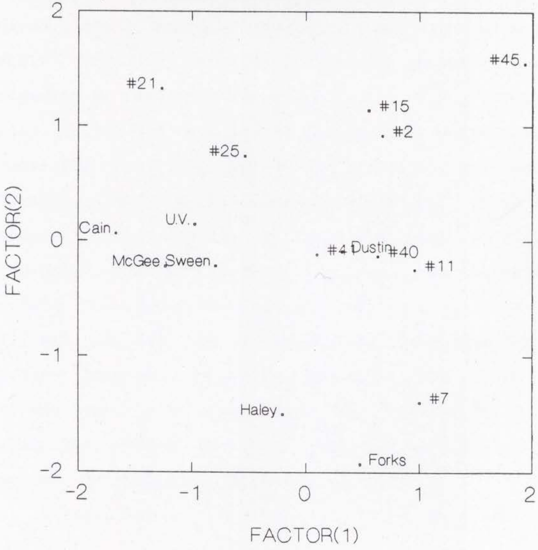
area. While the component loadings are quite low, the arrangement of stream sites is of interest: three of the reference streams were positively associated with factor 1, while Cains, McGee, and #15 were negatively associated with factor 1 (Figure 35). The remaining streams lay between these two groups, except for #11, which was situated to one side of the reference streams. Dustin and McGee had strong negative associations with factor 2, implying low shrub diversity, and high channel dimensions and organic matter concentration. Stream #25 was strongly influenced by stream size and tree diversity (which were positively associated with the third component). In general, the more recent cuts were situated closer to the three reference streams than the older cuts. Stream #40 was an exception in that it was the oldest cut, but had similar vegetation and channel dimensions to the more recent cuts.

All the substrate variables, including surface, streambed, and sediment pot measurements, were subjected to a PCA. The first component of a PCA of all the substrate variables explained 27.3% of total variance, and was highly positively correlated with bedrock and negatively correlated with coarse to very coarse gravels and total fines (Table 14). The second component, which accounted for 24.3% of the total variance explained, was strongly positively correlated with surface fines and negatively with cobble, while the third, accounting for 16.9%, was influenced by siltation,

Table 14. Principal components analysis of substrate variables with correlations between the first four principal components and the substrate variables. Dom. = Dominant; sfc. = surface; Subdom. = subdominant.

	Principal component			
	1	2	3	4
Eigenvalue	5.18	4.62	3.20	1.92
% Total Variance	27.29	24.32	16.85	10.11
Dominant surface bedrock	0.70	-0.06	0.27	0.56
Dom. sfc. cobble	-0.33	-0.48	0.12	-0.29
Dom. sfc. coarse gravel	-0.92	-0.04	0.12	-0.08
Dom. sfc. fine & medium gravels	-0.37	0.21	0.79	-0.03
Dom. sfc. fines	0.08	0.81	-0.32	-0.24
Subdominant surface bedrock	0.73	0.22	0.40	0.38
Subdom. sfc. cobble	0.19	-0.55	0.43	-0.50
Subdom. sfc. coarse gravel	-0.76	-0.46	0.06	0.14
Subdom. sfc. fine & medium gravels	-0.35	0.40	0.66	0.03
Subdom. sfc. fines	0.15	0.79	-0.19	0.15
Bedrock in substrate core	0.70	0.21	0.31	-0.49
Cobble in core	0.28	-0.76	-0.33	0.14
Medium and coarse gravels in core	-0.87	0.16	-0.14	0.38
Fines in core	-0.69	0.40	0.11	0.49
Sedimentation (>3.3mm)	0.15	-0.07	0.69	0.07
Sedimentation (<3.3mm)	0.33	0.40	0.69	0.17
75% Embeddedness	0.24	0.77	-0.50	-0.20
50% Embeddedness	-0.14	0.76	-0.11	-0.04
25% Embeddedness	-0.55	0.46	0.31	-0.55

Figure 36. Principal components analysis with substrate variables. The diagram depicts the ordination of sites based on factors 1 and 2.



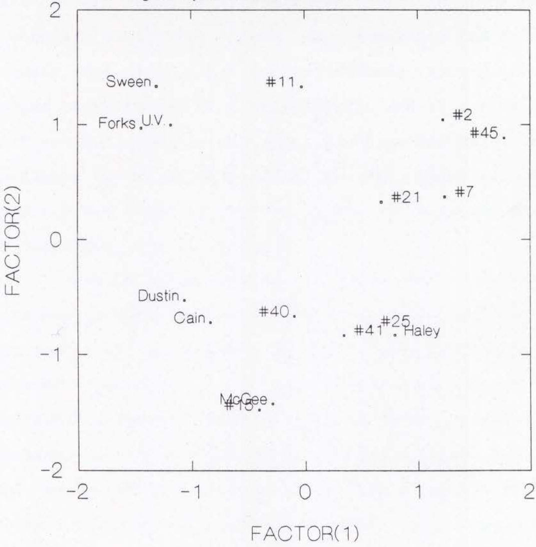
bedload movement, and fine to medium gravels (Table 14). Again, all of these values were quite low yet they provided an interesting ordination of the stream sites based on the substrate data (Figure 36). In Figure 36, three of the reference streams, excluding the Forks, were ordinated in proximity to each other, with negative associations to bedrock, and positive associations to coarse and very coarse gravels, and streambed fines. In contrast, the Forks had a positive association with factor 1, and a strong negative association with factor 2, which implied a negative correlation with core and surface fines, and medium and coarse gravels, and positive correlation with bedrock and cobble. Figure 36 indicated a distinct grouping of streams according to factors 1 and 3, with Cain, U.V., Sweeney, Haley, McGee, #25, and #21 in the top left corner, mostly negatively associated with bedrock and positively with streambed fines, gravels, and sedimentation. There also appeared to be a grouping of sites according to factor 2, with Haley, #7, and the Forks negatively associated with surface fines and positively associated with cobble; and #21, #15, #2, #25, and #45 positively associated with surface fines and negatively associated with cobble. Stream #45 was highly positively associated with factors 1, 2, and 3, implying strong negative or positive associations with most of the substrate variables.

A final PCA was used to simplify the data set. The first component accounted for only 27.6% of the total variance explained, and was highly positively correlated with surface bedrock, siltation, nitrogen, magnesium, and age of stand (Table 15). The second component, explaining only 22.8% of the total variance, was highly positively correlated with chloride, sodium, moss cover, canopy cover, and snag density, and negatively correlated with the maximum August temperature. The third component, explaining 14.8% of the total variance, was positively correlated with tree diversity and cobble. The ordination of streams based on this final set of variables can be seen in Figure 37. The three reference streams, Forks, Sweeney, and Upper Vault, are clumped in one corner of Figure 37, with positive associations to Cl, Na, moss and canopy cover, and snag density, and negative associations to surface bedrock, sedimentation, N, Mg, age of stand, and maximum temperature. They are distinguished from the newer cuts with respect to factor 1, but differ little with respect to factors 2 and 3. Thus the unlogged reference and recently cutover streams had similar values for Cl, Na, moss, canopy cover, snag density, maximum summer temperatures, tree diversity, and cobble, but differed with respect to concentrations of N and Mg, surface bedrock, and siltation. By contrast, the unlogged reference and older cutover streams both had negative associations with N, Mg, sedimentation and surface bedrock, but were distinct with respect to Cl, Na, moss and canopy cover, snag

Table 15. Principal components analysis of final set of summary variables with correlations between the first four principal components and the final variables. Arcsin = arc sine transformation; Log = log base 10 transformation; Sqr = square root transformation; Inv = inverse transformation.

	Principal component			
	1	2	3	4
Eigenvalue	4.13	3.42	2.21	1.45
% Total Variance	27.56	22.83	14.77	9.66
Inv Age of Stand	0.87	0.12	-0.01	0.13
Log Snag Density	-0.28	0.65	0.45	0.30
Tree Diversity	0.29	0.06	0.85	-0.10
Canopy Cover	0.12	0.78	0.17	0.05
Sqr Moss	-0.13	0.72	0.06	0.56
Log Chloride	-0.55	0.58	-0.43	-0.06
Log Magnesium	0.78	0.06	-0.20	-0.34
Log Sodium	0.24	0.69	-0.25	-0.46
Sqr Nitrogen	0.89	0.17	-0.11	0.09
Maximum Temp.	0.17	-0.82	0.14	0.39
Log Channel Width	-0.46	0.05	0.51	0.19
Surface Cobble	0.06	0.08	0.74	-0.45
Surface Bedrock	0.65	0.53	-0.05	0.27
Arcsin Surface Fines	0.26	-0.14	-0.28	0.45
Log Sedimentation	0.83	-0.16	0.26	0.12

Figure 37. Principal components analysis with a final set of physico-chemical variables. The diagram depicts the ordination of sites based on factors 1 and 2.



density, and maximum summer temperatures. Cains, the reference stream that was logged in the 1920s, behaved like the older cuts, in that it was similar to the other park streams with respect to factor 1, but differed with respect to factors 2 and 3.

4.3.2. Correspondence analysis

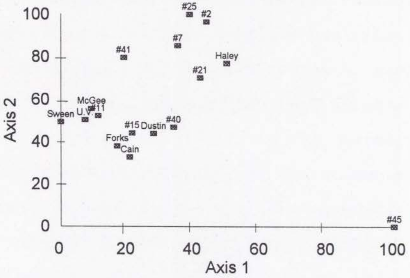
Correspondence analysis was used to ordinate streams based on the mean number of invertebrates per rockball for each stream. The first three axes accounted for most of the variance explained (86%) (Table 16). The results, shown in Figure 38, depict two clumps, one containing the reference streams, the three 1970s cutover streams, Dustin (which was logged prior to the more recent cut), and #11 (partially cut with a wide buffer). The other group consisted of the remaining 1980s cutovers except for #45, which was distinct from all the other streams as it was the only one dominated by simuliids, not chironomids.

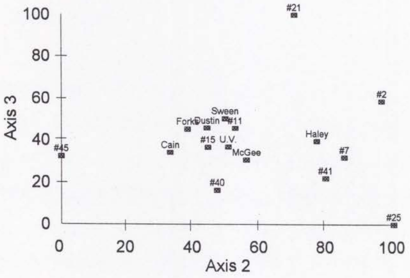
A more detailed picture was given with a correspondence analysis of each individual sample from each stream (as opposed to the mean number of individuals per rockball per stream) (Appendix 6). In these ordinations, we see that the variability amongst samples appeared to be greatest, in general, for the more recently cutover streams (#45, #2, #21, Haley, #7, #25) and least for the reference and older cutover streams (Sweeney, Upper Vault, Cain, Forks, McGee,

Table 16. The eigenvalues of the first four axes in the correspondence analysis. The percent worth of each eigenvalue is given.

