

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 m³ 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 Brunswick, 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. Similarly, 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.