Axis	Eigenvalue	Percent of total	Cumulative Percent
1	0.138	49.68	49.68
2	0.062	22.28	71.96
3	0.038	13.65	85.61
4	0.015	5.32	90.93

Figure 38. The arrangement of stream sites based on correspondence analysis of invertebrate data, using the mean number of invertebrates per rockball for each site. Axis 1 x Axis 2, and Axis 2 x Axis 3 are shown.





#40 and Dustin). Also, we can see that 4 of the 5 rockball samples from stream #45 were similar to those of other recently cutover streams, while the fifth was very distinct from all other rockball samples, and exerted a strong influence on the average, as seen in Figure 38.

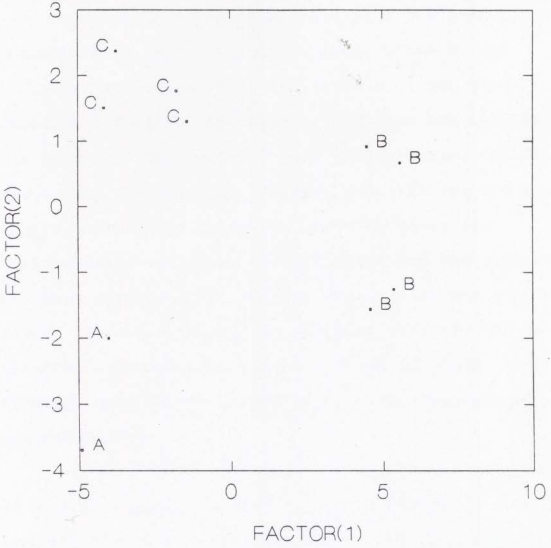
4.3.3. Discriminant function analysis

The division of sites into distinct groups based on various different sets of environmental variables was attempted with discriminant function analysis. F-statistics only showed highly significant differences among two 1980s cuts, one 1970s cut, and two reference streams on the basis of the maximum and range values of the thermograph data. Figure 39 shows the ordination of the stream sites, with each stream depicted by two points, representing July and August temperature data. Significant differences were also observed between three reference streams and the cutover streams, including Cains, on the basis of August average and maximum temperatures.

4.3.4. Canonical Correspondence Analysis

Of the 12 environmental variables put into the forward selected CCA, only one - age of the stand - explained a significant proportion of the variance of macroinvertebrate weighted averages ($P = 0.010$). Age of stand represented 33% of the total variance explained by all environmental

Figure 39. Discriminant function analysis using thermograph data. Group A includes stream #15, with no buffer strip; Group B includes streams #11 and #25, with buffer strips; and Group C includes the two reference streams, Sweeney and Upper Vault.



variables. In total, only 21% of the variance in the species data was accounted for by the environmental variables. Thus, the measured environmental variables could not predict most of the variation in the species composition of invertebrates.

Canonical correspondence analysis provides a combined ordination of the invertebrate taxa, streams, and environmental variables (Figure 40a,b). The direction and relative lengths of the arrows in Figure 40a and 40b infer the relative importance of each environmental variable for predicting the community composition. Figure 40b shows the approximate distributions of the taxa along each environmental variable. Oligochaetes had the highest weighted average with respect to siltation and surface fines. Chironomids and the Nymphomyiidae had the largest weighted averages with respect to age of stand, snag density, moss cover, stream width, and chloride concentrations.

Table 17 shows that age of stand ($r = 0.77$), nitrogen ($r = 0.72$), magnesium ($r = 0.70$), siltation ($r = 0.69$), bedrock ($r = 0.55$) and chloride ($r = - 0.60$) were strongly related to taxon axis 1, while bedrock ($r = 0.27$) and siltation ($r = 0.22$) were weakly related to taxon axis 2. Except for #11, all the more recently cutover streams were positively associated with Axis 1, while the reference and older cutover streams had near zero or negative correlations

Figure 40(a). Canonical correspondence analysis of invertebrate data and environmental variables. (a) shows the ordination of the 16 streams and 12 environmental variables.

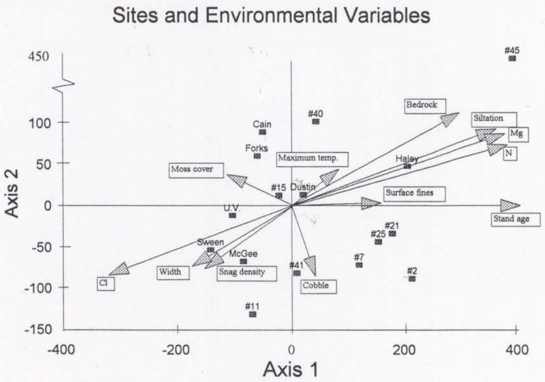


Figure 40(b). CCA, depicting the ordination of 20 taxa and 12 environmental variables.

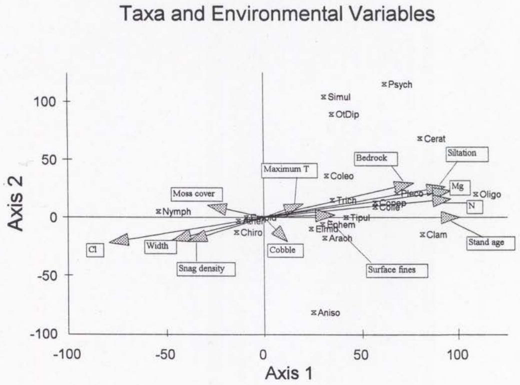


Table 17. Weighted correlation matrix (weight = sample total) of CCA, relating taxon axes to measured environmental variables.

	Axis			
	1	2	3	4
Surface cobble	0.078	-0.209	0.408	0.029
Moss cover	-0.209	0.093	-0.324	-0.119
Nitrogen	0.719	0.181	0.136	-0.041
Chloride	-0.599	-0.210	-0.586	0.054
Siltation	0.689	0.223	0.464	-0.079
Magnesium	0.703	0.201	0.235	0.504
Surface fines	0.281	0.004	-0.269	0.192
Maximum temp.	0.151	0.099	0.329	-0.608
Channel width	-0.336	-0.189	-0.048	-0.193
Snag density	-0.292	-0.179	-0.105	0.023
Age of stand	0.767	0.000	0.000	0.000
Surface bedrock	0.549	0.274	-0.087	-0.208
Fraction of Variance	0.27	0.33	0.16	0.11

with this Axis. This means that, in general, the reference streams tended to have higher than average snag density, chloride concentrations, and moss cover, and lower than average maximum summer temperatures, surface bedrock, fines, sedimentation, and concentrations of N and Mg. The more recent cuts had positive associations where the reference streams had negative, and vice versa. The older cuts were aligned somewhere between. The 2-dimensional depiction of these many relationships can, however, be somewhat misleading. For example, McGee is depicted with lower than average maximum temperatures and higher than average snag density, while in fact, McGee reached the second highest temperature recorded and had very few snags.

Stream #45 was strongly positively correlated with axes 1 and 2. From the correspondence and principal components analyses we know that this unusual pattern for #45 is partially a consequence of its invertebrate data, in which one particular sample on #45 had an unusually high number of simuliids. Stream #45 is also shown to have strong correlations with bedrock, siltation, aqueous concentrations of N and Mg, and surface fines. Stream #11 was also an exception, in that it was one of the more recently cut streams but was more closely associated with the reference and older cutover streams. This may be due in part to #11's large buffer, and its partially cut watershed.

5.0.0. DISCUSSION

5.1.0. Forestry-associated changes to habitat features

5.1.1. Water quality

Temperature

The data demonstrate that effects on a stream's thermal regime brought about by forest harvesting can be reduced by leaving forested strips along streambanks. The PCA ordination of maximum and average temperatures indicates that stream #2, in a recently cut watershed with a wide buffer strip, had a similar thermal regime to those of the three reference streams (Forks, Sweeney, and Upper Vault). Their water temperatures never exceeded 17°C, and the daily range rarely exceeded 4°C. Where no riparian vegetation was left along the cutover streams, high maximum temperatures and large daily fluctuations were recorded. These effects were apparently rather long-lasting: streams in two of the 1970s cutovers reached the highest temperatures, while Cains, tyhe watershed logged in the 1920s, had higher-than-expected maximum temperatures and daily temperature fluctuations. However, Cains may have been subject to other disturbances since logging, for example, agricultural uses, or flooding by beavers.

It is important to note that, even with buffer strips, the temperatures recorded in some streams were higher than expected when compared with the temperatures recorded in

Sweeney, Upper Vault and The Forks. For example, stream #45 had a 30-40-m-wide buffer strip along its sampling reach, and a variable-sized buffer strip (10-30-m-wide) upstream. Nevertheless, higher-than-expected maximum and average temperatures were recorded. Similarly, stream #25 was forested on the north side of the sampling reach, and had a 14-28-m protective strip along the south side. High water temperatures were also recorded on this stream. However, about 250 m above the sampling reach, the buffer strip disappeared, and the stream flowed unshaded through a large clearcut of low relief. Hewlett and Fortson (1982) also observed that deforestation in areas of gentle relief may elevate stream water temperatures, even with large riparian buffers. Sabean (1977), who measured stream temperatures in N.S., found that the re-entry of a stream into a canopied section resulted in a decrease in streamwater temperature, but at a slower rate than the initial increase when first exposed to direct solar radiation. He also observed that temperatures did not always return to their original levels.

Water Chemistry

Concentrations of nitrogen, potassium, and magnesium in streamwater were generally smallest in the reference streams, and greatest in the most recently cutover streams. Stream #45, cut five years before the study, and #7, most recently cut 4 years before, had 6-7 times greater N concentrations than the average of the reference streams. Also, the streams in the oldest clearcuts had slightly larger nutrient concentrations than those in the reference catchments, despite 15-20 years of regeneration. In a study of clearcutting in the Nashwaak watershed of west-central N.B., it was reported that higher than expected exports of K, Ca, Mg, and especially nitrate-N occurred for at least three years post-logging, with nitrate concentrations increasing from background levels of 0.1 mg/L to a maximum of 1.6 mg/L (Krause 1982). In this study, no immediate consequences of clearcutting were measured (i.e. within the first four years of harvesting), however elevated concentrations of dissolved substances in clearcut streams continued well beyond the time of cutting. Furthermore, the concentrations of streamwater N and K recorded in this study were probably smaller than they would have been at any other time of year, partly because nitrate concentrations increase during periods of high discharge, and partly because during the summer, biological activity reduces concentrations of nitrate and potassium in stream water (Johnson *et al.* 1969).

Stream #11 and Dustin Brook had similar water chemistry to that of the reference streams, as seen in the PCA ordination, while Cains fell between the oldest cutover streams and the other park streams. Thus, while Cains appears to be "leaky" compared with the other reference streams, #11 appears to have been well-protected by a buffer strip, while Dustin Brook remains an anomaly.

The longterm losses of nutrients from clearcut watersheds has various implications for site fertility. Losses of nitrate via leaching are accompanied by the loss of cations, including calcium, magnesium, and potassium, which are important as nutrients and in acid-neutralization capacity (Helyar 1976, cited by Patriquin *et al.* 1993).

Despite higher-than-expected losses of nitrates from the recently clearcut watersheds, no noticeable pattern in stream water pH was measured over the chronosequence of stands. This is in contrast to several other studies, which have documented either an increase in pH, due to elevated concentrations of Ca^{2+}, Mg^{2+}, K^{+}, and Na^{+} (Eshner and Larmoyeux 1963), or a decrease in pH, due to increased rates of nitrification in the surrounding soils (Likens *et al.* 1970). In addition, various studies of conversion of wetlands, moorlands and native-forest to tree plantations have documented the gradual acidification of soils and streamwater through the uptake of base cations by vegetation (Collier *et al.* 1989; Harriman and Morrison 1982; Jenkins *et al.* 1990). The most recently cutover stream, #7, had higher pH (7.3-7.6) and larger Ca concentrations than other streams. This observation could be related to soils or bedrock of #7, or it could indicate that samples from other streams were collected too many years post-harvesting to show any substantial effects. This could have implications for the acidification of the clearcut watershed, as well as for future site fertility.

Sedimentation

Of the 6 streams with the highest rates of sedimentation, four flowed below logging roads (#7, #2, and #45, #40). The first three of these streams were relatively steep and recently cut-over (within 7 years of the study), and discharge on the last, #40, was observed to increase considerably during rainstorms (the culvert had been dislodged above the sampling reach). In Narrow Mountain Brook of the Nashwaak Experimental Watersheds, N.B., sediment loads rarely reached 5 mg/l before road construction, but frequently exceeded this concentration thereafter, reaching 250 mg/l in one branch stream (Krause 1982).

Three other of the study streams were located below logging roads - #41, #21 and #11. Only one of these, #41, had higher than average substrate deposition for the sixteen study streams, and it was located approximately 1 km below a washed out logging road. The road above the sampling reach of #21 was quite old, and was well-vegetated around the stream crossing. The road above the sampling reach of stream #11, however, was still eroding into the stream channel. In Narrows Mountain Brook of the Naskwaak

Experimental Watersheds, it was found that concentrations of suspended sediment were greatly reduced by a riparian strip (Krause 1982). This may have been the case for #21 and for #11, which was protected by a wide buffer strip along one side, and a forest on the other.

Sediment loads are also increased by high stream velocities (Ontario MNR 1988). Stream #25 drained a large, nine-year-old clearcut that was only partially revegetated. The three sediment pots in this stream could have been buried as an indirect result of higher stormflows, which, according to Smith *et al.* (1993) would have increased total boundary shear stress acting on grains in the channel, as well as destabilized debris dams. In turn, this would have led to shifts in debris location and the release and scour of stored sediment. Alternatively, the sediment pots in stream #25 may have been washed away by stormflows. This is unlikely, however, as the sediment pots in both stream #2 and the Forks stayed in position, and the high gradient of both these streams would have resulted in large peakflows.

Sedimentation was negligible for Sweeney, Upper Vault, and the Forks, and slightly higher for Cains. Sediment movements in watersheds of mature forest are minimized by smaller peakflows, stable debris dams, and stable stream banks (Bormann and Likens 1979). Cains may still be recovering from past logging activities or some other occurrence during which riparian vegetation was cut or otherwise disturbed. The resulting scarcity of large trees and snags has reduced the amount of large woody debris entering the stream channel, without which stable debris dams cannot form. Small quantities of sediment were collected in the sediment pots of stream #15 and Dustin Brook, despite flowing through highly disturbed watersheds. This may be due to several factors: first, both streams had large numbers of debris dams (full of sawn timbers), which reduce flow velocities and store sediments; and second, previous log-driving in Dustin Brook would have scoured the streambed of fines, such that there was still a small proportion of fines in the substrates (as indicated by the substrate core analysis).

5.1.2. Stream substrates

The results of the substrate core analysis suggest that clearcutting has had little measurable effect on the composition of stream sediments. This is in keeping with Duncan and Ward (1985), who noted that the composition of stream gravels, and the amount of fine sediments (<2 mm in size), are more closely correlated with the lithology and soils of the drainage basin and the specific climatic, vegetative, topographic, and hydrologic factors present, than with forest management activities.

On the other hand, it should be noted that the surface substrate estimates showed significant differences in surface fines and coarse gravels between the four reference and the 12 cutover streams. These results agree with Cederholm *et al.* (1981, as cited in Duncan and Ward 1985), who showed that the percentage of sediment <0.85 mm in spawning gravels increased in proportion to basin area affected by timber harvesting and roads. Similarly, Adams (1980, as cited in Duncan and Ward 1985) found that watershed hydrology, especially magnitude of peak flows, was of particular importance in determining the composition of streambed gravels. Though water yields were not measured in this study, it is well accepted that peakflows increase with clearcutting of most or all of a watershed (Hetherington 1986).

The proportions of surface fines and embeddedness were greatest for #21, #45, #41, and #15. The first three of these streams flowed under logging roads, and #15 appears to have been used as a skidder trail. Eaglin and Hubert (1993) measured the amount of surface fines and embeddedness in 28 stream reaches affected by clearcutting and logging roads in Wyoming, and found positive correlations with the extent to which roads crossed watercourses within a drainage, and with the proportion of the drainage that was logged. However, streams #2, #40, #7 and #11 also flowed below roads, but did not have high proportions of surface fines.

5.1.3. Channel features

Streambank cutting reduced shading and the long-term input of large woody debris (LWD) into the stream channel. We did not quantify LWD in stream channels, but we can infer the future situation from the nature of the riparian vegetation. The lack of snags and large trees along Cains, McGee, #15 and Haley, and stumps in the riparian zones of #11, #40, Dustin and parts of #7 suggest that these streams will experience long-term large woody debris impoverishment. The consequences of this are: a reduction in channel complexity and number and volume of pools, thereby decreasing current heterogeneity and increasing stream power and erosiveness; a decrease in the amount of cover for stream biota; and a diminished capacity to retain, store, and process organic and inorganic matter (Trotter 1990; Hicks *et al.* 1991; Ehrman and Lamberti 1992). Large woody debris in streams draining clearcuts is further reduced by rapid breakdown and by the destabilization and erosion of natural debris dams (Golladay *et al.* 1989). A study done in Alaska showed that LWD in stream channels would be reduced by 70% ninety years after streambank logging of an old-growth forest, and recovery to pre-cutting amounts would take more than 250 years (Murphy and Koski 1989).

Streambank logging and selective removal of large trees in the riparian area results in decreased pool depth and hydraulic variability, and increases the frequency of riffles and runs (Welch *et al.* 1977; Murgatroyd and Ternan

1983; Sedell *et al.* 1988; Trotter 1990). Small pieces of debris entering streams in the postcutting phase tend to be flushed from the watershed, and scour the stream bottom of sediment and debris. Sedell *et al.* (1988) maintain that natural accumulations of large woody debris (on the west coast) have been observed to deflect current laterally, causing lateral migration of the channel, and producing midchannel bars, secondary channels and braided reaches. In this study, pre-logging channel data is unavailable. However, the paucity of large trees and snags along several streams indicates that the morphology of these streams will have been affected by a lack of LWD.

Stream width and depth are also affected by other factors, such as sediment and water input (Sullivan *et al.* 1987), mechanical disturbance, and successional vegetation. Alders that colonize disturbed riparian areas have shallow root systems, giving them little resistance to undercutting (Sedell *et al* 1988). Dustin Brook, for example, was bordered by alders, and was also the widest stream measured. Disturbance of stream banks by machinery may also widen stream channels, as was the case with the trail crossing below the sampling reach of Sweeney Brook, which reached 8.1 m in width at an unbridged crossing.

5.1.4. Riparian Vegetation

The density of snags and shrub-sized species in riparian areas provided an indication of disturbance history, as well as an indication of future ecosystem health. The absence of large trees and snags in a riparian forest indicates some form of disturbance: streambank or selective logging in the riparian zone, agricultural use, disease, windthrow, or flooding. The streamside forests of Cains and Dustin had small densities of large trees and snags, substantiating evidence of past disturbance. The high densities of shrubs along these and other study streams without wide buffer strips (McGee, #15, #7, Haley) showed that shrub-sized species, particularly *Alnus rugosa*, predominated in disturbed areas where snag density was small. This shift from mostly long-lived, large-sized canopy trees to small-stemmed, low-canopy, fast-growing shrub species can have a significant influence on the stream environment. The ecological functions of aquatic macro-invertebrates in headwater streams are strongly dependent on the type of energy inputs available (Stout *et al.* 1993). Initially, clearcutting riparian forests reduces the volume of leaf litter reaching streams. Rapid regrowth of early successional herbs, shrubs and trees allows for the recovery of inputs, but of a different quality. For example, leaves from many early successional forest trees are processed relatively rapidly and inefficiently, while those from mature forests are processed slowly or at moderate rates (Stout *et al.* 1993). Where coniferous plantations replace mixed-forests, the stream would receive fewer litterfall inputs, with a smaller seasonal peak in the fall (Webster and Patten 1979). Following perturbations such as clearcutting, complete recovery of the stream ecosystem is limited by the recovery rate of allochthonous inputs.

5.1.5. Overall patterns in habitat data

One important finding in the principal components analysis of a reduced set of physico-chemical factors was that the three reference streams, Sweeney, Forks, and Upper Vault, were closely associated with respect to the measured environmental variables. Cains differed from the other reference streams in its associations with chloride, sodium, moss cover, canopy cover, snag density, maximum August temperature, tree diversity, and cobble. According to the analysis, Cains was more like the older cutover streams than the other reference streams, and appeared to be recovering from some previous disturbance(s). Biotic regulation of major nutrient losses, sedimentation, and bedload movement had been restored in the Cains watershed, but more time will pass before large woody debris input, moss cover, canopy cover, riparian tree diversity, and channel and current heterogeneity reach pre-disturbance levels.

In retrospect, Cains was not a good choice as a reference stream; however, it was not known initially that Cains had been cut in the 1920s, or that it could have been an old pasture, or the site of a beaver dam. Similarly, the history of logdrives and cutting in the Dustin Brook catchment was not known before selection of this stream. It appears that these past perturbations may have confounded some of the results of this study, such that both streams behaved more like streams draining older 1970s clearcuts than the more recently cutover or older, reference streams. On the other hand, inclusion of these watersheds in this study has produced some potentially interesting insights into the longer-term effects of logging.

Another important finding in the principal components analysis was the clumping of the five most recently cutover streams and the three reference streams according to factors 2 and 3. This would imply that recently cutover streams were fairly well protected by buffer strips. Nevertheless, they differed with respect to nutrients, sedimentation and bedrock. Of these five streams, #11 appears to be most closely associated with the three park streams, possibly because only 65% of #11's watershed was cut, and one side of the sampling reach was left forested. We know from other studies that the impacts of clearcutting, such as increased discharge and sedimentation, increase with the proportion of the watershed that is logged and the extent to which roads cross watercourses (Eshner and Larmoyeux 1963; Rich and Gottfried 1976; Growns and Davis 1991; Eaglin and Hubert 1993).

5.2.0. Implications for fish habitat

Brook trout (*Salvelinus fontinalis*) are common throughout the freshwaters of Fundy National Park and the surrounding area (Woodley 1985). Brook trout abundance was not measured in this study, but according to Raleigh (1982) and Carlson *et al.* (1990), it is possible to utilize the measured physical and chemical variables to make inferences about the suitability of streams as trout habitat (Raleigh 1982; Carlson *et al.* 1990). Trout were observed in Cains, McGee, #41, #40, #21, #45, #25, and Haley. Large, impassable debris dams of logging slash may have prevented the movement of brook trout into the sampling reaches of #7, #15, and Dustin Brook. Most trout were observed in pools, deep runs, or below undercut banks.

Optimal brook trout habitat is characterised by clear, cold spring-fed water, silt-free rocky substrates, a 1:1 pool to riffle ratio, areas of slow deep water, well-vegetated streambanks, abundant instream cover, and stable water flow, temperature regimes, and streambanks (Raleigh 1982). According to Raleigh (1982), warm water temperatures are the single most important factor limiting the distribution and production of brook trout. The temperatures recorded on #15, #25, and McGee reached lethal levels for brook trout fry and adults (>23°C), while the large fluctuations and maximum temperatures observed in #41, #40, and #10 were sufficient to cause physiological stress (van Groenewoud 1977; Lynch *et al.* 1984). Nevertheless, brook trout were observed in four of these six streams: #41, #40, #25 and McGee. This is probably because (1) the maximum temperatures recorded were short-lived, and thus tolerable, and (2) refuge could be sought in cooler places. It is possible that higher-than-average temperatures recorded on cutover watersheds, in combination with nutrient enrichment, may have increased primary productivity, thereby increasing insect and trout production (Englert *et al.* 1982).

The dissolved oxygen concentrations measured in September, 1993, would not cause immediate injury or mortality to most stream organisms, but long-term exposure to the lowest observed O_2 concentrations (7.7 and 8.2 mg/l) would have been harmful to brook trout. Optimum oxygen concentrations for brook trout are > 7 mg/l at temperatures < 15°C, and > 9 mg/l at temperatures > 15°C (Raleigh 1982). The swimming performance and growth rates of salmonids are adversely affected as O_2 falls below these optimum concentrations. There were hot days in July and August when O_2 concentrations would have been considerably lower than those measured in September. Furthermore, the summer of 1993 was abnormally cool and wet, which suggests that dissolved oxygen concentrations were lower than those typically recorded.

Large woody debris plays an important role in creating and maintaining water depth and cover, both of which are essential components of good salmonid habitat. Instream logs produce a diversity of hydraulic gradients; create large, deep pools with ample cover and low velocity, enabling fish to hold their position against the current; and provide cover in the form of overhanging vegetation, submerged logs, stumps, roots, water depth, and undercut banks (Sedell *et al.* 1988; Hicks *et al.* 1991). Removal of LWD decreases the number and size of pools, and is often linked to a decrease in abundance of salmonids (Sedell *et al.* 1988). The absence of buffer strips along all or part of #15, McGee, #25, and #7, and of snags and large trees along Cains and Dustin indicates that these streams will have little natural input of LWD for decades to come.

Trout habitat quality is related to annual flow regime. Clearcutting of a watershed increases discharge during the plant growing season, which can result in greater baseflows during the critical summer low-flow period. Alternatively, the higher peak flows may destabilize spawning gravels, and enhance scouring, erosion and sedimentation (Reiser and Bjornn 1979; Englert *et al.* 1982). Higher rates of sedimentation and higher proportions of surface bedrock were measured in the more recently cutover streams. Brook trout are sensitive to both suspended and deposited enhance sediments. High turbidities may cause direct injury by coating gills - impairing water circulation and respiration, as well as indirect injury, by reducing visibility, thereby impairing the trout's ability to locate food (Raleigh 1982). Silt can also smother and suffocate eggs and alevins buried in stream gravels, and reduce the productivity of invertebrates (Cordone and Kelley 1961). Spawning fish tend to avoid silty areas, unless upwelling ground water is present (Meehan and Bjornn 1991). Elevated rates of sedimentation were observed in some cutover streams. It is possible that all the cutover streams had elevated sediment loads at one time or another, but have recovered since the time of cutting.

5.3.0. Effects of forestry activities on invertebrate taxa

Oligochaeta

The majority of oligochaetes are adapted to living in sediments ranging from sand to mud (Brinkhurst and Gelder 1991). In this study, we found the greatest numbers of oligochaetes in the rockballs of stream #21, #45, #2, and Haley. The latter 3 had high to medium rates of sedimentation and bedload movement, while #21 had a high percentage of streambed fines. From Figure 40b we see that, in this study, oligochaetes were closely associated with sedimentation, as well as with the more recently cutover streams. The greater abundance of oligochaetes colonizing rockballs in the more recently cutover streams agrees with the research of Cordone and Kelly (1961), who found that a change in the type of invertebrate species occurs with deposition of sediment, such that sediment-loving, burrowing organisms become more prevalent.

Chironomidae

Chironomids are the largest family of aquatic insects, often occurring in high densities and diversity, and accounting for at least 50% of the total macroinvertebrate species diversity in streams (Coffman and Ferrington 1984). Their life cycles and feeding habits vary widely (Hilsenhoff 1991): most chironomids are herbivores or detritivores, grazing on fine particles on and in the substratum, or filtering food from the water column. Chironomids have been found in a great range of conditions, including extremes of temperature, dissolved oxygen, pH, salinity, velocity, depth, and productivity. They also have short generation times and rapid colonization rates, enabling them to cope with fluctuating environments, and to build up large populations opportunistically (Newbold *et al.* 1980).

Because numerous studies have shown increases in chironomid abundance with disturbance, it might be expected that chironomids would constitute a relatively large proportion of the post-disturbance stream fauna. Sprules (1947, cited in Hynes 1970) found that the deposition of sandy silt reduced the total numbers of emerging insects, but increased the proportion of emerging chironomids. In

British Columbia, Culp and Davies (1982) observed that elevated inputs of alder detritus in the post-logging phase increased the biomass and density of Chironomidae and the mayfly *Baetis*. Similarly, both Newbold *et al.* (1980), and Noel *et al.* (1986) reported higher densities of total fauna, in particular Chironomidae and *Baetis*, in streams running through cutovers.

In this study chironomids numerically dominated most samples, but many more individuals were found in streams draining reference and older clearcuts than in streams draining more recent cuts. This may have partially been a consequence of the mesh size used to rinse invertebrates in the laboratory: tiny individuals of the subfamily Chironominae, known to tolerate warmer waters and low oxygen concentrations, could have slipped through the 300 µm mesh. Alternatively, habitat heterogeneity in the streams draining recent clearcuts could have been reduced by the deposition of fines and by scouring action, such that chironomids had fewer refuges and were more susceptible to predation (Wilzbach *et al.* 1986). Many plecopteran species, for example, are predatory, and plecoptera were more abundant in the rockballs placed in recently cutover streams. Walde and Davies (1984) showed that plecopteran predators can reduce total prey biomass, mean prey size, and the density of invertebrate populations. Alternatively, as colonists of wood, chironomids may have been affected by the quantity of woody debris in stream channels (Golloday and Webster 1988). Woody debris was not, however, quantified in this study.

Plecoptera

Plecoptera, or stoneflies, are associated with clean, cool running waters (Harper and Stewart 1984). They lack extensive gills and are generally intolerant of low dissolved oxygen concentrations. Most species are univoltine or semivoltine, and are thus unable to respond quickly to disturbance. Plecoptera are usually shredders or predators, feeding on leaf detritus or smaller insects.

In this study, Plecoptera displayed a smaller abundance in the rockballs in older stands. This might suggest that plecopteran species found in rockballs in this study are associated with conditions typical of disturbed habitats, such as bedrock, sedimentation, N and Mg concentrations, surface fines, and high maximum summer temperatures, than with conditions typical of the reference streams, such as high moss cover, Cl, and snag density. Wilzbach *et al.* (1986) found that high habitat complexity increased prey refuges, and in turn, decreased predator abundance. This would suggest that the higher number of Plecoptera in the streams in recent cutovers may reflect greater hunting success, which is a consequence of lower habitat complexity (such as high rates of sedimentation). It is unlikely that the smaller number of Plecoptera in the reference streams was a consequence of litter inputs, because the vegetation along reference streams was similar to the vegetation along the more recently cutover streams, as indicated by the principal components analysis of vegetation data.

Nymphomyiidae

Palaeodipteron walkeri is the only species of the rather primitive and rare Nymphomyiidae family found in the study region (Borror *et al.* 1989). The larvae feed on diatoms and algae in pebble and moss habitats of small, cold, rapidly flowing streams (Back and Wood 1979; Teskey 1984; Hilsenhoff 1991). Large numbers of *P. walkeri* are found as far north as the middle subarctic zone of Quebec, where average annual air temperatures are -3.7°C (Back and Wood 1979). In this study, no Nymphomyiidae were collected in rockballs in the seven most recently cutover streams, while a few were collected from each of the reference streams. The canonical correspondence analysis suggests a strong association between Nymphomyiidae and the reference streams; as well as a positive association between Nymphomyiidae and moss cover, snag density, stream width, and chloride concentrations; and a negative association with bedrock, sedimentation, surface fines, Mg and N concentrations, and high temperatures. *P. walkeri* did not occur in streams with measurable amounts of sedimentation. However, the greatest number of individuals was found in

#15, which was slow-moving and reached some of the highest temperatures recorded.

Thus, the small, delicate Nymphomyiidae may not be as sensitive to warm waters as the literature suggests. These insects may, however, be quite sensitive to some habitat changes associated with clearcutting. Further study may demonstrate that they could be used as an indicator of ecological integrity of streams.

Elmidae

Elmidae ("riffle beetles") usually live in oxygen-rich waters, and are typical inhabitants of the swifter portions of streams (White *et al.* 1984; Hilsenhoff 1991). Elmidae are semi-voltine, and feed on algae, decaying wood and detritus. In this study, Elmidae most abundantly colonized rockballs in older cuts, and least abundantly in recent cutovers and reference streams. The older cutover streams differed from the other streams with respect to chloride, sodium, moss cover, canopy cover, snag density and maximum August temperature. It is possible that the lower canopy closure and higher temperatures measured in these streams resulted in higher primary productivity, allowing a greater abundance of algae-eating Elmidae (Gurtz and Wallace 1984). Alternatively, Elmids in headwater streams are most commonly found clinging to moss on rocks, and with an abundance of moss in streams draining recent clearcuts as well as reference stands in this study, these insects are unlikely colonizers of gravel-filled rockballs (D. Giberson, pers. comm. 1995).

Simuliidae

Simuliidae are found in swift currents, adhering to solid surfaces, straining diatoms and other microscopic food particles from the water (Peterson 1984). Many species have short generation times, are highly opportunistic, and rapidly colonize disturbed areas. As filter-feeders, blackflies are moderately tolerant of sedimentation, particularly when the sediment has a large proportion of organics. Too much sediment, however, can have a smothering effect on blackflies (D. Giberson, pers. comm. 1995). In this study, streams with the largest proportions of surface fines and the highest rates of sedimentation did not have a smaller abundance of simuliids. In Czechoslovakia, simuliid larvae developed more rapidly in areas where streams received direct sunlight (Olejnicek 1986). This was thought to be a response to higher temperatures and the more rapid development of microorganisms suitable as food for simuliid larvae. In this study, however, no evidence of greater simuliid densities was found in the warmer, exposed reaches of the study streams. It is possible that rockballs were unsuitable for sampling blackflies, due to their contagious distribution of these insects.

5.3.1. Overall trends

There were fairly distinct differences between the invertebrate communities colonizing rockballs in the reference and older cutover streams, and those of the more recently cutover streams. Stream #11 and Dustin Brook were, however, exceptions. Stream #11 was most recently clearcut in 1986, and Dustin Brook in 1983. The invertebrate communities of both streams were more like the reference and older cutover streams than the more recently cutover streams. For #11, two partial cuts over several years may have resulted in a less-severe disturbance than would have occurred with a faster clearcutting of the watershed. Alternatively, both Dustin Brook and stream #11 had relatively high concentrations of organic carbon in streambed substrates. The larger-than-average amounts of detritus in these streams may have offset some effects of clearcutting.

This distinction between the reference and older cuts, and the more recent cuts was apparent in the results of the principal components analysis, in which the sites were arranged on the basis of physical and chemical variables alone. These two groupings differed with respect to surface bedrock, sedimentation, nitrogen, magnesium, and stand age. It would appear that these variables play a significant role in distinguishing the invertebrate communities. However, according to the canonical correspondence analysis, *only one* *variable* - stand age, had a significant influence over the invertebrate communities in this study. Thus *none* of the measured physical or chemical variables were able to explain the patterns in invertebrate abundance over the chrono-sequence of cutover and reference streams.

A common dispute among stream ecologists is whether streams are physically- or biologically-controlled environments. Some would argue that abiotic factors such as temperature, water chemistry, substrate, and current velocity are of far greater importance to the stream community than biotic factors, such as food availability, competition, and predation (Hart 1983). In this study we did not record biological characteristics such as algal standing crop, large woody debris, detritus collected in rockballs, abundance of brook trout and predatory insects, or competition amongst aquatic invertebrates, nor did we measure current velocity or discharge. Perhaps one or more of these unmeasured factors had a strong influence on the invertebrate assemblages found in this study. Among the variables that were measured, however, stand age was paramount in explaining the variation in the invertebrate data.

5.4.0. Problems with methodology

The device used in this study to sample invertebrates undoubtedly had a strong influence on the results. Minshall and Minshall (1977, cited in Silsbee and Larson 1983) found significantly larger numbers of organisms colonizing trays of introduced substrate than in natural stream substrate, as well as large differences in species composition between introduced and natural substrate. While the samples collected in their study may not have accurately described the local fauna of the streams, they found that the differences between streams were consistent between sampling methods, and probably represented real differences in the benthic communities. Coleman and Hynes (1970) found many times more animals in fully colonized samplers extending 30 cm into the streambed, than in Surber samples collected at the substrate surface. This would further suggest that the invertebrate assemblages sampled in this study do not accurately represent the complete invertebrate community of the study streams, although they do provide a repeatable, standardized indicator of colonization under differing environmental conditions and a constant substrate.

Other problems with the invertebrate sampling procedure include: sampling of riffle habitats only, sampling for one year only, and the use of 300 µm mesh, such that smaller-sized organisms would have escaped.

5.5.0. Conclusions

5.5.1. Effects of clearcutting and road-building on stream

ecosystems in the Fundy area

Biotic regulation of nutrient exports appeared to recover with time since cutting, with the smallest concentrations of dissolved substances recorded, in general, in the reference streams, and the greatest concentrations in the streams draining the most recent clearcuts. Stream #45, draining a five-year-old clearcut, and #7, most recently 30% cut 4 years before, had 6-7 times greater N concentrations than the average of the references. Therefore, elevated concentrations of dissolved substances in streams draining clearcuts continued for at least five years after cutting.

Continued maintenance of logging roads, road washouts, and peakflow events complicated monitoring of the recovery of sediment exports with time since clearcutting. Some of the lowest rates of sedimentation were recorded in the references Sweeney, Upper Vault, and the Forks, as well as in Dustin, draining a 1983 clearcut, and stream #15, draining a 1979 clearcut. However, one of the highest rates of sedimentation was measured in the sampling reach of stream #40, draining a 1973 clearcut, but located directly below a washed out logging road.

The thermal regimes of streams draining recent clearcuts were more like the thermal regimes of the reference streams than those of streams draining 1970s clearcuts. This is because legislation introduced in the early 1980s required that streams be protected from clearcutting by buffer strips. Before this, buffer strips were rarely left along streams, with the consequence that, in 1993, streams like #40, #15 and McGee still had higher maximum temperatures and greater diurnal fluctuations than the streams clearcut in the 1980s. Streams draining partially cutover watersheds, or with wide buffer strips along their length, had similar thermal regimes to the reference watersheds (especially stream #2).

The presence of buffer strips along streams draining recent clearcuts also ensured the rapid recovery of allochthonous inputs of biomass. Principal components analysis demonstrated that the vegetation along the three reference streams, Forks, Sweeney and Upper Vault, was most similar to the vegetation along the streams draining more recent clearcuts with relatively wide buffer strips (#45, #25, #11, #21, #2). These 8 streams were characterised by low shrub densities, high in-stream moss cover, and higher-than-average tree and snag densities. By contrast, vegetation along streams cutover in the 1970s and early 1980s, including stream #7 (cut in 1982, 1986 and 1989) and the reference, Cains (watershed cut in the 1920s and potentially disturbed since then), indicated that these streams will experience long-term impoverishment of large-dimension woody debris. Shrub density was high, and the most common shrub-sized species was *Alnus rugosa* or, occasionally, *Abies balsamea*.

Overall, the three reference streams, Sweeney, Forks, and Upper Vault, had many similar properties, with relatively high values for chloride and sodium concentrations, moss and canopy cover, and snag density; and relatively low values for surface bedrock, sedimentation, nitrogen and magnesium concentrations, and maximum August temperatures. The reference streams were distinct from the streams draining older clearcuts in their associations with chloride and sodium concentrations, moss and canopy cover, snag density, maximum August temperature, tree diversity, and cobble. Streams draining recent clearcuts were moderately well protected by buffer strips, but differed from the reference streams in their associations with nutrients, sedimentation and surface bedrock. Stream #11, with a wide buffer and partial cutovers in 1984 and 1987, was closely associated in many ways with the reference streams. In summary, the recovery of streams draining clearcuts was accelerated by buffer strips. Complete recovery of stream ecosystems is slower for streams left unprotected by trees.

There was a steady increase in the number of invertebrates colonizing rockballs with increasing age of stand. The greatest abundance was found in the rockballs on McGee, Sweeney, Cain, and Upper Vault. Chironomids were the most common group, and were largely responsible for this increasing trend in abundance. The small number of

Nymphomyiidae, represented by the species *Palaeodipteron walkeri*, also increased in abundance with age of stand, and were strongly associated with moss cover and snag density. Plecoptera, on the other hand, decreased in abundance with increasing age of stand, as did the Oligochaeta, which were correlated with sedimentation and surface fines. Few Elmidae were found in the reference and recently cutover streams, and more in the older cutover streams, while few Tipulidae were found in the older cutover streams and more in the recently cutover and reference streams.

Correspondence analysis indicated two groupings of streams based on the invertebrate data: (1) the reference streams, 1970s cutovers (#15, #40 and McGee), Dustin and #11; and (2) the 1980s cutovers. Although this differentiation appears to mimic groupings established in the principal components analysis, results of the canonical correspondence analysis reveal that the measured environmental variables could not predict most of the variation in the invertebrate assemblages. In fact, only one variable - age of stand - explained a significant proportion of the variance in the invertebrate data. The use of artificial substrates to sample stream invertebrates may have had a strong influence on these results.

Forest cutting and road building were not, for the most part, beneficial to trout habitat. The high temperatures in six cutover streams were sufficient to cause physiological stress, and on three cutover streams, mortality. The dissolved oxygen concentrations in at least two cutover streams may have adversely affected swimming performance and growth rates. High rates of sedimentation and bedload movement in at least three of the cutover streams could have destabilized spawning gravels, degraded spawning habitat, damaged gill membranes, and potentially interferred with foraging and food supply. Selection- and clear-cutting in the riparian zones of six streams have depleted the long-term supply of inputs of large woody debris. This will eventually degrade trout habitat by decreasing the number of pools, amount of cover, and retention of fine sediments.

The results of the principal components analyses have justified the use of a chronosequence approach to study the recovery of stream ecosystems following forest harvesting. This approach assumes that one can substitute a longer-term study of one individual stream before, during and after harvesting by using many streams draining different-aged clearcuts, and comparing these to reference streams draining mature forest. Although within-stream and between-stream conditions vary both spatially and temporally, it was hoped that patterns related to harvesting would emerge, despite differences in topography, surficial geology, watershed size, vegetation type, etc. Indeed, with no *a priori* grouping of streams on the basis of cutting history, a final principal components analysis of environmental variables depicted three distinct clumps of streams: the reference streams, the streams draining recent clearcuts, and the streams draining older clearcuts.

5.5.2. Recommendations for the protection of stream habitat in the vicinity of Fundy National Park:

1. **Manage stands of timber to supply longer-term inputs of large woody debris (LWD) to streams. This should preclude selective harvesting of large trees in buffer zones, which is permitted in the proposed Watercourse Buffer Zone Guidelines for Crown Forestry Activities.** Current guidelines suggest a minimum buffer width of 15 m, and allow selective harvesting within the buffer of no more than 30% of the merchantable basal area. This includes trees from across the range of diameters and species present. The very low density of snags along stream #15, McGee, Cain, and Haley, and presence of sawed stumps in the buffer zones of stream #11, #40, Dustin Brook, and parts of #7 indicate that there will be little long-term input of large woody debris into these streams, as well as little habitat for cavity-nesting mammals and birds. While there was a high density of snags in the riparian zone of stream #11, most of them were small trees, which decompose more rapidly than large-sized trees, and have a shorter residency time in the stream

channel. Inevitably, allowing cutting in buffer zones will result in larger trees being cut, and the depletion of valuable inputs of LWD into stream channels.

2. **Leave a buffer strip along the full length of the stream as well as protect feeder streams and springs.** Current guidelines suggest leaving a buffer strip $\geq$ 15-m along streams $\geq$ 0.5 m in width, and a 3-m buffer along streams < 0.5 m in width. However, thin or non-existent buffer strips along the upper parts of streams #45, #25, and #41 (all cutover in the 1980s) have contributed to the large temperature fluctuations and high maximum temperatures measured on these streams. Stream #25 flowed unprotected through a clearcut for at least 500 m, then entered a fairly wide buffer. By that time, stream temperatures could be very high, and remained high in the sampling reach, approximately 500 m downstream of the unbuffered clearcut. Furthermore, inputs of leaf detritus, the primary source of food energy in headwater streams, would have been seriously reduced by the replacement of riparian forest with an even-aged conifer plantation in the headwaters of stream #25.

3. **Where past logging activities have removed riparian vegetation, it may be useful to add digger logs to stream channels as a mitigation.** Clearcutting up to the banks along all or parts of stream #7, #25, #15, McGee and #40

will result in the longterm depletion of large woody debris in these stream channels. Placement of "digger logs" into degraded watercourses has been shown to increase the number of pools, sort gravels, create a diversity of physical habitats, reduce ice production, and stabilize stream ecosystems by retaining allochthonous inputs and reducing stream velocities (Rutherford pers. comm. 1995).

4. **When possible, cuts should be partial or selective, instead of clearcutting an entire watershed.** In this study, it was observed that stream #11 was more like the reference streams than other, more recently cutover streams. This could be because the watershed of stream #11 was not entirely clearcut: 35% was cut in 1984 and 30% in 1987. Perhaps the likeness of this stream to the reference streams had little relation to logging, however, various other studies have also documented decreased impacts of logging with the proportion of the watershed that is logged (Eshner and Larmoyeux 1963; Rich and Gottfried 1976; Martin and Pierce 1980; Krause 1982; Campbell and Doeg 1989; Growns and Davis 1991; Eaglin and Hubert 1993). According to Martin and Pierce (1980), nutrient exports are minimized by cutting lower portions of watersheds instead of upper portions, and by favoring progressive strip cutting or selective cutting over other methods of forest harvesting.

5. **Plan roads so that they avoid stream crossings.** High rates of sedimentation and high proportions of surface fines were not a uniform response to logging roads in this study. Nevertheless, of the 6 streams that had more than 100 g of sediment per pot, 4 crossed below logging roads. Of the 7 streams that had less than 20 g per pot, 6 did not have road-crossings. Two roads had washed out, resulting in the addition of large quantities of sediment into the stream channel. Currently, logging roads are built straight, with little regard to the topography of the land. As a result, they often cross many stream channels that could be avoided. With extra effort, roads could be planned to avoid as many stream crossings as possible.

6. **Where streams must be crossed, always build adequately-sized culverts or bridges.** Of the 12 streams draining cutovers, 7 crossed logging roads, and of these 7 logging roads, 2 had washed out. It seems that road engineers often underestimate streamflows (especially peakflows), and place under-sized culverts at stream crossings.

7. **Logging roads should either be maintained, to prevent erosion, or removed.** Active erosion of logging roads into stream channels was observed for streams #45, #2, #11, #7, #41, and #40. This could be prevented by

revegetating the sides of logging roads in the vicinity of water courses (as was the case for stream #21), or adding bales of hay to prevent the movement of sediment into stream channels. If a stream crossing is going to be maintained, it should be monitored; if not, the road should be removed, with minimum damage to the stream.

8. **If possible, future researchers should select streams which have had no prior history of disturbance.** Logging activities may have confounded the results for Dustin Brook and Cains, and possibly, several other streams, including #41, Upper Vault and Sweeney. Recovery from log driving and damming may take many decades, as can be seen with some of the major rivers flowing through Fundy National Park.

5.7.0. Summary

The purpose of this study was to examine the effects of forest harvesting and management on streams in Fundy National Park and vicinity. Sixteen streams were sampled, 12 draining 5 to 20-year-old clearcuts, and four draining reference streams within Fundy National Park, including Cains, which had been clearcut 70 years ago.

Before the Clean Environment Act (1977), forests were clearcut to the edges of streambanks and logs were either driven downstream to sawmills, or hauled out on logging roads. Since 1977, legislation has required greater protection of streams, so that most forest harvesting activities in the 1980s have required protective buffer strips.

In this study, I found that the removal of riparian vegetation in the three catchments clearcut in the 1970s has resulted in long-term perturbations to ecosystem function. In general, streambank cutting resulted in higher stream temperatures, changes in litter inputs, and long-term depletion of large woody debris. Recovery of these characteristics to pre-disturbance conditions could take many decades. Despite this, the invertebrate communities of the streams cutover in the 1970s were similar to those of the undisturbed, reference streams.

The recovery of the more recently cutover streams to pre-disturbance conditions was more rapid than the recovery of the older cutovers. This appeared to be a consequence of protective buffer strips. However, with or without buffer strips, most of the clearcut watersheds displayed disruptions of structure and function, including large concentrations of nutrients in streamwater (particularly nitrate); elevated rates of sedimentation and bedload movement; and widely fluctuating stream temperatures.

Partial cutting of a watershed appeared to mitigate the negative effects of forestry activities. Stream #11, which was 65% clearcut, resembled the reference streams more than the other recently cutover streams. Canopy cover, moss cover, and inputs of woody debris were maintained by a wide buffer along one side of the reach, and forest along the other. Furthermore, cutting was done in two stages, with 35% cut in 1984 and 30% in 1987.

The unstable, post-cutting stream environment led to a reduction in invertebrate abundance in rockballs (particularily Chironomidae). The Nymphomyiidae also appeared to be negatively affected by clearcutting. On the other hand, the algae-eating Elmidae, Plecoptera and silt-loving Oligochaetes were more abundant colonizing rockballs in clearcut streams. Not one of the physical or chemical stream characteristics measured in this study explained these patterns in invertebrate abundance. Instead, the age of the stand appeared to be most closely correlated with the invertebrate community.

Clearcutting and road-building tended to degrade trout habitat by increasing stream temperatures and sedimentation, and by destabilizing spawning gravels and thermal regime. Alternatively, higher temperatures in cutover streams may have enhanced trout development and survival.

This study provided data for this region so that both the staff of Fundy National Park and people in the Department of Natural Resources will have a better understanding of the local impacts of forestry activities. In addition, a number of recommendations were made to mitigate effects of harvesting on stream ecosystems. I hope this provides a useful reference to anyone wishing to protect the landscape, and in particular, the stream environment, from large-scale deforestation.

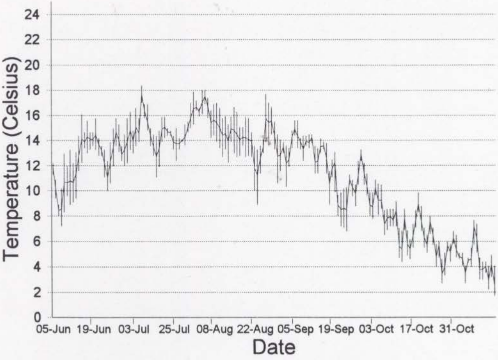
Appendix 1. Buffer strip widths (in metres) measured from each side of the stream bank, 70, 30 and 10 metres below the sampling reach, every 5 metres within the 25-m sampling reach, and 10, 30 and 70 metres above the sampling reach.

	Transect						---> Beginning of sampling reach					
	-70m		-30m		-10m		0m		5m		10m	
	left	right	left	right	left	right	left	right	left	right	left	right
STREAM												
45	+20	+20	+20	+20	+20	+20	+35	+35	+35	35	+20	+20
2	+20	+20	+20	+20	+20	+20	+20	+20	+20	+20	+20	+20
11	+30	27	+30	26	+30	25	+30	25	+30	25	+30	25
7	+30	+30	+30	+30	+30	+30	13	+20	12	+20	10	+20
21	+30	7	+30	11	21	11	19.5	12.5	28	10	26.5	10
Haley							+20	+20	+20	+20	+20	+20
25	25	+30	28	+30	25	+30	24	+30	26	+30	25.5	+30
Dustin	0	+20	24	+20	24	+20	26.5	+20	33	+20	+33	+20
41	+30	+30	+30	+30	+30	+30	+30	+30	+30	+30	+30	+30
15	0	0	0	0	0	0	0	0	0	0	0	0
McGee	0	0	0	0	0	0	0	0	0	0	0	0
40	+30	+30	+30	+30	12	+30	+20	+20	+20	+20	+20	+20

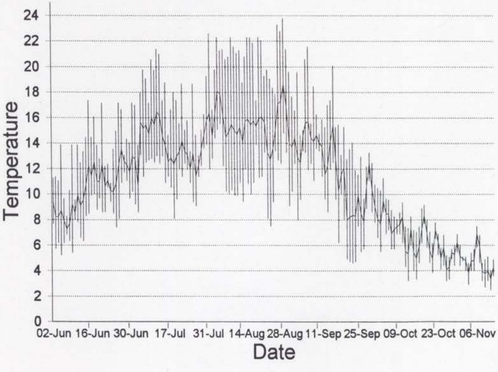
Note: Selective harvesting was observed in the buffer strips of streams #40, #11 and Dustin.
The left side of stream #11 and the right side of stream #25 were forested along their sampling reaches.
The buffer strip of stream #41 consisted of a very open forest with many budworm killed trees.

	Transect				End of sampling reach --->							
	15m		20m		25m		+10m		+30m		+70m	
	left	right	left	right	left	right	left	right	left	right	left	right
STREAM												
45	+20	+20	+20	+20	+20	+20	+20	+20	+20	+20	+20	+20
2	+20	+20	+20	+20	+20	+20	+20	+20	+20	+20	+20	+20
11	+30	25	+30	25	+30	25	+30	17	+30	17	+30	6
7	6	+20	5	+20	0	+20	0	+20	0	+20	0	+20
21	23	9.5	21	11	24	11	15.5	13	15	18	14	10
Haley	10	+20	5	+20	5	+20	7	+20	0	0	18	+20
25	24	+30	23	+30	15	+30	16	+30	18	+30	18	+30
Dustin	+33	+20	+30	+20	+30	+20						
41	+30	+30	+30	+30	+30	+30	+30	+30	+30	+30	+30	+30
15	0	0	0	0	0	0	0	0	0	0	0	0
McGee	0	0	0	0	0	0	0	0	0	0	0	0
40	+20	+20	+20	+20	+20	+20	0	0	+20	+20	+20	+20

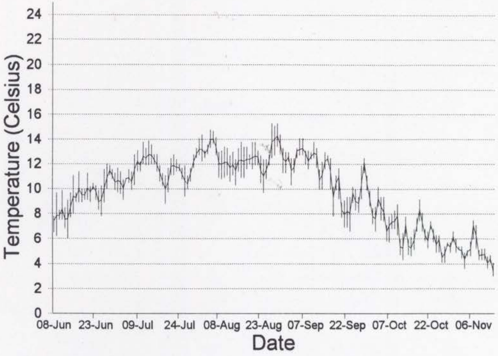
Appendix 2(a) . Mean, maximum and minimum daily temperature for stream #11, forested on one side, with a 6-27 m buffer strip on the other. Cutting occurred in this basin during 1984 and 1987.



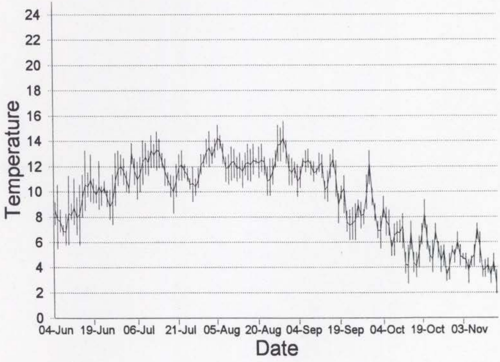
Appendix 2(b). Mean, maximum and minimum daily temperature for stream #15, with no buffer strip. Cutting occurred in this basin during 1979.



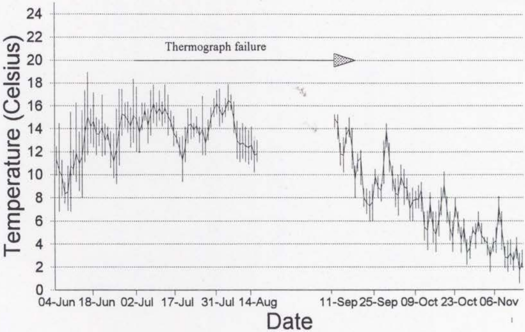
Appendix 2(c). Mean, maximum and minimum daily temperature for the reference stream, Sweeney Brook.



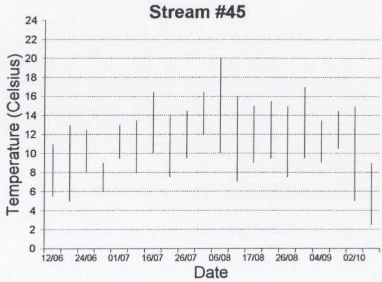
Appendix 2(d). Mean, maximum and minimum daily temperature for the reference stream, Upper Vault.

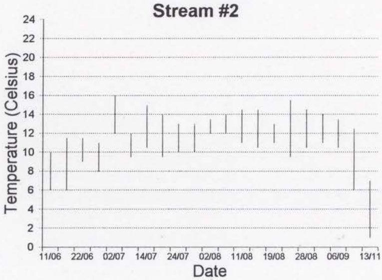


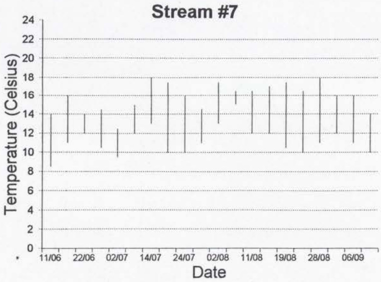
Appendix 2(e). Mean, maximum and minimum daily temperature for stream #25, which flows unprotected through a 1984 clearcut, then enters a buffer strip >15 m on one side, and forested on the other. Thermograph malfunctioned from early July to mid-September.

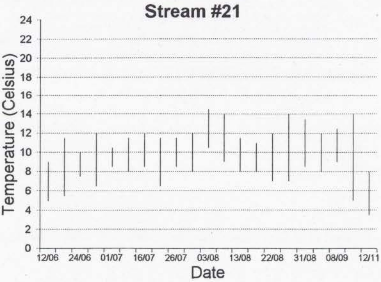


Appendix 3(a). Maximum/minimum thermometer data showing the 4-5 day temperature range of streamwater.
Streams #45, #2, #7 and #21 were cutover in the mid-late 1980s.

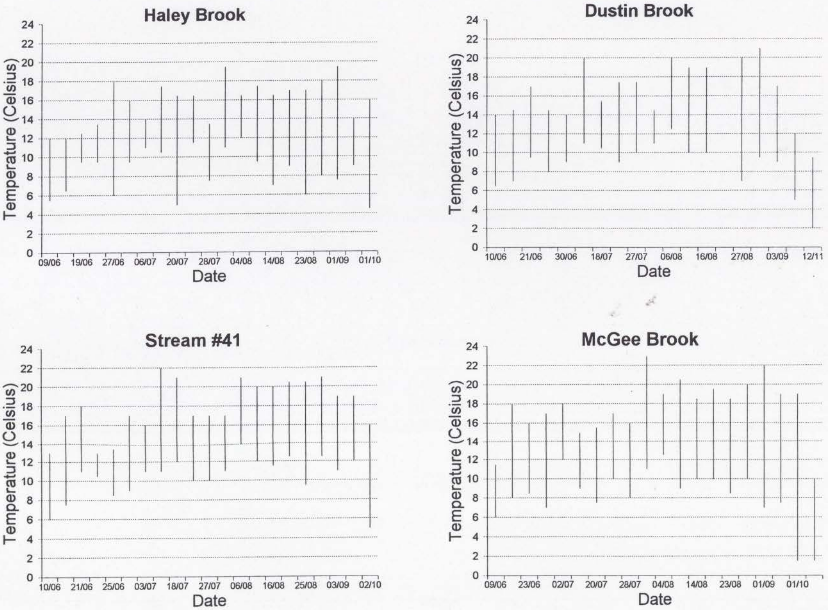




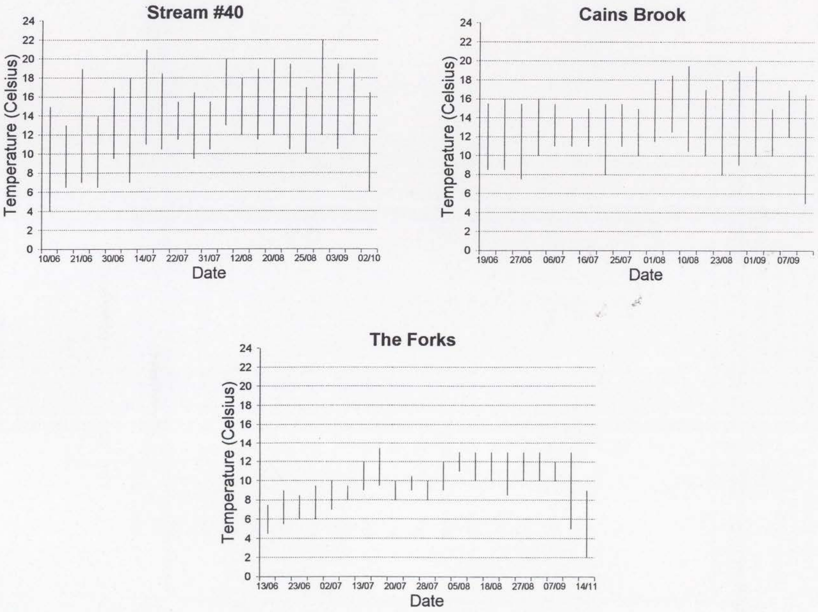




Appendix 3(b). Maximum/minimum thermometer data showing the 4-5 day temperature range of streamwater. Haley, Dustin and stream #41 were cutover in the early 1980s, and McGee was cutover in 1978.



Appendix 3(c) . Maximum/minimum thermometer data showing the 4-5 day temperature range of streamwater. Stream #40 was clearcut in 1973, and Cains and the Forks are reference streams.



Appendix 4. Pearson correlation coefficients for relationships between physical, chemical and biological variables. Variables with non-normal distributions were transformed in the following ways: age of stand (inverse transformed); snag, shrub and tree density, chloride, magnesium, alkalinity, sodium, channel area and width, and siltation (log transformed); moss cover and nitrogen (square root transformed); and surface fines (arcsine transformed).

	Age of stand	Snag density	Shrub density	Tree density	Tree diversity	Buffer width	Canopy cover
Age of stand	1.00						
Snag density	-0.17	1.00					
Shrub density	-0.04	-0.70*	1.00				
Tree density	0.26	0.64*	-0.65*	1.00			
Tree diversity	0.26	0.35	nc	nc	1.00		
Buffer width	-0.27	0.67*	-0.39	0.54*	nc	1.00	
Canopy cover	0.17	0.50*	-0.67*	0.54*	0.26	0.45	1.00
Moss cover	-0.05	0.71*	-0.80*	0.49	-0.04	0.40	0.46
Chloride	0.16	0.34	nc	nc	-0.47	nc	0.14
Magnesium	0.03	-0.35	nc	nc	0.17	nc	0.07
Alkalinity	0.28	-0.63*	0.66*	-0.32	nc	-0.39	-0.42
Potassium	0.75*	-0.15	-0.13	0.26	nc	-0.10	0.06
Sodium	0.32	0.12	nc	nc	-0.04	nc	0.45
Nitrogen	0.80*	-0.08	nc	nc	0.07	nc	0.10
Maximum temp.	0.09	-0.34	nc	nc	0.03	nc	-0.69*
Area	0.12	0.06	-0.04	-0.16	nc	-0.11	0.12
Width	-0.23	0.22	-0.21	-0.15	0.29	-0.11	0.06
Surface cobble	0.01	0.24	nc	nc	0.60*	nc	0.02
Surface bedrock	0.67*	0.12	nc	nc	0.03	nc	0.42
Surface fines	0.34	-0.15	nc	nc	0.01	nc	0.01
Siltation	0.62*	-0.25	nc	nc	0.41	nc	0.05

* significant at $P < 0.05$

nc Pearson r not calculated; known from larger matrix that variables unlikely to be significantly correlated.

Appendix 4. Continued.

	Moss cover	Chloride	Magnesium	Alkalinity	Potassium	Sodium	Nitrogen
Moss cover	1.00						
Chloride	0.44	1.00					
Magnesium	-0.15	-0.31	1.00				
Alkalinity	-0.47	nc	nc	1.00			
Potassium	0.14	nc	nc	0.30	1.00		
Sodium	0.14	0.45	0.38	nc	nc	1.00	
Nitrogen	0.08	-0.32	0.63*	nc	nc	0.32	1.00
Maximum temp.	-0.37	-0.60*	-0.17	nc	nc	-0.67*	0.12
Area	-0.08	nc	nc	-0.16	-0.11	nc	nc
Width	0.19	0.10	-0.53*	-0.29	-0.36	-0.11	-0.52*
Surface cobble	-0.09	-0.20	0.04	nc	nc	0.05	0.07
Surface bedrock	0.47	0.01	0.33	nc	nc	0.39	0.75*
Surface fines	0.02	-0.12	0.24	nc	nc	-0.11	0.16
Siltation	-0.13	-0.63*	0.59*	nc	nc	-0.02	0.62*

Appendix 4. Continued.

	Maximum temperature	Area	Width	Surface cobble	Surface bedrock	Surface fines	Siltation
Maximum temp.	1.00						
Area	nc	1.00					
Width	-0.01	0.74*	1.00				
Surface cobble	-0.07	nc	0.21	1.00			
Surface bedrock	-0.15	nc	-0.20	0.00	1.00		
Surface fines	0.14	nc	-0.13	-0.37	-0.04	1.00	
Siltation	0.35	nc	-0.12	0.08	0.51	0.10	1.00

* significant at P = 0.05

nc Pearson r not calculated; known from larger matrix that variables unlikely to be significantly correlated.

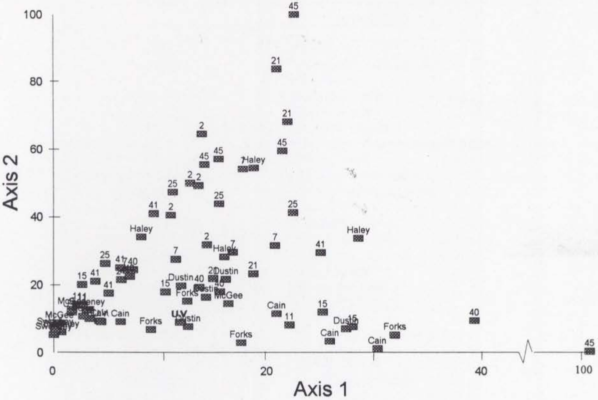
Appendix 5. Invertebrate abundance of each taxonomic group in each rockball for all 16 streams.

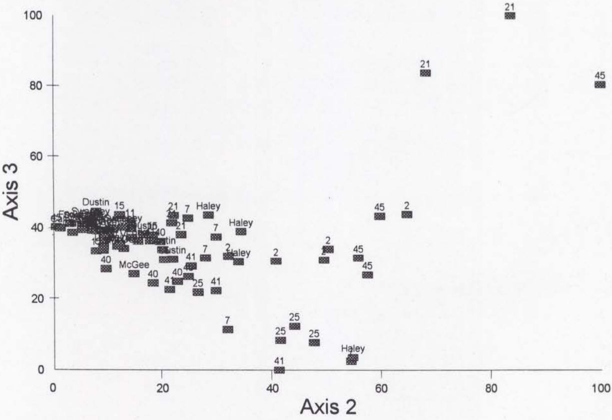
STREAM	ROCKBALL	EPHEMEROPTERA	PLECOPTERA	COLEOPTERA	ELMIDAE	TRICHOPTERA	DIPTERA									OLIGOCHAETE	PELECYPODA	COPEPODA	COLLEMBOLA	ARACHNIDA	ANISOPTERA	TOTAL PER	AVERAGE #
							CHIRONOMIDAE	SIMULIIDAE	TIPULIDAE	ATHERICIDAE	CERATOPOGONIDAE	EMPIDIDAE	NYMPHOMYIIDAE	PSYCHODIDAE	OTHER DIPT.							ROCKBALL	PER SAMPLE
#45	1	12	54			20	69	391	4					1		9						560	188.00
#45	2	6	35		2	3	52		2		1			1		3						105	
#45	3	1	15			5	15		1						1	10						48	
#45	4	3	31	1	1	4	47	5	4		1					7						104	
#45	5	17	23			16	60									7						123	
#2	1	4	15			7	43		1							1						71	174.20
#2	2	23	21			6	71		1							16						138	
#2	3	5	44			3	85		7							4						148	
#2	4	18	49			10	196	14	8							4						299	
#2	5	8	64			17	116		5			1				4						215	
#11	1	4	15			6	219	43	3		1					1						292	362.75
#11	2	2	13			2	196	1								3						217	
#11	3	13	22		2	9	377									3						426	
#11	4	5	36		1	3	462	2	3							4						516	
#7	1	3	18		1	19	175	2	7			1				5						231	201.20
#7	2	17	23		2	23	163	8	7			1				2						246	
#7	3	32	9			9	75	13							1							139	
#7	4	11	13	1	1	5	95	11	4							5						146	
#7	5	81	29			16	100	8	4							6						244	
#21	1	13	38			12	174	21	10							60	4					332	252.40
#21	2	9	30			30	196	27	5							4						301	
#21	3	10	15			25	208	21	3							9						291	
#21	4	4	19			6	69	5	4							32	3					142	
#21	5	3	14			17	153	2	4							3						196	
Haley	1	11	22		1	8	131	1	4			1				7						186	334.75
Haley	2	53	62		2	11	101	7	6					1						1		244	
Haley	3	30	151	1	4	50	361	96	14			4		4		7						722	
Haley	4	10	15		1	4	130	14	3			1				9						187	
#25	1	66	60		6	2	180	12	1							5						332	214.75
#25	2	55	40		6	4	119	22	2							2						250	
#25	3	29	12		6	1	154		2							2						206	
#25	4	15	13		3	1	37					1				1						71	
Dustin	1	7	14		3	36	211	21	1							2		1				296	283.60
Dustin	2	2	13		2	29	221	59	1				1									328	
Dustin	3	14	29		5	23	195	22	1			1				1			1			292	
Dustin	4	1	3			10	158	16		1		1				1	1					192	
Dustin	5	10	32	1	3	17	229	16	1							1						310	

Appendix 5. Continued.

STREAM	ROCKBALL	EPHEMEROPTERA	PLECOPTERA	COLEOPTERA	ELMIDAE	TRICHOPTERA	DIPTERA									OLIGOCHAETE	PELECYPODA	COPEPODA	COLLEMBOLA	ARACHNIDA	ANISOPTERA	TOTAL PER	AVERAGE #
							CHIRONOMIDAE	SIMULIIDAE	TIPULIDAE	ATHERICIDAE	CERATOPOGONIDAE	EMPIDIDAE	NYMPHOMYIIDAE	PSYCHODIDAE	OTHER DIPT.							ROCKBALL	PER SAMPLE
#41	1	2	4		1	2	13	3														25	128.80
#41	2	5	11			4	120	2	1							1					1	145	
#41	3	12	26		3	4	154	1								1						201	
#41	4	22	8		4	3	44					1										82	
#41	5	24	10		3	3	151															191	
#15	1	4	16		5	14	258	60								9						366	424.80
#15	2	6	27		4	46	302	17	2		1		1			2						408	
#15	3	19	7	3	17	4	222		1							5						278	
#15	4	9	16	6	12	14	509	10	5			7	4			1						593	
#15	5	22	25		6	1	328	91				2	3			1						479	
McGee	1	64	23		6	3	466	63	5		1	2	1			1				1		636	673.00
McGee	2	45	17		9	5	706	2	6	1		2				4						797	
McGee	3	9	16		1	4	551		3		1	1				1						587	
#40	1	32	12		1	6	184	22	2													259	243.20
#40	2	20	11		1	3	123	3								2						163	
#40	3	44	26		2	10	278	129														489	
#40	4	3	11		2	3	86	8							1	1						115	
#40	5	22	13			8	143	3	1													190	
Cain	1	7	10			8	350	105	1							1						482	518.20
Cain	2	16	17		1	18	506	21	1			3										583	
Cain	3	16	18		1	4	478	13	3	1		1										535	
Cain	4	25	17		1	17	323	60	6				3			2						454	
Cain	5	14	13	1		7	402	99				2					1					539	
Sweeney	1	2	9		1	5	502	3	3													525	592.50
Sweeney	2	1	6		1	10	533	1														552	
Sweeney	3	6	14			30	411	3	8						1	1						474	
Sweeney	4	2	6		1	22	776		5			1	2			3						818	
Upper Vault	1	37	10		1	5	506	48	1							1	1					610	810.00
Upper Vault	2	56	14		4	17	775	73	4			4				1						948	
Upper Vault	3	17	11		1	10	752		3			2	2			2						800	
Upper Vault	4	24	12			17	680	19	6			1	1	1		2						763	
Upper Vault	5	41	31			14	832	1	5			1	3			2						930	
Forks	1	3	21			18	434	4	5													485	240.20
Forks	2	2	13			7	170	55	4													251	
Forks	3	0	5			5	171	12	4													197	
Forks	4	1	9			4	77	6	1			1	1	1								101	
Forks	5	1	1			6	136	21	2													167	
Total per taxa		1242	1627	14	141	830	19095	1818	211	3	6	43	22	9	4	284	10	1	1	2	1		

Appendix 6. The arrangement of stream sites based on correspondence analysis of invertebrate data using each sample from every stream site. Axis 1 x Axis 2, and Axis 2 x Axis 3 are shown.





Appendix 7. Latin names and the corresponding common names of plant and fish species referred to in this study.

Latin name	Common name(s)
Plant species	
Abies balsamea (L.)	Balsam Fir
Acer pensylvanicum L.	Moosewood, Striped Maple
Acer rubrum L.	Red Maple
Acer saccharum Marsh.	Speckled Alder
Acer spicatum Lamb.	Sugar Maple
Alnus incana ssp. rugosa (Du Roi)	Mountain Maple
Amelanchier spp.	Juneberry, Serviceberry, Shadbush
Betula alleghaniensis Britt.	Yellow Birch
Betula papyrifera Marsh.	White or Paper Birch
Cornus rugosa Lam.	Round-leaved Dogwood
Cornus spp.	Dogwood
Corylus cornuta Marsh.	Beaked Hazelnut
Fagus grandifolia Ehrh.	American Beech
Fraxinus nigra Marsh.	Black Ash
Lonicera spp.	Honeysuckle
Nemopanthus mucronata (L.) Trel.	Mountain-Holly
Picea glauca (Moench) Voss	White Spruce
Picea rubens Sarg.	Red Spruce
Prunus pensylvanica L.F.	Pin, Bird or Fire Cherry
Prunus virginiana L.var.virginiana	Choke Cherry
Rubus spp.	Salmon Berry
Salix bebbiana Sarg.	Long-beaked Willow
Salix discolor Muhl.	Pussy-Willow
Salix spp.	Willow
Sambucus racemosa L.	Red-berried Elder
Sorbus americana Marsh.	American Mountain-Ash
Spirea spp.	Spirea
Thalictrum pubescens Pursh	Meadow-rue
Viburnum cassinoides L.	Witherod, Wild Raisin
Viburnum trilobum Marsh.	Highbush Cranberry
Fish species	
Oncorhynchus keta Walbaum	Chum Salmon
Oncorhynchus kisutch Walbaum	Coho Salmon
Oncorhynchus tshawytscha Walbaum	Chinook Salmon
Salmo salar L.	Atlantic Salmon
Salmo trutta L.	Brown Trout
Salvelinus fontinalis Mitchill	Brook Trout

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